University of New England

Ecological, biophysical and animal production responses to strategic-rest grazing in Australia and worldwide

Submitted by

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Original material declaration

I certify that the ideas, data collection, results, analyses, software and conclusions reported in this dissertation are entirely my own effort, except where otherwise acknowledged. I also certify that the work is original and has not been previously submitted for any other award, except where otherwise acknowledged.

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Note to the examiners

The thesis has been written in the style of a thesis by publication. Two of the chapters have been published in peer-reviewed journals (Journal of Applied Ecology and Agriculture, Ecosystems and Environment). In this thesis two separate terms are used for rotational grazing practices that incorporate extended rest: (i) In the global review, the term “strategic-rest grazing” encompasses all variations on grazing incorporating rest; (ii) in the regional study the term “short-duration grazing” relates to a sub-set of broader strategic-rest grazing practices in the region. For consistency, formatting of the thesis aligns with that suggested by the University of New England. As Chapters 2 through to 6 have been prepared as manuscripts, there is some repetition, for which I apologise in advance.

Sheep grazing, New England Tablelands, New South Wales

"The care of the Earth is out most ancient and most worthy,

and after all our most pleasing responsibility"

(Wendell Berry)

“It is important to be cautious about how 'grazing' is characterized. When properly managed on a sustainable basis grazing can be an effective practice for maintaining the biodiversity of most indigenous grasslands. In many cases unsustainable grazing and the removal of moribund biomass by domestic livestock is inappropriately undertaken, whereas if managed sustainably grazing can be an effective practice, in particular if used with an appropriate fire regime. We need to be careful 'not to judge' grazing practices and any decision-making needs to be based on sound conservation biology principles, and getting the right balance among conservation, traditional cultural values and development.”

(From IUCN, 2008, 'Life in a Working Landscape: Towards a Conservation Strategy for the World's Temperate Grasslands')

Abstract

Livestock grazing can facilitate the maintenance of biodiversity in landscapes or cause landscape degradation and biodiversity loss. With a global population expected to surpass 9 billion people by 2050, there will be increasing pressure on the world's grazing lands to produce protein while minimising impacts on landscapes. This thesis explores the potential for grazing that incorporates extended and planned rest (Strategic-rest grazing, hereafter SRG) to enable continuing livestock production while also maintaining biodiversity and biophysical functions.

In Chapter 2, I conducted a global meta-analysis comparing impacts of SRG to continuously grazed or ungrazed areas. I found that total groundcover and animal production per hectare were significantly greater with SRG compared to continuous grazing, while biomass, plant richness, plant diversity and animal weight gain did not differ between grazing treatments. Where the length of rest, relative to graze time increased with SRG, there were significant increases in biomass and further increases in groundcover and animal production per hectare in comparison to continuous grazing. These findings highlight the importance of incorporating the length of rest relative to graze duration into analyses comparing grazing systems. I found that the main focus of research around SRG differed between major geographic regions and climate zones. North American, Australian and New Zealand research mostly focused on short-term animal productivity, as did research in temperate areas. In contrast, research from Europe predominantly focused on biodiversity conservation. Research in more arid areas has focused largely on general sustainability for continuing animal production. Where richness and diversity of flora and fauna were compared between SRG and continuously grazed areas, responses were mostly favourable in SDG areas, or there was no difference. There were few examples of negative outcomes in SRG areas. Where

richness and diversity in SRG areas were compared to ungrazed areas there was often no difference between SRG and ungrazed. Despite the often-favourable responses for production and ecological outcomes with SRG, a very small number of studies have considered the potential to achieve animal production and biodiversity conservation simultaneously with SRG approaches. This suggests we have limited understanding of trade-offs and synergies between these two goals.

A localised study was undertaken of ground-layer biodiversity and landscape function outcomes in naturalised pastures in NSW, Australia. This study assessed grasslands on six properties managed with short-duration grazing (hereafter SDG, a form of SRG) and compared with outcomes on properties managed in ways more typical of the region (largely continuous and with unplanned rest; hereafter RP). With SDG management there was approximately 19% greater perennial herbaceous cover and a corresponding 14% lower cover of undesirable introduced annual plants. Significant improvement in attributes relating to landscape functioning were also seen with SDG management, with environmental factors less important in influencing these attributes. Pasture composition also differed between management approaches with increased cover of favourable forage species and reduced cover of species that increase under heavy grazing pressure with SDG management. Greater richness of native forbs was seen under RP, but no other identifiable differences in richness and Shannon-Wiener diversity was seen in the ground-layer of pastures managed in contrasting ways.

Insects are an important component of overall landscape biodiversity and are sensitive to changes in land-use and agricultural intensification. Insect richness and abundance were assessed on RP and SDG properties and found to be significantly higher on SDG sites. These increases were likely largely due to the greater cover of tall perennial plants and litter cover

and increased structural heterogeneity of the pasture sward with SDG management. These increases suggest there is potential for altered grazing practices to improve the capacity of grazed landscapes to provide ecosystem services from insects such as natural pest control and pollination, as well as provide food resources for wildlife.

This thesis has highlighted the potential to balance animal production, biophysical and biodiversity outcomes with grazing incorporating extended rest and that research to-date has largely been on animal production outcomes rather than biodiversity responses. Importantly, it highlights that minimal research has considered trade-offs and synergies between animal production and biodiversity conservation outcomes, and the potential to achieve both simultaneously. If we are to meet the growing demand for protein from the world's grazing lands, while also preventing landscape degradation and sustaining biodiversity, it is essential to fill this knowledge gap.

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Chapter 1 Introduction to thesis

1.1 Background

Livestock grazing occurs across more than 25% of the earth's surface and is an important contributor to food and economic security as well as human cultures (DeRamus, 2004; IUCN, 2008; Squires, 2012; Tudge, 2017; Webster, 2017). Although livestock grazing can be an important facilitator of biodiversity across landscapes (Hickman et al., 2004; IUCN, 2008), in many situations it is a major contributor to landscape degradation and biodiversity loss (MA, 2005; Steinfield et al., 2006). With a growing human population and increasing pressure on the world's agricultural lands, finding strategies to enable livestock grazing to co-exist with diverse and optimally functioning landscapes is a key challenge.

Changes in grassy ecosystems associated with livestock grazing are similar in many parts of the world (Dyksterhuis, 1949; Díaz et al., 2007). Under increasing grazing pressure, taller perennial species are replaced by more prostrate perennial species with annual plants and bare ground becoming dominant as grazing impacts increase (Dyksterhuis, 1949; Moore and Biddiscombe, 1964; Landsberg et al., 1999; Díaz et al., 2007). These changes resulting from heavy grazing pressure lead to losses of important biophysical ecosystem functions as well as substantial losses of biodiversity. Biophysical effects include considerably reduced potential for water infiltration (Moore and Biddiscombe, 1964; Yates and Hobbs, 2000) and the loss of soil and nutrients (Thurow, 1991; Yates and Hobbs, 2000). Additionally, the loss of perennial plants modifies the ground-layer micro-climate by increasing insolation levels and reducing humidity (King and Hutchinson, 1983; Neave and Tanton, 1989). Perennial plants and dense litter cover are also important habitat for animals, especially invertebrates

and microbes (Clarholm, 1985; Wardle et al., 2004; King and Hutchinson, 2007), which in turn are a food source for predatory and insectivorous animals (Recher and Lim, 1990; Nadolny, 1998; Losey and Vaughan, 2006).

Despite the negative impacts of heavy grazing, livestock can be important facilitators of diversity, when managed appropriately, as they remove excess standing plant litter and minimise the dominance of perennial species, allowing interstitial species to persist. In many parts of the world, diverse grasslands have co-evolved with large grazing herbivores (Frank et al., 1998) with traditional grazing practices based on natural and semi-natural habitats and local livestock breeds being significant contributors to biological and cultural diversity (Török et al., 2016; Varga et al., 2016). These traditional grazing practices have often implemented stocking strategies that varied with seasonal conditions and avoided over-exploitation of other areas in the landscape (Fernández-Giménez, 1999; Oba et al., 2000; Kioko et al., 2012; Zhang et al., 2015). Thus, in many circumstances, the production of food and fibre from livestock has co-existed alongside biodiversity.

1.2 Contemporary grazing practices

In contrast to many traditional grazing practices , modern grazing practices are typically sedentary with stock grazing in large units for extended periods or continuously (Earl and Jones, 1996; Kemp and Dowling, 2000; Teague et al., 2011). In the developed world, these systems are based on a paradigm of maximising animal production with minor value placed on other services from grasslands (Lang and Heasman, 2004; Bohlen and House, 2009). Prolonged grazing periods under sedentary management practices result in animals favouring certain areas due to factors such as proximity to water sources, position in the landscape, perceived or actual presence of predators, as well as learned and habitual behaviours (Taylor et al., 1985; Bailey et al., 1996; Fuhlendorf and Engle, 2001). These heterogeneous grazing

patterns can contribute to overall diversity if stocking rates are low in relation to the carrying capacity of the landscape but can lead to degradation in heavily used areas and under-use of other areas, leading to sub-optimal outcomes for ecological and productivity outcomes.

Livestock exclusion is an option commonly advocated for ecological restoration following negative impacts of grazing (Fleischner, 1994; Lunt et al., 2007; Prober et al., 2011). However, exclusion from grazing does not always achieve desired ecological outcomes in an appropriate timeframe (Hulme et al., 2002; Holechek et al., 2006; di Virgilio et al., 2019) and is not a viable approach for commercial grazing production, which impacts large areas of landscapes outside formal reserve systems. Historic landscape modification, including tree removal, cultivation and sowing of introduced pasture species and fertilisation to increase livestock productivity has a strong influence over current land condition. Nevertheless, grazing management that controls animal distribution across the landscape and incorporates extended periods of rest from grazing to allow grassland recovery, may be a way to lessen impacts from grazing while maintaining levels of livestock production that can contribute meaningfully to overall food and fibre demands.

1.2.1 Rotational and continuous grazing

Various terms have been used to describe rotational grazing systems that incorporate planned rest, including short-duration grazing (SDG), high-intensity, short-duration grazing (HISD), deferred grazing and various forms of rest-rotation grazing (for concise definitions of these grazing systems, see Allen et al., 2011). Many of these approaches to grazing management have been implemented historically (i.e., shepherding, transhumant practices) and in more recent decades with early advocates including Voison, 1959, Acocks, 1966 and Savory and Parsons 1980. A substantial body of research has considered the benefits or otherwise of rotational grazing compared with continuous grazing. This research has largely

considered animal performance and maintenance of vegetation and soil condition for continuing livestock production (Kemp and Dowling, 2000; Teague et al., 2013). For reviews and related discussions of this research, see Gammon (1978); Holechek et al. (2000); Briske et al. (2008); Teague et al. (2008).

While it is understood that tactical grazing management that alters the timing of grazing to achieve specific composition goals can be a valuable management tool (Jones, 1933; Lodge and Whalley, 1985; Kemp et al., 1996; Kemp et al., 2000), most authors to date conclude that there are few benefits of rotational grazing over continuous grazing for managing either the short or long-term productivity of grazing systems, or for promoting major compositional changes in pasture (Holechek et al., 2000; Dowling et al., 2005; Briske et al., 2008). More specifically, in relation to animal production, it is generally concluded that there are trends towards greater weight gain per animal in continuously grazed systems and greater animal production per unit area for rotational systems. Explanations given are that animals are able to graze more selectively in continuous systems leading to greater weight gain for individuals (Ellison, 1960; Briske et al., 2008), while for rotational systems reduced selectivity by animals generally means that more areas of the landscape are used (Joseph et al., 2002; Norton et al., 2013).

In contrast to grassland composition changes in response to rotational grazing, few studies have considered biophysical responses (i.e., soil compaction, water infiltration). However, studies by Teague et al. (2004) found reduced impacts of drought with rotational grazing (less impact of drought on perennial cover and bare ground) and favourable outcomes for a range of soil attributes with rotational grazing that incorporated rests of 40–80 days (Teague et al., 2011).

Limitations to conclusions include (1) that the majority of the research undertaken has been conducted in studies where the timing of grazing and stocking rate decisions have been made by researchers with the need to limit variability to fit within an experimental framework (Teague et al., 2013; Wilmer et al., 2018); (2) considered short-term outcomes and in small grazing units that rarely reflect commercial reality (Kemp and Dowling, 2000; Teague et al., 2013); (3) have been studies undertaken without the constraints of market, climate and socio-economic constraints that commercial grazing managers necessarily deal with (i.e. livestock grazing businesses exist within complex socio-ecological frameworks where they must constantly adapt to changing social, economic and ecological conditions; they are complex, creative, systems; (Provenza et al., 2013; Wilmer et al., 2018); (4) studies that rarely consider the length of rest relative to graze duration (di Virgilio et al., 2019), and (5) with some exceptions, been conducted over timeframes too short to identify the impacts of altered management (Teague et al., 2013; di Virgilio et al., 2019).

1.3 Rotational and planned rest grazing and biodiversity.

The ability to incorporate planned rest into grazing systems is likely to enable the integration of ecological and animal production outcomes to a greater degree than more continuous grazing (Dorrough et al., 2004) and there are claims that rotational grazing practices incorporating significant periods of planned rest can lead to improved biodiversity outcomes (Stinner et al., 1997; Lindsay and Cunningham, 2009; Kruse et al., 2016). In addition to planned and strategic rest, the reduced reliance on external nutrient inputs common in these systems is also likely to minimise impacts on native species that are often intolerant of nutrient enrichment (Isselstein et al., 2005; Dorrough and Scroggie, 2008). Despite these claims and likely potential, minimal research has been conducted considering outcomes for biodiversity in commercial-scale rotational grazing systems incorporating

planned rest (Dorrough et al., 2004; Dorrough et al., 2012; Nordborg, 2016) but see Chillo et al., (2019) and McDonald et al., (2019).

1.3.1 Plant biodiversity

The few studies that have explicitly considered impacts of rotational grazing on plant richness and diversity overall indicate a minor influence of grazing management (White et al., 1991; Beukes and Cowling, 2000; Hickman et al., 2004; Selemani et al., 2013). In the few studies that have found an effect of rotational grazing strategies on richness and diversity, management was less important than environmental factors such as position on slope (Guretzky et al., 2005; Guretzky et al., 2007; Fujita et al., 2009) and rainfall patterns (Weigel et al., 1989). It is also clear that excessive grazing pressure overrides the effects of any changes in management approach (O'Reagain and Turner, 1992; Ash and Stafford Smith, 1996; Provenza, 2003). However, where studies have been conducted on commercial farms and ranches they have mostly shown favourable changes to the compositional of plant communities with rotational grazing strategies (Earl and Jones, 1996; Jacobo et al., 2006; Teague et al., 2011; Chillo et al., 2015) although Dowling et al. (2005) concluded there were no significant overall responses across several properties spanning a large geographical area. There appear to be no studies showing negative influence of rotational grazing strategies on plant richness and diversity.

In contrast to the lack of supporting evidence for favourable plant biodiversity outcomes with rotational grazing strategies, several studies have demonstrated increased levels of plant richness and diversity where pastures are rested from grazing in particular seasons (Hulme et al., 2002; Leonard and Kirkpatrick, 2004; Ruthven, 2007; Chen et al., 2008; Lennartsson et al., 2012; Mavromihalis et al., 2013). Some studies of seasonal grazing have also indicated favourable outcomes in comparison to grazing exclusion (Humphrey and Patterson, 2000;

Kleinebecker et al., 2011). It is known that strategic grazing management with a particular goal in mind can alter grassland composition (Jones, 1933; Lodge and Whalley, 1985). Consequently, the favourable responses seen with seasonal grazing suggest that if enhancing biodiversity was an explicit goal of a grazing manager, rest periods could be timed to coincide with particular growth periods of plant species of interest (Dorrough et al., 2004).

1.3.2 Animal biodiversity

Less attention has been given to impacts of contrasting grazing strategies on animal biodiversity. As with plant diversity, impacts on animals are generally considered by comparing grazed areas with areas where livestock are excluded or the effects of lowered grazing intensity (Chillo et al., 2015; DeLonge et al., 2016; Eldridge et al., 2016). Where studies have considered impacts of rotational grazing strategies on animal biodiversity, most attention has been given to birds and insects, with relatively less attention around impacts on mammals and reptiles. Where ungrazed areas have been compared to rotationally grazed areas there are examples of negative impacts of rotational grazing (Temple et al., 1999; Isacch and Cardoni, 2011; Davis et al., 2014). However, it is more common that rotational grazing has either no impact (Sedivec et al., 1990; Danley et al., 2004; Arthur et al., 2008; Morón-Ríos et al., 2010; Ranellucci et al., 2012) or favourable impacts on animal biodiversity (Gebeyehu and Samways, 2003; Keene, 2009; Saunders and Fausch, 2012; Fonderflick et al., 2014) compared to ungrazed areas.

Where animal biodiversity in rotationally grazed areas is compared to continuously grazed areas, outcomes are commonly favourable with rotational grazing (Schulz and Guthery, 1988; Sedivec et al., 1990; Gebeyehu and Samways, 2003; Branson and Sword, 2010; Farruggia et al., 2012; Enri et al., 2017), particularly where rest occurs during particular seasons (Arthur et al., 2008; Johnson and Sandercock, 2010; Fonderflick et al.,

2014). It is rare that animal biodiversity is negatively impacted by rotational grazing in comparison to continuous grazing (Isacch and Cardoni, 2011; Dorrough et al., 2012; Goodenough and Sharp, 2016). Both Dorrough et al., (2012) and Goodenough and Sharp, (2016) indicated that there were opportunities to further alter management to optimise biodiversity outcomes. Additionally, Isacch and Cardoni, (2011) noted that contrasting grazing strategies impacted bird species differently, highlighting that a range of grazing strategies is necessary optimise outcomes for both livestock production and biodiversity.

1.4 Australian temperate grasslands

Temperate grassy ecosystems include the steppes of Eurasia, the North American Prairies, the South American Pampas, South African Veld and grasslands in temperate regions of Australia and New Zealand (Watkinson and Ormerod, 2001). These areas are some of the most hospitable and productive landscapes on earth and in many cases have been heavily modified by humans for agriculture (IUCN, 2008). They make up approximately 8% of the terrestrial surface area of the world and are the least protected biome on the planet (IUCN, 2008).

Changes in Australian temperate grassy ecosystems since European colonisation in the late 1700s have been considerable with up to 95% of the original vegetation having been heavily altered for agriculture by clearing, cultivation, livestock grazing and fertiliser application (Dorrough et al., 2004; McIntyre et al., 2004). The cessation of burning by Indigenous Australians also changed vegetation structure on a large scale (McIntyre et al., 2004; Gammage, 2011). More recently tree dieback in these regions is a major threat to ecological function and biodiversity (Landsberg and Wylie, 1988; Reid and Landsberg, 2000; Fischer et al., 2009).

The consequence of historic and ongoing agricultural intensification in these regions is that grasslands and grassy woodlands differ markedly from the original native ecosystems (Moore, 1970; Whalley et al., 1976; Prober et al., 2002). Broad structural changes, in addition to the cumulative loss of large mature trees through ring-barking, clearing episodes and tree dieback, include declines in the diversity of erect native perennial grasses and forbs (McIntyre and Lavorel, 1994; Pettit et al., 1995; Dorrough et al., 2004) and a shift towards greater dominance by prostrate and rosette plants with cool-season and annual growth patterns (Moore and Biddiscombe, 1964; Moore, 1970; Lodge and Whalley, 1989). In addition to losses of floral diversity, the changes in vegetation structure have negatively affected faunal diversity and a range of ecosystem services, that in turn, challenge the sustainability of commercial livestock grazing systems (Dorrough et al., 2004).

1.4.1 Dominant paradigm of agriculture in temperate Australia

Contemporary agriculture is dominated by a model that considers commodity production as the main goal with high levels of short-term productivity valued over other social, environmental and cultural services from these landscapes (Jones, 1996; Whalley, 2000; Holt-Giménez and Altieri, 2013). Modern temperate grazing systems are largely based on modified pastures with inputs such as phosphate-based fertilisers and forage legumes (Jones, 1996; Kemp et al., 1996). Donald (1970) and Wolfe (1972) considered native grasses to be generally inferior and in industry guides to temperate pasture management in Australia, native grasses receive very little attention (Wheeler et al., 1987). Despite recognition of the agronomic value of native grasses in pastures (Archer and Robinson, 1988; Robinson and Archer, 1988) and calls for research into native grasses for pasture production systems in the 1990s and early 2000s (Jones, 1996; Fitzgerald and Lodge, 1997) in response to declining pasture productivity and concerns around changing climate, only a small amount of research

has ensued (Garden et al., 2005; Norton et al., 2005; Sanford et al., 2005; Waters et al., 2005). In general, alternative grazing systems based on native and low-input species are poorly promoted despite the potential for native grasses to contribute forage value as well as soil protection especially in times of climate stress (Jones, 1996; Kemp et al., 1996; Nadolny, 1998; Dorrough et al., 2004; Firn, 2007; Williams, 2018).

1.4.2 Alternative model based on fewer inputs in temperate Australia

Alternative approaches to grazing management based on naturalised pastures are likely to be more resilient within the Australian context of unpredictable rainfall patterns (Whalley, 2000). This may be especially relevant with future climate change scenarios that predict increased temperatures and less frequent, more intense rainfall events (Hughes, 2003; Hennesey et al., 2008). Alternative models are likely to generate additional services from grazed landscapes such as mitigation of temperature extremes, carbon sequestration, biodiversity conservation and protection of water catchments in addition to commodity production.

1.5 Study region - the North West Slopes and Northern Tablelands of NSW

On the North West Slopes and Northern Tablelands of NSW, livestock grazing practices are typical of those in the remainder of temperate south-eastern Australia (Nadolny, 1998). The region is a plateau with peaks over 1500 m, rugged gorges to the east and a gently sloping fall to the west. Tertiary basalts are present at high elevations and overly a granite bathylith from the late Permian intruded into metamorphosed Palaezoic sediments known locally as trap rock (Norton, 1971). Soils in the region range from highly fertile to nutrient-

poor as a consequence of this varied geology (Norton, 1971; Nadolny, 1998). At the time of European settlement of the region, the vegetation varied from eucalypt forests to open grassy woodlands with grassy plains and frost hollows present ins some areas. Descriptions of the country from this time are contained in records of early explorers, pioneers and squatters (Norton, 1971; Curtis, 1989). On passing through the region near the Apsley River a few miles to the south of Walcha, NSW, the explorer John Oxley, commented: "… we proceeded through the finest open country, or rather park, imaginable: the general quality of the soil excellent…". Other writings from the time described country that was not densely wooded, but was "more like open park scenery" (John Everett, "Ollera" near Wandsworth; Norton, 1971); "fine, open country, with small plains and beautiful ridges"; Frederick Lamotte from Strathbogie, Norton, 1971), and with: "openness of countryside and thinness of wood" (Captain Henderson, 1851; Norton, 1971). In general, lower slopes and valley bottoms were open with large, mature eucalypts, while denser forested country was present on ridges and hilltops, open grasslands in valley bottoms were thought to be a minor component of the landscape (Oxley, 1820 in Curtis, 1989). Shrubs were present within some vegetation communities but were more common on poorer soils of the slopes than on the tablelands (Norton, 1971; Curtis, 1989). The open character of much of the grassy woodlands and open forests and grasslands of the region at the time of European occupation had likely been maintained by aboriginal burning (Norton, 1971; Gammage, 2011).

The region was first grazed from the late 1820s (Norton, 1971; Whalley et al., 1976). This was initially as large-scale stock movements across land claimed by squatters with livestock managed by shepherds (Norton, 1971; Nadolny, 1998). The pastoral industry was initially based on native grasslands, but introduced legume species were introduced as early as 1841 (Norton, 1971). Rye grass (*Lolium perenne* L.), cocksfoot (*Dactylis glomerata* L.), prairie grass (*Bromus spp.)* and sheep's burnett (*Sanguisorba minor* L.) were introduced in

the 1860s and other introduced plant species came in with the Goldrushes through the 1850s and 1860s (Norton, 1971). When land parcels became smaller as a result of the Robertson Land Acts of 1861, managers were expected to improve their land through tree clearing and the growing of crops. Swamps and lagoons were regularly drained, in part to combat liver fluke (*Fasciola hepatica*) and footrot which were common problems in the region (Norton, 1971).

The earliest botanical records from the area come from 1878 by Bentham and Mueller (in Norton, 1971). This was approximately 60 years after the initial introduction of livestock. Further records were made by Turner (1903) and Herbarium records from Gray (1961) also exist (from Norton, 1971). The combination of information gained from the diaries of early explorers and settlers, early official botanical records and remnant vegetation of roadside reserves, railways and historical cemeteries indicate that the composition of the original native pastures of the region is well defined (Lodge and Whalley, 1989).

In general, the herbaceous layer of grassy ecosystems in the region was dominated by tussocky C4 perennial graminaceous species. At higher elevations, the dominant (or co-dominant) species were *Themeda triandra* and *Sorghum leiocladum* along with *Poa sieberiana*. At lower elevations, *Poa* was replaced by a complex of species such as *Bothriochloa macra* and/or *decipiens*, *Aristida ramosa*, *Dichanthium sericeum*, and *Bothriochloa biloba*. Non-tussocky species likely to have been present were *Microlaena stipioides* and a range of *Rytidosperma* species. *Microlaena stipoides* in particular occurs today on fertile soils at higher elevations in the region as well as less fertile soils close to trees (Norton, 1971; Whalley et al., 1978). Annual species would have been uncommon in these original grasslands due to the dominance of the tussocky and other perennial grasses in the ground-layer (Whalley et al., 1978). It is also likely native herbaceous legumes such as

Desmodium varians, *Glycine tabacina* and *Cullen tenax* were common as they still are today in ungrazed areas (Whalley et al., 1978).

Early graziers had to manage significant challenges including extended dry periods and droughts, which settlers of European origin were generally unaccustomed to (Lines, 1991). These difficulties in dealing with the unpredictable climate were compounded by an inability to destock in dry times due to a lack of transport and market options typical of the time (Wright, 1985; Tongway and Ludwig, 1996). There were also substantial challenges associated with managing the 'winter feed-gap' – the Everett family at "Ollera" labelled August "starvation month" (Norton, 1971). This feed-gap was a consequence of the natural dominance of summer growing perennials and summer rainfall, combined with very cold temperatures during late winter. Extended dry periods and the winter feed-gap resulted in pastures, and particularly winter-active herbs, being heavily grazed, leading to major changes in pasture composition (Whalley et al., 1978). With these changes relatively unpalatable native species increased, often to the extent that they were viewed as invasive. However, while the introduced forage species that were a part of the flora by the mid 1800s produced considerable herbage mass under favourable conditions, they were not resilient in dry periods (Breakwell, 1923). The best pastures were considered to be a mix of native and introduced species, but native species needed to be managed carefully with judicious grazing management to ensure the persistence of more favourable species (Breakwell, 1923) and little has changed (Williams, 2018).

By the late 1980s and early1990s there was significant awareness of issues of environmental degradation arising from agricultural practices in Australia (White, 1997; Toyne and Farley, 2000). This recognition occurred alongside declining terms-of-trade of agriculture and rising costs of inputs (Chisholm, 1992; Kemp et al., 2000). Around the time,

graziers in temperate Australia began to look at ways of managing livestock that contrasted with conventional high-input models (Earl and Jones, 1996; Norton and Reid, 2013). In addition to general issues around environmental degradation and declining terms-of-trade, other important issues included an increasing acknowledgment of the lack of persistence of the perennial components of pastures and corresponding issues such as invasion of annual, weedy species, loss of productivity and a heavy reliance on fossil fuel-derived inputs to maintain pasture productivity (Nadolny, 1998; Kemp et al., 2000). At the time, the often premature death of native trees, along with a lack of regeneration of young trees (eucalypt dieback), was also reaching high levels (Landsberg and Wylie, 1988).

Graziers implementing alternative practices were influenced by methods such as the Savory Grazing method (Savory and Parsons, 1980) and the ideas of Voison (1959) and Acocks (1966). Cross and Ampt (2017) referred to the network of these graziers as an agroecological 'Community-of-Practice' with management practices including various forms of rotational grazing incorporating extended rest periods as a means of achieving overall holistic land and business management goals.

1.6 Purpose of this thesis

This thesis explores the knowledge gaps concerning the outcomes of grazing practices that incorporate extended periods of rest. The thesis forms two parts; part one explores major trends and gaps in the body of research investigating variations on grazing management that incorporates extended rest periods. The term strategic-rest grazing (SRG) is used for these global reviews; part two explores changes in biophysical and biodiversity attributes in the ground-layer on properties of the North West Slopes and Northern Tablelands of NSW managed with grazing that incorporates extended rest. Specifically, it focuses on the ground-layer of naturalised pastures on properties managed with short periods of grazing followed by

extended rests based on recovery of key forage species and desirable herbage mass in pastures. In this second part of the thesis, the term short-duration grazing (SDG) is used to refer to strategic-rest grazing practices. This study compares the ground-layer of naturalised pastures on these properties with the ground-layer of properties managed in ways more typical of regional practice where pastures are grazed more-or-less continuously, but with short periods of rest when forage becomes limited. This study was a part of a large-scale revegetation and restoration project, the Brigalow–Nandewar Biolinks, that aimed to restore biodiversity in an agricultural region that had been heavily impacted by agriculture in the past 150 to 200 years.

1.7 Structure of the thesis

Chapter 1 (this chapter) introduces the thesis, giving background to relevant research around strategic-rest grazing and the region and discussing research that is of relevance to the thesis.

Chapter 2 is a systematic literature review and meta-analysis examining peer-reviewed research comparing strategic-rest grazing (SRG) management with continuous grazing or grazing exclusion. This meta-analysis quantifies a range of outcomes including animal production, general sustainability and more specific biodiversity outcomes. This paper was co-authored with another PhD student and is published as a peer reviewed journal article (Journal of Applied Ecology).

Chapter 3 explores the literature in a more qualitative way by investigating the different foci of research into SRG in different regions and different climate zones of the world. It considers major trends and the extent to which this subset of grazing research integrates animal production and ecological or biodiversity conservation outcomes.

Chapter 4 examines generalised changes in plant responses and landscape function on 12 properties on the North West Slopes and New England Tablelands in northern NSW, Australia. It considers whether there are differences between properties managed with short-duration grazing (SDG), a form of strategic-rest grazing, for extended timeframes compared to practices that are more typical in the region. Specifically, it examines differences in: (1) landscape functioning, (2) major functional groups perennial and annual species and (3) responses of functional groups that are of interest from an animal production and general ecological functioning perspective.

Chapter 5 examines changes in plant richness and diversity of naturalised pastures on the same 12 properties on the North West Slopes and New England Tablelands of NSW in response to these contrasting management approaches and how this compares to the influence of environmental factors.

Chapter 6 examines insect biodiversity on 16 properties, eight which are managed with SDG and eight with RP, on the North West Slopes and New England Tablelands in NSW

Chapter 7 discusses the thesis outcomes as a whole and explores the implications of the findings and opportunities for further research.

Chapter 2 Ecological, biophysical and production effects of incorporating rest into grazing regimes: a global meta-analysis

Title of Article:	Ecological, biophysical and production effects of incorporating rest into grazing regimes: a global meta-analysis
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Candidate		Principal Supervisor

2.1 Abstract

1. Grazing can have considerable ecological impacts when managed inappropriately, however livestock production is a significant contributor to global food security and the removal of land from production is not always a viable option. Grazing management practices that incorporate periods of planned rest (i.e. strategic-rest grazing) may be an alternative to grazing exclusion or continuous grazing that could achieve ecological and animal production outcomes simultaneously.
2. We conducted a meta-analysis of global literature to investigate how strategic-rest grazing mediates ecological (i.e., plant richness and diversity), biophysical (plant biomass and groundcover) and production response variables (animal weight gain and animal production per hectare) compared to continuously grazed or ungrazed areas.
3. Overall, total groundcover and animal production per hectare were significantly greater under strategic-rest grazing than continuous grazing management, but biomass, plant richness, plant diversity and animal weight gain did not differ between grazing treatments. Increasing the length of rest relative to graze time under strategic-rest grazing was associated with an increase in plant biomass, groundcover, animal weight gain and animal production per hectare when compared to continuous grazing.
4. *Synthesis and applications.* Understanding both the ecological and animal production trade-offs associated with different grazing management strategies is essential to make informed decisions about best-management practices for the world's grazing lands. We show that incorporating periods of rest into grazing

regimes improves groundcover and animal production per hectare and that these benefits are more pronounced with increases in the length of time land is rested for. This extended rest also improves biomass production and weight gain compared to continuous grazing systems. Based on these meta-analyses, we recommend future research considers the duration of rest compared to graze time in comparisons of grazing systems.

Keywords: Biodiversity, biomass, continuous grazing, grazing exclusion, grazing management, groundcover, rotational grazing, weight gain.

2.2 Introduction

Livestock grazing is the single most extensive use of land on the planet, occupying ~25% of global land area and 66% of the world's agricultural land (FAO; Asner et al., 2004). The livestock industry employs over 1.3 billion people, is worth $1.4 trillion to the economy and provides about 33% of human protein intake (Thornton, 2010). However, the livestock sector is also a key driver of land-use change and can lead to degradation of ecosystem structure and function (Dorrough et al., 2004; Eldridge et al., 2016), biodiversity loss (MA, 2005; Steinfield et al., 2006) and soil degradation (Yates et al., 2000; Greenwood and McKenzie, 2001; MA, 2005; Steinfield et al., 2006) when poorly managed. With projected increases in human global population and corresponding demands for food production, pressures on grazing lands are likely to increase (FAO; Tilman et al., 2001; Steinfield et al., 2006). Therefore, the sustainable management of livestock to address food production needs whilst balancing environmental impacts is a major challenge.

Excluding livestock is often seen as a solution to conserve biodiversity and improve ecological condition (Pettit et al., 1995; Prober and Thiele, 1995; Spooner et al., 2002; Eldridge et al., 2016), though this strategy 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). However, livestock grazing may be compatible with maintaining ecological outcomes, if managed appropriately (WallisDeVries et al., 1998; Watkinson and Ormerod, 2001; Dorrough et al., 2004). In many parts of the world, large herbivores co-evolved with vegetation communities, whereby many plant species have adapted to grazing pressure and developed mechanisms to tolerate herbivory (Coughenour, 1985; Milchunas et al., 1988). In some circumstances, livestock can also 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 (Olff and Ritchie, 1998; Albert et al., 2015) and increasing plant community heterogeneity (Limb et al., 2018).

In contemporary grazing systems, land is commonly grazed continuously (year-long or throughout the entire grazing season) without periods of planned rest (Earl and Jones, 1996; Shakhane et al., 2013). Continuous grazing can result in uneven grazing patterns, i.e. patch grazing (Adler et al., 2001; Fuhlendorf and Engle, 2001), which can result in overgrazing of palatable and grazing-sensitive species (Teague and Dowhower, 2003; Briske et al., 2008; Teague et al., 2011; Norton et al., 2013). Severe overgrazing of large patches can also lead to land degradation processes, such as soil erosion (Blackburn, 1984). Incorporating periods of planned rest into grazing regimes, hereafter called strategic-rest grazing, is an alternative to continuous grazing management that is thought to reduce environmental degradation while maintaining or improving productivity (Hart et al., 1993; Norton, 1998; Teague et al., 2008). Previous reviews have concluded that there are few benefits for animal production and landscape sustainability from strategic-rest grazing systems, yet significant knowledge gaps remain (Holechek et al., 2000; Briske et al., 2008; Teague et al., 2013; Nordborg, 2016; Hawkins, 2017). Climate type, the length of time land is rested relative to time grazed and differences in stocking rates between treatments compared have been shown to affect responses to grazing (Briske et al., 2008; Teague et al., 2015; Eldridge et al., 2016). However, previous reviews into strategic-rest grazing have not assessed the importance of the length of rest periods and climate zone or compared both biodiversity and production metrics simultaneously, across multiple biomes or types of grazing systems that incorporate rest periods. In addition, there have been no reviews comparing effects of grazing with periods of rest with ungrazed areas on ecological or biophysical variables.

Here, we conduct the first quantitative, global meta-analysis to investigate the extent to which strategic-rest grazing (SRG) and varying length of rest periods influence ecological, biophysical and production outcomes across climate zones, compared to continuously grazed (CG) and ungrazed systems. Understanding the impacts of strategically managed livestock on both ecological and production objectives will increase the potential of land-sharing options for livestock grazing that are an alternative to grazing exclusion, which in many circumstances can have negative socio-economic consequences.

Specifically, we ask the following research questions:

1. How do differences in grazing management systems affect ecological outcomes (plant richness and diversity)?
2. How do differences in grazing management systems affect biophysical outcomes (biomass and groundcover)?
3. How do differences in grazing management systems affect animal production outcomes (weight gain and animal production per hectare)?
4. To what extent do climate, differences in stocking rate and the length of the graze and rest periods mediate ecological, biophysical and livestock production responses?

2.3 Methods

A systematic review of worldwide literature was conducted using Scopus, returning articles from 1950 until November 2017 to examine the effect of SRG on plant species richness, plant species diversity, groundcover, plant biomass, weight gain per animal and animal production per hectare. We searched for studies that compared SRG systems with either CG or ungrazed areas. Search terms were identified to address the scope of the study

and retrieve as many relevant studies as possible. 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* OR "ground cover" OR "groundcover" OR "bare ground") 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 in the analyses 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.

2.3.1 Meta-analyses

We compiled a dataset for each of the six response variables on which corresponding meta-analyses were conducted. In each dataset we collated the mean, standard deviation and sample size, along with the explanatory variables stocking rate difference, climate zone and the rest-to-graze ratio for each independent grazing contrast (comparing an SRG treatment with a CG or ungrazed treatment). Stocking rate difference referred to the difference in stocking rate (animal units ha^{-1} $year^{-1}$) between SRG and CG treatments in a contrast, where lower means the stocking rate of the SRG treatment was lower than the CG treatment, and higher means the stocking rate of SRG was greater than the CG treatment compared. Climate zone referred to the Koppen-Geiger climate classification where the study was undertaken (tropical, arid/semi-arid, temperate or cold/continental; Peel et al., 2007). The rest-to-graze

ratio referred to the length of time pasture was rested relative to the length of the graze period. For example, a pasture that is rested for 5 weeks and grazed for 1 week will have a rest-to-graze ratio of 5:1. Information on geographic region, stock type, method of calculation of richness, diversity and groundcover, the type of diversity index and unit of animal production per hectare was also recorded (see Appendix 1 for definitions). Where this information was not provided either in the text or as supplementary information, the studies were not included in meta-analyses. Where the same data were reported in multiple papers, data from only one paper was included.

A total of 220 articles were retained from the results of our literature search, however only 176 of these articles contained data suitable for the meta-analyses (Appendix 2).-Of these, we analysed 76 studies relating to biomass, 79 for individual animal weight gain, 38 for groundcover, 36 for animal production per hectare, 28 for plant richness and 18 for diversity (across both the SRG–CG and SRG–ungrazed datasets). Overall the majority of contrasts between SRG and CG treatments were compared at equal stocking rates for each response variable except plant richness Appendix 3).

We undertook meta-analyses using the metafor (v.1.9-6) and metagear (v.0.4) packages (Viechtbauer, 2010) within the R open-source software environment (Version 3.4.0; R core team 2017). We calculated the effect sizes of each comparison as the log response ratio ($\ln RR$) (Hedges and Olkin, 1985):

$$\ln\left(\frac{X_T}{X_R}\right) \qquad \textbf{Equation 1}$$

where X_T was the mean value of the response variable (in either the ungrazed 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 ungrazed 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, 2016). 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 variance among studies. The sampling variance of $\ln RR$ was calculated as:

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

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 and common treatments were weighted with a variance–covariance matrix that accounted for this dependency (Lajeunesse, 2011, 2016). See Appendix 7 for information on variance computation for studies with missing variance data.

Multi-level random-effect (MLRE) models were fitted for the effect sizes of each response variable for SRG–CG and SRG–ungrazed 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). We included a nested random term region/study/livestock type in all models. In the models of plant species richness, diversity and groundcover, the random effect was further nested to account for calculation method of how species richness and diversity were estimated in each

study. In models of the effect size of plant diversity, the random term was further nested to include the diversity index (Appendix 1). In the animal production per hectare models, we added an additional nested level for the type of unit (Appendix 1).

MLRE models were initially fitted without explanatory variables, to assess if the overall effect size differed significantly from zero (i.e. a null model). Null models included all studies of the response variable of interest. To explain variability in effect size, we fitted MLRE models with the explanatory variables of climate zone, stocking rate difference and rest-to-graze ratio as fixed effects, using only the studies where these data was available. It was not possible to calculate a rest-to-graze ratio for every study due to a lack of information provided in studies (Appendix 4). Due to the higher proportion of missing rest-to-graze information in datasets of plant richness, diversity, biomass and groundcover, the effect of rest-to-graze ratios were tested in separate models for these variables. Stocking rate difference was not tested for animal production per hectare studies, as this was confounded with the response. We tested the significance of individual factor levels by calculating their marginal means for each response variable. Effect-size heterogeneity in each model was assessed using the least squares extension of Cochran's Q-test (Q_E; Hedges & Olkin, 1985; Viechtbauer, 2010). A significant Q_E-value indicated that effect size differed more than expected due to sampling variability (Hedges & Olkin, 1985).

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) (Appendix 5; Viechtbauer, 2010). Model selection was guided by assessment of the Akiake's Information Criterion (AIC) and its small sample size correction (AICc). In selecting models for plant species richness and diversity, we focussed on AICc, given the small sample sizes. A difference in AIC or AICc value of >2 was considered better than the null model. We

report only the best fitting models in the results. Homogeneity of variance was assessed by visualising model residuals against fitted values. 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. All models had a significant amount of residual heterogeneity, despite inclusion of explanatory variables. Publication bias was assessed using Egger's regression test (Egger et al., 1997; Sterne & Egger, 2005), which is appropriate for use with MLRE models (Habeck and Schultz, 2015). See Appendix 7 for further information on publication bias.

2.4 Results

2.4.1 Biomass and groundcover

Overall, biomass and groundcover were significantly higher under SRG compared to CG management (biomass: $Z = -4.17$, $p < 0.001$; groundcover: $Z = -2.67$, $p = 0.008$; Figure. 2.1). However, differences in stocking rate affected the biomass result (Appendix 6), with biomass being significantly greater under SRG when the stocking rate of SRG was lower than that of CG ($Z = -3.38$, $p = <0.001$), but there was no difference in biomass between SRG and CG when the stocking rate of SRG was equal to ($Z = -1.83$, $p = 0.067$) or greater than CG ($Z = -0.71$, $p = 0.473$). Biomass and groundcover were significantly lower under SRG than ungrazed (biomass: $Z = 2.39$, $p = 0.017$; groundcover: $Z = 3.04$, $p = 0.002$, Figure. 2.1).

Biomass increased under SRG relative to CG as the rest-to-graze ratio increased ($Z = -16.31$, $p < 0.001$, Figure. 2.2a). When stocking rates of SRG were equal to CG there

was no difference in biomass between SRG and CG at low rest-to-graze ratios, but biomass was greater under SRG when rest-to-graze ratios were higher than 6:1. Biomass decreased in SRG relative to ungrazed systems as the rest-to-graze ratio increased ($Z = 7.46$, $p < 0.001$, Figure. 2.2b), becoming greater under ungrazed than SRG above a rest-to-graze ratio of 4:1.

Groundcover under SRG relative to CG increased as the rest-to-graze ratio increased ($Z = -29.52$, $p < 0.001$). When the SRG stocking rate was equal to CG, groundcover was always greater under SRG (Figure. 2.2c). Groundcover increased under SRG relative to ungrazed as the rest-to-graze ratio increased but was always greater under ungrazed than SRG ($Z = -4.92$, $p < 0.001$; Figure. 2.2d).

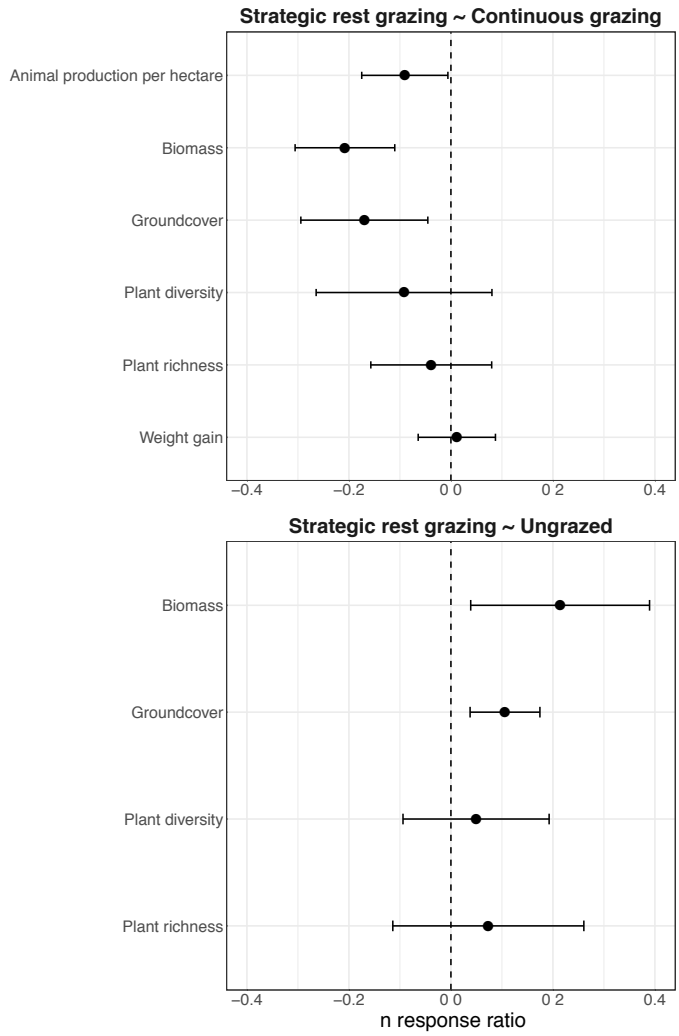

Figure 2-1 Effect size ±95% confidence intervals of null (overall) models for SRG–CG and SRG–ungrazed datasets

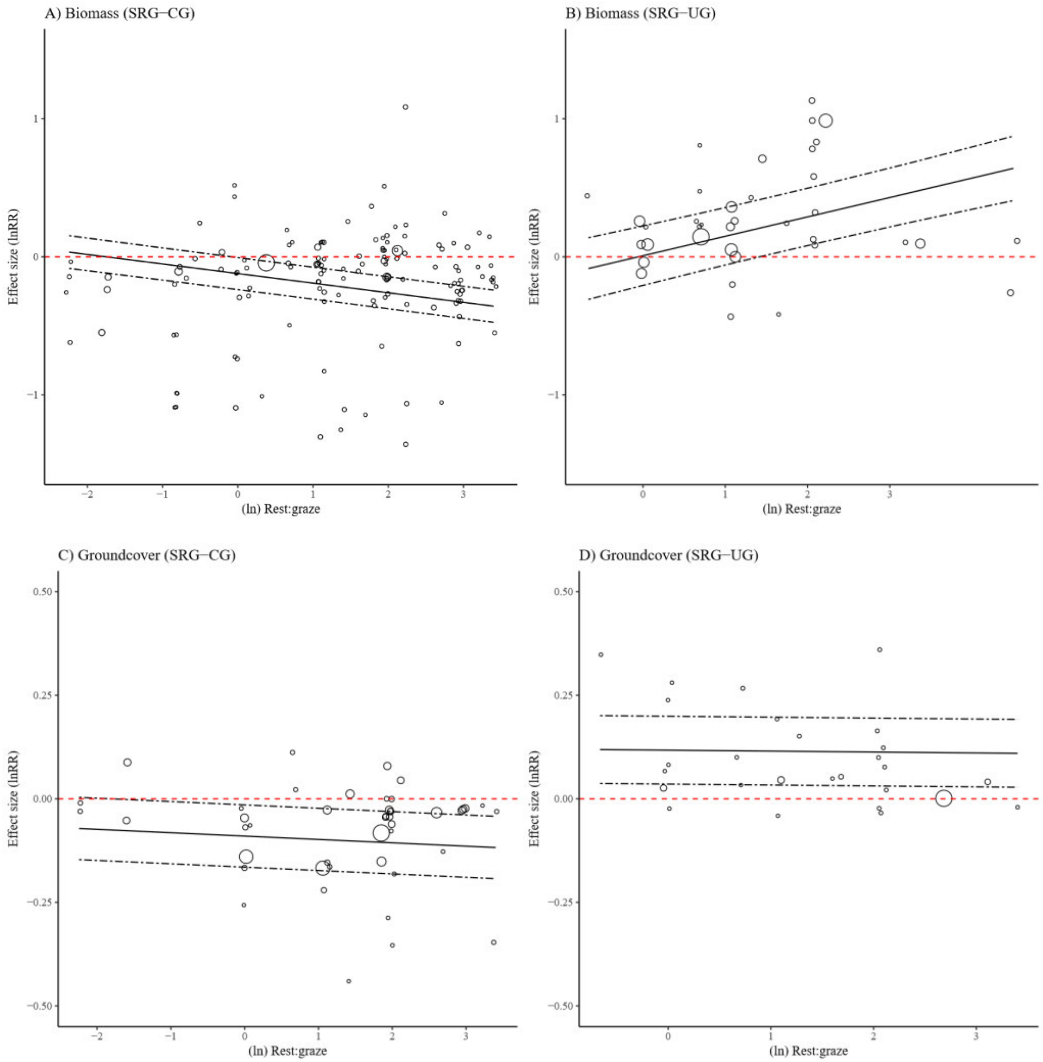

Figure 2-2 (a-d). Rest-to-graze ratio relationship with effect size for: (a) plant biomass SRG–CG contrast (at equal stocking rates only); (b) plant biomass SRG–ungrazed (UG) contrast; (c) groundcover SRG–CG contrast (at equal stocking rates only); d) groundcover SRG–ungrazed contrast; (e) weight gain per animal SRG–CG contrast; (f) animal production per hectare SRG–CG contrast; (g) plant richness SRG–CG contrast; and (h) plant richness SRG–UG contrast. Dotted lines represent the ±95% confidence interval. The observed effect sizes are drawn proportional to the inverse of the corresponding standard errors (i.e. larger circles reflect a smaller standard error and were given greater weighting in the analysis)

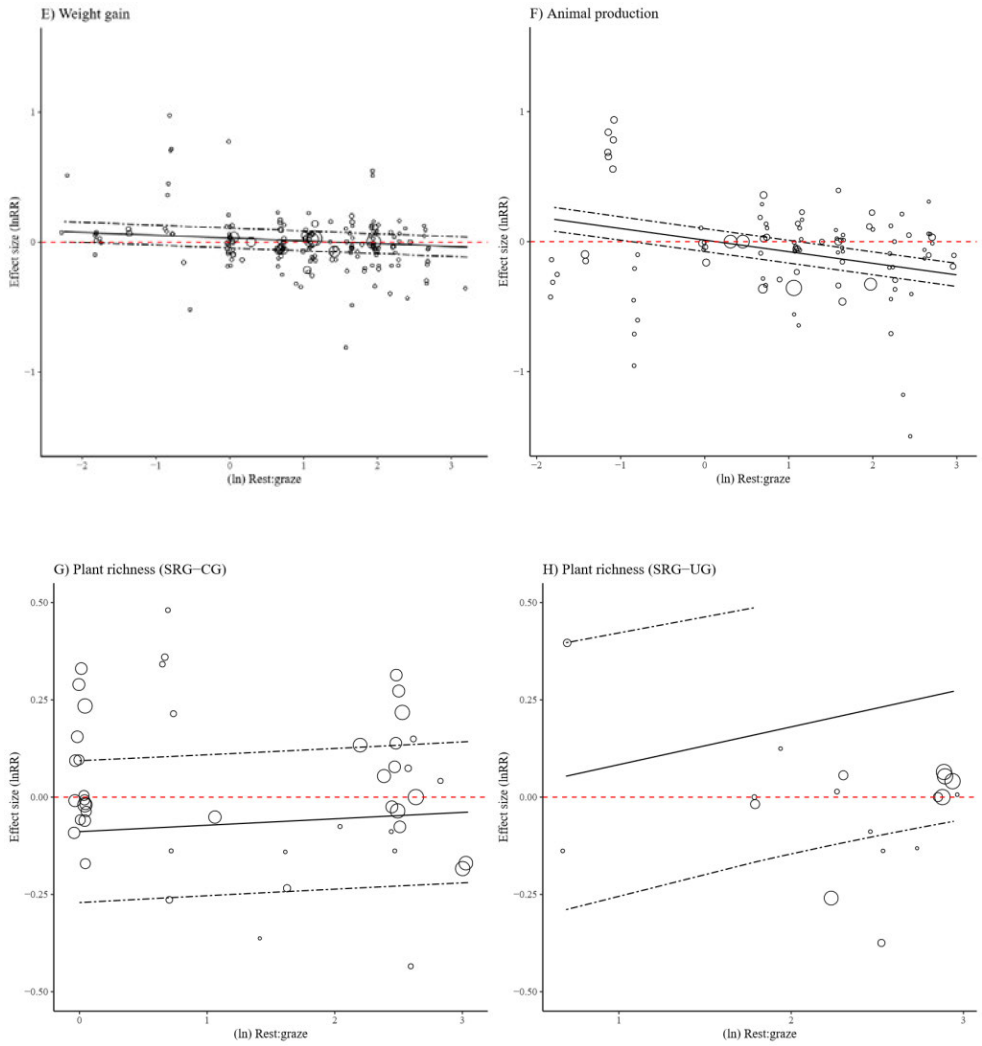

Figure 2-3 (e–g). Rest-to-graze ratio relationship with effect size for: (a) plant biomass SRG–CG contrast (at equal stocking rates only); (b) plant biomass SRG–ungrazed (UG) contrast; (c) groundcover SRG–CG contrast (at equal stocking rates only); (d) groundcover SRG–ungrazed contrast; (e) weight gain per animal SRG–CG contrast; (f) animal production per hectare SRG–CG contrast; (g) plant richness SRG–CG contrast; and (h) plant richness SRG–UG contrast. Dotted lines represent the ±95% confidence interval. The observed effect sizes are drawn proportional to the inverse of the corresponding standard errors (i.e. larger circles reflect a smaller standard error and were given greater weighting in the analysis).

2.5 Animal Weight Gain and Animal Production per hectare

Overall, there was no difference in weight gain between SRG and CG systems ($Z = 0.30$, $p = 0.765$) but animal production per hectare was significantly greater under SRG than CG ($Z = -2.09$, $p = 0.036$, Figure. 2.1). Results did not differ when the stocking rate difference or climate zone were considered

As the rest-to-graze ratio increased, there was greater weight gain and animal production per hectare under SRG compared to CG (weight gain: $Z = -8.51$, $p < 0.001$, Figure. 2.2e; animal production per hectare: $Z = -18.77$, $p < 0.001$; Figure. 2.2f). Significantly greater animal production per hectare under SRG was found at rest-to-graze ratios greater than 3:1, but at small rest-to-graze ratios (below 0.4:1) production per hectare was greater in CG systems. No significant difference was found for weight gain between the SRG and CG systems within the range of rest-to-graze ratios tested in our analysis.

2.6 Plant species richness and diversity

There was no significant difference overall between the grazing treatments for either plant species richness (SRG–CG: $Z = -0.64$, $p = 0.524$; SRG–ungrazed: $Z = 0.77$, $p = 0.443$) or plant diversity (SRG–CG: $Z = -1.05$, $p = 0.296$; SRG–ungrazed: $Z = 0.67$, $p = 0.500$, Figure. 2.1). There were also no significant differences in richness and diversity between SRG and CG when considered with the stocking rate difference. Richness decreased under SRG compared to CG as the rest-to-graze ratio increased ($Z = 4.25$, $p < 0.001$, Figure. 2.2g). Despite this, there was no significant difference in richness between SRG and CG within the range of rest-to-graze ratios tested in our analyses, at any stocking rate.

In arid and semi-arid climates, species richness under SRG was lower than in ungrazed areas ($Z = 2.20$, $p = 0.028$), but there were no differences between SRG and ungrazed in other climate zones. Richness decreased under SRG compared to ungrazed as rest-to-graze ratio in SRG systems increased ($Z = 2.41$, $p = 0.016$, Figure. 2.2h). Species richness was always significantly greater for ungrazed than SRG areas above rest-to-graze ratios of 44:1. There were no differences associated with explanatory variables in comparisons of plant diversity between SRG and ungrazed systems.

2.7 Discussion

Identifying the global and regional patterns of grazing management that drive plant diversity and production metrics across continents and biomes is critical if we are to understand the likelihood of achieving production and biodiversity outcomes simultaneously. This study has addressed significant literature gaps related to better understanding ecological, biophysical and production effects of grazing systems that incorporate periods of rest compared with continuously grazed or ungrazed systems. We synthesized the current literature to demonstrate that strategic grazing incorporating periods of rest is associated with increased groundcover and animal production per hectare relative to continuous grazing practices. Importantly, the benefits to biophysical and livestock production variables increased as the length of rest relative to grazing time increased.

Biomass and groundcover are two important indicators of ecosystem functioning as they are known to stabilise soil, provide a buffer from extreme temperatures, reduce soil erosion, improve infiltration and contribute to nutrient cycling through the decomposition and mineralisation of plant material (Clarholm, 1985; Yates et al., 2000; Gardiner and Reid, 2010). Standing biomass and ground litter also provide habitat for small animals including invertebrates, birds, reptiles and small mammals (King and Hutchinson, 2007). While

groundcover was greater under SRG than CG, the response of biomass was confounded by differences in stocking rates of the treatments compared. When compared at equal stocking rates there was no significant difference in biomass between SRG and CG. Despite this, biomass became greater under SRG than CG when a rest-to-graze time greater than 6:1 was applied. The lower biomass and groundcover found under SRG than ungrazed areas was unsurprising as livestock consume plant materials, trample plant and litter and disturb the soil surface. These results indicate that although SRG increases biomass and groundcover compared to more traditional CG practices, overall grazing impacts still override any effect of management system for these attributes, regardless of climate zone. This is consistent with previous conclusions (O'Reagain and Turner, 1992; Ash and Stafford Smith, 1996).

Continuous grazing is often considered to lead to greater weight gain as livestock are able to selectively graze preferred plants (Ellison, 1960; Joseph et al., 2002; Briske et al., 2008). In contrast, smaller grazing units under SRG can lead to greater and more uniform herbage production and utilisation with consequently greater animal production per hectare (Joseph et al., 2002; Norton et al., 2013; Williamson et al., 2016). Our meta-analyses partially support these findings, as we found no significant difference in animal weight gain between SRG and CG and greater animal production per hectare under SRG than CG. Importantly, our study revealed that both animal production metrics increased in SRG systems as the rest-to-graze ratio increased. This is consistent with Teague et al. (2015) who demonstrated a positive relationship between the number of paddocks (and therefore longer rest periods) in rotational grazing systems and ecological condition and profitability. Previous studies have rarely considered the influence rest-to-graze ratios in response to grazing management, and few studies have investigated outcomes under rest-to-graze ratios larger than 10:1, which may account for the often contrasting findings and contribute to the ongoing debate surrounding the benefits of SRG strategies such as rotational grazing (Briske et al., 2008;

Teague et al., 2013). The rest-to-graze ratio is likely an important component in achieving the goal of balancing productivity and ecological sustainability in grazing systems.

Livestock grazing is often considered to have negative impacts upon species richness and diversity (MA, 2005; Steinfield et al., 2006; Lunt et al., 2007). However, previous research has typically compared CG with ungrazed areas rather than grazing systems that incorporate periods of rest. The lack of difference in species richness and diversity between SRG and ungrazed areas in this study may be a result of periods of rest allowing plants to recover and regenerate from grazing events. Lower species richness under SRG compared to ungrazed areas in semi-arid environments likely reflects the greater susceptibility of these environments to land degradation and biodiversity loss (Holechek et al., 1999; Eldridge et al., 2016). Further, the lack of difference in plant species richness and diversity between SRG and CG areas is likely because stocking rate and overgrazing are more influential drivers of floristic richness and diversity than grazing management system (O'Reagain and Turner, 1992; Ash and Stafford Smith, 1996; Provenza, 2003). Reduced richness under SRG as the length of rest increased may be a reflection of more uniform grazing patterns under SRG. Although we did not observe differences in species richness and diversity between grazing treatments, it is possible that compositional changes occurred that were not captured with the metrics we used in this study. Grazing affects species differently, with some responding positively (increaser species) and others negatively (decreaser species), thereby cancelling out the effects (Eldridge et al., 2018). Species composition and functional trait information may therefore be more informative measures (Cadotte et al., 2011; Winfree et al., 2011).

2.7.1 Directions for future research

A large proportion of studies included in our meta-analyses 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. Only a small proportion of studies included in our meta-analyses examined the effects of SRG management with rest-to-graze ratios greater than ten, thus further research into SRG systems with larger rest-to-graze ratios is needed. The significance of the stocking rate difference and the rest-to-graze ratio in many of the models highlights the need for greater consideration of these factors in future studies. Future studies would also benefit from greater attention to other important factors such as the types of rest-grazing systems, the timing of rest periods relative to periods of key pasture growth (Jones, 1933; Lodge and Whalley, 1985), management cues, sampling methods, the length of time that the grazing treatment was imposed prior to research being undertaken and the dominant vegetation type in the pasture. The large amount of residual heterogeneity in the meta-analyses indicate that these and other unexplained factors were likely influencing outcomes. Similar issues were encountered by Hawkins (2017). Unfortunately, much of this unexplained variation is challenging to overcome in the context of complex agroecological systems influenced by environmental, social and economic factors that are difficult to replicate or control for in field experiments (Heady, 1961; Briske et al., 2008; Provenza et al., 2013; Teague et al., 2013). These differences may have masked potential benefits or disadvantages of particular grazing management practices and confounded results (Briske et al., 2008). While no differences in species richness or diversity between SRG and CG or SRG and ungrazed areas were found, we caution that the small sample sizes for these analyses alongside variability in the way data was reported may be influencing this result and suggest that greater research for these ecological attributes is needed. Importantly, ecological and animal production outcomes were rarely considered simultaneously in the studies investigated, suggesting that there is a limited understanding of trade-offs between ecological and animal production objectives.

Conclusion

We show that rest periods are important to grazing sustainability and productivity. Quantifying both biodiversity and production outcomes associated with different grazing practices is urgently needed to make informed decisions about best-management practices for sustainable management of the world's grazing lands for joint production and ecological outcomes. However, our meta-analysis revealed that the predominant focus of existing studies comparing strategic-rest with continuous grazing has been on biophysical (e.g. plant biomass and groundcover) and animal production measures (e.g. livestock weight gain or animal production per area) with relatively few studies focused on biodiversity measures (e.g. species richness and diversity). If we are to further our understanding of the effects of, and the trade-offs between different grazing strategies it is important to consider effects on ecological as well as biophysical and animal production variables.

S.M., R.L. and R.R. conceived the ideas of this paper. S.M. and R.L. undertook the literature search and data compilation. L.K. and S.M. undertook the statistical analyses and L.K. prepared the figures. S.M. and R.L contributed equally to the writing with L.K. and R.R. also contributing substantially to the writing and critical editing of this manuscript. All authors gave their approval for publication.

Chapter 3 Can we achieve ecological and animal production outcomes simultaneously with grazing that incorporates strategic rest?

3.1 Abstract

With increasing pressure on the world's grazing lands to meet protein needs for a growing human population and mitigate global threats to biodiversity, understanding livestock grazing strategies that can meet the dual outcomes of animal production and biodiversity conservation is imperative. We reviewed 285 studies considering ecological and livestock production outcomes for grazing systems incorporating planned rest (Strategic Rest Grazing, hereafter SRG). Most of the studies have been conducted in North America (42%) and Australia/New Zealand (combined, 26%). Studies from these regions have largely focussed on animal production outcomes such as weight gain, animal production for a given area, reproductive rates and animal health. Studies from Europe were the next most numerous (13% of research) and focused predominantly on biodiversity outcomes. Where floral and faunal ecological responses to SRG were compared to continuously stocked areas, the majority of studies recorded either a positive response for floral and faunal richness (S) and diversity (H') or there was no difference. Where richness and diversity in SRG areas were compared to ungrazed areas, there was most often no difference between contrasting management approaches. Only three studies have considered the potential for natural tree regeneration under SRG, with all three showing favourable outcomes for SRG compared to continuously grazed areas. Eight percent of studies considered animal productivity alongside general sustainability outcomes such as groundcover and biomass. However, less than 3% of studies had simultaneously considered biodiversity conservation outcomes (e.g. floral and

faunal richness and diversity) alongside animal productivity outcomes, suggesting that there is a substantial gap in our knowledge concerning the trade-offs and synergies between biodiversity conservation and livestock production. In order to meet the dual challenges of sustained food production from grazing animals and challenges around biodiversity conservation, this knowledge gap should be addressed.

3.2 Introduction

Livestock grazing impacts approximately 26% of the world's land area (Asner et al., 2004), is an essential contributor to food and fibre production and is integral to many cultures around the world (Kemp and Michalk, 2007; Strong and Squires, 2012). While livestock grazing is a major driver of biodiversity loss and land degradation (Steinfield et al., 2006), it can support biodiversity conservation if managed appropriately (Kirkpatrick et al., 2005; Metera et al., 2010; Caballero, 2015). Appropriately managed grazing can also play an important role in creating resilient and heterogeneous landscapes (Naveh, 2005; Henkin, 2011), with the presence of people managing herbivores in such landscapes often contributing positively to sustainable social–ecological systems (Carmona et al., 2013; Horcea-Milcu et al., 2018).

Where degradation occurs as a result of past livestock grazing practices, grazing exclusion is commonly advocated as a way of restoring landscapes (Fleischner, 1994; Pettit et al., 1995; Prober and Thiele, 1995; Eldridge et al., 2016). However, this strategy inevitably comes with a loss of food and fibre production that is unacceptable where livestock are a source of income and consequently, excluding livestock altogether is not an appropriate strategy across much of the world (Neilly et al., 2016). Land-sparing options that isolate areas from agriculture, such as grazing exclusion, also lead to greater reliance on intensive production systems for livestock (Dorrough et al., 2007). Such systems have undesirable

external impacts such as pollution, animal welfare issues and clearing of land for increased crop production to produce grain that could otherwise be used directly for human food (Webster, 2017). Livestock grazing explicitly managed with ecological as well as animal production goals is an alternative strategy to grazing exclusion that could potentially support biodiversity outcomes while avoiding or minimising degradation and the negative socio-economic consequences of removing land from production.

Continuous stocking in extensive grazing units is the dominant livestock management approach in contemporary grazing systems with rotational grazing practices incorporating planned or strategic rest (hereafter SRG i.e. strategic rest grazing) less common (Earl and Jones, 1996; Dorrough et al., 2004; Kahn et al., 2010; Teague et al., 2011; Scott et al., 2013). Rotational grazing strategies provide a means of controlling animal distribution in both time and space with control most commonly achieved by increased fencing infrastructure and paddock sub-division. Herding practices such as transhumance and shepherding could also be considered a form of rotational grazing as stock are strategically moved through landscapes according to seasonal growth patterns and pastures rested for extended periods (Squires, 2012; Meuret and Provenza, 2015). Two benefits arising from greater control over animal distribution are: (1) the prevention of excessive grazing of preferred plants, and preferred areas of the landscape, as animals can be moved before damage occurs; and (2) the control can enable managers to avoid areas of the landscape at particular times of the year or season thereby making strategic rest of targeted areas possible. For example, Scohier et al. (2013) found increased diversity of insects where sheep were excluded at peak flowering time and Mavromihalis et al. (2013) found increased abundances of grazing-sensitive species where sheep were excluded in spring (although annual weeds increased). Strategic rest also enables extended recovery that can assist the restoration of ecological attributes such as groundcover and the rehabilitation of eroded areas (Teague et al., 2004). Consequently, where there is

control over animal distribution, there is likely to be greater potential to incorporate strategic, planned and extended rest that may reduce impact from livestock grazing practices potentially enabling simultaneous achievement of ecological and animal production goals.

Reviews to date comparing SRG with more continuous grazing strategies have largely focused on how the former benefits plant and animal productivity (Heady, 1961; Van Poollen and Lacey, 1979; Briske et al., 2008) and to a lesser extent rangeland condition (O'Reagain and Turner, 1992; Holechek et al., 2000) focussing on how different grazing systems impact livestock productivity rather than biodiversity outcomes (Nordborg, 2016). Generally, it has been concluded that there is little difference in outcomes for animal production (i.e. weight gain, production per unit area, reproductive success) or rangeland sustainability (i.e. maintenance of biomass or groundcover) to sustain long-term production between contrasting management systems and that total grazing pressure is a much more important driver (Gammon, 1978; O'Reagain and Turner, 1992; Ash and Stafford Smith, 1996; Holechek et al., 2000; Briske et al., 2008). Biodiversity outcomes in contrast, are typically considered by comparing grazed and ungrazed areas or areas under different grazing intensities (Fleischner, 1994; Spooner et al., 2002; Dorrough et al., 2004; Isselstein et al., 2005; Holechek et al., 2006; Lunt et al., 2007; Foley et al., 2011; Eldridge et al., 2016; Cross and Ampt, 2017).

Globally, many grazed landscapes consist of open grassy woodlands such as the Spanish dehesas, Australia's grassy woodlands, montados in Portugal and waldhudes in central and north-western Europe. In many cases these landscapes are considered to have been maintained historically using fire and/or grazing regimes that enable the balance between ongoing recruitment of woody species and minimising shrub encroachment (Pott, 1998; Bignal and McCracken, 2000; Gammage, 2011; Horcea-Milcu et al., 2018). Around the world these ecosystems are declining from a lack of natural tree regeneration under heavy

grazing pressures and the ongoing intensification of agriculture (Dorrough et al., 2004; Tscharntke et al., 2005; Fischer et al., 2009; Carmona et al., 2013) or alternatively through land abandonment where releases of grazing pressure result in shrub and/or tree encroachment (Pott, 1998; Humphrey and Patterson, 2000). Although it varies with livestock type (Dumont et al., 2007; Pauler et al., 2019), under continuous stocking, natural recruitment of trees is uncommon (Vesk and Dorrough, 2006; Fischer et al., 2009; Carmona et al., 2013). Given the keystone role that long-lived trees have in contributing to biodiversity in grassy woodland ecosystems (Bennett et al., 1994; Fischer et al., 2010), if SRG practices were able to facilitate increased natural regeneration of trees, while also preventing shrub encroachment, this would be an important contributor to biodiversity conservation in grazed ecosystems.

To understand the extent to which land-sharing options such as SRG are viable, it is necessary to consider trade-offs and potential synergies between ecological and agricultural productivity outcomes. Ideally these different outcomes would be examined simultaneously in research studies. For example, animal production per hectare or individual animal weight gain alongside increases in a plant or animal species of conservation interest. A number of authors have advocated such an approach, with calls 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). If we are to gain an understanding of the potential for dual ecological and animal production outcomes in landscapes grazed by domestic livestock, it is essential to address this knowledge gap.

In this study, we set out to: (1) investigate ecological outcomes under SRG practices comparing general richness and diversity responses of flora and wildlife under contrasting grazing management systems; (2) consider outcomes for natural tree regeneration under contrasting grazing management regimes; (3) quantify the proportion of research that has considered animal production and ecological outcomes simultaneously (i.e. using an integrated approach), and (4) determine whether this degree of integration differs between major geographic regions and climate zones.

3.3 Methods

3.3.1 Search terms and data collation

A systematic literature review was conducted using Scopus, returning articles from 1950 until November 2017. We searched for studies comparing SRG where land is rested for a planned period with either continuously grazed (hereafter CG) systems or ungrazed 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* OR 'functional diversity' OR invasive OR weed control) 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. Where more than one study was published using the same dataset (one instance) these were treated as separate studies. In total, 285 articles were retained.

For each study, we recorded the geographical region in which the study was undertaken (Europe, Eurasia - East of the Caucasus Mountains), Middle East, Africa, North America, South America and Australia/New Zealand), climatic zone (whether tropical, semi-arid, temperate, continental) based on the Koppen–Geiger Climate Classification (Peel et al., 2007) and all reported above-ground biotic and animal production response variables (Table 3.1).

Table 3.1 Response variables recorded in the 285 studies. Each response variable is denoted according to whether it is most relevant to biodiversity conservation outcomes, most relevant to animal production or relates to both outcomes. Diversity is either Shannon-Weiner or Simpson's diversity index.

Response variables relating to biodiversity conservation	Response variables relating to both biodiversity conservation and animal production	Response variables relating to animal production
Bird abundance	Grassland sward structure	Animal health
Bird community composition	Herbaceous biomass	Animal reproductive success
Bird diversity	Invasive plant management	Forage intake
Bird nesting success	Plant abundance	Individual animal weight gain
Bird richness	Plant community composition	Milk or meat quality
Insect diversity	Plant reproductive success	Pasture quality
Insect evenness	Groundcover	Production per unit area
Invertebrate abundance		Specific plant species – animal production
Invertebrate composition		Wool quality
Invertebrate richness		
Mammal abundance		
Mammal community composition		
Mammal diversity		
Mammal evenness		
Mammal richness		
Plant diversity		
Plant evenness		
Plant functional diversity		
Plant richness		
Reptile abundance		
Reptile survival		
Specific plant species - biodiversity		
Tree regeneration		

3.3.2 Analyses

Where a quantitative ecological response variable was reported (i.e. flora and fauna richness and diversity, extent of tree regeneration) the effect of SRG relative to CG and ungrazed treatments was recorded as either significantly greater, significantly lower ($P \leq 0.05$) or no different. Due to the qualitative nature of species composition data, this variable was recorded as a difference or no difference between grazing systems. Where the statistical significance of comparisons was not provided in studies, we determined trends based on the study conclusions. Where opposing trends were present across multiple contrasts, responses were denoted as no difference. Where more than one study was published using the same dataset (one instance) these were treated as separate studies. We calculated: (1) the proportion of studies conducted in different regions and climate zones and (2) the proportion of SRG–CG and SRG–ungrazed comparisons reporting a greater, lesser or similar (i.e. not different) response for each variable within regions and within climate zones.

3.3.3 Main focus of papers

To determine the proportion of research from our initial search that focused on animal production or ecological outcomes, or that considered multiple outcomes simultaneously, we allocated studies to one of five categories based on the response variables reported in the studies. The five categories are shown in Table 3.2). Finally, we calculated the number and proportion of studies in each category in each region and climatic zone.

Table 3.2 The five categories of focus based on the predominant focus of studies

Category name	Details
Animal production	Primarily focused on short-term animal production (i.e. increases in individual animal weight gain, animal production per unit area, pasture intake, milk production, wool production etc.
General sustainability	Predominantly focused on sustainable grazing practices. For example, conserving indigenous grasses for the purpose of maintaining production, groundcover and preventing shrub encroachment or loss of key forage species, but without explicit consideration of animal production outcomes
Biodiversity conservation	Predominantly concerned with biodiversity conservation but with no explicit consideration of animal production outcomes
Integrated general sustainability	Simultaneously considered animal production and general sustainability outcomes
Integrated biodiversity conservation	Focused both on animal production and conservation outcomes (i.e. explicitly considered animal production alongside specific conservation outcomes such as conserving a focal species or ecological plant or animal community).

3.4 Results

Of the 285 studies retained for the analyses, we recorded 39 response variables (Table 3.1). The most commonly reported response variables were biomass (n = 112), plant composition (n = 111), livestock weight gain (n = 98) and groundcover (n = 48). The most common biodiversity attributes recorded other than plant composition were plant richness (n = 41) and plant diversity (Shannon-Weiner or Simpson's diversity; n = 26). The number of studies recording animal richness and diversity responses was 20 and seven, respectively.

Most studies were undertaken in North America (42%), followed by Australia/New Zealand (26%), Europe (13%) and Eurasia (8%) with the remaining 11% of studies being from Africa, South America and the Middle East. More than half of the research was conducted in temperate regions with the remaining research evenly split between semi-arid and cold/continental climates (26% and 22% respectively). Only a small number of studies came from the tropics (2%).

3.4.1 Richness and diversity outcomes

Approximately half as many studies investigated faunal richness and diversity (n = 21) in response to SRG compared to plant richness and/or diversity (n = 41). Three of these faunal studies investigated mammal richness and/or diversity, seven investigated bird richness and/or diversity and 11 investigated invertebrate richness or diversity.

Under SRG grazing, 38% of studies reported greater floral richness than CG grazing while in 52% of studies there were no differences between grazing systems. Only 10% of studies reported lower richness under SRG compared to CG (Figure 3.1a). For 44% of studies, there was an increase in floral diversity with SRG compared to CG and for 50% there was no difference. There was a lower level of floral diversity in only 6% of studies under SRG compared to CG (Figure 3.1b).

For faunal richness 50% of studies found increases under SRG compared to CG, 43% found no difference and 7% of studies found lower faunal richness under SRG. For faunal diversity, there was an increase with SRG compared to CG in 60% of studies, no difference in 40% of studies and there were no studies where faunal diversity was lower under SRG compared with CG.

Approximately 41% of studies investigating differences between SRG and ungrazed areas found increases in floral richness with SRG, 46% found no difference and 14% found reduced floral richness with SRG (Figure. 3.1c). Floral diversity was found to be greater in SRG compared to ungrazed areas in 27% of studies, no different in 47% of studies and lower in 27% of studies (Figure 3.1d).

In SRG compared to ungrazed areas, faunal richness was greater in 16% of studies, no different in 67% and lower in 16% (Figure 3.1c). Faunal diversity was greater in 25% of

studies comparing SRG to ungrazed areas and for 50% and 25% there was no difference or reduced diversity respectively (Figure 3.1d).

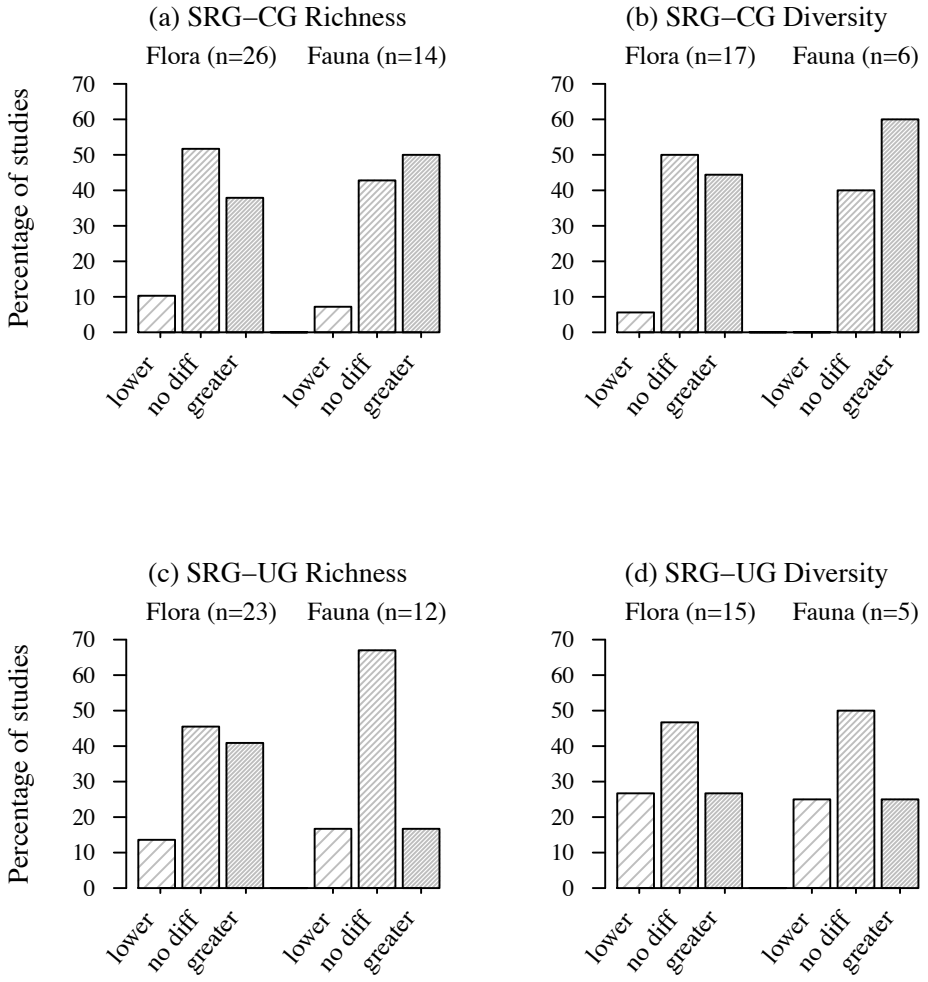

Figure 3-1 Richness and diversity responses for the two contrasts (SRG–CG and SRG–ungrazed) shown separately for plant richness and diversity and for animal richness and diversity. Figures (a) and (b) show responses for SRG compared to CG; (c) and (d) show responses for SRG compared to ungrazed areas. The Y-axis is percentage of studies with lower, no difference (no diff) or greater richness or diversity responses for SRG. SRG–CG refers to SRG–CG comparisons, SRG–UG refers to SRG–ungrazed comparisons. Lesser means a reduced response in SRG systems, no diff means no response and greater means an increased response under SRG.

3.4.2 Tree regeneration

Only three studies reported empirical data on the amount of tree regeneration in SRG systems compared to both CG and UG areas. All three of these reported increases in tree regeneration in SRG areas compared CG areas but less regeneration in SRG compared to ungrazed.

3.4.3 Area of focus

Of the 285 studies reviewed, 66% were focused on animal production and general sustainability with the remaining 34% having a biodiversity conservation focus. More specifically, 39% reported primarily animal production variables, 26% reported only general sustainability outcomes without consideration of animal production, 23% of studies reported ecological variables without considering animal production outcomes, 8% reported both animal production alongside more general sustainability variables such as groundcover and biomass, while only 3% reported both animal production and biodiversity conservation outcomes simultaneously.

3.4.3.1 Differences between geographical regions

North America dominated the research effort (42% of studies) followed by Australia/New Zealand (26% of studies). The predominant focus in these two regions was on animal production outcomes (North America: 40%, Australia/New Zealand: 53%; Figure. 3.2a and b). In both these regions there was also a significant amount of research into more general sustainability outcomes (North America: 25%, Australia/New Zealand: 25%; Figure 3.2 a, b). In North America, 22% of studies mainly considered biodiversity conservation outcomes (Figure 3.2a) while in Australia/New Zealand 15% of studies focused on biodiversity conservation (Figure 3.2b). A smaller proportion of research in these two regions combined considered sustainability alongside animal production outcomes (North America: 11%, Australia/New Zealand: 7%). Approximately 2.5% of studies in North America, and no studies in Australia/New Zealand, had considered the potential for simultaneous achievement of specific biodiversity conservation and animal production objectives (Figure 3.2a, b).

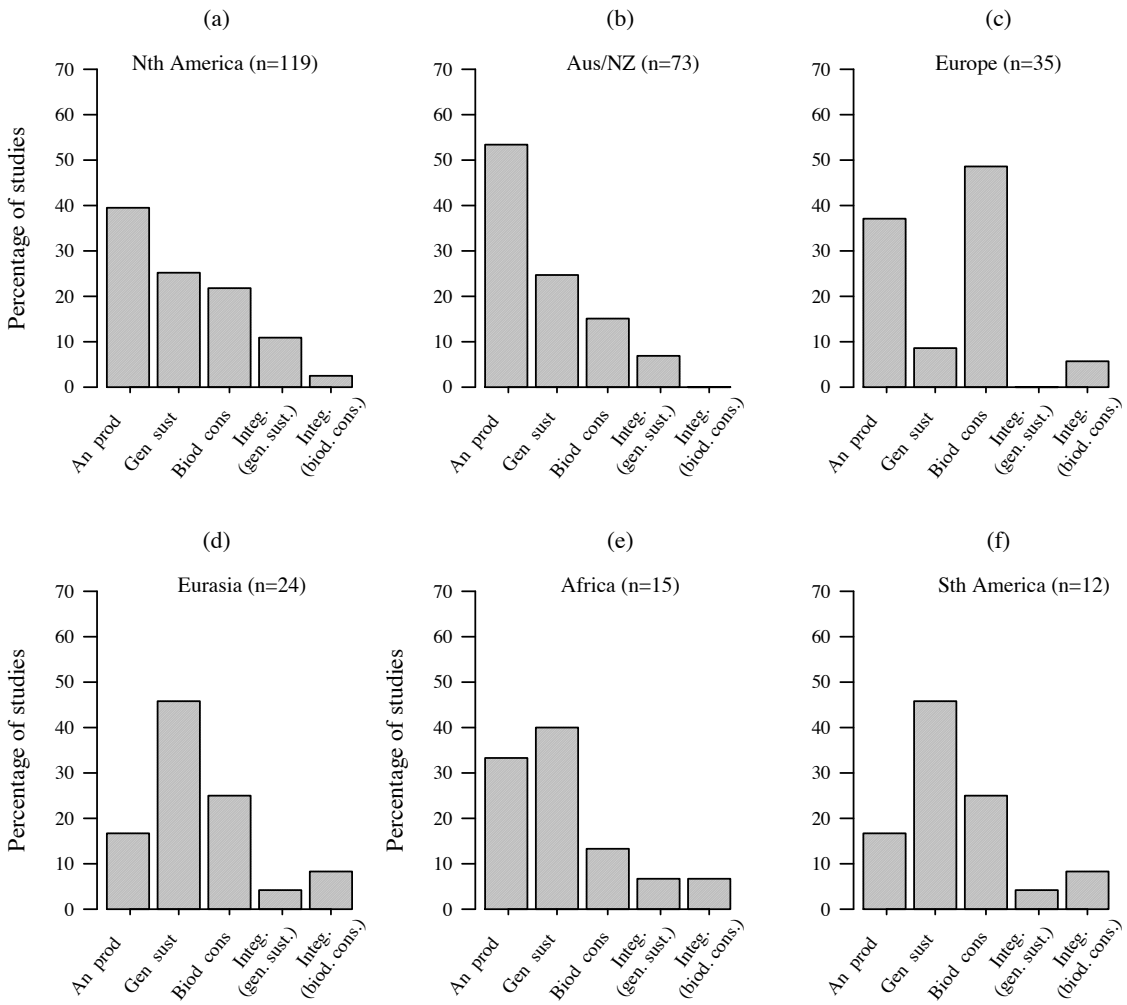

Figure 3-2 Percentage of papers in each category of dominant focus for the six regions. "An. prod." is predominantly short-term production focussed, "Gen. sust." is mainly focussed on general sustainability such as groundcover, biomass and preventing shrub encroachment with no explicit consideration of animal production outcomes. "Biod. cons." refers to studies with the major focus on specific biodiversity conservation outcomes but no explicit consideration of animal production outcomes. "Integ. (gen. sust.)" refers to studies that investigated general sustainability outcomes while simultaneously considering animal production outcomes. "Integ. (biod. cons.)" refers to studies that simultaneously considered biodiversity conservation outcomes as well as animal production outcomes. The number of studies (n) available for each of the regions is shown in parentheses

A substantial body of animal-production-focused research comes from Europe (37%; Figure 3.2c. In Eurasia the greatest focus has been on general sustainability (46%) with only 17% having focussed on animal production outcomes Figure 3.2d). More work has been done on specific biodiversity conservation outcomes in Europe (49%; Figure 3.2c) than elsewhere. Although still a low proportion of all the research conducted, an increased amount of research integrating animal production with biodiversity conservation has been conducted in Europe

(6%) and Eurasia (8%) than in the other major regions. We only found a small amount of research from Africa (5%), and even less from South America (4%).

3.4.3.2 Focus between different climate zones

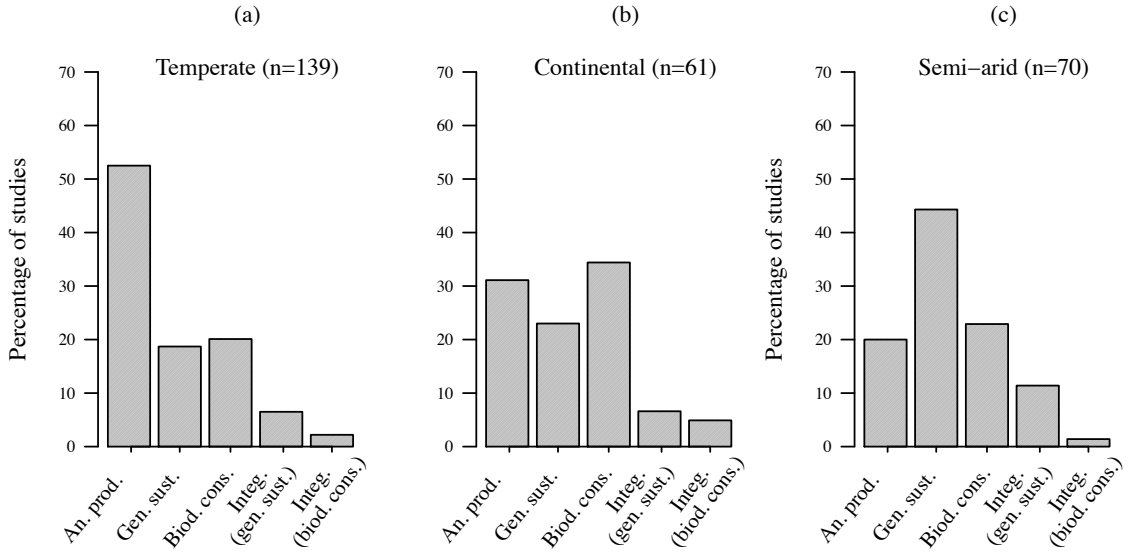

Figure 3-3 Percentage of papers from the three climatic zones with the most published studies showing the percentage for each of the five categories of focus. "An. prod" is predominantly short-term production focused, "Gen. sust." is mainly focussed on general sustainability such as groundcover, biomass and preventing shrub encroachment with no explicit consideration of animal production outcomes. "Biod cons." refers to studies with the major focus on biodiversity conservation outcomes but no explicit consideration of animal production outcomes. "Integ. (gen. sust.)" refers to studies that investigated general sustainability outcomes while simultaneously considering animal production outcomes. "Integ. (biod. cons.)" refers to studies that simultaneously considered conservation outcomes as well as animal production outcomes. The number of studies (n) available for each of the regions is shown in parentheses

Differences in major focus in the different climatic zones was apparent. Research in temperate areas mostly focussed on short-term animal production outcomes (53%) and a similar amount of attention has been given to general sustainability outcomes as to biodiversity conservation goals (19% and 20% respectively). In temperate areas, only 2% of the research considered both biodiversity conservation and animal production outcomes together and only 7% of studies considered general sustainability outcomes alongside animal production (Figure 3.3a). In areas with a continental climate, there was also a strong focus on animal production outcomes (31% of studies) with a slightly greater number of studies considering mainly biodiversity conservation (34%; Figure 3.3b). Twenty-three percent of studies in continental climatic zones focused on general sustainability outcomes with 7% and

5% considering the integration of general sustainability or biodiversity conservation, respectively, alongside animal production outcomes. Research in more arid areas was mostly around general sustainability (44%; Figure 3.3c) with 20% and 21% focusing on animal production and biodiversity conservation, respectively (Figure 3.3c). There was a greater amount of research in more arid areas (compared with other major climatic zones) that integrated animal production with general sustainability (11%; Figure 3.3c).

3.5 Discussion

To date, most previous reviews of SRG have focussed largely on animal production outcomes or on sustaining landscapes for continuing animal production rather than more specific biodiversity goals. The synthesis here indicates there is potential for improved ecological outcomes with SRG when compared to more continuous grazing practices but, despite this, only a small proportion of research has intentionally considered ecological and animal production outcomes simultaneously. The focus on animal production outcomes with little consideration of other values from rangelands is unsurprising given the dominant productivist paradigm of agriculture in contemporary times (Lang and Heasman, 2004; Richards and Lawrence, 2009) and while the production of food and fibre is clearly an important goal for human use of the world's grazing lands, there are many other goods and services that come from these landscapes as well including water purification (Egoh et al., 2011), carbon sequestration (Conant et al., 2001; Eyles et al., 2015), mitigation of temperature extremes (Pokorný et al., 2010), silviculture (Webster, 2013) and conservation of wildlife, recreation and cultural and spiritual values (IUCN, 2008). These alternative goods and services are important both for people that live in grazed landscapes, for the broader human population as well as for other wildlife.

The large proportion of studies showing improved rather than negative outcomes for SRG compared to CG highlights that grazing practices incorporating strategic rest have potential to mitigate impacts on biodiversity. Many factors, including but not limited to, stocking rate, livestock type and underlying productivity of a landscape influence ecological impacts of grazing. In ecologically focussed studies, stocking rates are often difficult to determine (McDonald et al., 2019a), but variations in stocking rates are likely to be playing a role. However, increases in heterogeneity are also likely to be important, with increases in spatial heterogeneity typically enhancing biodiversity outcomes in landscapes (Coughenour, 1991; Van Klink and WallisDeVries, 2018). Extended rest, seasonal or otherwise, is likely to increase temporal heterogeneity in a landscape, and it may be that an overall reduction in grazing impact due to this increased heterogeneity as a result of targeted, sometimes seasonal, rest might underlie the often-favourable outcomes for biodiversity attributes in studies reviewed here. It may also be the case that measurements on floral or faunal diversity are not taken during the grazed period of the rotation, thus altering outcomes and potentially leading to underlying bias.

Given the general agreement that excessive grazing pressure on preferred plants and parts of the landscape are more important drivers of landscape degradation and biodiversity loss than changes to livestock management (Ellison, 1960; Gammon, 1978; Thurow, 1991; Teague et al., 2013) it is unsurprising that our data finds that complete removal of grazing (i.e., ungrazed areas) very often produces positive outcomes for biodiversity conservation. However, strategies that exclude grazing animals altogether cannot adequately meet production outcomes. The lack of studies that have simultaneously investigated production and biodiversity outcomes means there is little quantitative data around the extent to which animal productivity is reduced where biodiversity goals are met, or alternatively the extent biodiversity is reduced where animal productivity goals are achieved. It is logical to conclude

that enhanced biodiversity outcomes requires increased retention of vegetative biomass (i.e. resources) to provide for life-forms other than livestock (Dorrough et al., 2004). It is likely also that outcomes will differ depending on the intensity of livestock grazing. For example, Farruggia et al. (2012) found that where stocking rates were high there was a clear benefit for butterfly abundance of resting pastures during a particular season, but where stocking rates were more lenient, benefits were less clear. Enri et al. (2017) found similar animal performance between a rotational grazing system that incorporated strategic rest to favour flower-visiting insects but did not consider differing stocking rates. Clearly, it is important to understand these trade-offs to determine the economic viability of grazing systems that do protect biodiversity. Such information could also contribute to the development of niche markets to support biodiversity-friendly agricultural practices, thus creating opportunities for the uptake of biodiversity friendly practices.

In this review, we found only three studies that had considered the potential for natural tree recruitment under SRG, which arguably lies between the extremes of heavy continuous grazing and livestock exclusion. Given the keystone role mature, long-lived tree species play in facilitating high levels of diversity in grassy woodland ecosystems (Fischer et al., 2010), this lack of research attention is an obvious gap in our understanding of how to improve biodiversity outcomes in grazed landscapes . This is especially relevant considering that the three studies (Arthur et al., 2008; Fischer et al., 2009; Carmona et al., 2013) showed favourable outcomes for SRG compared to CG. Increasing research effort into the potential of different grazing systems to support the natural regeneration of trees while minimising excessive woody encroachment, has potential to achieve substantial benefits for the integration of ecological and animal production outcomes.

Although most studies focused on animal production or supporting the sustainability of grazed lands for continued animal production, the major focus of research differed between major geographic regions and climatic zones. Of the three geographic regions where most of the research has been conducted, the greater focus on biodiversity conservation in European research was in contrast to research from North America and Australia/New Zealand combined where research mainly focussed on animal production. These differences are likely due to a range of complex factors but may possibly relate in-part to historical influences. For example, North America, Australia and New Zealand were all a part of an expanding empire that was aiming to increase its wealth and power but had minimal understanding of the ecology of colonised lands (Lines, 1991; Wallach, 2005). In contrast, grazing livestock had been a part of complex agroecosystems in Europe for many thousands of years, with knowledge of the management of those grazed landscapes embedded in cultural memory in these 'Old World' landscapes (Bignal and McCracken, 2000; Varga et al., 2016; Horcea-Milcu et al., 2018). The dominant culture of colonising powers was likely influenced by scientific and industrial revolutionary thinking prevalent at the time, with a corresponding world-view that sought to dominate, control and exploit, rather than understand and work with local ecology (Merchant, 1980; Massy, 2017). Under this type of world-view, alternative values of landscapes such as ecological, cultural and spiritual well-being are considered less important than technological and materialistic progress (Wallach, 2005). If the dominant focus on animal production and economic return of North American and Australian/New Zealand research was balanced with greater consideration of ecological outcomes from grazed landscapes, a greater understanding of how to achieve these dual goals may result.

The dominance of production-focused research is also most evident in temperate regions, with a greater emphasis on general sustainability in dryland regions and studies in

areas with continental climates lying in between these extremes. Reasons for this may be in the increased productivity and greater predictability of rainfall that exists in temperate climates compared to more arid areas (Sala et al., 1988; Knapp and Smith, 2001; Li et al., 2008; Eldridge et al., 2016) with less productive ecosystems more fragile and prone to degradation (Dorrough et al., 2004). Further, in areas of higher potential productivity, it is likely that costs for inputs intended to maximise productivity are more likely to be recovered than in areas of lower fertility with less predictable rainfall (Nadolny, 1998). It is therefore understandable that research in temperate areas is more likely to focus on enhancing productivity. Conversely, historically high stocking rates and associated heavy grazing of less productive, generally more fragile, dryland areas (Tongway and Ludwig, 2002) has likely resulted in the need to determine grazing management approaches that ensure greater sustainability and arrest productivity decline.

3.6 Conclusion

In the literature on SRG, most research has focused on short-term animal productivity and the management of landscapes for ongoing animal production. Less research has targeted biodiversity conservation outcomes, especially in North American and Australian/New Zealand region where most SRG research has been conducted. Differences in focus between major geographic regions are likely partly attributable to historical and cultural differences in approaches to managing landscapes and agriculture. Although most studies examining biodiversity conservation outcomes showed favourable responses with SRG compared to CG approaches, minimal research to-date has simultaneously investigated animal production alongside specific biodiversity conservation outcomes. This suggests there is a major gap in our understanding of the potential to achieve the dual outcomes of animal production and biodiversity conservation across the world's grazing lands.

Chapter 4 Short-duration rotational grazing leads to improvements in landscape functionality and increased perennial herbaceous plant cover

nb. the term short-duration grazing has been used instead of strategic-rest grazing throughout the next three chapters for the studies on commercial grazing properties in the North West Slopes and Tablelands region of NSW.

4.1 Abstract

Livestock grazing can lead to reduced groundcover and altered composition of pastures through the loss of palatable forage species and reduced litter cover. This negatively impacts landscape function and ultimately livestock production. Grazing livestock for short periods with high animal density, followed by long rests to allow pasture recovery (short-duration grazing), could be a way to address these issues. In naturalised pastures, we assessed landscape functioning and compared the abundance of six major plant functional groups at 36 sites on 12 commercial grazing properties. Six of the properties had been managed with short-duration grazing for more than 7 years (in most cases over 10 years), while the six control properties were managed with grazing that was more typical of the region (relatively continuous throughout the year with unplanned rests). Under short-duration grazing, there was approximately 19% greater foliar cover of perennial herbaceous species with a

corresponding 14% reduction in foliar cover of introduced annual plants. Attributes relating to biophysical functioning of the landscape were enhanced by short-duration grazing, with environmental factors less important in influencing these landscape function attributes. Higher-value forage species were also more abundant on short-duration grazing properties, especially at higher rainfall sites. Conversely, species that tend to increase under heavy grazing pressures, and are of lower forage value, were less abundant under short-duration grazing. Despite the changes in pasture composition in response to grazing management there was a large amount of unexplained variation in herbaceous community composition. This study demonstrates benefits for landscape function and naturalised pasture composition under short-duration grazing that has been in place for several years compared with more usual grazing practices.

Additional keywords: grassland composition, ground cover, litter, landscape function analysis, naturalised pastures, planned rest, increaser species

4.2 Introduction

Livestock grazing occupies approximately 26% of the world's land area (Asner et al., 2004) and can lead to landscapes of high diversity and function when managed appropriately (Watkinson and Ormerod, 2001; IUCN, 2008). However, livestock grazing can also be a major contributor to soil degradation processes and losses of biodiversity where stocking rates are inappropriately high (Steinfield et al., 2006). Globally, there are generalised responses of plant communities to livestock grazing, with the foremost being the loss of perennial herbaceous species (Díaz et al., 2007). For example, grazing promotes replacement of taller species by shorter plants with prostrate species becoming more common under high stocking rates and continuous grazing of areas (Dyksterhuis, 1949; Lodge and Whalley, 1989; Teague et al., 2011). Prolonged, inappropriately high stocking rates result in plant communities dominated by annual, ruderal species or bare ground (Dyksterhuis, 1949; Arnold, 1955; Ellison, 1960; Grigulis et al., 2001; Díaz et al., 2007; Eldridge et al., 2016).

From a livestock production perspective, an important role of perennial herbaceous species is forage provision (Kemp and Dowling, 2000). In combination with plant litter in interstitial spaces between tussocks, perennial plants also slow the movement of water downslope, facilitate water infiltration and reduce evaporation from the soil surface (Thurow, 1991; Yates and Hobbs, 2000; Roth, 2004; Tongway and Hindley, 2005) thereby protecting soils and promoting plant growth (Lang, 1979; Tongway and Hindley, 2005). Perennial tussock species and interstitial litter are also important for building organic matter in the soil, providing a habitat for above and below-ground invertebrates that are important for soil health (Clarholm, 1985; Wardle et al., 2004; King and Hutchinson, 2007) and that support biodiversity by providing a food source for higher trophic levels (Recher and Lim, 1990). Loss of perennial cover is also a major cause of soil structural decline (Greenwood and

McKenzie, 2001). Thus, livestock grazing practices that lead to substantial reductions in herbaceous perennial plants and litter cover result in losses of multiple important ecological functions as well as a reduction in forage availability for livestock.

Heavy stocking pressures, rather than grazing per se, is considered a primary driver of these degrading processes (Ellison, 1960; Gammon, 1978; Thurow, 1991; Greenwood and McKenzie, 2001; Teague et al., 2013). In large paddocks where stock forage continuously, heterogeneous grazing patterns are common with favoured plants and areas grazed selectively and more frequently. This leads to changes in grassland composition and soil degradation in some areas and underuse in other areas (Willms et al., 1988; Thurow, 1991; O' Connor, 1992; Bailey et al., 1996). These heterogeneous patterns in response to grazing become more apparent as paddock size increases (Senft et al., 1985; Stuth, 1991; Bailey et al., 1996).

Modern sedentary grazing practices restrict animal movements and typically have short non-grazing intervals. This style of management is common practice in commercial grazing systems in many rangeland ecosystems (Earl and Jones, 1996; Dorrough et al., 2004; Teague et al., 2011; Scott et al., 2013; Shakhane et al., 2013). An alternative to sedentary or continuous grazing practices is rotational grazing management that incorporates short graze periods with large mobs of livestock followed by extended periods of rest. In these systems, the time when the animals are returned is governed by the needs of the pastures rather than the animals - livestock are grazed for a short time and returned to pastures only after a monitored rest based on the recovery rates of key perennial pasture species. This style of management is described in McCosker (2000); Barnes et al. (2008); Scott et al. (2013) and Steffens et al. (2013) with early influences including Voison (1959); Acocks (1966) and Savory and Parsons (1980). In this study this style of management is referred to as short-duration grazing (hereafter SDG).

Grazing practices that aim to graze and rest pastures at key times can influence grassland species composition and overall production (Jones, 1933; Lodge and Whalley, 1985; Hidalgo and Cauhépé, 1991; Earl and Jones, 1996; Ash et al., 2011; Steffens et al., 2013; Zhang et al., 2015) and tactical rest is considered an important aspect of good grassland management (Cook et al., 1978; Kemp and Dowling, 2000; Behrendt et al., 2013). A large body of research has been conducted examining the impacts of varying rotational grazing regimes that strategically graze and rest pastures, compared with more continuous grazing practices on pasture composition and productivity (for reviews see Gammon (1978); Holechek et al. (2000); Briske et al. (2008)). While it is acknowledged that this research struggles to account for complex human and ecological factors (Briske et al., 2011; Provenza et al., 2013; Teague et al., 2013), general conclusions are that there are few advantages of rotational over continuous grazing practices with stocking rates being the most important driver impacting vegetation and pasture productivity (Gammon, 1978; Briske et al., 2008).

Regardless of these conclusions, many practitioners and advocates maintain that various forms of SDG result in improved animal production and pasture sustainability compared with more usual styles of management where grazing is relatively continuous throughout the year and rest is unplanned (Joyce, 2000; Mason and Kay, 2000; McCosker, 2000; Sparke, 2000; Norton and Reid, 2013). There is also a significant body of literature discussing reasons for the discrepancies between researcher-based systems and larger-scale studies that consider the complexity inherent in these systems (Norton, 1998; Briske et al., 2011; Barnes and Hild, 2013; Norton et al., 2013; Provenza et al., 2013; Steffens et al., 2013; Teague et al., 2013). Despite this extensive body of research, there is a surprising lack of research investigating outcomes on commercially managed properties, particularly over longer timeframes (Barnes et al., 2008; Barnes and Hild, 2013; Norton et al., 2013; Provenza et al., 2013). Many differences exist between researcher controlled and commercial operations. Two important

ones are that grazing units under continuous grazing are typically much larger than in research situations (Barnes et al., 2008; Norton et al., 2013; Teague et al., 2013) and that pasture changes are a response to complex and longer-term factors where managers have had to adapt to a complex array of influences including climatic, economic, social and political factors (Provenza et al., 2013). Therefore, while researcher-controlled, reductionist systems are valuable for understanding processes relating to soils, water, plants and herbivores (and their interactions), they are arguably less appropriate for understanding property or landscape-scale change (Teague et al., 2013). Working on commercial grazing properties that have been managed with SDG for a number of years provides an opportunity to circumvent the limitations of reductionist research frameworks.

We compared naturalised pastures on commercial properties managed with SDG with pastures on nearby properties managed in ways more typical of regional practice. We tested whether: (1) there would be changes in the abundance of herbaceous perennial plants and other major plant functional groups, and (2) the amount of overall ground cover and landscape function attributes would differ between SDG sites and RP sites. We hypothesised that there would be favourable outcomes for all factors under SDG management.

4.3 Methods

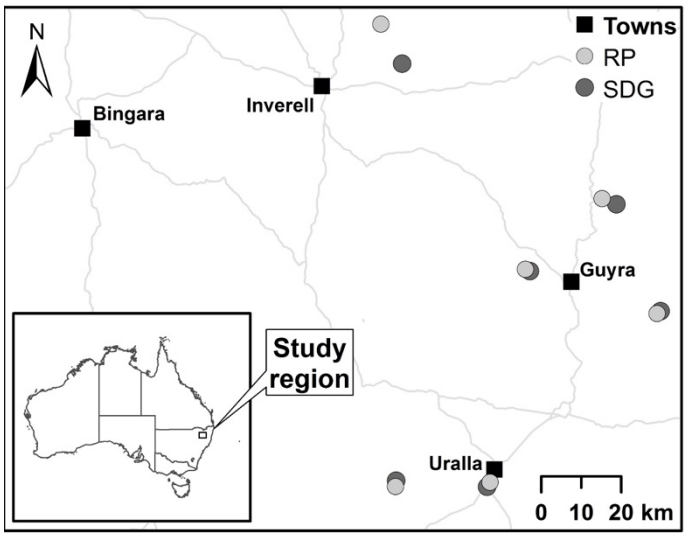

Figure 4-1 Location of the study region showing major towns (black squares) and the location of the 12 study properties. Properties that implemented grazing management more typical of regional practice are light grey circles; SDG properties are dark grey circles

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4.3.1 Location

The study was based on the Northern Tablelands and Slopes of New South Wales, Australia, between 29° 38′ S, 15° 46′ E and 30° 39′ S, 150° 27′ E (Figure. 4.1). Altitudes ranged from almost 700 m a.s.l. to just below 1400 m a.s.l. The climate of the region is classified as sub-humid with summer-dominant rainfall. Winter months are dominated by cool dry air from the continental interior or from the Southern Ocean. Severe frosts can occur in winter and there is a rainfall and altitudinal gradient across the region, with generally greater precipitation and lower evapotranspiration at higher elevations in the east (Lea et al., 1977; Lodge and Whalley, 1989). The geology of the region is complex and largely determines soil characteristics (Lea et al., 1977; Lodge and Whalley, 1989). Tertiary basalts are mostly found at higher elevations and are underlain by late Permian granites intruded into Palaeozoic sediments. Accordingly, there are three major soil types in the region, fertile basaltic soils, infertile coarse to fine granitic soils and relatively infertile soils that are derived from metamorphosed sediments known locally as 'trap' soils.

4.3.2 Site selection

We selected naturalised pasture sites (terminology from Allen et al. (2011) on 12 commercial grazing properties. Six of the properties were managed with SDG and six with what we defined as usual regional practice (hereafter RP). Five of the six paired sites were at higher tableland sites, with one property pair located on the western slopes of the study area. Details about management practices were determined through initial contact with managers, and then later with a formal questionnaire. On SDG properties, stock grazed intensively at high stocking densities for just a few days, in some cases 1–2 days, with pastures then rested until key forage species and pasture herbage mass had recovered. The decision to return stock was based on pasture monitoring with the length of rest varying with growing conditions. This planned rest resulted in overall rest-to-graze ratios (amount of time rested compared to amount of time grazed) of between 13:1 and 60:1 (median = 27:1). On individual properties this ratio varied at different times depending on climatic conditions. Some SDG managers implemented very short graze periods (1–2 days) while others grazed for several days. Smaller paddock sizes are necessary to implement the large rest-to-graze ratios typical of the SDG approach to management with paddock sizes on SDG properties between 2–16 ha (median = 10 ha) in this study. These smaller paddock sizes necessitate increased density of fencing and water infrastructure (Savory and Parsons, 1980; Sherren et al., 2012). Managers implementing SDG minimised the use of external inputs including minimising hand-feeding of stock during colder winter months and extended dry periods and limited application of inorganic phosphorus-based fertilisers (only one of the six SDG properties had applied superphosphate in the past 5–10 years as opposed to all six of the RP properties). One of the SDG properties applied poultry manure. On the properties examined, SDG management had been in place for between 7 and 25 years (median = 15 years). The contrasting RP management approach was more typical of grazing management on naturalised pastures in

the region. This was mainly continuous, with some rotation of stock where forage was limiting, but with relatively long graze and short rest periods in response to forage availability (Scott et al., 2013; Shakhane et al., 2013). In this study, rest periods on RP properties had rest-to-graze ratios that ranged from 0 to 3:1 (median = 0.5:1) and paddocks were large (median 20 ha, range 12–272 ha) in comparison to SDG paddocks. Superphosphate application on RP properties was typically on a calendar rotation (either annual or biennial) and stock were hand-fed to a greater extent than on SDG properties. The RP properties had been managed in the same way at least for the past 10 years and were judged to be well-managed by local standards.

Few pastoral managers had implemented SDG management in the region for greater than 10 years. Consequently, local networks were used to locate suitable SDG properties, with each property paired with a suitable neighbouring or nearby RP property. While it is acknowledged that this was not a random selection of participants, given the constraint of a low number of properties managed with SDG for an adequate timeframe it was the only practical way of selecting sites that respected neighbour relations and controlled for local soil and climatic conditions.

Livestock type were predominantly cattle (*Bos taurus*), with the exception of four properties (three SDG and one RP) that also ran sheep for fine-wool production. The study design and low number of suitable SDG properties precluded the two livestock types from being treated separately in the study.

Due to varying levels of engagement with producers, only coarse information on stocking rates could be obtained from the formal survey undertaken. Stocking rates are listed in Appendix 8. One of the criteria for the initial selection of properties for the study was that they were commercial enterprises. Therefore, while the stocking rate data was relatively

coarse, we consider that as grazing management is a trade-off between maximising yield and minimising the risk of degradation (Jakoby et al., 2015) it was a reasonable assumption that managers stock according to their profitability goals balanced with the need to sustain their resource-base.

4.3.3 Collection of plant data

We selected three open pasture sites with similar soil type, aspect and slope to the corresponding paired property on each of the six pairs of properties. Sites were at least 500 m apart ensuring that on SDG properties sites were representative of pastures at differing stages of the grazing rotation. One quadrat of 30 m^2 (5 × 6 m) representative of the surrounding pasture was surveyed for all vascular plant species in 2 years (n = 36, 2015; n = 35, 2016). We conducted surveys in late summer to early autumn when pasture species were easiest to identify (Reseigh, 2004; Shakhane et al., 2013). Species incidence was recorded, percentage of foliar cover estimated for all species in each quadrat and plant and litter cover visually estimated and combined to give overall cover.

Each site was broadly categorised according to the amount of herbage mass retained in pastures with categories ranging from very low (>300 kg ha^{-1}) to high >2300 kg ha^{-1}. Estimations were made using a modified photo-quadrat technique after Morgan et al. (2018). For the estimations, six 50 x 50 cm sub-quadrats within nineteen 5 x 6 m quadrats of differing biomass amounts were cut and oven-dried at 40°C for 48 hrs. These calibration samples were then weighed and average dry matter for each 5 x 6 m calibration quadrat calculated. Photographs of each plant survey plot were then compared to photographs of the biomass calibration plots and plant survey plots then allocated to one of six categories of pasture biomass. While greater accuracy could have been obtained using alternative methods

that quantified rather than broadly categorised biomass, this method was considered appropriate in the context of the study.

Plants were identified to species level either in the field or by collecting specimens for keying out. Nomenclature for all seed plants other than Poaceae followed Harden (2000–2002). For Poaceae it followed Ausgrass 2 (Simon and Alfonso, 2011) and Grasses of NSW (Jacobs et al., 2008). Plants were allocated to six functional groups to compare their relative abundance between the two commercial grazing regimes. These groups were: (1) all perennial species; (2) introduced annual and short-lived perennial species; (3) introduced C3 perennial pasture plants that historically had been brought into the region as forage and are considered desirable introduced perennial plants in a pasture mix; (4) native C3 perennial grasses that are considered good forage especially during the winter feed-gap (5) native C4 perennial grasses, and (6) two native perennial grasses, *Bothriochloa macra* (Steud.)S.T. Blake and *Cynodon dactylon* (L.) Pers., which, while being valuable ground cover, are tolerant of heavy grazing and increase in abundance under high grazing pressure (Cook et al., 1978; Oudtsdoorn, 1992). The introduced annual or short-lived perennial grass, *Eleusine tristachya* (Lam.) Lam., was also analysed separately as it was present in high abundance compared with other introduced annual species. Leguminous species were too rare across both management categories to be analysed as a separate functional group. A list of plant species and their functional group allocations is presented as supplementary material (Appendix 9).

The introduced C3 perennials included in the analysis were: the grasses *Dactylis glomerata* L., *Lolium sp.*, *Festuca arundinacea* Schreb., *Phalaris aquatica* L.; the legumes, *Trifolium fragiferum* L., *T. pratense* L. and *T. repens* L. and the forbs, *Plantago lanceolata* L. and *Cichorium intybus* L.. While the majority of these plants had been deliberately

introduced into the region for livestock production, they had not been sown into pastures in this study for at least the past 10 years, and in most instances much longer. Appendix 8 details current, past and historical fertiliser application and past and historical cultivation of sites.

4.3.4 Landscape function analysis

We performed landscape function analysis (LFA; Tongway and Hindley, 2005) to assess whether there would be a difference between management approaches in the capacity of pastures to retain soil/sediment, water and cycle nutrients in the ground-layer. Landscape function analysis assesses the potential of the soil surface to support the growth of plants and is useful for monitoring the restoration of degraded sites over time. It also enables an assessment across sites at one point in time (Read et al., 2016). We assessed eleven soil-surface indicators (SSI's; Appendix 10) and analysed these to produce two indices (on a scale of 0–100) that related to the likelihood of soil/sediment (stability) and water (infiltration) being retained in the landscape rather than lost as run-off. A third index (nutrient cycling) related to the potential for effective cycling of nutrients and corresponding decomposition and mineralisation processes at and near the soil surface.

We assessed a 20-m transect in the same location as the plant quadrat for landscape function. We divided each transect into patches and inter-patches and performed a landscape organisation assessment (Tongway and Hindley, 2005). We defined a patch type as an area of pasture with increased perennial grass and litter and decreased bare ground compared to surrounding (inter-patch) areas. Patches are more likely than inter-patches to retain soil, water and nutrients in situ (Tongway and Hindley, 2005). The landscape organisation assessment determined the extent that each patch-type contributed to the overall landscape, and therefore its contribution to the landscape function assessment. For each patch-type, five

replicates per transect were assessed for each SSI. These data were analysed to give quantitative values of (1) stability, (2) infiltration and (3) nutrient cycling (Tongway and Hindley, 2005).

4.3.5 Soil analyses

We assessed each site for soil nutrient status. At each site, and for each LFA patch-type, three 5-cm diameter soil cores were collected per patch type to a depth of 5 cm. We combined these samples for analysis. Where there were two patch-types per transect, the results of the analysis were weighted according to the landscape organisation assessment (i.e., the proportion of each patch-type in the landscape). Soil samples were kept cool and returned to the laboratory for oven-drying at 40ºC for 7 days until completely dry. They were then sealed and stored in a cool, dark place for later analysis.

Soil chemical analyses determined the nutrient levels in different patch-types at each site using methods from Rayment and Higginson (1992). We ground bulked samples to 2 mm following removal of large organic matter. We determined electrical conductivity (EC) and pH in a 1:5 soil to water extract, measured exchangeable cation concentrations (Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}) using atomic absorption spectroscopy and 1-M NH_4Cl, and determined available phosphorus with Colwell P Malachite Green in 0.5-M $NaHCO_3$. Samples measuring P were passed through charcoal to remove organic matter and absorbance in a spectrophotometer measured at 630 nm. We ground subsamples of the bulked soil samples to pass through a 0.5 mm sieve and analysed a homogeneous subsample of this (0.2 g) for total C and N content by combusting at 950°C using a LECO TruSpec Simultaneous Carbon and Nitrogen Elemental Analyser.

4.3.6 Statistical analyses

Beta-distributed generalised linear mixed models (GLMMs) were used to analyse the abundance of overall plant foliar and litter cover, the foliar cover of the six functional groups and of *E. tristachya* and the three LFA indices. The beta-distribution is appropriate for proportional data as it accommodates heteroscedasticity (Cribari-Neto and Zeileis, 2010). The two categories of management and the rest to graze ratio (continuous) were treated as fixed effects in the models. Other fixed effects were (1) soil-type (categorical, basalt or granite-derived); (2) year of data collection (categorical, 2015 and 2016); (3) mean annual rainfall at each site as determined from rainfall records for each property or from the nearest Bureau of Meteorology Weather Station (BOM, 2018) (continuous); (4) recent rainfall at each site (number of rainfall events ≥ 10 mm) for the preceding month (continuous); (5) the 5-months prior to sampling (continuous), and (6) nutrient levels in the top 5-cm of soil. Correlations of different factors were determined in R (R Core Team, 2018). Where factors were determined to be correlated with each other, only one was included to avoid over-fitting models. Property pairs (replicates) were blocked to incorporate the dependency of properties within a pair and treated as random. Models were developed using the GLMMADMB package in R (Fournier et al., 2012). The most parsimonious models with the lowest value of Akaike's Information Criterion (AIC) were retained for interpretation.

Plant community data was ordinated using non-metric multidimensional scaling (NMDS). This method has been used for a similar purpose (Fensham et al., 1999; Dowling et al., 2005). The three-dimensional solution had an acceptable stress-value (0.142) and was retained for interpretation. The direction of change for the six functional groups was related to community composition using a vector fitting approach (Envfit in R) and displayed

graphically. Analyses were conducted using the Vegan package in R (Oksanen et al., 2018) and group differences tested using the Adonis function in the Vegan package.

The soil nutrients, P, Ca, C and N and the two SSI classes, litter and soil surface roughness (SSR; see Appendix 10) that we considered contributed substantially to the LFA outcomes were modelled with linear mixed effects models (LME) using the NLME package in R (Pinheiro et al., 2017). For litter and SSR, each assessment point along transects was considered a data point, with transects treated as random effects. The same management and environmental variables as in the beta-distributed GLMM models were used. The LME models were compared using Restricted Maximum Likelihood tests to determine the significance of each explanatory variable in the simplest models with the lowest AIC.

Differences in the ratio of rest-to-graze were also analysed using LME models, with property pairs treated as random. Rest to graze ratios and management approach were strongly correlated (Spearman rank order correlation; $S = 0.70$, $p < 0.001$) with SDG being characterised by significantly larger rest to graze ratios than RP ($T = 4.39$, $p = 0.007$; Table 4.2). Models that included management approach had lower AICs than models that included rest-to-graze ratio. Therefore, management approach was used in final models in preference to rest-to-graze ratio.

4.4 Results

4.4.1 Plant abundance

Perennial herb cover was significantly greater on SDG than RP sites (Figure. 4.2a; Table 4.1) and was also influenced by geological substrate, with more perennial cover on basalt-derived soils than granite-derived soils (Figure. 4.2b; Table 4.1). Conversely, the cover

of introduced annual and short-lived perennial species was lower on SDG than RP sites (Figure. 4.2c; Table 4.1) and on basalt-derived soils (Figure. 4.2d; Table 4.1).

Eleusine tristachya was a major component of the introduced annual functional group. This species accounted for 16% of cover, on average, on RP sites, but only 3% on SDG sites (Table 4.1). The abundance of *E. tristachya* was also influenced by geological substrate, being more abundant on granite soils (Table 4.1).

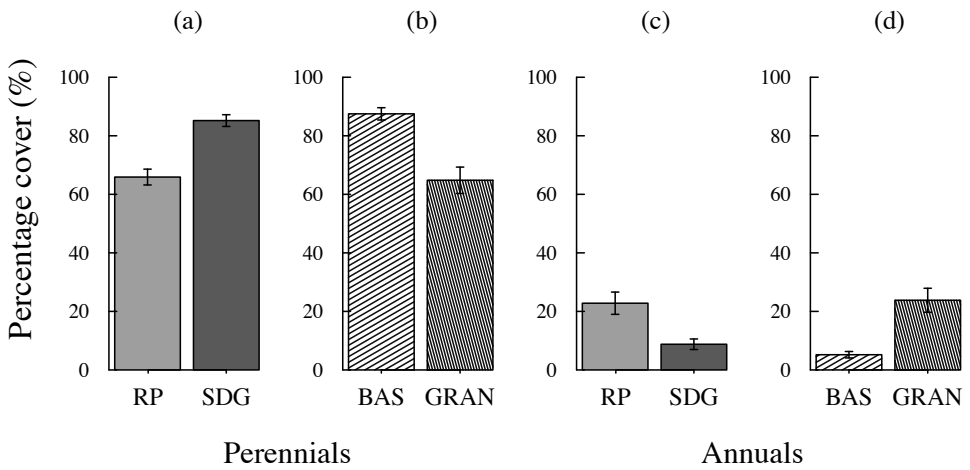

Figure 4-2 Mean percentage cover of perennial herbs and introduced annuals under contrasting management (a, c) and on soils derived from different geological substrates (b, d): BAS refers to basalt-derived soils; GRAN refers to granite derived soils. Error bars are ±1 s.e.m.

The increase in cover of introduced C3 perennial species with greater mean annual rainfall was significantly greater on SDG sites than RP sites (Figure. 4.3a and b; Table 4.1). Greater rainfall in the previous 5 months was significantly associated with a reduction in cover of these introduced C3 perennials (Table 4.1).

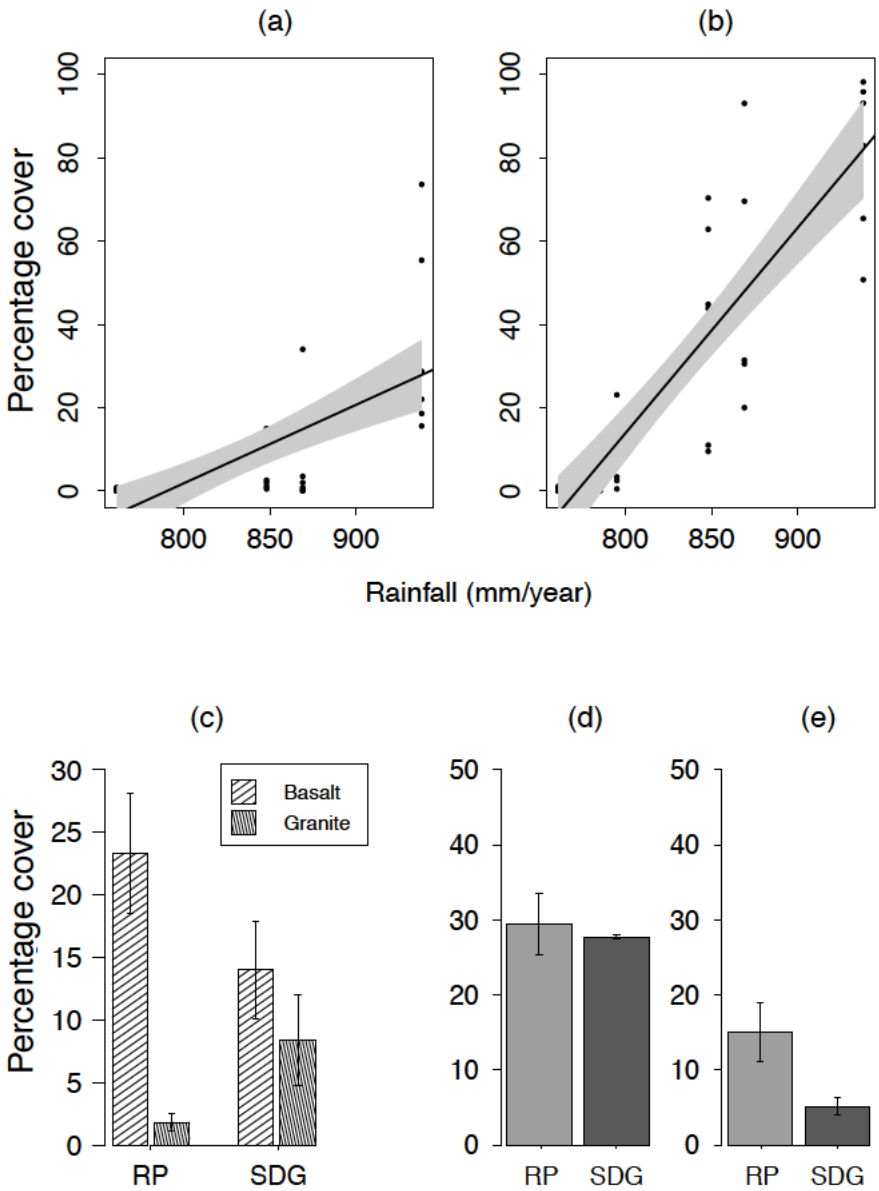

Figure 4-3 Relationship between mean percentage cover of introduced year-long perennials as mean annual rainfall increased on (a) RP (n = 35) and (b) SDG (n = 36) sites respectively; (c) average cover of native year-long green perennials under the two contrasting management approaches for the two geological substrates; (d) cover of warm-season perennial species and (e) cover of the increaser species, *C. dactylon* and *B. macra*, under the two contrasting management approaches. Least squares regression lines are shown in (a) and (b); error bars (c–e) ±1 s.e.m.

The abundance of native C3 perennial species responded differently to the two geological substrates depending on management approach. At RP sites, very little of this functional group was present on granite-derived soils while on SDG sites there was significantly greater cover of this group on this soil type (Figure. 4.3c; Table 4.1). On basalt-derived soils, there was greater cover of these species on RP sites compared with SDG sites (Figure. 4.3c), but this difference was not significant (Table 4.1).

The cover of native C4 perennials was similar between management approaches (Figure. 3d) but varied with mean annual rainfall, with greater cover of this functional group at lower rainfall sites (Table 4.1). Native C4 perennials were also present in significantly greater abundance where there had been greater rainfall in the previous month (Table 4.1).

The combined cover of *C. dactylon* and *B. macra*, was significantly greater at RP sites (Figure. 4.3e and Table 4.1) and decreased as Ca levels increased (Table 4.1).

Table 4.1 Results from GLMMADMB analyses for plant cover classes AIC's for full and final models are shown with Z – values and associated probabilities as superscripts (ns denotes not significant in the final model) Int. annuals are introduced annual and short-lived perennial herbaceous plants; IYLPs are introduced year-long perennial forage plants; NYLPs are native year-long green perennial forage plants; WSPs are native warm-season perennials; increasers are two species that increase under heavy grazing pressures (*Bothriochloa macra* and *Cynodon dactylon*); management refers to changes from RP to SDG; Mean rain is mean annual rainfall, geology is the change from granite to basalt derived soils; recent rain refers to the number of rainfall events greater than 10 mm over the previous month or the previous 5 months, as indicated by adjacent superscripts; Ca refers to calcium levels in the top 5 cm of the soil profile.

Model	***AIC***	***Management***	***Average rain***	***Geology***	***Recent rain***	***Management int. Mean Rain***	***Ca***	***Management int. geology***
Perennials	Full: – 86.7 Final: – 95.6	3.31, $p < 0.001$	*ns*	3.02, $p = 0.002$	*ns*	*ns*	*ns*	*ns*
Int. annuals	Full: – 209.9 Final: – 219.2	–2.54, $p = 0.010$	*ns*	–2.77, $p = 0.002$	*ns*	*ns*	*ns*	*ns*
IYLPs	Full: – 357.9 Final: – 347.7	*interaction*	*interaction*	*ns*	[b] 5.25, $p < 0.001$	1.96, $p = 0.05$	*ns*	*ns*
NYLPs	Full: –309.6 Final: -320	*interaction*	*ns*	*interaction*	*ns*	*ns*	*ns*	–2.42, $p = 0.01$
WSPs	Full: – 216.2 Final: – 223.0	*ns*	–6.53, $p < 0.001$	*ns*	[a] 2.28, $p = 0.007$	*ns*	*ns*	*ns*
Increasers	Full: – 500.1 Final: – 513.4	– 2.94**	*ns*	*ns*		*ns*	–1.95, $p = 0.051$	*ns*
Eleusine tristachya	Full: – 624.8 Final: – 635.1	–2.04, $p = 0.042$	*ns*	–3.99, $p < 0.001$	*ns*	*ns*	*ns*	*ns*

a – number of rainfall events (≥ 10mm) over the previous month; b – number of rainfall events (≥ 10mm) over the previous five months;

4.4.2 Non-metric multidimensional scaling

Non-metric multidimensional scaling showed a limited amount of separation of management approaches in ordination space (Figure. 4.4a). Separation between management approaches was less than the separation of plant communities according to geological substrate types (Figure. 4.4b). There was also a separation of plant community composition between tableland sites that were at a higher altitude and those that were on lower slopes to the west (Figure. 4.4c). Vectors showed a direction of change for plant functional groups that was consistent with the GLMM results (Figure. 4.4 a,b,c). While testing of differences between group means showed significant effects, the R^2 value for all three factors was low (management approach $R^2 = 0.06$, $p = 0.005$; geological substrate $R^2 = 0.15$, $p = 0.005$; between tablelands and western slopes sites: $R^2 = 0.13$, $p = 0.005$), suggesting most of the variation influencing these plant communities was not explained by our data.

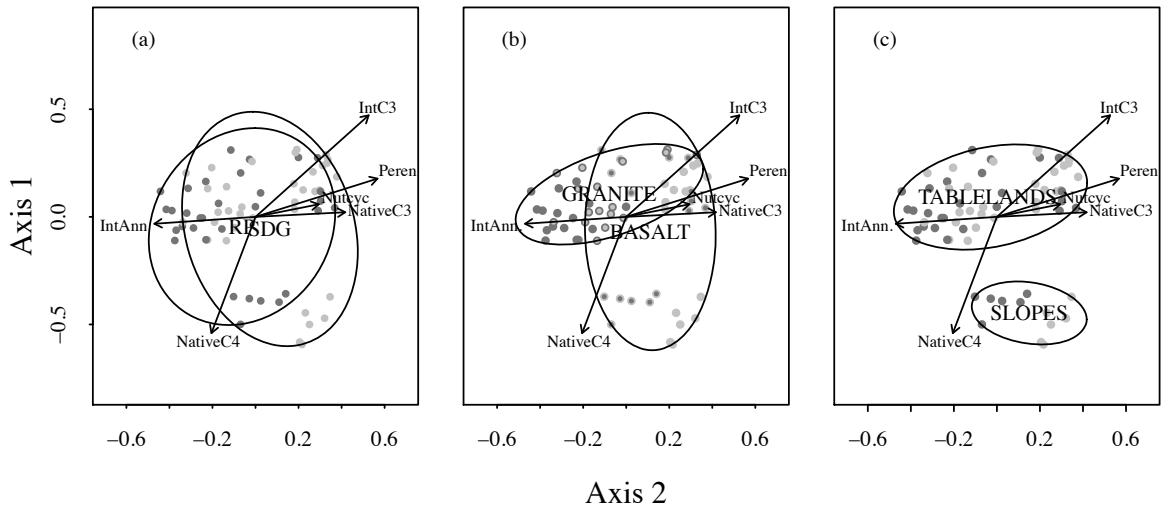

Figure 4-4 Multi-dimensional scaling analysis of pasture functional groups (perennials, introduced annual species, desirable introduced C3 pasture species, native C3 pasture species, native C4 pasture species and increaser species). Short-duration grazing sites are light grey symbols and regional practice sites are Dark grey symbols. Ellipses surround sites within each management category (a), that are basalt-derived (Bas) or granite-derived (Gran) (b) and (c) at higher tableland sites (Tablelands) compared to lower elevation sites on the western slopes (Slopes) of the study region. Arrows show the direction of change for significant variables with the relative length of each vector representing the relative magnitude of change for each plant functional group.

4.4.3 Ground-layer biophysical attributes

Mean (± 1s.e.m.) plant and litter cover was similar at SDG (98.5 ± 0.3%) and RP sites (95.5 ± 1.2%; $Z = 1.92$, $p = 0.055$; Figure. 4.5a). The site with the least overall cover (69%) was an RP site, while the least amount of overall cover at any SDG site was 93%. Across the 2 years surveyed, more SDG sites had ≥ 99% cover than RP sites (24 compared to nine sites, respectively). Three RP sites had < 80% cover compared to zero SDG sites (Figure 4.5a).

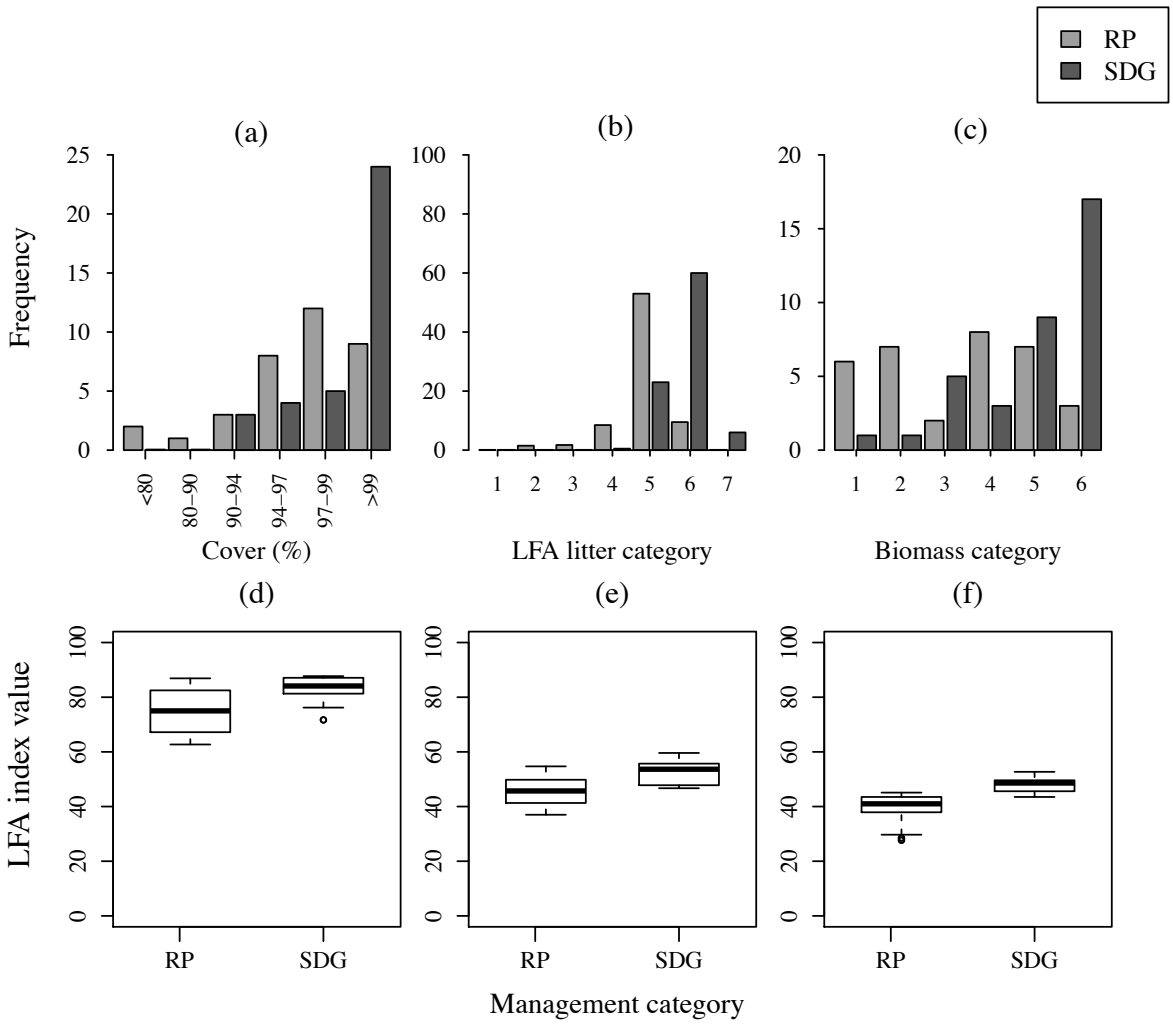

Figure 4-5 Frequency histograms of the two grazing management categories in relation to: (a) overall plant foliar and litter cover; (b) litter cover and depth from Figure. 4.5. Frequency histograms of the two grazing management categories in relation to: (a) overall plant foliar and litter cover; (b) litter cover and depth from LFA assessments (class1: < 10%, class 2: 10–25%, class 3: 25–50%, class 4: 50–75%, class 5: 75–100%, class 6: 100% and up to 20 mm thick, class 7: 100% and 20–70 mm thick), and (c) biomass categories (class 1: < 300 kg ha^{-1}, 2: 300–800 kg ha^{-1}, 3: 800–1300 kg ha^{-1}, 4: 1300–1800 kg ha^{-1}, 5: 1800–2300 kg ha^{-1}, 6: > 2300 kg ha^{-1}); (d, e, f) are the LFA indices for soil/sediment stability, water infiltration and nutrient cycling respectively. The dark line on the box plots represents the median value for each management approach, the extent of the boxes the inter-quartile range and extreme lines show the highest and lowest values (excluding outliers). Y-axis shows 0–100 reflecting that each index is scored out of 100.

Table 4.2 Results from GLMMADMB and Linear Mixed Effects analyses for the three landscape function indices, litter cover and depth, soil surface roughness (SSR) and the amount of retained herbage mass in pastures. AICs for full and final models are shown with *Z* - values for the GLMMADMB models and *T* - values for the Linear Mixed Effects models (ns denotes not significant in the model). Associated probabilities are shown as superscripts. Management refers to the change from RP to SDG, geological substrate is what soils are derived from at each site with granite is the reference substrate.

Model	*Model type*	*AIC*	*Management*	*Geology*
Stability	GLMMADMB	Full: – 272.5 Final: – 259.5	$Z = 2.91$, $p = 0.004$	*ns*
Infiltration	GLMMADMB	Full: – 297.1 Final: – 299.3	$Z = 2.85$, $p = 0.004$	*ns*
Nutrient cycling	GLMMADMB	Full: – 276.0 Final: – 279.6	$Z = 3.90$, $p < 0.001$	*ns*
Litter cover and depth	NLME	Null: 10159.3 Final: 8349.3	$T = 48.1$, $p < 0.001$	$T = 6.76$, $p < 0.001$
SSR	NLME	Null: 5269.7 Final: 4158.6	$T = 35.5$, $p < 0.001$ $T = -4.4$, $p = 0.007$	$T = -6.9$, $p < 0.001$
Rest-to-graze ratio	NLME	Null: 110.47 Final: 99.57		*na*

The amount of litter cover and depth determined by the LFA assessment was significantly greater on SDG than RP sites (Figure. 3.5b; Table 4.2). On SDG sites, 74% of points assessed had litter in class 6 or 7 (100% cover and up to 20mm depth; 100% cover and between 20mm and 70mm thick respectively) compared to only 13% of RP points assessed (Figure. 4.5b). Litter cover and depth also differed according to geological substrate, with granite-derived soils having significantly lower litter cover and litter depth than basalt-derived soils (Table 4.2).

The measure of SSR relating to micro-topographical relief and determined by LFA assessment was significantly greater at SDG sites than RP sites (Table 4.2).

Greater herbage mass was retained on SDG sites than on RP sites (Figure. 4.5c; Table 4.2).) with only three RP sites being in the higher category of pasture biomass (>2300 kg ha^{-1}) compared to 17 of the SDG sites. Conversely, over twice the number of RP sites (15) had retained less than 800 kg ha^{-1} in pastures compared to seven SDG sites (Figure. 4.5c).

Compared with RP sites, the greater perennial plant cover, litter cover and depth and SSR on SDG sites resulted in significantly higher values for the three LFA indices of soil stability (Figure 4.5d, Table 4.2), infiltration (Figure 4.5e, Table 4.2) and nutrient cycling (Figure 4.5f, Table 4.2).

Changes in soil nutrient status for major nutrients, P, N and C and Ca in the top 5-cm of the soil profile differed between basalt and granite-derived soils (Basalt-derived - $mg.kg^{-1}$: P = 70.4 ±8.32, N = 5.83 ±0.45, C = 0.48 ±0.04, Ca = 15.67 ±2.24; granite-derived - $mg.kg^{-1}$: P = 29 ±2.8, N = 3.92 ±0.45, C = 0.32 ±0.02, Ca = 3.57 ±0.26) with reduced levels of Ca were associated with greater cover of increaser species. However, no differences for these nutrients were found between management approaches.

4.5 Discussion

This study found that short-duration grazing management of naturalised pastures on commercial properties was associated with favourable outcomes for functional and compositional attributes of pastures compared to more usual management typical of the region. The improvements observed included greater persistence of perennial herbaceous species including native C3 grasses that are valuable as winter forage and corresponding reductions in the cover of introduced annual species and species that typically increase under heavy grazing pressure. Pastures under SDG management were also characterised by increased cover and depth of plant litter and greater retention of pasture biomass indicating improved landscape function. These findings supported our initial hypotheses. Pasture composition was also strongly influenced by the environmental variables of underlying geological substrate and local rainfall. However, there was a large amount of unexplained variation in pasture composition suggesting there were many factors unaccounted for in our analyses.

A key difference between the two management approaches was the ratio of the time pastures were rested for compared to the time pastures were grazed (rest to graze ratio). Despite variability within categories, on SDG sites this ratio was significantly larger, ranging from 13:1–60:1 (median = 27:1), than on RP sites where it ranged from 0 to 3:1 (median = 0.5:1). For SDG sites, the length of rest depended on local environmental conditions and was based on the recovery of key perennial forage species, while on RP sites rest was less planned and generally in response to limited forage availability. Extended rest allows taller perennial forage plants the opportunity to gain stature (Teague et al., 2013). Over long time-frames, this may impart a competitive advantage over relatively prostrate species. Our findings are consistent with Teague et al. (2011), who similarly investigated responses where rest-to-graze ratios were of a similar magnitude.

The regular pasture monitoring by SDG managers is also likely to facilitate proactive adjustment of stocking rates preceding and during dry times. This monitoring enables reductions in grazing pressure before feed shortages eventuate thereby minimising excessive grazing of more palatable, less grazing-tolerant plants that are typically of taller stature. The smaller grazing units necessary to achieve extended rest periods on SDG properties likely support this decision-making process with managers able to more easily visualise and effectively monitor plant recovery where pastures are not being repeatedly grazed (Steffens et al., 2013; Teague et al., 2013). Thus, the differences seen here may be due to avoidance of heavy grazing of more palatable and taller perennial species at key times during a season or in extended dry times, and over a timeframe of several years. This is likely long enough to see changes in pasture composition independent of cycles of extended dry periods and times when forage is limited.

Landscape function analysis was an important component of this study. While a large body of work has considered pasture compositional responses to SDG styles of management, including groundcover and herbage mass changes, relatively little research has considered how attributes that relate to soil processes vary between rotational and continuous grazing (but see Teague et al., 2011). This may be because these characteristics do not directly relate to animal production, but rather to more general ecosystem processes and consequently receive less attention (Whalley, 2000). Although quantifying the benefits arising from these improvements was outside the scope of this study, the significant improvements seen here under SDG management are highly relevant to maintaining productivity, ecological function and resilience of grazing landscapes (Teague et al., 2004; Müller et al., 2007; Teague et al., 2011; Teague et al., 2015). When considered in the context of: (1) large areas of the landscape; (2) with climate change scenarios resulting in increased temperatures and rainfall events that are likely to be of greater magnitude (Hughes, 2003; Hennesey et al., 2008) and (3) across several years, these improvements are likely to be highly relevant to the ecological and economic sustainability of commercial grazing enterprises.

We suggest that the accumulation of litter to the extent seen on SDG sites in this study may not be achievable under continuous grazing practices. The extended rest that is common under SDG management, followed by grazing with relatively high animal densities during grazing periods is likely to allow tall grasses and mature plant tissue to be trampled by animals and turned into litter. This is particularly true of tall native species that, without trampling by dense mobs of livestock, can remain standing for a long time (Whalley, 2017). While there appears to be limited work investigating this hypothesis, a study by Beukes and Cowling (2000) did show clear differences in the amount of litter present in areas stocked at much higher densities (4x) and rested for 12 months between grazing intervals. Chamane et al. (2017) also found much greater depth of litter under high density, short-duration rotational

grazing system compared to a lower density rotational system. The physical breakdown by livestock is likely to facilitate contact of dry matter with soil, thereby providing a food source for soil organisms (Whalley, 2000). This increased litter load has important consequences for a range of important ecosystem processes including nutrient cycling (McNaughton et al., 1988), water infiltration and aeration of the soil (Tongway and Hindley, 2005) and creation of seedling microsites (Milton, 1992). Litter is also vital for conserving resources such as water, soil and nutrients and to moderate the environment at the soil surface (Beukes and Cowling, 2000). While there may be a trade-off of reduced abundances of legumes under high litter loads (Leigh et al., 1995), the combination of improved landscape function from deep litter layers (including reduced loss of soluble nutrients such as P and N in run-off (McCaskill and Cayley, 2000), increased soil surface micro-topography and strong tussocky perennial grasses, in addition to favourable compositional changes may be an important outcome of SDG management that is rarely considered in studies of sustainable grazing systems. Management practices that allow taller perennial species to gain stature, combined with intermittent trampling by relatively dense mobs of livestock to turn plant material into litter that benefits soil processes, may be necessary to achieve these dual outcomes of improved pasture composition and landscape function.

The relationship of LFA to biophysical outcomes has been verified in a range of environments (Yates and Hobbs, 2000; Ata Rezaei et al., 2006; Maestre and Puche, 2009; Zucca et al., 2013) and LFA has been used as a research tool in temperate grazed landscapes (McIntyre and Tongway, 2005; Read et al., 2016). Although there is uncertainty around the ability of the nutrient cycling index to predict some aspects of nutrient availability in grazed areas (Eldridge and Delgado-Baquerizo, 2018), plant litter is important for the cycling of P and N in pastures (Leech, 2009). Therefore, while litter plays an important role in minimising the loss of nutrients in run-off and facilitates their availability, further work would be

valuable to quantify the benefits of LFA indices, particularly the nutrient cycling index, in heavily grazed areas. With increased certainty around the quantification of these indices, LFA would be a valuable tool to improve the ability of graziers to understand and monitor the biophysical condition of pastures.

Although only broadly categorised, the greater retention of pasture biomass at SDG sites was a further difference between management approaches. This likely reflects more timely adjustments of stocking rates in response to changing conditions that is a characteristic of SDG management. Research literature and industry extension material regarding the sustainable management of naturalised pastures in temperate Australia recommends retention of around 1500 kg ha^{-1} or more of dry matter in pastures (Kemp et al., 2000; Kahn and Earl, 2007; Badgery et al., 2008; Dorrough et al., 2008). In this study, pasture biomass levels were often below that amount on RP properties, suggesting that feed budgeting and timely adjustments to stocking rates happen less under more usual management practices on naturalised pastures.

While changes in abundance for several functional groups in our study were significant, there was also large amount of unexplained variation that did not relate to management approach. Likely contributing to this were inherent site differences such as legacy impacts of previous cultivation practices and fertiliser history that are known to influence pasture composition (Scott and Whalley, 1982; Reseigh, 2004). These factors could not be included in analyses due to low sample sizes and difficulty obtaining this information. However, pasture compositional changes were only one aspect of the study, with improvements to landscape function also being an important component of sustainable pastures that we assessed (Whalley, 2000; Dorrough et al., 2008). It is likely that these landscape function changes are a major aspect of the improvements claimed by advocates for SDG styles of

management. We suggest that these advocates may not be distinguishing between compositional and landscape function attributes.

A unique aspect of this study is that it examined outcomes for pasture composition and landscape functioning over timeframes upwards of ten years on commercial grazing properties (i.e., at the ranch scale). In commercial situations, decisions are made by grazing managers rather than researchers and are in response to difficult to predict changes in markets alongside a variable and unpredictable climate (Teague et al., 2013). This type of study is unusual, with only four studies we are aware of having investigated outcomes on commercial properties where livestock are controlled by grazing managers (Earl and Jones, 1996; Dowling et al., 2005; Jacobo et al., 2006; Teague et al., 2011). Earl and Jones (1996), Jacobo et al. (2006) and Teague et al. (2011) all concluded that there were generally favourable outcomes with an SDG approach to management. However, Dowling et al. (2005) found no consistent response to time-controlled grazing (a variation of SDG management) across several sites. Dowling et al. (2005) used multiple statistical approaches, finding evidence of favourable composition at some time-controlled grazing sites with ANOVA techniques but few differences with alternative techniques. They considered their results to be complicated by variable pasture compositions at the start of the investigation, ultimately concluding there were no advantages for time-controlled grazing. While the trends found by Dowling et al. (2005) are similar to trends we found in our investigation, there are some key differences between our study and that of Dowling et al. (2005) that may account for the differing conclusions. These are: (1) the concentration of our study within a more limited geographical region which is likely to have resulted in less variation in composition among paired sites, (2) the use of linear mixed model statistical techniques enabled us to account for variation between paired property locations and (3) the extended time that SDG management had been in place for in our study would have encompassed multiple periods of climatic stress (i.e. dry

times) with management therefore impacting pasture composition over long timeframes. While Dowling et al. (2005) investigated pasture attributes across six years, they do not state how long time-controlled grazing management had been in place for prior to their study (although two of the sites they used were also used by Earl and Jones, 1999). These extended timeframes are likely to be of critical importance as significant pasture changes are a response to long-term management factors (Teague et al., 2013) such as adaptively responding to changing market and variable climatic conditions. Consistent with our findings, the study by Jacobo et al. (2006) also considered litter cover, finding increases in rotational grazing systems. Similarly, Teague et al. (2011) considered a range of biophysical attributes on properties where an SDG style of management had been in place for more than 9 years and found positive responses for several attributes relating to landscape functioning. Neither Earl and Jones (1996) nor Dowling et al. (2005) presented findings on litter. The combination presented here of pasture compositional changes (that vary substantially across geographic areas and over time), combined with an examination of landscape function attributes is in line with the findings of Teague et al. (2011) particularly and demonstrates important changes occurring under SDG management that are likely to benefit the resilience of grazing businesses.

It is likely that SDG management enables managers to avoid excessive grazing of desirable plants and areas of the landscape to a greater extent than with more usual practices. In conjunction with improved landscape function, the outcome is a ground-layer with greater surface micro-topography that in-turn positively impacts landscape function and pasture resilience. There are key features of SDG management that support this outcome. These are: (1) strategically planned fencing and water infrastructure that enables control of grazing pressure; (2) smaller grazing units leading to more homogenous use of pastures by grazing animals; (3) the intentional monitoring of pasture growth and recovery with the aim of

matching stocking rates to pasture availability and (4) the increase in vegetation height that occurs with extended rests, followed by dense mobs of livestock trampling mature plant material, allows the accumulation of litter at the soil surface. While the first three of these factors are arguably achievable under continuous grazing practices with smaller paddocks and improved water infrastructure, we suggest it is unlikely that ground-layer litter could accumulate to the same extent under more continuous grazing practices. This hypothesis would benefit from targeted research.

Obtaining explicit financial data was outside the scope of this study. However, as SDG management necessitates intensive fencing and water infrastructure (i.e. an additional cost for grazing managers), this needs to be factored into consideration around adoption of these practices. Case-studies from practitioners of SDG claim that costs for this infrastructure are recovered within 2 years (Wright, 2017). A bio-economic analysis of an approach similar to SDG which incorporated tactical rest to promote the perennial grass component of pastures considered economic and environmental outcomes over a 20-year timeframe (Jones et al., 2006). This analysis concluded that tactical rest grazing had economic and environmental sustainability benefits compared with grazing that did not incorporate rest (Jones et al., 2006). While environmental outcomes such as salinity, biodiversity, erosion etc. can be difficult to place an exact value on, the planned fencing and water infrastructure that helps achieve these multiple outcomes is essential for managing pastures during extended dry periods. Government support for this infrastructure may be more appropriate than reactive drought relief policies (Sherren et al., 2012). Although decisions to alter practices will depend on the overall context of the grazing businesses, there appears to be reasonable evidence that the additional set-up costs to implement SDG should not be a deterrent to this approach to managing livestock.

Importantly, although ordination techniques should be viewed as graphical representations rather than robust statistical analyses (Oksanen, 2019), the ordination did demonstrate that there was substantial variation in pasture composition that was not explained by management approach, or any of the other factors we examined. The outcomes of this study should be considered within that context.

In addition to management approach, underlying geological substrate and rainfall patterns were important drivers of pasture composition with interactions also found between grazing management and these environmental factors. In particular, higher mean annual rainfall and altitude were associated with a greater abundance of high-value introduced pasture species. Historical land management factors would also be influencing these patterns. Due to higher rainfall and higher fertility of basalt-derived soils at increased elevations, pastures have been sown and fertilised more frequently with introduced species than sites at lower altitudes and with lower soil fertility (Keys, 1996). However, the increased abundance of introduced C3 perennial species under SDG management at these higher elevations suggests that planned and extended rest is supporting the retention of higher-value forage species in pastures. Rapid response to rainfall was also evident in native grasses in our surveys, thus highlighting the value of these species as an important component of resilient pastures.

There are limitations to this study that need to be acknowledged. These include that plant surveys were only taken during late summer and that soil samples were only taken to 5-cm depth. Due to the increased variability that is common in these surface layers (Pulido et al., 2016), we may have overlooked associations of soil nutrients with pasture attributes. Due to resource limitations, detailed and specific data on animal production could not be obtained. However, coarse data on stocking rates was collated, and while in two cases this data was

unreliable, stocking rates under both management approaches were very similar. Therefore, the positive responses seen on SDG sties in the study cannot be attributed to lower stocking rates.

Finally, it is possible that the SDG managers in this study were particularly astute and motivated observers who were more concerned with the composition and function of their pasture-base than the paired RP property managers. To minimise the possible influence of these differences in management skill and motivation over the results we sought to use only RP properties that were judged to be well-managed by local standards. However, we acknowledge that this is a flaw in the study design that was difficult to avoid.

Conclusion

The results of this study suggest that in a commercial context, SDG is a management strategy that is able to successfully address issues that are of ongoing concern in temperate pastures in the region. While it is uncertain exactly what aspects of management are responsible for the improvements in ground-layer composition and functioning observed, key characteristics of sustainable grazing management such as retaining adequate pasture biomass and proactively adjusting stocking rates to prevent excessive grazing during times of feed shortage are likely to be facilitated by the regular movements of stock and targeted pasture monitoring that occur under SDG. While these practices are possible under continuous grazing, it may be more difficult to implement appropriate rest for targeted species where stock are still present in paddocks. We also suggest it is unlikely that the depth of non-standing plant litter can develop to the same extent under continuous stocking practices as seen here under SDG, and that it would be valuable to explore this further. While there were challenges associated with working on commercial properties, including lack of availability of detailed information for some factors, this study has provided an insight into some

favourable outcomes that are occurring under SDG management where it has been in place for several years. In some ways our results are at odds with the conclusions of many studies and reviews in this area of research. We consider these differences to be due to having investigated the effects of long-term changes in management as well as the additional consideration of landscape function that is an integral component of managing grazed pastures for resilience.

Chapter 5 Current grazing practices interacting with environmental factors and historic management drives grassland composition and diversity

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5.1 Abstract

Livestock grazing is a widespread use of temperate grasslands and can reduce grassland plant diversity. Identifying commercial livestock grazing practices that are compatible with high levels of grassland diversity is desirable. We determined whether grazing livestock for short periods with high animal density, followed by long rests to allow pasture recovery (short-duration grazing), would lead to altered composition and improved richness and diversity of herbaceous species in grasslands compared to more usual commercial grazing practices. Over 2 years, we assessed the richness, Shannon-Wiener diversity (H') and composition of native and introduced herbaceous species in naturalised pastures at 36 sites on 12 commercial grazing properties. Six of the properties had been managed with short-duration grazing for more than 7 years, in most cases over 10 years, while the six other (control) properties were managed with grazing more typical of the region (i.e. relatively continuous grazing throughout the year with unplanned rests). Differences in management approach influenced composition of pastures more than richness and diversity, with SDG management resulting in favourable composition of both native and introduced components of pastures. Across both management types, native and introduced herbaceous species contributed similar amounts of cover (approximately 45% for both) to pastures. The extent of introduced cover was the most important determinant of the richness, diversity and cover of native species, with these attributes declining as introduced cover increased. Total species richness (native plus introduced) was influenced by environmental and management factors. There was greater native forb diversity with management more typical of the region, reduced richness overall during the drier sampling periods and increased richness overall P levels in the soil surface layer were higher. Increased P in the soil surface layer also resulted in higher plant diversity. This study highlights the importance of environmental factors in driving

richness and diversity and that while alternative approaches to grazing management influence composition, they do not necessarily influence species richness or diversity of grasslands that have been modified by past agronomic practices.

Additional keywords: grassland composition, groundcover, litter, planned rest, bare ground, rest-rotation, strategic rest grazing, overgrazing, temperate grasslands

5.2 Introduction

Temperate grasslands occupy 8% of the earth's terrestrial surface (Watkinson and Ormerod, 2001) and occur on every continent, except Antarctica. They are among the most diverse but also the most endangered biomes in the world (Pärtel et al., 2005; Dostálek and Frantík, 2012) and due to their suitability for human settlement, have been extensively modified for agriculture, including livestock grazing (Henwood et al., 2010). Ongoing agricultural intensification and demands on the world's agricultural lands to produce food and fibre as the world's human population increases (Tilman et al., 2002; Foley et al., 2005) means that these grasslands are under growing pressure, and their conservation an ongoing challenge (Henwood et al., 2010).

In many regions temperate grasslands have evolved with large herbivores (McNaughton, 1979; Oesterheld et al., 1992; Milchunas et al., 1998). These grazers influence grassland diversity by reducing the abundance of dominant plant species (Belsky, 1992; Tremont, 1994; Díaz et al., 2007) and removing moribund plant biomass and litter (Frank et al., 1998; Dorrough et al., 2004). If grazing pressure is moderate in relation to the productive capacity of landscapes and herbivores can graze over large areas, selective grazing results in heterogeneous vegetation patterns that contribute to overall landscape diversity (Adler et al.,

2001; Fuhlendorf and Engle, 2001). Consequently, where managed appropriately, grazing has the potential to facilitate high levels of diversity in grasslands.

Heavy grazing pressures typically result in losses of floral and faunal biodiversity (Fleischner, 1994; Eldridge et al., 2016) as well as degradation of important landscape processes such as water infiltration, nutrient cycling and soil conservation (Yates and Hobbs, 2000; Steinfield et al., 2006). Negative impacts of grazing can be reduced through lowering grazing intensity or by complete exclusion of grazing animals (Noy-Meir et al., 1989; Dumont et al., 2009; Prober et al., 2011). However, these strategies, particularly grazing exclusion, come with substantially reduced levels of agricultural productivity.

In addition to grazing livestock, environmental factors influence biodiversity and composition of grasslands. Environmental factors can be intrinsic to a landscape, such as local climate and soil type, but are also mediated by historic (Pärtel et al., 2005; Gustavsson et al., 2007; McIntyre and Lavorel, 2007) and current management factors (Isselstein et al., 2005; Klimek et al., 2007; McIntyre and Lavorel, 2007). Management practices impacting biodiversity include the cultivation and sowing of pastures, herbicide application and top-dressing and nutrient enrichment with phosphate-based fertilisers and the addition of forage legumes (McIntyre and Lavorel, 2007). While these practices lead to higher short-term productivity (Dorrough et al., 2004; Isselstein et al., 2005), when combined with heavy grazing pressure, they result in the loss of grazing and nutrient-intolerant native species from once diverse natural grasslands that were typically lower in nutrient status than grasslands modified for production (Isselstein et al., 2005; Pärtel et al., 2005). Furthermore, although short-term production is lower in unfertilised, unmodified grasslands (Pärtel et al., 2005) the diversity in unmodified grasslands can lead to increased resilience making native and

naturalised pastures a valuable component of pasture production systems helping to balance productivity and environmental outcomes (Tilman et al., 2006; Wrage et al., 2011).

One strategy to reduce livestock impacts, and potentially restore diverse grasslands is the implementation of management that reduces nutrient inputs and incorporates short periods of grazing with extended rest periods. In this study this type of management is referred to as short-duration grazing (hereafter SDG). A community-of-practice of advocates and practitioners of these approaches exists, commonly with a goal of achieving environmental outcomes while maintaining profitability (Stinner et al., 1997; Alfaro-Arguello et al., 2010; Cross and Ampt, 2017; Gardner and Thackway, 2019). Previous authors have suggested that there may be potential to successfully integrate livestock production with biodiversity conservation using these approaches (Dorrough et al., 2004; Fischer et al., 2009). However, while there are claims that these practices can lead to increased biodiversity in comparison with more conventional practices (Stinner et al., 1997; Alfaro-Arguello et al., 2010; Ampt and Doornbos, 2011), there is little supporting empirical data (Nordborg, 2016); but see McDonald et al. (2019b).

In this paper, we explore the potential for pastures that have been modified for livestock production to be restored to high levels of diversity through reductions in nutrient input and altered grazing management. Specifically, we investigated: (1) whether there are increases in native, introduced and overall diversity of plants in the ground-layer of naturalised grasslands on commercial livestock grazing properties managed with SDG compared to more usual regional practice. We also wanted to determine the relative importance of grazing management and environmental factors to current grassland composition.

5.3 Methods

5.3.1 Location of study

This study was conducted on the Northern Tablelands and Slopes of New South Wales, Australia, between 29° 38′ S, 15° 46′ E and 30° 39′ S, 150° 27′ E (Figure. 5.1). Study sites ranged in altitude from almost 700m a.s.l. to just below 1400 m a.s.l. The region's climate is sub-humid with summer-dominant rainfall. Severe frosts occur in winter and a rainfall and altitudinal gradient exists across the region, with generally greater precipitation and lower evapotranspiration at higher elevations in the east (Lea et al., 1977; Lodge and Whalley, 1989). The geology of the region is complex and largely determines soil characteristics (Lea et al., 1977; Lodge and Whalley, 1989). Higher elevations are commonly Tertiary basalts underlain by late Permian granites intruded into Palaeozoic sediments. These geological processes have resulted in three major soil types in the region: fertile basaltic soils, infertile coarse to fine granitic soils and relatively infertile soils derived from metamorphosed sediments (known locally as 'trap' soils).

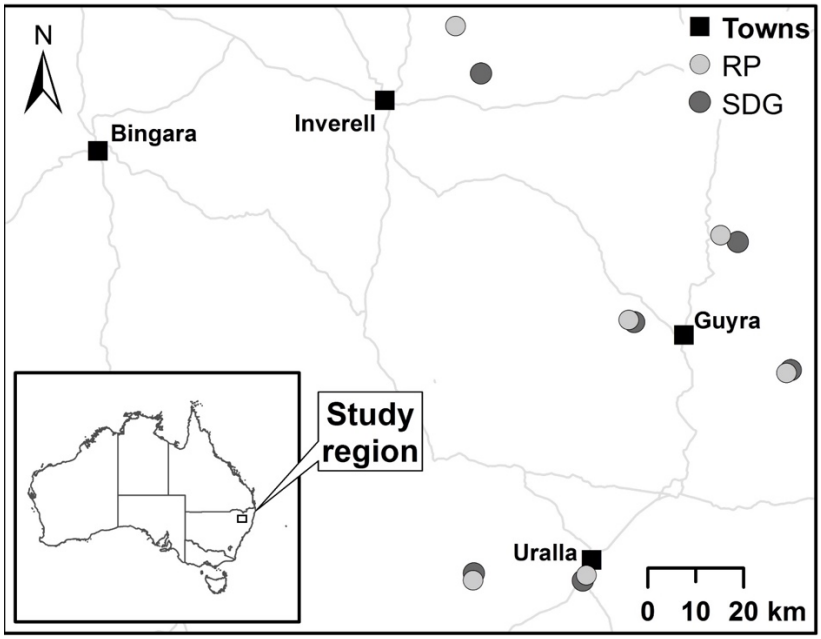

Figure 5-1 Location of the study region showing major towns (black squares) and the location of the 12 study properties. Properties that implemented grazing management more typical of regional practice (RP) are light grey circles; SDG properties are dark grey circles.

5.3.2 Site selection

On 12 commercial grazing properties we selected sites in naturalised pastures (terminology from Allen et al. (2011). Six of the properties were managed with SDG and six with what we defined as usual regional practice (hereafter RP). Five of the six paired locations were at higher elevations on the Northern Tablelands with one property pair located on the North West Slopes. Details about management practices were determined through initial contact with managers and greater detail was obtained later with a formal questionnaire. Typical practice for stock management on SDG properties was to graze stock intensively at high stocking densities for short periods (ranging from 1–2 days up to 1 week), with pastures subsequently rested until key forage species and pasture herbage mass had recovered. Decisions to return stock to pastures were based on pasture monitoring with the length of rest varying in accordance with seasonal growing conditions. This planned rest resulted in overall rest-to-graze ratios (amount of time rested compared to amount of time grazed) ranging between 13:1 and 60:1 (median = 27:1). Some SDG managers implemented very short graze periods (1–2 days) while others grazed for several days and rested for longer periods accordingly. Smaller paddock sizes, and correspondingly increased density of fencing and water infrastructure, are necessary to implement the large rest-to-graze ratios typical of the SDG approach and paddock sizes on SDG properties ranged in size from 2–16 ha (median = 10 ha). In this study, managers implementing SDG minimised their use of external inputs. This included minimal hand-feeding of stock during colder winter months and extended dry periods and limited application of inorganic phosphorus-based fertilisers (only one of the six SDG properties had applied superphosphate in the past 5–10 years as opposed to all six of the RP properties; Appendix 8). One of the SDG properties applied poultry manure. The reduced reliance on hand feeding of stock was achieved by pro-active adjustments of livestock numbers to levels matching the amount of feed available in pastures.

On the properties examined, SDG management had been in place for between 7 and 25 years (median time = 15 years).

The contrasting RP management approach was more typical of grazing management of naturalised pastures in the region. This was mainly continuous, although with some rotation of stock where forage was limiting, but with relatively long graze and short rest periods in response to forage availability (Scott et al., 2013; Shakhane et al., 2013). Rest-to-graze ratios on RP properties ranged from 0 to 3:1 (median = 0.5:1) and paddocks were large (median 20 ha, range 12–272 ha) in comparison to SDG paddocks. Superphosphate application on RP properties was typically on a calendar rotation (either annual or biennial) and stock were hand-fed to a greater extent than on SDG properties. The RP properties had been managed in the same way at least for the past 10 years.

Few pastoral managers had implemented SDG management in the region for greater than 10 years. Consequently, local networks were used to locate suitable SDG properties, with each property paired with a suitable neighbouring or nearby RP property. While this was not a random selection of participants, given the constraint of a low number of properties managed with SDG for adequate timeframes, it was the only practical way of selecting sites that respected neighbour relations and controlled for local soil and climatic conditions.

Livestock were predominantly cattle (*Bos taurus*), with four properties (three SDG and one RP) also running sheep (*Ovis aries*) for fine-wool production. The study design and low number of suitable SDG properties precluded the two livestock types from being treated separately in the study. Due to varying levels of engagement with producers, only coarse information on stocking rates could be obtained (Appendix 8). One of the criteria for the initial selection of properties for the study was that they were commercial enterprises. Therefore, we assumed that managers stocked according to their production goals balanced

with the need to sustain their resource base since grazing management is a trade-off between maximising yield and minimising the risk of degradation (Jakoby et al., 2015).

5.3.3 Collection of plant data

On each property, we selected three open pasture sites with similar soil type, aspect and slope to the corresponding paired property. Sites were at least 500 m apart ensuring that on SDG properties sites were representative of pastures at differing stages of the grazing rotation. All sites were located in pastures that had been altered historically through either top-dressing with phosphate-based fertilisers and pasture legumes and/or cultivated and sown with introduced forage species. However, sites varied in the degree of modification (i.e. number of times cultivated, top-dressed and fertilised and amount of fertiliser applied. At each site, a quadrat of 30 m^2 (5 × 6 m) representative of the surrounding pasture was surveyed for all vascular plant species in 2 successive years (2015, n = 36; 2016, n = 35). Surveys were conducted in late summer to early autumn when pasture species in the study region were easiest to identify (Reseigh, 2004; Shakhane et al., 2013). Species incidence was recorded, and percent foliar cover was estimated for all species in each quadrat. Plant and litter cover were visually estimated and summed to give overall cover.

Plants were identified to species level either in the field or by collecting specimens for subsequent determination. Nomenclature for all seed plants other than Poaceae followed Harden (2000–2002). For Poaceae, Ausgrass 2 (Simon and Alfonso, 2011) and Grasses of NSW (Jacobs et al., 2008) were used. Appendix 9 is a list of plant species and their incidence according to management approach and type of soil-substrate. We grouped plants according to whether they were native or introduced, and whether grass or forb. We determined overall species richness and Shannon–Wiener diversity (H') for all plants combined and for native grasses and forbs.

5.3.3.1 Rare and infrequent species

A small number of plants found in surveys were considered rare, infrequent or intolerant of high levels of grazing and P in temperate Australian grasslands (Whalley et al., 1978; McIntyre and Lavorel, 2001; Dorrough et al., 2011; OEH, 2019): the C4 grasses, *Themeda triandra* Forssk. and *Bothriochloa biloba* S.T. Blake., the perennial C3 grass *Microlaena stipoides* (Labill.) R.Br., and the forbs, *Tricoryne eliator* R. Br., *Einadia nutans* (R.Br.) A.J. Scott., *Murdannia graminea* (R. Br.) G. Bruckn., *Plantago varia* R. Br. and *Wahlenbergia sp*. Responses of these species to management are briefly discussed but the incidence and cover of these plants was too low for statistical analysis.

Sites were broadly categorised according to the amount of pasture herbage mass retained with categories ranging from very low (>300 kg ha^{-1}) to high (>2300 kg ha^{-1}). Estimations were made using a modified photo-quadrat technique after Morgan et al. (2018). For the estimations, six 50 x 50-cm sub-quadrats within nineteen 5 x 6-m quadrats of differing biomass amounts were cut and oven-dried at 40°C for 48 hrs. These calibration samples were then weighed and average dry matter for each 5 x 6-m calibration quadrat calculated. Photographs of each plant survey plot were then compared to photographs of the biomass calibration plots and plant survey plots then allocated to one of six categories of pasture biomass.

5.3.4 Long-term average rainfall and recent rainfall

Long-term average and recent rainfall data for each of the paired property locations was collated using a combination of BOM weather data (BOM, 2018) and property records where available. For the two months preceding sampling, rainfall was higher in 2015 than the long-

term median for those months at each location, while in 2016 it was either lower (property pairs 1, 4, and 6) or very similar (property pairs 2, 3 and 5; Figure. 5.2).

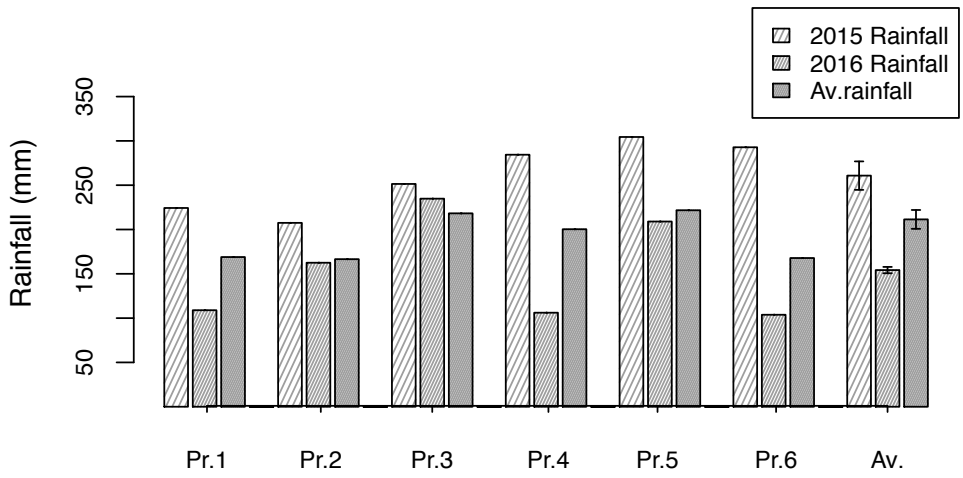

Figure 5-2 Rainfall (mm) for each property pair (pr) location for the 2 months prior to plant surveys in 2015 and 2016 and the long-term median for these same 2 months at each location. Average rainfall across all property pair locations for the 2 months prior to sampling in 2015 and 2016 as well as long-term median for that time period is also shown (Av.). Error bars for the average values are ±1 s.e.m.

5.3.5 Litter cover and depth

Litter cover and depth was determined from Landscape function analysis (LFA; detailed explanation and methods are provided in Chapter 4 and Appendix 10). Categories of low, medium and high litter cover were attributed. These categories were: high litter cover (100% cover over more than 80% of the landscape); medium litter cover (between 100% and 75% cover over 80% of the landscape) and low litter cover (less than 75% cover over 80% of the landscape).

5.3.6 Soil analyses

We assessed each site for soil nutrient status. For each LFA patch-type (see LFA methods in Chapter 4), three 5-cm diameter soil cores were collected per site to a depth of 5 cm. We combined these samples for analysis. Where there were two patch-types per transect, the results of the analysis were weighted according to the landscape organisation assessment (i.e., the proportion of each patch-type in the landscape). Soil samples were kept

cool and returned to the laboratory for oven-drying at 40ºC for 7 days until completely dry. They were then sealed and stored in a cool, dark place for later analysis.

Soil chemical analyses determined the nutrient levels in different patch-types at each site using methods from Rayment and Higginson (1992). We ground bulked samples to 2 mm following removal of large organic matter. We determined electrical conductivity (EC) and pH in a 1:5 soil to water extract, measured exchangeable cation concentrations (Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}) using atomic absorption spectroscopy and 1-M NH_4Cl, and determined available phosphorus (mg/kg) with Colwell P Malachite Green in 0.5-M $NaHCO_3$. Samples measuring P were passed through charcoal to remove organic matter and absorbance in a spectrophotometer measured at 630 nm. We ground subsamples of the bulked soil samples to pass through a 0.5-mm sieve and analysed a homogeneous subsample of this (0.2 g) for total C and N content by combusting at 950°C using a LECO TruSpec Simultaneous Carbon and Nitrogen Elemental Analyser.

5.3.7 Statistical analyses

Potential differences in species richness (S) and Shannon–Wiener diversity (H') of native forbs and grasses in response to management approach were modelled with linear mixed effects models (NLME) using R statistical software (Pinheiro et al., 2019). The two categories of management (SDG and RP) and the rest-to-graze ratio (continuous) were considered fixed effects in models. Other fixed effects were geological substrate (basalt or granite-derived), year of data collection (2015 and 2016), retained biomass (six categories) and the three categories of litter cover and depth derived from the LFA assessments. Continuous variables were: long-term average rainfall, altitude above sea-level, recent rainfall (number of rainfall events ≥10 mm) at each property pair location (at the individual level where data were available), absolute amount of rainfall for the previous 6 weeks prior to

sampling at each paired property location, and nutrient levels (N, C, P, K, Na, Mg, Ca) in the top 5 cm of soil. Where factors were determined to be correlated with each other, only one factor was included to avoid over-fitting of models. Property pairs (replicates) were blocked to incorporate the interdependence of properties within a pair and treated as a random effect. The NLME models were compared using Restricted Maximum Likelihood tests to determine the significance for each response variable. The simplest models with the lowest value of Akaike's Information Criterion (AIC) were retained for interpretation with difference in AIC value of >2 was considered better than the null model. Factors were considered significant if the test-statistic (*T*-value) was significant at the $p = 0.05$ level).

Differences in the rest-to-graze ratio were analysed using LME models, with property pairs treated as random. Rest-to-graze ratios and management approach were strongly correlated (Spearman rank order correlation, $r_s = 0.70$, $p < 0.001$) with SDG having significantly larger rest-to-graze ratios than RP ($T = 4.39$, $p = 0.007$). Models that included management approach had lower AICs than models that included rest-to-graze ratio. Therefore, management approach was used in final models in preference to rest-to-graze ratio.

Beta-distributed generalised linear mixed models (GLMMADMB) were used to assess the cover of native herbaceous plants. The beta-distribution is appropriate for proportional data as it naturally accommodates heteroskedasticity (Cribari-Neto and Zeileis, 2010). The same management and environmental variables as applied to the NLME models were used. These models were developed using the GLMMADMB package in R (Fournier et al., 2012), the simplest models with the lowest AIC values were retained for interpretation and factors considered significant if the test-statistic (*Z*-value) was significant at the $p = 0.05$ level)

5.3.7.1 NMDS, vector fitting and ordination

Dissimilarity matrices of the plant cover (abundance) data for the native and introduced pasture components were ordinated using non-metric multidimensional scaling (NMDS; metaMDS package in R). Solutions with three-dimensions had acceptable stress values for both plant groups (0.139 and 0.142 for native and introduced components, respectively) and were retained for interpretation. Continuous predictor variables were related to community composition using a vector fitting approach (Envfit in R) with the resulting R^2 value giving the relationship between the vector and the composition of the community in multi-dimensional space. Analyses were conducted using the Vegan package in R (Oksanen et al., 2018).

5.4 Results

Some 137 herbaceous plant species were recorded across all sites: 122 species were found on SDG sites (native: 68, introduced: 54) and 99 on RP sites (native: 54, introduced: 45). Overall, 76 species were native and 61 were introduced. On granite-derived soils there were 53 native and 44 introduced species, and on basalt-derived soils there were 59 native and 52 introduced species. Appendix 9 is a list of plant species with their presence recorded by each management category and geological substrate type.

5.4.1 Species richness and diversity

Year of assessment, rainfall in the previous 5 months and the amount of P in the top 5-cm of the soil profile influenced total species richness in pastures (Table 5.1). Plant richness was significantly lower in 2016, the drier of the two years and where there had been more rainfall in the previous 5 months and richness increased with increasing P levels in the top 5-

cm of soil (Table 5.1). Although there were more species overall with SDG management, no significant differences in overall richness were found between management approaches.

There was almost identical richness of all native and all introduced species at sites between the years sampled, with the richness of both native and introduced species being lower in 2016 (Figure. 5.3a and b). There was greater richness of native grasses than native forbs in both years (Figure. 5.3a), whereas richness of introduced forbs was greater than richness of introduced grasses in both years (Figure. 5.3b).

Shannon-Wiener diversity (H') across all sites was 1.64. On SDG and RP sites diversity overall was 1.61 and 1.67, respectively, in 2015 and in 2016 it was 1.69 and 1.59, respectively; neither difference was significant. Diversity on basalt-derived and granite-derived soils was 1.61 and 1.67, respectively. The differences between geological substrate type were influenced by average annual rainfall with diversity being lower on granite-derived soils at sites with higher annual rainfall (Table 5.1). Increased Colwell P in the top 5-cm of the soil profile was also associated with greater overall diversity (Table 5.1).

There was no influence of grazing management approach on richness or diversity of native herbaceous plants with the exception of native forbs, which had reduced richness with SDG management (Table 5.1). Native richness (overall) was lower in the second (drier) year of sampling (Figure. 5.3a; Table 5.1) with this reduction being attributable to both reduced native forb and grass richness (Figure 5.3a). Native richness decreased as introduced cover increased (Figure. 5.4a; Table 5.1) and this was due to changes in the richness of native grasses rather than native forbs (Figure. 5.3a; Table 5.1). Native richness increased as P levels increased (Table 5.1; Figure. 5.4b).

Diversity of native species was only influenced by the amount of introduced cover, increasing as levels of introduced cover declined (Table 5.1). Introduced cover did not significantly alter the diversity of native forbs or native grasses alone.

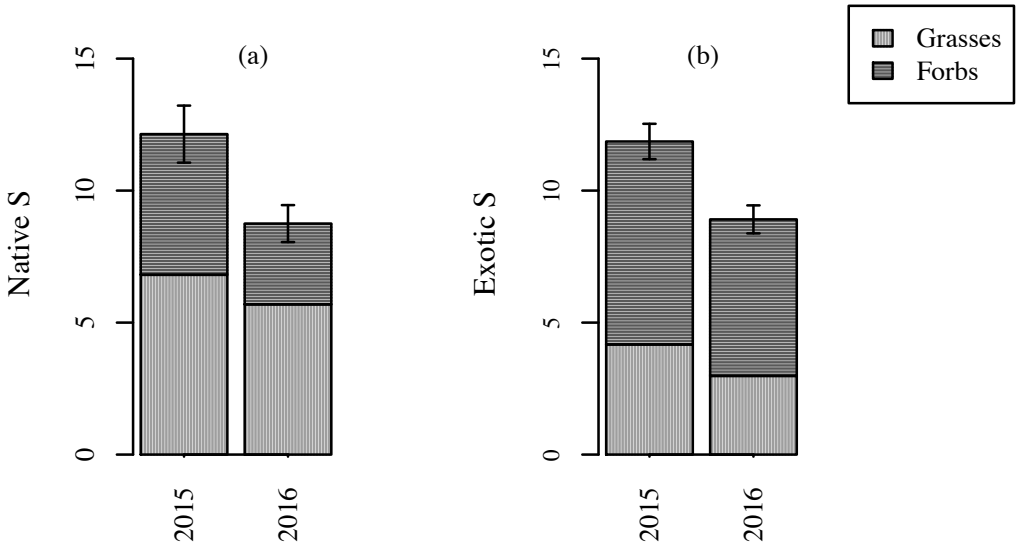

Figure 5-3 Species richness of (a) native and (b) introduced herbaceous plants in 2015–2016. Grasses and forbs are shown separately in each column. Error bars are ±1 s.e.m.

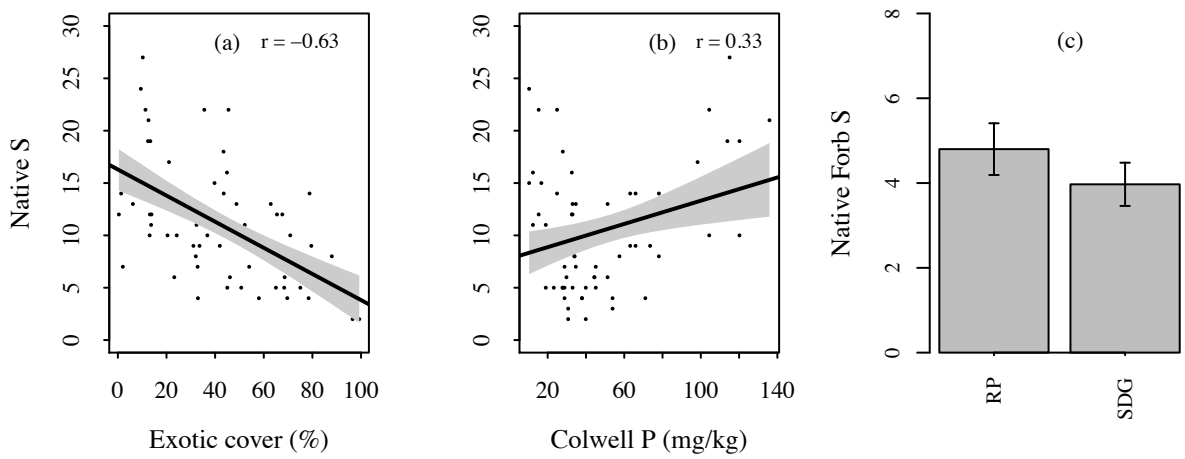

Figure 5-4 Modelled relationships of native herbaceous species richness in relation to (a) introduced herbaceous cover (%) and (b) Colwell P. (c) shows the richness of native forbs with contrasting management approaches; error bars are ± 1s.e.m. 'r' refers to Pearson's product moment correlation coefficient. The grey area in both plots refers to the 95% confidence interval for linear models between the two variables.

5.4.2 Cover of native herbaceous species

Greater native cover was present on RP sites, on basalt-derived soils and at sites with lower average rainfall (Table 5.2). Increased rainfall in the month preceding sampling also

resulted in increased native cover (Table 5.2). Although amount of overall native cover was significantly higher on RP sites (Figure. 5.5a; Table 5.2), the composition of this native component of pastures differed between the two management categories with RP sites characterised by higher proportions of species that tend to increase under heavy grazing pressure (Figure. 5.4a; Table 5.2).

The amount of cover of introduced species was influenced by geology and average annual rainfall, being lower on basalt-derived soils and at sites with lower rainfall. Management approach did not alter the amount of cover by introduced species. However, the composition of the introduced component did differ between management approaches, with RP sites having a significantly greater proportion of introduced annual species compared to introduced perennial species (Figure. 5.4b; Table 5.2).

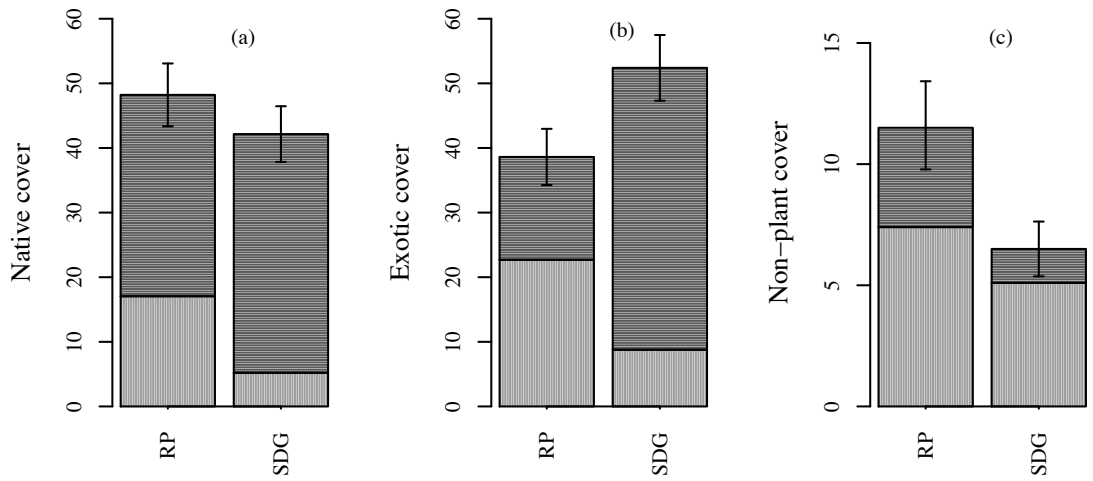

Figure 5-5 Groundcover occurring with the two contrasting management approaches. Native cover (a) of increaser species (*C. dactylon* and *B. macra*; light grey) and all other species (dark grey), (b) introduced cover of annual (light grey) and perennial species (dark grey), and (c) non-plant groundcover, litter cover (light grey) and bare ground (dark grey). Error bars are ±1 s.e.m.

Table 5.1 Values and associated probabilities from linear mixed effects (S and H' - Z-values) and GLMMADMD models (cover -T-values) for species richness (S) and diversity (H') (all herbaceous species, all native herbaceous species and native grasses and native forbs) for native and introduced plant cover. Geology is the difference between basalt and granite-derived soils with result the change from basalt to granite-derived soils. Year refers to the two years of assessment (2015 and 2016) and management approach reflects changes from RP to SDG. Av. rain refers to average annual rainfall and recent rain to the number of significant rainfall events (>10 mm) for the previous month [a] and 5 months [b]; litter categories are low and high with these litter categories reflecting the depth as well as the cover of litter as determined from the LFA assessments.

Response variable (AIC full, final)	***Management***	***Geology***	***Year***	***Av._Rain***	***P***	***Int. cover***	***Recent rain***	***Litter***	***Geol:av. rain***
Overall S (444.2, 446.0)	*ns*	*ns*	–5.380, p < 0.001	*ns*	3.108, *p* = 0.005	*ns*	[b] –2.044, *p* = 0.05	*ns*	*ns*
Overall H (86.97, 77.3)	*ns*	*interaction*	*ns*	*interaction*	2.891, *p* = 0.008	*ns*	*ns*	*ns*	–4.303, *p* = 0.05
Introduced S (380.6, 366.9)	*ns*	*ns*	–3.434, *p* = 0.002	*ns*	*ns*	*ns*	[b] –2.636, *p* = 0.013	*ns*	–4.303, *p* = 0.05
Native S (412.8, 412.6)	*ns*	*ns*	–4.256, *p* < 0.001	*ns*	3.007, *p* = 0.006	–3.645, *p* < 0.001	*ns*	*ns*	*ns*
Native H (118.7, 100.9)	*ns*	*ns*	*ns*	*ns*	*ns*	–3.569, *p* = 0.001	*ns*	*ns*	*ns*
Native Grass S (352.4, 338.8)	*ns*	*ns*	–2.545, *p* = 0.0158	*ns*	*ns*	–3.487, *p* = 0.001	*ns*	*ns*	*ns*
Native Grass H (117.9, 95.1)	*ns*	*ns*	2.322, *p* = 0.027	*ns*	*ns*	*ns*	[a] 2.280, *p* = 0.029	*ns*	*ns*
Native forb S (326.1, 328.6)	–2.956, *p* = 0.032	–5.535, *p* = 0.012	–5.297, *p* < 0.001	–6.073, *p* <0.001	*ns*	*ns*	*ns*	(low–high) 2.611, *p* = 0.016	*ns*

Table 5.2 T-values and associated probabilities GLMMADMD models for ground-layer cover for native, introduced, introduced annual species, the two species that increase under heavy grazing pressures (*Bothriochloa macra* and *Cynodon dactylon*) and cover of bare ground and litter (i.e., non-plant cover). Geology is the difference between basalt and granite-derived soils with result the change from basalt to granite-derived soils and management reflects changes from RP to SDG. Av. rain refers to average annual rainfall and recent rain to the number of significant rainfall events (>10 mm) for the previous month [a] and 5 months [b]; changes in response to the litter categories are from low and high, with these litter categories reflecting the depth as well as the cover of litter as determined from the LFA assessments.

Response variable *(AIC full, final)*	***Management***	***Geology***	***Altitude***	***Recent rain***	***Ca***	***Av. Rain***	***Litter***
Native cover (–39.4, –141.6)	–3.66, $p < 0.001$	–3.21, $p = 0.001$	ns	[a] 3.62, $p < 0.001$	ns	–7.83, $p = 0.001$	(low–high) 3.49, $p < 0.001$
Introduced cover (1.7, –4.8)	*ns*	3.14, $p = 0.002$	*ns*	*ns*	*ns*	5.75, $p < 0.001$	*ns*
Int. annuals (219.9, 209.9)	–2.54, $p = 0.011$	2.77, $p = 0.006$	*ns*	*ns*	*ns*	*ns*	*ns*
Increasers (513.4, 500.1)	–2.94, $p = 0.003$	*ns*	*ns*	*ns*	–1.95, $p = 0.051$	*ns*	*ns*
Cover of litter and bare ground (–205.8, –216)	–2.69, $p = 0.007$	*ns*	–3.98, $p < 0.001$	*ns*	*ns*	*ns*	*ns*

5.4.3 NMDS ordination – differential responses of native and introduced species

Non-metric multidimensional scaling of both the native and introduced components of the herbaceous plant community (Figure. 5.6 and 5.7) showed the strong effect of the correlated factors of altitude and average annual rainfall (r_s = 0.86, p = 0.029; Spearman rank order correlation) for both the native and introduced pasture components. The vector showing increasing perennial introduced cover was closely aligned with these two environmental variables (Figure. 5.6 a and b). Management approach was significantly associated with the composition of the introduced component of pastures (Figure. 5.7b). Changes in the cover of introduced annual species and bare ground were closely associated with changes in native composition and independent of the influences of altitude and rainfall and perennial introduced cover and (Figure. 5.6a and b). Underlying geological substrate and soil nutrient levels were also significantly related to native plant community composition (Figure 5.6b and 5.7b). The number of rainfall events >10 mm in the previous month was important for the native component of pastures (Figure. 5.6 and 5.7).

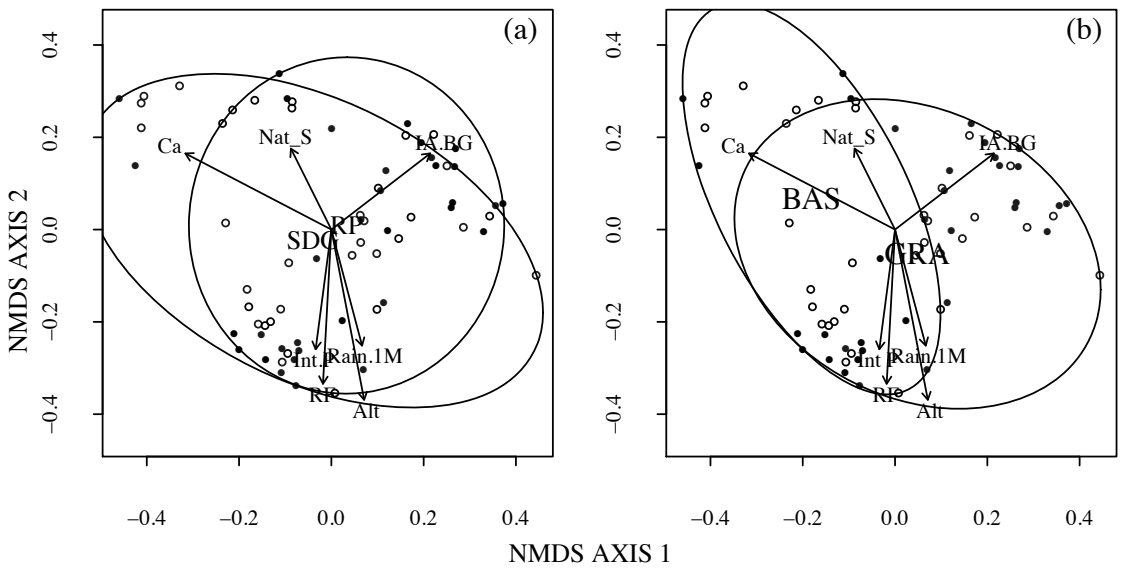

Figure 5-6 Non-metric multi-dimensional scaling ordinations (in 3-dimensional space) showing the first two axis for native plant communities based on standardised-abundance (cover) data. Arrows represent the direction of increasing gradients for vectors, with the length of arrows proportional to correlations between the variable and the ordination. Ellipses surround sites within contrasting categories of (a) management (R^2 = 0.04, p = 0.066) and (b) geological substrate (R^2 = 0.33, p < 0.001) with centroids shown by respective text (RP, SDG, GRA - granite and BAS - basalt. With the exception of native plant richness (Nat_S; R^2 = 0.18, p = 0.003) only significant vectors with an R^2 > 0.3 are shown. R^2 values for significant vectors are: Altitude (Alt): 0.67, p < 0.001, average rainfall (RF): 0.54, p < 0.001,Introduced perennial cover (Int.P): 0.33, p < 0.001, introduced annuals and bare ground (IA.BG): 0.35 p < 0.001, rainfall in the previous month (Rain.1M): 0.32, p < 0.001and Ca: R^2 = 0.61 p < 0.001. R^2 values of other soil nutrients significantly correlated with the ordination, and positively associated with Ca, were Mg: 0.60, p < 0.001; K: 0.59, p < 0.001, C: 0.28, p < 0.001, P: 0.21, p = 0.002; N: 0.20, p < 0.001.

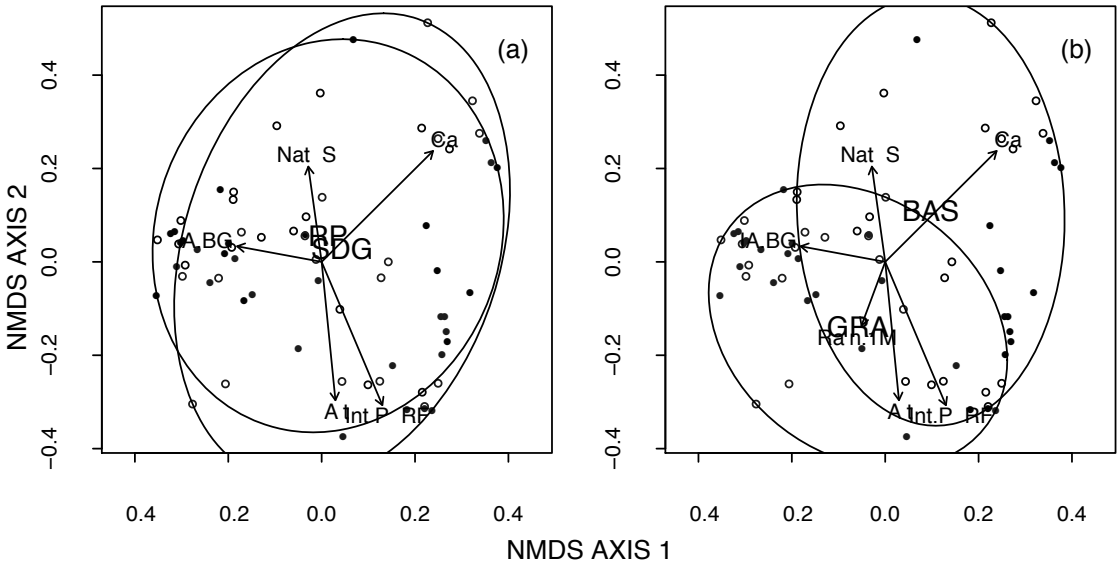

Figure 5-7 Non-metric multi-dimensional scaling ordinations (in 3-dimensional space) showing the first two axis for introduced communities based on standardised-abundance (cover) data. Arrows represent the direction of increasing gradients for vectors, with the length of arrows proportional to correlations between the variable and the ordination. Ellipses surround sites within contrasting categories of (a) management (R^2 = 0.12, p = 0.002) and (b) geological substrate (R^2 = 0.3, p <0.001) with centroids shown by respective text (RP, SDG, GRA - granite and BAS - basalt). With the exception of native plant diversity (Nat_S; R^2 = 0.27, p <0.001), only significant vectors with an R^2 > 0.3 are shown. R^2 values for significant vectors are: Altitude (Alt) - 0.62, p < 0.001, rainfall (RF): 0.66, p < 0.001 Introduced perennial cover (Int.P: 0.72, p <0.001, introduced annuals and bare ground (IA.BG): 0.22, p <0.001, Ca: R^2 = 0.5, p <0.001. R^2 values of other soil nutrients significantly correlated with the ordination, and positively associated with Ca, were Mg: 0.49, p <0.001; K: 0.47, p <0.001, C: 0.19, p <0.001, P: 0.2, p = 0.003; N: 0.15, p = 0.005.

5.4.4 Rare and infrequent species

Few species in these categories were identified during our surveys. Although statistical analysis could not be done due to the low number of species in this category, there appeared to be much greater cover and frequency of sensitive C3 and C4 grasses in both years of the survey at SDG than RP sites. There were similar frequencies of sensitive forb species with both management approaches but a higher cumulative cover of forbs in 2015 on RP sites (Table 5.3).

Table 5.3 Frequency and cover (cumulative percentage cover across all sites within a category) of plants that are considered infrequent in grazed pastures of the region and sensitive to the combination of grazing and phosphorus (Whalley et al 1978; McIntyre and Lavorel, 2001; Dorrough et al., 2011; OEH, 2019). Frequency (number of occurrences) and cumulative cover (percentage) across all sites within a category for each year are shown. Plants included are the C4 grasses *Themeda triandra* and *Bothiochloa biloba*, the C3 grass *Microlaena stipoides* and the forbs *Tricoryne eliator*, *Einadia nutans*, *Murdannia graminea*, *Plantago varia* and *Wahlenbergia sp*. The forb category includes the legume *Glycine clandestina*. Statistical analyses could not be performed due to the low occurrence of rare and infrequent species.

Category	***Attribute***	***C4 grasses***	***C4 grasses***	***C3 grasses***	***C3 grasses***	***Forbs***	***Forbs***
Year		*2015*	*2016*	*2015*	*2016*	*2015*	*2016*
RP	Frequency	0	0	9	7	14	10
	Cum. cover	0	0	132.2	89.1	8.9	0.3
SDG	Frequency	6	5	13	10	17	6
	Cum. cover	136.5	151.7	213.2	116.8	1.7	1.2

5.5 Discussion

Commercial livestock grazing in temperate Australia is a major driver of biodiversity loss and there is potential for alternative grazing management approaches to reduce impacts on ground-layer biodiversity (Dorrough et al., 2004; Mavromihalis et al., 2013; McDonald et al., 2019b). Despite many studies considering broad compositional changes under contrasting grazing management approaches (including SDG) from an agronomic and production perspective (Earl and Jones, 1996; Dowling et al., 2005; Ash et al., 2011; Teague et al., 2011), few have explicitly investigated how SDG and similar rotational grazing approaches influence richness and diversity of herbaceous species in pastures as well as biodiversity more generally (Nordborg, 2016), particularly on commercial grazing properties over longer timeframes. In this study, two significant differences were found between management approaches. Firstly, the composition of the introduced component of pastures varied significantly between management approaches and there was an increased richness of native forbs under RP management. There were relatively minor differences in other richness and diversity indices and in the composition of the native component of the community. The compositional changes rather than diversity differences, in addition to improved landscape functioning under SDG management (Lawrence et al., 2019) are likely to be responsible for many of the benefits claimed by practitioners and advocates of these altered approaches to grazing management.

Across both management types, native and introduced herbaceous species contributed similar amounts of cover (approximately 45% for both) in pastures. While all pastures were naturalised, they had all been modified historically through a combination of grazing, cultivation, sowing of introduced pastures and nutrient enrichment. These impacts result in the loss of native perennial species, particularly taller tussock species such as *Themeda*

triandra, *Poa sieberiana*, *Sorghum leiocladum* and *Dichanthium sericeum*. These species were abundant in grasslands prior to European colonisation and have been replaced by a range of perennial and annual introduced species (Moore, 1970; Norton, 1971; Whalley et al., 1978). In these naturalised pastures, heavy continuous grazing pressure leads to greater dominance by increaser species such as *Bothriochloa macra* and introduced annual/ruderal species as well as bare ground, particularly where pastures have been sown in the past (Cook et al., 1978; Whalley et al., 1978). While *B. macra* and *Cynodon dactylon*, another common increaser species, (Oudtsdoorn, 1992) are valuable groundcover during extended dry periods and droughts (Moodie, 1934; Whalley et al., 1978), and *B. macra* can be a valuable forage species if it is a part of the pasture mix (Archer and Robinson, 1988; Robinson and Archer, 1988). However, these two species are still considered invasive in pastures if they dominate (Cook et al., 1978).

The only richness response that grazing management influenced was native forbs which were lower on SDG sites. The reduced richness of all plants found in the second year of sampling was likely due to the drier weather conditions preceding sampling. Reduced richness was also found where there had been increased rainfall in the previous 5 months. This surprising result may be due to new growth following rainfall leading to ephemeral, and less dominant, forb species being selectively grazed (McIntyre et al., 1993). Additionally, the presence of ephemeral forbs may have been overlooked where there was vigorous growth of taller perennial species in response to recent rainfall. Lower richness of native forbs with SDG management may be a consequence of increased cover of tall perennial species that prosper under SDG (Teague et al., 2013; Lawrence et al., 2019) with native forbs competitively excluded by these taller grasses.

Despite a lack of significant difference in overall richness between management approaches, 122 compared to 99 plant species were recorded on SDG and RP sites, respectively. The lack of difference identified in analyses may be due to the considerable variation in richness at different sites with SDG management with a low species richness dominated by tall palatable perennial species (largely introduced species) seeming to be present at SDG sites that had been heavily fertilised in the past (basalt and granite-derived soils). In contrast, on less fertile soils that had been less modified historically, higher levels of native diversity were present (R. Lawrence *pers. obs*.).

The increased native richness, but not introduced richness, associated with increased P levels in the top 5-cm of the soil profile was counter to expectations. Low nutrient soils are generally associated with higher levels of plant diversity (Prober et al., 2002; Blomqvist et al., 2003; Isselstein et al., 2005) with enrichment by P leading to reduced native species richness (Beadle, 1954; Yates and Hobbs, 2000; Dorrough and Scroggie, 2008). Relatively few studies have looked at species richness in relation to fertiliser application and assumptions that fertiliser application directly results in declines in species richness may be misleading as it is difficult to disentangle confounding effects of past cultivation, heavy grazing and fertiliser application on species composition (Whalley and Lodge, 1987; Dorrough et al., 2012). It is likely that the increased P levels are associated in part with underlying geological substrate, with native forb diversity higher on basalt-derived soils that were also characterised by higher P levels (Basalt-derived soils - P = 70.4 ±8.32; granite-derived soils - P = 29 ±2.8) than granite-derived soils. It may also be that species persisting in these grasslands following decades of modification for agricultural production are relatively tolerant of both grazing and high available P.

We found that as the cover and depth of leaf litter increased there were associated increases in the richness of native forbs as well as greater cover of native species. Native grasses typically have a lower ratio of N:C resulting in slower decomposition rates (King and Hutchinson, 1983). Consequently, where native grass cover is high, and grassland structure permits the build-up of litter, the resulting deeper litter layer may enable greater retention of seeds leading to increased availability for the germination and survival of native forb species. This possibly contributes to increased cycling of P, thereby contributing to the unexpected increased native richness seen with greater P levels. While introduced perennial forage species contribute high-value (high N:C) forage for animal production, native species that are more resistant to breakdown (Moretto et al., 2001; Whalley, 2017) contribute to protection of the soil surface. In this way, native species provide a valuable contribution to the pasture mix through improvements to ecological function such as improved water infiltration, retention of soil, sediment and nutrients and providing litter for invertebrate detritivores (and associated species) that contribute to the cycling of nutrients in the soil surface-layer. These native grasses, particularly tall tussock species that were more common on SDG sites, support important ecological functions in grazed pastures, especially in times of climatic stress.

For grazing systems to balance productivity alongside long-term sustainability goals, high proportions of perennial species in pastures are necessary (Mason and Kay, 2000). Ideally these perennial species are a combination of high-value forage species as well as those that can persist during times of climate stress (palatable, perennial and persistent) (Kemp et al., 2000; Stafford et al., 2000). In this study, SDG pastures demonstrated these characteristics to a greater degree than the more usual regional practice pastures. Alternative grazing management practices may also have the potential to contribute to the conservation of native grasslands outside of formal reserve areas with areas of high conservation value such as travelling stock reserves having been exposed to intermittent grazing (Davidson et al.,

2005; Spooner and Morris, 2012). There are also examples of diverse native grasslands being sustained under commercial grazing conditions (Norton and Reid, 2013; Gardner and Thackway, 2019). However, our findings suggest that where pastures have been heavily modified previously by high grazing pressure cultivation and nutrient enrichment for increased livestock production, high levels of native biodiversity are unlikely to occur under commercial grazing pressures regardless of management approach.

Limitations of this study include the confounding of livestock types and a low number of sites assessed in the context of considerable variation in landscape characteristics and management history between sites.

Conclusion

Plant richness responses were complex and influenced by a combination of past environmental and historic management factors as well as current management approaches. Although the composition of SDG pastures was characterised by higher levels of perennial species, the native diversity, particularly of forbs, was lower compared to more usual regional practice suggesting that overall benefits differ between the two management approaches. Although there was minor variation in overall richness and diversity between management approaches, the composition of pastures clearly differed, with SDG pastures exhibiting compositional characteristics such as increased perennial cover and reduced levels of weedy species compared to more usual regional practice but the latter having increased richness of native forbs. Overall SDG pastures were not highly diverse native grasslands unless on less-fertile sites that had been relatively unmodified historically. ~~However, benefits for pasture composition were clear where there was incorporation of short graze periods and extended and monitored pasture rest into grazing management practices.~~

Chapter 6 Increased diversity and abundance of parasitoid hymenopterans under short-duration grazing suggests positive outcomes for insects with alternative approaches to grazing management

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6.1 Abstract

Insects are sensitive to land-use change and agricultural intensification is associated with large-scale declines in insect biodiversity. As livestock grazing impacts most of the world's grasslands, which are an important component of agricultural lands, managing livestock in ways that support a high diversity and abundance of insects can play a substantial role in offsetting the impacts of agricultural intensification on insect populations. In this study, we compared insect biodiversity in naturalised pastures on eight properties where livestock were managed with short-duration grazing (SDG) with eight properties where livestock were managed in ways more typical or conventional regional practices (RP). With SDG, grasslands were rested for extended times based on the recovery of plants after grazing and inputs were lower than with grazing management more typical of regional practice which was relatively continuous throughout the year. We found significant increases in species richness and abundance of parasitoid hymenopterans under SDG relative to RP management suggesting favourable outcomes for insect biodiversity with SDG approaches to managing livestock. Parasitoid wasps abundance and richness were positively associated with the ground-layer attributes of greater perennial plant cover and increases in the cover and depth of litter that characterise SDG. There was also a strong response of parasitoid wasp diversity and abundance to recent rainfall. These increases suggest there is potential for altered grazing practices to improve the capacity of landscapes to provide ecosystem services including natural pest control, pollination and food resources for wildlife. If SDG grazing management practices were implemented across large scales, they could potentially contribute to insuring against insect decline in landscapes modified for agricultural production.

Additional keywords:-nutrient cycling, invertebrates, nestedness, continuous grazing, mosaic landscapes, livestock production, indigenous grasslands, rotational grazing

6.2 Introduction

Insects are an essential link in the food-webs of all terrestrial ecosystems. They are one of the most diverse groups of terrestrial animals (Chapman, 2009) and act to facilitate nutrient and energy transfer from plants to higher trophic levels (Seastedt and Crossley, 1984). Insects also support a myriad of essential ecological functions, including pollination, dung burial, pest control and seed dispersal (Losey and Vaughan, 2006; Prather et al., 2013) and are an essential food source for wildlife (Losey and Vaughan, 2006). Insect populations are sensitive to changes that are occurring on local as well as global-scales (Conrad et al., 2006; Potts et al., 2010; Hallmann et al., 2017; Sánchez-Bayo and Wyckhuys, 2019) including agricultural intensification and high levels of pesticide use (Benton et al., 2002; Woodcock et al., 2016), landscape simplification (Benton et al., 2002; Tscharntke et al., 2005) and climate change (Fox, 2013). Understanding how changes in land use intensity impacts insects is critical to their conservation in landscapes modified for agricultural production.

Grasslands cover approximately 40% of the world's surface area (Suttie et al., 2005) and support a great abundance and diversity of insects (Tscharntke and Greiler, 1995). As livestock grazing impacts most of the world's grasslands, managing livestock in ways that support high biodiversity of insects can potentially play an important role in offsetting impacts of agricultural intensification on insect populations, while allowing continued food and fibre production. Impacts of grazing livestock on insects can be direct or indirect (Kruess and Tscharntke, 2002; Pöyry et al., 2006; Van Klink et al., 2015). Large herbivores directly influence insect communities by decreasing available food and habitat, unintentional predation and trampling during grazing, and indirectly through changes in plant diversity, community structure and abiotic conditions (Van Klink et al., 2015). Reducing the intensity

of grazing is one way of altering the impacts on vegetation and insects, with reductions in grazing intensity leading to increased insect species richness across multiple groups (Kruess and Tscharntke, 2002; WallisDeVries et al., 2016). Strong drivers of insect responses are the increased vegetation height and greater structural complexity that occur with lower grazing impacts (Southwood et al., 1979; Woodcock et al., 2007). Providing extended periods in a grazing cycle where grasslands are free from grazing animals is an alternative, or additional, way of reducing the impacts of grazing animals (Davis et al., 2014; Kruse et al., 2016), particularly where livestock are removed at critical times during the lifecycle of a target insect species (Scohier et al., 2013).

Livestock grazing approaches such as short-duration grazing (SDG hereafter) aim to rest pastures for extended periods after short, intense bursts of grazing with high number of animals. In these systems, livestock only return to areas when vegetation has recovered to a target herbage mass, or a plant species of interest has recovered, and to facilitate appropriate length of rest from grazing it is commonplace to adjust livestock numbers during times of climate stress. Such practices retain vegetation of greater height and structural complexity in comparison to grazing most or all of the time which typically leads to simplification of grassland, especially at high grazing intensity in relation to the productive capacity of the landscape (Curry, 1994; Lawrence et al., 2019; McDonald et al., 2019a). Approaches that implement grazing for short periods followed by extended rest often also place increased emphasis on the value of biodiversity in grazing systems (Stinner et al., 1997; Norton and Reid, 2013; Cross and Ampt, 2017). There is also evidence that these types of grazing approaches are more likely to facilitate natural tree regeneration, providing greater structural complexity and resources in landscapes, than more continuous grazing practices (Fischer et al., 2009). For these reasons, it is likely that SDG could lead to increased abundance and diversity of insects compared with more continuous grazing practices, particularly if rest

periods coincide with critical stages of insect lifecycles. Few studies have specifically investigated whether differential responses of insect populations occur where livestock are managed with approaches similar to SDG (i.e. strategic and planned rest; but see Saunders and Fausch (2007; 2012); Dorrough et al. (2012).

As impacts of agricultural practices intensify, insects with particular traits can decline more rapidly than others resulting in the non-random loss of insect species from communities (Loo et al., 2002; Rader et al., 2014). Consequently, where non-random species loss occurs, communities in simplified habitats are unlikely to be a random subset of those found in less disturbed habitats. Comparing functional traits of organisms in communities from habitats of differing land-use intensity can indicate whether such non-random species loss is occurring. This is of interest because of the disproportionate loss of ecological functions that arises as species, or groups of species, with particular functions are lost (Cardinale et al., 2012).

Due to their abundance and diversity, measuring insect communities in their entirety is impractical. An alternative method to determine insect diversity and abundance is to use an indicator group. While no indicator group is perfect for every use, the great diversity of parasitoid hymenopteran wasps, along with their close ecological relationship with practically all other insect groups (Gauld and Bolton, 1988; Quicke, 1997) means that abundance and species richness within this group corresponds closely to abundance and richness of several other groups of insects in agricultural grasslands (Anderson et al., 2011).

In this study, we investigate insect species richness and abundance, using parasitoid wasps as an indicator group, on commercial livestock grazing properties where livestock were managed with SDG and compared with more continuous grazing management typical of RP. Specifically, we asked (1) whether there was increased parasitoid wasp richness and abundance in areas managed with SDG (2) whether nearby woody vegetation cover was

influencing the abundance and diversity of parasitoid wasps and (3) whether parasitoid hymenopteran communities in pastures managed with RP were a non-random subset of parasitoid communities in pastures managed with SDG (i.e., whether non-random species loss was occurring).

6.3 Methods

6.3.1 Study region

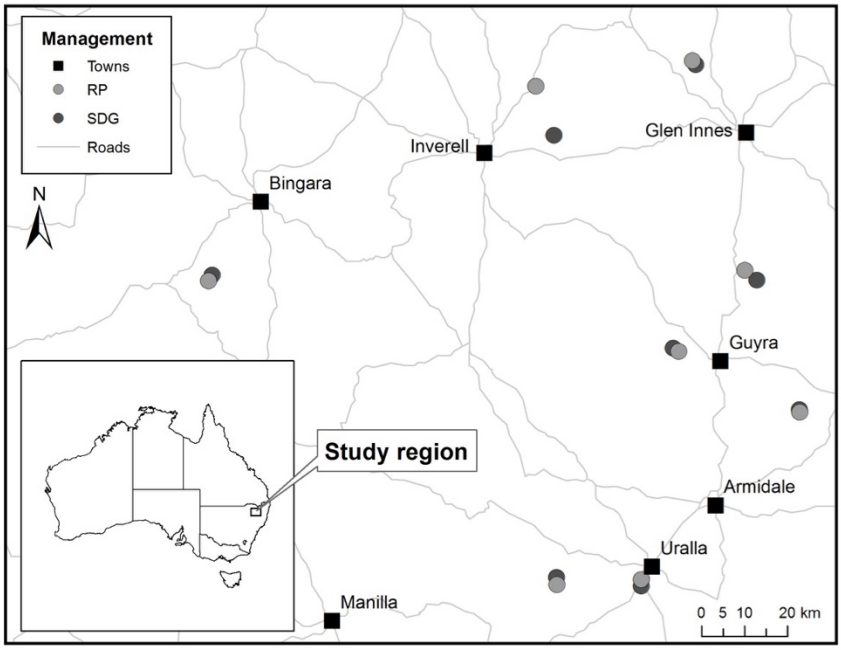

Figure 6-1 Location of the study region showing major towns (black squares) and the location of the 16 study properties. Properties that implemented grazing management more typical of RP are light grey circles; SDG properties are dark grey circles.

The study was conducted on the Northern Tablelands and North West Slopes of New South Wales, Australia, between 29° 38′ S, 15° 46′ E and 30° 39′ S, 150° 27′ E (Figure. 6.1). Altitudes ranged from almost 700 m a.s.l. in the west to just below 1400 m a.s.l. in the east. The climate of the region is sub-humid with summer-dominant rainfall and winter months dominated by cool, dry air from the continental interior or from the Southern Ocean. Severe frosts often occur in winter and there is a rainfall and altitudinal gradient across the region,

with generally greater precipitation and lower evapotranspiration at higher elevations in the east (Lea et al., 1977; Lodge and Whalley, 1989). The geology of the region is complex and largely determines soil characteristics (Lea et al., 1977; Lodge and Whalley, 1989). Tertiary basalts are mostly found at higher elevation and are underlain by late Permian granites intruded into Palaeozoic sediments. Accordingly, there are three major soil types in the region: fertile basaltic soils, infertile coarse to fine granitic soils and relatively infertile soils derived from metamorphosed sediments known locally as 'trap' soils.

6.3.2 Site selection

We selected naturalised pasture sites (terminology of Allen et al., 2011) on 16 commercial grazing properties. Twelve of these properties were the same properties as those used for data collection in Chapters 4 and 5 with an additional four properties used in this study. Eight of the properties were managed with SDG and eight neighbouring or nearby properties with matching soil types, aspects and areas of naturalised pastures managed with what we defined as usual regional practice (RP hereafter). Six of the eight paired properties were higher elevation tableland sites, with two property pairs located on the slopes of the study region. Details about management practices were determined through initial contact with managers, and then later with a formal questionnaire. On SDG properties, stock grazed intensively at high stocking densities for just a few days, in some cases 1–2 days, with pastures then rested until key forage species and pasture herbage mass had recovered. The decision to return stock was based on pasture monitoring with the length of rest varying with growing conditions. This planned rest resulted in overall rest-to-graze ratios (amount of time rested compared to amount of time grazed) of between 13:1 and 60:1 (median = 27:1). On individual properties, this ratio varied at different times depending on climatic conditions. Some SDG managers implemented very short graze periods (i.e., 1–2 days) while others

grazed for several days at a time. Smaller paddock sizes are necessary to implement the large rest-to-graze ratios typical of SDG with paddock sizes on SDG properties between 2–16 ha (median = 10 ha) in this study. These small paddock sizes necessitate increased density of fencing and water infrastructure (Savory and Parsons, 1980; Sherren et al., 2012). Managers implementing SDG minimised the use of external inputs including minimising hand-feeding of stock in winter and extended dry periods and limiting application of inorganic phosphorus-based fertilisers (only one of the eight SDG properties had applied superphosphate in the past 5–10 years, as opposed to all eight of the RP properties). One of the SDG properties applied poultry manure. On the properties examined, SDG management had been in place for between 7 and 25 years (median = 15 years). The contrasting RP management approach was more typical of grazing management on naturalised pastures in the region. This was mainly continuous grazing, with some rotation of stock where forage was limiting, but with relatively long graze and short rest periods in response to forage availability (Scott et al., 2013; Shakhane et al., 2013). In this study, rest periods on RP properties had rest-to-graze ratios that ranged from 0 to 3:1 (median = 0.5:1) and paddocks were large (median 20 ha, range 12–272 ha) in comparison to SDG paddocks. Superphosphate application on RP properties was typically on a calendar rotation (either annual or biennial) and stock were hand-fed to a greater extent than on SDG properties. The RP properties had been managed in the same way for at least the past 10 years.

Few pastoral managers had implemented SDG management in the region for >10 years. Consequently, local networks were used to locate suitable SDG properties, with each property paired with a suitable neighbouring or nearby RP property. While it is acknowledged that this was not a random selection of participants, given the constraint of a low number of properties managed with SDG for an adequate timeframe, it was the only practical way of selecting sites that respected neighbour relations and controlled for local soil

and climatic conditions. Livestock were predominantly cattle (*Bos taurus*), with the exception of four properties (three SDG and one RP) that also ran sheep (*Ovis aries*) for fine-wool production. The study design and low number of suitable SDG properties precluded the two livestock types from being treated separately in the study.

Due to varying levels of engagement with producers, only coarse information on stocking rates could be obtained from the formal survey undertaken. Stocking rates are listed in Appendix 8. One of the criteria for the selection of properties for the study was that they were commercial enterprises. Therefore, while the stocking rate data was relatively coarse, we consider it a reasonable assumption that managers stock according to their profitability goals balanced with the need to sustain their resource base as grazing management is a trade-off between maximising yield and minimising the risk of degradation (Jakoby et al., 2015).

6.3.3 Insect sampling

From mid-January until late February 2014, insects were sampled using a sweep net. On each property, three separate transects of approximately 250 m were sampled with 150 sweeps each. On the SDG properties, one transect was sampled in the most recently grazed paddock with the lowest biomass in the grazing cycle, a second transect was sampled in the paddock that was currently being grazed by livestock (intermediate biomass) and a third transect in the paddock where livestock were next to be moved to (i.e., with the highest biomass due to the longest rest in the grazing cycle). On the RP comparison property, three separate transects at least 500 m apart were sampled in paddocks where livestock were currently grazing. Due to the nature of management, livestock were grazed at lower density in the RP comparison paddocks. This sampling strategy was judged optimal to account for the equivalent level of grazing and retained pasture herbage on comparison properties.

Following collection, insects were placed in a 70% ethanol solution for transport to the laboratory and subsequent sorting.

Due to their abundance and diversity, measuring insect communities in their entirety is impractical and the use of an indicator group is an alternative way of assessing insect populations. Parasitoid hymenopterans are a useful indicator group as they occupy a high trophic position and have a close ecological relationship with arthropod communities (Gauld and Bolton, 1988; Quicke, 1997). Their abundance and species richness also corresponds with that of several other groups of insects in agricultural grasslands (Anderson et al., 2011). Hence parasitoid hymenopterans were used in this study as an indicator of overall insect diversity. Following the sorting of insect collections to order, parasitoid hymenopterans were identified to the lowest taxonomic level possible given available expertise. At a minimum this was to family level, but in many cases to sub-family or genus.

6.3.4 Woody vegetation cover

Surrounding woody vegetation communities can influence parasitoid diversity and abundance (Altieri, 1993) and is usually reduced by heavy continuous grazing practices (Vesk and Dorrough, 2006; Fischer et al., 2009). To determine whether wooded vegetation cover influenced insect biodiversity, the cover of woody vegetation was recorded in circles of radii, 100, 200 and 500 m, from the central point of insect collections. These distances were selected because parasitoid wasps do not typically travel over large distances and while dispersal distance depends on body size, 500 m is generally considered the upper limit for foraging range (Kruess and Tscharntke, 1994, 2000). Vegetation cover was determined using ArcGIS (ESRI, 2014).

6.3.5 Landscape function analysis

To better understand the relationship between pasture attributes that might influence parasitoid abundance and diversity we used the nutrient cycling index from Landscape Function Analysis (LFA; Tongway and Hindley, 2005) as this index was heavily influenced by perennial plant cover and cover and depth of litter in our analysis. Detailed explanation and methods are provided in Chapter 4 and Appendix 10). Insects were sampled towards the end of the summer when recent rainfall was below average (Figure. 6.2). LFA assessments were done at the end of the following winter. Although separated in time, there was very little growth in the intervening period. Therefore, the LFA assessments were considered relevant to conditions at the time of collection of parasitoids.

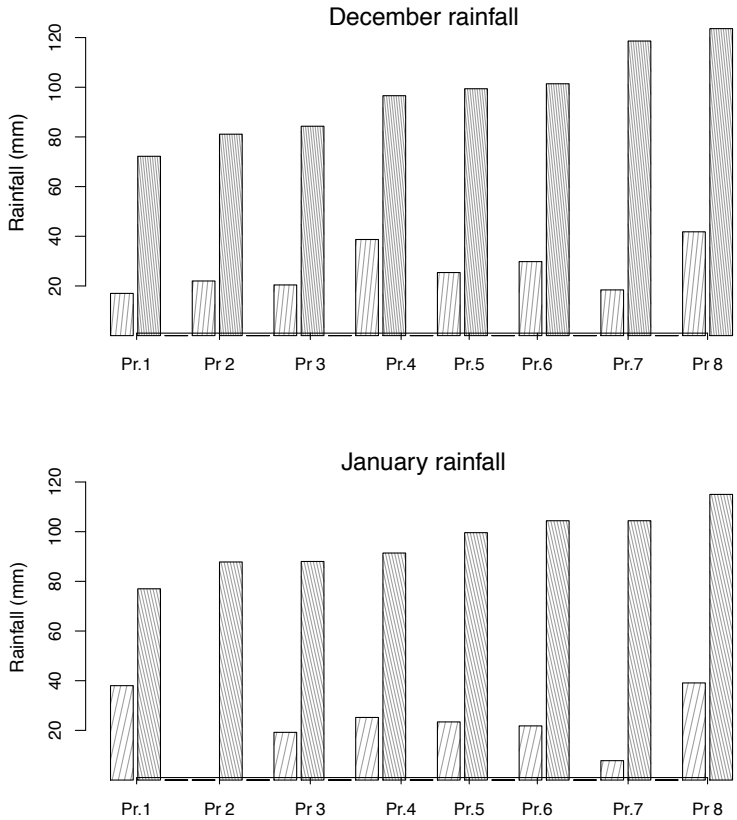

Figure 6-2 Rainfall (mm) for the two months preceding sampling for parasitoid wasps (December 2013 and January 2014 - light grey bars) at each location alongside the long-term median monthly rainfall for December and January (darker grey bars) for each location. Pr. 1 = property pair 1, Pr. 2 = property pair 2 etc.

6.3.6 Statistical analyses

To model insect diversity and abundance in response to management approach and other covariates, a Poisson Generalised Linear Mixed Model (GLMM) with a log-link function was used with the MASS package in R (Venables and Ripley, 2002). The Poisson distribution with a log-link function is typically used for count data (Quinn and Keogh, 2002). Fixed covariates used in the models were management type (categorical, SDG vs RP), number of significant rainfall events (>10mm) in the preceding month and preceding 2 months (continuous), time (in years) since conversion to SDG, percentage of nearby woody vegetation cover (continuous) and the LFA nutrient cycling index (continuous). For one of the property-pairs, the median value for nutrient cycling for each management type from the seven other locations was used because we were unable to undertake LFA at this location. We considered this acceptable due to the low amount of variation in the nutrient cycling index for each management approach (Figure 4.5; Chapter 4) and the substituted values were conservative. Property pairs (replicates) were used as a random intercept in models. Models were checked for goodness-of-fit using the dispersion statistic (residual deviance divided by degrees of freedom) and were considered significant if the test statistic (*Z*-value) was significant at the $p = 0.05$ level.

The LFA nutrient cycling index and the rest-to-graze ratios were both closely associated with management approach (Spearman rank order correlation: $r_s = 0.81$, $p < 0.001$ and $r_s = 0.70$, $p < 0.001$ respectively). Consequently, these three explanatory variables could not be used together in analyses. Nevertheless, due to the high likelihood that perennial cover and increased cover and depth of litter (which both contributed substantially to the LFA nutrient cycling index) supported higher insect richness and abundance through provision of food and

habitat resources (Luff, 1966; Dennis et al., 1998), we assessed nutrient cycling independently of management approach.

Dissimilarity matrices of parasitoid abundance data were ordinated using non-metric multidimensional scaling (NMDS; metaMDS package in R). The three-dimensional solution was retained (stress value = 0.086). Continuous predictor variables were related to community composition using a vector fitting approach (Envfit in R) with the resulting R^2 value giving the relationship between the vector and the composition of the community in multi-dimensional space. Differences between groups were tested using the Adonis function. Analyses were conducted using the Vegan package in R (Oksanen et al., 2018).

We tested whether parasitoid communities from species-poor sites were a subset of parasitoid communities from species-rich sites (i.e., nested), with nested systems indicating that species loss is non-random (Rader et al., 2014). The level of nestedness was determined using the NODF function in R (Almeida-Neto et al., 2008). This metric is based on the extent of paired overlap and decreasing fill of a matrix where rows are ordered according to descending frequency of species and columns ordered according to descending frequency at sites. The percentage overlap of species for each pair of columns and rows is determined, with both outputs included in analyses (Ulrich and Gotelli, 2007; Almeida-Neto et al., 2008; Ulrich et al., 2009). The resulting metric was tested for significance against 100 null matrices using the null model described in Patterson and Atmar (1986). Analyses were performed using the 'vegan' package in R (Oksanen et al., 2018). For all statistical analyses we used R 3.6.1 (R Core Team, 2018).

6.4 Results

Across 16 properties, 742 individual parasitoid wasps were collected. Of these, there were 254 unique morphospecies in 25 parasitoid families. Some 77% (573) of individuals were from the superfamily Chalcidoidea, with the remainder from Ceraphranoidea, Chrysidoidea, Cynapoidea, Evanoidea, Ichneumonidea, Platygasteroidea, Proctotrupoidea and Vespoidea (Table 6.1). For 149 (63%) of these species, only a single individual was found. All individuals were identified at least to family level; 66 (28%) to sub-family and 29 (12%) to either genus or subgenus; 604 individuals (81%) were from families that were always parasitic, or in rare cases predaceous, with the remaining 138 individuals (19 %) from either Eulophidae or Eurytomidae (which includes both parasitic wasps and gall formers). Five hundred and seventy-four individuals (90%) were terebrant (with an ovipositor that bores and deposits eggs) compared with only 21 (10%) aculeate individuals (with an ovipositor that is modified into a sting).

6.4.1 Parasitoid species richness and abundance

Parasitoid species richness and abundance were both significantly greater on SDG properties than RP properties ($Z = 3.05$, $p = 0.002$ and $Z = 2.52$, $p = 0.012$, respectively; Figure. 6.3a and d). Richness and abundance were also both significantly greater where there had been more substantial (>10 mm) rainfall events in the previous month (richness: $Z = 3.06$ $p = 0.002$; abundance: $Z = 2.67$, $p = 0.007$; Figure. 6.3a and c). Models were under-dispersed (dispersion parameter = 0.48 and 0.14 for richness and abundance, respectively), suggesting the significance levels of the *Z*-statistic are conservative for both. Models run using the LFA nutrient cycling index instead of management approach were also closely associated with greater richness and abundance of parasitoids ($Z = 3.20$, $p = 0.001$; $Z = 3.34$, $p < 0.001$;

Figure. 6.3b and d). These models were also under-dispersed (dispersion statistic = 0.38 and 0.14 for richness and abundance, respectively). There was no influence of nearby woody vegetation cover on either parasitoid richness or abundance.

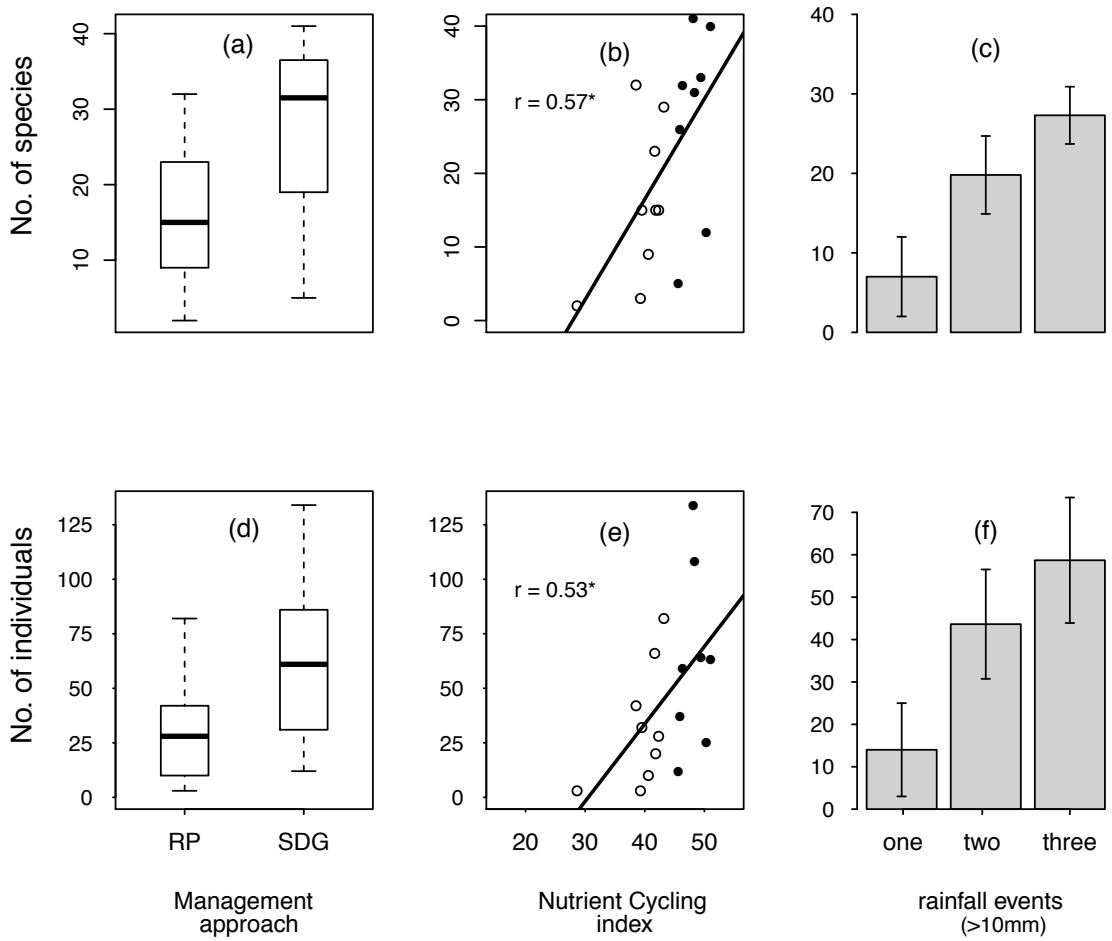

Figure 6-3 Relationships of parasitoid richness and abundance in response to management approach, LFA nutrient cycling index and recent rainfall. Boxplots show differences in (a) richness and (d) abundance under contrasting management approaches. Scatterplots in (b) and (d) show changes in response to LFA nutrient cycling index (black circles are SDG sites and open circles are RP sites); 'r' is Spearman's rank order correlation, with asterix denoting the *p*-value (associated *p*-values are 0.012 and 0.028 for species richness and abundance respectively). Barplots in (c) and (f) show parasitoid richness and abundance respectively in response to recent rainfall events (> 10 mm). For species richness the AIC for the model that included nutrient cycling was 135.4 and for the model that included management approach was 135.0 (null model AIC = 143.25) and for species abundance the AIC was 166.8 for both models (null model = 172.6).

6.4.2 Community composition

The composition of parasitoid communities between contrasting management approaches showed some separation in ordination space but was not quite significant at the $p = 0.05$ level (Figure. 6.3). However, the length of time since conversion to SDG was significant in influencing parasitoid community composition (Figure. 6.3), with greater

separation in ordination space as the length of time under SDG management increased. The rest-to-graze ratio was also associated with changes to parasitoid community composition, with greater changes compared to RP as this ratio increased, but as with management approach, this effect was not quite significant at the $p = 0.05$ level (Figure. 6.3).

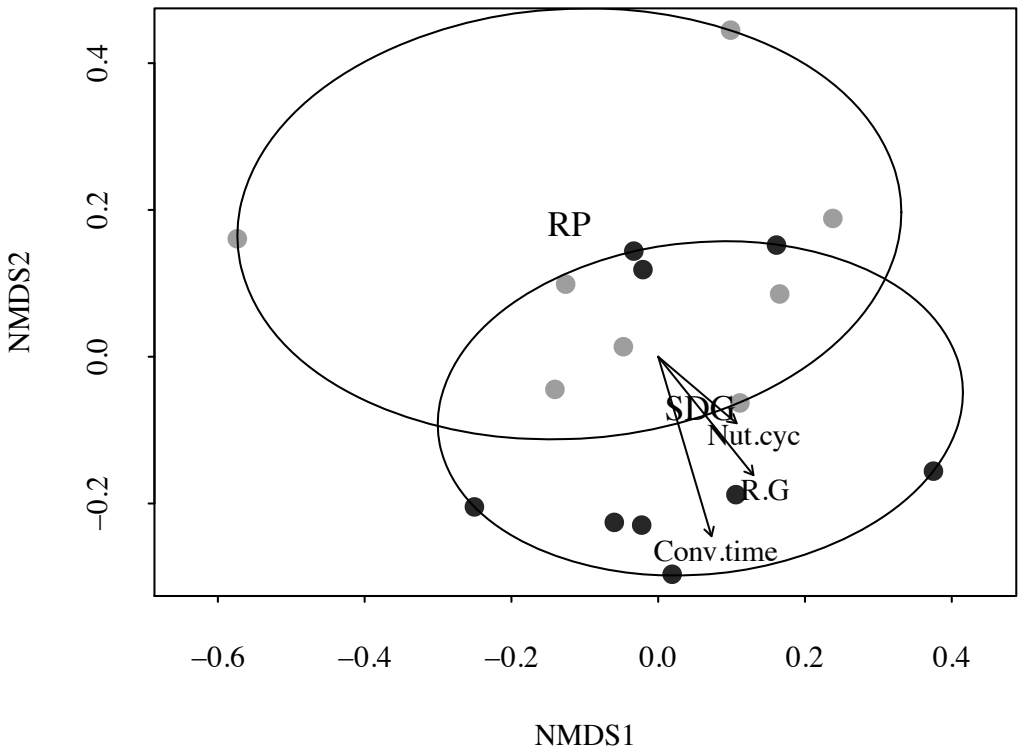

Figure 6-4 Multi-dimensional scaling plots of insect communities. Communities of insects on SDG properties are shown with black circles and those on more RP properties are shown with light-grey circles. Ellipses surround communities within each of the two management categories. Arrows show the direction of change for rest-to-graze ratio (R.G), LFA nutrient cycling index (Nut.cyc) and the time since conversion to SDG. The length of each vector represents the relative magnitude of change in parasitoid communities. The length of time since conversion to SDG management was significant ($R^2 = 0.48$, $p = 0.01$); the rest-to-graze ratio was almost significant at the 0.05% level ($R^2 = 0.34$, $p = 0.07$ and the difference between management approaches was not significant ($p = 0.07$) and had a low R^2 value (0.15).

Table 6.1 Number of individual wasps (abundance) and no of species found (No. of species) in each parasitoid hymenopteran family. Wasp type and guild refers to whether species in the family are known to be parasitoids (P), predaceous wasps (Pr) or gall formers (Ga; some families include both parasitic and gall-forming species). Guild refers to whether the family lays eggs in adult hosts (A), eggs (E), larvae or nymph (L), pupae (P), are hyperparasitoids (H; lay their eggs in other parasitoids). No. of insect orders is the number of insect orders that the parasitoid family is known to parasitise or predate on.

Family	Abundance	No. of species	Wasp type and guild	No. of insect orders
Aculeate families				
Aphelinidae	12	10	E H L P	5
Braconidae	46	10	E H L	15
Ceraphronidae	5	4	E	1
Chalcididae	84	9	E H L	5
Diapridae	16	11	H P	3
Encyrtidae	184	57	A E H L P	13
Eucharatidae	10	3	L	1
Eucoilidae	3	2	L	1
Eulophidae	113	49	A E H L P Ga	8
Eupelmidae	56	10	E L P	5
Eurytomidae	25	8	P H Ga	1
Evanidae	7	1	L H P	4
Figitidae	1	1	L	1
Ichneumonidae	2	2	L H P	7
Megaspiliidae	1	1	H L	2
Myrmaridae	22	7	E H	5
Pteromalidae	17	14	E H L P	9
Scelionidae	45	20	E	4
Tetracampidae	4	3	E H	3
Torymidae	19	4	E H L	5
Trichogrammatidae	27	7	E	7
Terebrant families				
Bethylidae	35	17	Pr	2
Dryinidae	4	2	L	1
Mutillidae	1	1	A E H L P	4
Pompilidae	1	1	E H Pr	2

The nestedness analysis suggested that parasitoid communities on species-poor sites were random subsets of the communities found on species-rich sites ($Z = 9.83$, $p = 0.18$). No significant differences between management approaches were found for egg, larval or pupal guilds. There were also no differences in the abundance of parasitoids belonging to groups that include hyperparasitoids between the two management approaches.

6.5 Discussion

Insects are key components of grassland ecosystems and at the core of most terrestrial food webs (Seastedt and Crossley, 1984; Herbst et al., 2013) and insect biodiversity is threatened by multiple anthropogenic actions (Sánchez-Bayo and Wyckhuys, 2019). Due to the vast area of land that livestock grazing impacts, managing grazing in ways that facilitate increased biodiversity of insects and more complex resilient trophic webs could contribute substantially to counteracting these threats. In this study, we found large differences in the abundance and richness of parasitoid wasps with contrasting approaches to grazing management, with a much greater diversity and abundance of insects on properties managed with SDG in comparison with more usual regional practice.

Increased perennial herbaceous cover, including greater retention of herbage mass, and increased cover and depth of plant litter on the soil surface are associated with SDG management (Lawrence et al., 2019) are likely to be important drivers of the insect responses seen here. These ground-layer attributes provide essential food and habitat resources for lower trophic level insects that parasitoid hymenopterans depend on (Luff, 1966; Dennis et al., 1998). In general, as evidenced by our assessment of ground-layer plant communities, these resources were substantially less available on RP compared to SDG sites despite stocking rates being roughly equivalent (Lawrence et al., 2019). Tussocky perennial grasses and litter also intercept rain, dew and frost, with these grasses enhancing retention of leaf litter in interstitial spaces more than relatively prostrate species, thus supporting micro-environments that favour invertebrate species including insects (Recher and Lim, 1990; Dennis et al., 1998; McIntyre and Tongway, 2005). The close association of the LFA nutrient cycling index with both increased insect richness and abundance of parasitoids reflects this increase in longer-lived, robust perennial species. Although plant diversity is considered to

have a substantial influence on insect biodiversity (Andow, 1991), vegetation structure is considered to be at least as important (possibly more) than plant diversity (Southwood et al., 1979; Koricheva et al., 2000; Haddad et al., 2001; Woodcock et al., 2007) and is likely to be a more important factor than plant richness and diversity (which differed little between management types; Chapter 5) or plant community composition. Further, the extended times where pastures are undisturbed (i.e., sometimes as long as 90 days during rest periods) are likely to positively contribute to these outcomes by allowing a range of insect species to complete their lifecycles because of the lack of disturbance from grazing and trampling of livestock.

The clear response of insect biodiversity to changes in landscape management in commercial grazing operations highlights that there are opportunities to address concerns around insect decline, where grazing is a part of the agricultural matrix, through altered grazing management on a large-scale. However, there are a several questions that more detailed work could answer. These include examining insects from a range of trophic levels to determine whether responses are consistent across multiple groups; further exploring the importance of extended rest periods, particularly during critical times of insect lifecycles and examining if there are changes to functional diversity in insect communities in response to contrasting management approaches. Determining the extent to which the increased diversity and abundance of insects seen on SDG properties benefits vertebrate wildlife. Connected to this is determining SDG management also supports other landscape characteristics such as the presence of coarse woody debris, greater protection of remnant vegetation and increased natural tree regeneration/decreased clearing of living and dead trees that facilitate the conservation of biodiversity more generally. Finally, there would be value in determining the potential for these altered approaches to grazing to contribute to crop protection and pollination services in landscapes where livestock grazing is a part of a complex mosaic of

farmland consisting of multiple land-uses including cropping and horticulture (*sensu* Tscharntke et al., 2002; Steffan-Dewenter, 2003). This would be especially valuable where livestock grazing is a part of a mixed farming system that includes cropping and grazing enterprises.

There is a low adoption rate of SDG approaches to grazing management in the region of study (R. Lawrence *pers. obs.)* as well as contention around the benefits (or otherwise) of these approaches (Briske et al., 2008; Briske et al., 2011; Provenza et al., 2013; Teague et al., 2013). However, the debate largely focuses around benefits for short and long-term animal productivity with limited attention placed on alternative goods and services that come from grazed landscapes (Chapter 3). Studies that have considered insect responses to approaches similar to SDG have generally shown favourable outcomes (Farruggia et al., 2012; Saunders and Fausch, 2012; Enri et al., 2017) with the study by Enri et al. (2017) that explicitly considered insect biodiversity as well as animal production, finding no reductions in animal productivity where strategic rest from grazing was applied. This is an area in need of further research to maximise ecosystem services from insects in agricultural landscapes as well as to contribute in some way to buffering insect populations against declines in response to agricultural intensification.

Despite the differences in abundance and richness of parasitoids between management approaches there are clear limitations to this study. These are mainly the low sample size arising from one collection event. This limits the conclusions that can be made and may also have resulted in no evidence of whether parasitoid communities on species poor sites were a subset of those on more species rich sites (nestedness). The strong response seen despite this small sample size indicates that insect responses to the contrasting grazing management approaches seen here are worthy of further investigation. This would also enable a greater

understanding of the range of landscape attributes insects were responding to beyond just grazing management and attributes relating to nutrient cycling in the ground-layer.

Conclusion

Insect biodiversity is an important component of grazed landscapes and altered approaches to livestock management such as SDG have the potential to reduce the impact of heavy impacts of grazing livestock on insect populations. In addition to responding to increases in vegetation structure and the abundance of perennial tussock species under SDG management, the lack of disturbance due to the extended rest of SDG management likely allow insects to complete lifecycles thereby supporting higher biodiversity of insects in comparison to more usual RP that was largely continuous throughout the year. Although positive impacts on insect populations can be achieved with reduced grazing intensity, such a strategy comes with unacceptable losses of animal production. Although it requires more accurate quantification, stocking rates across both management approaches were similar and so losses of animal production are unlikely to be happening with SDG. The clear increase in richness and abundance of insects seen in this study, alongside global concerns for insect biodiversity with land-use change and intensification, and in the context of the importance of insects for crop pollination and pest control services, provide a compelling reason to further investigate responses of insect communities to altered grazing practices.

Chapter 7 Synthesis and general conclusions

Livestock grazing is a major contributor to landscape degradation and biodiversity loss on a global scale. It is also an essential contributor to food and economic security for billions of people worldwide (DeRamus, 2004; Michalk et al., 2019) and when managed appropriately can co-exist with and even facilitate high levels of biodiversity (IUCN, 2008). Yet, the relationships between plant and animal diversity, landscape function and grazing are complex and relate to current management practices as well as historic impacts relating to agronomic production. Due to this complexity many important aspects of grazing management that alter impacts of livestock remain poorly studied. Areas where research is lacking include understanding appropriate stocking rates and exploring further the potential of extended and targeted rest to simultaneously achieve meaningful levels of animal production as well as biodiversity conservation, while also considering additional benefits such as carbon sequestration, pollination and pest control in crops, water purification and social well-being that is likely to co-occur with biodiversity conservation in landscapes managed also for animal production.

This thesis set out to further our understanding of the potential for various forms of SRG to contribute to continuing animal production alongside improved biophysical and biodiversity conservation outcomes in comparison to more continuous grazing management practices. These combined goals are necessary to maintain livestock production while achieving improved stewardship of the world's grazed lands. To understand global trends in grazing practices upon ecological and production metrics, I conducted a review and meta-analysis of literature around grazing practices incorporating rest (Chapters 2 and 3), followed by an empirical investigation of ground-layer biophysical and biodiversity responses across

several properties managed in contrasting ways in the New England region of NSW, Australia (Chapters 4 to 6).

This chapter will briefly summarise the findings of these reviews and studies and discuss them in the context of the potential to effectively achieve both ecological and animal production outcomes in livestock grazing systems. It will also identify existing gaps in our understanding and propose future research directions to increase knowledge in this field.

7.1 Summary of thesis findings

The global meta-analyses presented in Chapter 2 compared a range of animal production and ecological responses and highlighted benefits for animal production and the biophysical outcomes groundcover and biomass in response to SRG. These benefits were especially apparent where the length of rest relative to graze duration increased (rest-to-graze ratio). This is a factor that has rarely been included in analyses comparing grazing systems previously.

In considering the extent to which studies investigating animal production were simultaneously considering aspects of biodiversity conservation, and the potential to integrate these dual outcomes, it is clear that minimal attention has been given to understanding the potential to manage for both outcomes across the world's grazing lands. Although there are rational reasons for this, it is clearly a major gap in our understanding of how to best meet the competing demands of continuing food and fibre production while also addressing biodiversity decline.

In the regional study, favourable outcomes were found for a range of attributes with SDG management in comparison to more usual regional practice. Where pasture composition and biophysical condition of the ground-layer were assessed, substantial improvements

relating to landscape functioning and pasture composition for livestock productivity were seen.

Finally, a small number of studies have considered responses of insect communities to grazing management that incorporates extended rest. These studies have found largely favourable responses. Despite the limitations of the data presented the clear increases in richness and abundance of parasitoid hymenopterans seen in the study with SDG management suggest there are substantially improved habitats for insects with SDG grazing compared to more usual practices that are largely continuous. This has implications for a range of ecological processes such as the provision of food resources for wildlife and other services supported by invertebrates in grasslands.

7.2 Research significance

The global review of literature around SRG presented here, alongside the study of commercial grazing properties managed for long timeframes with SDG (a form of SRG), has demonstrated a range of benefits of incorporating extended and planned rest into grazing systems for both ecological and production outcomes. In the global review as well as the regional study, the length of rest time in relation to graze duration was important. This is a factor that is rarely considered in studies comparing rotational and continuous grazing (di Virgilio et al., 2019; McDonald et al., 2019a). Appropriate stocking rates, generally around 75% of maximum, are considered the major driver enabling long-term sustainability for continuing livestock production in extensive and naturalised grasslands (Michalk et al., 2019), although where biodiversity outcomes are a consideration livestock distribution is also important (Tonn et al., 2019). Implementing appropriate stocking rates does not necessarily require the incorporation of strategic or lengthy rest periods. However, the fencing and water infrastructure required to implement extended, monitored and strategic rest enables increased

control over animal distribution across landscapes. This then enhances the capacity to manage for identified goals such as the recovery of key forage species or protection of sensitive ecological areas such as riparian zones or areas of higher biodiversity value. Planned and strategic rest is therefore likely to improve the capacity to optimise multiple outcomes in the broader landscape as well as pastures.

To-date, it is clear that most ecological studies considering biodiversity responses to SRG report variables relating to the management of livestock inadequately. Similarly, studies focusing on animal production outcomes are rarely considering the potential for biodiversity conservation. With a greater effort to understand grazing management approaches that promote additional or alternative goods and services from grazed landscapes, significant gains could be achieved.

7.3 Future research directions

A range of opportunities to further our understanding of the potential to achieve biodiversity, as well as meaningful levels of animal production in grazed landscapes, are highlighted by this thesis:

(1) The importance of incorporating the length of rest relative to graze duration in research needs to be considered. This was found to be an important factor in both the global review and regional study, and rarely intentionally considered. It was particularly difficult to determine in studies that focused on ecological outcomes. In addition to the length of rest relative to graze duration, the timing of rest, in relation to plant and animal species of interest is also likely to be important and should be considered.

(2) landscape function differed substantially between contrasting management approaches examined in the regional study. Although LFA has been used and verified in a

range of environments (Yates and Hobbs, 2000; Ata Rezaei et al., 2006; Maestre and Puche, 2009; Zucca et al., 2013), and is a useful tool for monitoring change, the quantitative benefit indicated by each index is unverified in temperate grazed pastures. As this is a robust and efficient monitoring technique, considerable value would be gained through quantification of these LFA values across a range of soil types in temperate grazed pastures.

(3) the substantial differences in insect diversity and abundance suggests there is potential to alter grazing management from year and season-long, continuous grazing to that which incorporates targeted and planned rest to support increased insect biodiversity in agricultural landscapes. More detailed investigation into the drivers of insect biodiversity in response to grazing such as multiple sampling events alongside detailed collection of relevant factors likely to be influencing insect communities would assist this understanding. While it is likely that simple increases in height and structural heterogeneity of the grassland sward were influencing insect responses in this study, other factors such as recent climate patterns, nearby vegetation, fertiliser regimes and timing of rest periods are also likely to influence insect responses and require more detailed investigation. Examining these factors was outside the scope of this study but given the importance of insects to multiple ecological and ecosystem services, this is an important area for future investigation.

(4) Dorrough et al. in 2004 suggested there is potential with rotational grazing practices that incorporate rest to balance livestock production with biodiversity conservation. It was outside the scope of the study of properties in this thesis to consider biodiversity in mid and upper-storey vegetation, and for wildlife other than insects in all vegetation layers. While Dorrough et al. (2012) investigated diversity responses of insects, reptiles and plants, and Fischer et al. (2009) tree regeneration, in both those studies it was noted by the authors that there was considerable variation in skill levels of managers, length of rest periods and the

duration of time since management had changed to SRG practices. Thus, there is still a significant research gap and a need for further investigation of the potential of these outcomes to achieve multiple outcomes in a commercial-production context. Dorrough et al., (2004) also hypothesised that stocking rates would likely need to be reduced to achieve optimal biodiversity (Fig. 7.1). This thesis has highlighted that due to the general lack of animal production data in ecological studies, the extent to which this is true remains unclear.

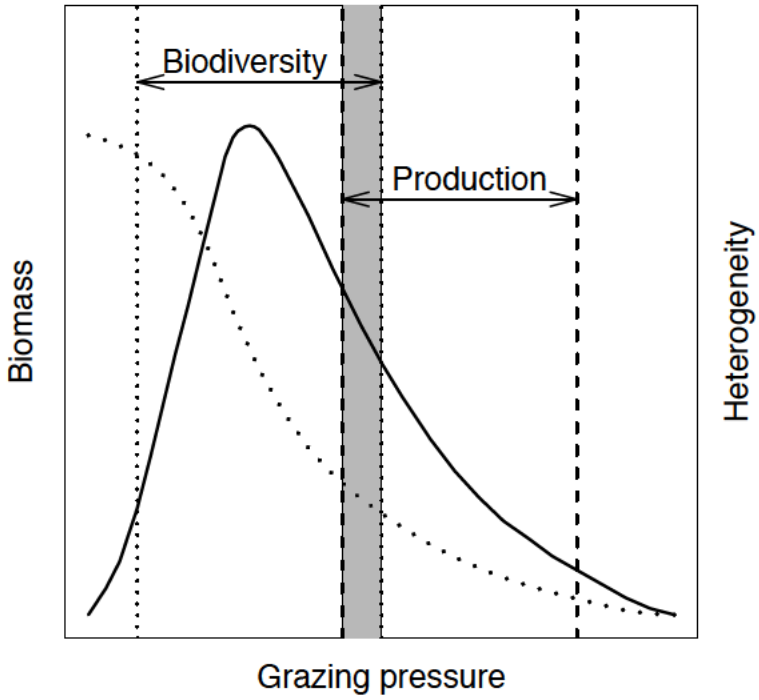

Figure 7-1 Hypothesised relationship between vegetation biomass (dashed line), structural heterogeneity (solid line) and grazing pressure (x-axis). Optimal conditions for biodiversity are likely to occur when biomass and heterogeneity are high, and in co contrast, optimal livestock production is likely to occur where biomass is moderate, and heterogeneity is low (figure recreated from Dorrough et al., 2004).

(5) while it is clear that there are opportunities to improve biodiversity outcomes within grazed landscapes using SRG it is unclear what the optimal number of paddocks (or grazing areas where livestock are shepherded) is for a given area of land. This is relevant because of costs associated with intense fencing and water infrastructure (or labour) required to implement extended rest periods that can impact profitability of businesses (Briske et al., 2014). This infrastructure can also inhibit the movement of wildlife across landscapes (Connelly et al., 2000; Silcock, 2018) thus negatively impacting biodiversity. It is also

important to acknowledge that heavy grazing pressures are a bigger driver of landscape degradation and biodiversity loss than altered approaches to grazing management (O'Reagain and Turner, 1992; Ash and Stafford Smith, 1996). Moderate grazing pressures in large paddocks also lead to heterogeneous grazing patterns (Adler et al., 2001; Fuhlendorf and Engle, 2001; Tonn et al., 2019) which, while not maximising animal production outcomes, can positively contribute to biodiversity outcomes if stocking numbers are moderate and conservative. Therefore, attention needs to be given to explicitly understanding optimal grazing and rest regimes in the context of associated infrastructure and labour costs for the optimal achievement of both animal production and biodiversity conservation outcomes.

(6) There is compelling evidence that Earth's biological systems are critically threatened by global warming (IPCC, 2018) as well as biodiversity decline (UN, 2019). In addition to the need to balance the management of grazed landscapes with biodiversity conservation, there are also opportunities to sequester carbon in grazed landscapes (Conant et al., 2001). Practices that enable the sequestration of carbon also likely in many cases to support improved biodiversity conservation outcomes (De Deyn et al., 2011). In addition to the need for future research to ascertain the potential to integrate animal production and biodiversity conservation, implications for carbon emissions or sequestration should also be considered.

Conclusion

The range of favourable outcomes seen in response to grazing practices that incorporate extended rest indicates there is potential with this approach to balance animal production, general landscape sustainability and biodiversity conservation goals. However, there are substantial gaps in our knowledge around how to successfully integrate multiple outcomes. In addition to protecting biodiversity outside formal reserve systems, improving biodiversity of animals such as insects has additional benefits for wildlife conservation and the provision of

ecosystem services such as pest control and pollination in agricultural landscapes where grazing is a part of the agricultural matrix, as well as many other ecological functions. The incorporation of extended, planned and strategic rest is likely to be a useful tool to help facilitate an appropriate balance between animal production and alternative outcomes in the world's grazed landscapes.

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

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Appendices

Appendix 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, 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 or points (for richness, diversity, ground cover and biomass) 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/semi-arid, (3) temperate, and (4) cold/continental	Based on Koppen-Geiger climate classification (Peel *et al.*, 2007)
Rest-to-graze ratio	Length of time an area of land was rested relative to length of time an area of land was grazed	Information provided in text. Length of rest time ÷ length of graze time (relative to the length of a typical grazing season or the length of continuous grazing treatment)
Stocking rate difference	Difference in stocking rate between SRG and CG contrasts, with SRG as the reference (4 levels): (1) greater, (2) equal, (3) lower, (4) unknown	Information provided in text

Appendix 2. List of studies included in meta-analyses with response variables from each paper included in the meta-analyses denoted. 'R' = plant species richness; 'D' = plant diversity; 'B' = biomass; 'G' = ground cover; 'W' = individual animal weight gain; 'P' = animal production per hectare; SRG–CG = contrast/s between strategic rest grazing and continuous grazing; SRG–UG = contrast/s between strategic rest grazing and ungrazed area.

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
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 (2): 185–190.					Y	Y	Y	
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.			Y	Y			Y	
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 (4):437–447.					Y	Y	Y	
Anderson, D. 1988. Seasonal stocking of tobosa managed under continuous and rotation grazing. Journal of Range Management: 41 (1): 78–83.			Y		Y		Y	Y
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.				Y			Y	
Badgery, W. B. 2017. Longer rest periods for intensive rotational grazing limit diet quality of sheep without enhancing environmental benefits. African Journal of Range & Forage science 34 (2): 99–109.			Y	Y			Y	
Badgery, W., G. Millar, D. Michalk, P. Cranney, and K. Broadfoot. 2017b. The intensity of grazing management influences lamb production from native grassland. Animal Production Science 57 (9): 1837–1848.						Y	Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Badgery, W., G. Millar, K. Broadfoot, D. Michalk, P. Cranney, D. Mitchell, and R. Van de Ven. 2017a. Increased production and cover in a variable native pasture following intensive grazing management. Animal Production Science 57 (9): 1812–1823.			Y				Y	
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 (2): 83–89.					Y		Y	
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.			Y		Y		Y	
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.					Y	Y	Y	
Biondini, M. E., and L. Manske. 1996. Grazing frequency and ecosystem processes in a northern mixed prairie, USA. Ecological Applications 6 (1): 239–256.			Y		Y		Y	Y
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 (90): 41–51.			Y				Y	
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.					Y		Y	
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 (4): 403–408.				Y	Y	Y	Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
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 (2): 94–96.					Y		Y	
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 (4): 261–264.						Y	Y	
Broadbent, P. 1964a. 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 (1): 218–223.					Y		Y	
Broadbent, P. 1964b. 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.					Y	Y	Y	
Bryant, H., R. Blaser, R. Hammes, and W. Hardison. 1961. Comparison of continuous and rotational grazing of three forage mixtures by dairy cows. Journal of Dairy Science 44 (9): 1742–1750.			Y			Y	Y	
Bungenstab, E., A. Pereira, J. Lin, J. Holliman, and R. Muntifering. 2011. Productivity, utilization, and nutritive quality of dallisgrass (Paspalum dilatatum) as influenced by stocking density and rest period under continuous or rotational stocking. Journal of Animal Science 89: 571–580.					Y	Y	Y	
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.					Y		Y	
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.					Y	Y	Y	
Burns, J., and D. Fisher. 2011. Stocking strategies as related to animal and pasture productivity of endophyte-free tall fescue. Crop Science 51 (6): 2868–2877.					Y	Y	Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
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 48 (1): 81–84.			Y				Y	
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.					Y	Y	Y	
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 (4): 657–663.	Y		Y				Y	
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.			Y					Y
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 (8): 1047–1058.					Y		Y	
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 (2): 161–168.			Y		Y		Y	
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.	Y	Y		Y				Y
Derner, J. D., and R. H. Hart. 2007a. Grazing-induced modifications to peak standing crop in northern mixed-grass prairie. Rangeland Ecology & Management 60 (3): 270–276.			Y				Y	
Derner, J. D., and R. H. Hart. 2007b. Livestock and vegetation responses to rotational grazing in short-grass steppe. Western North American Naturalist 67 (3): 359–367.				Y	Y		Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
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.					Y		Y	
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.			Y				Y	
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 (4): 271–284.					Y		Y	
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): 1–8.			Y				Y	Y
Donnelly, J., A. Axelsen, and F. Morley. 1970. Effect of flock size and grazing management on sheep production. Australian Journal of Experimental Agriculture 10 (44): 271–278.					Y		Y	
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 50 (6): 647–651.			Y					Y
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.	Y						Y	
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.			Y				Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
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 (1): 39-52.			Y				Y	
Enri, S. R., M. Probo, A. Farruggia, L. Lanore, A. Blanchetete, and B. Dumont. 2017. A biodiversity-friendly rotational grazing system enhancing flower-visiting insect assemblages while maintaining animal and grassland productivity. Agriculture, Ecosystems & Environment 241: 1–10.			Y		Y		Y	
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.					Y		Y	
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.	Y							Y
Fisher, C. E., and P. T. Marion. 1951. Continuous and rotation grazing on buffalo and tobosa grassland. Rangeland Ecology & Management/Journal of Range Management Archives 4 (1): 48-51.					Y	Y	Y	
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.	Y	Y						Y
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 (3): 627–635.	Y	Y					Y	Y
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 (6): 681–684.			Y	Y			Y	Y

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
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 (1): 82–89.	Y							Y
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 (8): 977–991.						Y	Y	
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 55 (2): 135–138.			Y				Y	
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 (1):163–171.					Y		Y	
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.	Y						Y	Y
Gutman, M., and N.G. Seligman. 1979. Grazing management of Mediterranean foothill range in the upper Jordan river valley. Journal of Range Management 32 (2): 86–92.			Y		Y	Y	Y	
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 43 (1): 64–68.					Y	Y	Y	
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 52 (6): 590–599.					Y	Y	Y	

Study	*R*	*D*	*B*	*G*	*W*	*P*	*SRG–CG*	*SRG–UG*
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.		Y	Y				Y	
Hafley, J. 1996. Comparison of Marshall and Surrey ryegrass for continuous and rotational grazing. Journal of Animal Science 74: 2269–2275.					Y	Y	Y	
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 (2): 97–103.			Y				Y	
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 (4): 560–565.			Y				Y	Y
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 (36): 161–174.	Y		Y	Y			Y	
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 (1): 62–76.			Y		Y	Y	Y	
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 (5): 992–1000.			Y				Y	Y
Harrington, G., and D. Pratchett. 1974. Stocking rate trials in Ankole, Uganda: I. Weight gain of Ankole steers at intermediate and heavy stocking rates under different managements. The Journal of Agricultural Science 82 (3): 497–506.					Y	Y	Y	

Study	*R*	*D*	*B*	*G*	*W*	*P*	*SRG–CG*	*SRG–UG*
Hart, R. H., J. Bissio, M. J. Samuel, and J. W. Waggoner Jr. 1993. Grazing systems, pasture size, and cattle grazing behavior, distribution and gains. Journal of Range Management, 46 (1): 81–87.					Y		Y	
Hart, R. H., M. J. Samuel, J. Waggoner, and M. Smith. 1989. Comparisons of grazing systems in Wyoming. Journal of Soil and Water Conservation 44 (4): 344–347.					Y		Y	
Heitschmidt, R., J. Frasure, D. Price, and L. Rittenhouse. 1982a. Short duration grazing at the Texas Experimental Ranch: Weight gains of growing heifers. Journal of Range Management 35 (3): 375–379.					Y	Y	Y	
Heitschmidt, R., M. Kothmann, and W. Rawlins. 1982b. Cow-calf response to stocking rates, grazing systems, and winter supplementation at the Texas Experimental Ranch. Journal of Range Management 35 (2):204–210.			Y				Y	
Heitschmidt, R., S. Dowhower, and J. Walker. 1987. Some effects of a rotational grazing treatment on quantity and quality of available forage and amount of ground litter. Journal of Range Management 40 (4):318–321.					Y	Y	Y	
Hellgren, E. C., A. L. Burrow, R. T. Kazmaier, and D. C. Ruthven III. 2010. The effects of winter burning and grazing on resources and survival of Texas horned lizards in a thornscrub ecosystem. The Journal of Wildlife Management 74 (2): 300–309.				Y				Y
Hidalgo, L., and M. Cauhépé. 1991. Effects of seasonal rest in aboveground biomass for a native grassland of the Flood Pampa, Argentina. Journal of Range Management 44 (5): 471–475.			Y				Y	
Hill, B., and D. Saville. 1976. Effect of management system and stocking rate on the performance of ewes and lambs. Australian Journal of Experimental Agriculture 16 (83): 810–817.					Y		Y	
Holechek, J. L., T. J. Berry, and M. Vavra. 1987. Grazing system influences on cattle performance on mountain range. Journal of Range Management 40 (1):55–59.					Y		Y	

Study	*R*	*D*	*B*	*G*	*W*	*P*	*SRG–CG*	*SRG–UG*
Holmes, W., and H. E. S. Osman. 1960. The feed intake of grazing cattle. I. Feed intake of dairy cows on strip and free grazing. Animal Science 2:131–139.					Y		Y	
Holst, P., D. Stanley, G. Millar, A. Radburn, D. Michalk, P. Dowling, R. Van de Ven, S. Priest, D. Kemp, and W. M. King. 2006. Sustainable grazing systems for the Central Tablelands of New South Wales. 3. Animal production response to pasture type and management. Australian Journal of Experimental Agriculture 46 (4): 471–482.			Y		Y		Y	
Hulme, P., B. Merrell, L. Torvell, J. Fisher, J. Small, and R. Pakeman. 2002. Rehabilitation of degraded Calluna vulgaris (L.) Hull-dominated wet heath by controlled sheep grazing. Biological Conservation 107 (3): 351–363.		Y					Y	Y
Hyder, D. N., and W. Sawyer. 1951. Rotation-deferred grazing as compared to season-long grazing on sagebrush-bunchgrass ranges in Oregon. Journal of Range Management 4 (1): 30–34.					Y		Y	
Jacobo, E. J., A. M. Rodríguez, N. Bartoloni, and V. A. Deregibus. 2006. Rotational grazing effects on rangeland vegetation at a farm scale. Rangeland Ecology & Management 59 (3): 249–257.		Y		Y			Y	
James, J. J., J. Davy, M. P. Doran, T. Becchetti, P. Brownsey, and E. A. Laca. 2017. Targeted grazing impacts on invasive and native plant abundance change with grazing duration and stocking density. Rangeland Ecology & Management 70 (4): 465–468.			Y					Y
Jochims, F., C. Poli, P. Carvalho, D. David, N. Campos, L. Fonseca, and G. Amaral. 2013. Grazing methods and herbage allowances effects on animal performances in natural grassland grazed during winter and spring with early pregnant ewes. Livestock Science 155: 364–372.					Y		Y	
Kahn, L. P., J. M. Earl, and M. Nicholls. 2010. Herbage mass thresholds rather than plant phenology are a more useful cue for grazing management decisions in the mid-north region of South Australia. The Rangeland Journal 32 (4): 379–388.			Y	Y				Y

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Kauffman, J. B., W. C. Krueger, and M. Vavra. 1983. Effects of late season cattle grazing on riparian plant communities. Journal of Range Management 36 (6):685–691.			Y					Y
Kioko, J., J. W. Kiringe, and S. O. Seno. 2012. Impacts of livestock grazing on a savanna grassland in Kenya. Journal of Arid Land 4 (1):29–35.			Y					Y
Kirby, D. R., M. F. Pessin, and G. K. Clambey. 1986. Disappearance of forage under short duration and seasonlong grazing. Journal of Range Management 39 (6): 496–500.			Y				Y	
Kitessa, S., and A. Nicol. 2001. The effect of continuous or rotational stocking on the intake and live-weight gain of cattle co-grazing with sheep on temperate pastures. Animal Science 72 (1): 199–208.			Y			Y	Y	
Kothmann, M. M., G. W. Mathis, and W. J. Waldrip. 1971. Cow-Calf Response to Stocking Rates and Grazing Systems on Native Range (La Produccion de Vacas y Becerros a Differentes Cargas y Sistemas de Pastoreo con Rotacion en Pastizales Naturales). Journal of Range Management 24 (2): 100–105.					Y	Y	Y	
Kreuter, U., G. Brockett, A. Lyle, N. Tainton, and D. Bransby. 1984. Evaluation of veld potential in east Griqualand using beef cattle under two grazing management systems. Journal of the Grassland Society of Southern Africa 1: 5–10.					Y	Y	Y	
Kruse, M., K. Stein-Bachinger, F. Gottwald, E. Schmidt, and T. Heinken. 2016. Influence of grassland management on the biodiversity of plants and butterflies on organic suckler cow farms. TUEXENIA 36: 97–119.	Y	Y					Y	
Laycock, W., and P. Conrad. 1981. Responses of vegetation and cattle to various systems of grazing on seeded and native mountain rangelands in eastern Utah. Journal of Range Management 34 (1): 52–58.				Y			Y	
Lennartsson, T., J. Wissman, and H.-M. Bergström. 2012. The effect of timing of grassland management on plant reproduction. International Journal of Ecology 2012: 1–9.	Y	Y					Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Leonard, S. W., and J. Kirkpatrick. 2004. Effects of grazing management and environmental factors on native grassland and grassy woodland, Northern Midlands, Tasmania. Australian Journal of Botany 52 (4):529–542.	Y						Y	
Levine, N., D. Clark, R. Bradley, and S. Kantor. 1975. Relationship of pasture rotation to acquisition of gastrointestinal nematodes by sheep. American Journal of Veterinary Research 36 (10): 1459–1464.					Y		Y	
Lodge, G., and K. King. 2006. Soil microbial biomass, labile and total carbon levels of grazed sown and native pastures in northern New South Wales. Australian Journal of Agricultural Research 57 (8): 837–845.			Y	Y			Y	
Lodge, G., S. Murphy, and S. Harden. 2003. Effects of grazing and management on herbage mass, persistence, animal production and soil water content of native pastures. 1. A redgrass–wallaby grass pasture, Barraba, North West Slopes, New South Wales. Australian Journal of Experimental Agriculture 43 (8): 875–890.			Y	Y	Y		Y	
Lomas, L. W., J. L. Moyer, G. Milliken, and K. Coffey. 2000. Effects of Grazing System on Performance of Cow-Calf Pairs Grazing Bermudagrass Pastures Interseeded with Wheat and Legumes. The Professional Animal Scientist 16 (3): 169–174.			Y		Y		Y	
Lucas, R. W., T. T. Baker, M. K. Wood, C. D. Allison, and D. M. Vanleeuwen. 2004. Riparian vegetation response to different intensities and seasons of grazing. Journal of Range Management 57 (5): 466–474.	Y	Y	Y	Y				Y
Ma, L., F. Yuan, H. Liang, and Y. Rong. 2014. The effects of grazing management strategies on the vegetation, diet quality, intake and performance of free grazing sheep. Livestock Science 161 (1): 185–192.			Y		Y	Y	Y	
Manley, W., R. Hart, M. Samuel, M. Smith, J. Waggoner Jr, and J. Manley. 1997. Vegetation, cattle, and economic responses to grazing strategies and pressures. Journal of Range Management 50 (6): 638–646.				Y			Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Martin, S. C., and K. E. Severson. 1988. Vegetation response to the Santa Rita grazing system. Journal of Range Management 41 (4): 291–295.			Y				Y	
Mathews, B. W., L. E. Sollenberger, and C. R. Staples. 1994. Dairy heifer and bermudagrass pasture responses to rotational and continuous stocking. Journal of Dairy Science 77 (1): 244–252.					Y		Y	
Mavromihalis, J., J. Dorrough, S. Clark, V. Turner, and C. Moxham. 2013. Manipulating livestock grazing to enhance native plant diversity and cover in native grasslands. The Rangeland Journal 35 (1): 95–108.				Y			Y	Y
McCaughey, W., K. Wittenberg, and D. Corrigan. 1997. Methane production by steers on pasture. Canadian Journal of Animal Science 77: 519–524.					Y		Y	
McCollum III, F. T., R. L. Gillen, B. R. Karges, and M. E. Hodges. 1999. Stocker cattle response to grazing management in tallgrass prairie. Journal of Range Management 52 (2): 120–126.					Y	Y	Y	
McCollum III, F., and R. L. Gillen. 1998. Grazing management affects nutrient intake by steers grazing tallgrass prairie. Journal of Range Management 51 (1): 69–72.			Y				Y	
McDonald, R., M. Van Leeuwen, and C. Harris. 1986. Effect of topping pastures 2. Lamb growth during summer. New Zealand Journal of Experimental Agriculture 14: 291–296.			Y		Y		Y	
McIlvain, E., and D. Savage. 1951. Eight-year comparisons of continuous and rotational grazing on the Southern Plains Experimental Range. Journal of Range Management 4 (1):42–47.					Y		Y	
Metzger, K., M. Coughenour, R. Reich, and R. Boone. 2005. Effects of seasonal grazing on plant species diversity and vegetation structure in a semi-arid ecosystem. Journal of Arid Environments 61 (1): 147–160.	Y	Y					Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Morley, F., D. Bennett, and G. McKinney. 1969. The effect of intensity of rotational grazing with breeding ewes on phalaris-subterranean clover pastures. Australian Journal of Experimental Agriculture 9 (36): 74–84.			Y				Y	
Moyo, B., S. Dube, C. Moyo, and E. Nesamvuni. 2011. Heavily stocked 5-paddock rotational grazing effect on cross-bred Afrikaner steer performance and herbaceous vegetation dynamics in a semi-arid veld of Zimbabwe. African Journal of Agricultural Research 6 (10): 2166–2174.			Y		Y		Y	
Murray, R., and J. Klemmedson. 1968. Cheatgrass range in southern Idaho: seasonal cattle gains and grazing capacities. Journal of Range Management 21 (5): 308–313.					Y		Y	
Nie, Z., and R. Zollinger. 2012. Impact of deferred grazing and fertilizer on plant population density, ground cover and soil moisture of native pastures in steep hill country of southern Australia. Grass and Forage Science 67: 231–242.				Y			Y	
Norman, H., M. Wilmot, D. Thomas, E. Barrett-Lennard, and D. Masters. 2010. Sheep production, plant growth and nutritive value of a saltbush-based pasture system subject to rotational grazing or set stocking. Small Ruminant Research 91 (1): 103–109.					Y		Y	
Nyako-Lartey, Q., and R. Baxter. 1995. The effects of different grazing regimes on the population dynamics of small mammals in the Eastern Cape. Transactions of the Royal Society of South Africa 50: 143–151.			Y	Y	Y		Y	Y
Oates, L. G., and R. D. Jackson. 2014. Livestock management strategy affects net ecosystem carbon balance of subhumid pasture. Rangeland Ecology & Management 67 (1): 19–29.			Y				Y	
Oates, L. G., D. J. Undersander, C. Gratton, M. M. Bell, and R. D. Jackson. 2011. Management-intensive rotational grazing enhances forage production and quality of subhumid cool-season pastures. Crop Science 51 (2): 892–901.				Y			Y	Y

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Odadi, W. O., J. Fargione, and D. I. Rubenstein. 2017. Vegetation, wildlife, and livestock responses to planned grazing management in an African pastoral landscape. Land Degradation & Development 28: 2030–2038.	Y	Y					Y	
Olson, K. C., R. D. Wiedmeier, J. E. Bowns, and R. L. Hurst. 1999. Livestock response to multispecies and deferred-rotation grazing on forested rangeland. Journal of Range Management 52 (5): 462–470.					Y		Y	
Ortega, I. M., S. Soltero-Gardea, F. C. Bryant, and D. L. Drawe. 1997. Evaluating grazing strategies for cattle: deer forage dynamics. Journal of Range Management 50 (6): 615–621.			Y				Y	
Owensby, C. E., E. F. Smith, and K. L. Anderson. 1973. Deferred-rotation grazing with steers in the Kansas Flint Hills. Journal of Range Management 26 (6):393–395.			Y		Y		Y	
Paine, L. K., and C. A. Ribic. 2002. Comparison of riparian plant communities under four land management systems in southwestern Wisconsin. Agriculture, Ecosystems & Environment 92 (1): 93–105.	Y						Y	
Paine, L. K., D. Undersander, and M. D. Casler. 1999. Pasture growth, production, and quality under rotational and continuous grazing management. Journal of Production Agriculture 12 (4): 569–577.			Y	Y			Y	
Pandey, C. 1998. Effects of soil water and grazing on herbaceous plant cover and cover: biomass relations in a seasonally dry tropical savanna. Tropical Ecology 39: 201–209.	Y						Y	Y
Pavlů, V., M. Hejcman, L. Pavlů, and J. Gaisler. 2003. Effect of rotational and continuous grazing on vegetation of an upland grassland in the Jizerské Hory Mts., Czech Republic. Folia Geobotanica 38: 21–34.	Y						Y	
Phillip, L., P. Goldsmith, M. Bergeron, and P. Peterson. 2001. Optimizing pasture management for cow-calf production: The roles of rotational frequency and stocking rate in the context of system efficiency. Canadian Journal of Animal Science 81: 47–56.			Y		Y	Y	Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Popp, J., W. McCaughey, and R. Cohen. 1997. Grazing system and stocking rate effects on the productivity, botanical composition and soil surface characteristics of alfalfa-grass pastures. Canadian Journal of Animal Science 77: 669–676.			Y	Y	Y	Y	Y	
Pulido, R., and J. Leaver. 2003. Continuous and rotational grazing of dairy cows–the interactions of grazing system with level of milk yield, sward height and concentrate level. Grass and Forage Science 58: 265–275.					Y		Y	
Raffaele, E., and T. T. Veblen. 2001. Effects of cattle grazing on early postfire regeneration of matorral in northwest Patagonia, Argentina. Natural Areas Journal 21: 243–249.	Y							Y
Reardon, P. O., and L. B. Merrill. 1976. Vegetative response under various grazing management systems in the Edwards Plateau of Texas. Journal of Range Management 29 (3): 195–198.			Y				Y	Y
Renfrew, R. B., and C. A. Ribic. 2001. Grassland birds associated with agricultural riparian practices in southwestern Wisconsin. Journal of Range Management 54 (5):546–552.				Y			Y	
Rogler, G. A. 1951. A twenty-five year comparison of continuous and rotation grazing in the Northern Plains. Journal of Range Management 4 (1): 35–41.					Y		Y	
Rosi, M., S. Puig, M. Cona, F. Videla, E. Mendez, and V. Roig. 2009. Diet of a fossorial rodent (Octodontidae), above-ground food availability, and changes related to cattle grazing in the Central Monte (Argentina). Journal of Arid Environments 73 (3): 273–279.		Y						Y
Ruiz-Mirazo, J., and A. B. Robles. 2012. Impact of targeted sheep grazing on herbage and holm oak saplings in a silvopastoral wildfire prevention system in south-eastern Spain. Agroforestry Systems 86 (3): 477–491.	Y	Y		Y				Y
Ruthven, D. 2007. Grazing effects on forb diversity and abundance in a honey mesquite parkland. Journal of Arid Environments 68 (4): 668–677.	Y	Y						Y

Study	*R*	*D*	*B*	*G*	*W*	*P*	*SRG–CG*	*SRG–UG*
Sanford, P., B. Cullen, P. Dowling, D. Chapman, D. Garden, G. Lodge, M. Andrew, P. Quigley, S. Murphy, and W. M. King. 2003. SGS Pasture Theme: effect of climate, soil factors and management on pasture production and stability across the high rainfall zone of southern Australia. Australian Journal of Experimental Agriculture 43 (8): 945–959.			Y				Y	
Sanjari, G., B. Yu, H. Ghadiri, C. A. Ciesiolka, and C. W. Rose. 2010. Effects of time-controlled grazing on runoff and sediment loss. Soil Research 47 (8): 796–808.				Y			Y	
Sanjari, G., H. Ghadiri, and B. Yu. 2016. Effects of time-controlled and continuous grazing on total herbage mass and ground cover. Journal of Agriculture and Rural Development in the Tropics and Subtropics 117 (1): 165–174.			Y	Y			Y	
Saunders, W. C., and K. D. Fausch. 2007. Improved grazing management increases terrestrial invertebrate inputs that feed trout in Wyoming rangeland streams. Transactions of the American Fisheries Society 136: 1216–1230.			Y	Y			Y	
Saunders, W., and K. D. Fausch. 2012. Grazing management influences the subsidy of terrestrial prey to trout in central Rocky Mountain streams (USA). Freshwater Biology 57: 1512–1529.			Y				Y	Y
Savadogo, P., L. Sawadogo, and D. Tiveau. 2007. Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. Agriculture, Ecosystems & Environment 118 (1–4): 80–92.			Y	Y				Y
Scott, D. 2001. Sustainability of New Zealand high-country pastures under contrasting development inputs. 7. Environmental gradients, plant species selection, and diversity. New Zealand Journal of Agricultural Research 44 (1): 59–90.	Y						Y	
Sedivec, K. K., T. A. Messmer, W. T. Barker, K. F. Higgins, and D. R. Hertel. 1990. Nesting success of upland nesting waterfowl and sharp-tailed grouse in specialized grazing systems in southcentral North Dakota.			Y		Y	Y	Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
General technical report RM-Rocky Mountain Forest and Range Experiment Station, US Department of Agriculture, Forest Service (USA). 71–92.								
Seefeldt, S. S., and S. D. McCoy. 2003. Measuring plant diversity in the tall threetip sagebrush steppe: influence of previous grazing management practices. Environmental Management 32 (2): 234–245.	Y	Y						Y
Selemani, I. S., L. O. Eik, Ø. Holand, T. Ådnøy, E. Mtengeti, and D. Mushi. 2013. The effects of a deferred grazing system on rangeland vegetation in a north-western, semi-arid region of Tanzania. African Journal of Range & Forage Science 30 (3): 141–148.	Y		Y	Y			Y	
Smith, R., E. Biddiscombe, and W. Stern. 1973. Effect of spelling newly sown pastures. Australian Journal of Experimental Agriculture 13 (64): 549–555.			Y		Y		Y	
Smoliak, S. 1960. Effects of deferred-rotation and continuous grazing on yearling steer gains and shortgrass prairie vegetation of southeastern Alberta. Journal of Range Management 13 (5): 239–243.					Y		Y	
Southwood, O., and G. Robards. 1975. Lucerne persistence and the productivity of ewes and lambs grazed at two stocking rates within different management systems. Australian Journal of Experimental Agriculture 15 (77): 747–752.					Y		Y	
Stejskalová, M., P. Hejcmanová, V. Pavlů, and M. Hejcman. 2013. Grazing behavior and performance of beef cattle as a function of sward structure and herbage quality under rotational and continuous stocking on species-rich upland pasture. Animal Science Journal 84: 622–629.					Y	Y	Y	
Sternberg, M., M. Gutman, A. Perevolotsky, E. D. Ungar, and J. Kigel. 2000. Vegetation response to grazing management in a Mediterranean herbaceous community: a functional group approach. Journal of Applied Ecology 37 (2): 224–237.	Y						Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Sternberg, Marcelo, Carly Golodets, Mario Gutman, Avi Perevolotsky, Eugene D. Ungar, Jaime Kigel, and Zalmen Henkin. 2015. Testing the limits of resistance: a 19-year study of Mediterranean grassland response to grazing regimes. Global Change Biology. 21 (5): 1939-1950.		Y					Y	
Sun, Y., J. Angerer, and F. Hou. 2015. Effects of grazing systems on herbage mass and liveweight gain of Tibetan sheep in Eastern Qinghai-Tibetan Plateau, China. The Rangeland Journal 37: 181–190.			Y		Y	Y	Y	
Teague, W., and S. Dowhower. 2003. Patch dynamics under rotational and continuous grazing management in large, heterogeneous paddocks. Journal of Arid Environments 53 (2): 211–229.				Y			Y	
Teague, W., S. Dowhower, and J. Waggoner. 2004. Drought and grazing patch dynamics under different grazing management. Journal of Arid Environments 58 (1): 97–117.			Y	Y			Y	
Teague, W., S. Dowhower, S. Baker, N. Haile, P. DeLaune, and D. Conover. 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. Agriculture, Ecosystems & Environment 141: 310–322.			Y	Y			Y	Y
Teague, W., S. Dowhower, S. Baker, R. Ansley, U. Kreuter, D. Conover, and J. Waggoner. 2010. Soil and herbaceous plant responses to summer patch burns under continuous and rotational grazing. Agriculture, Ecosystems & Environment 137: 113–123.				Y			Y	
Thornton, D., and G. Harrington. 1971. The effect of different stocking rates on the weight gain of Ankole steers on natural grassland in western Uganda. The Journal of Agricultural Science 76 (1): 97–106.					Y		Y	
Tothill, J., C. McDonald, and G. McHarg. 2009. Effect of wet season rotational grazing on pasture and animal production in buffel (Cenchrus ciliaris)-siratro (Macroptilum atropurpureum) pastures in south-east Queensland. Tropical Grasslands 43: 162–170.					Y		Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Tupper, G. 1978. Sheep production on a Danthonia caespitosa-Stipa variabilis grassland in response to fertilizers and deferred grazing. Australian Journal of Experimental Agriculture 18 (91): 210–217.			Y		Y		Y	
Vermeire, L. T., R. K. Heitschmidt, and M. R. Haferkamp. 2008. Vegetation response to seven grazing treatments in the Northern Great Plains. Agriculture, Ecosystems & Environment 125: 111–119.			Y				Y	
Volesky, J.D., J.K. Lewis, and C.H. Butterfield. 1990. High-performance short-duration and repeated-seasonal grazing systems: effect on diets and performance of calves and lambs. Rangeland Ecology & Management/Journal of Range Management Archives, 43 (4):310-315.					Y	Y	Y	
Waller, R. A., P. Sale, G. Saul, and G. Kearney. 2001a. Tactical versus continuous stocking in perennial ryegrass-subterranean clover pastures grazed by sheep in south-western Victoria. 1. Stocking rates and herbage production. Australian Journal of Experimental Agriculture 41 (8): 1099–1108.			Y				Y	
Waller, R. A., P. Sale, G. Saul, and G. Kearney. 2001b. Tactical versus continuous stocking in perennial ryegrass-subterranean clover pastures grazed by sheep in south-western Victoria. 3. Herbage nutritive characteristics and animal production. Australian Journal of Experimental Agriculture 41 (8): 1121–1131.					Y	Y	Y	
Wang, C., B. Tas, T. Glindemann, K. Mueller, A. Schiborra, P. Schoenbach, M. Gierus, F. Taube, and A. Susenbeth. 2009. Rotational and continuous grazing of sheep in the Inner Mongolian steppe of China. Journal of Animal Physiology and Animal Nutrition 93: 245–252.			Y		Y		Y	
Wang, T., Z. Zhang, Z. Li, and P. Li. 2017. Grazing management affects plant diversity and soil properties in a temperate steppe in northern China. Catena 158: 141–147.	Y	Y	Y	Y			Y	
Wang, Z., X.-j. Yun, Z.-j. Wei M. Schellenberg P, Y.-f. Wang, X. Yang, and X.-y. Hou. 2014. Responses of Plant Community and Soil Properties to Inter-Annual Precipitation Variability and Grazing Durations in a Desert Steppe Within Inner Mongolia. Journal of Integrative Agriculture 13 (6): 1171-1182.			Y					Y

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Warn, L., and H. Frame. 2002. Effects of grazing method and soil fertility on stocking rate and wool production. Wool Technology and Sheep Breeding 50 (3): 510–517.						Y	Y	
Weltz, M., and M. K. Wood. 1986. Short-duration grazing in central New Mexico: effects on sediment production. Journal of Soil and Water Conservation 41 (4): 262–266.			Y	Y			Y	Y
White, M. R., R. D. Pieper, G. B. Donart, and L. W. Trifaro. 1991. Vegetational response to short-duration and continuous grazing in southcentral New Mexico. Journal of Range Management 44 (4):399–403.			Y	Y			Y	
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.			Y		Y		Y	
Winder, J. A., and R. F. Beck. 1990. Utilization of linear prediction procedures to evaluate animal response to grazing systems. Journal of Range Management, 43 (5): 396–400.					Y		Y	
Wong, N. K., and J. W. Morgan. 2012. Experimental changes in disturbance type do not induce short-term shifts in plant community structure in three semi-arid grasslands of the Victorian Riverine Plain managed for nature conservation. Ecological Management & Restoration 13 (2): 175–182.				Y				Y
Wyatt, W., J. Gillespie, D. Blouin, B. Venuto, R. Boucher, and B. Qushim. 2013. Effects of year-round stocking rate and stocking method systems on cow-calf production in the gulf coast region of the United States: Costs, returns, and labor considerations. The Professional Animal Scientist 29 (1): 16–26.					Y		Y	
Xiuxia, B., Y. Jin, L. Shurun, G. Jimuse, W. Puchang, and L. Yong. 2008. Effects of Different Grazing on the Typical Steppe Vegetation Characteristics on the Mongolian Plateau. Nomadic Peoples 12 (2): 53–66.	Y			Y			Y	

Study	***R***	***D***	***B***	***G***	***W***	***P***	***SRG–CG***	***SRG–UG***
Zhang, Y., D. Huang, W. B. Badgery, D. R. Kemp, W. Chen, X. Wang, and N. Liu. 2015. Reduced grazing pressure delivers production and environmental benefits for the typical steppe of north China. Scientific Reports 5: 16434.						Y	Y	
Zhou, X., J. Wang, Y. Hao, and Y. Wang. 2010. Intermediate grazing intensities by sheep increase soil bacterial diversities in an Inner Mongolian steppe. Biology and Fertility of Soils 46 (8): 817–824.		Y	Y					Y
Zimmer, H. C., V. B. Turner, J. Mavromihalis, J. Dorrough, and C. Moxham. 2010. Forb responses to grazing and rest management in a critically endangered Australian native grassland ecosystem. The Rangeland Journal 32 (2): 187–195.	Y						Y	Y

Appendix 3 Stocking rate differences for SRG–CG contrasts for each response variable (where reported), with SRG as the reference. Total number of contrasts, number in bracket is as a percent of total contrasts

	Biomass	Ground cover	Weight gain	Plant richness	Plant diversity
Greater	24 (14)	11 (15)	30 (16)	12 (22)	2 (11)
Equal	112 (65)	36 (51)	139 (73)	16 (29)	7 (39)
Lower	22 (13)	8 (11)	20 (11)	18 (33)	3 (17)

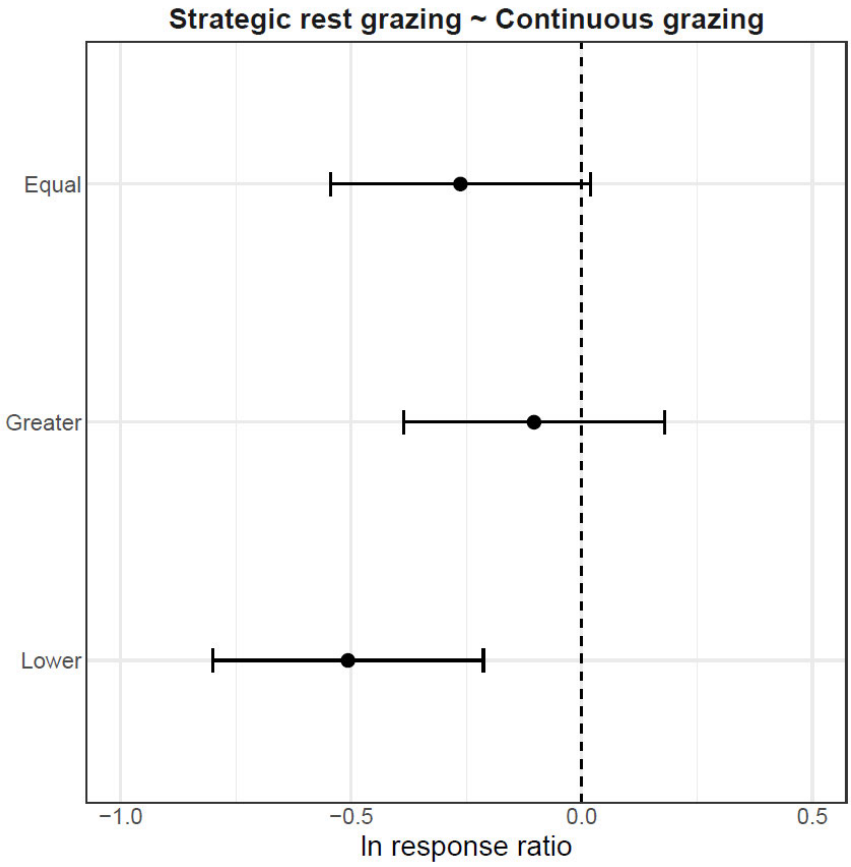
Appendix 4 Proportion of papers with missing data for each response variable dataset. 'n' = the number of studies and the number of contrasts. 'SD' and 'rest-to-graze' = the number of contrasts with incomplete data (with percentages in parentheses) for standard deviations (SD) and rest-to-graze ratios. SRG = strategic-rest grazing, CG = continuous grazing, UG = ungrazed

Response		**Missing data types**		
	n	SD	rest-to-graze	Stocking rate difference
Plant richness				
SRG-CG	19, 55	32 (58%)	6 (11%)	9 (16%)
SRG-UG	13, 27	11 (41%)	8 (30%)	NA
Plant diversity				
SRG-CG	10, 18	4 (22%)	1 (6%)	6 (33%)
SRG-UG	10, 24	4 (17%)	9 (38%)	NA
Biomass				
SRG-CG	66, 173	62 (36%)	30 (18%)	15 (9%)
SRG-UG	21, 63	18 (29%)	26 (41%)	NA
Groundcover				
SRG-CG	31, 71	25 (37%)	22 (33%)	16 (23%)
SRG-UG	13, 35	20 (57%)	8 (23%)	NA
Weight gain				
SRG-CG	79, 190	85 (45%)	2 (1%)	1 (1%)
Animal production per hectare				
SRG-CG	36, 98	43 (44%)	3 (3%)	NA

.

Appendix 5 Results of likelihood ratio test and omnibus tests for best-fit models

Best-fit model parameters	Likelihood-ratio test		Omnibus test	
	χ^2	*P*	Q_M	*P*
Biomass (SRG–CG)				
Stocking rate	149.93	< 0.001	154.38	< 0.001
Rest-to-graze + stocking rate	396.28,	< 0.001	401.47	< 0.001
Biomass (SRG–ungrazed)				
Rest-to-graze	55.34	< 0.001	55.62	< 0.001
Ground cover (SRG-CG)				
Rest-to-graze + stocking rate	10323.94	< 0.001	10337.12	< 0.001
Ground cover (SRG–ungrazed)				
Rest-to-graze	24.15,	< 0.001	24.16,	< 0.001
Weight gain				
Rest-to-graze	83.59	< 0.001	83.65	< 0.001
Animal production per hectare				
Rest-to-graze	352.24	< 0.001	352.24	< 0.001
Richness (SRG–CG)				
Stocking rate	103.00,	< 0.001	103.53	< 0.001
Rest-to-graze + stocking rate	121.48	< 0.001	121.87	< 0.001
Richness (SRG–ungrazed)				
Climate	5.27	0.072	8.74	0.013
Rest-to-graze	5.77	0.016	5.83	0.016



Appendix 6 Effect size ± 95% confidence intervals of biomass null models with stocking rate differences, comparing strategic-rest grazing relative to continuous grazing.

Appendix 7 Additional method information

Variance computation

A significant proportion of publications reviewed did not provide measures of variance (Appendix 4). In some cases, composite means were estimated (i.e., yearly-means were averaged within a study). In these cases, standard deviations were calculated along with these averages. The sampling variances were then estimated using each study's overall sample size as opposed to the number of years. Arguably, this may introduce bias. However, to designate these studies with a reduced sample size would ultimately exhibit a greater bias, as it would lessen the influence of these studies, which in some cases had large sample sizes, at the

expense of incomplete data-reporting. Where available, we estimated variances using known transformations for other reported statistics (i.e., *T*-values and interquartile ranges). Where standard deviations could not be derived from reported data, we imputed missing standard deviations within each contrast (SRG–CG, SRG–ungrazed) within each dataset using the co-efficient of variation (CV), or the mean to SD ratio (Bracken, 1992; Koricheva et al., 2013). The CV is calculated from all complete cases (i.e., within a dataset and grazing type). Missing standard deviations are then imputed using this pooled CV in conjunction with the sample sizes from the incomplete datasets. Imputation was undertaken using the impute SD function within metagear (v.0.4) package (Lajeunesse, 2016).

Publication bias

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. 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 (Egger et al., 1997; Habeck and Schultz, 2015). Asymmetry between effect size and sampling variance suggesting publication bias was observed in the SRG–ungrazed comparison for plant richness ($p = 0.082$), the SRG–CG comparison for biomass ($p = 0.006$) and the SRG–CG comparison for groundcover ($p = 0.011$).

Appendix 8 Details for each property about the overall approach to pasture sowing and for individual sites on each property, whether application of inorganic phosphorus-based fertilisers is calendar based, the approximate time since last cultivation for each site and whether sites had been cultivated historically. Where data was available, stocking rates (DSE ha^{-1}) are given. Average stocking rates for SDG properties were 8.7 (granite-derived soils) and 10.7 (basalt-derived soils). For RP properties average stocking rates were 7.5 (granite-derived soils) and 10.5 (basalt-derived soils). Stocking rates for two of the RP sites were either unavailable or seemed unreliable and were not included. Individual sites on each property are referred to as i, ii or iii. NA means the site had never been cultivated.

Management	Property pair	Regular pasture sowing on property (Y/N)	Current calendar-based fertiliser application and/or top-dressing	Last cultivation event (approximate number of years ago)	Cultivated historically (i.e. > 25 years ago) (Y/N)	Stocking rate in a year of average rainfall (DSE/ha)
SDG[1]	1	N	N	> 25 (all three sites	Y (all three sites)	7
	2	N	N	8, (all three sites)	Y (all three sites)	8
	3	N	N	17 (all three sites)	Y (all three sites)	14
	4	Y	N	15 (i), 15 (ii), 25 (iii)	Y (all three sites)	10
	5	N	N	12 (all three sites)	Y (all three sites)	11
	6	N	Y*	8 (all sites)	Y (all three sites)	8
RP[2]	1	N	Y	> 25 (all three sites)	Y (all three sites)	5
	2	Y	Y	NA (all three sites)	N (all three sites)	10
	3	Y	Y	15 (i), > 25 (ii), > 25 (iii)	Y (all three sites)	11
	4	Y	Y	15 (i), 15 (ii), NA (iii)	Y (i), Y (ii), N (iii)	NA
	5	Y	Y	12 (i), 6 (ii), 3 (iii)	Y (all three sites)	NA
	6	Y	Y	7 (i), NA (ii), NA (iii)	Y (i), N (ii), N (iii)	10

* application of poultry manure

Appendix 9 List of plant species with presence according to management approach, underlying geological substrate and functional groups each species was allocated to. Perennial herbaceous, introduced annual plants and native C3 and C4 perennial grasses were determined following Harden (2000–2002) Jacobs et al. (2008) and Simon & Alfonso, (2011). The introduced annual group includes short-lived perennial species. The introduced C3 perennial pasture plants includes preferred forage plants that have been introduced into the region of study previously (Lodge and Whalley, 1989; Shakhane et al., 2013). Increaser species refer to perennial species known to increase under heavy grazing pressure (Oudtsdoorn, 1992; Jacobs and Harden, 2008; Kioko et al., 2012). Introduced species are denoted by an asterix.

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
Family - Poaceae										
Tribe – Andropogoneae										
Bothriochloa biloba	y	y		y	y				y	
Bothriochloa decipiens	y	y		y	y				y	
Bothriochloa macra	y	y	y	y	y				y	y
Cymbopogon refractus	y	y		y	y				y	
Dicanthium sericeum	y	y	y	y	y				y	
Eulalia aurea	y		y		y				y	
Sorghum leiocladum	y	y	y	y	y				y	
Themeda avenacea	y			y	y				y	
Themeda triandra	y		y		y				y	
Tribe – Aristideae										
Aristida leptopoda	y		y	y	y				y	
Aristida ramosa	y		y	y	y				y	
Tribe – Cynodonteae										
Chloris truncata	y	y	y		y				y	
Cynodon dactylon	y	y	y	y	y				y	y
Tribe - Danthonieae										
Rytidosperma bipartita	y	y	y	y	y			y		
Rytidosperma caespitosum			y		y			y		
Rytidosperma pilosa	y	y	y		y			y		
Rytidosperma racemosa	y	y	y	y	y			y		
Tribe - Eragrostideae										
Eragrostis brownii	y	y	y		y				y	
Eragrostis leptostachya	y	y	y	y	y				y	
Eragrostis trachycarpa	y	y	y	y	y				y	
Tribe - Erharteae										
Microlaena stipoides	y	y	y	y	y			y		

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
Tribe – Paniceae										
Digitaria brownii	y		y		y				y	
Digitaria divaricatissima	y	y		y	y				y	
Eriochloa procera	y			y	y				y	
Panicum decompositum		y		y	y				y	
Panicum effusum	y	y	y		y				y	
Cenchrus purpurascens	y	y		y	y				y	
Tribe - Poeae										
Dichelachne micrantha	y	y	y	y	y			y		
Poa annua	y		y			y				
Poa sieberiana	y	y	y	y	y			y		
Tribe - Stipeae										
Austrostipa aristiglumis	y			y	y			y		
Austrostipa scabra		y	y	y	y			y		
Austrostipa verticillata	y	y		y	y			y		
Tribe – Triticeae										
Anthosachne sp.	y	y		y	y			y		
Tribe - Zoysieae										
Sporobolus creber	y		y	y	y				y	
Tribe - Bromeae										
**Bromus brevis*	y	y	y	y		y				
**Bromus catharticus*	y	y	y	y		y				
Tribe – Cynodonteae										
**Eleusine tristachya*	y	y	y	y	y					
Tribe - Eragrostideae										
**Eragrostis cilianensis*		y	y	y		y				
**Eragrostis lugens*	y		y		y					

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
Tribe – Paniceae										
**Digitaria sanguinalis*		y	y	y		y				
**Panicum gilvum*	y	y	y	y		y				
**Setaria parviflora*	y	y	y	y	y					
**Setaria pumila*		y	y	y		y				
Tribe – Paspaleae										
**Paspalum dilatatum*	y	y	y	y	y					
Tribe - Poeae										
**Phalaris aquatica*	y	y	y	y	y		y			
**Anthoxanthum odoratum*	y	y	y		y					
**Avena fatua*	y		y			y				
**Briza minor*	y		y			y				
**Phleum pratense*	y			y	y					
**Holcus lanatus*	y	y	y	y	y					
**Festuca arundinacea*	y	y	y	y	y		y			
**Vulpia bromoides*	y		y	y		y				
**Lolium sp.*	y	y	y	y	y		y			
**Dactylis glomerata*	y	y	y	y	y		y			
Tribe - Triticeae										
**Hordeum leporinum*			y							
Family – Anthericaceae										
Tricoryne elatior	y	y	y		y					
Family – Asteraceae										
Ammobium alatum	y		y	y	y					
Calotis cuneifolia	y		y		y					
Calotis lappulacea	y			y	y					
Chrysocephalum apiculatum	y	y	y		y					
Craspedia variabilis	y			y	y					
Cymbonotus lawsonianus	y		y		y					
Euchiton involucratus	y	y	y	y	y					
Leptorhynchos squamatus	y		y		y					
Vittadinia cuneata	y	y		y	y					
Family – Fabaceae										
Desmodium sp.	y		y	y	y					
Glycine sp.	y	y	y	y	y					
Lespedeza juncea	y		y	y	y					

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
Family – Cyperaceae										
Carex inversa	y	y	y	y	y					
Cyperus fulvus		y		y	y			y		
Cyperus gracilis		y		y	y			y		
Fimbristylis dichotoma	y	y	y	y	y			y		
Juncus subsecundus	y	y	y	y	y					
Family – Campanulaceae										
Wahlenbergia communis	y		y	y	y					
Wahlenbergia gracilenta	y	y	y	y	y					
Family – Chenopodiaceae										
Einadia nutans	y	y	y	y	y					
Chenopodium pumilio	y	y	y	y		y				
Family – Commelinaceae										
Murdannia graminea	y	y	y		y					
Family – Convovulaceae										
Convolvulus erubescens	y	y	y	y	y					
Dichondra repens	y	y		y	y					
Family – Euphorbiaceae										
Chamaesyce drummondii	y	y	y		y					
Family – Geraniaceae										
Geranium solanderi	y	y	y	y	y					
Family – Lamiaceae										
Mentha satureioides	y	y	y	y	y					
Family – Lomandraceae										
Lomandra multiflora	y		y		y					
Family - Malvaceae										
Hibiscus trionum		y		y		y				
Family – Nyctaginaceae										
Boerhavia dominii	y			y	y					
Family – Onagraceae										
Epilobium billardierianum		y		y	y					

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
Family – Oxalidaceae										
Oxalis perennans	y	y	y	y						
Family – Plantaginaceae										
Plantago varia	y	y		y	y					
Family – Polygonaceae										
Rumex brownii	y	y	y	y	y					
Family - Portulacaceae										
Portulaca oleracea	y	y	y	y		y				
Family – Rubiaceae										
Asperula conferta		y	y	y	y					
Family – Rosaceae										
Acaena agnipila	y	y	y	y	y					
Family – Urticaceae										
Urtica incisa		y		y	y					
Family – Zygophyllaceae										
Tribulus micrococcus		y		y	y					
Family – Pteridaceae										
Cheilanthes sp.	y	y	y	y	y					
Family - Amaranthaceae										
**Amaranthus powellii*	y	y	y	y	y					
Family – Apiaceae										
**Cyclospermum leptophyllum*	y	y		y		y				
Family – Araliaceae										
**Hydrocotyle bonariensis*		y		y		y				
Family – Asteraceae										
**Bidens pilosa*	y	y	y	y		y				
**Carthamus lanatus*	y	y		y		y				
**Centaurea solstitialis*	y	y		y		y				
**Cichorium intybus*	y	y	y	y	y		y			
**Cirsium vulgare*	y	y	y	y		y				

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
**Conyza sp.*	y	y	y	y		y				
**Gamochaeta purpurea*	y	y	y	y		y				
**Hypochaeris glabra*	y	y	y	y		y				
**Hypochaeris radicata*	y	y	y	y	y					
**Lactuca serriola*	y	y		y		y				
**Leucanthemum vulgare*	y			y	y					
**Schkuhria pinnata*	y		y	y	y					
**Sonchus oleraceus*	y		y	y		y				
**Taraxacum officinale*	y		y	y						
Family – Fabaceae										
**Medicago lupulina*	y			y	y		y			
**Medicago polymorpha var. vulgaris*	y	y	y		y					
**Medicago sativa*	y	y		y	y		y			
**Trifolium arvense*	y		y	y		y				
**Trifolium campestre*				y		y				
**Trifolium fragiferum*	y	y		y	y		y			
**Trifolium pratense*	y			y	y		y			
**Trifolium repens*	y	y	y	y	y		y			
**Trifolium subterraneum*	y	y	y		y		y			
Family – Caryophyllaceae										
**Cerastium glomeratum*	y			y		y				
**Dianthus armeria*	y	y	y	y		y				
**Paronychia brasiliana*	y	y	y	y	y					
Family – Brassicaceae										
**Lepidium bonariense*	y	y	y	y		y				
**Sisymbrium officinale*	y		y	y		y				
Family – Plantaginaceae										
**Plantago lanceolata*	y	y	y	y	y		y			
Family – Rubiaceae										
**Richardia stellaris*		y	y		y					
Family – Gentianaceae										
**Centaurium erythraea*	y	y		y		y				
Family – Malvaceae										
**Malva neglecta*	y	y	y	y		y				
**Modiola caroliniana*	y	y	y	y		y				

Plant species	Short-duration grazing	Regional Practice	Granite-derived soils	Basalt-derived soils	Perennial plants	Introduced annual plants	Introduced C3 perennial pasture plants	Native C3 perennial grasses	Native C4 perennial grasses	Increaser species
Family – Polygonaceae										
**Acetosella vulgaris*	y	y	y	y	y					
Family – Scrophulariaceae										
**Verbascum thapsus*	y	y	y	y		y				
Family – Solanaceae										
**Solanum nigrum*	y	y	y			y				
Family – Verbenaceae										
**Verbena bonariensis*	y	y		y	y					

Appendix 10 The three Landscape Function Analysis (LFA) integrative indices of soil stability, infiltration and nutrient cycling with 11 soil surface assessment indicators (SSI) adjacent to descriptions of related soil processes. Blocks denote SSI's that are integrated into each index (after Tongway and Hindley, 2005 and Read et al. 2016).

Indices			Soil surface assessment indicators	Soil processes
Stability	Infiltration	Nutrient cycling		
■			Perennial vegetation canopy and rock cover	Protect the soil and reduces compacting and erosional force of raindrop impact on the soil surface
	■	■	Perennial vegetation basal (grass) or canopy (trees and shrubs) area	Surrogate for root biomass and contribution of below ground biomass to soil processes
■	■	■	Litter cover, depth, origin and decomposition	Strongly related to carbon, nitrogen and other minerals in the surface soil layer
■		■	Cryptograms (biological crust cover)	Stabilise the soil surface and indicate plant available nutrients in the surface soil layer
■			Crust brokenness	Broken, brittle crusts are unstable and prone to erosion. Smooth crusts are less vulnerable
■			Erosion type and severity	Accelerated erosion caused by the interaction of management and climate
■			Deposited materials	Alluvium, litter and other material transported provide resources and may form productive alluvial fans
	■	■	Soil surface roughness (microtopography)	Ease of soil surface disturbance relates to resistance to erosion and conversely water capture or run-off
■	■		Surface resistance to disturbance	Ease of soil surface disturbance relates to resistance to erosion and conversely water capture or run-off
■	■		Slake test	Assesses the stability of the soil aggregates and their water erosion potential
	■		Soil texture	Relates to infiltration and water holding potential of soil with different textural characteristics

Appendix 11. Published journal article (Agriculture, Ecosystems and Environment, 2019, volume 281, pages 134–144). Chapter 4 is slightly altered from the published journal article based on PhD thesis reviewer comments.

Title of Article:	Short-duration rotational grazing leads to improvements in landscape functionality and increased perennial herbaceous plant cover
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Contributions:	Rachel Lawrence - 85%, R.D.B Whalley - 5%, Nick Reid - 5%, Romina Rader - 5%
Signed:	

Rachel Lawrence [redacted]		Romina Rader [redacted]
Candidate		Principal Supervisor

Short-duration rotational grazing leads to improvements in landscape functionality and increased perennial herbaceous plant cover

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Abstract

Livestock grazing can lead to reduced groundcover and altered composition of pastures through the loss of palatable forage species and reduced litter cover. This negatively impacts landscape function and ultimately livestock production. Grazing livestock for short periods with high animal density, followed by long rests to allow pasture recovery (short-duration grazing), could be a way to address these issues. In naturalised pastures, we assessed landscape functioning and compared the abundance of six major plant functional groups at 36 sites on 12 commercial grazing properties. Six of the properties had been managed with short-duration grazing for more than 7 years (in most cases over 10 years), while the six control properties were managed with grazing that was more typical of the region (relatively continuous throughout the year with unplanned rests). Under short-duration grazing, there was approximately 19% greater foliar cover of perennial herbaceous species with a corresponding 14% reduction in foliar cover of introduced annual plants. Attributes relating to biophysical functioning of the landscape were enhanced by short-duration grazing, with environmental factors less important in influencing these landscape function attributes. Higher-value forage species were also more abundant on short-duration grazing properties, especially at higher rainfall sites. Conversely, species that tend to increase under heavy grazing pressures, and are of lower forage value, were less abundant under short-duration grazing. Despite the changes in pasture composition in response to grazing management there was a large amount of unexplained variation in herbaceous community composition. This study demonstrates benefits for landscape function and naturalised pasture composition under short-duration grazing that has been in place for several years compared with more usual grazing practices.

Additional keywords: grassland composition, ground cover, litter, landscape function analysis, naturalised pastures, planned rest, increaser species

1. Introduction

Livestock grazing occupies approximately 26% of the world's land area (Asner et al., 2004) and can lead to landscapes of high diversity and function when managed appropriately (Watkinson and Ormerod, 2001; IUCN, 2008). However, livestock grazing can also be a major contributor to soil degradation processes and losses of biodiversity where stocking rates are inappropriately high (Steinfield et al., 2006). Globally, there are generalised responses of plant communities to livestock grazing, with the foremost being the loss of perennial herbaceous species (Díaz et al., 2007). For example, grazing promotes replacement of taller species by shorter plants with prostrate species becoming more common under high stocking rates and continuous grazing of areas (Dyksterhuis, 1949; Lodge and Whalley, 1989; Teague et al., 2011). Prolonged, inappropriately high stocking rates result in plant communities dominated by annual, ruderal species or bare ground (Dyksterhuis, 1949; Arnold, 1955; Ellison, 1960; Grigulis et al., 2001; Díaz et al., 2007; Eldridge et al., 2016).

From a livestock production perspective, an important role of perennial herbaceous species is forage provision (Kemp and Dowling, 2000). In combination with plant litter in interstitial spaces between tussocks, perennial plants also slow the movement of water downslope, facilitate water infiltration and reduce evaporation from the soil surface (Thurow, 1991; Yates and Hobbs, 2000; Roth, 2004; Tongway and Hindley, 2005) thereby protecting soils and promoting plant growth (Lang, 1979; Tongway and Hindley, 2005). Perennial tussock species and interstitial litter are also important for building organic matter in the soil, providing a habitat for above and below-ground invertebrates that are important for soil health (Clarholm, 1985; Wardle et al., 2004; King and Hutchinson, 2007) and that support

biodiversity by providing a food source for higher trophic levels (Recher and Lim, 1990). Loss of perennial cover is also a major cause of soil structural decline (Greenwood and McKenzie, 2001). Thus, livestock grazing practices that lead to substantial reductions in herbaceous perennial plants and litter cover result in losses of multiple important ecological functions as well as a reduction in forage availability for livestock.

Heavy stocking pressures, rather than grazing per se, is considered a primary driver of these degrading processes (Ellison, 1960; Gammon, 1978; Thurow, 1991; Greenwood and McKenzie, 2001; Teague et al., 2013). In large paddocks where stock forage continuously, heterogeneous grazing patterns are common with favoured plants and areas grazed selectively and more frequently. This leads to changes in grassland composition and soil degradation in some areas and underuse in other areas (Willms et al., 1988; Thurow, 1991; O' Connor, 1992; Bailey et al., 1996). These heterogeneous patterns in response to grazing become more apparent as paddock size increases (Senft et al., 1985; Stuth, 1991; Bailey et al., 1996).

Modern sedentary grazing practices restrict animal movements and typically have short non-grazing intervals. This style of management is common practice in commercial grazing systems in many rangeland ecosystems (Earl and Jones, 1996; Dorrough et al., 2004; Teague et al., 2011; Scott et al., 2013; Shakhane et al., 2013). An alternative to sedentary or continuous grazing practices is rotational grazing management that incorporates short graze periods with large mobs of livestock followed by extended periods of rest. In these systems, the time when the animals are returned is governed by the needs of the pastures rather than the animals - livestock are grazed for a short time and returned to pastures only after a monitored rest based on the recovery rates of key perennial pasture species. This style of management is described in McCosker (2000); Barnes et al. (2008); Scott et al. (2013) and Steffens et al. (2013) with early influences including Voison (1959); Acocks (1966) and

Savory and Parsons (1980). In this study this style of management is referred to as short-duration grazing (hereafter SDG).

Grazing practices that aim to graze and rest pastures at key times can influence grassland species composition and overall production (Jones, 1933; Lodge and Whalley, 1985; Hidalgo and Cauhépé, 1991; Earl and Jones, 1996; Ash et al., 2011; Steffens et al., 2013; Zhang et al., 2015) and tactical rest is considered an important aspect of good grassland management (Cook et al., 1978; Kemp and Dowling, 2000; Behrendt et al., 2013). A large body of research has been conducted examining the impacts of varying rotational grazing regimes that strategically graze and rest pastures, compared with more continuous grazing practices on pasture composition and productivity (for reviews see Gammon (1978); Holechek et al. (2000); Briske et al. (2008)). While it is acknowledged that this research struggles to account for complex human and ecological factors (Briske et al., 2011; Provenza et al., 2013; Teague et al., 2013), general conclusions are that there are few advantages of rotational over continuous grazing practices with stocking rates being the most important driver impacting vegetation and pasture productivity (Gammon, 1978; Briske et al., 2008).

Regardless of these conclusions, many practitioners and advocates maintain that various forms of SDG result in improved animal production and pasture sustainability compared with more usual styles of management where grazing is relatively continuous throughout the year and rest is unplanned (Joyce, 2000; Mason and Kay, 2000; McCosker, 2000; Sparke, 2000; Norton and Reid, 2013). There is also a significant body of literature discussing reasons for the discrepancies between researcher-based systems and larger-scale studies that consider the complexity inherent in these systems (Norton, 1998; Briske et al., 2011; Barnes and Hild, 2013; Norton et al., 2013; Provenza et al., 2013; Steffens et al., 2013; Teague et al., 2013). Despite this extensive body of research, there is a surprising lack of research investigating

outcomes on commercially managed properties, particularly over longer timeframes (Barnes et al., 2008; Barnes and Hild, 2013; Norton et al., 2013; Provenza et al., 2013). Many differences exist between researcher controlled and commercial operations. Two important ones are that grazing units under continuous grazing are typically much larger than in research situations (Barnes et al., 2008; Norton et al., 2013; Teague et al., 2013) and that pasture changes are a response to complex and longer-term factors where managers have had to adapt to a complex array of influences including climatic, economic, social and political factors (Provenza et al., 2013). Therefore, while researcher-controlled, reductionist systems are valuable for understanding processes relating to soils, water, plants and herbivores (and their interactions), they are arguably less appropriate for understanding property or landscape-scale change (Teague et al., 2013). Working on commercial grazing properties that have been managed with SDG for a number of years provides an opportunity to circumvent the limitations of reductionist research frameworks.

We compared naturalised pastures on commercial properties managed with SDG with pastures on nearby properties managed in ways more typical of regional practice. We tested whether: (1) there would be changes in the abundance of herbaceous perennial plants and other major plant functional groups, and (2) the amount of overall ground cover and landscape function attributes would differ between SDG sites and RP sites. We hypothesised that there would be favourable outcomes for all factors under SDG management.

2. Methods

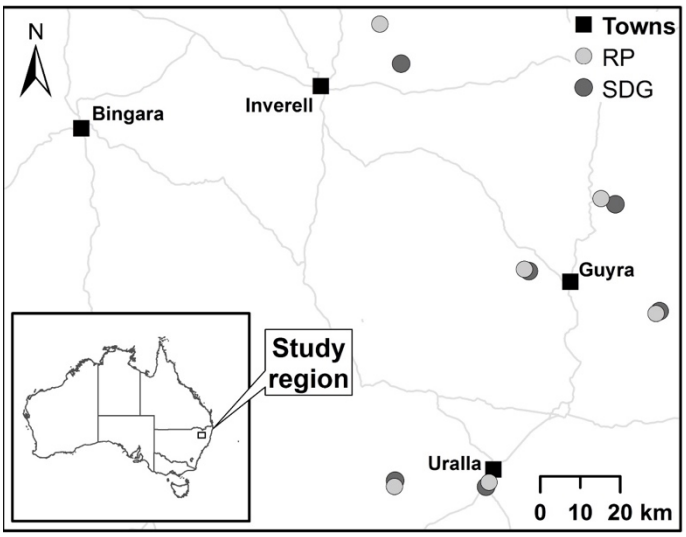

Figure 1 Locat on of the study reg on show ng major towns (b ack squares) and the ocat on of the 12 study propert es. Propert es that mp emented graz ng management more typ ca of reg ona pract ce are ght grey c rc es; SDG propert es are dark grey c rc es

2.1 Location

The study was based on the Northern Tablelands and Slopes of New South Wales, Australia, between 29° 38′ S, 15° 46′ E and 30° 39′ S, 150° 27′ E (Figure. 1). Altitudes ranged from almost 700 m a.s.l. to just below 1400 m a.s.l. The climate of the region is classified as sub-humid with summer-dominant rainfall. Winter months are dominated by cool dry air from the continental interior or from the Southern Ocean. Severe frosts can occur in winter and there is a rainfall and altitudinal gradient across the region, with generally greater precipitation and lower evapotranspiration at higher elevations in the east (Lea et al., 1977; Lodge and Whalley, 1989). The geology of the region is complex and largely determines soil characteristics (Lea et al., 1977; Lodge and Whalley, 1989). Tertiary basalts are mostly found at higher elevations and are underlain by late Permian granites intruded into Palaeozoic sediments. Accordingly, there are three major soil types in the region, fertile basaltic soils, infertile coarse to fine granitic soils and relatively infertile soils that are derived from metamorphosed sediments known locally as ‘trap’ soils.

2.2 Site selection

We selected naturalised pasture sites (terminology from Allen et al. (2011) on 12 commercial grazing properties. Six of the properties were managed with SDG and six with what we defined as usual regional practice (hereafter RP). Five of the six paired sites were at higher tableland sites, with one property pair located on the western slopes of the study area. Details about management practices were determined through initial contact with managers, and then later with a formal questionnaire. On SDG properties, stock grazed intensively at high stocking densities for just a few days, in some cases 1–2 days, with pastures then rested until key forage species and pasture herbage mass had recovered. The decision to return stock was based on pasture monitoring with the length of rest varying with growing conditions. This planned rest resulted in overall rest-to-graze ratios (amount of time rested compared to amount of time grazed) of between 13:1 and 60:1 (median = 27:1). On individual properties this ratio varied at different times depending on climatic conditions. Some SDG managers implemented very short graze periods (1–2 days) while others grazed for several days. Smaller paddock sizes are necessary to implement the large rest-to-graze ratios typical of the SDG approach to management with paddock sizes on SDG properties between 2–16 ha (median = 10 ha) in this study. These smaller paddock sizes necessitate increased density of fencing and water infrastructure (Savory and Parsons, 1980; Sherren et al., 2012). Managers implementing SDG minimised the use of external inputs including minimising hand-feeding of stock during colder winter months and extended dry periods and limited application of inorganic phosphorus-based fertilisers (only one of the six SDG properties had applied superphosphate in the past 5–10 years as opposed to all six of the RP properties). One of the SDG properties applied poultry manure. On the properties examined, SDG management had been in place for between 7 and 25 years (median = 15 years). The contrasting RP management approach was more typical of grazing management on naturalised pastures in

the region. This was mainly continuous, with some rotation of stock where forage was limiting, but with relatively long graze and short rest periods in response to forage availability (Scott et al., 2013; Shakhane et al., 2013). In this study, rest periods on RP properties had rest-to-graze ratios that ranged from 0 to 3:1 (median = 0.5:1) and paddocks were large (median 20 ha, range 12–272 ha) in comparison to SDG paddocks. Superphosphate application on RP properties was typically on a calendar rotation (either annual or biennial) and stock were hand-fed to a greater extent than on SDG properties. The RP properties had been managed in the same way at least for the past 10 years and were judged to be well-managed by local standards.

Few pastoral managers had implemented SDG management in the region for greater than 10 years. Consequently, local networks were used to locate suitable SDG properties, with each property paired with a suitable neighbouring or nearby RP property. While it is acknowledged that this was not a random selection of participants, given the constraint of a low number of properties managed with SDG for an adequate timeframe it was the only practical way of selecting sites that respected neighbour relations and controlled for local soil and climatic conditions.

Livestock type were predominantly cattle (*Bos taurus*), with the exception of four properties (three SDG and one RP) that also ran sheep (*Ovis aries*) for fine-wool production. The study design and low number of suitable SDG properties precluded the two livestock types from being treated separately in the study.

Due to varying levels of engagement with producers, only coarse information on stocking rates could be obtained from the formal survey undertaken. Stocking rates are listed in Appendix 8. One of the criteria for the initial selection of properties for the study was that they were commercial enterprises. Therefore, while the stocking rate data was relatively

coarse, we consider that as grazing management is a trade-off between maximising yield and minimising the risk of degradation (Jakoby et al., 2015) it was a reasonable assumption that managers stock according to their profitability goals balanced with the need to sustain their resource-base.

2.3 Collection of plant data

We selected three open pasture sites with similar soil type, aspect and slope to the corresponding paired property on each of the six pairs of properties. Sites were at least 500 m apart ensuring that on SDG properties sites were representative of pastures at differing stages of the grazing rotation. One quadrat of 30 m^2 (5 × 6 m) representative of the surrounding pasture was surveyed for all vascular plant species in 2 years (n = 36, 2015; n = 35, 2016). We conducted surveys in late summer to early autumn when pasture species were easiest to identify (Reseigh, 2004; Shakhane et al., 2013). Species incidence was recorded, percentage of foliar cover estimated for all species in each quadrat and plant and litter cover visually estimated and combined to give overall cover.

Each site was broadly categorised according to the amount of herbage mass retained in pastures with categories ranging from very low (>300 kg ha^{-1}) to high >2300 kg ha^{-1}. Estimations were made using a modified photo-quadrat technique after Morgan et al. (2018). For the estimations, six 50 x 50 cm sub-quadrats within nineteen 5 x 6 m quadrats of differing biomass amounts were cut and oven-dried at 40°C for 48 hrs. These calibration samples were then weighed and average dry matter for each 5 x 6 m calibration quadrat calculated. Photographs of each plant survey plot were then compared to photographs of the biomass calibration plots and plant survey plots then allocated to one of six categories of pasture biomass. While greater accuracy could have been obtained using alternative methods

that quantified rather than broadly categorised biomass, this method was considered appropriate in the context of the study.

Plants were identified to species level either in the field or by collecting specimens for keying out. Nomenclature for all seed plants other than Poaceae followed Harden (2000–2002). For Poaceae it followed Ausgrass 2 (Simon and Alfonso, 2011) and Grasses of NSW (Jacobs et al., 2008). Plants were allocated to six functional groups to compare their relative abundance between the two commercial grazing regimes. These groups were: (1) all perennial species; (2) introduced annual and short-lived perennial species; (3) introduced C3 perennial pasture plants that historically had been brought into the region as forage and are considered desirable introduced perennial plants in a pasture mix; (4) native C3 perennial grasses that are considered good forage especially during the winter feed-gap (5) native C4 perennial grasses, and (6) two native perennial grasses, *Bothriochloa macra* (Steud.)S.T. Blake and *Cynodon dactylon* (L.) Pers., which, while being valuable ground cover, are tolerant of heavy grazing and increase in abundance under high grazing pressure (Cook et al., 1978; Oudtsdoorn, 1992). The introduced annual or short-lived perennial grass, *Eleusine tristachya* (Lam.) Lam., was also analysed separately as it was present in high abundance compared with other introduced annual species. Leguminous species were too rare across both management categories to be analysed as a separate functional group. A list of plant species and their functional group allocations is presented as supplementary material (Appendix 9).

The introduced C3 perennials included in the analysis were: the grasses *Dactylis glomerata* L., *Lolium sp.*, *Festuca arundinacea* Schreb., *Phalaris aquatica* L.; the legumes, *Trifolium fragiferum* L., *T. pratense* L. and *T. repens* L. and the forbs, *Plantago lanceolata* L. and *Cichorium intybus* L.. While the majority of these plants had been deliberately

introduced into the region for livestock production, they had not been sown into pastures in this study for at least the past 10 years, and in most instances much longer. Appendix 8 details current, past and historical fertiliser application and past and historical cultivation of sites.

2.4 Landscape function analysis

We performed landscape function analysis (LFA; Tongway and Hindley, 2005) to assess whether there would be a difference between management approaches in the capacity of pastures to retain soil/sediment, water and cycle nutrients in the ground-layer. Landscape function analysis assesses the potential of the soil surface to support the growth of plants and is useful for monitoring the restoration of degraded sites over time. It also enables an assessment across sites at one point in time (Read et al., 2016). We assessed eleven soil-surface indicators (Appendix 10) and analysed these to produce two indices (on a scale of 0–100) that related to the likelihood of soil/sediment (stability) and water (infiltration) being retained in the landscape rather than lost as run-off. A third index (nutrient cycling) related to the potential for effective cycling of nutrients and corresponding decomposition and mineralisation processes at and near the soil surface.

We assessed a 20-m transect in the same location as the plant quadrat for landscape function. We divided each transect into patches and inter-patches and performed a landscape organisation assessment (Tongway and Hindley, 2005). We defined a patch type as an area of pasture with increased perennial grass and litter and decreased bare ground compared to surrounding (inter-patch) areas. Patches are more likely than inter-patches to retain soil, water and nutrients in situ (Tongway and Hindley, 2005). The landscape organisation assessment determined the extent that each patch-type contributed to the overall landscape, and therefore its contribution to the landscape function assessment. For each patch-type, five

replicates per transect were assessed for each SSI. These data were analysed to give quantitative values of (1) stability, (2) infiltration and (3) nutrient cycling (Tongway and Hindley, 2005).

2.5 Soil analyses

We assessed each site for soil nutrient status. At each site, and for each LFA patch-type, three 5-cm diameter soil cores were collected per patch type to a depth of 5 cm. We combined these samples for analysis. Where there were two patch-types per transect, the results of the analysis were weighted according to the landscape organisation assessment (i.e., the proportion of each patch-type in the landscape). Soil samples were kept cool and returned to the laboratory for oven-drying at 40ºC for 7 days until completely dry. They were then sealed and stored in a cool, dark place for later analysis.

Soil chemical analyses determined the nutrient levels in different patch-types at each site using methods from Rayment and Higginson (1992). We ground bulked samples to 2 mm following removal of large organic matter. We determined electrical conductivity (EC) and pH in a 1:5 soil to water extract, measured exchangeable cation concentrations (Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}) using atomic absorption spectroscopy and 1-M NH_4Cl, and determined available phosphorus with Colwell P Malachite Green in 0.5-M $NaHCO_3$. Samples measuring P were passed through charcoal to remove organic matter and absorbance in a spectrophotometer measured at 630 nm. We ground subsamples of the bulked soil samples to pass through a 0.5 mm sieve and analysed a homogeneous subsample of this (0.2 g) for total C and N content by combusting at 950°C using a LECO TruSpec Simultaneous Carbon and Nitrogen Elemental Analyser.

2.6 Statistical analyses

Beta-distributed generalised linear mixed models (GLMMs) were used to analyse the abundance of overall plant foliar and litter cover, the foliar cover of the six functional groups and of E. tristachya and the three LFA indices. The beta-distribution is appropriate for proportional data as it accommodates heteroscedasticity (Cribari-Neto and Zeileis, 2010). The two categories of management and the rest to graze ratio (continuous) were treated as fixed effects in the models. Other fixed effects were (1) soil-type (categorical, basalt or granite-derived); (2) year of data collection (categorical, 2015 and 2016); (3) mean annual rainfall at each site as determined from rainfall records for each property or from the nearest Bureau of Meteorology Weather Station (BOM, 2018) (continuous); (4) recent rainfall at each site (number of rainfall events □ 10 mm) for the preceding month (continuous); (5) the 5-months prior to sampling (continuous), and (6) nutrient levels in the top 5-cm of soil. Correlations of different factors were determined in R (R Core Team, 2018). Where factors were determined to be correlated with each other, only one was included to avoid over-fitting models. Property pairs (replicates) were blocked to incorporate the dependency of properties within a pair and treated as random. Models were developed using the GLMMADMB package in R (Fournier et al., 2012). The most parsimonious models with the lowest value of Akaike's Information Criterion (AIC) were retained for interpretation.

Plant community data was ordinated using non-metric multidimensional scaling (NMDS). This method has been used for a similar purpose (Fensham et al., 1999; Dowling et al., 2005). The three-dimensional solution had an acceptable stress-value (0.142) and was retained for interpretation. The direction of change for the six functional groups was related to community composition using a vector fitting approach (Envfit in R) and displayed

graphically. Analyses were conducted using the Vegan package in R (Oksanen et al., 2018) and group differences tested using the Adonis function in the Vegan package.

The soil nutrients, P, Ca, C and N and the two SSI classes, litter and soil surface roughness (SSR; see Appendix 10) that we considered contributed substantially to the LFA outcomes were modelled with linear mixed effects models (LME) using the NLME package in R (Pinheiro et al., 2017). For litter and SSR, each assessment point along transects was considered a data point, with transects treated as random effects. The same management and environmental variables as in the beta-distributed GLMM models were used. The LME models were compared using Restricted Maximum Likelihood tests to determine the significance of each explanatory variable in the simplest models with the lowest AIC.

Differences in the ratio of rest-to-graze were also analysed using LME models, with property pairs treated as random. Rest to graze ratios and management approach were strongly correlated (Spearman rank order correlation; $S = 0.70$, $p < 0.001$) with SDG being characterised by significantly larger rest to graze ratios than RP ($T = 4.39$, $p = 0.007$; Table 3). Models that included management approach had lower AICs than models that included rest-to-graze ratio. Therefore, management approach was used in final models in preference to rest-to-graze ratio.

3 Results

3.1 Plant abundance

Perennial herb cover was significantly greater on SDG than RP sites (Figure. 2a; Table 2) and was also influenced by geological substrate, with more perennial cover on basalt-derived soils than granite-derived soils (Figure. 2b; Table 2). Conversely, the cover of introduced annual and short-lived perennial species was lower on SDG than RP sites (Figure. 2c; Table 2) and on basalt-derived soils (Figure. 2d; Table 2).

Eleusine tristachya was a major component of the introduced annual functional group. This species accounted for 16% of cover, on average, on RP sites, but only 3% on SDG sites (Table 2). The abundance of *E. tristachya* was also influenced by geological substrate, being more abundant on granite soils (Table 2).

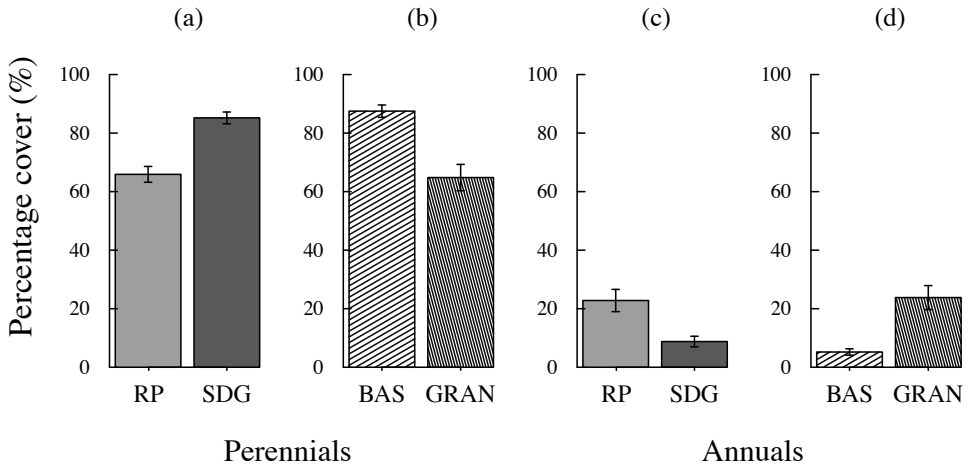

Figure 2 Mean percentage cover of perenn a herbs and ntroduced annua s under contrast ng management (a, c) and on so s der ved from d fferent geo og ca substrates (b, d): BAS refers to basa t-der ved so s; GRAN refers to gran te der ved so s. Error bars are ±1 s.e.m.

The increase in cover of introduced C3 perennial species with greater mean annual rainfall was significantly greater on SDG sites than RP sites (Figure. 3a and b; Table 2). Greater rainfall in the previous 5 months was significantly associated with a reduction in cover of these introduced C3 perennials (Table 2).

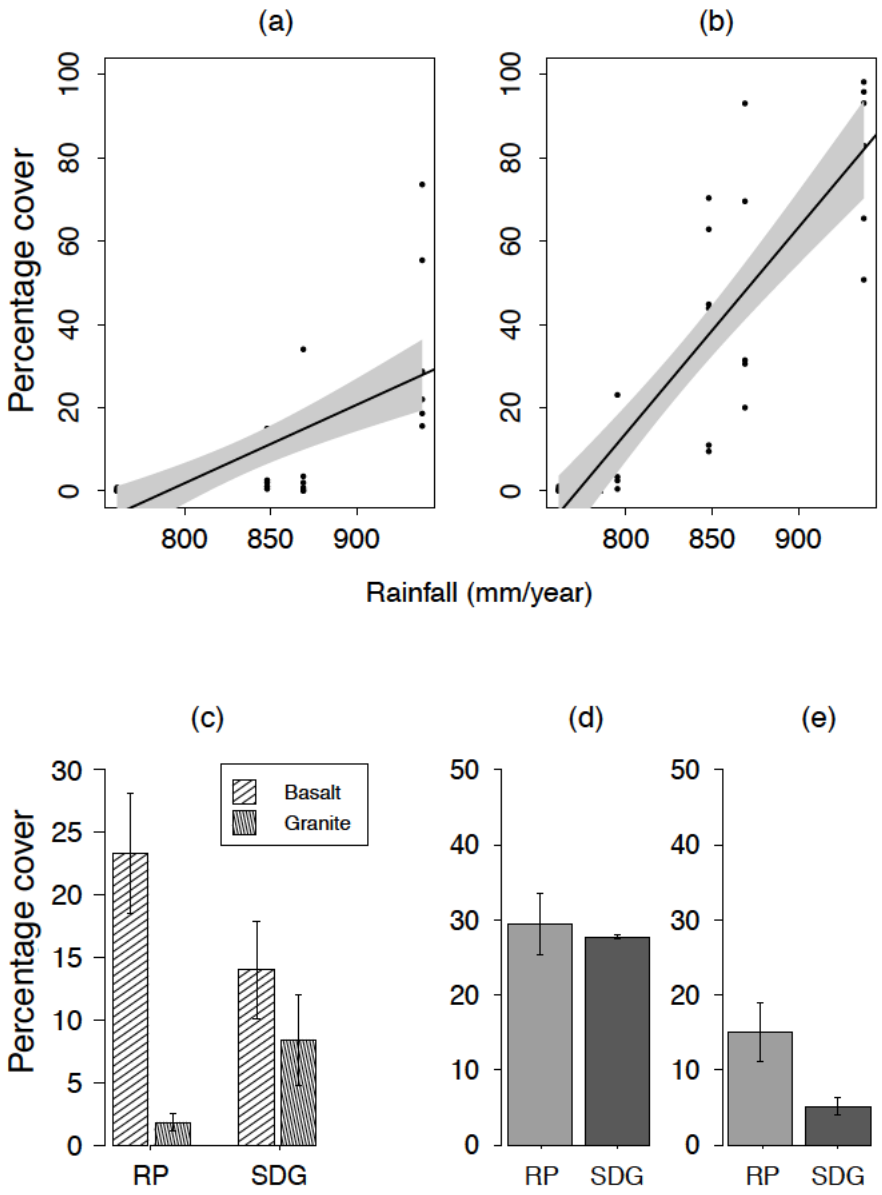

Figure 3 Re at onsh p between mean percentage cover of ntroduced year- ong perenn a s as mean annua ra nfa ncreased on (a) RP (n = 35) and (b) SDG (n = 36) s tes respect ve y; (c) average cover of nat ve year- ong green perenn a s under the two contrast ng management approaches for the two geo og ca substrates; (d) cover of warm-season perenn a spec es and (e) cover of the ncreaser spec es, *C. dactylon* and *B. macra*, under the two contrast ng management approaches. Least squares regress on nes are shown n (a) and (b); error bars (c–e) ±1 s.e.m.

The abundance of native C3 perennial species responded differently to the two geological substrates depending on management approach. At RP sites, very little of this functional group was present on granite-derived soils while on SDG sites there was significantly greater cover of this group on this soil type (Figure. 3c; Table 2). On basalt-derived soils, there was greater cover of these species on RP sites compared with SDG sites (Figure. 3c), but this difference was not significant (Table 2).

The cover of native C4 perennials was similar between management approaches (Figure. 3d) but varied with mean annual rainfall, with greater cover of this functional group at lower rainfall sites (Table 2). Native C4 perennials were also present in significantly greater abundance where there had been greater rainfall in the previous month (Table 2).

The combined cover of *C. dactylon* and *B. macra*, was significantly greater at RP sites (Figure. 3e and Table 2) and decreased as Ca levels increased (Table 2).

Table 2 Res ts from GLMMADMB ana yses for p ant cover c asses. AIC s for fu and f na mode s are shown w th Z – va ues and assoc ated probab t es as superscr pts (ns denotes not s gn f cant n the f na mode) Int. annua s are ntroduced annua and short- ved perenn a herbaceous p ants; IYLPs are ntroduced year- ong perenn a forage p ants; NYLPs are nat ve year- ong green perenn a forage p ants; WSPs are nat ve warm-season perenn a s; ncreasers are two spec es that ncrease under heavy graz ng pressures (*Bothriochloa macra* and *Cynodon dactylon)* management refers to changes from RP to SDG; Mean ra n s mean annua ra nfa , geo ogy s the change from gran te to basa t der ved so s; recent ra n refe s to the number of ra nfa events greate than 10 mm ove t e prev ous month or the prev ous 5 months, as nd cated by adjacent superscr pts; Ca refers to ca c um eve s n the top 5 cm of the so prof e.

Model	*AIC*	*Management*	*Average rain*	*Geology*	*Recent rain*	*Management int. Mean Rain*	*Ca*	*Management int. geology*
Perennials	Full: – 86.7 Final: – 95.6	3.31, $p < 0.001$	*ns*	3.02, $p = 0.002$	*ns*	*ns*	*ns*	*ns*
Int. annuals	Full: – 209.9 Final: – 219.2	–2.54, $p = 0.010$	*ns*	–2.77, $p = 0.002$	*ns*	*ns*	*ns*	*ns*
IYLPs	Full: – 357.9 Final: – 347.7	*interaction*	*interaction*	*ns*	[b] 5.25, $p < 0.001$	1.96, $p = 0.05$	*ns*	*ns*
NYLPs	Full: –309.6 Final: -320	*interaction*	*ns*	*interaction*	*ns*	*ns*	*ns*	–2.42, $p = 0.01$
WSPs	Full: – 216.2 Final: – 223.0	*ns*	–6.53, $p < 0.001$	*ns*	[a] 2.28, $p = 0.007$	*ns*	*ns*	*ns*
Increasers	Full: – 500.1 Final: – 513.4	– 2.94**	*ns*	*ns*		*ns*	–1.95, $p = 0.051$	*ns*
Eleusine tristachya	Full: – 624.8 Final: – 635.1	–2.04, $p = 0.042$	*ns*	–3.99, $p < 0.001$	*ns*	*ns*	*ns*	*ns*

3.2 Non-metric multidimensional scaling

Non-metric multidimensional scaling showed a limited amount of separation of management approaches in ordination space (Figure. 4a). Separation between management approaches was less than the separation of plant communities according to geological substrate types (Figure. 4b). There was also a separation of plant community composition between tableland sites that were at a higher altitude and those that were on lower slopes to the west (Figure. 4c). Vectors showed a direction of change for plant functional groups that was consistent with the GLMM results (Figure. 4 a,b,c). While testing of differences between group means showed significant effects, the R^2 value for all three factors was low (management approach $R^2 = 0.06$, $p = 0.005$; geological substrate $R^2 = 0.15$, $p = 0.005$; between tablelands and western slopes sites: $R^2 = 0.13$, $p = 0.005$), suggesting most of the variation influencing these plant communities was not explained by our data.

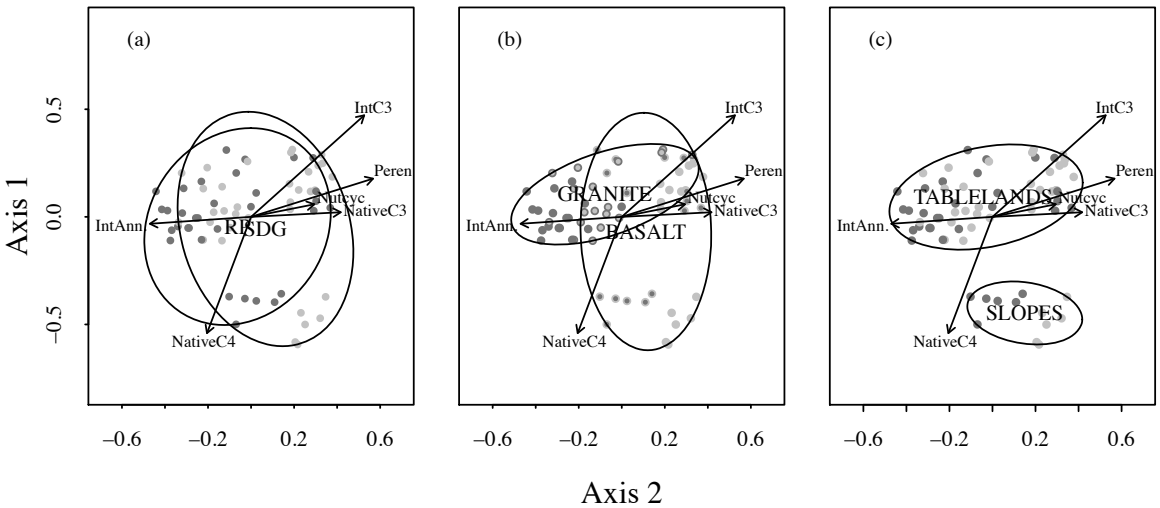

Figure 4 Mu t -d mens ona sca ng ana ys s of pasture funct ona groups (perenn a s, ntroduced annua spec es, des rab e ntroduced C3 pasture spec es, nat ve C3 pasture spec es, nat ve C4 pasture spec es and ncreaser spec es). Short-durat on graz ng s tes are ght grey symbo s and reg ona pract ce s tes are Dark grey symbo s. E pses surround s tes w th n each management category (a), that are basa t-der ved (Bas) or gran te-der ved (Gran) (b) and (c) at h gher tab e and s tes (Tab e ands) compared to ower e evat on s tes on the western s opes (S opes) of the study reg on. Arrows show the d rect on of change for s gn f cant var ab es w th the re at ve ength of each vector represent ng the re at ve magn tude of change for each p ant funct ona group.

3.3 Ground-layer biophysical attributes

Mean (± 1s.e.m.) plant and litter cover was similar at SDG (98.5 ± 0.3%) and RP sites (95.5 ± 1.2%; $Z = 1.92$, $p = 0.055$; Figure. 5a). The site with the least overall cover (69%) was an RP site, while the least amount of overall cover at any SDG site was 93%. Across the 2 years surveyed, more SDG sites had ≥ 99% cover than RP sites (24 compared to nine sites, respectively). Three RP sites had < 80% cover compared to zero SDG sites (Figure 5a).

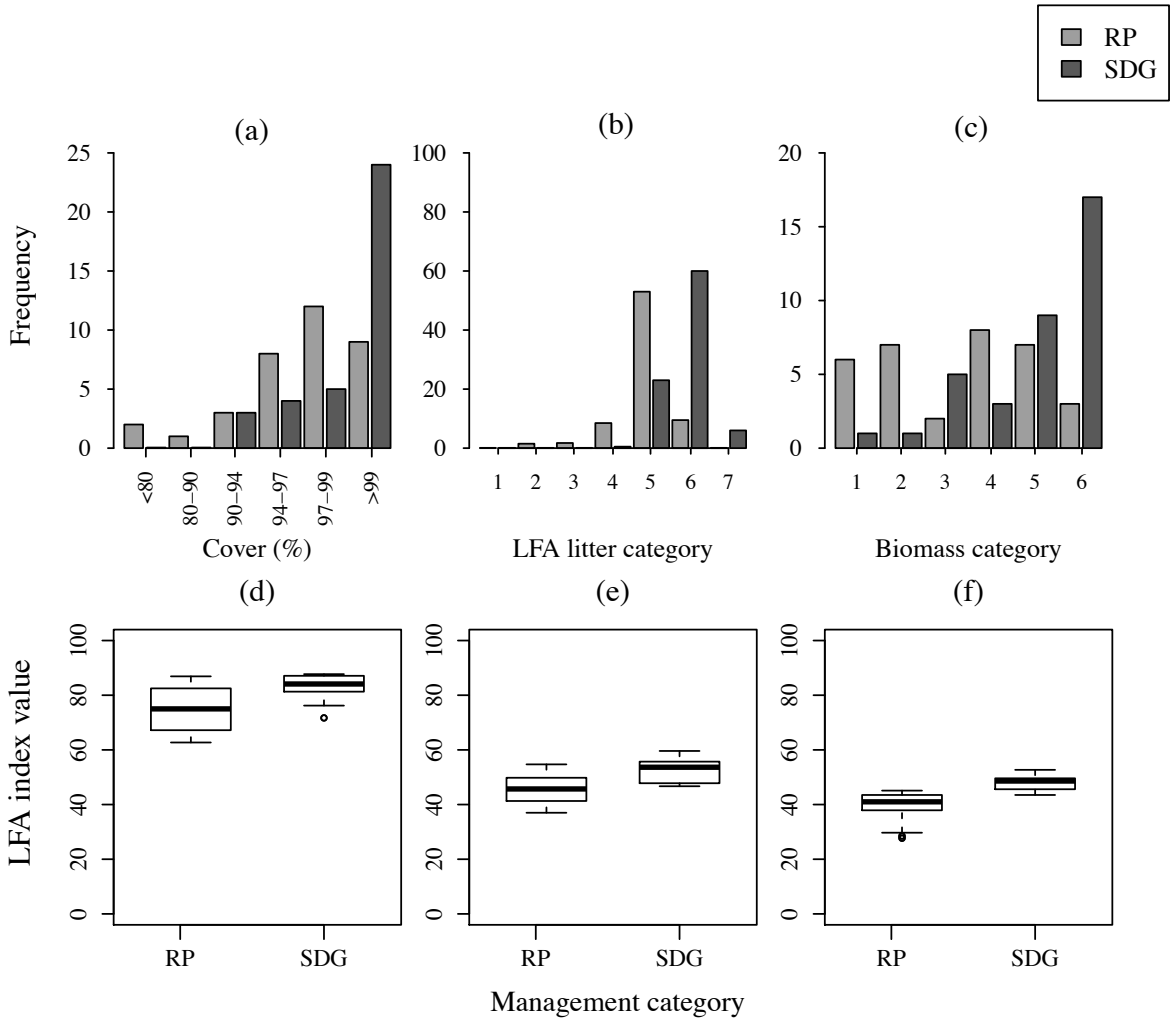

Figure 5 Frequency h stograms of the two graz ng management categor es n re at on to: (a) overa p ant fo ar and tter cover; (b) tter cover and depth from F gure. 4.5. Frequency h stograms of the two graz ng management categor es n re at on to: (a) overa p ant fo ar and tter cover; (b) tter cover and depth LFA assessments (class1: < 10%, class 2: 10–25%, class 3: 25–50%, class 4: 50–75%, class 5: 75–100%, class 6: 100% and up to 20 mm thick, class 7: 100% and 20–70 mm thick), and (c) biomass categories (class 1: < 300 kg ha^{-1}, 2: 300–800 kg ha^{-1}, 3: 800–1300 kg ha^{-1}, 4: 1300–1800 kg ha^{-1}, 5: 1800–2300 kg ha^{-1}, 6: > 2300 kg ha^{-1}); (d, e, f) are the LFA indices for soil/sediment stability, water infiltration and nutrient cycling respectively. The dark line on the box plots represents the median value for each management approach, the extent of the boxes the inter-quartile range and extreme lines show the highest and lowest values (excluding outliers). Y-axis shows 0–100 reflecting that each index is scored out of 100.

Table 3 Results from GLMMADMB and Linear Mixed Effects analyses for the three landscape function indices, litter cover and depth, soil surface roughness (SSR) and the amount of retained herbage mass in pastures. AICs for full and final models are shown with *Z* - values for the GLMMADMB models and *T* - values for the Linear Mixed Effects models (ns denotes not significant in the model). Associated probabilities are shown as superscripts. Management refers to the change from RP to SDG, geological substrate is what soils are derived from at each site with granite is the reference substrate.

Model	*Model type*	*AIC*	*Management*	*Geology*
Stability	GLMMADMB	Full: – 272.5 Final: – 259.5	$Z = 2.91$, $p = 0.004$	*ns*
Infiltration	GLMMADMB	Full: – 297.1 Final: – 299.3	$Z = 2.85$, $p = 0.004$	*ns*
Nutrient cycling	GLMMADMB	Full: – 276.0 Final: – 279.6	$Z = 3.90$, $p < 0.001$	*ns*
Litter cover and depth	NLME	Null: 10159.3 Final: 8349.3	$T = 48.1$, $p < 0.001$	$T = 6.76$, $p < 0.001$
SSR	NLME	Null: 5269.7 Final: 4158.6	$T = 35.5$, $p < 0.001$ $T = -4.4$, $p = 0.007$	$T = -6.9$, $p < 0.001$
Rest-to-graze ratio	NLME	Null: 110.47 Final: 99.57		*na*

The amount of litter cover and depth determined by the LFA assessment was significantly greater on SDG than RP sites (Figure. 5b; Table 3). On SDG sites, 74% of points assessed had litter in class 6 or 7 (100% cover and up to 20mm depth; 100% cover and between 20mm and 70mm thick respectively) compared to only 13% of RP points assessed (Figure. 5b). Litter cover and depth also differed according to geological substrate, with granite-derived soils having significantly lower litter cover and litter depth than basalt-derived soils (Table 3).

The measure of SSR relating to micro-topographical relief and determined by LFA assessment was significantly greater at SDG sites than RP sites (Table 3).

Greater herbage mass was retained on SDG sites than on RP sites (Figure. 5c; Table 3).) with only three RP sites being in the higher category of pasture biomass (>2300 kg ha^{-1}) compared to 17 of the SDG sites. Conversely, over twice the number of RP sites (15) had retained less than 800 kg ha^{-1} in pastures compared to seven SDG sites (Figure. 5c).

Compared with RP sites, the greater perennial plant cover, litter cover and depth and SSR on SDG sites resulted in significantly higher values for the three LFA indices of soil stability (Figure 5d, Table 3), infiltration (Figure 5e, Table 3) and nutrient cycling (Figure 5f, Table 3).

Changes in soil nutrient status for major nutrients, P, N and C and Ca in the top 5-cm of the soil profile differed between basalt and granite-derived soils (Basalt-derived - $mg.kg^{-1}$: P = 70.4 ±8.32, N = 5.83 ±0.45, C = 0.48 ±0.04, Ca = 15.67 ±2.24; granite-derived - $mg.kg^{-1}$: P = 29 ±2.8, N = 3.92 ±0.45, C = 0.32 ±0.02, Ca = 3.57 ±0.26) with reduced levels of Ca were associated with greater cover of increaser species. However, no differences for these nutrients were found between management approaches.

4. Discussion

This study found that short-duration grazing management of naturalised pastures on commercial properties was associated with favourable outcomes for functional and compositional attributes of pastures compared to more usual management typical of the region. The improvements observed included greater persistence of perennial herbaceous species including native C3 grasses that are valuable as winter forage and corresponding reductions in the cover of introduced annual species and species that typically increase under heavy grazing pressure. Pastures under SDG management were also characterised by increased cover and depth of plant litter and greater retention of pasture biomass indicating improved landscape function. These findings supported our initial hypotheses. Pasture composition was also strongly influenced by the environmental variables of underlying geological substrate and local rainfall. However, there was a large amount of unexplained variation in pasture composition suggesting there were many factors unaccounted for in our analyses.

A key difference between the two management approaches was the ratio of the time pastures were rested for compared to the amount of time pastures were grazed for (rest to graze ratio). Despite variability within categories, on SDG sites this ratio was significantly larger, ranging from 13:1–60:1 (median = 27:1), than on RP sites where it ranged from 0 to 3:1 (median = 0.5:1). Effectively the two management categories equated to two categories of rest to graze with typically extended rest on SDG sites and little rest on RP sites. For SDG sites, the length of rest depended on local environmental conditions and was based on the recovery of key perennial forage species, while on RP sites rest was less planned and generally in response to limited forage availability as is common for management of naturalised pastures in this region (Scott et al., 2013; Shakhane et al., 2013). Extended rest allows taller perennial forage plants the opportunity to gain stature (Teague et al., 2013), thereby imparting a competitive advantage over relatively prostrate species (such as *C. dactylon* and *B. macra* and many annual, weedy species). Our findings are consistent with Teague et al. (2011), who similarly investigated responses where rest to graze ratios were of a similar magnitude. The lack of rest and recovery for plants is likely contributing to an increased abundance of relatively short species at RP sites. While it is rare that research examining rotational grazing practices has looked in systems where rest-to-graze ratios exceed 10:1 (unpublished data), modelling by Teague et al. (2015) demonstrated the importance of extended (although not too long) pasture rests.

The regular pasture monitoring by SDG managers is also likely to facilitate proactive adjustment of stocking rates preceding and during dry times. This monitoring enables reductions in grazing pressure before feed shortages eventuate thereby minimising excessive grazing of more palatable, less grazing-tolerant plants that are typically of taller stature. The smaller grazing units necessary to achieve extended rest periods on SDG properties likely support this decision-making process with managers able to more easily visualise and

effectively monitor plant recovery where pastures are not being repeatedly grazed (Steffens et al., 2013; Teague et al., 2013). Thus, the differences seen here are likely due to avoidance of heavy grazing of more palatable and taller perennial species over a timeframe of several years. This is long enough to see changes in pasture composition independent of cycles of extended dry periods and times when forage is limited.

Landscape function analysis was an important component of this study. While a large body of work has considered pasture compositional responses to SDG styles of management, including groundcover and herbage mass changes, relatively little research has considered how attributes that relate to soil processes vary between rotational and continuous grazing (but see Teague et al., 2011). This may be because these characteristics do not directly relate to animal production, but rather to more general ecosystem processes and consequently receive less attention (Whalley, 2000). Although quantifying the benefits arising from these improvements was outside the scope of this study, the significant improvements seen here under SDG management are highly relevant to maintaining productivity, ecological function and resilience of grazing landscapes (Teague et al., 2004; Müller et al., 2007; Teague et al., 2011; Teague et al., 2015). When considered in the context of: (1) large areas of the landscape; (2) with climate change scenarios resulting in increased temperatures and rainfall events that are likely to be of greater magnitude (Hughes, 2003; Hennesey et al., 2008) and (3) across several years, these improvements are likely to be highly relevant to the ecological and economic sustainability of commercial grazing enterprises.

We suggest that the accumulation of litter to the extent seen on SDG sites in this study may not be achievable under continuous grazing practices. The extended rest that is common under SDG management, followed by grazing with relatively high animal densities during grazing periods is likely to allow tall grasses and mature plant tissue to be trampled by

animals and turned into litter. This is particularly true of tall native species that, without trampling by dense mobs of livestock, can remain standing for a long time (Whalley, 2017). While there appears to be limited work investigating this hypothesis, a study by Beukes and Cowling (2000) did show clear differences in the amount of litter present in areas stocked at much higher densities (4x) and rested for 12 months between grazing intervals. The physical breakdown by livestock is likely to facilitate contact of dry matter with soil, thereby providing a food source for soil organisms (Whalley, 2000). This increased litter load has important consequences for a range of important ecosystem processes including nutrient cycling (McNaughton et al., 1988), water infiltration and aeration of the soil (Tongway and Hindley, 2005) and creation of seedling microsites (Milton, 1992). Litter is also vital for conserving resources such as water, soil and nutrients and to moderate the environment at the soil surface (Beukes and Cowling, 2000). While there may be a trade-off of reduced abundances of legumes under high litter loads (Leigh et al., 1995), the combination of improved landscape function from deep litter layers (including reduced loss of soluble nutrients such as P and N in run-off (McCaskill and Cayley, 2000), increased soil surface micro-topography and strong tussocky perennial grasses, in addition to favourable pasture compositional changes may be an important outcome of SDG management that is rarely considered in studies of sustainable grazing systems. Management practices that allow taller perennial species to gain stature, combined with intermittent trampling by relatively dense mobs of livestock to turn plant material into litter that benefits soil processes, may be necessary to achieve these dual outcomes of improved pasture composition and landscape function.

The relationship of LFA to biophysical outcomes has been verified in a range of environments (Yates and Hobbs, 2000; Ata Rezaei et al., 2006; Maestre and Puche, 2009; Zucca et al., 2013) and LFA has been used as a research tool in temperate grazed landscapes

(McIntyre and Tongway, 2005; Read et al., 2016). Although there is uncertainty around the ability of the nutrient cycling index to predict some aspects of nutrient availability in grazed areas (Eldridge and Delgado-Baquerizo, 2018), plant litter is important for the cycling of P and N in pastures (Leech, 2009). Therefore, while litter plays an important role in minimising the loss of nutrients in run-off and facilitates their availability, further work would be valuable to quantify the benefits of LFA indices, particularly the nutrient cycling index, in heavily grazed areas. With increased certainty around the quantification of these indices, LFA would be a valuable tool to improve the ability of graziers to understand and monitor the biophysical condition of pastures.

Although only broadly categorised, the greater retention of pasture biomass at SDG sites was a further difference between management approaches. This likely reflects more timely adjustments of stocking rates in response to changing conditions that is a characteristic of SDG management. Research literature and industry extension material regarding the sustainable management of naturalised pastures in temperate Australia recommends retention of around 1500 kg ha□1 or more of dry matter in pastures (Kemp et al., 2000; Kahn and Earl, 2007; Badgery et al., 2008; Dorrough et al., 2008). In this study, pasture biomass levels were often below that amount on RP properties, suggesting that feed budgeting and timely adjustments to stocking rates happen less under more usual management practices on naturalised pastures.

While changes in abundance for several functional groups in our study were significant, there was also large amount of unexplained variation that did not relate to management approach. Likely contributing to this were inherent site differences such as legacy impacts of previous cultivation practices and fertiliser history that are known to influence pasture composition (Scott and Whalley, 1982; Reseigh, 2004). These factors could not be included

in analyses due to low sample sizes and difficulty obtaining this information. However, pasture compositional changes were only one aspect of the study, with improvements to landscape function also being an important component of sustainable pastures that we assessed (Whalley, 2000; Dorrough et al., 2008). It is likely that these landscape function changes are a major aspect of the improvements claimed by advocates for SDG styles of management. We suggest that these advocates may not be distinguishing between compositional and landscape function attributes.

A unique aspect of this study is that it examined outcomes for pasture composition and landscape functioning over timeframes upwards of ten years on commercial grazing properties (i.e., at the ranch scale). In commercial situations, decisions are made by grazing managers rather than researchers and are in response to difficult to predict changes in markets alongside a variable and unpredictable climate (Teague et al., 2013). This type of study is unusual, with only four studies we are aware of having investigated outcomes on commercial properties where livestock are controlled by grazing managers (Earl and Jones, 1996; Dowling et al., 2005; Jacobo et al., 2006; Teague et al., 2011). Earl and Jones (1996), Jacobo et al. (2006) and Teague et al. (2011) all concluded that there were generally favourable outcomes with an SDG approach to management. However, Dowling et al. (2005) found no consistent response time-controlled grazing (a variation of SDG management) across several sites. Dowling et al. (2005) used multiple statistical approaches, finding evidence of favourable composition at some time-controlled grazing sites with ANOVA techniques but few differences with alternative techniques. They considered their results to be complicated by variable pasture compositions at the start of the investigation, ultimately concluding there were no advantages for time-controlled grazing. While the trends found by Dowling et al. (2005) are similar to trends we found in our investigation, there are some key differences between our study and that of Dowling et al. (2005) that may account for the differing

conclusions. These are: (1) the concentration of our study within a more limited geographical region which is likely to have resulted in less variation in composition among paired sites, (2) the use of linear mixed model statistical techniques enabled us to account for variation between paired property locations and (3) the extended time that SDG management had been in place for in our study would have encompassed multiple periods of climatic stress (i.e. dry times) with management therefore impacting pasture composition over long timeframes. While Dowling et al. (2005) investigated pasture attributes across six years, they do not state how long time-controlled grazing management had been in place for prior to their study (although two of the sites they used were also used by Earl and Jones, 1999). These extended timeframes are likely to be of critical importance as significant pasture changes are a response to long-term management factors (Teague et al., 2013) such as adaptively responding to changing market and variable climatic conditions. Consistent with our findings, the study by Jacobo et al. (2006) also considered litter cover, finding increases in rotational grazing systems. Similarly, Teague et al. (2011) considered a range of biophysical attributes on properties where an SDG style of management had been in place for more than 9 years and found positive responses for several attributes relating to landscape functioning. Neither Earl and Jones (1996) nor Dowling et al. (2005) presented findings on litter. The combination presented here of pasture compositional changes (that vary substantially across geographic areas and over time), combined with an examination of landscape function attributes is in line with the findings of Teague et al. (2011) particularly and demonstrates important changes occurring under SDG management that are likely to benefit the resilience of grazing businesses.

It is likely that SDG management enables managers to avoid excessive grazing of desirable plants and areas of the landscape to a greater extent than with more usual practices. In conjunction with improved landscape function, the outcome is a ground-layer with greater

surface micro-topography that in-turn positively impacts landscape function and pasture resilience. There are key features of SDG management that support this outcome. These are: (1) strategically planned fencing and water infrastructure that enables control of grazing pressure; (2) smaller grazing units leading to more homogenous use of pastures by grazing animals; (3) the intentional monitoring of pasture growth and recovery with the aim of matching stocking rates to pasture availability and (4) the increase in pasture height that occurs with extended rests, followed by dense mobs of livestock trampling mature plant material, allows the accumulation of litter at the soil surface. While the first three of these factors are arguably achievable under continuous grazing practices with smaller paddocks and improved water infrastructure, we suggest it is unlikely that ground-layer litter could accumulate to the same extent under more continuous grazing practices. This hypothesis would benefit from targeted research.

Obtaining explicit financial data was outside the scope of this study. However, as SDG management necessitates intensive fencing and water infrastructure (i.e. an additional cost for grazing managers), this needs to be factored into consideration around adoption of these practices. Case-studies from practitioners of SDG claim that costs for this infrastructure are recovered within 2 years (Wright, 2017). A bio-economic analysis of an approach similar to SDG which incorporated tactical rest to promote the perennial grass component of pastures considered economic and environmental outcomes over a 20-year timeframe (Jones et al., 2006). This analysis concluded that tactical rest grazing had economic and environmental sustainability benefits compared with grazing that did not incorporate rest (Jones et al., 2006). While environmental outcomes such as (salinity, biodiversity, erosion etc.) can be difficult to place an exact value on, the planned fencing and water infrastructure that helps achieve these multiple outcomes is essential for managing pastures during extended dry periods. Government support for this infrastructure may be more appropriate than reactive

drought relief policies (Sherren et al., 2012). Although decisions to alter practices will depend on the overall context of the grazing businesses, there appears to be reasonable evidence that the additional set-up costs to implement SDG should not be a deterrent to this approach to managing livestock.

Importantly, although ordination techniques should be viewed as graphical representations rather than robust statistical analyses (Oksanen, 2019), the ordination did demonstrate that there was substantial variation in pasture composition that was not explained by management approach, or any of the other factors we examined. The outcomes of this study should be considered within that context.

In addition to management approach, underlying geological substrate and rainfall patterns were important drivers of pasture composition with interactions also found between grazing management and these environmental factors. In particular, higher mean annual rainfall and altitude were associated with a greater abundance of high-value introduced pasture species. Historical land management factors would also be influencing these patterns. Due to higher rainfall and higher fertility of basalt-derived soils at increased elevations, pastures have been sown and fertilised more frequently with introduced species than sites at lower altitudes and with lower soil fertility (Keys, 1996). However, the increased abundance of introduced C3 perennial species under SDG management at these higher elevations suggests that planned and extended rest is supporting the retention of higher-value forage species in pastures. Rapid response to rainfall was also evident in native grasses in our surveys, thus highlighting the value of these species as an important component of resilient pastures.

There are limitations to this study that need to be acknowledged. These include that plant surveys were only taken during late summer and that soil samples were only taken to 5-cm

depth. Due to the increased variability that is common in these surface layers (Pulido et al., 2016), we may have overlooked associations of soil nutrients with pasture attributes. Due to resource limitations, detailed and specific data on animal production could not be obtained. However, coarse data on stocking rates was collated, and while in two cases this data was unreliable, stocking rates under both management approaches were very similar. Therefore, the positive responses seen on SDG sties in the study cannot be attributed to lower stocking rates.

Finally, it is possible that the SDG managers in this study were particularly astute and motivated observers who were more concerned with the composition and function of their pasture-base than the paired RP property managers. To minimise the possible influence of these differences in management skill and motivation over the results we sought to use only RP properties that were judged to be well-managed by local standards. However, we acknowledge that this is a flaw in the study design that was difficult to avoid.

Conclusion

The results of this study suggest that in a commercial context, SDG is a management strategy that is able to successfully address issues that are of ongoing concern in temperate pastures in the region. While it is uncertain exactly what aspects of management are responsible for the improvements in ground-layer composition and functioning observed, key characteristics of sustainable grazing management such as retaining adequate pasture biomass and proactively adjusting stocking rates to prevent excessive grazing during times of feed shortage are likely to be facilitated by the regular movements of stock and targeted pasture monitoring that occur under SDG. While these practices are possible under continuous grazing, it may be more difficult to implement appropriate rest for targeted species where stock are still present in paddocks. We also suggest it is unlikely that the depth of non-

standing plant litter can develop to the same extent under continuous stocking practices as seen here under SDG, and that it would be valuable to explore this further. While there were challenges associated with working on commercial properties, including lack of availability of detailed information for some factors, this study has provided an insight into some favourable outcomes that are occurring under SDG management where it has been in place for several years. In some ways our results are at odds with the conclusions of many studies and reviews in this area of research. We consider these differences to be due to having investigated the effects of long-term changes in management as well as the additional consideration of landscape function that is an integral component of managing grazed pastures for resilience.

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