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Livestock grazing management and biodiversity conservation in Australian temperate grassy landscapes

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Abstract. There is an increasing interest in the development of livestock grazing management strategies that achieve environmental sustainability and maintain or improve the long-term production capacity of commercial grazing systems. In temperate Australia, these strategies are generally focussed on reducing perennial pasture decline, soil loss, acidity, and salinity. An additional challenge facing land managers and researchers is developing grazing strategies that also maintain and enhance local and regional biodiversity. However, few studies have assessed the compatibility of management practices for maintaining long-term productivity and biodiversity conservation. We still have only a very basic understanding of the effects of different grazing strategies and pasture management on biodiversity and this is a major impediment to the development of appropriate and compatible best management practice. We argue that although there is an increasing desire to find management strategies that protect and enhance biodiversity without hindering long-term agricultural production, in many cases this may not be possible. Current knowledge suggests that compatibility is most likely to be achieved using low-input systems in low productivity (fragile) landscapes, whereas in highly productive (robust) landscapes there is less opportunity for integration of productive land-use and biodiversity conservation. There is an urgent need for improved communication and collaboration between agronomic and ecological researchers and research agencies to ensure that future programs consider sustainability in terms of biodiversity as well as pasture and livestock productivity and soil and water health.

Additional keywords: sustainability, ecosystem function, vegetation heterogeneity, grazing strategies, perennial grasses, agriculture.

Introduction

In temperate Australia, conservation of native biodiversity is rarely viewed as an integral component of economically and socially viable livestock production. In this paper, biodiversity is defined as the composition of native biota and the interactions among biota and between biota and their environment. The structural and functional components of an ecosystem are also considered as part of this definition as they determine the arrangement of species and habitats in space and time. Research examining the long-term sustainability of livestock and pasture production has also often excluded consideration of biodiversity conservation. Even so it is increasingly believed that the level of internal regulation of ecosystem function in agro-ecosystems is partly dependent on plant and animal diversity (Altieri 1999). It has been argued that biodiversity may enhance production of certain ecological services including nutrient recycling, regulation of microclimate and local hydrological processes, suppression of undesirable organisms, and detoxification of noxious chemicals (Altieri 1999).

In this paper we consider whether livestock grazing systems can be enhanced to improve biodiversity outcomes in the temperate grasslands and grassy woodlands of Australia. To achieve this we firstly examined the effects of past management strategies on biodiversity and demonstrate that the loss of native biodiversity is linked to other aspects

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Fig. 1. The temperate climate zone of Australia estimated using the Köppen classification scheme. This approximates the distribution of temperate grassy vegetation types in southern Australia prior to European settlement (source: Bureau of Meteorology).

of landscape degradation. Consideration of the processes responsible for change may also help to identify grazing systems that can cater for biodiversity and sustainable management of soils, water, vegetation, and livestock. Following this we examine whether some landscapes are more susceptible to biodiversity loss or landscape degradation than others and thus whether strategies for maintaining biodiversity should vary within and among landscapes. We suggest a framework that places landscapes on a continuum from fragile to robust (as per Tongway and Ludwig 2002) and argue that different strategies will be required along this continuum. The final section provides a critique of specific livestock management strategies that are either currently practiced or have been recommended from research. We consider whether these strategies may cater for biodiversity conservation in grazed landscapes.

The temperate grassland and grassy woodland zone of southern Australia

The temperate zone of southern Australia covers much of Victoria, Tasmania, and the eastern portion of NSW and includes isolated areas in southern South Australia and the south-west of Western Australia (Fig. 1). At the time of European settlement, much of this area was dominated by perennial grassland or grassy woodlands (Groves and Williams 1981; Orchard and Thompson 1999) and it was these areas that became the initial focus for livestock and associated agricultural land-use. Although other vegetation types occur in this region, they are not the subject of this paper.

Since European settlement, the extent of native-dominated grassland and grassy woodland vegetation has significantly declined (McDougall and Kirkpatrick 1994; Yates and Hobbs 1997). In some cases, more than 95% of the native vegetation has been cleared and sown to exotic pastures or crops (Table 1). In remaining native grasslands and grassy woodlands, grazing, fertilisers, and exotic plant species invasion have led to further biodiversity loss and habitat modification (Moore 1970; Whalley *et al*. 1978; McIntyre and Lavorel 1994*a*, 1994*b*; Prober and Thiele 1995; Hamilton 2001). These modifications have resulted in improved livestock production but have also had considerable effect on soil structure, nutrient cycling, water,

Table 1. The estimated extant percentage cover of three vegetation types on private land in temperate Australia
Disturbed vegetation has been sown to exotic pastures or crops. Woodland is all areas not cleared of native tree cover. Areas of native pasture include vegetation that has been fertilised and/or invaded by exotic plant species and is not necessarily dominated by native plants

Region	%Disturbed	%Woodland	%Native pasture	Reference
Victorian Basalt Plain, Vic	>95	<1?	0.1–5	P. Quigley (unpublished data); McDougall and Kirkpatrick 1994
Ararat Region, SW Vic.	61	2	37	Dorrough and Moxham (unpublished data)
Monaro Tablelands, NSW	31.5	22.3	46.3	Garden *et al*. 2000*a*
Southern Tablelands, NSW	49.7	19.4	30.9	Garden *et al*. 2000*a*
Central Tablelands, NSW	41.6	10.8	47.6	Garden *et al*. 2000*a*

pasture productivity and palatability, and thus on the long-term sustainability and productivity of the livestock industries.

The development of local and regional management strategies aimed at mitigating the environmental effects of the livestock industries, while maintaining or improving their productivity, has become an increasing focus in these landscapes. On-ground strategies and research have focussed on: (*1*) increasing the drought tolerance and persistence of pastures (Kemp and Culvenor 1994; Kemp and Dowling 2000); (*2*) reducing soil and nutrient loss (Greenwood and McKenzie 2001), acidification (Kemp and Dowling 2000), and the effects and extent of salinity (Mitchell *et al*. 2001); (*3*) preventing or minimising invasion by low palatability or toxic plants (Grigulis *et al*. 2001); and (*4*) controlling invertebrate pest outbreaks (Delfosse 1993).

A result of recent pastoral research has been an increasing recognition of the contribution of native perennial plant species to livestock production in temperate Australia (Dowling *et al*. 1993; Lodge 1994; Johnston 1996; Garden *et al*. 2000*b*, 2001; Kemp and Dowling 2000), which contrasts significantly with attitudes of previous decades (Donald 1970). In some regions within the temperate zone, 60–70% of land supporting livestock enterprises is unsown native pasture or woodland (Table 1) and further areas previously sown are likely to have been re-invaded by native plants (Whalley *et al*. 1978; Garden *et al*. 2001).

Another branch of research has focussed on understanding the loss of native fauna and flora (McIntyre and Lavorel 1994*b*; Prober and Thiele 1995; Prober 1996; Bennett and Ford 1997) and developing strategies to maintain or restore biodiversity within the pastoral landscape (McIntyre *et al*. 2002). This research has clearly demonstrated that although many native perennial grasses may be widespread in the temperate zone, other biota have significantly declined. However, most research has focussed almost entirely on vascular plants and compared sites that have been intermittently grazed (densities unknown), continuously grazed (densities estimated), and ungrazed. Experimental research considering effects of alternative grazing strategies on native biodiversity is currently lacking.

Biodiversity loss and land degradation in temperate grazing lands

Livestock grazing management practices have resulted in considerable changes to the structure and composition of both the understorey and overstorey vegetation of temperate grassy vegetation. Three general changes to the understorey vegetation have been observed: (*1*) a decline in the diversity of native perennial forb species (McIntyre and Lavorel 1994*a*; Tremont and McIntyre 1994; Pettit *et al*. 1995), (*2*) a shift towards species with cool-season growth (Moore and Biddiscombe 1964; Moore 1970), and (*3*) an increasing dominance by annual plant species (Moore and Biddiscombe 1964; Moore 1970; Pettit *et al*. 1995). The overstorey has been modified by clearing while grazing by livestock, and other factors such as increased soil nutrient availability, altered insect–plant dynamics, and waterlogging have led to further declines in remnant woodland tree cover, via a condition known as tree die-back (Landsberg *et al*. 1990; Reid 2000). As we shall discuss below, these considerable changes in vegetation have also affected native fauna, soil microbial activity, nutrient cycling, soil hydrology, and the sustainability of grazing systems.

Prior to European settlement, perennial grass and forb species dominated the ground layer of temperate grassy ecosystems, whereas annuals were minor components (Costin 1954; Willis 1964; Moore 1970; Tremont and McIntyre 1994). Provision of watering points, supplemental feeding during droughts, and addition of fertilisers and nitrogen-fixing legumes enabled land managers to maintain herbivore densities at levels much higher than those before European settlement. In addition to increased herbivore densities, the foot pressure exerted by most introduced ungulates is greater than of native macropods (Bennett 1999). As a result of high herbivore densities and high rates of biomass removal, the rate and extent of bare ground creation is increased, particularly in late summer when soil moisture is low. This disturbance regime has favoured the establishment of invasive annual plant species, most of which are exotic, at the expense of native perennial species (Moore 1970). This pattern is particularly pronounced in the south of the temperate zone where rainfall is primarily winter-dominant. The typical grazing management in much of temperate Australia, set-stocking in relatively large paddocks (>40 ha), also results in uneven grazing pressure, increasing herbivore selectivity and loss of palatable species (Leigh and Holgate 1978).

The shift from diverse native perennial grasslands to exotic cool-season pastures dominated by annuals underlies significant environmental changes that have negative effects on ecosystem function and livestock productivity. The loss of perennial plant cover is a major factor in increasing the susceptibility of grasslands to weed invasion (Grigulis *et al.* 2001), and continual selection of palatable plant species by grazing animals favours increased dominance by less palatable plant species (Moretto and Distel 1999). These vegetation changes gradually reduce the productivity of grasslands for grazing by livestock. Dominance by annual species increases drought susceptibility and variability of pasture production, with peaks of production in spring and feed shortages in late summer and early autumn. Where exotic annual plants have replaced diverse native perennial grasslands or woodland, water capture and use is reduced, particularly in summer and autumn months when annual species are not actively growing (Johnston 1996, 2003). Consequently, rates of soil erosion can be high and increased rates of deep drainage can lead to dryland salinisation (Johnston 2001).

The sowing of annual nitrogen-fixing legumes and addition of phosphate fertilisers is considered to have been one of the major advances in Australian agriculture because it has greatly promoted grass growth without the need for costly nitrogen fertilisers (Smith 2000). However, these practices are generally more detrimental to the persistence of native biodiversity than grazing alone (Dorrough and Ash 1999; Fairfax and Fensham 2000; Dorrough 2001; Hamilton 2001; McIntyre and Martin 2001). In addition, in pastures dominated by cool-season exotic annuals and perennials, there can be incomplete utilisation of available nitrates following summer rain, hastening acidification of soils (Helyar *et al.* 1990; Johnston 2001). Acidification has now been observed throughout temperate Australia and, although lime can ameliorate acidity, it is only effective and economical if acidification occurs within the top 0–15 cm of soil, not when it occurs at depth (Simpson 1999). Some of the sown exotic species (e.g. phalaris and lucerne) are sensitive to increasing acidity and this can result in even further annual-plant dominance (Simpson and Langford 1996).

High livestock densities can also lead to loss of litter cover and soil microtopography, degradation of surface soil structure (Yates *et al.* 2000), and reduced richness and cover of microphytic soil crust flora (Rogers and Lange 1971). Soil crust flora play important roles in reducing erosion (Eldridge 1998), increasing water infiltration and nutrient cycling (Rogers *et al.* 1966), and may also have positive benefits for seedling establishment (Scarlett 1994). Declines in the abundance of microphytic soil crusts may also have hastened invasion by exotic annual flora, degraded invertebrate habitat, and led to general declines in ecosystem function, particularly in areas with low resource availability.

In conjunction with changes in plant species composition and richness, modifications to vegetation structure through livestock grazing has affected fauna habitat. High herbivore densities reduce litter cover and vegetation biomass and create a more even sward structure. The loss of perennial tussocks and increased bare ground may have effectively eliminated the habitat of many grassland fauna, particularly those requiring closed tussock vegetation. Some of the remaining species, such as the Earless Dragon (*Tympanocryptis pinguicolla*) (Smith 1994) and the Plains Wanderer (*Pedionomus torquatus*) (Baker-Gabb *et al.* 1990), are apparently favoured by an open tussock structure, but other species, such as the Striped Legless Lizard (*Delma impar*), are dependent on relatively closed perennial swards (Coulson 1990).

Most studies on invertebrate communities in grazing systems indicate that more intensively managed grazing systems have lower invertebrate richness and abundance (Rushton *et al.* 1989; Siepel *et al.* 1989; Dennis *et al.* 1997; Downie *et al.* 1999; Di Giulo *et al.* 2001). In particular, the abundance of most micro-, meso-, and macro-invertebrate groups, predatory invertebrates, and detritivorous and herbivorous invertebrates declines as grazing intensity increases (Hutchinson and King 1970, 1980; King and Hutchinson 1976, 1983; Abensperg-Traun *et al.* 1995, 1996). Insects such as the ptunarra brown butterfly and many species of xanthorhoine geometrid moths, whose larval stages feed on native grasses, are adversely affected by grazing, fire, fertiliser application, and the sowing of exotic plant species (Neyland 1993; McQuillan 1999). Changes in the composition and richness of invertebrate communities are apparently due to many factors including declines in plant species richness, loss of tussock-forming plant species, changes in ground surface microtopography and microclimate, increased compaction of soil, and fertiliser inputs (King *et al.* 1976; Hutchinson and King 1980; King and Hutchinson 1983; Di Giulo *et al.* 2001).

Not all research has found livestock management systems to have negative effects on invertebrates. Although ungrazed woodlands and native pastures are associated with a greater diversity of invertebrate orders, some studies have found total invertebrate abundance and biomass to be greater in exotic pastures (Hutchinson and King 1980; Bromham *et al.* 1999). Ants also tend to be more abundant in heavily grazed areas (Hutchinson and King 1980), most likely because increased bare ground, caused by grazing, favours opportunist species.

Value of native biodiversity to livestock production systems

Despite the history of negative effects of pastoral grazing systems on Australian biodiversity, it is increasingly understood that grazing, or at least some form of biomass removal, is essential for the maintenance of plant biodiversity in most grasslands (Lunt 1991; Gilfedder and Kirkpatrick 1994; Freidel and James 1995; Fensham 1998). Herbivore grazing can create gaps and reduce litter cover and the stature of dominant plants, leading to increased rates of plant invasion, thus increasing diversity (Armesto and Pickett 1985; O'Connor 1991; Reader and Buck 1991; Lavorel *et al*. 1994; Morgan 1998*a*). In the absence of grazing, short-lived species dependent on establishment from seed and low-statured, shade-intolerant species decline, leading to overall loss of plant species richness (Belsky 1992; Tremont 1994) and a gradual reduction in structural complexity. These results suggest that, in areas historically managed for livestock production, opportunities for maintaining native biodiversity do exist.

It is evident that many of the same processes that lead to the loss of biodiversity also contribute to declining productivity. Although grazing can be compatible with persistence of some native species, biodiversity is likely to continue to decline in high-input grazing systems (grazing systems that include high rates of resource additions and pasture replacement) that aim to maximise production and profit. Low intensity and low-input grazing systems (utilising minimal input of resources, except those required to replace nutrients lost from the system) based on perennial native plant species can be implemented with lower costs and risks, but short-term financial benefits may be less (Crosthwaite and Malcolm 2000). High-input systems often become annual-dominated and, although annual pastures use less water and less available solar energy than perennial pastures, because little of their production goes into roots or long-term structural components, their above-ground production is generally greater. However, by shifting to a more diverse perennial system, there can be an increase in the long-term persistence and yield stability of the pasture (Tilman *et al*. 1996, 2001), with long-term production benefits (Lefroy 2001).

There is anecdotal evidence that livestock utilising native grassland have fewer health problems than those grazed on improved and fertilised pastures (Rankin 1993; Crosthwaite 1996; Crosthwaite and Macleod 2000). Low-input native grasslands are perceived as 'clean' pastures, providing stock with a diverse diet and reducing disease and intestinal worms (Crosthwaite and Macleod 2000). Observations and research suggest that pest outbreaks are most common in high-input introduced and degraded pastures (Reid 2000), whereas small remnants of native grasslands with a tussock structure in western Victoria were found to have no exotic slugs and very few red-legged earth mites (Horne 1999). The wingless grasshopper does not thrive in perennial native grasslands and has become a pest of pastures as summer-growing native grasses have been progressively replaced by introduced, winter- and spring-growing annual grasses and legumes (Farrow and Baker 1993). Colonisation rates by dispersing winged adults are greater in overgrazed paddocks (Baker 1993) and in pastures disturbed by ploughing and sowing; overgrazing of annual pastures and droughts can lead to outbreaks.

In certain landscapes, particularly on low fertility and acidic soils, native pastures can be equally or more productive than sown exotic perennial pastures (Jones 1996; Simpson and Langford 1996). Regardless of soil fertility, the productive capacity of pasture types varies between and within particular years and, at certain times, native-dominated pastures can be as or more productive than sown pastures (Jones 1996; Simpson and Langford 1996). As a result, native plant-dominated areas are used by some graziers to fill the late summer–early autumn feed gap when active growth by exotic cool-season perennial and annual grass pastures has ceased.

Significant production value can also arise from non-grass components of the native grasslands. In terms of species richness, most vascular plant species in temperate grasslands are inter-tussock forbs (Tremont and McIntyre 1994). Although they constitute a small percentage of the actual vegetative cover, these species are often highly palatable and can form a large part of the diet of domestic livestock (Leigh and Holgate 1978; Wimbush and Costin 1979). However, many perennial inter-tussock forbs are susceptible to grazing effects, and although they may contribute considerably to livestock production when common, their value decreases dramatically as they become rare.

Should strategies vary across the landscape? Resource availability and ecosystem resilience

The year-round availability of resources to plants is likely to play a large role in determining the resilience of ecosystems to disturbances or stress, such as that imposed by livestock grazing. On this basis we consider ecosystems to fall along a continuum from fragile (low resource availability and supply) to robust (high resource availability and supply) as per Tongway and Ludwig (2002) (Table 2). The resilience of ecosystems is likely to be lower in 'fragile' landscapes where water and/or nutrients limit plant growth and persistence (Fig. 2*a*). In fragile landscapes, reductions in plant cover due to grazing are more likely to result in soil degradation and extreme micro-climates, limiting the potential for plant growth (Ludwig and Tongway 1996; Yates *et al*. 2000) and re-establishment (Anderson *et al*. 1996; Ludwig and Tongway 1996; Yates *et al*. 2000). In contrast, in robust

Table 2. Abiotic characteristics of robust and fragile landscapes

Variable	Robust	Fragile
Soil nutrient availability	High	Low
Soil structure	Heavy	Light
Erodability	Low	High
Temporal variability in available soil moisture	Low	High
Mean monthly soil moisture availability	High	Low
Length of plant growing season	Long	Short

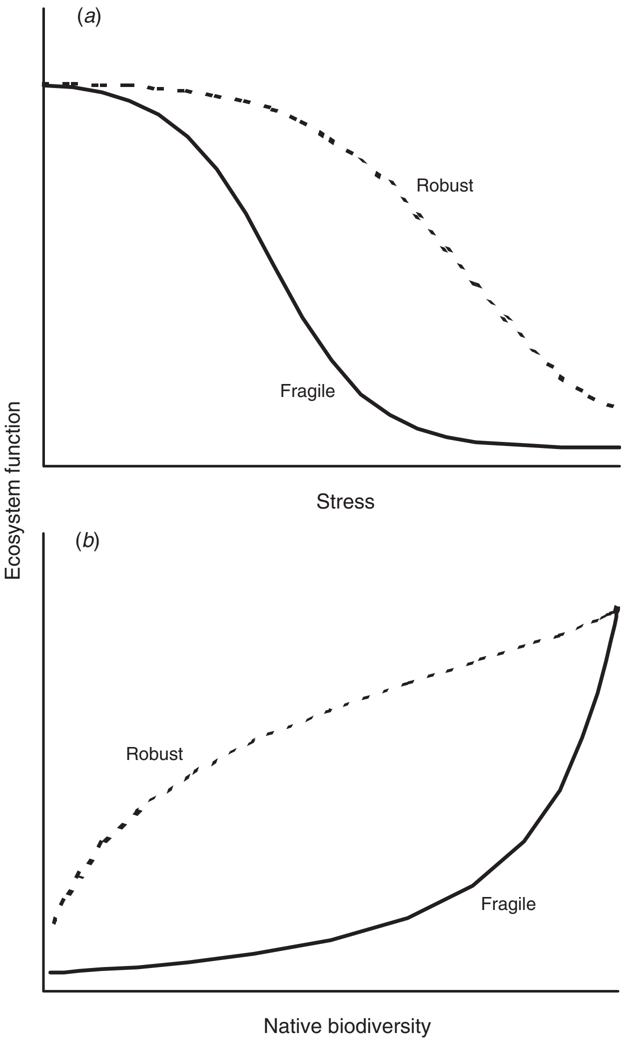

Fig. 2. Relationship between ecosystem function, stress, and biodiversity in fragile and robust landscapes. (*a*) As stress increases, ecosystem function declines as a result of loss in perennial grass cover, soil nutrients, and organic matter, and deterioration of soil structure. This is likely to be most rapid in fragile landscapes. (Derived from Tongway and Ludwig 2002.) (*b*) If the relationship between stress and native biodiversity is similar in most landscapes, then the dependency of ecosystem function on native biodiversity is likely to be greater in fragile landscapes.

landscapes, grazing-sensitive native plant species might be replaced with productive or invasive species at high grazing intensity, thus maintaining ecosystem functions (Fig. 2*a*).

There has been no detailed comparative work undertaken within Australia examining whether the relationship between native biodiversity and grazing differs between fragile or robust parts of the landscape. However, if the relationship is similar in both fragile and robust environments (assumed to be a non-linear negative function), in fragile landscapes the threshold beyond which there is long-term loss in production and functioning is likely to be very closely linked to native biodiversity (Fig. 2*b*). In robust landscapes, native biodiversity may contribute little to function or production (Fig. 2*b*).

These patterns suggest that alternative strategies need to be applied, depending on the fragility or robustness of environments. In those landscapes dominated by robust environments it is unlikely that successful biodiversity conservation can be achieved through direct integration with grazing systems without significant loss in productivity. In these cases, strategies solely targetted at biodiversity conservation should have greatest success. In these landscapes, many native fauna and flora are already restricted to small isolated remnants. Their persistence may be dependent upon enhancing biodiversity values in parts of the intervening landscape where the current focus is on production of agricultural products. In contrast, in less arable fragile landscapes, although grazing and associated management do lead to significant modification of vegetation, many native species persist under these regimes (Fensham 1998; Landsberg *et al.* 1999; McIntyre and Martin 2001). In this latter case, incorporating biodiversity conservation into production systems is also likely to have benefits for livestock productivity and the development of sustainable low-input systems.

Within landscapes, resource availability is both spatially and temporally patchy. Thus, within any one region, property or paddock, resilience and rates of ecosystem recovery following disturbance will vary through both space and time. The size and relationship of fragile and robust patches in a landscape will have considerable bearing on how an area could be managed to optimise productivity and biodiversity conservation. Where localised patches with high resource availability occur in a predominantly low resource landscape, these areas can be subject to intense grazing pressure in contrast to the surrounding landscape (McNaughton 1984). An alternative state can also be reached where high resource patches become dominated by large plant species when grazing pressure is light (van de Koppel *et al*. 1996). In the first case, palatable plant species and native fauna dependent on high resource patches are likely to decline as a result of severe grazing (Morton 1990). In the second case the lack of grazing will result in the loss of shade-intolerant plants (Tremont 1994). Production outcomes are also likely to vary, with maximum utilisation in the first scenario and under-utilisation in the second.

The above observations indicate that subdivision of the landscape into manageable units based on inherent landscape productivity and physical features (environmentally defined management 'cells') should be a first priority. This is in line with current trends in grazing management based on land capability (Simpson and Langford 1996). Across the landscape the timing and intensity of grazing should be varied to provide a wide range of grazing regime defined niches for grazing-susceptible and -tolerant species to persist and disperse. Management in some cells may focus primarily on productivity, whereas in other cells it may focus on particular products, weed control outcomes, or rare species and communities. Management should also aim to account for temporal variation in climate, disturbances, plant growth rates, and habitat quality.

Best management practices and thresholds

Factors such as loss of vegetative cover, soil erosion, weed invasion, soil structure decline, acidification, and salinity reduce the capacity of graziers to increase productivity in the long term (Crosthwaite and Malcolm 2000). As a result, there is a growing body of research that investigates how current grazing management practices can be modified to improve sustainability and long-term productive potential of grazing systems (for a review, see Kemp and Dowling 2000). Although native biodiversity may be integral for the development of low-input grazing systems and many of the processes influencing pastoral sustainability have contributed to biodiversity loss (see above), little research has been undertaken that compares the effects of different grazing strategies for maintaining or enhancing biodiversity in these ecosystems. In addition, experimental grazing trials undertaken in diverse native vegetation often fail to collect and/or present the data that would be necessary for assessment of their effects on native biodiversity. As a result, those management strategies that have been proposed tend to be based on anecdotal evidence and limited experimental results.

In this section we discuss current management strategies and more recently recommended strategies in relation to their potential effects on biodiversity and long-term livestock productivity. We consider the strategies at both the paddock and landscape scale as it is recognised that strategies that have negative effects on biodiversity, ecosystem function, or production within a paddock may be offset across a property or region.

Grazing management

Under the typical continuous grazing systems (otherwise known as set stocking), animals are run in medium- to large-sized paddocks (>40 ha) and stock are either moved when vegetation or livestock condition begins to deteriorate or are fed conserved fodder or grain. Livestock production from continuous grazing can be high, as animals are allowed to selectively graze. However, continuous selection of the most palatable plant species can lead to their loss from the grassland and a gradual deterioration in perennial grass content and productivity. For these reasons, strict continuous grazing is no longer regarded as a sustainable grazing regime (Kemp *et al*. 2000). Continuous grazing with sheep commonly results in the transport of nutrients from the broader paddock to localised areas in sheep camps (Hilder and Mottershead 1963), increasing the need for fertiliser additions and increasing the risk of soil acidification in the broader paddock. As discussed above, it is generally accepted that continuous grazing is a major cause of native biodiversity loss. However, if grazing densities are low, in large paddocks where there is considerable spatial distribution of vegetation, soils, and topography, grazing effect can be patchy. Although this can lead to over-utilisation in some patches with subsequent bare ground and weed invasion (Johnston 2001), under-utilisation can also occur (Willms *et al*. 1988), enabling localised persistence of grazing-intolerant species (Huntly 1991). This is less likely to occur in grazing systems where grazing densities are high and pressure is even (e.g. rotational grazing systems, see below).

Under many circumstances, livestock grazing has been shown to be compatible with the conservation of a high diversity of native grassland species (McIntyre and Lavorel 1994*b*; Fensham 1998). Thus, it has been argued that in diverse native grasslands, if past grazing management regimes are known, then these should be continued (Diez and Foreman 1997; Foreman 1999; Milne *et al*. 1999). This approach, known as '*status quo* management' (Diez and Foreman 1997), is based on the premise that past grazing management has led to the current composition of the grassy vegetation and maintaining that management strategy, if it is known, should maintain the current biota. Therefore, the underlying assumption is that the current composition of vegetation and fauna is stable and current levels of stress are not leading to irreversible changes in ecosystem function. However, concerns have been raised that such a management strategy fails to acknowledge that although many species may persist in grazed landscapes, their population growth rates may be negative (McIntyre and Lavorel 1994*b*; Kirkpatrick *et al*. 1995; Dorrough and Ash 2004). Indeed there is little reason to assume that these grazed ecosystems should be stable or that under current grazing management they are not likely to shift to alternate, less diverse states as a result of interactions with climatic conditions or other environmental factors (e.g. drought or fire). Indeed, much ecological research has emphasised the instability of grazing systems, particularly in semi-arid rangelands (van de Koppel *et al*. 1997).

Terrick Terrick National Park in the Victorian Riverina is often cited as an example of successful '*status quo*' management (Foreman 1999). Continuing past grazing pressure has been argued to be essential for maintaining the diversity of native annual forbs (Foreman 1997; Milne *et al*. 1999). Despite this, exclosures established for 7 years indicated that the abundance and diversity of native annual forbs were more closely correlated to climatic patterns than to grazing (Conway 2000), and the diversity of native perennial plants increased and bare ground declined under grazing exclusion (Foreman 1997; Conway 2000). These results suggest that reductions in grazing intensity and frequency could improve landscape function and with little or no negative biodiversity effect.

Variations in continuous grazing management, based on strategic resting of pastures, could benefit biodiversity and the persistence of dominant native perennial grasses. Strategic resting can enhance flowering, growth, and survival of plant species (Lodge and Whalley 1985; FitzGerald and Lodge 1997; Ash and McIvor 1998). It can also influence invertebrate biodiversity, with potential flow-on effects through to litter and soil structure and upon some of the food web processes concerning fungi and bacteria.

Decisions regarding the timing of strategic rest from grazing can be based on: (*1*) livestock condition, (*2*) vegetation biomass, and (*3*) plant phenology. Approaches based on livestock condition are considered to be the least sustainable, leading to long-term adverse effects on vegetation and soil structure and function (Jones 1996; Lodge *et al*. 1998). Quantitative approaches based on perennial grass biomass are now widely recommended and some general rules have been developed (Kemp *et al*. 2000). Biomass-based strategies can provide a more flexible framework than those based on phenology and do not require knowledge about individual plant growth and reproductive cycles. However, phenologically based strategies provide more accurate predictions about changes in plant community composition over time.

The basis of the phenological approach is to identify the critical periods of a plant's lifecycle, including flowering, rapid growth, or recruitment, when rest from grazing may be beneficial. For example, it has been suggested that spring and summer rest from grazing may promote and maintain species richness within diverse grasslands by allowing spring and early summer flowering species to set seed and replenish root reserves (Diez and Foreman 1997; Milne *et al*. 1999; Eddy 2002). Similar recommendations have been made to increase the cover of exotic perennial pasture species, particularly following drought events (Dowling *et al*. 1996; FitzGerald and Lodge 1997; Lodge *et al*. 1998). Although such a strategy should increase the flowering and seed production of native perennial plants, there has been no experimental work undertaken to determine whether this results in increased local plant diversity or higher probabilities of persistence for grazing susceptible plant species. No information is available on the benefits or otherwise for fauna, although theory would suggest that an increase in the density of flowering plants should promote beneficial invertebrates such as predators, parasitoids, and pollinators (Wratten *et al*. 1998).

Regardless of the lack of experimental research, some private landholders and government agencies are using spring resting to manage diverse native-dominated grasslands (J. Bellchambers, pers. comm.; P. Foreman, pers. comm.). Landholders utilising this strategy have remarked on an increase in flowering perennial species, but it is unknown whether this represents an increase in flowering of existing plants, an increase in species richness, or increased establishment of pre-existing species.

Spring resting could result in short-term reductions in livestock production and economic returns (Crosthwaite and Malcolm 2001). The feasibility of spring resting is also dependent on access to pastures that would not suffer from grazing pressure at this time (e.g. exotic perennial pastures). However, since perennial plant persistence and available summer and autumn forage should be increased by spring resting, it is possible that resting may have little long-term effect on the economic output of a farm, and may have positive outcomes during drought.

Spring resting could have some undesired ecological consequences. High rainfall and warm weather in spring can result in rapid plant growth, leading to rank pastures by the time the stock are introduced in summer (K. Calvert, pers. comm.). It is also possible that increased competitive effects in spring may reduce the abundance of some native plant species and, in sites with a high cover of exotic cool-season annual species, resting in spring may promote their abundance in the soil seed bank.

Gap formation in late summer and early autumn is a key precursor to invasion by exotic annuals and declines in native perennial species (Grigulis 1999; Dorrough 2001; Grigulis *et al*. 2001). Therefore, controlling grazing in late summer, during times of high soil moisture deficit, is also critical. One solution to enable utilisation of available summer forage and reduce competitive effects of grasses without enhancing weed invasion, is short-duration crash grazing (very high density of grazing animals over a short period of time) in mid-summer (T. Barlow, pers. comm.). This reduces competitive effects of dominant grasses and provides summer-growing grasses with enough time to regrow and fill large gaps prior to autumn germination of annual plants. Successful implementation of crash grazing strategies may be dependent on establishment of infrastructure similar to that necessary for cell or rotational grazing systems (see below).

Alternative grazing strategies requiring intensive management and infrastructure such as intensive rotational grazing, have been argued to provide increased benefits for biodiversity, productivity, and sustainability (Norton 1998). Rotational grazing generally describes a controlled grazing system using high stocking rates of livestock moved among many small paddocks with the intention that grazing time, and therefore grazing pressure, in any one area is minimised and rest is maximised (McCosker 2000). Rotational grazing has been shown to lead to an improvement in the utilisation of pasture, a reduction in selective grazing, and a maintenance of pasture cover (Earl and Jones 1996; Sanford *et al*. 2003). Observations of high plant species diversity in intermittently grazed stock reserves (McIntyre and Lavorel 1994*b*) have also been the basis of suggestions that rotational grazing strategies might be more beneficial for biodiversity than modified set stocking (Barlow 1998). However, grazing frequency in stock reserves is typically lower than in commercial rotational grazing systems and therefore it is unknown whether similar outcomes would be observed.

On the Monaro Tablelands of NSW, rotational grazing after 3–5 years was found to have no effect on either small- or large-scale plant species richness, although it did reduce spatial heterogeneity in vegetation structure (Bruce 1998). Spatial structural variation is important for plant species co-existence in grassland communities, and grazing-susceptible and twining or climbing plant species that co-occur with tussocks may be less likely to persist if structural heterogeneity declines. Low structural heterogeneity may also have negative consequences for fauna utilising fine-scale vegetation mosaics such as that created by patchy shrubs and tussocks and inter-tussock spaces (e.g. small- to medium-sized strictly terrestrial vertebrate fauna and terrestrial invertebrates). However, rotational grazing could increase between-paddock heterogeneity and this may favour some of the more mobile fauna species.

Other than limited research on dominant grass species, as far as we are aware no research has yet been published examining the effects of rotational grazing systems on native biodiversity in diverse temperate grasslands or grassy woodlands. Because of the potential for these grazing strategies to provide for both production and conservation goals, there is an urgent need for such research to be undertaken.

Vegetation structure

In Australian grasslands and rangelands, palatable perennial grasses are the main functional group driving ecosystem function (Hodgkinson 1992; Freudenberger *et al*. 1997; Kemp and Dowling 2000). These species form the vegetation matrix, structuring fauna habitat and the availability of resources in space and time for inter-tussock plant species. Retaining the dominant palatable perennial grasses should therefore be a key management goal for both sustaining productivity (Kemp *et al*. 2000) and biodiversity (McIntyre *et al*. 2000) in temperate grazing lands.

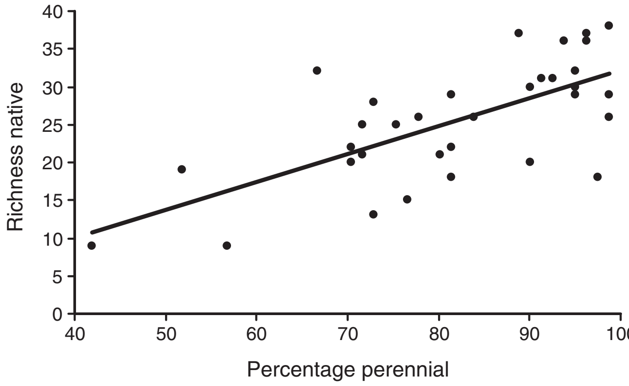

Fig. 3. The relationship between the richness of native plant species and the percentage of eighty-one 0.0011-m^2 quadrats occupied by perennial grasses in thirty-two 390-m^2 sites ($R^2 = 0.46$, $F_{1,31} = 26.8$, $P < 0.001$). All sites were located within grazed grasslands on the Monaro Tablelands, NSW, and excluded areas that had been cultivated and sown to exotic perennial grass species. (J. Dorrough, J. Ash, and S. Bruce, unpublished data.)

Maintaining perennial grass cover and increasing standing biomass of dominant perennials can result in increased microbial biomass and reductions in runoff, groundwater discharge, and nitrate leaching (White *et al*. 2000). The implications of an increased cover of dominant perennials for native biodiversity are little known, although should this involve an increase in the cover of native perennial plant species, some positive gains are likely (Fig. 3). Alternatively, an increase in the cover of exotic perennial grasses is more likely to have negative outcomes (Morgan 1998*b*).

From a biodiversity perspective, it has been recommended that native perennial tussock-forming grasses should be no less than 60–70% of the ground cover (McIntyre *et al*. 2000). From a production point of view, perennial grass biomass generally needs to be maintained above 0.5–1.5 t dry matter/ha to prevent death of grasses in drought and minimise weed invasion (Kemp *et al*. 2000). Optimal production value will be obtained if the sward is even and kept at active stages of growth. In contrast, for biodiversity, the structural heterogeneity of vegetation should be maximised (Fig. 4) and tall and medium tussock-forming species should dominate the perennial grass layer. Although there may be minor production advantages of a variable vegetation structure dominated by tall tussocks (e.g. protection of lambs), diversifying the vertical structure of the pasture can result in a reduction in the utilisable forage, particularly in sheep enterprises.

Long-ungrazed vegetation has low vegetation heterogeneity, resulting in low habitat value and high competition for space and light for subordinate forb species (see above). Light to moderate grazing intensity reduces total

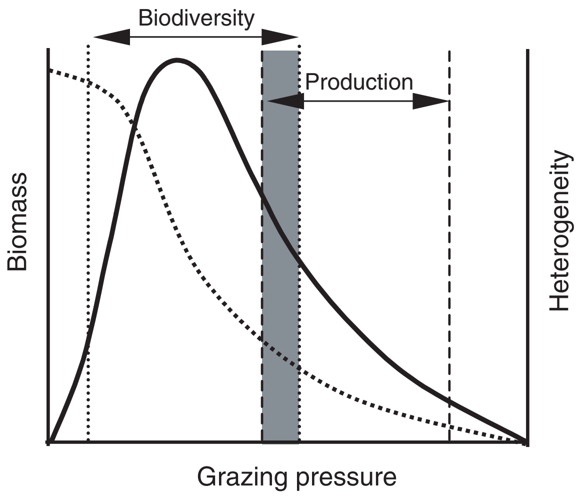

Fig. 4. Hypothesised relationship between vegetation biomass (dashed line), structural heterogeneity (solid line), and grazing pressure under a set-stocked grazing regime. Optimal conditions for biodiversity are likely to occur when biomass and heterogeneity are high, and in contrast, optimal livestock production is likely to be achieved when heterogeneity is low and biomass is moderate.

biomass, but increases structural heterogeneity (Fig. 4). With further increases in grazing pressure, vegetation biomass declines and heterogeneity reaches a peak and then begins to decline. Under rotational grazing regimes, heterogeneity may decline at lower grazing pressure. Fig. 4 indicates that production may need to be compromised if total biomass and vegetation structural heterogeneity are optimised. Research is required to parameterise these relationships.

Fertilisers

Sustainability in agroecosystems implies that there is limited use of non-renewable resources. Even so, the addition of non-renewable fertilisers is an important strategy in increasing native and exotic pasture production and replacing nutrients lost through leaching, stock movement, and the export of commodities (Cayley *et al*. 1987, 1999). At present, little is known about the effects of interactions between nutrient additions and grazing for native biodiversity, although negative effects are expected. In Australia, increased soil fertility in conjunction with disturbance or grazing can lead to high rates of invasion by annual plant species and declines in dominance by native perennial grasses (Trumble and Fraser 1932; Hobbs and Atkins 1988; Hobbs and Huenneke 1992; Robinson *et al*. 1993). Similar patterns have been observed along natural fertility gradients (Bridgewater and Backshall 1981; McIntyre and Lavorel 1994*a*). Fertiliser addition can also facilitate higher livestock grazing densities (Cayley *et al*. 1999), with subsequent negative effects on the persistence of grazing-susceptible species. Increased fertiliser can also decrease the number of soil mesofauna, ants and spiders (Siepel *et al*. 1989; Di Giulo *et al*. 2001). The threatened Golden Sun Moth (*Synemon plana*) inhabits grasslands in south-eastern Australia that have a cover greater than 40% of *Austrodanthonia* spp. and low levels of phosphorus (O'Dwyer and Attiwill 1999). The addition of fertiliser to these grasslands increases weed cover, especially the introduced perennial ryegrass (*Lolium perenne*), and reduces the cover of *Austrodanthonia* spp., rendering the sites unsuitable for *S. plana* (O'Dwyer and Attiwill 1999).

Increased nutrient loss through surface run-off and deep drainage can also occur as a result of fertiliser application (Ridley *et al*. 2003). This can have effects on in-stream biodiversity (ANZECC 2000) and alter plant community composition within adjacent remnant vegetation (Hobbs and Huenneke 1992). Ridley *et al*. (2003) state that 'planting wide buffer strips to protect waterways is an essential component of more environmentally sustainable farming systems'. Similar consideration should also be given to isolated remnant vegetation adjoining fertilised pastures.

Grassland replacement

Replacement of native grasslands with a mix of exotic grasses and legumes in conjunction with high rates of fertiliser is a recommended practice for improving animal production in much of temperate Australia (Malcom *et al*. 1996; Cayley *et al*. 1999). However, in many cases, sown pastures have degraded over time, becoming increasingly dominated by annual grasses and broad-leafed weeds (Kemp and Dowling 1991; Garden *et al*. 2001). Pasture replacement strategies also have negative effects on the persistence of many species of native fauna and flora (Dorrough and Ash 1999; Fairfax and Fensham 2000; Dorrough 2001; Hamilton 2001; McIntyre and Martin 2001). Nitrogen fixation by introduced legumes is also instrumental in increasing annual grass and broad-leafed weed biomass and soil acidification in pasture. Despite this, annual legumes are still considered to play an essential role in sustainable grazing systems (Kemp *et al*. 2000), although perhaps this role should be examined.

There has been recent interest in the development of native grass pasture species for use on less fertile, saline, or acidic soils (Johnston *et al*. 2001; Mitchell *et al*. 2001). However, any replacement strategy, whether it uses native or exotic grass species, is likely to lead to biodiversity and sustainability effects (Jones 1996). A better approach is to develop grazing strategies that maximise the benefit of retaining and recruiting persistent native species.

It has been argued that intensively utilised improved pastures (i.e. high input, exotic species) can play a role in conserving biodiversity by off-setting production losses from less intensively utilised areas of a property (Crosthwaite and Malcolm 2000). In addition, run-down introduced pastures have both low biodiversity and production values (Crosthwaite and Malcolm 2000) and may have additional adverse environmental effects (e.g. nitrate leaching, weeds, and soil loss) that can be reduced through appropriate intensification (Cayley *et al*. 1999; White *et al*. 2000). Maximising production in certain areas of a property can be further justified by considering the cost of restoring biodiversity in fertile productive areas compared with enhancing or maintaining biodiversity in areas with lower fertility and/or a less intense history of land use.

Fire

Prior to European settlement, fire is thought to have been a major factor structuring grassland and grassy woodland communities (Lunt 1997; Lunt and Morgan 2002). Many of the more significant species-rich grassland remnants in south-eastern Australia have a history of regular burning and little or no livestock grazing (Stuwe and Parsons 1977). Fire plays an important role in maintaining the tussock and inter-tussock structure in native grasslands, enabling co-existence of a diversity of small native forb species (Lunt 1991; Morgan 1998*a*; Morgan and Lunt 1999). Since European settlement, fires have generally been suppressed, although burning has been used in conjunction with grazing to reduce dominance by large tussocks, particularly in drainage lines. The effect of fire on invertebrates is dependent upon timing, intensity, and extent of the fire, and on the length of the vegetation recovery period. The recovery response depends upon feeding habit (herbivore, predator, detritivore, fungivore), distance to source and mode of dispersal, and whether they survive the fire by going underground (Warren *et al*. 1987). The activity of some invertebrates, such as ants, increases dramatically after fire (Andersen and Yen 1985).

Because many species-rich grasslands have had a long history of regular burning, fire has often been used in grazed grasslands resumed for conservation. However, in sites with a long history of grazing, imposing a management regime of regular burning can quickly result in invasion by exotic plant species (Lunt 1990; Lunt and Morgan 1999). Contrary to this, recent trials in the Victorian Riverina have shown that a combination of fire and grazing can be used successfully to enhance native plant abundance and reduce weed cover in previously cropped and grazed grasslands (I. Lunt, pers. comm.). Further research is required to investigate the role that fire may play in maintaining biodiversity in grazed grasslands and grassy woodlands. If beneficial effects can be shown, trials are then required to determine how fire could be used within a commercial grazing enterprise without detrimental effects on production.

Invertebrate pest control

Invertebrate pests consume foliage, roots, and sap, transmit plant pathogens, and predispose plants to pathogens. This leads to loss in pasture quality, which affects animal production. There is a lack of information to evaluate pest effects on productivity because of a lack of records on pest occurrence (time and space), density of pests, level of damage, and a lack of methods to relate losses in pasture quality and quantity to livestock production (Madin 1993).

Invertebrate pests can be native species that have responded to better quality food associated with managed pastures (e.g. armyworms, cutworms, wingless grasshopper, redheaded pasture cockchafer, blackheaded pasture cockchafer, and the black field cricket) or can be introduced species without their natural enemies (aphids, sitona weevil, red legged earthmite, lucerne flea, and white snails) (Cullen 1993; Panetta *et al*. 1993). Different species are favoured by different factors: height of pasture, botanical composition, occurrence of bare patches of ground, amount of plant cover, proximity of alternative food plants, and habitat such as trees (Panetta *et al*. 1993).

Chemical applications are commonly used to control pests, but farmers are increasingly preferring to minimise or avoid the use of insecticides, and allow control by natural enemies or through manipulation of grazing regimes, pasture composition, and cropping frequency (Panetta *et al*. 1993). The effects of such management activities upon pest populations and non-target organisms are still largely unknown (Panetta *et al*. 1993). Chemical insecticides can be expensive, cause adverse damage to non-target animals, and may leave chemical residues in the soil and produce. The need to apply insecticides has to be ascertained first, and any application has to be properly timed. Some farmers apply fertilisers to compensate for pasture losses to pests. Although this may be more cost-effective than applying insecticides, this strategy may exacerbate existing problems or promote other insect pests (McQuillan 1993).

Tillage is also sometimes used to reduce numbers of soil insect pests, but zero tillage can increase populations of detrivorous and predaceous invertebrates (Robertson *et al*. 1994; Horne and Edward 1998) and generally does not lead to increased numbers of soil insect pests (Wilson-Rummenie *et al*. 1999). As a result, better long-term pest management outcomes might be achieved in zero-tillage systems.

Landscape-scale thresholds

The direct effects of grazing and other land management practices clearly play an important role in the persistence of native biodiversity within temperate grasslands and grassy woodlands (see above). Landscape processes, primarily the extent and spatial pattern of remnant and modified vegetation, also influence patterns of native biodiversity, ecosystem processes, and pasture production within agricultural landscapes (Loyn 1987; Young *et al*. 1996; Davies and Margules 1998; Walpole 1999; McAlpine *et al*. 2002). Thus, it is necessary for grazing management to not just consider the requirements of biodiversity within individual paddocks, but also the context of the paddocks within the property or the region.

Theory indicates that the effect of landscape-scale habitat loss on species persistence is non-linear (Koopowitz *et al.* 1994). Recent research has focussed on attempting to quantify these theoretical models to determine whether there are clear landscape-scale thresholds of vegetation cover below which there is an irreversible and disproportionate loss of biodiversity (Bennett and Ford 1997). Results of a study on birds in the Northern Plains region of Victoria, an intensive agricultural region, showed that below 10% tree cover, bird species richness dramatically declined (Bennett and Ford 1997). Many other species appear to have also declined in abundance, suggesting that future extinctions may occur even in the absence of further clearing (Loyn 1987). In some areas that were once heavily wooded, tree cover is now well below 10%, emphasising that revegetation and protection of existing vegetation will be essential for the persistence of many species (Bennett and Ford 1997).

Based on both empirical and theoretical research on habitat clearance, it has been recommended that native vegetation cover should be retained over 50% of the landscape (McAlpine *et al.* 2002), with lower targets (approx. 30%) suggested at the property scale (Walpole 1999; McIntyre *et al.* 2000). Below these thresholds there is likely to be significant loss of biodiversity, disruption of ecosystem processes, and loss of pasture production. In most landscapes in south-eastern Australia, native grassland and grassy woodland vegetation cover has already been reduced well below these thresholds (Table 1) and broad-scale restoration of native vegetation appears necessary. The most likely focus for restoration will be pastures where native grasses still persist but other components of their natural system have been lost through grazing, nutrient application, and disturbance. However, there has been no attempt to restore these vegetation types at broad-scales and success at small-scales has been poor (Morgan 1999). Research is required to determine: (*1*) the composition, extent, and spatial arrangement of restored native vegetation desirable for broad-scale nature conservation objectives; (*2*) the technical aspects of broad-scale grassy ecosystem reconstruction; (*3*) the economic costs of broad-scale revegetation at farm and regional scales.

Conclusions

It has been argued that the conservation of biodiversity in agricultural regions is hampered by mismatches between the scales of biological processes and units of management as well as regional and local policy and planning structures (Briggs 2001). This review has shown that insufficient knowledge of key ecological processes is also a major factor in preventing the integration of biodiversity conservation with agriculture. At present there are few empirical data examining the effect of different grazing strategies (i.e. season of rest, rotational systems) on biodiversity in remnant grassy vegetation in south-eastern Australia. The limited information indicates that year-round set-stock grazing, fertiliser application, and pasture replacement have negative effects on the persistence of native grassland species. Intermittent grazing strategies are more likely to have long-term biodiversity and production benefits, particularly if they can be manipulated such that vegetation heterogeneity can be maintained through space and time.

Not only do we lack essential information on management effects for biodiversity but, in many cases, there is a lack of knowledge of biodiversity itself. Current knowledge of biodiversity in grasslands and grassy woodlands of south-eastern Australia is primarily restricted to vascular plants, threatened species of vertebrates, and pest invertebrates, although this too is limited. Much of our current knowledge is primarily descriptive. There is a dearth of knowledge regarding certain key elements of biodiversity such as invertebrates (Yen 1999) and non-vascular plants, their ecological functions, and the interactions between all these elements. A certain level of understanding of the roles of these biodiversity elements is necessary before we can determine whether proposed best management practices will lead to long-term sustainability.

Increasing the cover and abundance of perennial grasses is one of the most important strategies for ensuring sustainability. The majority of research in temperate Australia has only focussed on the role of dominant perennial grasses; however, grazing strategies to favour these species may also have benefits for other aspects of biodiversity, particularly if the dominant grasses are native. Conflict is most likely to arise with regard to vegetation structure. Cover and structure optimal for the persistence of perennial native grasses and maintenance of maximum livestock production will not always coincide with optimal co-existence of native fauna and flora.

Considerable collaborative research, extension, and policy development are required before a sustainable and biodiverse grazing system can be achieved in Australian temperate grassy ecosystems. It is essential that agronomists and ecologists develop joint projects to tackle many of the issues raised in this paper. In particular, effort is required to identify the links between long-term sustainability in production and biodiversity. It will also be necessary to re-train extension staff to ensure that they are aware of both biodiversity conservation and sustainable grazing management strategies. Government, industry, and universities should actively foster joint research, development, and application and this may require new approaches to both the structure and funding allocation of our agricultural research institutions.

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