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Diversity and Sustainability in Grassy Eucalypt Ecosystems

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Preface

Grassy woodlands, characterised by an overstorey of eucalypts and a grass-dominated understorey, form a major ecosystem of eastern Australia, stretching from northern Queensland through to the riverine plains and foothills of northern Victoria.

The plant communities found in this ecosystem change significantly as we travel from south to north, but many of the ecosystem processes and functional relationships, and hence principles for sustainable management, are similar.

In the southern part of its range, the grassy woodland ecosystem has been substantially modified through clearing and grazing for agriculture; in many areas intensification of land use has led to severe fragmentation of the ecosystem, to the stage where the long-term viability of the remnants is in doubt. As we travel northward, land use becomes generally less intense and, in northern New South Wales and much of Queensland, although the ecosystem has been (and continues to be) cleared and modified for agriculture, it could best be described as forming a variegated rather than fragmented landscape.

This report examines how general principles of sustainable land use might be applied to grassy woodlands, with a particular focus on the extensive regions in Queensland where the ecosystem is still relatively intact. It particularly examines the importance of conserving biodiversity in order to maintain healthy, fully-functioning ecosystems and a sustainable resource base for productive use, and as an intrinsic goal. It examines some of the crucial ecological processes important in grassy woodlands and the factors that affect them.

The report then goes on to examine current land use and management practice, and identifies their relative sustainability in terms of likely effects on ecosystem function and health. The report also flags and discusses a range of indicators that land managers, both public and private, might use to assess the effects and relative sustainability of current practice; an important aim is that these indicators should support a process of adaptive management, where management practice continually evolves and is improved by assessing its impacts on key characteristics of the ecosystem.

This report was developed by the authors as a scoping exercise to identify priorities for further research and development aimed at improving the ecological sustainability of, and the natural resource base for, the extensive grazing industries of northern Australia. A major program of research to examine the economic profitability and the ecological sustainability of improved land and grazing management has been established as a result of this scoping study. It is jointly funded by the Meat Research Corporation and LWRRDC under the MRC's North Australia Program. Many of the concepts and principles outlined in this report could also be applied to grassy woodland ecosystems and their management elsewhere in Australia, and to other ecosystems.

Phil Price
Executive Director
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1 Objectives and Scope of the Paper

1.1 Background

This paper discusses the broad issue of ecological sustainability in the context of the grassy ecosystems of eastern Australia. These ecosystems are of particular interest because of the progressive land-use changes that may damage their health.

Grazing by domestic livestock is widely considered to be the management factor of most importance to grassy ecosystems. It is certainly the most widespread and persistent factor, and has been operating continuously on these communities since the nineteenth century (Shaw 1957). However, other technologies and factors are increasingly influential in grazing management, including livestock breeding, feeding technologies, cultivation, sown species, fertilisation, tree clearing, and changed fire regimes. These processes are inextricably interrelated to each other and to grazing intensity. For example, hardier cattle breeds and supplementary nutrition have enabled grazing pressure to be increased in Queensland pastures, particularly during drought (Scanlan *et al*. 1994). Sown pasture technologies, often involving a suite of disturbances (eg. cultivation, fertiliser, herbicide) increase potential animal production and generally lead to higher stocking rates. All these processes are aspects of the general phenomenon of land-use intensification: efforts to exploit fully the productive capacity of the ecosystems. The general tendency in Australia is towards intensified and more-restricted land-use options (Cocks and Walker 1994).

Examples of unsustainable management of the grassy ecosystems are most prominent in southern Australia, where land-use intensification is most advanced. There is a widespread perception in Queensland that many of these problems are not relevant to that State. In this paper we argue that although land use is not yet as intensive in Queensland as in southern Australia, the processes of land-use intensification are in place, and the risks of unsustainable management are similar.

1.2 Scope of the paper

In this report, literature is selectively reviewed in order to: i) identify the various elements of ecological sustainability; ii) assess potential indicators of ecological sustainability; and iii) identify the relationship between the indicators and productivity. We discuss the concepts in the context of the ecology of, and land use in grassy ecosystems, and identify potential areas for research, comparing various methodologies. The extensive sub-tropical grassy ecosystems in Queensland (the review region) are the focal point of the discussion. These are most commonly represented as eucalypt woodlands with a grass understorey. Despite this focus, many of the principles and issues raised are relevant to a wider range of communities, and extend beyond the review region.

We draw on the rangeland literature and use the term ‘rangeland’ in referring to areas where domestic stock grazing of natural plant communities is the dominant land use. This definition differs from that used in the Draft National Strategy for Rangeland Management (NRMWG 1996), which included only arid and semi-arid regions and some tropical savanna regions—the least intensively used parts of the country. However, Burrows (1980) reported that 96% of Queensland rural holdings were native pastures and no single statistical division had less than 80% of its area under native pasture. The reality is that rangeland use intergrades with more-intensive land uses (both temporally and spatially) and the potential to intensify land use exists in most regions, even the arid and semi-arid zones, where nutrient supplements (Bastin 1989) and sown pasture species are widely used (Anon. 1990; Humphries *et al.* 1993; Lonsdale 1994).

Overriding all definitional boundaries is the particular challenge for R & D to develop management systems that enable diverse, native ecosystems to coexist with pasture technologies—that is, the aim is to maintain productivity, without compromising ecological sustainability. The charter to develop technologies and management systems that are consistent with the goals of ecological sustainability is the responsibility of research institutions and their funding agencies.

2 Sustainability—Concepts and Uses

2.1 Brief history of sustainability as a concept

Development of the concept

References to unsustainability stretch back to ancient Greece, to nineteenth century Australian agriculture, and to the US and UK in the early twentieth century (Reeve 1990). However, the concept and use of the term sustainability has been evolving most rapidly over the past few decades. It continues to be adapted within a number of disciplines including ecology, resource economics, social theory and development studies. Definitions have been reworked and reiterated so many times that some see it as no longer useful (eg. Gatto 1995). However, others may see it as a concept that is nascent, requiring better definition and measurement or, alternatively, a better understanding of its implications (Shearman 1990).

In a review of sustainability in an Australian context, Dovers (1990) identified five major concerns that led to the emergence and evolution of the concept:

- use and degradation of resources at a rate that will compromise their future availability;
- accumulation of wastes to the extent of compromising future use of resources;
- reduction of biological diversity, reducing non-human life and its future use by humans;
- unsatisfactory models of human population growth and development that lead to socially undesirable situations (eg. crowding, overconsumption); and
- inequitable models of development.

Sustainability has evolved as an integrating concept which recognises the interrelationships between these issues (Dovers and Norton 1994). Milestones in the recent development of sustainability as a concept include a book discussing the issues facing Australia (Birch 1975). An Australian national conservation strategy was published in 1984 (Anon. 1984); this was the first of a series of reports on sustainability produced at State and regional levels.

At the international level, the Brundtland Report (World Commission on Environment and Development 1987) has been a major force in developing the concept of sustainability. So too was the United Nations Conference on Environment and Development in Rio de Janeiro in 1992 (United Nations 1992) which maintained the integrating agenda incorporating ecological (species and ecosystems), economic (population issues and resource use), and social issues (food security, peace and urbanisation) among other concerns (Dovers and Norton 1994). At that time, the Australian Government was developing its National Strategy for Ecologically Sustainable Development (Anon. 1992) and the Food and Agriculture Organization of the United Nations was preparing an international framework for evaluating sustainable land management (Smyth and Dumanski 1993). Together with many other reports, these have reflected, and further publicised, concern about the effects of humans on the biosphere and the long-term viability of our activities.

Disciplinary approaches to sustainability—internal contradictions

The concept of ecological sustainability is frequently integrated with other applications of the term ‘sustainability’, most notably social sustainability, economic sustainability, socio-economic sustainability and sustainable development (eg. Shearman 1990; Goldman 1995; Young 1995). These tend to reflect disciplinary approaches rather than sharply defined categories of sustainability, and many discussions and applications of sustainability combine one or more of these considerations such as economic and ecological sustainability (eg. Faeth 1993; Hitzhusen 1993; Fievez *et al.* 1994), social and ecological sustainability (eg. Dovers and Norton 1994; Young 1995); social, economic and ecological sustainability (World Commission on Environment and Development 1987; United Nations 1992; Conway 1993; Farshad and Zinck 1993; Smyth and Dumanski 1993; Goldman 1995). Some papers restrict discussion to ecological sustainability in the context of human land-use, for example in forestry (Norton and Mitchell 1994) and agricultural land-uses (Kleinman *et al.* 1995).

Dovers and Norton (1994) considered ecological sustainability to be “…a state where exploitation was undertaken in an ecologically conservative manner within apparently safe limits flowing from an integrative assessment of current and potential threats”. Ecological sustainability tends to be regarded as a very important component of the broader concept, but is more explicitly discussed in relation to the exploitation of natural ecosystems, for example in relation to forestry and grazing (eg. Michalk and Kemp 1993; Cairns and Meganck 1994). At this scale issues such as maintenance of biodiversity and ecosystem processes are more tangible. These concerns extend to national and global concerns over biodiversity and ecosystem processes (including air and water quality), and lead to concerns about the general intensification of land use and the need for conservative development (Cocks and Walker 1994).

Economic sustainability tends to be phrased in terms of ‘sustainable development’, reflecting a preoccupation with socio-economic rather than ecological issues. Economic sustainability also has a variety of definitions, depending on the economic school from which it derives. Even within this single disciplinary approach, Gatto (1995) has found internal inconsistency in definitions. He links this to the impossibility of a continuous increase of capital productivity and the non-substitutability of at least some natural resources. Other perspectives on economic sustainability encompass ideas such as:

- “sustainable income” (Fievez *et al.* 1994);
- “sustainable economic growth” (Pearce *et al.* 1989; Hitzhusen 1993); and
- the acceptability of constrained deterioration of the land base if productivity can be increased to compensate for lost income (Fievez *et al.* 1994).

These concepts might conflict with definitions that stress ecological sustainability over long time-scales (Dovers and Norton 1994). They also challenge the view that world resources are being so depleted that they cannot sustain greater levels of exploitation (Farshad and Zinck 1993). Goldman (1995) argued that ecologically sustainable management (in the sense of being stable) is not socially sustainable, as may be seen from the extent to which rapacious resource users have displaced cultures which used resources more moderately. Schaller (1993) recognised two extreme views about sustainability—first that it can be achieved by fine-tuning conventional agriculture; second that it will not be attainable until conventional agriculture is redesigned. These and other perspectives tend to stretch the capacity of the term ‘sustainability’ to embrace all viewpoints. They also illustrate the extent to which sustainability concepts are value-laden and highly dependent on their context.

Sustainability in agricultural (cropping) areas

In the literature, concepts of sustainability are most often discussed in the context of the management of agricultural systems (Yunlong and Smit 1994). This usually refers to systems involving cropping, although the terminology is not always clearly defined. Users of the term ‘agricultural sustainability’ may implicitly include rangelands in their discussions, but the two systems, because of their differences, generally need to be considered separately. Cropping is the exploitation of a simplified, largely human-created vegetation, whereas rangeland activities exploit a largely natural (although sometimes human-adapted) diverse vegetation.

Hansen (1996) described four ways in which the term agricultural sustainability had been interpreted and defined by authors:

- Sustainability as an ideology—this incorporates the ethical and philosophical dimensions, including the concept of stewardship and conservative farming.
- Sustainability as a set of strategies—general and specific techniques for maintaining the land resource.
- Sustainability as the ability to fulfil a set of goals—identification of various social, economic and ecological states that could be achieved by sustainable management.
- Sustainability as the ability to continue—the maintenance of production over time.

Thus it can be seen that the use of the term in relation to agriculture can be as complex as the more generic concept, as both uses encompass ecological, economic and social issues. At this general level, approaches to agricultural sustainability are not obviously distinct from those relevant to rangelands. However, where cropping lands are considered, sustainability is most concerned with maintaining the soil resource (ESDWG 1991; Hamblin 1992; Farshad and Zinck 1993; Smyth and Dumanski 1993; Jansen *et al.* 1995) for agricultural productivity (eg. Pankhurst 1994; Kleinman *et al.* 1995). There is also concern for the on-site and off-site effects of soil and nutrient loss (Smyth and Dumanski 1993; Pesek 1994). Less frequent concerns are about the minimisation of chemical residues and maintenance of crop diversity (Farshad and Zinck 1993; Reeve 1990; Pesek 1994; Jansen *et al*. 1995), and maintenance of “system resilience’ (Conway 1993; Hansen 1996).

More contradictions—stability in agricultural vs. rangeland systems

Describing agricultural sustainability in terms of an ecosystem’s response to perturbation is intuitively appealing (eg. see Altieri *et al.* 1983; Greenland and Szabolcs 1994; Hansen 1996), and elements of stability are an inherent part of the sustainability concept (Conway 1993). However, Goldman (1995) found there was little empirical evidence for, or exploration of, `this phenomenon. There is also some confusing use of the terminology as, depending on the author, the term ‘stability’ may or may not be distinguished from ‘resilience’. The terminology of Begon *et al.* (1990) is followed in this paper:

> The stability of a community (or ecosystem) measures its sensitivity to disturbance (perturbation). …Stable communities are by definition those that persist. There are various types of stability: Resilience describes the speed with which a community returns to its former state after it has been perturbed and displaced from that state. Resistance describes the ability of the community to avoid displacement in the first place.

In agricultural ecosystems, stability may be highly dependent on the way that the system is managed, and human intervention is seen to be able to accelerate recovery of the system after a perturbation such as drought (Lal 1994; Goldman 1995). In rangeland systems, the pre-existing vegetation type is the most important factor in the production system, and this perception dominates management considerations. The inherent resistance or resilience of the system to perturbation is important. Management involves correctly estimating how much perturbation the ecosystem will resist, or be resilient to. Thus agriculture can enhance stability through management, while the management of grazed natural communities needs to be sensitive to their inherent levels of stability. Management to enhance stability in an agricultural context may result in loss of the natural community, ie. loss of the resource base of rangelands.

Defining sustainability in the broadest sense

We have been critical of attempts to define sustainability by recognising the subjectivity and internal contradictions inherent in the concept, but the question remains—how to pursue a concept that is fundamentally important to the management of our natural resources? At the broadest level, the definition proposed in the Brundtland report is quite satisfactory:

> Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs.
>
> (World Commission on Environment and Development 1987)

However, while it is broad enough for general direction, some have found it too general for operational guidance (eg. Dovers and Norton 1994; Gatto 1995). The Ecologically Sustainable Development Working Group, in its final report on agricultural sustainability (a report that covers both cropping and rangeland regions) extended the concept to a set of general principles for ecologically sustainable development as follows:

- improvement in material and non-material well-being;
- intergenerational equity;
- intragenerational equity;
- maintenance of ecological systems and protection of biodiversity;
- global ramifications, including international spillovers, international trade and international cooperation; and
- dealing cautiously with risk, uncertainty and irreversibility.

(ESDWG 1991)

In the following year, the Commonwealth Government used the same principles to derive the following definition of ecologically sustainable development:

> ... using, conserving and enhancing the community’s resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased.
>
> (Anon. 1992)

We will adopt these as appropriate goals for sustainability, recognising as Gatto (1995) did, that there is a variety of views representing different priorities and optimisation criteria. In reality the path of development will be a negotiated compromise between different parties. The views of natural scientists are no more value-neutral than those of other parties (Reeve 1990; Lélé and Norgaard 1996). Ecologists concerned with sustainability and protection of biodiversity tend to have priorities focused in those directions. Our operational principles will thus draw more strongly on concepts of ecosystem health than on the all-encompassing notions of sustainability. These principles and potential indicators are discussed in Sections 2.2, 5.1 and 5.2.

2.2 Sustainability in rangelands

Are sustainability definitions adequate for rangelands?

We have noted some aspects of sustainability which may set rangelands apart from agricultural and highly managed pasture lands, but published criteria for defining sustainability tend to group all grazed ecosystems together. The Ecologically Sustainable Development Working Group did not offer a different definition of sustainability for those regions in which rangeland use was recognised (ESDWG 1991). The draft National Rangelands Strategy did not attempt to define sustainability in rangelands but did recognise that different regions may need different management to achieve sustainability (NRMWG 1996).

Discussions on sustainability in rangelands may focus on the specifics of the ecological impacts and land-use management rather than on tight definitions (eg. Morton and Price 1994). Michalk and Kemp (1993) saw sustainability in grazing systems (both native and sown) as a matter of nutrient cycling, the movement of soil and water and the removal of nutrients. Hutchinson (1992) specified a range of goals for sustainable grazing enterprises:

- to conserve the soil resource and its physical structure;
- to enhance soil fertility, nutrient cycling, and the retention of nutrients;
- to sustain nutritious and productive plants;
- to optimise forage use within the constraints of responsible management;
- to monitor and analyse changes in plant and soil resources;
- to develop grazing enterprises that are resilient and adaptive to market change;
- to assess the long-term benefits and costs of sustainable resource use/development; and
- to develop long-term strategies to promote sustainable and environmentally responsible development.

These goals further highlight the inherent contradictions in the sustainability of grazed natural ecosystems (rangelands) as against intensively managed pasture (or cropping) lands. Previously we noted that the increase of system resilience through management could be inconsistent with the maintenance of the natural ecosystems used as rangelands. Furthermore the goals of enhancing soil fertility and of providing nutritious, productive plants which could sustain production, may not always be consistent with environmental health as measured by other criteria. Such increases in fertility, arising from the more intensive use of rangeland, may lead to unsustainability. This is further discussed in Sections 3.3, 3.4 and 4.4.

Ecological sustainability—some general issues for rangelands

Livestock grazing and its effects are the primary concerns of sustainable rangeland management. Native and feral animals may also put extra pressure on pastures. There is evidence that some levels of stocking in Queensland may be unsustainable because of deterioration in pasture condition (Tothill and Gillies 1992; Lawrence *et al.* 1994). Some of the many ramifications of the direct and indirect effects of grazing on sustainability are listed in Table 1. The processes are interrelated and their boundaries are arbitrary, as illustrated in the column listing indirect effects. Some of these processes are evident in the review region, and are discussed in more detail in Sections 3.3 and 3.4.

In addition to grazing, a second issue for pasture sustainability is the replacement of native communities with sown (generally exotic) grasses and legumes. This may require the complete or partial replacement of the community and has many ecological ramifications. The main reason for replacement is to develop pastures for higher levels of animal productivity and this is the land management issue that has been the major research focus for humid regions (see many articles in *Tropical Grasslands*). Despite concerns about the stability of sown pastures, the technology has been further promoted as a solution through the use of different cultivars and better management to develop “sustainable pasture systems”—those in which animal production could be maintained (Teitzel 1992). However, these trends run counter to the experiences in southern regions, where pasture improvement has been associated with soil acidification and vegetation clearance (LWRRDC 1995), tree decline (Landsberg and Wylie 1991) and issues such as dryland salinity (LWRRDC 1995). All these have more to do with unsustainable land-use.

A third key issue concerning rangeland sustainability, and one that is linked to the previous two, is that of determining the spatial scales at which to manage and observe ecosystem processes. Michalk and Kemp (1993) considered the unit of management (ie. the paddock or property) to be the relevant one for assessing sustainability but acknowledged that larger units are also relevant. This is clearly the case for the issues outlined above, where the aggregation of management practices at the property level can result in thresholds of intensity or extent. Exceeding these can, in turn, lead to emergent issues such as salinity and tree decline at the regional scale.

2.3 Biodiversity as a factor in sustainability

Biodiversity conservation and sustainability concepts—an uneasy relationship

The contribution that biodiversity makes to ecological sustainability is not well understood, as may be seen from the very varied approaches to it in the sustainability literature. The scientific study of diversity has been conducted for half a century; the term biodiversity was coined in the 1980s and according to Ghilarov (1996) was always connected with the politics of environmental technology. Wilson (1992) described biodiversity as the variety of organisms considered at all levels. This includes the variety of species and their genes, the variety of organisms at higher taxonomic levels, as well as the variety of ecosystems which comprise the communities of organisms and their environment.

The confusion about the role of biodiversity in the sustainability equation is evident in the fact that some authors include it and others do not, apparently according to the context in which the issue is being discussed. For a particular land-use, most notably cropping or plantation forestry, the main objective is to maintain plant monocultures and their productivity (Hamblin 1992). There may be interest in maintaining access to biodiversity but only insofar as it provides genetic material for the development of new crops and varieties (eg. Anon 1992). However, the maintenance of biodiversity for these purposes is a global issue. In this context, biodiversity can be seen as part of the 'environmental assets' to be maintained for the next generation (Pearce *et al.* 1989). From the global or regional perspective therefore, cropping could sometimes be viewed as an unsustainable activity because of its direct and indirect effects on the natural ecosystems that support biodiversity.

Even within a highly simplified cropping landscape, the functional issues of biodiversity cannot be dispensed with. The role of a diverse soil biota in maintaining soil processes is still unclear (Elliot and Lynch 1994), but it appears that crop and habitat diversification can contribute positively to ecosystem processes such as pest control (Altieri *et al.* 1983).

Table 1. Manifestations of unsustainability in pasture ecosystems resulting from grazing

Ecosystem attribute affected	Examples of effects	Indirect effects
Species composition	Loss of grazing sensitive flora and fauna	Loss of pest controlling organisms
	Decline in dominant perennial grasses	Decline in plant productivity
	Increase in exotic species	Decrease in native species
Vegetation structure	Increase in shrubs	Fauna/flora habitat altered
	Decrease in shrubs	" " "
	Loss of plant/litter cover	Soil losses increase; fauna/flora habitat altered
	Disruption of soil crusts	Soil movement and nutrient cycling effects
	Changes to riparian zones	Effects on water quality
Vegetation dynamics	Reduced capacity to recover from grazing	Change in species composition/vegetation structure/plant productivity
	Reduced capacity to resist the effects of grazing	" " "
	Loss of regeneration niches	" " "
Soil processes	Soil acidification increase	Change in species composition
	Soil compaction increase	Soil and water loss increase
	Soil loss increase	Decline in soil fertility
Water movement	Loss of water	Decline in potential plant productivity
Nutrient cycling	Loss of recycling capacity	Decline in soil fertility
Disturbance processes	Endogenous fire frequency lost (fewer fires)	Increase in shrubs
	Endogenous grazing regime lost (grazing more intensive)	All the above effects

Sources: Harrington *et al.* (1984); Hamblin (1992); Hofmann and Ries (1991); Davidson and Davidson (1992); Bunn *et al.* (1993); Michalk and Kemp (1993); Barrett *et al.* (1994); Fleischner (1994); McIntyre and Lavorel (1994a); Robinson and Traill (1996).

Apart from productivity considerations, simplified systems may potentially face local or regional issues of biodiversity conservation. Despite severe disturbance, some highly modified systems may contain important biota. One example is the weed flora of rice in south-west New South Wales which are comprised predominantly of native wetland plants of the region (McIntyre and Barrett 1985). Within the review region, cleared areas of brigalow (*Acacia harpophylla*) may support local birds.

To an extent, the importance of biodiversity in rangelands is a more clear-cut issue, as this land use requires the resources and ecosystem services provided by a natural plant community. The extensiveness and relative integrity of rangelands also make them attractive targets for the conservation of ecosystems *per se*. At the same time, their extensiveness can lead to complacency about their status or make management interventions politically sensitive.

Rangelands are susceptible to the processes of ecosystem simplification through the application of sown pasture technologies which boost and sustain productivity. In relation to sustainable forestry, Noss (1993) said:

> To a point (although we do not know precisely where the point is), ecosystems can be simplified and brought under our control and yet still function in the sense of cycling nutrients and transforming energy into useful products. They might even be aesthetically attractive—a pastoral scene for example—although impoverished in many ways that we do not understand.

This comment is particularly relevant to the review region. We know from experiences elsewhere in Australia, that simplification of native communities has led to ecosystem collapses. A crucial question is how much diversity do we need to avoid reaching these thresholds of collapse? Or have these thresholds been reached in some areas without production or conservation problems yet appearing? Questions about spatial distribution are equally important. In which parts of the landscape is it most critical to maintain biodiversity?

How might sustainability be linked to the conservation of biodiversity?

Diversity and stability

Working from the principles developed by Odum (1983) and others that biological systems of increasing complexity have a greater ability to use solar energy, Giampietro *et al*. (1992) defined ‘biophysical capital’ as the ability of an ecosystem to use solar energy for generating biophysical processes which stabilise biosphere structure/function. Human exploitation of these systems can be seen as the diversion of energy flows from these stabilising processes in order to stabilise human society. Giampietro *et al*. (1992) regarded the stability of the dynamic equilibrium between biophysical capital and human-technological capital as an indicator of the sustainability of human activity.

If we accept the existence of a positive relationship between higher biophysical capital and higher species diversity, as proposed by Wright (1983), diverse systems can be seen as having high levels of stability. However, this is a highly questionable generalisation and the relationship depends on the particular community, the way it is perturbed and the way stability is measured (Begon *et al.* 1990; Johnson *et al.* 1996). Systems with high biophysical capital are not necessarily resilient and complex diverse ecosystems such as coral reefs and rainforest may collapse under exploitation (Margalef 1968). In the absence of human-induced disturbances, these same systems may be very stable. McIntyre *et al.* (1996) stressed the importance of the endogenous disturbance regime in determining the stability of a system under human-induced disturbance. A community of low diversity may be more resilient or resistant to a disturbance than one of high diversity (Begon *et al.* 1990) if it has been exposed to analogous disturbances over evolutionary time (Begon *et al.* 1990; McIntyre *et al.* 1996).

Because of the complexity of the issues, there continues to be much debate about the role of diversity in ecosystem dynamics (Costanza 1992). Schaeffer *et al.* (1988) identified a range of possible relationships between ecosystem structure (including its diversity) and function. These relationships included tight linkages, no linkages and no apparent linkages. Hypothetically, change in function could also occur without change in structure. Where there are no linkages, a loss of some diversity could occur without corresponding loss of function. This scheme is more realistic than broad generalisations in that it acknowledges variation between ecosystems in the way that the constituent species interact.

In the case of seasonal herbaceous communities (including grasslands), a review by Johnson *et al.* (1996) showed there does appear to be a positive correlation between species diversity and functional stability. Coexistence is achieved through phenological differences that allow different species to partition their use of seasonally available resources (water and nutrients). Tilman and Downing (1994) related species diversity to stability under drought. However, Silva (1996) has pointed out the importance of species composition in determining stability in savanna: in the case of a drought for example, stability is dependent on the presence of drought-tolerant species.

Ecological redundancy and keystone species

Examples of loose linkages between system diversity and function are covered by the concepts of ecological redundancy and keystone species. Ecological redundancy can be said to exist within a community if there are many species within one or more functional groups (Hobbs *et al.* 1992; Walker 1992). The particular functional classification used needs to relate to functions that are important to the ecosystem. The concept has most application when there are widespread but declining species and ecosystems, in which the species are poorly known (Walker 1995). Lavorel and Noble (1992) raise two pertinent questions:

i) Is a certain level of redundancy necessary for ecosystem persistence?
ii) Is redundancy an artefact of ignoring the role of particular species in carrying out particular ecosystem functions under specific sets of environmental conditions?

The answer to question i) would be 'no' in the case of keystone species (ie. species in an ecosystem that do not have any functional analogue—only one species in a particular functional group). While it has been demonstrated that under certain conditions, some species have particularly strong interactions (eg. Paine 1992) there is a lack of information about the general occurrence of keystone species across the range of community types. Without this information, Mills *et al.* (1993) have argued that the application of the keystone concept will not advance the conservation of biodiversity. Possible examples of keystone species in rangelands are mammals that influence vegetation structure, soil and water processes (Archer 1989; Brown and Heske 1990; Walker 1995) and trees and shrubs which determine fauna assemblages, microclimate and soil and water processes (West 1993 and references therein).

If we were to dissect ecosystem function finely enough, it might be possible to view redundancy as an artefact, answering question ii) with a 'yes'. However, this is not really the point of the exercise. The question is not whether any function is affected, but whether significant functions are so affected as to lose the integrity of the ecosystem. The answer is largely in the eye of the beholder: the ecosystem will still continue to function in some form, regardless of its level of simplification (Ghilarov 1996). While not actually denying the existence of keystone species, rangeland scientists have tended to 'see' redundancy in rangeland ecosystems fairly readily (eg. Johnson and Mayeux 1992; West 1993; Walker 1995), possibly because of the importance to this land use of productivity, which could potentially attain high levels if species were lost.

Ecological redundancy and functional groups

Redundancy and functional groups are important concepts to explore and develop, as they lie at the core of the relationship between diversity and sustainability. The concept of redundancy has tended to be visualised as a means by which we can protect the critically important components of the ecosystem (ie. representatives of all functional groups) yet be less concerned about the 'redundant' elements. Realistically, with our inadequate knowledge both of ecosystems and their functions, it is unlikely that we will have the luxury of being able to identify and then selectively manage the functionally important components to ensure their survival.

A more conservative and pragmatic application of a functional group approach is to distinguish functions in terms of the processes (usually some form of disturbance) thought to threaten particular systems (eg. grazing in rangelands in Australia). This makes no assumptions about redundancy or which ecosystem functions are more important. It assumes that the threatening disturbance can result in the removal of a component of the ecosystem, but that by discovering the various tolerances of disturbance by different groups of species, it will be possible to identify the most threatened elements and the traits that determine their vulnerability. By maintaining all 'disturbance' functional groups over the landscape, there is a greater likelihood that ecosystem integrity and most of the ecosystem functions can be maintained in some form.

Diversity and productivity

One ecosystem function of particular interest to humans is that of maintaining productivity. In simple systems, a theoretical relative yield advantage is obtained from a polyculture (two or three species of crop) compared with a monoculture (Vandermeer 1981). Similarly, for annual plants and several trophic levels, Naeem *et al*. (1994) reported a relationship between diversity and above-ground biomass, although the design and simplification of the experimental systems has been questioned (André *et al.* 1994). As diversity is the product of the interaction of multiple causes, there is no compelling evidence to encourage extrapolation of this pattern beyond these simple systems to more complex ones (Bulla 1996; Ghilarov 1996).

Ecological theory (Tilman 1973; Grime 1979) suggests that communities with limited resources (stressed) and low productivity have low diversity, as do high productivity communities where resources are abundant and competition drives interactions. Highest diversity is associated with intermediate levels of productivity. Bulla (1996) studied these relationships in savanna grasslands using peak standing biomass and found the results to conform with theory. The position of peak diversity (a product of species richness and equitability) along the biomass gradient varied from 300–600 g/m^2, depending on the grassland type. Species richness (the number of species per unit area) showed a similar pattern to diversity, but equitability (the evenness in amount of each species) declined in proportion to peak standing biomass.

A rationale for maintaining biodiversity in the review region

What arguments does ecological theory provide for the conservation of biodiversity in grazing lands?

Not surprisingly, ecological theory provides no firm guidelines about how much biodiversity needs to be maintained. Nor is it clear which ecosystem functions (= services) are critical to human life support systems. It may be that biodiversity maintenance is more dependent on the interactions of different ecosystems than on individual ones (Perrings *et al.* 1992). Another important gap in our understanding is the relationship between soil biodiversity and the resilience of soil ecosystems. However, this understanding is hindered as much by the difficulty of measuring microbial diversity (Elliot and Lynch 1994; Pankhurst 1994) as it is by conceptual complexities.

Nonetheless, some important points can be made about the role of biodiversity in rangelands:

- The more our actions divert energy from the biophysical processes of ecosystems to meet our own needs, the more likely we are to de-stabilise that system. Thus, theory supports the intuitively held notion that intensification of land use is riskier, because the processes maintaining the systems are likely to collapse. In some cases, there may also be loss of the services that the systems provide for us. Loss of biodiversity is one of the outcomes of this intensification.
- Although it may be possible to link particular components of diversity to specific ecosystem functions, this has rarely been achieved for an entire ecosystem. For a particular ecosystem, it may be possible to determine that some taxa are more critical to ecosystem function than the rest, but beyond this, we only know that the more species that are removed from an assemblage, the greater is the chance of losing ecosystem function.
- The generality of the questions posed makes the diversity/stability issue generally difficult to resolve, as the question boils down to how particular species in specific systems interact (Johnson *et al.* 1996). For grasslands, it appears that species diversity at smaller scales contributes to the stability of the community. The components of stability (resistance and resilience) relate to an ecosystem's response to single perturbations, whereas disturbances such as grazing and fertilisation tend to be chronic. At the landscape scale, the concept of resilience is probably important, as the temporal and spatial patterns of disturbances are variable and a diversity of land-use intensities may assist recovery from particular localised perturbations. However, the location of areas of high diversity may determine the potential for the recovery (or resilience) of the system.
- Although productivity is not linearly related to diversity, it can be argued that the presence of a diversity of plant life-forms and ecological types will provide the means to maintain productivity over longer time spans, when ecological conditions are likely to change. For example, the diverse dicotyledonous (dicot) component of grasslands can contribute significantly to animal production in some circumstances (McKeon *et al.* 1988), even though grasses are generally considered the major source of production. The adequate representation of different functional types at the community and landscape scale is likely to provide this important buffering ability in the face of climatic, land-use and land-condition changes.

Ideally, it would be possible to present managers with clear illustrations of the functional relationships between levels of biodiversity, sustainability and production. However, Australian examples tend to be isolated, fragmented and derived mainly from examples of ecosystem collapse in other regions. Research is needed to elucidate some of these relationships and present them in a positive light to land managers and planners.

In addition, the value of Australia's fauna and flora is not necessarily lost on land holders. People value nature in terms of moral attention and aesthetic appreciation (Sagoff 1992) and land managers are no exception. In many cases, losses of biodiversity can occur because of the unintentional actions of managers who fail to see the connection between what they do and the subsequent effects, which may be delayed or diffuse.

From a regional perspective, the maintenance of biodiversity is an important component of flexibility, particularly in relation to future land-use options. The irreversibility of biodiversity loss demands that decisions that will cause it be well considered for their future implications.

2.4 Ecosystem health as an alternative to sustainability

Definitions of ecosystem health

Apart from the inherent contradictions which different users may introduce to the sustainability concept, there is the difficulty of judging sustainability on an appropriate time scale. Several authors have acknowledged the impossibility of this by proposing the abandonment of fixed goals for sustainability (Pearce *et al.* 1989; Lee 1993; Schaller 1993). If sustainability is viewed as a set of objectives, rather than immediately achievable goals, there is less need to define those objectives tightly. Instead, they form the signposts indicating what direction to aim for.

The concept of ecosystem health has become increasingly adopted by policy makers and scientists (Rapport 1992a; Committee on Rangeland Classification 1994), because of the difficulties of applying sustainability concepts. Ecosystem health is potentially useful as it describes the current state of the system without making predictions about its future condition on particular time scales. Its analogy to human health is part of its inherent appeal. Human health is a complex, relativistic concept that is quite well understood by the majority of people.

Although ecosystem health could be a useful concept for communication, a brief perusal of the following definitions soon reveals the same problems of subjectivity and quantification as encountered with sustainability.

> Ecosystem health is an absence of disease (Schaeffer *et al.* 1988)
>
> ... a state of nature (whether managed or pristine) that is characterized by systems integrity: that is, a healthy nature exhibits certain fundamental properties of self-organizing complex systems. (Rapport 1992a).
>
> … an ecosystem that is "stable and sustainable—that is if it is active and maintains its organization and autonomy over time and is resilient to stress" (Haskell *et al*. 1992)
>
> Health is a normative concept ... a general index of system health [is] made up of three components: vigor, organization and resilience. (Costanza 1992).
>
> ... the degree to which the soil and the ecological processes of rangeland ecosystems are sustained. (Committee on Rangeland Classification 1994).
>
> A healthy soil will be a productive soil. (Pankhurst 1994).

The health concept is not without its critics. Like sustainability, it is a value-laden term (Rapport 1992a). Wicklum and Davies (1995) have argued that it is not an appropriate analogy as it requires an acceptance of homeostatic processes and a recognition of optimum condition, neither of which is appropriate for ecosystems. While this is an important point, the suggestion of these authors, that quality and sustainability are less misleading terms and should be adopted, is highly questionable. Another feature of the health concept that may be viewed as a weakness or strength, depending on the observer, is that it focuses on ecosystem attributes and ignores the human element of land use.

Ecosystem health in rangelands

Some of the difficult aspects of defining ecosystem health arise from the failure to differentiate between 'evolved' ecosystems, containing assemblages that have developed over an evolutionary time scale (see McIntyre *et al.* 1996) and simplified, relatively recent and highly managed systems such as modern cropping landscapes. In the former, a dynamic equilibrium has been reached between the biota, the environment and the endogenous disturbance regime. The concept of integrity can be entertained and there is a baseline from which to measure stability and resilience. In an intensive agricultural ecosystem of relatively recent origin, there may be no integrity to maintain in the sense of complex interactions between organisms and it is difficult to measure resilience if the dynamics are essentially chaotic.

In Australian rangelands, there is a set of evolved 'natural' ecosystems which are being exploited for animal production. The notion of ecosystem health can apply in the sense that a healthy system will maintain its inherent productive capacity and the interactions between the majority of the biota will be maintained. From this viewpoint, the elements of illness (as the opposite of health) listed below and derived from Schaeffer *et al*. (1988), are quite appropriate:

- falling numbers of native species;
- overall levels of regressive succession;
- changes in standing biomass;
- changes in gross or net primary energy production;
- changing relative amounts of energy flow to grazing and decomposer food chains;
- changes in mineral macronutrient stocks; and
- changes in both the mechanisms of, and the capacity for, damping undesirable oscillations.

The significance of these processes is that both decreases and increases in productivity or nutrient levels are recognised as indicators of decline in health. This is an important feature. In simplified cropping ecosystems, increases in these factors are usually viewed as a sign of health.

3 The Ecology of Grassy Eucalypt Ecosystems

3.1 What are grassy ecosystems?

Definition

Essentially, grassy ecosystems are those in which the ground layer is dominated by grasses. In Queensland, the term 'savannas' is sometimes used. These are tropical grasslands, burnt from time to time, with scattered trees and alternating wet/dry seasons (Bourlière and Hadley 1970). However, we prefer the term grassy as it does not ascribe a particular overstorey structure to the ecosystem. In addition, the term 'savanna' tends to preclude the existence of an ecotone between temperate, sub-tropical and tropical grassy ecosystems. In fact, grassy ecosystems of humid and sub-humid climates occur along the entire north-south transect of eastern Australia.

In Queensland, grassy ecosystems are usually termed 'pasture', ignoring the particulars of tree canopy structure and reflecting the ubiquity of livestock grazing across the landscape. 'Native pasture' refers to areas that are dominated by native grassland species. Because there are few, if any, communities that are entirely native, this terminology is applied loosely. 'Sown' (or 'improved') pasture, in the strictest sense, refers to areas that have been cultivated and sown with exotic species. In practice, the boundary between native and sown pastures is fuzzy. Native pastures may be oversown with exotics species without the use of soil cultivation. Furthermore, many native pastures have been colonised by exotic species that have spread and become naturalised in areas beyond the points of deliberate or accidental introduction.

General description

Structural character

Grassy ecosystems occur as grasslands (no trees), grassy woodlands (up to 30% tree cover) and grassy open forests (up to 70% tree cover) (Specht 1970; Kirkpatrick and Duncan 1987). In the review region, woodlands are the predominant structural form of grassy ecosystem, and these intergrade with grassy forests (Ridley 1962). Shrubs may occur in small amounts and grassy understoreys may intergrade with shrub-dominated understoreys. The structure is determined by biotic, anthropogenic and environmental factors (Trémont and McIntyre 1994). Factors which reduce tree survival include fluvial and aeolian erosion, rockiness, herbivory, shading, frost and fire (Kirkpatrick *et al.* 1988).

The structure of the ground layer comprises a matrix of perennial tussock grasses, with interstitial grasses and forbs occupying part of the inter-tussock spaces. A variable amount of bare ground and litter also occurs between the tussocks (Trémont and McIntyre 1994). The structure is similar to that described by Grubb (1986) in his matrix/interstitial model, although the representation of annuals in the native interstitial flora is low in Australia (Trémont 1994; McIntyre *et al*. 1995). Although forbs[1] make up the majority of the species in the ground layer (Patton 1936; Lodge and Whalley 1989; McIntyre *et al.* 1993; Trémont 1994), they form the minority of the biomass (see Shaw and Bissett 1955; Shaw and Mannetje 1970; McIvor 1981).

Floristic character

In terms of numbers of species, temperate grassy vegetation is dominated by the Poaceae with large representations of the Asteraceae, Fabaceae, Liliaceae, Orchidaceae, Juncaceae and Cyperaceae. An array of other dicot families is represented by small numbers of species (Trémont and McIntyre 1994). This pattern is similar in the tropical and sub-tropical grassy vegetation of humid and sub-humid regions (Grice and McIntyre 1995). In general, grassy species can be considered to be sparsely distributed (*sensu* Rabinowitz 1981), with wide geographic ranges and low levels of dominance where they occur (McIntyre 1992; McIntyre *et al.* 1993). A major difference between climatic regions is winter rainfall dominance in temperate zones and summer rainfall dominance in the tropics which favour the growth of C_3 and C_4 grasses respectively (Groves and Williams 1981).

Distribution

General soil and climatic distribution

Although other vegetation types are present, grassy vegetation forms an almost continuous band from Tasmania to Cape York Peninsula and across tropical

[1] Herbaceous dicotyledons, terrestrial orchids and members of the Iridaceae and Liliaceae

northern Australia (Moore 1970; Burrows *et al.* 1988b). Grassy vegetation occurs on a variety of substrates, although parent materials of lowest nutrient status tend not to support grassy understoreys.

Species distributions

In general, species of grassy vegetation are widely distributed. For example, taxa recorded on the northern tablelands of New South Wales had an average geographical range of 4.5 Australian states (McIntyre *et al.* 1993). *Themeda triandra* occurs as a dominant from Tasmania to the tip of Cape York Peninsula. Other grasses such as *Heteropogon contortus* and *Poa* spp. are frequent as dominants over smaller, but still impressive ranges. Although temperate and tropical ecosystems tend to be partitioned in studies of vegetation, these patterns provide evidence of continuity between vegetation in different regions on a north-south gradient. This continuity is apparent in a sample of 413 herbaceous species found in temperate (because of high altitude) grassy vegetation in the northern tablelands of New South Wales, at the boundary of the temperate and tropical zones. Fifty species span four contiguous tropical and temperate states, while the majority of species span two temperate states and into Queensland (Fig. 1).

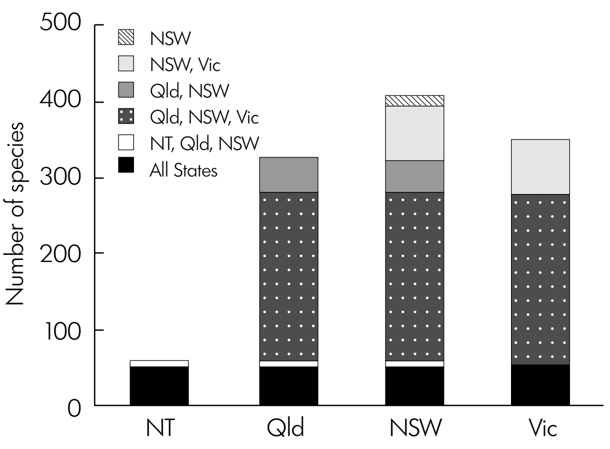

Fig. 1. Distributions of a sample of 413 temperate grassland species recorded on the northern tablelands of New South Wales. Species are those listed in McIntyre *et al.* (1993) for which distributions were available. The distribution combinations are for the four states only, most of the species had ranges extending beyond these. Five species had disjunct distributions over these four states and were excluded from the sample.

3.2 Grassy ecosystems in the review region

Regional classifications of grassy ecosystems

There is no satisfactory classification available for the grassy ecosystems in the review region. The state Department of Environment is currently completing an inventory of ecosystems in Queensland (Sattler and Williams, in preparation), based on thirteen bioregions in which regional ecosystems are identified. Regional ecosystems are vegetation mapping units that are linked to geology, rainfall and landform patterns within bioregions. The bioregions relevant to the review region are South Eastern Queensland and the Brigalow Belt (North and South) and the Desert Uplands (as mapped in Thackway and Cresswell 1995).

Specht *et al*. (1974) classified vegetation types on the basis of the structural and floristic attributes of the tallest stratum (height and density), with no reference to the nature of the understorey. This scheme does not distinguish between woodlands and forests with grassy or shrubby understoreys. With the exception of grasslands, in which the dominant grass is generally identified, this scheme does not contribute to our understanding of the types of grassy ecosystems in the region.

A third scheme developed by Weston *et al.* (1981) is derived from a vegetation classification comprising 35 vegetation zones mapped by J.A. Carnahan in 1973 and reproduced as maps in Queensland and Australian atlases (Carnahan 1976; Queensland Premier's Department 1976). In the scheme, the 35 zones were aggregated into 14 native pasture 'communities'. Short, highly simplified descriptions of each of these communities are presented in Weston (1988), and within the broad categories, refinements have been presented in Tothill and Gillies (1992). It is noteworthy that beyond these descriptions, there are no accounts that systematically integrate tree and understorey information and no quantification of the floristics of grassy ecosystems, and hence no information on the relative abundance of species. Without it there is little chance of assessing or monitoring their conservation status.

Of the fourteen native pasture communities identified for the entirety of Queensland, six occur in eastern sub-tropical Queensland (Weston 1988):

1) *Blady grass*. These are grasslands of restricted extent in high rainfall areas on the coastal lowlands, where rangeland use is minimal. Characteristic grasses are *Imperata cylindrica* and *Themeda triandra*. Extensive development makes landscape fragmentation an issue for these grasslands.

2) *Queensland blue grass*. Grasslands of *Dichanthium sericeum* are also of limited extent on the cracking clays of the Darling Downs. As the cropping potential of these grasslands is so high, their management as rangelands is now rare, and the grasslands are largely in a fragmented condition.
3) *Brigalow pastures*. Open forests and forest of *Acacia harpophylla* are extensive on fertile soils in the western part of the review region. Grasses are sparse in these communities, unless they are cleared, in which case colonisation by native species of *Chloris*, *Paspalidium*, *Dichanthium*, *Sporobolus* and *Eragrostis* may occur. The mixture of cropping and grazing, and the capacity of brigalow to regenerate after clearing makes these landscapes fragmented to variegated, depending on the level of development.
4) *Mitchell grass*. This community is dominated by *Astrebla* spp. and is subject to rangeland use. However, with the exception of some small eastward occurring areas of Mitchell grass, this community is primarily an arid and semi-arid region of inland Australia and not of importance in the review region.

The two final communities, Black spear grass and *Aristida*/ *Bothriochloa* pastures are most relevant to the discussion, as they represent extensive grassy ecosystems that are used primarily as rangelands. In the following sections, the accounts of Weston (1988) and Tothill and Gillies (1992) are used to summarise these types in the review region.

Black spear grass

This region comprises eucalypt woodlands and open forest that form the most extensive grassy ecosystems in the humid and sub-humid regions (700–1200 mm rainfall), and form a coastal belt that ranges north from the Queensland/NSW border, to Cooktown. In the southern speargrass region (south of Rockhampton) the soils are variable, but mainly duplex and generally free-draining. Important eucalypts in this zone are *E. crebra*, *E. maculata*, *E. tereticornis* and *E. tessellaris*. Not mentioned in Weston (1988) but also common, is *E. melanophloia*.

Eucalyptus tereticornis is a characteristic tree of lower slopes and river flats (Fox 1967). At least two other species are differentiated on landscape position with *E. crebra* dominating upper slope positions and *E. melanophloia* more abundant on the mid-slopes (Taylor and Cook 1993). Fox (1967) observed *E. melanophloia* on soils of higher clay content than *E. crebra,* which is consistent with their observed landscape positions.

Observations in south-east Queensland by Ridley (1962) show that grassy understoreys occur on basic rocks, and the change to a shrubby understorey in open forests can be quite abrupt with a substrate change to acidic. He also observed that grassy understoreys may occur on acidic rocks where fires are frequent. These patterns are consistent with the three major types of grassy vegetation on the northern tablelands of New South Wales (McIntyre and Lavorel 1994b). In NSW, the grassy ecosystems on granitic substrates occurred on the more fertile substrates (granodiorites and ademellites), with shrubby sclerophyll formations occurring on the infertile granites and leucoademellites (described as acid granites; Roberts 1983).

The understorey dominant that characterises this pasture type is the perennial tussock grass *Heteropogon contortus* (black speargrass) which grows on a variety of soil types (Burrows *et al.* 1988b). Other dominant perennial grasses can occur including *Bothriochloa bladhii*, *B. decipiens*, *Themeda triandra*, *Dichanthium sericeum*, *Cymbopogon refractus, Aristida* spp. and *Capillipedium spicegerum* (Partridge 1993), but there are no synthetic accounts of how floristics change with environment. Most studies of the herbaceous flora are site-specific and limited in the extent to which floristics are described (see Grice and McIntyre 1995). Historical accounts show that before European settlement, *Themeda triandra* was the major dominant (O'Shanesy 1882; Everist and Marriot 1955; Bisset 1962). In the Burnett district, the shift to *H. contortus* occurred within 35 years (Fox 1967). The compositional shift is variously attributed to grazing, drought or both.

Under current management, tree densities are controlled by poisoning. This has led to extensive modification of the woodlands and forests. Tree density has been associated with variation in understorey composition. In grassy woodlands of various species (mainly *E. crebra*, *E. populnea*, *E. melanophloia*), mainly andropogonoid grasses occurred in the open areas (specifically *Heteropogon contortus, Chrysopogon fallax, Bothriochloa decipiens* and *Themeda triandra*) while native legumes and non-Andropogonoid grasses (eg. *Aristida* spp. and *Paspalidium distans*) were correlated with treed areas (Scanlan and Burrows 1990). Where active tree clearing of *E. crebra* took place, the dominant *Aristida ramosa* was replaced by *Bothriochloa decipiens* and *Themeda australis* increased (Walker *et al.* 1986).

Aristida–Bothriochloa pastures

West of the speargrass zone, and extending into the semi-arid regions, are the *Aristida–Bothriochloa* pastures. In central and southern Queensland, this pasture type forms a mosaic with the brigalow communities. It is a composite of vegetation types (eight in Weston 1988; fifteen in Tothill and Gillies 1992). The 15-unit classification of Tothill and Gillies is used in this account, as the locations of the composite communities are identified. Six communities are described for the sub-humid sub tropics. Four are eucalypt woodlands, a fifth is characterised by *Acacia* and the sixth by a *Callitris–Casuarina* overstorey.

The four eucalypt woodland communities are summarised in Table 2. *Eucalyptus populnea* is the most characteristic overstorey component. No single grass species is common to all components, although *Bothriochloa decipiens* is most characteristic. Despite representing a wide range of ecological characteristics, the species of *Aristida* have not generally been distinguished in these classifications, and delineation within this group would assist in the characterisation of communities.

Aristida–Bothriochloa pastures intergrade with spear grass pastures. They have tree species (eg. *E. melanophloia*), and many grass species in common. A study of *Aristida*, in both spear grass and *Aristida–Bothriochloa* pastures found six of a total of twenty species in common and in both pasture types the distribution of species varied with their landscape position (McIntyre and Filet 1997).

3.3 Ecological processes—factors affecting diversity

Patterns of plant diversity—environment and endogenous disturbances

To our knowledge, there are no significant studies that explicitly describe plant diversity in the grassy ecosystems in the review region. There have been vegetation descriptions (eg. Neldner and Paton 1986) that include details of herbaceous communities, but no specific analyses of factors affecting diversity. At small scales, the species richness (alpha diversity) of temperate herbaceous communities can be extremely high (Trémont and McIntyre 1994). Richness at 1 m^2 scale at one subtropical and two tropical sites (all stock-grazed) was similar between sites (McIvor and McIntyre 1996), but much lower than the richness of comparable scales in temperate grassland (Lunt 1990; Trémont 1993). In temperate communities, there was a tendency for total species richness (30 m^2 scale) to decline on more fertile substrates and this trend was significant for rare native species, which were a subset of all natives (McIntyre and Lavorel 1994a).

We know of no studies which describe the effects of marsupial grazing on species diversity or productivity in sub-tropical woodlands. Fire is a second

Table 2. Local pasture units of *Aristida–Bothriochloa* pastures associated with eucalypt woodland and occurring in sub-tropical Queensland, east of Roma.

Name of local pasture unit	Eucalypt species	Soils/lithology	Dominant grasses	Other comments
49 *Aristida–Thyridolepis*	*E. populnea* *E. melanophloia*	Neutral red earths	*Bothriochloa decipiens* *Thryidolepis mitchelliana* *Aristida jerichoensis*	Transitional state with mulga communities
51 *Bothriochloa–Chloris–Aristida*	*E. populnea*	Hard-setting duplex soils and red earths	*Bothriochloa ewartiana* *B. decipiens* *Dichanthium affine* *Chrysopogon fallax* *Aristida* & *Chloris* spp.	Frequently associated with brigalow or bluegrass communities
52 *Aristida–Eragrostis* (southern sandy)	Many species, not listed	Hard-setting to sandy surfaced duplex soils derived from granite and sandstone	*Bothriochloa decipiens* *Cymbopogon refractus* *Aristida & Chloris* spp. *Eragrostis* spp.	Located between the Darling Downs and the southern brigalow
54 *Bothriochloa–Stipa–Danthonia*	Not listed	Shallow, dense and loamy on ‘ traprock’ or sandy duplex on granite and sandstone	*Bothriochloa decipiens* *Dichanthium affine* *Aristida & Chloris* spp. *Eragrostis* spp.	Near NSW border with temperate affinities

Source: Tothill and Gillies (1992)

endogenous disturbance (present over evolutionary time-scales) for these systems and burning can maintain species richness in temperate grasslands (Stuwe and Parsons 1977). Fire reduces the litter and the dominant grass matrix. Diversity is thought to be controlled by the competitive effects of both litter and the larger grasses (Grubb 1986) and theoretically can be maintained by fire. However, forb species vary in their tolerance to the physical effects of fire (Purdie 1977a,b), as do grasses (Andrew 1986; Orr *et al.* 1991). Community responses can therefore be variable, depending on the specific make-up of the assemblage being burnt as well as the timing, intensity and frequency of the fire. Interpretation of fire effects in the field is inevitably tied up with grazing/burning interactions, as fire predisposes grassy communities to grazing. In the Australian tropics, fire can be used to manipulate grazing pressure within paddocks (Andrew 1986).

Control of floral diversity—exogenous disturbances

Exogenous disturbances and fire

Exogenous disturbances are those that are fairly new to the community and differ greatly from endogenous disturbances in that there has been relatively little time to adapt to them. For this reason, the effects of exogenous disturbances can be quite different. In the case of fire, a particular fire regime may be endogenous, but a different fire regime may be exogenous. Thus a change to a fire regime can be an exogenous disturbance. However, in the case of fire, these distinctions are largely academic, as we have no records of endogenous fire regimes.

Livestock grazing

Grazing by domestic livestock is the most important exogenous disturbance in rangelands. There have been numerous studies on the effects of grazing on grassy communities in Queensland (see review in Grice and McIntyre 1995); however, their value to questions of diversity is limited by the fact that they focus on a subset of larger grasses and, rather than considering the broader issues of community change under grazing, the data presentation is production-oriented.

The existence of a positive response of species richness to intermediate levels of grazing is fairly well accepted in the literature (Trémont and McIntyre 1994). The logic is that with no grazing, the tussock grass matrix expands, litter accumulates and interstitial species are suppressed (Trémont 1994). At intermediate levels, the dominant matrix is reduced and interstitial species flourish, while at severe levels, many species simply cannot tolerate the physical damage.

Studies in temperate communities suggest that there may be a more complex picture in the review region if the plant community is divided up into components. McIntyre and Lavorel (1994a) showed that while total species and native species showed an approximation of the humped response, rare native species richness was adversely affected by moderate levels of grazing and exotic species were unaffected by grazing. These were comparisons of single plots from many sites. There are also complex patterns of diversity within selectively grazed paddocks that are related to grazing behaviour. Trémont (1993) found four distinct plant communities that differed in their life-history attributes and grazing tolerance, which were related to sheep grazing within 0.6 ha plots. Patchy grazing is also characteristic of pastures in the review region, but communities were not found to be as distinctive (Wandera 1993).

Comparison of different studies can be difficult if the sampling covers different portions of the grazing gradient. For example, McIvor and McIntyre (1996) compared stocking rates imposed on vegetation previously managed as commercial pastures, and so sampled within the highest level of stocking in the McIntyre and Lavorel study. The former study found richness to increase with stocking rate, although in the subtropical site, richness declined in the most heavily grazed quadrats within the highest stocked paddocks (S. McIntyre, unpublished data).

A second related issue that arises when studies are compared is the management history of the assemblage being studied. If the community has been heavily grazed in the past, the grazing-sensitive species may have been eliminated. The response to subsequent grazing treatments will be different from the response of a community that contains grazing-sensitive species. Thus in studies of diversity and grazing, the context becomes critical to the interpretation of the results. However, an important and consistent result in both temperate and tropical regions is that even heavily-grazed native pastures support many native species, and when the total area is considered, a considerable amount of biodiversity coexists with livestock production.

Other disturbances

Other disturbances of most relevance to the review region are soil disturbance, nutrient additions, water additions (through altered soil surface-drainage characteristics) and tree killing. In addition, accidental and deliberate plant introductions are common (Neldner and Paton 1986) and their invasiveness is dependent on the interactions with these disturbances. While there is a large literature describing the effects of nutrients, soil disturbances and exotic plants on ecosystems, there are few which specifically describe these effects in the review region. However, there are some generalisations that are relevant to grassy ecosystems in the review area:

- Nutrients reduce native plant diversity, and increase exotic species, in communities adapted to nutrient-poor soils (Hobbs and Huenneke 1992; Marrs 1993) ;
- Large-scale soil disturbance reduces native and rare plant diversity (Stuwe 1986; McIntyre and Lavorel 1994a) ;
- Water enrichment reduces native plant diversity, and rare natives are more sensitive to water enrichment than natives in general (McIntyre and Lavorel 1994a);
- Exotic species increase generally with increased levels of disturbance (Saunders *et al.* 1990; Groves and Burdon 1986) ;
- Exotic species are associated with reduced densities of rare species in grassy vegetation, even when the effects of disturbance and environmental variation are removed from the analysis (McIntyre and Lavorel 1994a);
- Some exotic species are associated with lower diversity and susceptibility to invasion (notably perennial grasses, Lonsdale 1994), while others appear to have no effect on species richness (McIntyre 1993) ;
- Disturbances tend to be correlated and applied as a regime specific to particular land-uses. Consequently, their individual effects can sometimes be difficult to identify (Weir 1977; McIntyre and Lavorel 1994b).

Diversity at broad spatial scales

Most of the previous discussion has concerned measurement of diversity at small spatial scales. For plants, some of this information can be aggregated to build a regional scale perspective, eg. by comparing community structure between sites to build up a regional picture of variation or by extrapolating the effects of disturbance on diversity. Explicit studies of landscape diversity have not been conducted for grassy ecosystems in Australia, but there is evidence that spatial aggregation and temporal extrapolation of the effects reported in the previous section would be valid. The same disturbances operating for a longer time in Victoria have resulted in the large scale destruction and modification of grassy vegetation and the endangering of many component species (Lunt 1991). Conceptual models of landscape modification are discussed in section 5.3, together with some hypotheses about landscape diversity.

For an understanding of faunal diversity patterns, landscape structure becomes more important, since mobile animals, particularly birds, temporally partition their use of resources over very large distances. However, there are indicators of faunal habitat quality that are available from site data and some of the information can be aggregated to landscape and regional scale in the same ways as it can for plants.

Habitat structure and its relationship to diversity

Is there a relationship between plant diversity and habitat structure?

The effects of disturbance may affect habitat structure and have indirect effects on plant diversity. This was illustrated by the matrix interstitial model in previous sections. At larger scales, the spatially variable effects of grazing may contribute to beta diversity (species turnover) at moderate levels but reduce it at high levels when grazing is non-selective. Another source of structural variability is the presence of trees and shrubs. Standing and fallen timber may contribute to micro-habitat heterogeneity (variation in light, water, nutrients and grazing protection) thus directly influencing the diversity of plants as well as a range of other biota. This has been well recognised in forests (Maser and Trappe 1984) but not in woodlands.

Habitat structure and faunal diversity

As for vegetation, the published literature on faunal diversity in Queensland grassy ecosystems is still in a ‘natural history’ phase, and few principles have been developed. But again, there is ample evidence from related temperate ecosystems that fauna are greatly affected by intensification of land use. Fauna depend on structural components in both the herbaceous and woody strata, although their specific needs may vary. We hypothesise that the loss of tussock structure from grazing and pasture improvement has probably been important in the decline of some grassland fauna in eastern Australia. Evidence for this is the use of tussocks as habitat (eg. Rufous Bettongs) and the persistence of some species in regions where tussock grasslands are relatively intact, compared with their decline elsewhere (eg. Bush Stone-curlew, Diamond Firetail; Robinson and Traill 1996).

Barrett *et al.* (1994) have listed the following structural elements as associated with the presence of woodland birds that are sensitive to habitat modification:

- stands of eucalypt forest or woodland ≥ 6 ha in extent;
- maintenance of indigenous flora;
- lightly grazed or ungrazed areas of tussock grassland;
- moderate levels of mistletoe;
- a range of tree age classes;

- some shrubs;
- fallen trees and woody debris;
- riverine vegetation

These elements are also likely to include, specifically, tree hollows and leaf and stick litter (Robinson and Traill 1996).

3.4 Factors affecting ecosystem processes

Ecosystem services

Grassy ecosystems are widely used to produce animal products but they also provide an array of services—these include (West 1993) contributing to the maintenance of the gaseous composition of the atmosphere; amelioration of climate; formation, fertility and stability (including control of erosion) of soils; disposal of wastes and provision of clean water supplies; cycling of nutrients; natural control of pathogenic and parasitic organisms; and pollination of crops. The long-term viability of ecosystems and their ability to continue to produce both economic products and services depend on maintaining the temporal and spatial continuity of the cycling and fluxes of carbon (energy), nutrients and water through the system. Disruption of this continuity leads to changes in ecosystem composition and structure. The outputs from systems must be balanced by inputs, or the natural resources will be mined out.

Natural systems support a complex array of soil animals and micro-organisms which are directly important for fluxes of energy and nutrients through the systems, and indirectly influence water cycles. Primary producers and soil organisms strongly influence each other and a change in the composition of one may result in major changes in the composition of the other (Medina 1996).

Carbon fluxes

The primary production by photosynthesis is the basis of production from grasslands and this is consumed by herbivores, fire or decomposers. The rate of consumption varies widely (fire > herbivores > decomposers). Nitrogen concentration in the herbage influences the balance of consumption between these groups. The nitrogen requirements decrease in the order herbivores > decomposers > fire, and as the N concentration drops the balance shifts away from herbivores to fire. N concentration also influences the rate of consumption, with decomposers more affected than herbivores and fire not affected. The C remaining in the soil is very important for soil structure, as aggregate stability depends on organic matter bonds between soil particles.

Nutrient fluxes

Although many factors determine soil fertility (geology, climate, vegetation), in the short term the availability of soil nutrients depends strongly on nutrient cycling and a productive soil needs both a favourable structure and an efficient nutrient cycling system. In grasslands, N and P are nearly always limiting to plant growth. The availability of N and P (and S) is inherently linked to the turnover of organic material and is determined by competition between biological and geochemical sinks (Attiwill and Adams 1993).

The soil microbial biomass functions both as a nutrient source and a sink. This is of particular importance in grassland ecosystems where the annual flux of nutrients through microbial biomass may be several times greater than plant uptake (Ruess and McNaughton 1987).

Water fluxes

The availability of water largely controls plant growth in rangelands and also affects other processes such as soil erosion. Some of the deleterious changes in rangelands can result from alterations to the hydrological cycles which are in balance in virgin landscapes. Decreases in evapotranspiration (because of reduced infiltration or lower water use by altered vegetation) lead to increases in runoff and/or deep drainage to balance the cycle, with a consequently increased likelihood of erosion and salinity (Williams and Chartres 1991).

The importance of plant cover for reducing runoff and soil erosion is well recognised (see review by Gifford 1985); it intercepts and absorbs the energy of falling raindrops; it impedes the flow of runoff water and so increases infiltration; and it resists the erosive force of flowing water (Osborn 1952). Many studies have shown that below some critical cover level, runoff and soil loss increase rapidly (Packer 1951; Copeland 1963; Elwell and Stocking 1976; Lang 1979; Costin 1980; Snyman and van Rensburg 1986; Lawrence and Cowie 1992; Silburn *et al.* 1992; Zobisch 1993; McIvor *et al.* 1995b; Scanlan *et al.* 1996).

Both point and landscape level scales are important—fluxes at the point scale determine the availability of water for biological activity at that site and they also determine the water moving as runoff and deep drainage which has effects elsewhere. Riparian areas are critical in these landscape-scale movements—they act as buffer strips to filter sediments and slow overland flow, and to stabilise stream banks.

Management factors

The management factors considered below are grazing, fire, fertiliser, timber treatment and the use of exotic species. These are considered separately, but there are interactions between them—for example, cattle concentrating their grazing on burnt areas may compact the soil in these areas and reduce infiltration. The relationship between soils and vegetation is also interactive: vegetation affects soils directly through the supply of organic matter, and indirectly by modifying soil temperature and moisture regimes, soil chemistry, and stability of the soil surface.

Grazing

The principal objective of livestock grazing systems is to maintain/improve forage production and/or harvesting efficiency. Maintenance of ecosystem health depends on balancing increased harvesting with the need to maintain soil characteristics (eg. aggregate stability) which depend on cover and organic matter in the soil. This involves short and long term considerations and requires compromises between what is optimal for plant, soil and animals respectively.

One of the most direct effects of grazing on soil processes is the removal of plant cover and exposure of soil surfaces. It is important to maintain plants and litter on the soil surface so as to reduce the amount of soil detached by raindrops, to increase the flow depth and to reduce the flow velocity so that less soil is detached and washed away. Soil cover does not only protect the soil surface; it also favours biological activity and improves the condition of the soil, increasing infiltration.

A number of studies have shown that heavy grazing can alter the numbers, composition and activity of animal and microbial populations in the soil (eg. Holt and McIvor 1994 ; Holt *et al.* 1996) with consequent effects on cycling. Grazing influences the nutrient inputs, outputs and transformations (Archer and Smeins 1991). Grazing reduces the amount of C (and other nutrients) immobilised in standing dead material and litter (Biondini and Manske 1996). Reductions in litter input because of grazing, combined with high rates of soil respiration, can lead to significant declines in soil organic C (Bridge *et al.* 1983; McIvor *et al.* 1995a).

Under grazing, nutrients otherwise tied up in standing dead material and litter are recycled more rapidly, redistribution of nutrients within plants is reduced, levels of soil biological processes increase (accelerating N turnover), and nutrients are concentrated in dung and urine in forms more available to plants (Frost *et al.* 1986; Wedin 1995; Chaneton *et al.* 1996). Nutrients may also be lost because of this increased mobility in the soil profile. Other causes of nutrient loss through grazing can be caused by the harvesting of livestock and deposition of dung and urine in stock camps and unproductive areas (yards).

Grazing affects nutrient mobility through the following processes: i) the concentration of nutrients in small volumes of soil where they exceed the short-term plant requirements and lead to gaseous and leaching losses and ii) chemical fixation (eg. P) and immobilisation in organic forms (Haynes and Williams 1993). Grazing animals can consume 40% of above-ground nutrients (Chaneton *et al*. 1996) and recycle twice as many nutrients as the amount removed from the site as animal biomass (Marrs 1993). Of the nutrients ingested from grazing, 60–90% are returned in dung and faeces (Haynes and Williams 1993). The resulting increased mineralisation rates, and nutrient cycling, may stimulate nutrient uptake and production (Marrs 1993; Chaneton *et al.* 1996) although continued heavy grazing can lead to reductions in N and P uptake and herbage production (Ash and McIvor 1995; McIvor *et al.* 1995a).

Many studies have shown that increases in grazing pressure increase soil compaction, bulk density and penetrometer resistance, and decrease aggregate stability, sorptivity, and hydraulic conductivity and thus decrease water infiltration (Gifford and Hawkins 1978; Willat and Pullar 1983; Thurow 1991; Fleischner 1994; McIvor *et al.* 1995a; Holt *et al.* 1996). Impacts can vary with soil type. Van Heveren (1983) found that heavy grazing increased the bulk density on fine-textured soils, but not on coarse-textured soils. Mott *et al.* (1979) showed that under heavy grazing at Katherine, grass clumps are destroyed, and the soil surface quickly collapses and forms seals with lower organic C in the surface, lower infiltration, and lower sorptivity and hydraulic conductivity than on grassed areas.

As mentioned earlier, cover is important for infiltration and grazing can reduce infiltration by reducing standing crop/litter biomass and cover. Grazing may also influence infiltration by altering grass composition; Thurow *et al.* (1988) showed that infiltration was greater on soils dominated by tussock grasses than on similar soils dominated by stoloniferous grasses, because of the maintenance of more stable cover under tussock grasses.

Fire

Fire affects ecosystem processes directly by removing dry material and litter, and altering the micro-climate, and indirectly by influencing plant growth and soil activity. Grassland fires are seldom hot enough to directly oxidise organic matter deeper than a few millimetres in the soil, but burning can either increase (because of the decay of roots of plants killed by fire and an increase in primary productivity) or decrease soil organic matter (because of increased microbial activity with soil warming, a temporary elimination of litter) (Daubenmire 1968).

Burning removes a proportion of all nutrients, with most nitrogen and sulphur being volatilised. Chapman (1967) found that 95% of the N, 26% of the P and 21% of the K were removed from the burnt area. Nutrients not lost as smoke are changed to simple salts and deposited as ash. These are water-soluble and hence immediately available and may be taken up by vegetation, leached (eg. K), or they may become less available or fixed in the short term (eg. P) (Marrs 1993). In contrast to forest fires, grassland fires produce too little ash to act as a significant fertiliser (Daubenmire 1968).

Frequent burning has little direct effect on the soil (Teague and Smit 1992) ; most effects are indirect and result from changes in the vegetation and are confined to the surface soil. Burning may speed up the rate of nutrient cycling by reducing litter, especially components such as woody leaf litter and dead wood which decompose slowly (Frost *et al.* 1986). N, C and S are lost through volatilisation and removal in smoke, but the overall significance of the losses has not been assessed. Even though losses may be small in many tropical grasslands, because of the small amounts of nutrients in dry, dormant material (eg. 4–6 kg N/ha—Norman and Wetselaar 1960; Tothill 1983a) such losses may prevent increases in soil N and reinforce N limitation (Wedin 1995).

A number of studies have shown that soil organic matter and total N levels are lower on burnt areas and that more frequent and intense fires have greater effects (Harrington and Ross 1974; Brookman-Amissah *et al*. 1980; Frost *et al.* 1986; Jones *et al.* 1990). Available P levels may increase slightly (Frost *et al.* 1986) and McKelvin and McKee (1986) in a long-term burning trial found greater P (and N) uptake from soil from burned than unburnt plots.

Fire bares the soil surface by reducing litter and herbaceous plant cover; this can result in increased runoff and soil movement (McIvor *et al.* 1995b) although not all studies have shown these effects (Roundy *et al.* 1978; Emmerich and Cox 1992).

Although there are erosion risks and small nutrient losses associated with fire, it is an endogenous disturbance that will always be part of extensively managed grassy ecosystems. Fire has considerable benefits. From a production perspective, fire promotes higher quality, accessible forage. It is also useful to manipulate plant composition in maintaining a tree/ grass balance, controlling herbaceous composition and manipulating grazing pressures within paddocks.

Fertilisation

Application of fertiliser to deficient pastures can produce large yield responses and increases in the organic C in vegetation and subsequently in litter and the soil. These production benefits must be weighed against the potentially destabilising effects of nutrient addition to ecosystems that have evolved under nutrient-poor conditions, as is common in Australian landscapes.

Fertiliser addition to low fertility soils provides an obvious direct increase in the nutrient supply but can also have indirect effects via changes to plant growth and composition or changes to the soil microbial population. For example, fertiliser application can change the mycorrhizal community with adverse effects on plant growth (Johnson 1993).

Fertiliser can indirectly affect the water cycles by increasing transpiration from the increased growth, and also by improving water infiltration because of increased soil cover and improving the soil surface condition. However, fertilisers have the potential to enter ground and surface waters, with attendant problems of eutrophication.

Tree clearing

Tree clearing or killing frequently increases herbage production (see studies in eucalypt woodlands in Australia by Walker *et al.* 1972, 1986; Gillard 1979; Tothill 1983b; Winter *et al.* 1989; Scanlan and Burrows 1990) and thus the amount of C from this source that enters the cycle; however the input from the trees is lost.

Removal of live trees can temporarily increase available nutrient levels. Lawrence *et al*. (1993) showed that clearing and burning of brigalow resulted in a flush of plant-available N, P and K but the effect was short lived and after three years, soil organic C and total N had returned to pre-clearing levels. Similarly, Tunstall *et al*. (1981) found soil ammonium and nitrate levels were higher with trees killed than with live trees.

Although the effect is not universal, many authors have shown nutrient levels to be higher under tree canopies than in the inter-tree zones, both in the review region (Ebersohn and Lucas 1965; Christie 1975; under *Eucalyptus populnea*) and elsewhere (eg. Kellman 1979; Belsky *et al*. 1989; Georgiadis 1989; Mordelet *et al.* 1993; Isichei and Muoghalua 1992). Where this occurs, tree removal will lead to the local environment becoming more uniform, reducing the heterogeneity of potential habitat for herbaceous species.

Just as nutrient levels may be higher under tree canopies, the soil's physical characteristics can also differ. Joffre and Rambal (1988) found that trees induced changes in soil properties and thus in water flows; bulk density was lower, structure better, and organic matter higher, with increased infiltration and greater soil water storage.

Tree clearing can have major effects on the flux of water through ecosystems. Trees can extract more water from deep in the soil than grasses (Williams *et al.* 1997) and when they are removed, deep drainage can increase (Thorburn *et al.* 1991; Williams *et al.* 1997) although, in a study of clearing in small brigalow catchments, ground-water recharge increased in the year after clearing but not later when pasture had been established (Lawrence *et al.* 1991; Thorburn *et al.* 1991; Lawrence *et al.* 1993).

Dryland salinity is a consequence of disturbing the hydrological cycle that was in balance in a natural landscape (Williams and Chartres 1991). The problem and the solution have been linked to changes in tree populations, which can influence the water balance of the landscape (Thorburn 1991; LWRRDC 1995). When water use by replacement vegetation is less than that of the native vegetation, more water moves through the profile and below the root zone. This water enters the regional groundwater and salt in the soils or groundwater is relocated to seepage areas where it accumulates after evaporation.

Exotic plants

Pasture development, including the sowing of exotic species, generally leads to increased productivity. Among the selection criteria for exotic sown species for the review region have been high productivity and response to fertiliser. As a result, exotics tend to have higher productivity than their native equivalents. Unfortunately, comparisons have nearly all been confounded with the use of fertilisers in the sown pasture and not in the native control (Grice and McIntyre 1995). The growth and stature of exotic species can reduce the resource supply to smaller, slower growing species.

Native grasslands have complex communities of invertebrates. Changes to sward structure, composition (including loss of native plant species) and productivity can lead to the elimination of species which rely on native species for food and habitat (D'Antonio and Vitousek 1992) and result in simplified communities that tolerate the disturbance regime. Wardle *et al.* (1995) showed that ragwort invasion influenced soil microbial biomass, saprophytic micro-arthropods and soil macro-fauna.

Exotic plants can change ecosystems by differing from native species in resource acquisition (particularly N) and/or resource use efficiency (Vitousek 1990). There can be strong feedbacks between species composition and N cycling (Wedin and Tilman 1990) with species producing nutrient-poor litter leading to reduced soil nutrient supply—a grass species need affect only the small labile pool of soil organic matter (<3% of the total) to cause a tenfold divergence in N mineralisation rates (Wedin 1995).

Average legume growth can contribute 40–210 kg N/ha/year to pasture systems (Henzell 1968) and the increased N mineralisation after legume sowing in pastures can contribute the equivalent 10–90 kg/ha of fertiliser nitrogen to the following crop (Jones *et al.* 1991). Increased N mineralisation in legume pastures (Wetselaar 1962; Crack 1972; Myers 1976) under warm, moist conditions during the early wet season can make more N available for plant growth but risks of loss by leaching are also high.

Exotic species frequently use water differently to native species and this can either lead to lower water use (and increased runoff and/or deep drainage) or greater water use and the drying out of soil profiles (Begg 1959, Donald 1970). For example, Williams *et al.* (1997) showed *Stylosanthes scabra* (seca stylo) to use less water than native woodland, but more water than native grassland. The efficient use of water can enable a species to outcompete associate species, as is shown by the replacement of *Agropyron spicatum* by the introduced *A. desertorum* in portions of the Great Basin of the USA (Eissenstat and Caldwell 1988).

Exotic plants and soil acidification

Legume pastures are associated with declines in soil pH levels in southern Australia (Williams 1980). Legumes cause the acidity, and the use of fertilisers enhances legume growth and accelerates the effects. The dominant factor in these changes is nitrate leaching and base stripping from the exchange complex. Other factors are proton accumulation because of the nitrification process, removal of nutrients in agricultural products, and organic matter accumulation. Soil acidification can lead to phytotoxic levels of Al and Mn (Haynes and Williams 1993) and deficiencies of molybdenum and calcium with consequent effects on plant and animal productivity.

4 Land Use in the Review Region

4.1 Land-use history of the review region

Management by Aborigines, and other endogenous disturbances

Grassy vegetation seems to have appeared in Australia in the mid to late Miocene age, approximately ten million years ago, during a period of increasing aridity (Smith 1982). The association of grassy vegetation with fire has probably been a long one. Although there are numerous accounts from explorers and early European settlers about the use of fire by indigenous people (Pyne 1991), there are no descriptions of the fire regimes that were applied to grassy ecosystems in the review region. The only thing we can say with confidence is that fire regimes in Australia have changed since European colonisation (Smith 1982). In the tropical monsoon grassland of the Northern Territory, Stocker and Mott (1981) suggest that fires were of low intensity and lit early in the dry season. Fires started by electrical storms are also presumably part of the endogenous pattern of burning.

Other disturbances would have included grazing by native mammals and small-scale disturbances from digging both by marsupials and humans (Gott 1983). These processes would have been far less intensive and widespread than the exogenous soil disturbances created by livestock and humans at present. This is because of the small size of the native animals and the variation in their population according to the availability of water. But as for fire, current observations will not reveal the details of the endogenous disturbance regimes under which these ecosystems would have evolved.

Pastoral settlement

European settlement of eastern Queensland took place between 1840 and 1860 (Shaw 1957). Initially, livestock included sheep and cattle, but the husbandry of sheep declined in areas where *Heteropogon contortus* replaced *Themeda australis* as a result of the new, heavier, stocking regime (Shaw 1957). The sharp callus of *H. contortus* seed caused problems in sheep and they were progressively replaced by cattle between 1880 and 1940. Although heavier than pre-settlement regimes, stocking was relatively light. Management of the vegetation included:

> fire—most commonly applied in the late dry season, and not every year (Bisset 1962) and;
> tree clearing—originally ring barking was used, but after 1960, stem injection of herbicides was common (Burrows *et al*. 1988a).

Intensification of land-use

Plant introductions

A number of technological and other developments made possible the more intensive exploitation of the region which has become most evident since the 1970s. Plant introduction has been voluminous since the 1930s, when the systematic introduction of exotics began (Miles 1949; Harrison 1986). Although the extreme seasonality of forage quality caused problems with tropical pastures (Wilson and Haydock 1971; McIvor *et al.* 1983; Miller and Stockwell 1991), at least part of the enthusiasm for exotic species arose from a general culture in which exotics were considered superior. The first work on the forage quality of a native grass was that of Miles (1949), conducted in 1940–45, four years after the first systematic introductions (Shaw and Bissett 1955). By the mid-1980s, 8000 grass and 22 000 legume accessions had been imported into Queensland (Mannetje 1984). An unspecified number have been grown in the open ground, but only 100 cultivars have been released (Mannetje 1984).

Sown pastures

Together with cultivar release, various techniques have been developed for establishing exotic species (reviewed by Cook *et al.* 1993). These involve either 1) replacement of the native ground layer through cultivation or herbicide treatment, followed by sowing grass legume mixtures; or 2) oversowing legumes into existing native or sown pasture through partial disturbance. In some cases, seed is broadcast without any disturbance. In most cases, fertilisers are needed for establishment, and maintenance fertiliser is recommended to maintain the pastures (Teitzel 1992).

The area of sown pastures in 1978 was calculated by Weston *et al*. (1981). Only small percentages of the total areas used for pasture had been sown, with less than 5% in all regions (Table 3). Walker and Weston (1990) recalculated the sown area as a percentage of the pasture for which the economic benefit of sowing was easily attainable (rather than percentage of the total of all pastures). The overall revised figure in 1990 showed that around 40% of the area in Queensland suitable for sown pasture, had already been sown.

A number of factors affect pasture sowing decisions. When animal products are highly profitable, investment in sown pasture tends to increase (Walker and Weston 1990). Other factors include favourable seasons, tax incentives and suitable species. There have been some problems in maintaining the stability of sown pastures (Teitzel 1992) and maintaining an appropriate grass/legume balance has also been a problem (Jones *et al.* 1997). The higher levels of management that sown pastures need to maintain their composition and productivity could also be a disincentive for sowing. The establishment and ongoing maintenance of sown pastures can be risky because of commodity prices and seasonal conditions (Clem *et al*. 1993; McIntyre 1996).

Livestock

Numbers of livestock increased dramatically in the 1970s and peaked in the late years of the decade (Walker and Weston 1990). This coincided with exponential growth in the use of sown pastures and with a number of other changes, notably feed supplements, the use of hardier *Bos indicus* cattle and low cattle prices in 1973. These changes have increased not only the numbers of cattle, but the total grazing pressure and the amount of pasture use (Mannetje 1984; Tothill and Gillies 1992). However, fertilising natural pastures is not regarded as an efficient means of improving diet quality in this region (Norman 1962), hence the use of either sown pastures or nutritional supplements for cattle.

Tree clearing

In general, the removal of trees in grassy ecosystems is thought to increase the productivity of the ground layer, although this effect declines along a south to north gradient (Burrows *et al.* 1990). For these reasons, clearing trees has become the major pasture management activity in eucalypt woodlands in southern Queensland. In an unknown proportion of woodlands, there is a continuing cycle of thinning and regeneration. Burrows *et al.* (1988a) attribute this to the pattern of clearing, which commonly involves thinning trees to a savanna landscape, with scattered large trees. As regeneration is associated with these trees, regrowth tends to be spread across the paddocks. For this reason, Burrows *et al*. have recommended that clearing occur on an all-or-nothing basis, retaining strips of woodland rather than scattered trees.

4.2 Conservation status of grassy ecosystems

In the south-east Queensland bioregion, over 40% of the grassy ecosystems are considered to be ‘of concern’ in terms of their conservation status (Department of Environment, unpublished data). Compared to all ecosystems in the bioregion (25% of ecosystems are ‘of concern’ overall), this is high, although the proportion of ‘endangered’ grassy ecosystems is relatively low (7% compared with 13% overall in south-east Queensland). Casual appraisal of the landscape suggests that extensive grassy ecosystems still occur in the region. The low levels of pasture improvement also suggest that the understorey could be in comparatively good condition overall, but there has been no assessment of this. However, the reservation status of grassy communities is low (approximately 70% with no, or low levels of reservation—Department of Environment, unpublished data) and there are threatening processes at work (eg. grazing, overclearing of trees, pasture improvement).

Table 3. Extent of sown pasture development in 1978 and proportion of the total area that was estimated to be suitable for sowing, including the area already sown (Weston *et al.* 1981) for five pasture communities in south-eastern Queensland.

Pasture type	Total area of pasture type (ha)	Area sown to pasture in 1978 (ha)	% of total sown in 1978	% of total area with potential to be sown
Black spear grass	25,000,000	569,000	2	42
Queensland blue grass	2,370,000	99,000	4	31
Aristida–Bothriochloa pastures	33,526,000	612,000	2	38
Blady grass	2,718,000	140,000	5	27

Although a reserve system is necessary to protect the most sensitive components of the ecosystems and provide reference areas for the future, we argue that in grassy ecosystems, even relatively good reservation status will not guarantee the maintenance of biodiversity. There are particular features of grassy vegetation which suggest that they have evolved as extensive, relatively continuous systems. This explains the lack of endemism in the flora and the existence of many sparse species—widespread, but never dominant (McIntyre 1992). Main (1984) argued that sparse species differed from others in needing extensive areas of reservation. This is an unlikely option, given the current land-use of grassy systems. For these reasons, conservation practices need to be integrated with rangeland management if the future of the grassy flora and fauna is to be assured.

4.3 Sustainability issues in the review region

Livestock grazing and pasture degradation

The effects of livestock grazing on the condition of grasslands is the most widely recognised sustainability issue in the region, with the greatest concern being the loss of animal production from the degradation of pastures as a result of damage to soil and changes in pasture composition.

Tothill and Gillies (1992) assessed the condition of all major pasture types in northern Australia, using a rapid appraisal approach in which working groups of experts were consulted to fill out a condition assessment matrix. This matrix recognised three levels of condition based on the extent of soil changes and the amount of ‘undesirable’ species in the herbaceous vegetation. While the floristic indicators are somewhat subjective, the exercise gives the most comprehensive picture of the condition of grassy vegetation. From 15 to 20% of the black speargrass pastures were rated as degraded and from 35 to 60% were deteriorating. *Aristida–Bothriochloa* pastures were in much better condition, with 0 to 10% of the area being considered degraded.

A difficulty in managing the impact of grazing is that there is no direct relationship between pasture degradation and a decline in animal production and producers prefer to look at the condition of their animals rather than examine that of their pasture (Lawrence *et al.* 1994). McIntyre (1996) and Ash *et al.* (1993) found that losses of palatable perennial grasses did not result in less animal liveweight gain in the short-term. Longer-term effects are likely to result from reduced potential productivity in the grazing-tolerant grasses with the consequence of greater chances of forage deficits, further overgrazing and disruption of soil/plant/micro-organism processes. However, lack of sustainability in the long term has not been clearly demonstrated in trials, and producers may be reluctant to take the risks seriously.

In general, livestock grazing on the native grasslands reduces the incidence of the dominant perennial grasses and increases unpalatable species such as *Aristida* spp. At the highest grazing pressures, perennial grasses are replaced by annual grasses and forbs (McArthur *et al.* 1994; McIvor and Scanlan 1994; Orr *et al.* 1994). Such models of vegetation change have tended to reflect consensus opinion rather than deriving conclusions from quantitative data. When pasture changes are quantified, the generalisations and details of change pathways may vary from the models (McIntyre 1996). However, the general trend towards loss of palatable perennial species seems to be widespread and has been incorporated in the South Australian land condition index published by Lange *et al.* (1994).

Soil erosion

Erosion of grazing lands is an issue of moderate to high concern in the sub-coastal catchments relevant to the review region (Anon. 1993). There is high concern on the more extensively managed Burdekin–Haughton catchment which has the most extensive land-use and more marginal systems for production. The related water quality issue—nutrient enrichment of surface waters—is also of moderate to high concern, with greatest concern in the Brisbane Valley, where land use is most intensive overall.

Soil salinity

There has been considerable debate about the risks of salinity associated with tree clearing in the tropics (Hughes 1984; Gillard *et al.* 1989; Williams and Chartres 1991; Burrows 1991a,b; Thorburn *et al.* 1991). Although mean evaporation exceeds mean rainfall in almost all areas, there can be periods during the wet season when rainfall exceeds evaporation. Although Thorburn *et al.* (1991) found no significant recharge of brigalow, pasture or crop catchments when the pastures and crops were established, eucalypt woodlands may use more water than herbaceous communities (Probert and Williams 1986; Williams *et al.* 1997) leading to substantial deep drainage and leaching of salts (Williams and Chartres 1991) where trees are removed or lost. Even relatively small amounts of deep drainage can cause salinisation (Allison and Peck 1987) and effects may take decades to occur (Gillard *et al.* 1989; Williams and Chartres 1991).

Despite this debate, salinisation is listed as an issue of moderate to very high importance in the catchments relevant to the review region (Anon 1993). The exception is the Burdekin–Haughton catchment, where concern is low. The first records of tree decline associated with water-table salting date from 1920 in south-east Queensland. Since that time, there has been a steady increase in reported occurrences of salinity, mostly in central and southern Queensland (Hughes 1979 in Wylie *et al.* 1993). Hughes (1984) found a marked association between clearing of trees and later outbreaks of seepage salting. Developing salinity problems are not always recognised by property owners and there is a complex relationship between the occurrence of salting and tree dieback. Nonetheless, of 161 properties surveyed in 1981–83, 35% reported salting of one or more of their water sources and 28% had identified soil salinisation on their properties (Wylie *et al.* 1993).

Hughes (1984) reported that 8000 ha of land in Queensland were seriously affected by seepage salting as a direct consequence of land clearing and predicted the expansion of these areas and the development of salting in new areas. Preliminary identification of vegetation characteristic of critical intake and runoff areas is in progress, but Hughes argued that stepped-up research was needed to identify potential problem areas more accurately.

Loss of biodiversity

The effects on plant diversity from the elimination of grazing-sensitive species receive little or no attention in the review region. Fauna are also probably being affected by changes to the grassy woodland structure and floristics. However, these processes have not yet been demonstrated in a critical decline of species such as has occurred in NSW and Victoria.

Tree dieback

There is a strong body of work on the occurrence (Landsberg 1988) and ecology (Landsberg and Wylie 1983) of tree dieback in south-east Queensland. In a summary report of surveys in the review region, Wylie *et al.* (1993) noted that dieback occurred in all the 70 shires inspected and was particularly severe in 20 shires of the Fitzroy, Wide Bay–Burnett, Moreton and Brisbane regions and on parts of the Darling Downs. Occurrences were most severe where land management was most intensive (improved pasture and fertiliser) and on older established properties. Of 161 Queensland properties surveyed in 1981–83, 17% recorded no dieback, 65% had slight dieback, 13% moderate and 5% severe. Despite these results little concern has been expressed about tree decline in the review region. In fact tree clearing is the major management activity and there have been concerns about the restrictiveness of tree-clearing guidelines.

Weed invasions

A number of sown pasture species have extensively naturalised in Queensland, spreading from deliberate plantings and accidental introductions. They have invaded non-pasture sites (eg. railway lines and roadsides) and replaced native species across an estimated five million ha in Queensland (Walker and Weston 1990), the most successful in the review region being buffel grass (*Cenchrus pennisetiformis*, *C. ciliaris*), Indian bluegrass (*Bothriochloa pertusa*), couch grasses (*Digitaria didactyla*, *C. dactylon*), Rhodes and panic grass (*Chloris gayana*, *Panicum maximum*) and Townsville stylo (*Stylosanthes humilis*).

Producers are most concerned about woody species which compete with pasture grasses, but there are a range of other weed issues, including the on- and off-site effects of sown pasture species. This is a source of controversy within the research and extension community, but has not yet become an issue for producers.

Soil acidification

Soil pH levels have declined over extensive areas of southern Australia under legume pastures (Williams 1980) and preliminary measurements on tropical pastures have also shown some substantial falls in pH (Jones *et al.* 1996). The cause of acidification is considered to be legume sowing, with the process accelerated by fertiliser. In practice fertiliser use and legume sowing in the review region are very positively correlated, as it is not thought economical to fertilise native grasslands.

4.4 Identifying unsustainable land use in the review region

Is it possible to discover whether current land-use or land-use trends in the review region are unsustainable? This cannot be proven directly, but examples of ecosystem collapse and losses of production elsewhere in Australia provide some indications. The most dramatic example of unsustainable use of grassy woodlands is dryland salinisation which has resulted from tree clearing, and is extensive in Western Australia, Victoria and New South Wales (Conacher and Conacher 1995). Currently, tree-clearing guidelines are being developed for Queensland with the aim of conserving the original tree cover in such a way that tree retention occurs in all regional ecosystems. This is strategically useful for biodiversity conservation. However, there is currently no process in Queensland whereby tree retention is linked preferentially to groundwater recharge and/or discharge areas (Paul McDonald, Department of Lands, pers. comm.). This has been the main method of preventing the development of the problem elsewhere in Australia (LWRRDC 1995). Moreover, the negotiations for vegetation retention percentages are based on no more than best-guess estimates of what is sustainable. This question also requires an examination of other land-use practices that will directly or indirectly affect the viability of retained vegetation.

The fact that tree regrowth is still a major management issue tends to obscure the fact that tree decline is occurring in some areas. To explain this paradox, we hypothesise that the transition from a stage at which regrowth is considered a ‘problem’ (in the sense that control methods are needed) to a stage of actual tree decline (in which the regeneration capacity of the tree population is lost) is extremely rapid. The stability of both stages is probably extremely high and a large amount of effort is required to move from one stage to the other (Fig. 2). In effect, a number of clearing cycles and other management influences (eg. grazing and pasture improvement) are needed to damage the regeneration potential of the woodlands. However, when this happens, the pastures rapidly ‘flip’ from regrowth to tree decline. There are no short-term financial incentives for managers to avoid this transition. On the contrary, they are likely to enjoy the respite from the cost of controlling trees as the ‘tree decline’ stage is entered. Nonetheless, it is costly and difficult to re-establish trees once the regeneration potential is lost.

Is the loss of trees in itself an indication of unsustainability? In terms of ecological processes, perhaps not, if there are no dryland salinity problems. But in terms of fauna conservation, tree loss is a major problem. Figure 3 summarises the pathways leading to the development of tree dieback in rural environments. Landsberg and Wiley (1991) have identified the known, and the hypothesised, factors leading to tree death, and the feedback loops that accelerate the process. It is an extremely complex process that is influenced by the effects of grazing, pasture improvement, use of fertilisers and tree killing. However, the overriding issue is that the intensification of any or all aspects of rangeland use is likely to move the system towards unsustainability.

4.5 Identifying sustainable land use in the review region

Are there levels of rangeland exploitation that are ecologically sustainable? This is a difficult question to answer as there have been only 200 years of European settlement. From the extensive areas of land that are already degraded (Conacher and Conacher 1995), it is evident that many of the practices so far used in Australia are unsustainable. Nor is there any evidence that Queensland has taken a more conservative approach. The relatively better condition of ecosystems in Queensland (in terms of less salinisation and biodiversity loss) reflect the fact that the region is at an earlier stage of land intensification, rather than any strategic planning to reduce impacts.

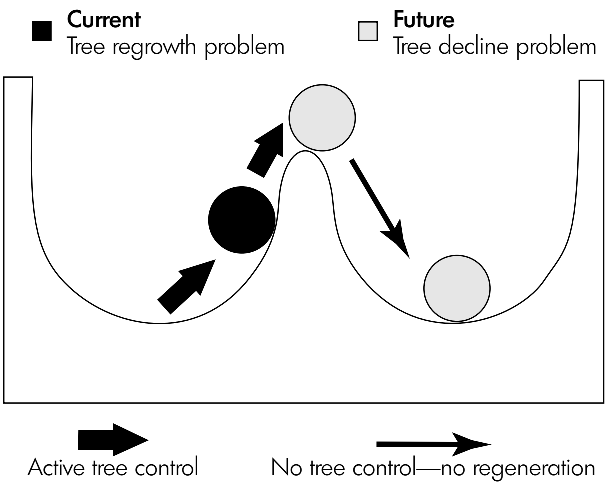

Fig. 2. Conceptual model of the stability of ‘tree regeneration’ and ‘tree decline’ stages. Both stages are very stable (represented by the deep hollows) and considerable management effort is required to move from one to the other (represented by the effort required to push the ball into the left or right hollow). The regrowth stage represents a situation where there is high potential for regeneration—tree thinning to increase pasture production stimulates regeneration. Without continued thinning, tree densities would stabilise and a woodland would form. If sufficient repeated tree-clearing led to a loss of regeneration potential, the ecosystem would move rapidly into a tree decline stage.

The importance of reference areas in determining ecological sustainability

It is likely that there will be levels of exploitation (intensity of land use and extent of particular land-uses) that are ecologically sustainable. This will vary across ecosystem types. However, these levels are still to be discovered. This requires the measurement of impacts by comparing managed areas with reference areas that have not been affected. At broader scales, it would be necessary to compare levels of ecosystem health in equivalent ecosystems that had been subjected to varying degrees of land intensification. Methodologically, this research poses a number of problems:

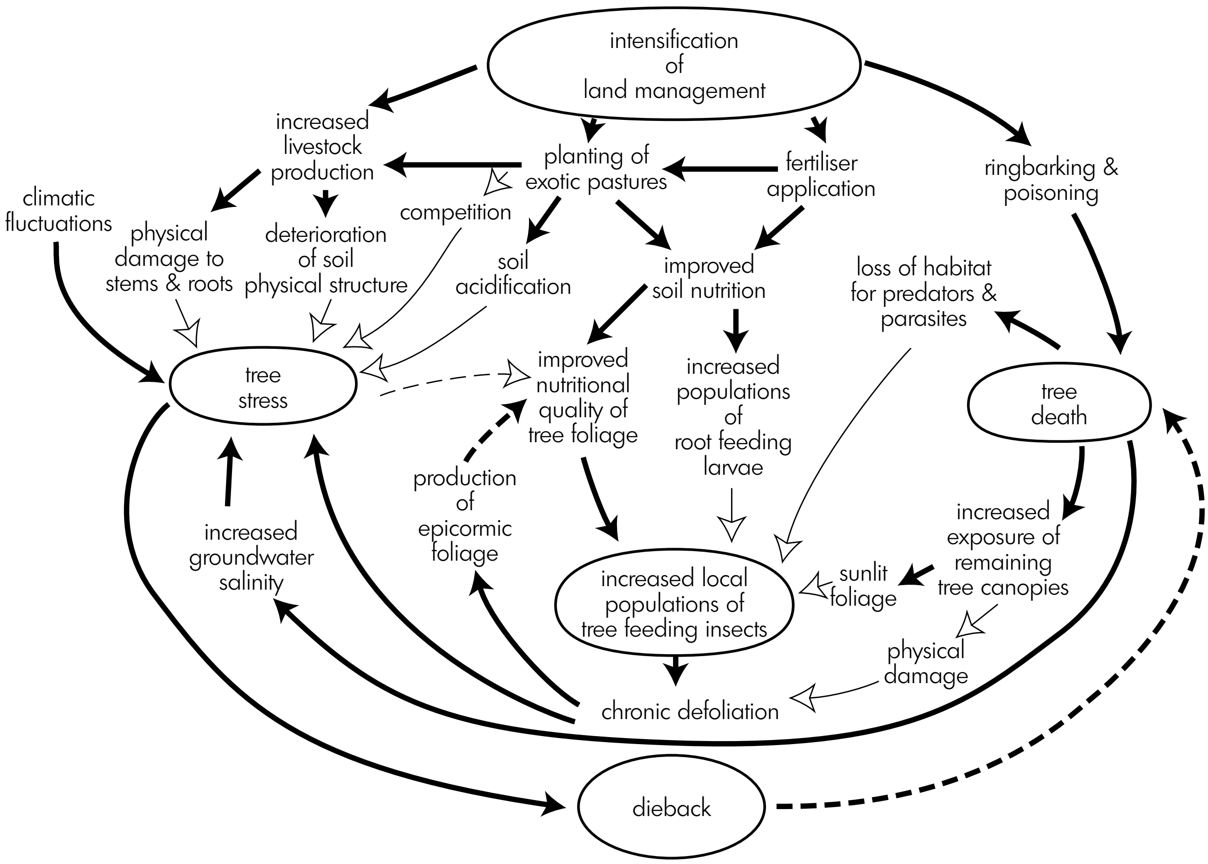

Fig. 3. Model of the processes leading to the development of tree dieback in grazed ecosystems. Black arrows indicate pathways identified through research. White arrows indicate pathways that are more speculative. Modified from Landsberg and Wylie (1991).

- Reference areas may not be sufficient in extent or have had their endogenous disturbance regimes maintained;
- The land available for sampling may not have been managed at the appropriate intensity for sufficient lengths of time;
- Land developments tend to occur as periodic episodes, and be uniform in nature across a particular land type, in which case the appropriate land management ‘treatments’ may not be available.

Ideally, land-use planning would lead to a process whereby the incorporation of such ‘experiments’ would occur when areas are developed.

Limitations to conservation and sustainability in the review region

There are many potential constraints to the achievement of sustainability. Enterprises may not be economically viable in the short term without higher levels of exploitation. There may be cultural constraints to the adoption of a different management philosophy and specific management practices. Planning and policy processes may be inadequate, or ineffectively administered. All these may contribute to land use that falls far short of the ideals of sustainability. For these reasons, it is possible for research and development to examine questions of sustainable land management without having defined an exact end-point. We know its general direction.

Despite the barriers, there is a genuine desire for producers to maintain their life-style and the ecological sustainability of their properties. Our view is that even within the current management paradigm, there is room for managers to move their practices towards better ecological health. Although this may not guarantee sustainability in the longer term, the changes will be in the right direction and increase the probability of sustainable land-use. This is particularly so for the review region, where land use is still relatively extensive. Incremental change may not be ideal, but is probably realistic. According to Nassauer (1995) people modify landscapes according to what they believe their neighbours will think or through cautious assessments of market expectations. Innovation in the design or management of the landscape occurs within the realm of convention. Changing the way people design and manage landscapes will require change in the way people read social characteristics into landscapes. At least initially, these changes will be slow.

5 Indicators of Ecosystem Health and Conservation Status

5.1 Indicators of health and sustainability

Ideal/theoretical indicators

The use of indicators has a long history which Rapport (1992b) described as having three stages: in the 17–18th century when there was recognition of severe local environmental degradation; from the 1960s when ecologists began to understand ecosystem responses to stress; and currently, where there is interest in the transformation of ecosystems in the context of socio-economic and cultural change. The use of sets of indicators is consistent with the need to meet the contemporary demand for a range of ecological and social and economic goals (Hansen 1996).

A great deal has been written about ecological indicators in the broadest sense (eg. Costanza *et al*. 1992; Hamblin 1992; McKenzie et al. 1992; Greenland and Szabolcs 1994). A major exercise in reviewing and defining ecosystem health indicators specifically for rangelands was conducted by the Committee on Rangeland Classification (1994). This group recognised three states of health: healthy, at risk and unhealthy. These were based on the evaluation of three criteria:

- degree of soil stability and watershed function;
- integrity of nutrient cycles and energy flows;
- the presence of functioning recovery mechanisms.

The committee recognised the need for multiple criteria and a strong element of judgement in the overall assessment of health. They recognised ecological redundancy in the sense that different plant species may have similar effects in maintaining ecosystem processes—hence an avoidance of explicit floristic criteria as indicators.

A matrix of indicators for rangeland health

The Committee on Rangeland Classification developed a matrix for the evaluation of rangeland health which is summarised in Table 4. The criteria implicitly incorporate elements of ecosystem resilience in the sense of the system's ability to regain health if the degrading process (presumably grazing) is halted. In this respect, loss of soil function is the most important, followed by nutrient cycling and recovery mechanisms. The authors consider soil stability and watershed function to be the most important criteria because of the irreversibility of soil loss. An 'unhealthy' ranking of this criterion should overrule higher rankings of other criteria at a particular site.

The appropriate spatial scales for assessment were not discussed in the report, although the observations range from individual plant scale to tens or hundreds of metres. A potential difficulty in applying these indicators is that some ecosystems (notably those of limited rainfall) may have naturally patchy plant distributions, large bare areas and substantial areas of soil erosion, all within a healthy ecosystem. These difficulties are overcome by the assessment methods of Tongway (Tongway 1994; Tongway and Hindley 1995) which employ most of the soil and nutrient cycling indicators, but in a spatially explicit context which permits differences between ecosystem types to be accounted for. In general, the patterns of resource loss and capture that can be observed across the landscape are finer-grained in mesic regions than those in arid areas.

Although the indicators described in the matrix represent a milestone for measuring ecosystem function in relation to processes, there is still a need for further research and development (Committee on Rangeland Classification 1994). Refined indicators need to be derived from improved models that describe the temporal and spatial dynamics of rangelands. A difficulty with the current indicators is their narrow focus on the continued use of rangelands for animal production. Other values or land uses which currently exist, or which may become more important in the future, need to be accounted for. It is notable that there are no indicators relating to the maintenance of biodiversity in rangelands. These need to be developed for assessment at property and regional scales.

5.2 Landscape and land-use indicators

Landscape indicators within a land use

Noss (1990) proposed a framework of indicators for monitoring biodiversity based on the hierarchical organisation of ecosystems (regional/landscape;

community/ecosystem; population/species; genetic) and three structural components—composition, structure and function. Each of these twelve combinations of organisational level and component has appropriate indicators and methodologies. Applying this general framework to sustainability in forestry areas, Noss (1993) developed a list of indicators of forest landscape condition that included factors affecting biodiversity maintenance. This list included indicators concerning habitat structure, the intensity and spatial patterning of land use and specific disturbance, and demographic parameters of sensitive species. The list is large, and managers would be expected to use it selectively.

The analogy between native forestry and rangeland use is strong. Both are based on the exploitation of extensive native ecosystems, where much of the vegetation is modified but not destroyed (see the discussion on variegated landscapes in Section 5.3). For this reason, it is simple to translate most of the forestry indicators to equivalent indicators of rangeland health (Table 5) if forest harvesting is considered a land-use analogous to grazing. The rangeland indicators vary in their relevance to property and regional management and these have been shown in the table. The principles covered by the indicators are as follows:

Table 4. The rangeland health evaluation matrix developed by the Committee on Rangeland Classification.

Indicator	State of health		
	Healthy	At risk	Unhealthy
Soil stability and watershed function			
Soil A-horizon	Mostly intact	Present, but distribution fragmented	Absent or associated only with obstructions
Pedestalling	None	Present but on mature plants only; no exposed roots	Most plants and rocks pedestalled; roots exposed
Rills and gullies	Absent or with muted features	Small, and not connected in a dendritic pattern	Well defined, active and with dendritic pattern
Scouring or sheet erosion	None visible	Patches of bare soil and scours developing	Well developed and contiguous
Sedimentation or dunes	No soil deposition	Soil accumulating around small obstructions	Soil accumulating in large barren deposits around large obstructions
Distribution of nutrient cycling			
Plant distribution	Well distributed across site	Fragmented distribution	Clumped with large bare areas
Litter	Uniform across site	Becoming associated with obstructions	Largely absent
Root distributions	Rooting throughout available soil profile	Absence of roots from portions of the available soil profile	Rooting in one portion of the available soil profile
Distribution of photosynthesis	Occurs throughout period suitable for plant growth.	Occurs during one portion of the period	Little or no activity during most of the period
Recovery mechanisms			
Age-class distribution	All ages represented	Seedling and young plants missing	Primarily old or deteriorating plants present
Plant vigour	Plants display normal growth form	Plants developing abnormal form	Most plants in abnormal form
Germination microsites	Present and distributed across the site	Microsites degrading because of soil movement and soil crusts	Most germination and establishment inhibited by these factors

Source: Adapted from Committee on Rangeland Classification (1994).

- That disturbances/land use may have variable effects on the biotic communities and that stratification of disturbance intensities may reduce impacts.
- That land use in one part of the landscape may affect another and there are configurations of land-use intensity that may go beyond the critical levels needed for ecosystem/species viability. These thresholds are not specified as they vary from species to species (With and Crist 1995).
- That sensitive species may need particular habitat elements and that, in addition to the provision of these, the population structure of sensitive species may need specific monitoring.

Regional landscape indicators

Forman (1995) did not develop specific indicators but developed a range of principles to guide optimal patterns of land use, incorporating landscape-ecological attributes. Two of these relate specifically to land-use applications:

> Aggregate with outliers: Land containing humans is best arranged ecologically by aggregating land-uses, yet maintaining small patches and corridors of nature throughout developed areas, as well as outliers of human activity, spatially arranged along major boundaries.
>
> Indispensable patterns: Top-priority patterns for protection, with no known substitute for their ecological benefits, are a few large natural-vegetation patches, wide vegetated corridors protecting water courses, connectivity for movement of key species among large patches and small patches and corridors providing heterogeneous bits of nature throughout developed areas.

These principles have emerged from a decade of work in landscape ecology dealing with land mosaics, and could be applied at property, district and regional level for a range of land uses. The principles are derived mainly from American and European landscapes and therefore often more intensive land uses. In the review region, the maintenance of a few large natural vegetation patches may point to the need for an extensive reservation system, but given the extent of grazing and the need for large reservation areas for grassy ecosystems (McIntyre 1992), these large patches may have to exist as part of the rangeland matrix.

5.3 Appropriate models of landscape change

The dominant model of landscape change—fragmentation

Habitat fragmentation is a conceptual model of human-induced landscape change that has strongly influenced conservation theory and practice in recent decades. Earlier theoretical work on equilibrium island biogeography spawned research on minimum viable populations and on metapopulations (Simberloff 1988). These theories, as applied to conservation, assume that habitat fragmentation occurs. This process involves the destruction of intact biotic communities over large areas, leaving small fragments that are isolated from each other by a matrix of land that is hostile to most of the biota in the fragments (Saunders *et al.* 1991).

The types of conservation management associated with fragmented landscapes include the development of a reserve system as the major conservation strategy, the creation of corridors to connect fragments, and emphasis on protection of rare species (McIntyre *et al*. 1996). In highly fragmented landscapes near populated areas, considerable resources may be directed towards rehabilitation and protection of fragments. Many of these activities are not the most appropriate for effective conservation of biota on rangelands. This is because the fragmentation model of landscape change is not relevant to land that is managed for low intensity animal production.

Variegated landscapes

The effects of exogenous disturbance at the landscape scale can be viewed as a continuum, with intact communities being least affected. With increasing levels of exogenous disturbance, the community is progressively destroyed, to the point where the community no longer forms the habitat matrix and is reduced to fragments, or in extreme cases, completely obliterated. In the intermediate stages of this fragmentation process, the landscape is variegated (McIntyre and Barrett 1992). Here, the community still forms the landscape matrix, but the fabric of the matrix is modified or destroyed in places.

Modification of intact, variegated or fragmented vegetation is the result of exogenous disturbances occurring at a finer spatial scale or disturbances of lower intensity. Modification does not result in complete destruction of the community, although it may affect its species composition, ecosystem processes or structure. The processes leading to modification may result in destruction if the effects are permanent and incremental and the processes persist. Some hypothetical pathways of landscape change are presented in Figure 4.

Arid rangelands of central Australia are largely intact but have been unevenly modified over most of their area by feral and domestic livestock. Similarly, vegetation fragments in a largely cleared landscape can be modified by a wide range of exogenous disturbances, including the off-site effects of other land uses (Hobbs and Huenneke 1992).

Examples of variegated landscapes include forestry areas, where extensive wooded areas form the landscape matrix. In this case, previously harvested regrowth forests represent modified communities, while small areas of permanent clearing and exotic plantations represent patches that have been destroyed. Rangelands may be considered variegated if there are areas of destruction (human settlement, cropping) as well as areas of modification (sown pasture, livestock grazing, fertilisation).

Table 5. Indicators of forest landscape condition (after Noss 1993) and equivalent indicators that could be used in the review region, where rangeland use is subjected to land-use intensification. ✓ denotes indicators that are most relevant to the property (Pty) or regional (Regn) scale in the review region.

Forest landscape condition indicator	Equivalent rangeland landscape condition indicator	Relevant scale	
		Pty	Regn
Forest age (a product of tree harvesting practices)	**Grazing intensity**		
Frequency distribution of age classes for each forest type and across all types	Distribution of grazing intensities for each land type and across all types	✓	✓
Average and range of tree ages within defined seral stages	Average and range of tree and perennial grass ages and density within defined grazing intensities	✓	
Forest structure	**Woodland structure**		
Ratio of areas of natural forest of all ages to areas in clear-cuts and plantations	Ratio of areas of natural woodland pastures of all grazing intensities to areas of cropping and sown and/or fertilized pasture	✓	✓
Abundance and density of snags, downed logs and other defined structural elements and patches	Abundance and density of fallen timber, litter and dead standing trees.	✓	
Spatial dispersion of structural elements and patches	Spatial dispersion of fallen timber, litter and dead standing trees.	✓	
Foliage density and layering (profiles) and horizontal diversity of foliage profiles in stand	Presence and diversity of shrub and small tree species	✓	
Canopy density and size and dispersion of canopy openings	Canopy health—size and dispersion of incidences of tree crown dieback and mistletoe infestation	✓	
Patch size and related variables	**Patch size and related variables**		
Patch size frequency distribution for each seral stage and forest type and across all stages and types	Patch size frequency distribution for each grazing intensity and land type and across all intensities and types	✓	
Patch size diversity index	Patch size diversity index	✓	
Size frequency distribution of late-successional interior forest patches	Size frequency distribution of lightly-grazed, unsown, unfertilized, uncleared vegetation patches on all land types	✓	✓
Total amount of late-successional interior forest habitat	Total amount of the above patches	✓	✓
Total amount of forest patch perimeter and edge zone	not applicable		

Table 5. Continued.

Forest landscape condition indicator	Equivalent rangeland landscape condition indicator	Relevant scale	
		Pty	Regn
Forest patch perimeter: area ratio	lightly-grazed, unsown, unfertilized, uncleared vegetation patch perimeter: area ratio	✓	
Edge zone : interior zone ratio	not applicable		
Fractal dimension	Fractal dimension	✓	✓
Patch shape indices	Patch shape indices	✓	✓
Fragmentation indices	Fragmentation indices	✓	✓
Patch isolation and related variables	**Patch isolation and related variables**		
Interpatch distance (mean, median, range) for all forest patches and late successional forest patches	Interpatch distance (mean, median, range) for all lightly-grazed, unsown, unfertilized, uncleared vegetation patches	✓	✓
Juxtaposition measures (% area within a defined distance from patch occupied by different habitat types, length of patch border adjacent to different habitat types)	Juxtaposition measures for high and low intensity land uses.	✓	✓
Structural contrast (magnitude of difference between adjacent habitats measured for various structural attributes)	Structural contrast (magnitude of difference between adjacent habitats measured for various structural attributes)	✓	
Fire regime	**Fire regime**		
Frequency, return interval or rotation period	Frequency, return interval or rotation period	✓	
Areal extent	Areal extent	✓	✓
Intensity or severity	Intensity or severity	✓	
Seasonality or periodicity	Seasonality or periodicity	✓	
Predictability or variability	Predictability or variability	✓	
Roads	**Roads**		
Road density for different classes of road	Road density for different classes of road		✓
Percentage of forest in roadless area	Not applicable		
% and areas of existing roadless areas retained and the end of each decade	Not applicable		
Miles of road constructed, reconstructed or closed at the end of each decade	Not applicable		
Amount of roadless area restored through permanent road closures and revegetation	Not applicable		
	Areas affected by cropping/sown pastures/ salinisation/tree decline		
	% of area affected at the end of each decade	✓	✓
	% increase of area affected at the end of each decade	✓	✓
Sensitive species	**Sensitive species**		
Demographic parameters	Demographic parameters	✓	
Genetic and health parameters	Genetic and health parameters		✓

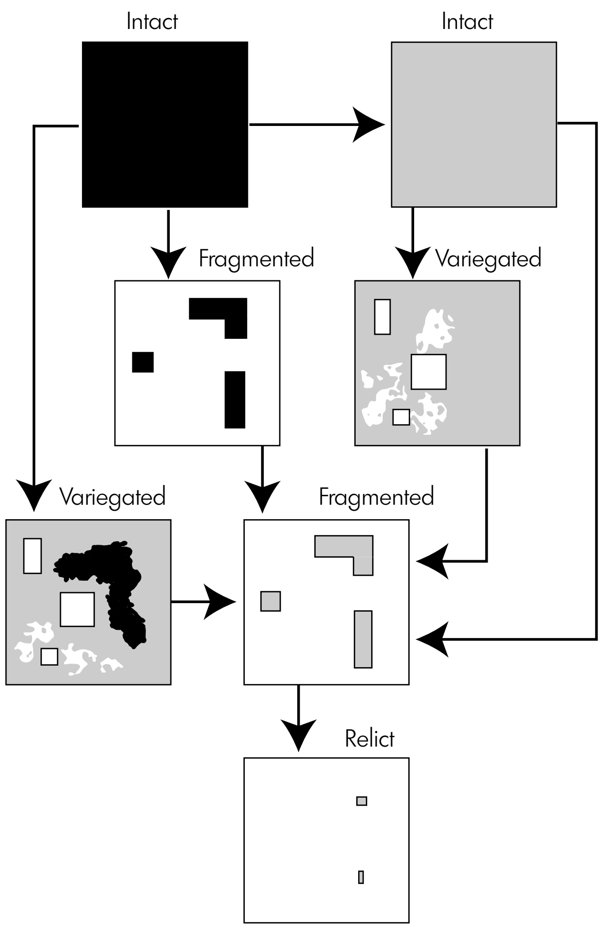

Fig. 4. Various pathways of landscape change that can occur in Australian landscapes, showing intact, variegated, fragmented and relict landscapes. Shading indicates where vegetation is destroyed (white) or where remaining vegetation is unmodified (dark shading) or modified (light stipple). Re-drawn from McIntyre *et al*. (1996).

Why is it important to differentiate between different states of landscape change?

In eastern Australia, grassy ecosystems have been variously affected by pastoral uses, depending on the length of time disturbances have prevailed and the intensity of land-use. In Tasmania, Victoria and some of New South Wales, grassy ecosystems have become fragmented (Lunt 1991; Kirkpatrick and Gilfedder 1995; Prober 1996). In other parts of New South Wales (eg. the northern tablelands), the landscapes are variegated (McIntyre and Lavorel 1994a) and in the review region, the grassy ecosystems are generally variegated and sometimes intact.

It is important to differentiate between different landscape states because the key to the sustainable management of grassy vegetation is to maintain it as an extensive ecosystem. The vital feature of intact and variegated landscapes is the existence of the vegetation as a matrix. Experience elsewhere has demonstrated the vulnerability that fragmentation brings to ecosystems. This is demonstrated through the appearance of salinisation, tree dieback and losses of genetic diversity (Prober and Brown 1994). By imposing a ‘fragmentation’ mind set and diverting resources only into small reserves and the rarest species, we may unwittingly permit the process of fragmentation to proceed in otherwise intact habitats. In that process, many relatively common species of birds and plants with moderate tolerance of disturbances will be pushed towards endangerment (McIntyre and Barrett 1992; McIntyre *et al*. 1992). Inappropriate use of the fragmentation model is still widespread in Australia and reflects the dominance of this paradigm in the conservation literature world wide.

Conservation strategies for variegated landscapes

In developing strategies for variegated landscapes, the ‘binary’ approach to conservation must be challenged, both at regional and at property management level. A common view is that land is either reserved for nature or for human exploitation. An alternative view is that, to a significant extent, nature conservation can be integrated with livestock production. Fortunately this view is aligned with more recent trends towards acknowledging native pastures as economically productive.

The goal of conservation in rangelands is to maintain connected habitats for as wide a range of native species as possible. A prerequisite for this is quantifying the way disturbances affect the community in question, particularly in relation to management variables. It is then possible to stratify intensity and types of management in such a way as to ensure that a sufficient variety of habitat states is available across the landscape for the majority of species to persist. Such stratification needs to take into account the different spatial scales at which species are operating. Planning also needs to recognise the existing areas of highest conservation value and attempt to minimise threats to these areas. This is because variegated landscapes will contain a component of the most disturbance-sensitive biota, which will effectively be ‘fragmented’ within a modified landscape matrix.

Rather than attempting to manage the species assemblage as a whole, groups of species that share habitat requirements (eg. intolerance of particular types of disturbance) need to be identified. This would be an appropriate compromise between the single-species approach to plant conservation, which tends to be impractical, and a whole community approach which fails to take into account the varying requirements of the component species.

Some management regimes may maximise animal production, others may maximise biodiversity. These two factors could potentially conflict. However, they may also complement each other within a sustainable management system: ungrazed areas along water courses may protect water quality, retained stands of trees may shelter stock, while lightly grazed areas can provide emergency fodder during droughts. Achieving balanced land-use across rangelands will protect biodiversity as well as a range of natural resources and ecosystem services. For some particularly sensitive species or vegetation types, reservation may be the only option for survival. The risk is that with prevailing conservation priorities, these will continue to be the primary focus at the expense of the majority of species. Without integrated management across the entire landscape, our net losses will be greater in the long run.

Property management planning

The on-ground implementation of management principles for variegated habitats is most likely to be achieved through a property management planning process. Seven vegetation management guidelines have been proposed for property planning, based on the conservation principles relevant to variegated landscapes (McIntyre 1994):

- take into account existing vegetation condition;
- avoid juxtaposing high-intensity land uses or severe disturbances with areas of conservative management;
- stratify management intensity so that areas of moderately intensive pasture utilisation act as buffers between extremes;
- combine low-intensity management for herbaceous species with management for tree regeneration, erosion control or protection of waterways;
- consider using low or medium-intensity pasture use to connect areas of conservation significance;
- manage areas of low-intensity land use in order to maintain the diversity of herbaceous vegetation;
- manage for the entire flora by adopting a diversity of approaches as different management methods will favour different species.

Birds show similar patterns of response to disturbance as herbaceous plants (Barrett *et al.* 1994) and are likely to also be catered for with this style of property planning. The application of these guidelines needs to be examined on some case-study properties, to test their practicality and efficacy and to make any necessary refinements. It is also essential to recognise the role of regional planning and management to account for the ecological processes (eg. hydrological processes, fauna that use extensive landscapes) that operate at broader spatial scales than within properties.

6 A Research Approach to Meet Information Needs

6.1 Information needs

The following accounts of perceived information needs are based on informal canvassing of views and interpretations of the current management environment.

Information needs for regional planning

General needs

Information used in the regional planning process tends to be at the broadest scales and concerned with natural resource inventory and description, combined with patterns of land use. In terms of ecological processes, there is some interest in factors contributing to land degradation. Much of the current natural resource inventory is put together by state departments. Systematic inventory data collection and mapping has been, by necessity, collected at broad scales (eg. aerial photography and remote sensing), using geological information, vegetation structure and tree composition to describe regional ecosystems. This approach does not handle modified ecosystems well, and areas tend to be classified simply as vegetated or cleared, neglecting those nuances of management effects that are very important in grassy ecosystems in rangelands. More recently the development of tree-clearing guidelines in Queensland has raised questions of the minimum area and configurations of retained vegetation that will ensure its viability.

Potential vs actual information needs

The demand for information in some areas is not at present high, but it may well increase. Local government planners have only recently moved to consider development decisions in a regional context. The processes of environmental impact assessment have not tended to place rigorous demands on our knowledge base. The review of the use of leasehold land may also lead to changed lease conditions—from a previous emphasis on infrastructure and production factors, to issues of vegetation conservation and land-use impacts.

Information needs of property managers

Case-study work with producers in the Brigalow Belt has raised issues of on-farm biodiversity planning. The information needs of producers which Kay Dorricot has described (pers. comm.) include basic inventory: they want to know the plants and vertebrate animals on their farms and what the conservation status of these species is. They also need to identify the areas of their properties with the highest conservation value and to understand the effects of management on such values. At a more fundamental level, they wish to understand what biodiversity is, what are its benefits and what creates a 'clean/green' image. Our own work had also discovered a need for basic training in plant identification, both for production and conservation purposes.

Property management planning

Not all producers will be interested in taxonomic details, but they may be interested in conservation in a general sense, and this is where land use (both current and historical) can provide useful predictors of current status and possible changes in the future. Other methods for the rapid assessment of conservation value could be the use of structural indicators such as dead and living trees, shrub layers and tussock grass structure. Where there is a need for rapid assessments and general guidelines for property planning, the variegated landscape guidelines discussed in Section 5.3 become highly relevant, as they can be implemented without a detailed knowledge of the biota. As a research approach, the identification of functional groups in relation to management factors allows an appropriate stratification of management intensities to be determined.

Indicators of ecosystem health and productivity

Some producers will have no interest at all in biodiversity but will be interested in the plant and animal production aspects. We suggest that these will be encapsulated in the management principles for biodiversity maintenance, which are likely to be more conservative. However, simple indicators of soil and pasture condition are important for paddock monitoring, which may be conducted independently of property planning exercises. Lawrence *et al*. (1994) have also shown that there is a need for clear demonstrations of the link between pasture condition and profitability. If there are no short-term links (as current data suggest) and long-term links are difficult to demonstrate, the only course is the difficult one of appealing to the producers' altruism.

Tree clearing guidelines

With the development of tree-clearing guidelines, there has been much demand for definitive answers to some of the most intractable ecological questions:

- What proportion of tree cover needs to be retained for sustainable land management?
- What are appropriate tree configurations—scattered, in strips or in blocks?
- What is the minimum width for a viable retained strip?
- How wide should a riparian buffer be?

These questions are difficult because the viability/sustainability issue involves time spans of many decades. In addition, there is a complex of ecological interactions which potentially determine the viability of a particular stand—the biology of the plants, the adjacent land-uses, and hydrological phenomena, as well as the climate and other stochastic factors.

Developing a vision for landscape management

Because the goals of sustainability tend to be abstract and intangible, we need to create a vision which planners, researchers and managers can adopt.

> ... persuading someone to accept the reasonableness of a decision must sometimes consist of more than convincing the recipient of the rationality of the choice. (Shelly and Bryan 1964)

There needs to be an element of inspiration in the images created. Some might argue that economic circumstances are the sole determinants of landscape configuration but research has shown that people have consistent preferences for particular landscape features. At a more abstract level Kaplan and Kaplan (1982, in Nassauer 1995) theorised about human landscape preferences and our ability to cope with stress in the environment. Landscapes preferred by humans were those that offer exploration: including complexity (rich, intricate elements) and mystery (with something yet to be discovered). Landscapes also need to be understandable: both coherent (orderly) and legible (accessible to finding one's way). More specifically, Nassauer (1995) found an overall preference for savanna-like landscapes with canopy trees or water views and allowing views across the landscape. All these elements are eminently suited for inclusion in sustainable landscapes and have similarities to Forman's (1995) principles for ecologically optimal patterns of land use.

A well as providing factual information and educational material, research and extension specialists will have to share the elements of a vision which managers can develop and work towards themselves. This will make demands on current resources and on the techniques of research communication.

6.2 Summary of review sections 2–5

Sustainability concepts

As a catch-all term, sustainability reflects society's general concern for the impacts of humans on the environment and the implications for our longer-term social and economic well-being. At the broadest level, sustainable development is the ability to meet the needs of the present, without compromising the ability of future generations to meet their own needs.

Because of the value systems inherent in the concept, and the range of disciplines that are bought to bear on it, there are contradictions in the application of sustainable land-use. The reality of management is therefore likely to be the outcome of compromise between parties who have different views of the issue.

Such contradictions are apparent in the consideration of ecological sustainability in rangelands. The intensification of land use that may be associated with the economic and social sustainability of rangelands (eg. severe grazing, pasture improvement) may threaten longer term ecological sustainability if it exceeds threshold levels.

Biodiversity and sustainability

The role of biodiversity as a factor in the sustainable land-use of rangelands is complex. The extensive natural communities that comprise rangelands gives them an inherent value that is recognised by society. The role of this diversity in maintaining ecological stability is not clear cut, but the more species that are removed from an assemblage, the greater the chance of losing ecosystem functions.

In grasslands, it appears that species diversity at finer scales does contribute to stability of the community. The adequate representation of different functional types at the community and landscape scale is likely to provide an important buffer against climatic, land-use and management changes.

Ecosystem health

The concept of ecosystem health is useful in that it describes the current condition of the ecosystem rather than predicting long-term outcomes. Schaeffer *et al.*'s (1988) definition of ecosystem health recognises that departures in the productive capacity from the inherent level of the ecosystem can be indicators of decline in health. This is relevant to rangelands in the review region, where both increases and decreases in the productive capacity can be associated with unsustainability.

Ecology of grassy ecosystems

The eucalypt grassy ecosystems of humid and sub-humid regions form a continuum along the entire east coast of Australia, and there are similarities in the floristics, structure and functioning both of temperate and of tropical systems. In the review region, grasslands, woodlands and, to a lesser extent, grassy forests occur on a wide range of the more fertile geological substrates.

Although endogenous ('natural') disturbances serve to maintain diversity in grassy systems, exogenous disturbances (heavy livestock grazing, soil disturbance, fertilisers, tree clearing and changed fire regimes) can reduce diversity, particularly of sensitive species. Intensification of land use (eg. pasture improvement, cropping) tends to result in combinations of different disturbances occurring as regimes.

Apart from the direct effects on the biota, intensification can lead to changes in the habitat structure and result in indirect effects on fauna and flora. Exotic species, deliberately and accidentally introduced, are aided by intensification, and can also reduce the diversity of native biota. These phenomena have not been studied in the review region.

Exogenous disturbances also influence ecosystem processes through direct effects and higher-order interactions. These can increase primary production and livestock production, but may disrupt ecosystem services through alterations to carbon and nutrient cycling, hydrological cycles and community dynamics.

Sustainability in the review region

In the review region, intensification of use through pasture sowing is limited in comparison with southern regions. While the potentially damaging effects of grazing on pasture composition and soil condition are well recognised, there is less awareness of other declines in ecosystem health such as dryland salinity, tree dieback and soil acidification. These are the most serious threats to sustainability in southern regions and land-use trends in the review region make it likely that they will become issues there in the future.

Although there is little information on the biological status of grassy ecosystems in the review region, their protection in conservation reserves is generally poor. Because of the extensive nature of the systems and their current land-use, it is suggested that a reserve system alone, while necessary, will not guarantee the long-term health of ecosystems.

The related concerns of loss of biodiversity and loss of ecosystem function could both be tackled by limiting (the intensity and/or extent of) intensified land-use. The greatest unknown is determining the amount and configuration of these land uses that is necessary to assure long-term sustainability. This is a relevant question at property, landscape and regional level.

The question of sustainability is complicated by the long time-lags that may be experienced before ecosystem collapse is evident, and the complex emergent processes that may appear at spatial scales differing from the ones that the management units may be operating on. However, on the basis of our existing knowledge, we can recommend some immediate and achievable research and management actions. We know what direction we need to move in, even if the destination is not clearly marked.

Strategies for research and management

A number of indices of ecosystem health are presented for investigation, refinement and use, at both small and large scales. These relate to production, ecosystem process and biodiversity.

Rather than focusing on the management of rare species and the process of biological fragmentation, conservation research, we argue, should be directed towards the integration of conservation management with rangeland use. Priority needs to be given to maintenance of the vegetation matrix, and to i) refining existing strategies for stratification of land-use intensities and ii) determining acceptable levels and configurations of intensive land-uses, that will achieve a balance for ecosystem health and production.

The information needs of planners and land managers are similar in that they need basic information about the ecosystems, their biota and their functioning, as well as the impacts of management. This is necessary to understand the problems and initiate basic management. At the operational level, they need indicators for the rapid assessment of ecosystem health and conservation status at a range of spatial scales. Following from this is the requirement for guidelines for appropriate levels and types of modification and suitable spatial configurations of land use.

Beyond the bare facts, managers need to understand why modifications of their actions may be both desirable and necessary and to share a vision for a landscape that is both desirable and sustainable. This assumes a certain degree of commonality between the interests of ecosystem health and the land managers. Where this commonality does not already exist, it needs to be encouraged.

6.3 Determining appropriate research methodologies

Strengths and weaknesses of different research approaches

In a review of experimental methods in ecology, Diamond (1986) listed three main experimental approaches:

- Laboratory experiments (LEs)—perturbations are produced by the experimenter in the laboratory/ glasshouse;
- Field experiments (FEs)—perturbations are produced by the experimenter in the field;
- Natural experiments (NEs)—unplanned perturbations occur in the field. Two types are distinguished: natural snapshot experiments (NSEs) are comparisons of communities assumed to have reached a quasi-steady state with respect to a perturbing variable; natural trajectory experiments (NTEs) are comparisons of the same community at various times before, during and after a perturbation (eg. a volcanic eruption or storm).

Each of these methods differs greatly in the way problems can be dealt with addressed, and it is essential to understand these strengths and weaknesses for the development of appropriate research programs. These are summarised in Table 6.

Most of the rangelands research relevant to the review region has consisted of laboratory experiments (usually as glasshouse experiments), or more commonly field experiments. There have been very few natural trajectory experiments, presumably because large one-off perturbations are not particularly relevant to the ecosystems and because long-term observations are generally not funded. The strengths of laboratory and field experiments is that independent variables can be best controlled through the active regulation or selection and layout of field sites (site matching).

A range of experimental approaches is required

Laboratory and field experiments are most effective in determining the specific effects of individual variables. However, they are limited in the spatial and temporal scales measured, and they lack realism and generality. Natural snapshot experiments are the opposite—poor identification of individual variable effects—but they can observe spatial and temporal scales unavailable in other approaches and provide the best levels of realism and generality. Natural snapshot experiments were used in the northern tablelands to determine the effects of disturbances arising from management on the structure, composition and conservation status of grassy vegetation (McIntyre and Lavorel 1994a, b; McIntyre *et al*. 1995). We suggest that some emphasis be put on them, as many of the sustainability issues in the

Table 6. Strengths and weakness of different types of experiments in ecology.

Experimental attribute	Type of experiment[a]			
	LE	FE	NTE	NSE
Regulation of independent variables	Highest	Medium / low	None	None
Site matching	Highest	Medium	Medium/ low	Lowest
Ability to follow trajectory	Yes	Yes	Yes	No
Maximum temporal scale	Lowest	Lowest	Highest	Highest
Maximum spatial scale	Lowest	Low	Highest	Highest
Scope (range of variables)	Lowest	Medium/ low	Medium/ high	Highest
Realism	None/low	Higher	Highest	Highest
Generality	None	Low	High	High

[a] LE = laboratory experiments; FE = field experiments; NTE = natural trajectory experiments; NSE = natural snapshot experiments

Source: Diamond (1986).

review region (and elsewhere) are manifest at temporal and spatial scales beyond the capacity of field experiments to deal with them. However, the best approach of any research program is to combine a range of experimental approaches, as the information obtained will be complementary.

6.4 Research questions and relevant scales for their investigation

Biological inventory

State and regional scale mapping

Broad-scale mapping of regional ecosystems is being conducted by state departments and forms the foundation for land-use planning. It is also important for research planning and design, as it provides the regional context and allows representative or extensive ecosystems to be chosen for smaller-scale studies. The assessment of some land-use patterns in relation to particular ecosystems is also appropriate at this scale. By combining land-use and ecosystem inventory it is possible to determine what ecosystems are threatened and, in some cases, what taxa are also threatened. The reservation status of ecosystems can also be established. As extensive areas need to be inventoried, most of this information has to be obtained remotely, with limited ground truthing.

Regional, district and landscape scales

Information on the distribution and status of some biota (eg. fauna and ground flora) can not always be obtained remotely, unless clear correlations have been established between remotely mapped habitat data and the occurrence of these taxa. More detailed ground surveys can be used to assess species distributions directly and establish the presence or absence of habitat correlates. Because they are more intensive, the surveys need to be strategically conducted on representative areas for specific groups of organisms and to establish particular habitat patterns. These surveys, if designed appropriately, can also answer some of the specific questions outlined below.

What effects do disturbances and land-uses have on the conservation status of ecosystems?

As described above, broad-scale mapping can be used to document ecosystem status, but often only in terms of larger physical changes such as destruction through clearing and modification through shrub invasion. However, there are other land-uses and types of modification that cannot be observed by these methods such as the effects of grazing management and pasture sowing on the status of herbaceous communities and fauna. Natural snapshot experiments can be used to examine some of the effects of management on the status of communities. This requires a knowledge of the management history of the sites being sampled—information that is not always available, or is so time-consuming to gather that it limits the sites sampled and the usefulness of the experiment.

Where there are broad correlations between a general land-use and management, land use can be substituted for a management history: for example, comparison of conservation reserves, stock routes and commercial pastures provides a range of stocking intensities imposed over significant time scales. In these experiments, it may also be necessary to document species rarity during the sampling process. This is because vulnerable species may not be recognised as such in statewide lists of endangered species. This is particularly the case for herbaceous grassland species, which are poorly documented and may be widespread in distribution but occur sparsely within their range.

Particular land-uses often involve correlated disturbances. With appropriately stratified sampling and analytical techniques some of the disturbances can be identified as individual effects in natural snapshot experiments. The best way of determining the effects of individual disturbances however, is to set up manipulations in field experiments. A major limitation of this approach will be to observe the effects for a sufficient amount of time or over appropriate spatial scales.

What effects do disturbances and land-uses have on the functioning of ecosystems?

Where ecosystem functioning can be assessed by sampling a number of sites over limited areas, information on function can be collected as part of the natural snapshot experiments described above. Most of the health indicators in Table 4 could be used in this way. Data on local erosion, nutrient cycling and recovery mechanisms in herbaceous vegetation can be aggregated to build a regional picture of ecosystem processes in relation to management factors.

However, where the ecosystem function is an emergent property at the regional or landscape scale, aggregation of individual site data cannot predict the effects of disturbances. Tree dieback, soil salinity and some hydrological processes are examples of disruptions to ecosystem processes that involve spatially explicit, landscape-scale considerations. Specific studies at relevant scales are necessary to examine these—for example, identification of critical areas for vegetation retention to avoid dryland salinity.

What are the effects of exotic plant species?

The questions and experimental approaches described above are also relevant to the topic of exotic species and the effects of pasture sowing on herbaceous communities. Several questions are pertinent to the issue of exotics and their effects: What are the most abundant exotics? Which sown species are the most persistent and have spread beyond sown areas? Which species appear to have the greatest impact on native and rare plants? What are the biological attributes of these species? What is the composition of sown pastures at different times after sowing? These questions can be most easily answered by classifying the plant assemblage into exotic, native and rare native components and analysing the relative change in the three groups over a range of habitats and disturbances/land uses.

Indicators of ecosystem health

Of the possible indicators of ecosystem health listed in this review, some are already reasonably well established (eg. Table 4 and the indicators of Tongway 1994) and can be used to establish links between ecosystem health, management factors and other attributes of interest such as productivity and diversity, as described above. Others are proposed indicators that may need confirmation, or indicators that are difficult to measure and need simplification or substitution. The landscape indicators of Table 5, regional land-use indicators of Forman (1995) and indicators of fauna habitat (Section 3.3) fall into this category. These will need to be examined specifically or incorporated as potential indicators in parallel with other studies.

Spatially explicit studies

What are the patterns of biodiversity in variegated landscapes?

There has been no spatially explicit examination of variegated landscapes and the patterns of diversity associated with them. These could usefully be conducted at the paddock, property and district scale, although the level of detail would have to be scaled to the total area over which data were collected. Within paddocks, the effects of grazing behaviour would be apparent and if comparable areas, subjected to different grazing regimes, were available, the impact of grazing on species richness, species turnover and landscape diversity could be quantified. At the property and district scale, the effects of land use on landscape diversity could be quantified. Such studies would begin to answer the question of what spatial arrangements of land uses are optimal for ecosystem health, for example, the role of riparian vegetation in protecting water quality and the identification of incompatible arrangements of land use.

What thresholds of land use are associated with emergent problems?

This is an important question for property and regional planning, but difficult to research because of the long time-lags that may be experienced between the imposition of land uses and the manifestation of unsustainable land-use. Methodologically, the question might be approached by comparing landscapes with different levels of a particular land-use. However, the land uses would have to have been imposed for equivalent (long) periods of time. Some idea of thresholds may be obtained by comparing extent and intensity of land use in grassy ecosystems from southern regions, with and without evidence of emergent problems such as tree decline and dryland salinity.

Synthesis

Functional groups

A methodology to link community change along disturbance gradients to biological attributes of the organisms in the ecosystem has been proposed (Lavorel *et al.* 1997). This involves the synthesis of community data with information on the characteristics of species from different response groups. It would allow a degree of generalisation to be achieved in the description of community change as well as determination of the range of disturbance intensities required to enable all functional groups to exist across a landscape. Stratification of disturbance intensities is a strategy to maximise the range of species able to survive in rangelands, without requiring specific management of individual plant species—an impractical approach for extensively managed landscapes.

Linking processes

There are questions of particular interest that involve an understanding of the links between processes at the plant community, landscape and regional scale. They also concern the identification of links between biodiversity status and other changes to ecosystem status such as pasture and soil condition and productivity, as well as energy, nutrient and water fluxes. Synthesis of the results of research described above will provide answers to some these questions. For example, simultaneous measurement of the impact of land use on the conservation status and land condition of different ecosystem types will determine which communities are most susceptible to degradation and in need of special protection. However, not all questions will be resolvable.

Explicit research approaches to understand these links are likely to develop over time as the issues and specific questions become more clearly articulated. At the moment, methodologies to study the linking of scales for specific management problems are poorly developed.

Management

While ecological research can start to define what levels and configurations of land use might be ecologically sustainable, the research process needs to take account of the realities of current land-use patterns and practices and the economic and social limitations to the adoption of new management. We suggest that surveys and on-farm experimental work can help to achieve this. Links with socio-economic studies at the property and regional levels are also essential to develop management principles that are both practical and acceptable to stakeholders.

Conclusions

There are many issues in the area of sustainable rangeland use. As information is currently restricted in quantity, there are plentiful research opportunities.

Despite the broad range of research outlined above, we have still been selective in our choice of topics and methodologies. We have intentionally not included studies of individual species and populations, as the priority is on the understanding of community dynamics. While we recognise that some key species may be subsequently selected for specific studies, we suggest that a priority for community and ecosystem aspects will be more cost-effective. This is in keeping with the limited information about the ecosystems in the review region, their extensiveness and the coarse level of management that they are subjected to. For similar reasons, we propose that a focus on higher plants and larger fauna is more appropriate at this stage than invertebrate or microbial studies. Although the smaller fauna are critical for aspects of ecosystem functioning, the paucity of knowledge about microbial and invertebrate ecology means that the research is less likely to be translated into management principles at the property and regional scale. Essentially, we are assuming that other indicators of ecosystem health such as soil and vegetation condition and land use, will function as ‘umbrella’ indicators for invertebrate and microbial ecosystem health.

Even from amongst the list of topics outlined, priorities will need to be set as no single research group can deal with all the issues exhaustively. There will be trade-offs between:

i) the geographical area covered by the research;
ii) the depth at which questions can be approached;
iii) the range of questions that can be tackled;
iv) the range of temporal and spatial scales to be investigated.

We propose to maximise iii) and iv) at the expense of i) and ii) as a practical compromise that will produce useable research outcomes. With respect to i), it is unrealistic to cover the review region or even an entire bioregion systematically by using the methodologies required to unravel some of the questions relating to land-use effects and biodiversity. This is because the complexity of data-processing and analysis limits the size of the data set that can be processed into meaningful results. In addition, natural snapshot experiments need to be conducted over a relatively short period to control for inter- and intra-seasonal variation. A third consideration is the extreme level of floristic detail required for biodiversity assessment, whereby multiple species groups (that are widely used in pasture research) need to be minimised (ie. the degree of lumping reduced). This considerably expands the field and laboratory effort needed to collect, process and identify specimens.

Given the type of data collection proposed, there are many potential research questions that could be raised. While no single experimental design could tackle all the issues outlined above, a set of complementary natural snapshot experiments, conducted at different spatial scales, will shed light on a range of issues relating to species inventory, patterns of diversity, indicators of ecosystem health and land-use impacts. We suggest that it would be a poor use of data to focus only on very few questions at great depth. With regard to the spatial scales of data collection, we suggest that there is more to be gained from data collection at a range of scales (regional, property, paddock and plot), in order to collect information that is both sufficiently detailed to understand the processes, and sufficiently broad to provide perspective and context.

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