

**Rangelands Monitoring:**

# **Developing an Analytical Framework for Monitoring Biodiversity in Australia's Rangelands.**

*Background paper 1.*

*A Review of Changes in Status and Threatening Processes.*



**TROPICAL SAVANNAS CRC**

Cooperative Research Centre for the Sustainable Development of Tropical Savannas

**National Land & Water Resources Audit**

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## **SUMMARY**

- (1) This document aims to provide background information for the establishment of a rangeland-wide monitoring program for biodiversity. To that end, it seeks to highlight what components of biodiversity have undergone change in status, are currently undergoing change in status, or may be predicted to be most susceptible in the future. It also seeks to provide an assessment of the factors which are associated with change in status of biodiversity, and to indicate in which regions biodiversity has changed, is changing, and is likely to change.
- (2) The most substantial changes to rangeland biodiversity have been in the terrestrial mammal fauna, which has suffered catastrophic decline in many rangeland areas. This loss has particularly affected larger dasyurids and rodents, bandicoots and smaller macropods. In most rangeland regions, loss occurred in an initial rapid response to pastoral settlement. In most rangeland regions, declines continued well after this initial shock, and are continuing.
- (3) The bird fauna of many rangeland regions has suffered regional extinctions and pronounced change. The proportion of species in decline is less than that for mammals, but is nonetheless substantial. Declines appear to be continuing across much of the rangelands. In contrast to the mammal losses, there is not a tight ecological segregation of declining versus non-declining species.
- (4) Counterpointed to these declining birds and mammals, some species have increased in abundance and distribution across the rangelands. Typically these have been favoured by the provision of artificial water sources and by vegetation change associated with pastoralism. Examples include crested pigeon, galah and large kangaroos. In general, increases in these species do not match decreases in other species to achieve a neutral biodiversity outcome, as the increaser species are typically widespread generalist “weedy” species.
- (5) There is less evidence for change in the reptile, frog and invertebrate faunas of the rangelands, but this needs qualification because of the even poorer historic baseline information. As a range of studies suggests that reptiles and some invertebrate groups show substantial responses to some land-use and management practices which are extensive across the rangelands, it is likely that there have been unrecorded broad-scale changes in these faunas.
- (6) Information about changes in the status of plant species is diffuse. One estimate suggests that about 17 species of plants have become extinct in the rangelands since European settlement, with about another 250 species declining to the point of being nationally recognised as threatened.
- (7) Biodiversity has been affected in all rangeland environments, although the principal factors providing impact generally vary between environments. Feral predators have

had a pervasive impact on faunal biodiversity across almost all rangeland environments, with the possible exception of tropical eucalypt woodlands and rainforests (which generally lack foxes).

- *Chenopod shrublands* have been affected particularly by pastoralism and grazing by feral animals (rabbits and camels), with the most pronounced biodiversity changes being in floristic composition and an extensive decline of a high proportion of their relatively few bird species.
- *Tussock grasslands* have also been affected primarily by pastoralism, but also by invasion of weed species. Biodiversity changes have probably been less pronounced, but have included alteration of floristic composition and regional declines of some vertebrate species.
- *Hummock grasslands* have been affected mainly by change in fire regimes, which has contributed to loss of some mammal species and changed abundance of some bird and plant species.
- *Mulga and other Acacia woodlands* have been affected largely by grazing (by livestock and feral animals), change in fire regimes and, in some bioregions, clearing. These factors have led to substantially reduced extent in some bioregions, altered vegetation structure, and change in floristic composition.
- *Tropical and subtropical eucalypt woodlands*. The tropical eucalypt forests have probably fared better than all other rangeland environments, but their biodiversity has undergone some changes due to the synergistic effects of changed fire regimes, pastoralism and weed invasion. These factors have also affected subtropical eucalypt woodlands, which are now also being cleared at an unprecedented rate.
- *Semi-arid eucalypt woodlands*. Mallee woodlands occur across much of the southern rangeland margin. Their biodiversity has been affected mainly by changed fire regime and clearing, and much of their mammal and bird fauna is greatly reduced in range or abundance. The taller and more open woodlands of semi-arid areas in all rangeland jurisdictions have undergone major changes through the compounded pressure of changed fire regimes (mostly fire exclusion), pastoralism, feral animals, weed invasion and clearance. Over very large areas, they have changed markedly in structure and floristic composition, and these changes (and the pressures themselves) have contributed to widespread decline in many bird and mammal species.
- *Rainforests* occupy a very small proportion of the high rainfall northern rangelands, but provide a highly distinctive contribution to the rangeland biota. Their condition has been affected mainly by changed fire regimes, the trampling impacts of livestock and feral animals, and weed invasion.
- *Wetlands and riparian areas* also occupy only a small proportion of the rangelands, however they have suffered a disproportionately large impact, particularly from pastoralism, feral animals, weed invasion, and modification of hydrological regimes. This impact has included markedly changed floristic composition, vegetation structure, ecological processes and at least localised losses of some distinctive animal and plant species.

- (8) Biodiversity in the rangelands has been affected by many threatening processes, often operating synergistically. We summarise the extent and impacts of these processes, including change in fire regime, pastoralism, feral predators, other feral animals, weeds, mining, hunting and harvesting of native species, clearing, horticulture, pesticides, changed water regimes and climate change.
- (9) For each of the 46 rangeland bioregions, we provide a summary of biodiversity status and of the occurrence of threatening processes.

## **INTRODUCTION**

Since European settlement, the highly distinctive environments and biota of the Australian rangelands have been substantially re-shaped. Much of the mammal fauna has disappeared; the composition, structure and relative extent of many ecosystems has altered drastically; and the forces which had previously crafted the rangeland ecology have been diluted, transformed or replaced with different sets of drivers.

In some cases, aspects of this change have been documented and causes identified. But in general, there is little information on:

- what has changed;
- where these changes have occurred;
- the rate of change;
- whether changes are ongoing; and
- what factors have caused change.

This review aims to collate and synthesise the available material on these subjects. It is framed as a resource document for the development of a biodiversity monitoring program across the rangelands, rather than as a definitive and exhaustive account of biodiversity in Australian rangelands.

As companion documents for the development of biodiversity monitoring in the rangelands, we provide elsewhere (Background Papers 2 and 3 in this series) reviews of existing pastoral monitoring programs and of programs which aim to survey or monitor rangeland biodiversity, and a critical appraisal of biodiversity monitoring programs generally (Background Paper 4 in this series). Together these documents aim to provide an indication of what components of the biodiversity of the Australian rangelands are most likely to change in status, what threatening processes are most likely to continue to alter ecosystem processes and species compositions, and how and where we should be concentrating our efforts to look for, record and manage change.

This present work comprises three main sections:

- an overview of changes in the biodiversity of Australian rangelands, with representative illustrations from different strands of biodiversity;
- an overview of factors which may have contributed to these changes; and
- a digest of changes and threatening processes for each of the 46 bioregions (as defined by Thackway and Cresswell 1995) which are entirely or largely within the defined rangeland area (Appendix A).

To some extent the same material is necessarily re-packaged within these three separate treatments. This is a necessary consequence of trying to report on different foci.

In some sections, we provide considerable material quoting from historical reports of rangeland biodiversity. We justify this on the grounds that it is easy for us to assume that what we have today is more or less the normal or natural state of the rangelands, a misconception that should be corrected when it is strikingly at variance with the perceptive commentary of some of the observers lucky enough to have seen these environments a century or so ago.

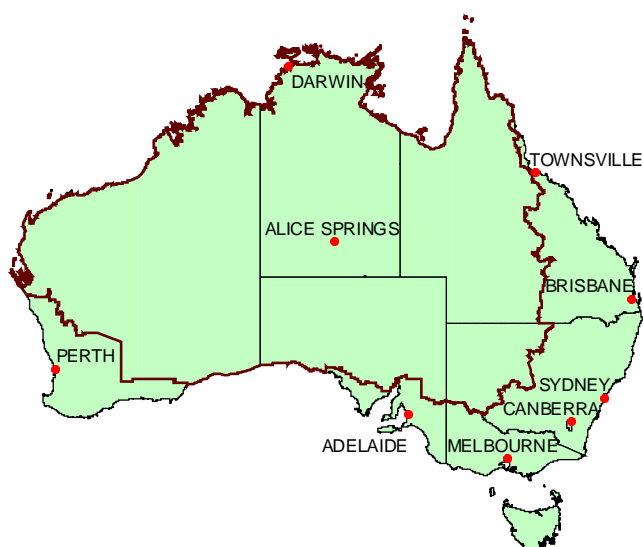
In this review, we emphasise change and loss rather than retention and gain. Many rangeland species have been largely unaffected by land use and management changes, and some native species have benefitted, typically from increased predictability of water supply. Over extensive areas, the rangelands continue to support a rich and distinctive biota, and we in no way intend to derogate the value of what remains. Indeed, the asset of a vast and “unspoiled” nature, with its myriad and strange wildlife, underpins a major rangeland industry, tourism, and the diversity of native plants continues to provide critical support for the most extensive rangeland industry, pastoralism. But this review is focused on the weakness in the fabric, the groups of species which are most susceptible to loss, the processes which are most likely to cause loss, and the regions in which loss has been most substantial – in order to guide monitoring onto those aspects most likely to change.

## DEFINITIONS

### *The Australian rangelands*

*Rangelands* are defined, somewhat loosely as “land where livestock are grazed extensively on native vegetation, and where the rainfall is too low or erratic for agricultural cropping or for improved pastures” (National Rangeland Management Working Group 1996). As delineated by ABARE (Fig. 1), this includes about 70% of Australia (for a total rangeland area of around 5,500,000 km<sup>2</sup>), and all or part of five jurisdictions (Queensland, New South Wales, South Australia, Northern Territory and Western Australia). Notably, and somewhat perversely, this defined area includes some desert lands too remote or harsh for pastoralism, and some lands where rainfall is adequate for cropping and improved pastures. We use this defined area here, although in a few cases, choose to present some information from adjacent areas, where this illuminates aspects which are poorly known for the rangelands proper.

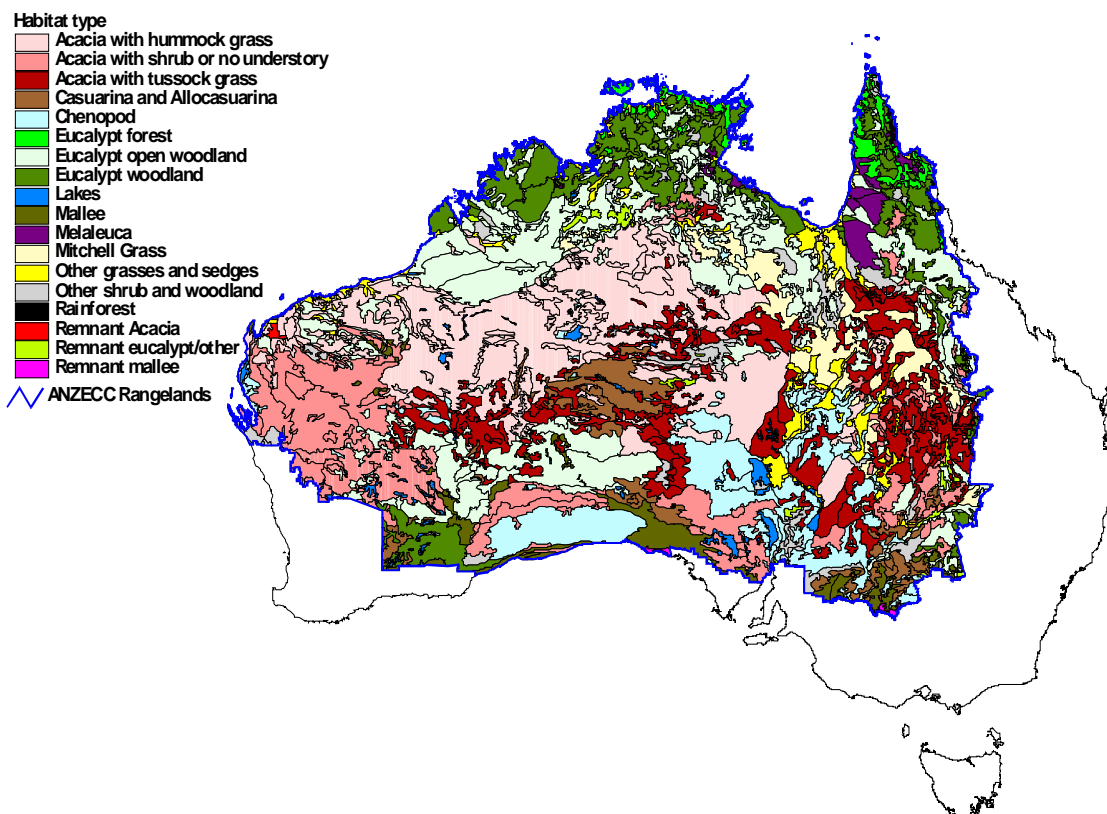
**Figure 1. Rangeland area (encompassed by solid black line) as defined by ABARE, and used in this report.**



In this review, we generally limit ourselves to consideration of terrestrial systems, largely to avoid overlap with other concurrent projects of the National Land and Water Resources Audit.

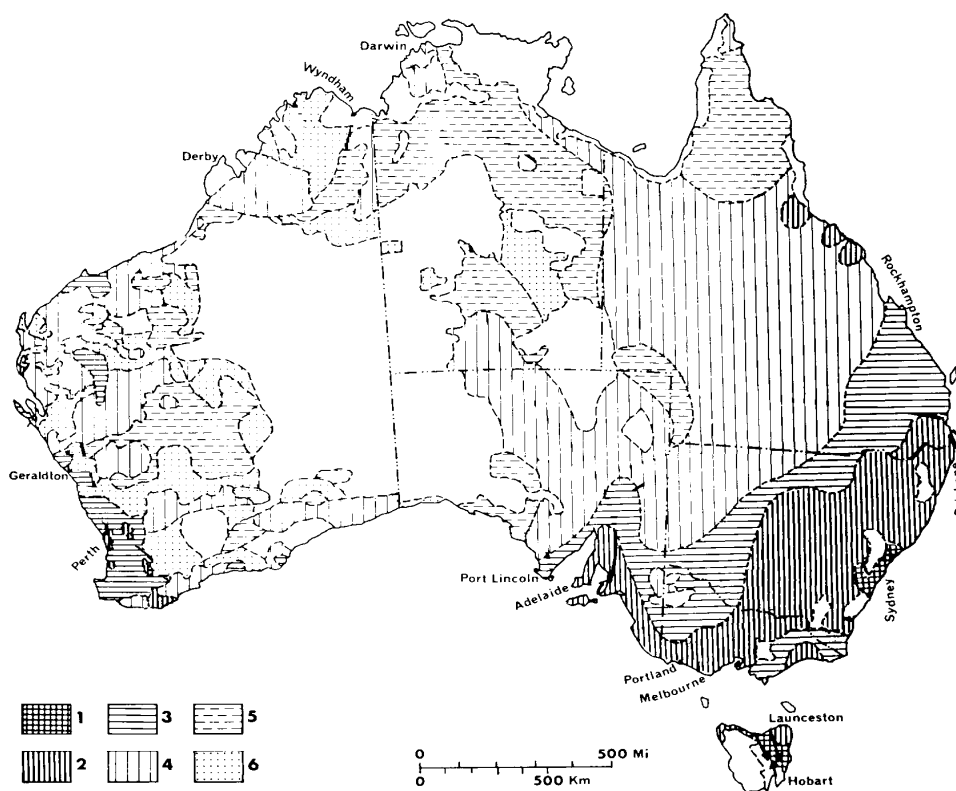
The Australian rangelands comprise an extremely heterogeneous array of climates, landforms and vegetation, including deserts, semi-arid grasslands, chenopod shrublands, tropical eucalypt forests, and *Acacia* woodlands (Fig. 2; Table 1). This environmental diversity subverts any simple description of biodiversity patterning, trends in its change, or explanation for such change.

**Figure 2. Distribution of broad vegetation types in the rangelands.**

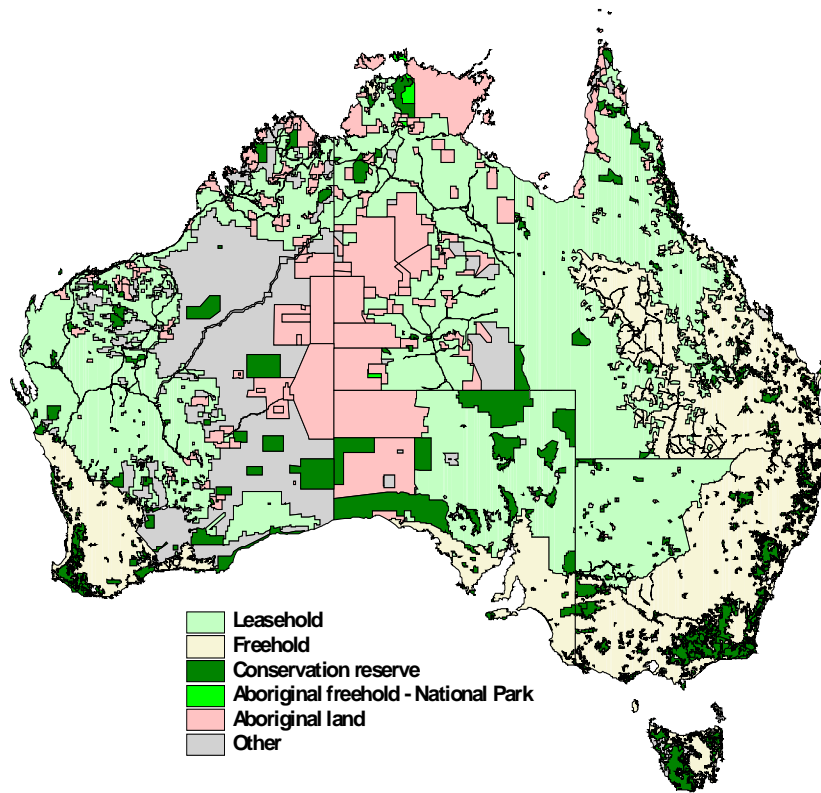


Superimposed on this environmental variation, the rangelands have been subjected to a range of land-uses and development history (Fig. 3), with very mixed tenure (Fig. 4) further complicated by contrasting administrative regulations between the six jurisdictions sharing governance of the rangelands.

**Figure 3. Historical pattern of pastoral settlement across the rangelands.** Time periods: 1. (grid) occupied by 1830; 2. (narrowly spaced vertical lines) by 1845; 3. (horizontal lines) by 1860; 4. (widely spaced vertical lines) by 1880; 5. (dashed horizontal lines) by 1900; 6. (dots) post 1900. (From Heathcote 1977).



**Figure 4. Broad land tenure types across the rangelands.**



## *Biodiversity*

Biodiversity is a term encompassing the variety of living things and the linkages which connect them – formally defined in Australia’s National Biodiversity Conservation Strategy (1993) as:

*the variety of all life forms – the different plants, animals and micro-organisms, the genes they contain, and the ecosystems of which they form a part.*

This definition explicitly recognises three levels within a biodiversity hierarchy – ecosystems, species and intra-specific genetic variability.

In this paper we concentrate on ecosystem diversity, ecological processes (the webs which bind species and ecosystems) and species. We pay scant regard to genetic diversity, because it is unarguably the least significant level of biodiversity, and because its information content is relatively well nested within information about the next level of hierarchy (species). Even more justification is the purely pragmatic, that there is so little relevant information available about genetic diversity of native species in rangeland systems.

To appreciate the significance of change, biodiversity needs to be considered at a range of scales. For example, it is widely recognised that the provision of artificial water sources has led to increases in the number of many rangeland bird species (such as galah *Cacatua roseicapilla* and crested pigeon *Ocyphaps lophotes*), and hence to local bird species richness. But these “increaser” species typically occur across many environments and over a huge range, and hence their increase at any one locality will not necessarily enrich biodiversity at a property or regional scale. Conversely, if the provision of an artificial water source is accompanied by increased pressure on an environment or species which has only a very localised distribution, the loss or decline of that environment or species is to the net detriment of property- and regional-scale biodiversity, even if the number of (regionally widespread) “increaser” species is greater than that of the number of (localised) “decreaser” species.

## CONTEXT – POLICY AND PRECURSORS

This review doesn't spring unheralded from a vacuum. Rather, recent attention to biodiversity and its loss, and to the rangelands, has prompted a spate of related overviews and reviews which singly cover large segments of this topic and/or which together provide the policy or conceptual framework or the information foundation. We present here a brief introduction to some of the more significant precedents, especially those which provide the policy framework for the recognition of obligation to protect biodiversity, and where this obligation intersects with rangeland activities.

### *The policy framework*

The *National Strategy for Ecologically Sustainable Development* was endorsed by the Council of Australian Governments in 1992, with an objective of ensuring that economic development should be accompanied by the conservation of biological diversity and the maintenance of essential ecological processes.

The *International Convention on Biological Diversity*, adopted by Australia in June 1993, provides a global mechanism for the conservation of biodiversity, and recognition of its value for the present and future generations. Within Australia, the *National Strategy for the Conservation of Australia's Biological Diversity* (Commonwealth of Australia 1996) provides a strategic framework for the conservation and management of Australia's biological diversity and, *inter alia*, explicitly recognises the need for:

- integration of biological diversity, conservation and natural resource management;
- management of threatening processes; and
- improvement of the knowledge and understanding of biodiversity.

The *National Strategy for the Conservation of Australia's Biological Diversity* also recognises a set of key principles, including that:

- *It is vital to anticipate, prevent and attack at source the causes of significant reduction or loss of biological diversity; and*
- *central to the conservation of Australia's biological diversity is the establishment of a comprehensive, adequate and representative system of ecologically viable protected areas integrated with the sympathetic management of all other areas, including agricultural and other resource production systems.*

It also includes an objective to “*manage biological diversity on a regional basis, using natural boundaries to facilitate the integration of conservation and production-oriented management*”, with one step in this objective being to identify the major threats to biodiversity in each region.

The *Australian National Strategy for the Conservation of Australian Species and Communities Threatened with Extinction* (Endangered Species Advisory Council 1992) recognised that a significant proportion of Australia's natural heritage had been lost in the preceding 200 years, and that threatening processes were continuing to whittle away our biodiversity. It identified major threatening processes as habitat destruction, habitat change and degradation, introduced animals, introduced plants, disease, direct exploitation, climatic changes and interactions of these factors.

Although principally (but not exclusively) directed at environments outside the rangelands, the *National Forests Policy Statement 1992* provided a number of more generally applicable goals and protocols. It recognised that biodiversity conservation was most efficiently achieved within a regional framework, in which all values could be considered within a rational and integrated planning process. It set a goal of maintaining an extensive and permanent native forest estate, managed in an ecologically sustainable manner. It recognised the need to protect the complexity of biological interactions, such as "the maintenance of ecological processes that sustain forest ecosystems". It recognised the need for the protection of the forest estate through the development of a conservation reserve system which was comprehensive, adequate and representative.

The *Australian Guidelines for Establishing the National Reserve System* (Commonwealth of Australia 1999) further extended recognition of the need for a bioregional approach to the evaluation of conservation values and threatening processes, and for conservation planning. A major tool for this conservation planning was the development of nationally-agreed Australian bioregions ("*Interim Biogeographic Regionalisations for Australia*": Thackway and Cresswell 1995). Information on the biodiversity and/or reservation status of these bioregions was collated for Queensland by Sattler and Williams (1999), for the Northern Territory by Connors *et al.* (1996), for Western Australia by Hopkins *et al.* (1996) and for New South Wales by Benson (1999).

Although never formally endorsed, the *Draft National Strategy for Rangeland Management* (National Rangeland Management Working Group 1996) provides the most critical foundation for this review, and was an important catalyst for the proposal for the development of a biodiversity monitoring program in the rangelands. This document noted that around 13% of Australia's rangelands were severely degraded, 16% were affected by sheet and rill erosion, 17% were affected by woody weeds, 25% of land cover had been significantly disturbed, and 12% of arid zone mammals had become extinct. It recognised that the principal threats to rangeland biodiversity were over-grazing by domestic stock, feral animals and excessively high numbers of some native animals; use of waters in the Great Artesian Basin; vegetation clearance; feral predators; weeds; changes in fire regime; and climate change. It specified the long-term goal that "*The biological diversity, essential ecological processes and other assets of the natural environment of Australia's rangelands are protected and improved*", and a number of objectives and actions relevant to biodiversity monitoring and management, including:

- *Objective 3.4. Improved condition and functioning in rangeland ecosystems*

- Action 3.4.5. Recognise and change practices which are not ecologically sustainable.
- *Objective 3.5. Fragile and degraded lands managed in a sustainable way which protects, and restores where possible, commercial and non-commercial values ... and which protects biodiversity.*
- *Objective 4.1. Identification of the full range of indigenous biodiversity and its conservation status on a bioregional basis, including identification of the threats to biodiversity.*
- *Objective 4.3. Appropriate nature conservation measures applied across all land tenures ... sufficient to allow the maintenance of indigenous biodiversity on a bioregional basis.*
- *Objective 4.4. Rehabilitation of threatened species and ecosystems.*
- *Objective 4.5. Long-term monitoring of biodiversity.*
  - Action 4.5.1. Incorporate biodiversity attributes in natural resource assessment programs;
  - Action 4.5.2. Implement early warning programs to prevent the decline of potentially threatened species and ecosystems
- *Objective 7.3. An expanded monitoring and evaluation program across the rangelands that incorporates indicators of production, biodiversity, climate and socio-economic factors on a regional and biophysical unit basis, for the assessment of rangeland condition and trend.*

Many of these objectives were retained or re-worded in the subsequent *National Principles and Guidelines for Rangeland Management* (ANZECC & ARMCANZ 1999). This document described the first of three goals as:

- “*Conservation and management of the natural environment. The rangelands are intrinsically valuable because of their biodiversity, areas of high wilderness quality and other conservation values. Due to many past practices that are now known to have worked against natural ecological processes, significant areas have been degraded, some beyond economic rehabilitation. Natural habitats, species of flora and fauna and economic opportunities have been lost. As an important natural resource, the rangelands require sensitive management and, where feasible, restoration of ecological values on a landscape scale.*”

The *Principles and Guidelines* also noted that:

*“To conserve the ecological patterns and processes in the rangelands, land managers need to work with the uncertainties of climate; use fire as a management tool; manage total grazing pressure, invasive species and those areas which help maintain landscape ecological systems (for example, the critical habitats provided by water holes, mound springs, and where fertility is greater, such as in depressions and around trees); and protect key areas from tree clearing ... and protect areas of high nature conservation value.”*

The *Principles and Guidelines* provide a rationale for the development of a biodiversity monitoring program in the rangelands, explicitly through

- “*Guideline 16 – Rangeland condition and trends should be assessed through an expanded monitoring and evaluation program that incorporates indicators of production, biodiversity, water resource condition, climate and socio-economic factors on a regional and biophysical unit basis; and the associated recommended actions*
  - *16.1. Governments, in consultation with rangeland users and managers, should develop agreed criteria and indicators for ecologically sustainable rangeland management and business viability;*
  - *16.2. Develop comprehensive, cost-effective monitoring programs to underpin implementation of criteria and indicators for ecologically sustainable rangeland management at a range of scales; and*
  - *16.3. The opportunity created by the National Land and Water Audit must be used to establish a national rangeland monitoring program to determine trends in the long-term health of Australia’s rangelands at a regional, land type and property level.”*

These guidelines and directions complement the development of indicators for monitoring and reporting on the health and condition of Australian environments generally (State of the Environment reporting: Saunders *et al.* 1998; ANZECC 1998 ), and for forest environments in particular (Montreal Process: Commonwealth of Australia 1998). Many of the indicators proposed for these two sets of reporting can be directly applicable to the rangelands, or can readily be adapted for more specific rangeland reporting. The most relevant of those accepted for National State of the Environment reporting include:

(for biodiversity)

- native habitat clearing;
- habitat fragmentation;
- fire regimes;
- introduced species;
- pollution;
- extinct, endangered and vulnerable species;
- extent and condition of native vegetation;
- populations of selected species;
- terrestrial protected areas;
- recovery plans; and
- area revegetated;

(and for land)

- changes in land use;
- area of land under best management practice;
- potential for erosion;
- extent of perennial vegetation cover;
- wind erosion;
- rising watertables;
- area affected by dryland salinity;
- area affected by acidity;
- quantities of biocides sold; and
- status of highly contaminated sites.

The definition of, monitoring for, and reporting on, ecosystem health is a closely related activity, which may underpin much of the value and interpretation of indicators. This work is being undertaken currently as part of a complementary National Land and Water Resources Audit project (Theme 7 Project 4 “Landscape Health”).

Other streams of the National Land and Water Resources Audit also help provide critical related information for monitoring biodiversity in the rangelands, and help craft a frame of reference for this project. The most relevant of these include:

<b>Theme</b>	<b>Project</b>	<b>related/background issues</b>
1. Water availability	1. Characterisation, categorisation, allocation and use of Australia’s water resources	will provide information on the use and management of water resources across the rangelands.
2. Dryland salinity	1. Defining the nature, extent, hazards and costs of dryland salinity across Australia	will assess prevalence and potential of this threatening process, explicitly including review of impacts upon biodiversity.
3. Vegetation management	2. Accessing, auditing and compiling digital vegetation data	will provide information on the extent of different vegetation types across the rangelands, and historical changes in this extent.
4. Rangelands monitoring	1. Assessing change in ecosystem function and trends in climate, fire and intensity of use that impact upon the ecosystem	will provide information for the rangelands on existing ecosystem function monitoring schemes, some threatening factors or management practices (e.g. fire, intensity of land use), and information on the extent of disruption to ecosystem function.
	2. Determining trends in economic, social and institutional factors that	will provide trend data on land use, explicitly seeking linkages with those of landscape function.

	influence land use and management in rangelands	
5. Agricultural productivity and sustainability	1. Trends in land use change, productivity and enterprise diversification	will provide information on historic and projected trends in land use
	5. Reporting land use change, practices, and degradation and the sustainability of agricultural enterprises	will provide reporting mechanisms for assessment of sustainable land use
	6. future monitoring framework for Australia's land resources	will help collate monitoring across a range of projects
6. Capacity for change	1. Valuing the natural resource base and costs of resource degradation	will help establish a valuation mechanism for natural resources
	2. Providing an integrated socio-economic and bio-physical database system	will help include biophysical data sets into a framework for reporting and policy development
	4. Reviewing institutional structures, their capacity for change and response to major resource management issues, and effectiveness of their intervention	will provide mechanisms for enhancing the institutional delivery of natural resource management
7. Ecosystem health	1. surface water quality. 2. waterway condition	these will provide information on the health and management of aquatic systems
	4. landscape health assessment	will assess status, trends, condition and management requirements for every bioregion, and provide recommendations from a nature conservation and management perspective.
	5. riverine vegetation scoping study	will identify integrity of riparian lands
	6. integration, and catchment-scale reporting	will provide assessment of condition at catchment scale, and tools/information for their management

The relationship between biodiversity protection, sustainability and land use practices is further linked through the development of certification for acceptable environmental standards. For the major industry of the rangelands, pastoralism, the most relevant international standard is ISO 14001 (AS/NZS 1996). AFFA (1999) proposed a target of

15 per cent of agricultural produce to come from properties that have an accredited environmental management system such as ISO 14000 by 2010, suggesting that accounting for environmental sustainability will become a more formal and recognised process across much of the rangeland area.

### *Biodiversity in Australian rangelands.*

There has been no prior comprehensive review of biodiversity in the Australian rangelands, probably because the rangeland area does not correspond closely with a biogeographic or ecological entity. However, there have been a number of major overviews of biodiversity in what is essentially the core rangeland area, the arid and semi-arid areas of central Australia, extending from southwestern Queensland and western New South Wales to the western deserts of Western Australia. These include edited volumes covering many aspects of the biogeography and evolution of the central Australian biota (Cogger and Cameron 1984; Barker and Greenslade 1982), of the changing status of the environments and biota in that area (Morton and Mulvaney 1996), and of conservation issues generally (Messer and Mosley 1983).

Several important papers have also described a framework for the ecological functioning and management of the arid rangelands (Griffin and Friedel 1985, 1996; Friedel *et al.* 1990; Morton 1990; Morton *et al.* 1995). These note especially the ecological significance of the highly variable and unpredictable climate in central Australia, with extended periods of drought punctuated by irregular rainfall events and flooding, with fire further superimposed as a discontinuous environmental modifier. The primacy of this climatic fluctuation as a driving force in the arid rangelands provides a major impediment to the development of a biodiversity monitoring program, as many species respond with extremely high year-to-year variability in abundance, distribution and habitat relationships (Carstairs 1974; Newsome and Corbett 1975; Noble 1977; Ford 1978; Hollands 1979; Wyndham 1983; Davies 1984; Caughley *et al.* 1984, 1985; Cairns and Grigg 1993; Masters 1993; Kingsford and Porter 1993; Southgate and Masters 1996; Dickman *et al.* 1999; Hunter and Elder 1999).

Morton (1990) noted that sites which provided the best buffer to the climatic variability formed critical refugia for much of the biota, allowing it to survive through the bad years. However the functioning of many of these refugia was now substantially compromised by degradation due to the concentration of exotic predators, livestock and rabbits on these very sites during critical periods. Morton *et al.* (1995) provided more expansive descriptions and inventories for some of these refugia, most notably including wetlands and rocky ranges. These latter refugia function not only to protect widespread species during years to decades of low rainfall, but also serve as safe havens for species which are adapted to higher rainfall regimes, otherwise marooned during centuries or millenia of relatively adverse climate, and hence may maintain highly disjunct relictual populations (Keast 1959; Latz 1996). Such sites are also major centres of endemism and speciation for species with poor dispersal abilities, such as camaenid land snails (Solem 1993).

Patterns of speciation and species richness in the arid and semi-arid rangelands have been investigated for many taxa (e.g., Barker and Greenslade 1982; Pianka 1969, 1981;

Hnatiuk and Maslin 1988; Schodde 1990). While species richness for most groups is generally low compared with temperate Australia and the wet tropics, some groups are exceptionally speciose. There is an extraordinary richness of ants and termites in Australian arid (Morton 1985; Morton and James 1988) and semi-arid zones (Andersen 1992), which far surpasses that in comparable climates on other continents. The number of species of reptiles, and especially skinks, in arid and semi-arid rangelands in Australia is also remarkable (Pianka 1981; Morton and James 1988). A range of hypotheses has been proposed to account for this extraordinary richness, including usurpation of the roles of “absent” non-reptile competitors, the physical structure of the extensively dominant understorey hummock grasses, fine-scale habitat specificity and turnover of species between environments within an intricate environmental mosaic, and unusually high abundances of some invertebrate prey (most notably, termites) (Morton and James 1988).

The main rangeland area outside the arid and semi-arid zones is the monsoonal tropics of northern Australia, extending from the Kimberley to Cape York Peninsula. This area has a distinctly different biota to that of the arid and semi-arid rangelands (Kikkawa and Pearse 1969; Bowman *et al.* 1988; Whitehead *et al.* 1992; Franklin *et al.* in press). For most plant and animal groups, species richness declines down the generally north-south rainfall gradient (Bowman 1996; Woinarski *et al.* 1999c). As with the arid rangelands, localised features of unusual topographic relief (most notably the sandstone escarpments of Arnhem Land and the north Kimberley) provide long-term refugia, housing many species with disjunct populations, acting as centres of endemism, and maintaining high species richness for particular groups (especially those with relatively low dispersal ability) (Specht 1958; Carlquist 1979; Solem and McKenzie 1991; Solem 1993). In contrast, many species associated with the lowland eucalypt savannas have extremely extensive longitudinal ranges, and there is relatively little turnover in species across large scales (Woinarski *et al.* 1999c). Also in common with the arid rangelands, there are few groups or regions with exceptionally high species richness, again with the notable exception of ants (Andersen 1992).

## **THE STATE OF RANGELAND BIODIVERSITY**

This review aims to provide a source document for the design and establishment of a rangeland-wide biodiversity monitoring program. Ironically (and underlining the need), with the lack of an existing such program, it is extremely difficult to provide a comprehensive assessment of the state of rangeland biodiversity. The many fragments of relevant knowledge are widely scattered and extraordinarily heterogeneous in their scope and focus. Even when brought together, there remain major gaps in our knowledge.

While this information patchiness and inadequacy is probably generic in Australia, these traits are accentuated in the rangelands compared to the more settled areas of the coastal fringe, where environmental research is typically more dimensional, more intensive and has a longer historical trace, and where the greater human population density has allowed a far greater concentration of informal notes on changes in biodiversity.

The sections below represent our attempt to marshal the knowledge about trends in the status of rangeland biodiversity. We report initially by the “level” of biodiversity (species or taxonomic groups of species, communities, and processes), and within each level, attempt to distill material on the extent of change, where and when change has occurred, and what factors may have contributed to that change. These elements of the argument obviously re-appear across various levels of biodiversity, and hence weave their way through this account. In Appendix A, we re-sort the information at a bioregional level, to summarise the extent of change and major threatening processes for each of the 45 rangeland bioregions. Despite our lamentation of the inadequacy of the literature, this review is not exhaustive. We have omitted many accounts reporting on localised aspects of biodiversity or on individual species, which contribute relatively little additional information to the overview.

As noted in the previous section, much of the biota of the rangelands (especially in arid and semi-arid areas) may undergo major fluctuations in response to climatic variation. This irregular cycling presents a formidable problem for the evaluation of status and its change, and may require unusually long periods of observation before long-term trends can be segregated from highly variable short-term responses (Noble 1977).

## Species

### **Mammals**

Mammals provide the starkest and most compelling picture of biodiversity change in the Australian rangelands. The extent of loss of the native mammal fauna in the arid and semi-arid rangelands over the last 200 years is amongst the worst extinction records on the globe (Short and Smith 1994). This change has been documented in studies or reviews considering the entire mammal fauna, suites of species, and individual species at a range of scales from local to national.

Even preceding European colonisation, the Australian mammal fauna has been characterised by a turbulent history, with recurring periods of major losses, spurts of speciation, distributional changes, and re-ordering of the numerically dominant species-groups (Murray 1984; Archer *et al.* 1998). One extremely marked transition occurred in the period 60,000 to 20,000 years ago, when most of the distinctive marsupial megafauna (and less charismatic groups of species) declined to extinction. This loss was undoubtedly due to environmental changes (Murray 1984; Baird 1989; Flannery 1990; Bowman 1998) which were both substantial and rapid. In turn, these environmental changes probably reflected relatively sudden and marked climatic fluctuations, probably compounded by the introduction of land management imposed by Aboriginal colonists. The relative influence of climate change and Aboriginal management practices upon the loss of megafauna remains contested (Flannery 1990, 1994; Bowman 1998). Many of the fossil sites which attest to the catastrophic loss of mammals over this period are in what is now defined as rangelands (Horton 1984). The evidence from these sites provides a valuable reminder of the dynamic nature of biota, and the relative recency of much of the rangeland biotic assemblages.

Subfossil (Baynes 1984; Morton and Baynes 1985; Baynes and Baird 1992; Baynes and Jones 1996; Baynes and Johnson 1996) and other evidence (e.g. rock art: Calaby and Lewis 1977; Murray and Chaloupka 1984; Chaloupka and Murray 1986) demonstrates that rangeland mammal assemblages continued to change up to the point of European colonisation. The introduction of the dingo, around 4000 years ago, coincided with a rapid decline of the thylacine *Thylacinus cynocephalus* and Tasmanian devil *Sarcophilus harrisii* across the Australian mainland (Archer 1984), possibly due to associated newly introduced diseases or parasites and/or (less probably) competition. The build-up of dingo abundance also affected a broad range of prey species (Baird 1991a), undoubtedly including many small mammals. The arrival and impact of feral cats may have been an even closer prelude to the watershed of European colonisation, with some evidence of their introduction via shipwreck on the Western Australian coast from the seventeenth century and/or assisted passage with Macassan trepang fishermen in northern Australia, followed by rapid spread to parts of western and central Australia (Carnegie 1898; Finlayson 1961; Burbidge and McKenzie 1989). These recently arrived predators may

have disturbed mammal assemblages across parts of the rangelands, but continuing changes in the abundance and distribution of some mammal species in the few thousand years preceding European colonisation (Finlayson 1961) may also have been a response to ongoing environmental change.

Change accelerated dramatically following European colonisation. Explicit records of the timing of onset of decline in rangeland mammals can be gleaned from sites broadly spread across rangeland areas: from southwestern New South Wales (courtesy of good documentation of early scientific collection), central Australia (through early scientific collections, chronicling of Aboriginal knowledge and analysis of subfossil material), parts of north-western Australia (mostly from subfossil deposits) and central Queensland (mostly from more limited documentation of early collections). In the sections below we illustrate these patterns of change in the mammal fauna from representative areas of the rangelands.

#### *Examples of changing status in rangeland mammals – western New South Wales.*

In western New South Wales, 27 species of native mammal have disappeared over the last two centuries (Dickman *et al.* 1993; Dickman 1994), from a total of 71 species present in 1788. The fate of extinction has fallen inequitably across mammal taxonomic groups and body sizes (Table 1): the regionally extinct species were all marsupials (principally the smaller macropods, bandicoots and larger dasyurids) and rodents – if bats are excluded from the original tally, the proportion of species lost is 47% (27 of 57). This tally and proportion of losses is strikingly higher than that recorded for the non-rangeland areas of the State: 14 species lost (=21% of the non-bat fauna) from the Great Divide, and 3 species lost (=6% of the non-bat fauna) from coastal New South Wales (Dickman 1994).

**Table 1. Loss of mammal species from the Western Division of New South Wales, 1788-1995** (modified from Dickman 1994).

<i>mammal group</i>	<i>no. spp. present in 1788</i>	<i>no. now extinct</i>	<i>% extinct</i>
monotremes	2	0	0
dasyurids <35g	8	0	0
dasyurids >35g	5	3	60
numbat	1	1	100
bandicoots	4	4	100
possums, koala	5	0	0
wombat	1	1	100
macropods <2kg	4	4	100
macropods 2-10kg	4	3	75
macropods >10kg	5	0	0
rodents <35g	5	2	40
rodents >35g	12	9	75
bats	14	0	0
dingo	1	0	0

The Blandowski expedition of 1856-57 to the area around the junction of the Murray and Darling Rivers provides a critical landmark for dating the decline. Comprehensive lists from this survey (Krefft 1866; Wakefield 1966) mark the last records for 13 mammal species in the rangelands of western New South Wales. These records are on the cusp of the arrival of livestock and extensive land modification. Dickman *et al.* (1993) and Dickman (1994) considered that this pre-dating provided good evidence that the loss of these species was not due to environmental change, but rather to the spread of feral cats, a case he supported by noting that cats were almost certainly present in the region in the early nineteenth century, that the native mammals which disappeared first were associated with open habitats and were relatively small (and hence most at risk to predation by cats), and that the timing of regional extinction here was coincident with that of other small mammals elsewhere in Australia where cats were known to have been present. But the attribution of guilt is not that clearcut. In the course of his collection, Krefft (1866) recorded conspicuous contrasts in the mammal fauna across the (expanding) pastoral frontier, for example noting (with prescience) for the greater stick-nest rat *Hapalotis (=Leporillus) conditor*,

*“It is one of the many species which will soon be extinct, as I found that it had already retreated before the herds of sheep and cattle across the Murray. Only a few empty nests were occasionally met with south of that river “;*

for the (also, now extinct) pig-footed bandicoot *Chaeropus ecaudatus*,

*“This singular animal ... is disappearing as fast as the native population. The large flocks of sheep and herds of cattle occupying the country will soon disperse those individuals which are still to be found in the so-called settled districts”*; and

for the burrowing bettong *Bettongia lesueur*,

*“This ... has long retreated before the herds of cattle with which the plains bordering on the Murray are now stocked: and it is no longer to be found south of the river, so, at least, the natives assured me”*.

Not all of the doomed mammals were already in decline at this time, suggesting that a range of threatening factors was in operation, that the threats were operating inequitably across the landscape, and/or that there were substantial differences between species in susceptibility. For example, for the lesser stick-nest rat *L. apicalis*, Krefft recorded:

*“I ... found it in great numbers upon ... both sides of the Murray. It also occurs on the Darling, and I have no doubt that the late lamented Explorers called Rat Point ... after this.”*

But it obviously declined rapidly soon after, for the last definite record of the species in New South Wales was in 1857, and anecdotal records suggest that it persisted in greatly reduced numbers, fading until eventual extinction in the late nineteenth century (Dickman 1993).

After the rapid loss of a tranche of species from the rangelands of New South Wales at initial contact, there was a subsequent wave of species loss at or before the turn of the twentieth century (Dickman *et al.* 1993; Dickman 1994). This loss followed the build-up of livestock numbers over extensive areas of the rangelands, subsequent (and consequent) land clearance and vegetation degradation, and the associated spread of rabbits and foxes. Species lost at this time included the bilby *Macrotis lagotis*, bridled nailtail wallaby *Onychogalea fraenata*, and brush-tailed bettong *Bettongia penicillata* (Dickman 1994).

Curiously, current trends in the status of the residue mammal fauna of New South Wales rangelands are less clearly etched – although Dickman (1994) suggests “continuing declines of species today”. This uncertainty is perhaps because it is easier to see species loss in hindsight rather than to recognise the pattern from typically patchy records of contemporary information. But the difficulty of recognising current trends in status is despite a widespread recognition of the problem of species loss, despite advances in mammal survey techniques, and despite substantial improvements in the amount and collation of survey information and tools for its analysis.

Not all native mammals have declined in these rangelands: Caughley (1986) notes that there were some increaser species (notably the large kangaroos), favoured by the increased availability of water, altered vegetation and reduction in dingo numbers.

Dickman (1994) also noted that the description of extent of historical change and its timing was affected by collection biases. While bats probably have suffered relatively little over the last 200 years, this conclusion should be tempered by the extremely poor record of the bat fauna over at least the first 100 years of settlement, rendering baseline information scant and unreliable. While Krefft gained most of his extraordinarily comprehensive collection of marsupial and rodent specimens through the adroitness of local Aboriginal people, for bats he noted despondently:

*“Owing to the superstitions of the natives, who look upon every Bat as a departed friend and relative, who, according to their ideas with regard to the transmigration of souls, has seen better days among themselves, has thrown spear and boomerang, and feasted upon Kangaroo, Wallaby, and Emeu, the number of Bats collected during my journey was very limited indeed”.*

Additional to this meagre early record, the ascription of changing status of bats has been complicated by a long-maintained taxonomic confusion, only recently approaching resolution through genetic studies. For all mammals, collection methods have also changed substantially, with consequent changes in the apparent abundance of species. For example, Dickman (1994) noted a number of apparent recent range expansions for the small dasyurids *Ningauia yvonneae*, *Planigale tenuirostris* and *P. gilesi*, which probably more reflect increasing use of pitfall trapping than real increases in distribution or abundance.

#### *Examples of changing status in rangeland mammals – central Australia.*

Change in the mammal fauna of central Australia is probably the best documented case of loss of biodiversity in the Australian rangelands, courtesy of some unusually good baseline collections (most notably those of the 1894 Horn Expedition and its network of collectors continuing to supply material after the expedition: Spencer 1896; Calaby 1996; Kerle and Fleming 1996), sub-fossil material (Morton and Baynes 1985; Baynes and Johnson 1996), documentation of detailed information from Aboriginal residents (Johnson and Roff 1982; Burbidge *et al.* 1988; Baker and the Mutitjulu Community 1992; Reid *et al.* 1993; Nesbitt *et al.* 1993; Baker and Nesbitt 1996; Pearson and the Ngaanyatjarra Council 1997), reporting of trends by an extraordinarily productive and perceptive mammalogist (H.H. Finlayson), over a 25 year period coinciding with the period of major decline (Finlayson 1930, 1932, 1935a, 1935b, 1939, 1940, 1941, 1943, 1958, 1961), and substantial recent survey efforts (Burbidge and Fuller 1979; Gibson 1986; Gibson and Cole 1988; Copley *et al.* 1989; Reid *et al.* 1993).

The recent compilation of all records by Baynes and Johnson (1996) listed 59 species of native mammals present in “central Australia” pre-European contact. Of these, at least 22 species are now extinct in the area. These losses have occurred inequitably across mammal groups (Table 2), with the highest proportion of extinctions among the larger dasyurids and larger rodents, the bandicoots and smaller macropods. This taxonomic/size

pattern is strikingly similar to that reported for western New South Wales (Table 1 above).

**Table 2. Loss of mammal species from central Australia, 1890s-1990s** (modified from Baynes and Johnson 1996).

<i>mammal group</i>	<i>no. spp. present in 1788</i>	<i>no. now extinct</i>	<i>% extinct</i>
monotremes	1	0	0
dasyurids <30g	9	0	0
dasyurids >30g	5	4	80
numbat	1	1	100
bandicoots	5	4	80
possums, koala	1	0	0
marsupial mole	1	0	0
macropods <2kg	4	4	100
macropods 2-10kg	2	1	50
macropods >10kg	2	0	0
rodents <20g	4	0	0
rodents >20g	10	7	70
bats	12	1	8
dingo	1	0	0

Decline in the mammal fauna began soon after the establishment of European settlement in central Australia. By 1921, less than thirty years after the Horn Expedition recorded an extraordinarily diverse mammal fauna, one of the collectors associated with that expedition wrote:

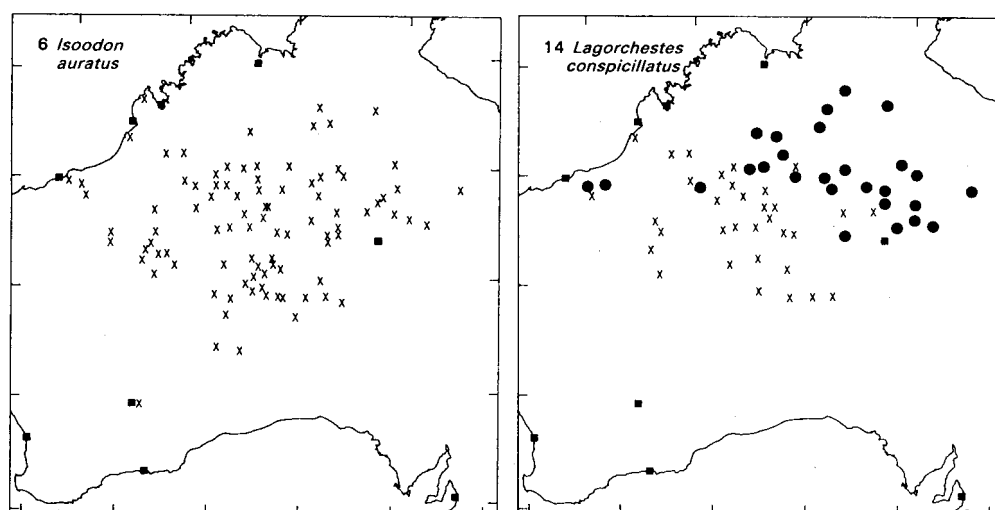
*“during the years since you last saw it, there have been many changes in this country. The rabbits have supplanted the marsupials, and the indigenous plants are gradually giving way to inferior kinds of herbage”* (P.M. Byrne describing Charlotte Waters, quoted in Calaby 1996).

Two major zoological expeditions to central Australia in the 1930s are important landmarks. Otto Lipfert travelled along the Canning Stock Route (Great Sandy Desert bioregion) in 1930-31 and collected the now regionally extinct mala *Lagorchestes hirsutus* (which he noted to be “the most common mammal”), desert bandicoot *Perameles eremiana*, golden bandicoot *Isoodon auratus* (“quite numerous”), western quoll *Dasyurus geoffroii* and red-tailed phascogale *Phascogale calura*, although he noted that “the nest-building rat [*Leporillus apicalis*], said to have been found 24 or 26 years ago, was, however, not seen this trip” (quoted in Burbidge and McKenzie 1983). Finlayson found a thriving mammal fauna across large areas of central Australia, in a series of expeditions between 1931 and 1935, but noted that many of these species had

declined sharply, or were no longer to be found, in his next expeditions from 1950 to 1956 (Finlayson 1961). Finlayson (1961) noted that declines were geographically patchy (roughly correlating with the occurrence of rabbits, foxes and/or pastoralism), and the timing variable between species. For some species, Finlayson's final expedition marked the last records of survival – for example, he noted in 1961 that the now regionally extinct golden bandicoot “survives in considerable numbers” [in the western deserts], and that the desert bandicoot “still persists”.

The most comprehensive information on the timing of decline is the compilation of Aboriginal knowledge presented in Burbidge *et al.* (1988). This updated and expanded the similar records of Finlayson (1961) and added a substantially finer scale of records (Fig. 5a,b). For most species, Aboriginal informants stated extinctions occurred mostly 25-40 years beforehand (i.e. in the period late 1940s to early 1960s), with generally earlier declines in the south (northern South Australia) than in the north (Tanami Desert and Great Sandy Desert), but typically very rapid decline at any one location. The timing of these extinctions varied between species, with earlier loss for pig-footed bandicoot *Chaeropus ecaudatus*, crescent nailtail wallaby *Onychogalea lunata* and stick-nest rats *Leporillus* spp., and more recent extinctions for golden bandicoot, desert bandicoot and numbat *Myrmecobius fasciatus* (Friend 1990).

**Figure 5. Example of information on decline of central Australian mammal species.** Crosses indicate sites where Aboriginal informants indicated local loss. Dots indicate sites where Aboriginal informants reported ongoing presence. (a) golden bandicoot *Isoodon auratus*. (b) spectacled hare-wallaby *Lagorchestes conspicillatus*. From Burbidge *et al.* (1988).



The decline of the central Australian mammal fauna did not simply cease with the pulse of extinctions between the 1920s and the 1960s. Since Burbidge *et al.*'s review (1988), one further mammal species (the mala *Lagorchestes hirsutus*) has become extinct in central Australia. The range and/or abundance of several other species continues to decline (brush-tail possum, black-footed rock-wallaby *Petrogale lateralis*, bilby, marsupial moles *Notoryctes caurinus* and *N. typhlops*, and probably kowari *Dasyercus byrnei*, mulgara *D. cristicauda*, ampurta *D. hillieri*, and fawn hopping-mouse *Notomys cervinus*), with ongoing loss of populations and/or regional extinctions, and yet other taxa are recognised as endangered or vulnerable, but with inadequate information to determine current trends (including sandhill dunnart *Sminthopsis psammophila*, dusky hopping-mouse *N. fuscus*, plains rat *Pseudomys australis* and central rock-rat *Zyzomys pedunculatus*) (Kerle *et al.* 1992; Lee 1995; Maxwell *et al.* 1996; Gibson and Cole 1996; Lundie-Jenkins and Findlay 1997; Copley and Alexander 1997; Pearson and Kinnear 1997; Brandle *et al.* 1999; Moseby *et al.* 1999).

Periods of drought appear to have been associated with particular spikes in the decline of central Australian mammals (Johnson and Southgate 1990; Braithwaite and Muller 1997).

Environmental changes in central Australia have not disadvantaged all native mammals. Large macropods (notably red kangaroo *Macropus rufus* and common wallaroo *M. robustus*) have increased in many areas (Finlayson 1961), presumably because of the establishment of artificial water sources, and possibly through changes in plant species composition.

Introduced species are now among the most numerous mammals in central Australia, and they are implicated in the decline of the native mammal fauna. Rabbits *Oryctolagus cuniculus* arrived in the Lake Eyre Basin in 1889 or 1890, and had reached the Musgrave Ranges by 1901, Lake Amadeus in 1902, and near Tennant Creek by 1905, suggesting an extraordinarily rapid spread across central Australia (Finlayson 1961). The spread of the fox *Vulpes vulpes* was notably slower: it was reported first from Anna Creek (northern South Australia) in 1910, advanced slowly to reach the Everard and Musgrave Ranges in small numbers by 1932, and then spread across the rest of the arid zone over the next two decades (Finlayson 1961). Records of feral cat *Felis catus* in central Australia in the nineteenth century hundreds of kilometres from European settlement in 1891 suggest a long-standing presence (Carnegie 1898; Finlayson 1961). House mice, and camel are now widespread and abundant in many rangeland types of central Australia, and goats are common in some more localised areas.

There are several overlapping and/or alternative explanations for the spectacular loss of mammals from central Australia, although different factors have probably contributed in varying proportions to the loss of particular species. Finlayson's (1961) review was the first major consideration of threatening processes. He proposed five factors:

1. *Long-term climatic change* (increased aridity). He suggested that this was a factor in the isolated, relict populations of some mammal species (explicitly

- including red-tailed phascogale *Phascogale calura*, brush-tailed bettong *Bettongia penicillata* and, perhaps, brushtail possum, lesser stick-nest rat *Leporillus apicalis* and ghost bat *Macroderma gigas*), and speculated that the central Australian populations of these species were marginal and highly susceptible to any environmental change.
2. *Aboriginal hunting*. Finlayson suggested that this was generally a minor factor, though may have contributed to the extinctions of highly localised species and species with low population size, and that extensive burning may have disadvantaged non-burrowing species.
  3. *Pastoralism*. Finlayson considered that “*this is no doubt a major cause of decline and perhaps the chief one*”, affecting especially “*the surface nesting forms*”, and that “*it has constantly been observed on stocking virgin country, that many native species disappear long before any question arises of competition for food.*”
  4. *Introduced pests*. “*The three major scourges, the rabbit, the fox, and the feral house cat ... together have had an effect in certain districts only to be described as catastrophic.*” Finlayson noted that the fox was then not evenly distributed across the arid zone, but that where present in appreciable numbers, native mammal populations could be ‘stripped’ with remarkable speed, and especially so for colonial species. However, he also noted that feral cats were widespread across central Australia well before the major crash in native mammals, and suggested that this broad distribution before the advent of major extinctions implied that they were a less ‘virulent’ threat.
  5. *Disease*. While noting that disease, poisoning (through natural agents) and excessive heat had all been observed to affect native mammals, Finlayson considered that none of these was “*a major cause of loss*”.

An additional, alternative, explanation was proposed by Bolton and Latz (1978). They noted that across large areas of central Australia, traditional Aboriginal land management practices were dislocated or abandoned following European contact. The most ecologically significant of these practices was fire management, which traditionally produced a fine scale seral mosaic. This intricacy has been lost in the last 50 years (Latz and Griffin 1978; Kimber 1983; Low 1986; Latz 1995). As the intricacy of environments resulting from traditional burning practices provided the varied range of resources needed through time by local populations of mammals, these populations would have collapsed rapidly on the cessation of traditional burning. Research on the now regionally extinct mala provided some support for this explanation (Lundie-Jenkins 1993), although a study on the population dynamics of several species of marsupials on one Western Australian island provided results which were not consistent with this “Vegetation Mosaic Hypothesis”, and prompted the authors to refute its general applicability across central Australia (Short and Turner 1994).

McKenzie (1981) and Burbidge and McKenzie (1989) examined the pattern of declines and extinctions and demonstrated that loss was highly concentrated in non-flying mammals in the weight range 35 g – 4.2 kg (the “critical weight range”), that mammals which sheltered in rock-piles were less extinction-prone, and that, across Western

Australia, extinctions and declines were far more likely to have occurred among the arid mammal fauna and/or in the more arid portions of the distribution of wide-ranging species. They concluded that these patterns were due largely to resource (productivity) loss which affected particularly the energetic costs of medium-sized mammals in arid areas. They also recognised that idiosyncratic factors were responsible for the losses of some species, most notably recognising the role of foxes in the decline of some species (notably rock-wallabies) but that other species had declined in areas well before the arrival of foxes.

Morton (1990) provided a major synthesis of the threads of evidence of decline in central Australian mammals, and many of the components in the above models. He noted that the most striking factor in decline was that it affected particularly mammals and not birds nor reptiles, that most of the mammals affected were medium-sized, and that most were herbivores or omnivores. He noted that many of these medium-sized mammals occurred in central Australia largely through dependence upon those highly restricted patches in the environmental mosaic there that were relatively fertile and which provided some succour during drought. It was just these critical refuge patches in which the impacts of introduced herbivores were most concentrated, and in which introduced predators may have congregated. During periods of adverse climate, the impacts of this extra load of exotic herbivores (often at extraordinarily high densities) and predators, would have been fatal. The loss fell unequally on the medium-sized mammals, because the larger species could move more widely between refugia, and the smallest could occur more abundantly and equitably across the landscape.

Smith and Quin (1996) provided a more quantitative synthetic analysis, for the Australian rodents, and demonstrated that declines were greatest for species

- with larger body size;
- which occurred in arid areas or in temperate woodlands;
- which had lower reproductive rates;
- which did not use burrows or natural cavities for shelter; and
- which included a high proportion of herbage (leaves and stems) in their diets.

They also noted that declines were most severe in areas where rabbits and foxes were abundant, and less severe in areas where dingoes were abundant. For small rodents, abundance of the feral cat was the best predictor of decline.

They explained this patterns with a “hyperpredation” hypothesis, that declines were most likely where the abundance of predators (cat, fox, dingo) had been greatly elevated and sustained by the introduction of rabbits and house mice. Cats and foxes were also associated with severe declines in species with low reproductive rates in areas where rabbits and mice were scarce, and where habitats may have been modified by grazing or frequent burning.

The primacy of feral predators in the decline of at least some mammals of arid and semi-arid has been further advanced by correlative or experimental studies, focussed on numbats (Friend 1990), possums (Kerle *et al.* 1992) and small to medium-sized

macropods (Kinnear *et al.* 1988; Gibson *et al.* 1994; Christensen and Burrows 1994; Short *et al.* 1994; Copley and Alexander 1997; Short 1998). In many of these cases, the responses are very dependent on which of the main predators (fox, cat, dingo) is present, and there may be complicated and compensatory responses of one predator species to the control of another species.

### *Examples of changing status in rangeland mammals – Kimberley.*

In contrast to the two rangeland areas discussed above, consideration of the changing status of mammals in the Kimberley is hampered by a far more fragmentary baseline record, with little detailed documentation of Aboriginal knowledge, a very limited sub-fossil record and only a thin veneer of information from early collections. The most notable of the latter are the records of Knut Dahl (1897, 1926) who collected widely in north-western Australia between 1894 and 1896, including five months collecting in the southwestern Kimberley. In contrast, there is now a good record of the current Kimberley mammal fauna from a number of relatively recent studies at regional or sub-regional level (e.g., McKenzie *et al.* 1975, 1977, 1978; Kitchener 1978; Kitchener *et al.* 1981; McKenzie 1981*b*, 1983; Woinarski *et al.* 1992), most of which include some discussion of changing status.

McKenzie (1981*a*) provides the most focussed consideration of change in the Kimberley mammal fauna. He noted that the limited sub-fossil evidence suggested that the mammal assemblage of the southeastern and southwestern Kimberley had changed in response to climatic oscillations in the middle Holocene, with species associated with high rainfall contracting in range to the north Kimberley well before the advent of Europeans. The arrival of pastoralism around the 1890s signalled a subsequent spate of changes in the mammal fauna of the lower rainfall southeastern and southwestern Kimberley. In the southwestern Kimberley, five of the 28 non-bat mammal species present then have become regionally extinct. The timing of this decline is unclear, largely because of a large gap in collecting effort for nearly 50 years following Dahl. Dahl's 1895 collections marked the last record from the southwestern Kimberley of the burrowing bettong, brush-tailed phascogale, golden bandicoot (although there was an unconfirmed record of this species in 1971) and the golden-backed tree-rat *Mesembriomys macrurus*. The last record of the pale field-rat *Rattus tunneyi* from the southwestern Kimberley was soon after, in 1899. Dahl's (1897) record is clear that three of these species, at least, were very common at the time. He noted that, for the burrowing bettong in the sandy country around Roebuck Bay,

*“the ground was nearly everywhere and in all directions excavated by the burrows of this little Macropod ... all the scrubs, and especially the slopes ... are inhabited by countless numbers”;*

the golden bandicoot was

“very numerous in the coast country around Roebuck Bay ... great numbers being brought to me”;

and for the golden-backed tree-rat

“the houses of settlers ... are always tenanted by (this species)”.

In the southeastern Kimberley, Kitchener (1978) reported the absence (probable extinction) of golden-backed tree-rat, sugar glider *Petaurus breviceps* and common brushtail possum *Trichosurus vulpecula*, suggesting that these species had disappeared because their riverine woodland habitat had been degraded by regular burning and the direct and indirect pressure of livestock. Based on archaeological and palaeontological material, he noted also that the golden bandicoot, bilby and northern quoll *Dasyurus hallucatus* were present in the area at the time of European entry, but were now extinct, probably due to

“the combination of intense overgrazing by stock and donkeys in recent years, and the introduction of feral cats”.

In contrast to this record of decline, the mammal fauna of the more mesic north Kimberley appears to have suffered no major historic change, at least up to the point of the most recent survey records, two to three decades ago (McKenzie *et al.* 1975, 1977, 1978; Kitchener *et al.* 1981).

Not all mammal species have declined in the Kimberley. Gooding and Harrison (1952) describe large increases in the population and range of the agile wallaby *Macropus agilis*, across most of the pastoral country, with increase apparently commencing in the 1880s, due to relaxation of predation pressure (from Aboriginal hunting, eagles and dingos), greater availability of water, and vegetation change. They note control programs had been implemented, including large-scale shooting schemes and strychnine poisoning.

### *Examples of changing status in rangeland mammals – Top End*

Trends in the status of the mammal fauna of the Top End of the Northern Territory largely recapitulate those of the Kimberley, except that the Top End has a more comprehensive baseline record courtesy of more frequent, more intensive and more extensive collecting, associated with exploration parties and records from the early colonial outposts (summarised in Parker 1973).

As with the Kimberley, decline in the mammal fauna follows a south-north (drier-wetter) gradient. For the relatively high rainfall Cobourg Peninsula, comparison of (somewhat) recent survey records and a good baseline suggested that no mammal species had apparently disappeared over the 130 years since European settlement (Frith 1973; Calaby and Keith 1974). However, this settlement was transitory and, except for the spread of

feral water buffalo *Bubalus bubalis*, horses *Equus caballus* and banteng *Bos javanicus*, unaccompanied by extensive environmental modification.

The extent of change across the remainder of the higher rainfall Top End (including Arnhem Land, Kakadu and the Darwin-Katherine area) is less clear. While most mammals present at European settlement are still widespread, some species have declined. Dahl (1897) noted for the brush-tailed rabbit-rat *Conilurus penicillatus*, that

”in Arnhem Land is everywhere common in the vicinity of water”,

a status corroborated by Collett (1897),

“numerous all over Arnhem Land, and in great numbers on the rivers on the lowlands”,

and further supported by the large series (40 individuals) collected in the Alligator Rivers region by Tunney in 1902-1903 (Thomas 1904). This species is now highly localised, uncommon, and apparently still declining across the mainland Top End (with the notable exception of Cobourg Peninsula), although still common on some Northern Territory islands (Woinarski *et al.* 1999; Woinarski *in press*).

Dahl (1897) also noted that the brush-tailed phascogale *Phascogale tapoatafa* was also at least locally common:

“on the rivers Mary and Katherine it was frequently observed. In fact, nearly everywhere inland it was very constant, and on a moonlight walk one would generally expect to see this little marsupial”.

This species is now very rarely recorded in the region, with the exception of some localised populations at Kakadu National Park.

The golden bandicoot *Isoodon auratus* was collected from both high rainfall (South Alligator River area) and lower rainfall (Roper River area) sections of the Top End prior to 1920, but now appears to be extinct across most of the Top End, with the exception of one island and possibly the far northeast of Arnhem Land (Southgate *et al.* 1998). The golden-backed tree-rat is known from only three specimens in the Top End (collected in the lower rainfall Gulf Country in 1901, the Mary River in 1903 and the South Alligator River catchment in 1969). The lack of records across the many recent surveys suggests that, if still present, it is now extremely rare (Woinarski *in press*).

In addition to the golden bandicoot and golden-backed tree-rat, there have been pronounced regional declines and extinctions in the lower rainfall areas of the Top End (Victoria River District, Gulf Country) for northern quoll, northern brown bandicoot *Isoodon macrourus*, common brushtail possum, black-footed tree-rat *Mesembriomys gouldii*, and pale field-rat *Rattus tunneyi* (Parker 1973; Johnson and Southgate 1990; Braithwaite and Griffiths 1996; Woinarski *in press*). Several of these species persist in

the region only on the Pellew Islands (Johnson and Kerle 1991) and, for fewer species, the mainland coastal fringe. The relatively poor historic record for most of the Gulf Country and Victoria River District precludes any accurate dating for the time of decline.

#### *Examples of changing status in rangeland mammals – mitchell grass downs*

Mitchell grasslands occupy an extensive band of the semi-arid northern interior, from the Barkly Tablelands to north-central New South Wales, with scattered outliers elsewhere in northern Australia. This relatively simple and homogeneous environment supports comparatively few mammal species. Evidence for change is constrained by the scant baseline record. The most comprehensive early collection was made by Stalker in 1905-06 on the Barkly Tablelands (Thomas 1906). However this record is an imprecise foundation, as it post-dates the spread of pastoralism by several decades, the collection may include records from a far wider geographical area, and it was made at an unfavourable period, when “no rains of any value (had) fallen for several years”. Of 12 (non-bat) native mammal species recorded then, three (northern quoll, common brushtail possum and carpentarian antechinus *Pseudantechinus mimulus*) are probably now regionally extinct.

The historical record for Queensland mitchell grass grasslands is also very sparse, but declines have been reported for Julia Creek dunnart *Sminthopsis douglasi* (Woolley 1992), spectacled hare-wallaby *Lagorchestes conspicillatus* and bilby *Macrotis lagotis* (Southgate 1990; Gordon *et al.* 1990; McFarland 1992; Johnson 1997), and at least some of these are probably largely ongoing.

#### *Examples of changing status in rangeland mammals – central Queensland*

There has been substantial loss of mammals in central Queensland. In the Brigalow Belt area, Gordon (1984) noted the regional extinction of eight species (western quoll *Dasyurus geoffroii*, long-nosed bandicoot *Perameles nasuta*, bilby *Macrotis lagotis*, brush-tailed bettong *Bettongia penicillata*, white-footed rabbit-rat *Conilurus albipes*, plains rat *Pseudomys australis*, Darling Downs hopping-mouse *Notomys mordax* and canefield rat *Rattus sordidus*) from a total of 44 native terrestrial mammal species, with at least four of the surviving species (northern brown bandicoot *Isoodon macrourus*, spectacled hare-wallaby *Lagorchestes conspicillatus*, bridled nailtail wallaby *Onychogalea fraenata* and northern hairy-nosed wombat *Lasiorchinus krefftii*) having vastly reduced ranges and/or abundance (Gordon and Lawrie 1980; Gordon *et al.* 1990; Crossman *et al.* 1994; Filet *et al.* 1997).

Some of these losses have been specific to the rangeland portion of the distribution of particular species. For example, Gordon *et al.* (1990) noted for the northern brown bandicoot, that “although (it) is common to abundant in coastal areas of Queensland, its

*status in central inland areas is rare to extinct. It appears that this has occurred because of its sensitivity to grazing animals”.*

The onset of decline for many mammal species in central Queensland began almost immediately after European settlement, probably through a combination of environmental change due to pastoralism, deliberate hunting and poisoning, disease and the spread of feral animals, particularly cats (Barnard 1925, 1934; Finlayson 1931, 1934). Barnard (1934) noted that about 1902 “*the country was overrun with domestic cats gone wild*”. Finlayson (1934) noted that, by 1930, apparently because of over-hunting and disease, the koala *Phascolarctos cinereus* had been “*reduced to such an extent that it is now a rare animal in many parts ... where it was formerly very plentiful*”. For bandicoots and dasyurids “*and indeed with most other small terrestrial or partly terrestrial mammals*”, he noted that most “*underwent a sudden diminution in the late eighties of last century [i.e. 1880s], and though some species have made brief recoveries from time to time, they have not persisted, and at the present time are reduced to vanishing point*” ... “*the real nature of the causes underlying these declines are obscure, owing partly to the absence of reliable contemporary records ... In different parts of the country floods, fires, droughts, disease, and closer settlement are all confidently advanced as having been severally responsible, and no doubt they have all contributed. But it is significant that the first notable diminution took place at a time when the country was still very sparsely occupied, and secondly, that the causes have been highly selective (affecting macropods and possums far less than bandicoots and dasyurids).*”

### Synopsis

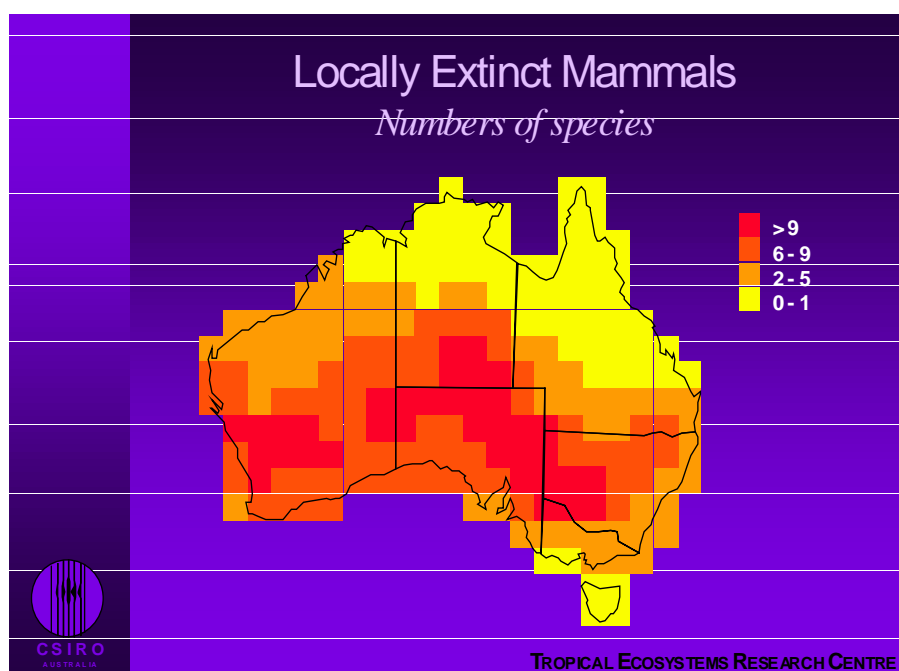
There is a repeated pattern of major loss of mammals across the rangelands. The loss has been most pronounced in arid and semi-arid areas (Fig. 6), and among certain groups of mammals (larger rodents, smaller macropods, larger dasyurids, bandicoots). Despite the distinctness of this pattern, the losses cannot be linked clearly to a single threatening process, but rather involve either idiosyncratic responses or a complex interaction of factors.

Declines in at least some species occurred almost immediately after the advent of European settlement to a rangeland area, with (at least regional) extinction occurring very rapidly (2-3 decades) thereafter for some species. Extinctions in the central and western deserts appear to have peaked between 1930 and 1960.

Change in the rangeland mammal assemblages is continuing. Within the area of greatest level of extinctions (central Australia), there is ongoing decline for many species which survived the initial wave of extinctions. Within the area which has been least scarred by mammal loss (the monsoonal tropics of northern Australia), there appears to be current declines for many of the species (or groups of species) which have become regionally extinct elsewhere.

In contrast to these declines, a small proportion of native mammals (most notably, some large macropods) have generally been favoured by environmental changes in the rangelands over the last century, and are now far more abundant across the rangelands.

**Figure 6. Loss of terrestrial mammal species across Australia**, modified from Woinarski and Braithwaite (1990). Values in  $1^{\circ} \times 1^{\circ}$  cells represent the number of species assumed to be present in 1788 minus the number present now. Note that this figure probably under-represents the loss of species across the tropical savannas.



## Birds

As with mammals, the bird communities in Australian rangelands appear to have been highly dynamic even before European settlement. The most spectacular components of this fauna (including flamingoes and the giant flightless mihirungs) were lost around 60,000 years ago, probably because of relatively rapid climate change, possibly exacerbated by altered fire regimes (Rich and Baird 1986; Baird 1991b).

Since European settlement, the rate of change in the bird fauna of the rangelands has increased rapidly, although the extent of this change has not matched that of the mammal fauna. One contested (Garnett 1999) argument is that this is simply a time lag, and that regional and complete extinctions of birds will become increasingly evident over the next few decades, reaching a level comparable to that for mammals (Recher and Lim 1990; Recher 1999).

Substantial reviews of the status of birds have been published for extensive portions of the rangelands – most notably for western New South Wales (Smith and Smith 1994; Smith *et al.* 1994, 1995) and for arid/central Australia (Reid and Fleming 1992), with more anecdotal accounts for the monsoonal tropics of northern Australia (Woinarski 1993a; Franklin 2000) and the rangelands of the southern half of Western Australia (Curry and Hacker 1990; Saunders and Curry 1990). There have been very many accounts of the changing status of birds in more localised areas of the rangelands (e.g. Barnard 1925, 1934a, 1934b; Read 1999; Johnstone *et al.* in press), and of changes in status within the rangelands of individual species (e.g. Garnett and Crowley 1995a,b) or groups of species (e.g. Franklin 1999).

Overviews of the extent of loss in Australian birds generally suggest that decline and regional loss has been greatest outside the rangelands, in the most intensively modified agricultural areas of temperate south-eastern and south-western Australia (Garnett 1992), where local losses of up to 25% of species have been recorded (e.g. Saunders 1989; Robinson 1993; Saunders and Ingram 1995; Keast 1995; Robinson and Traill 1996; Traill *et al.* 1996; Reid 1999), and are continuing at a rapid rate. Change in the composition of the rangeland avifauna has not yet been this profound.

### *Examples of changing status in rangeland birds – western New South Wales.*

Of 291 bird species and subspecies (excluding vagrants) reported in the Western Division of New South Wales since European settlement, six species (2% of the original fauna) - black bittern *Ixobrychus flavicollis*, lewin's rail *Rallus pectoralis*, little wattlebird *Anthochaera chrysoptera*, regent honeyeater *Xanthomyza phrygia*, night parrot *Pezoporus occidentalis* and thick-billed grass-wren *Amytornis textilis* - have become regionally extinct, an additional five species (magpie goose *Anseranas semipalmata*, flock bronzewing *Phaps histrionica*, squatter pigeon *Geophaps scripta*, purple-crowned lorikeet *Glossopsitta porphyrocephala* and tawny-crowned honeyeater *Phylidonyris melanops*) are “effectively extinct” (occurring now only as rare vagrants), nine other

species (letter-winged kite *Elanus scriptus*, black-breasted buzzard *Hamirostra melanosternon*, mallee-fowl *Leipoa ocellata*, Australian bustard *Ardeotis australis*, plains-wanderer *Pedionomus torquatus*, bush stone-curlew *Burhinus grallarius*, Bourke's parrot *Neopsephotus bourkii*, masked owl *Tyto novaehollandiae* and striated grass-wren *Amytornis striatus*) have “undergone particularly pronounced declines”, and a further 83 species have become less abundant and/or have contracted in range (i.e. a total of 35% of species have declined) (Smith and Smith 1994). Some of the changes have been startling: for example the flock bronzewing is now very seldom recorded in the District, even as a vagrant, but when this area was first settled by pastoralists (in the 1860s), it was reported as occurring “in countless multitudes ... (and in) ... immense assemblage(s)” (North 1914).

In contrast, 83 taxa (29%) have increased, including four species which have newly colonised the region (cattle egret *Ardea ibis*, little egret *Egretta garzetta*, azure kingfisher *Alcedo azurea*, banded whiteface *Aphelocephala nigricincta*) and another four species which “have greatly expanded their distributions” (plumed whistling-duck *Dendrocygna eytoni*, little corella *Cacatua sanguinea*, bar-shouldered dove *Geopelia humeralis* and superb fairy-wren *Malurus cyaneus*) (Smith and Smith 1994; Smith *et al.* 1994).

This change in bird species composition has been uneven across vegetation types (Smith and Smith 1994), with a concentration of decreasing bird species in chenopod shrublands, eucalypt woodland and mallee “scrub”, and relatively little change in the bird communities of the three other broad vegetation types considered (wetlands, low shrubland/grassland and “non-eucalypt” woodland/scrub). Illustrating the fragility of habitat specific species, Smith and Smith (1994) noted that a group of bird species (including little wattlebird, regent honeyeater and tawny-crowned honeyeater) were known in the Western Division only from records in 1849 from a relatively restricted *Banksia* heathland on sandhills along the Darling River. By 1948, this heathland had been eliminated by overgrazing and erosion, no *Banksia* now occurs anywhere in the Western Division and, as a consequence, these birds have been eliminated.

There are also cases of some individual bird species contrasting with the general trends, with the most notable example being the plains-wanderer, a grassland species which has suffered a major decline in western New South Wales (and throughout its range), principally through over-grazing but more recently through conversion of grasslands to intensive cropping (Baker-Gabb *et al.* 1990).

Most increaser bird species have been associated with wetlands, whose occurrence and extent has increased substantially in the region. Again, the trends are patchy, as the characteristics of wetlands today are very different to those 200 years ago, and some wetland bird species have been disadvantaged (Briggs 1994).

The chief factors implicated in the decline of birds in western New South Wales were listed by Smith and Smith (1994) as habitat degradation caused by over-grazing (principally by sheep and rabbits, but also by cattle, feral goats and increased numbers of

kangaroos) and introduced predators (foxes, cats), with reduced native mammal populations also affecting a smaller suite of predatory birds.

Ecological characteristics of declining bird species include relative habitat-specificity, sedentariness, small clutch size, and ground-foraging (although there was also a high proportion of ground-foragers among species which have increased in abundance) (Smith *et al.* 1994).

Declines in birds of the Western Division of New South Wales have been variable in rate and incidence. Major drought from 1895 to 1903 accentuated land degradation due to livestock and rabbits and marked a critical phase for many bird species. But decline has continued inexorably since, and is continuing for many species, including malleefowl (Brickhill 1985), grey falcon *Falco hypoleucos* (Olsen and Olsen 1986), plains-wanderer (Baker-Gabb *et al.* 1990) and bush stone-curlew (Maher 1988).

#### *Examples of changing status in rangeland birds – central Australia.*

Reid and Fleming (1992) collated information from many local and regional studies to describe a reasonably consistent picture of substantial levels of change in the bird fauna of arid Australia, with the status of about half of the 230 species in the arid zone bird fauna having changed since European settlement.

Of these arid zone birds, 19 species (8% of the arid zone fauna) are now recognised nationally as threatened, 12 species (5%) have declined and/or are at risk in two or more of the 14 regions they considered, and a further 40 species (17%) have declined in at least one region (although some remain common or have even increased in other arid regions). Reid and Fleming (1992) did not consider any bird had become extinct in the arid zone, although the night parrot *Pezoporus occidentalis* must have become extremely close. Serventy (1977) noted that up to the 1890s it “*was encountered by virtually every traveller, no matter how lacking in ornithological knowledge or skill*”, but that it has been reported extraordinarily infrequently since, despite many specific searches (Garnett *et al.* 1993).

Across the arid zone, there have been increases in range or abundance since European settlement for 45 species (20% of the original bird fauna), including galah *Cacatua roseicapilla*, crested pigeon *Ocyphaps lophotes*, magpie-lark *Grallina cyanoleuca* and pied butcherbird *Cracticus nigrogularis*. Most of these species are ground-feeding birds favoured by degraded landscapes.

General conclusions from the review by Reid and Fleming (1992) were that:

- there are major differences between environments in the proportion of species affected, with a high proportion of birds associated with chenopod shrublands (such as white-winged fairy-wren *Malurus leucopterus*, thick-billed grass-wren *Amytornis*

*textilis*, redthroat *Pyrrholaemus brunneus*, slender-billed thornbill *Acanthiza iredalei* and rufous fieldwren *Calamanthus campestris*), riparian or floodplain (such as grey grass-wren *Amytornis barbatus* and plains-wanderer *Pedionomus torquatus*) environments declining, but relatively few species associated with mulga woodlands or with the canopy of riparian woodlands affected;

- birds which feed at ground and/or low shrub height have been most adversely affected;
- sedentary bushbirds are more at risk than nomads;
- there are taxonomic biases in the proportion of declining species, with a relatively high proportion of parrots, pigeons, fairy-wrens, quail-thrush and thornbill-type birds in decline;
- there are geographic biases, with species with generally northerly distributions declining most in the arid southern part of their range, and birds with a generally southern distribution declining most in the arid north of their range.

They considered that the most important factor undermining the bird communities was vegetation change due principally to the impact of exotic herbivores (especially livestock and rabbits). They also considered that altered fire regimes, provision of artificial water sources, and feral predators were involved in some of these changes in status. In contrast to the situation for central Australian mammals, declines appear to have been relatively gradual (other than where accelerated by periods of drought) and continuing. Based on the (then) lack of regeneration of dominant plant species in the southern arid zone, they predicted that the rate of decline of some of these bird species was likely to increase.

#### *Examples of changing status in rangeland birds – pastoral Western Australia.*

The bird fauna of pastoral Western Australia has been reported to be relatively resilient. Of 118 bird species (excluding water birds and vagrants) recorded from the Murchison River catchment (Murchison bioregion), Saunders and Curry (1990) listed three regional extinctions and a further 5 declines, a rate of decline they noted to be far less than that in the nearby non-rangeland agricultural district to the immediate south (for which 14 species had become regionally extinct and another 24 species declined out of a bird fauna of 131 species). The regionally extinct species in the Murchison area are two ground-feeding granivores (night parrot and scarlet-chested parrot) and a small insectivore/granivore which forages on the ground and in low foliage (thick-billed grass-wren); the other five declining species also feed on the ground (malleefowl, little button-quail *Turnix velox*, white-winged fairy-wren *Malurus leucopterus*, rufous fieldwren and grey currawong). Increasing species in the Murchison area include at least three species which have colonised over the last century (Bourke's parrot, banded plover, red-tailed black cockatoo), and 18 species which have increased in range or abundance (including emu *Dromaius novaehollandiae*, wedge-tailed eagle *Aquila audax*, inland dotterel *Charadrius australis*, crested pigeon *Ocyphaps lophotes*, galah, welcome swallow *Hirundo neoxena*, grey-crowned babbler *Pomatostomus temporalis*, yellow-rumped thornbill *Acanthiza chrysorrhoa*, banded whiteface *Aphelocephala nigricincta*, spiny-cheeked honeyeater *Acanthogenys rufogularis*, zebra finch *Taeniopygia guttata*, magpie-

lark, black-faced wood-swallow *Artamus cinereus*, pied butcherbird, western bowerbird *Chlamydera guttata*, little crow *Corvus bennetti* and torresian crow *C. orru*). All of these species have increased directly due to changes brought about by the pastoral industry, either through the increased occurrence of water sources and/or the development of a more open ground layer (Saunders and Curry 1990).

In the adjacent Carnarvon Basin, Johnstone *et al.* (in press) noted no regional extinctions (with the possible exception of pheasant coucal *Phasianus centropus*), around 10-15% of 279 recorded species increasing and a similar proportion decreasing over the last century. These species with changing abundance were broadly consistent with those discussed above.

Similar patterns of declining and increasing species have occurred more extensively in Western Australian rangelands, with Curry and Hacker (1990) reporting 11 declining species and 20 increasing species across the entire Western Australian arid zone. This study is not considered in detail here as it was included within the review by Reid and Fleming (1992) for arid Australia generally.

#### *Examples of changing status in rangeland birds – northern Australia.*

Across many environments in northern Australia, a range of bird species has declined, in many cases to regional extinctions (Woinarski 1993a; Franklin 2000). Examples include purple-crowned fairy-wren *Malurus coronatus* and white-browed robin *Poecilodryas superciliosa* in riparian strips in the Kimberley (Smith and Johnstone 1977; Rowley 1993), partridge pigeon *Geophaps smithii* in tropical open forests in the northern Kimberley and monsoonal Northern Territory (Johnstone 1981; Braithwaite and Werner 1987), black treecreeper *Climacteris picumnus melanota* in tropical open forests on Cape York Peninsula (Garnett and Crowley 1995a), star finch *Neochmia ruficauda* and golden-shouldered parrot *Psephotus chrysopterygius* in grasslands on Cape York Peninsula (Garnett and Bredl 1985; Garnett and Crowley 1995b, 1995c, 1997), gouldian finch *Erythrura gouldiae* across much of the savanna woodlands of northern Australia (Tidemann 1993), and flock bronzewing *Phaps histrionica* across the mitchell grasslands (MacGillivray 1932).

The extent and timing of change was examined systematically by Franklin (1999), for granivorous birds (mostly quail, button-quail, parrots, cockatoos, pigeons and finches) across northern Australia. Based on detailed analysis of the historical record, he concluded that 12 of 49 native species had declined (including one extinction and two taxa now critically endangered), three had increased, and four exotic species had become established. Declines were concentrated mostly in central Queensland, though had occurred across virtually the entire area of northern Australia. In the most affected areas, the major wave of decline occurred shortly after the establishment of pastoralism. However, across most of the area, declines are continuing.

Despite the at least superficial appearance of a largely intact natural environment across northern Australia, it is apparent that the loss of bird species in this area has been due to environmental modification, largely because of the impacts of cattle grazing and/or (the often inter-related) changed fire regimes. For example, of the decline in partridge pigeons, Johnstone (1981) noted:

*“There is little doubt that the great decline in numbers in the Northern Territory and east Kimberley is due to over-grazing by domestic stock. This is not only evident with the Partridge Pigeon but also with its close relative the Squatter Pigeon (Geophaps scripta scripta) and other grassland species, such as the Paradise Parrot (Platycercus pulcherrimus) and the Black-throated Finch (Poephila cincta cincta), in eastern Queensland. All are species which favour woodland savannah especially in broad river valleys; this habitat, often the first taken up by pastoralists, is vulnerable to degradation by over-grazing. All four species are now extinct over most of their former range, and all declined rapidly between 1860 and 1920.”*

Likewise, in the Victoria River District:

*“(Cattle) ... concentrate about water and it is then that they have their most direct impact on birds. The birds which depend on grassy habitats are rendered homeless. Birds find the understorey eaten or broken down; their source of food and cover destroyed. Alien plants, Parkinsonia and burrs, find an entry and create new habitats which very few birds can use”* Boekel (1980b); and

*“there is ample evidence to show that Malurus coronatus has suffered a drastic decline since the establishment of the cattle industry. There is no evidence to show that the decline has halted. The next big drought may well be the last for this species, unless positive action is taken to prevent cattle from destroying its habitat”* (Boekel 1979).

For the star finch *Neochmia ruficauda* on Cape York Peninsula, Holmes (1998) noted:

*“... it is most plausible that the species has declined because of pervasive changes in land use. Livestock grazing is almost certainly the most significant factor. Habitat degradation during the dry season or drought would be especially marked near water sources. Since it is grasses and low shrubbery near watercourses and swamps that support the star finch, its foraging and nesting sites would be overgrazed and trampled.”*

And, of the flock bronzewing *Phaps histrionica* in the mitchell grass plains:

*“Gradual overstocking of the country with sheep, cattle and horses, followed by devastating drought, by depriving the birds of their natural shelter and food supply brought their numbers perilously near to the vanishing point”* (MacGillivray 1932).

*Examples of changing status in rangeland birds – central Queensland.*

The rapid change in bird fauna following European entry to the rangelands is best documented for central Queensland (Table 3), where a pioneering pastoralist happened also to be an extremely competent naturalist and kept detailed records of change for a 60 year period following pastoral settlement (Barnard 1925, 1934a). Over this period, from an original (1870) tally of 183 resident or regularly recorded species he chronicled the (property- and local-level) extinction of 12 species (black bittern *Ixobrychus minutus*, black-breasted buzzard *Hamirostra melanosternon*, black-breasted button-quail *Turnix melanogaster*, glossy black cockatoo *Calyptorhynchus lathami*, paradise parrot *Psephotus pulcherrimus*, azure kingfisher *Alcedo azurea*, yellow-tufted honeyeater *Lichenostomus melanops*, eastern whipbird *Psophodes olivaceus*, little shrike-thrush *Colluricincla megarrhyncha*, black-faced monarch *Monarcha melanopsis*, varied triller *Lalage leucomela* and russet-tailed thrush *Zoothera heinei*), substantial declines in a further 23 species, and colonisation by another five species (apostlebird *Struthidea cinerea*, crested pigeon *Ocyphaps lophotes*, cockatiel *Nymphicus hollandicus*, galah *Cacatua roseicapilla* and yellow-throated miner *Manorina flavigula*). The losses included the complete extirpation of the paradise parrot, the only bird species to have become extinct since European settlement of the Australian mainland (Chisholm 1922). A variety of often inter-related (but sometimes idiosyncratic) factors caused these changes, though the main cause was identified by Barnard (and his brother) as the stock on which his livelihood depended:

*“The herds of cattle and sheep fed down the grasses till they completely killed out many of the best kinds, thus destroying the food of the flocks of seed-eating parrots and finches, also quail and pigeons. By making pads (miles in length) going to the waterholes to drink, the earth washed into the swamps and waterholes till they gradually silted up, till now, in all those fine creeks and swamps there is not a drop of water left”* (Barnard 1934b); and

*“Now the stock have trampled away the long grass along the creeks, and the small waterholes have silted up and disappeared”* (Barnard 1925).

This environmental degradation is consistent with the loss of granivorous birds and riparian species such as the black bittern and azure kingfisher. The species which increased are those favoured by proliferation of artificial water sources and an open ground layer.

Barnard (1925, 1934b) also noted that changes in the bird fauna were magnified by the prolonged drought of 1902, which appears to have exacerbated the environmental degradation due to pastoralism.

But change was not restricted to the initial shock of pastoral establishment. Over the period since Barnard's last records (1930s), a further 13 species (spotted harrier *Circus*

*assimilis*, black-tailed native-hen *Gallinula ventralis*, white-eared honeyeater *Lichenostomus leucotis*, crested bellbird *Oreoica gutturalis*, brown songlark *Cinclorhamphus cruralis*, musk lorikeet *Glossopsitta concinna*, singing honeyeater *Lichenostomus virescens*, hooded robin *Melanodryas cucullata*, crested shrike-tit *Falcunculus frontatus*, dusky wood-swallow *Artamus cyanopterus*, black-throated finch *Poephila cincta*, tawny grassbird *Megalurus timoriensis* and diamond firetail *Stagonopleura guttata*) have declined to probable property- and local- level extinction (J. Woinarski & C. Catterall *unpubl.*), a consistent rate of about two species lost per decade since European settlement. These latter losses include species from a broad range of habitats, and suggest responses to ongoing gradual change in the floristic and structural composition of most vegetation types, most probably due to the compounded impacts of grazing and changed fire regimes. A further 50 species appear to have declined further over the last 65 years, indicating that a total of 48% of the bird fauna present at the time of European settlement had undergone a substantial decline in the last 130 years.

**Table 3. Summary of changes in the bird fauna at Coomooboolaroo station, central Queensland, 1870-1999** (from Barnard 1925, 1934a; and Woinarski and Catterall *unpubl.*). Totals are from an original bird fauna 185 resident or regularly recorded species in 1870; and note that some cases of changes in population trends means that the final column is not necessarily simply the sum of the cells from the two time periods.

time period	1870-1934	1934-1999	1870-1999
no. extinct species	12 (6.5%)	13 (7.0%)	25 (13.5%)
no. other declining species	23 (12.4%)	50 (27.0%)	64 (34.6%)
no. increasing spp.	6 (3.2%)	22 (11.9%)	7 (3.8%)
no. new spp.	5	8	13

### Synopsis

There are some consistent patterns in the change in bird fauna across the rangelands, and some regional differences which help suggest causes for the declines.

Firstly, the changes are substantial, with extinctions reported for four of the five areas considered above (Table 4) and declines for a large proportion of the original bird fauna, varying from 7% (in the region with the least adequate historical record) to 48% (in that with the best historical record). The original bird fauna has also changed through the colonisation by some species not originally present and/or by increase in the abundance or distribution of some of the original fauna.

There is a reasonable degree of consistency in the species or types of species which have been most disadvantaged. A high proportion of decrease(d) species forage(d) on the ground or in the understorey; most are (or were) relatively sedentary; many are (or were) habitat

specialists (with a high proportion of these associated with chenopod shrublands or riparian areas). In contrast, the increaser species are typically associated with more open ground (bare areas or low grasses) and/or benefit from the provision of artificial water sources and other infrastructure (Davies 1977). The archetypal increaser species are galah *Cacatua roseicapilla* and crested pigeon *Ocyphaps lophotes*, which have had continental-wide range expansions and increases in abundance (Serventy 1977), in some cases to the direct detriment of other native species (Rowley 1990). The loss of relatively specialised localised species and their “replacement” by widespread weedy species may not change local species richness, but it diminishes biodiversity at a national scale.

**Table 4. Summary of changing status of bird faunas across five rangeland areas.**

change in status	western NSW	arid Australia	Murchison, WA	northern Australia (granivores only)	central Queensland
no. spp. in original fauna	291	230	118	49	185
no. now extinct	6	0	3	1	25
no. other declines	89	29	5	11	64
<i>total % declines</i>	<i>32.6</i>	<i>12.6</i>	<i>6.8</i>	<i>24.4</i>	<i>48.1</i>
no. new spp.	4	0	3	4	13
no. other increases	79	45	18	3	7
<i>% increases</i>	<i>28.1</i>	<i>19.6</i>	<i>17.4</i>	<i>13.5</i>	<i>10.1</i>

The ecological traits and habitat associations of the decreaser bird species render them particularly vulnerable to the impacts of overgrazing (by livestock or feral animals). This factor is probably the single greatest cause of change in the rangeland bird fauna, and has been explicitly linked to many declines. In many cases, the period of most dramatic change in bird faunas has been when the degradation due to pastoralism has been exacerbated by widespread severe drought (Barnard 1925, 1934b; Smith and Smith 1994).

Disentangling the relative impacts of pastoralism from those of extreme climatic events and from other factors which accompanied pastoral settlement (e.g. introduction of feral predators, changed fire regimes, and more localised impacts of tree-clearing and use of pesticides) is almost impossible, as many of these impacts are conflated or operate in a compound way. To some extent, the relative effects of different impacts can be considered by contrasting changes in different regions. There are some substantial

differences between rangeland regions in the extent, timing and rate of changes in the bird fauna. The higher rate of change in the western New South Wales rangeland bird fauna than that of Western Australian is consistent with later, sparser settlement and (consequently) less severe land degradation in the latter area (Smith and Smith 1994). The bird fauna in the south-eastern Australian rangelands also suffered calamitously when the effects of severe drought from 1895 to 1903 were compounded by overstocking and massive rabbit plagues (Smith and Smith 1994). This drought was far less severe in the western Australian rangelands.

While vegetation changes due to grazing by livestock and feral animals is clearly a major cause of change in the rangeland bird fauna, many bird species may have idiosyncratic responses to a range of disturbance factors. For example, predation by foxes may be a major factor in the decline of malleefowl (Priddel 1990). But this species also provides a good illustration of the complexity of factors associated with decline, as it has also been shown to be disadvantaged by grazing by livestock and rabbits (which reduces the abundance and year-round availability of its major food item, seeds), habitat fragmentation, and fire regimes which are either too frequent or too extensive (Frith 1962; Benshemesh 1990; Priddel 1990; Priddel and Wheeler 1990; Woinarski 1999).

Many of these threatening factors continue to disturb the rangelands. While the most susceptible bird species and environments probably crashed in the initial shock of pastoral settlement, many species continue to decline, or have not yet recovered, despite often more benevolent land management practices operating today. In many cases, the threatening processes are acting far more insidiously than the initial impact of pastoral settlement (often soon followed by catastrophic drought), and hence the environmental changes (and resultant declines in bird fauna) are subtle, but nonetheless inexorable.

## Reptiles and amphibians

Compared to mammals and birds, there is appreciably less historical information on the reptiles and amphibians of the rangelands and, perhaps as a consequence, no previous suggestion of major change or substantial conservation problem for these groups (e.g. Recher and Lim 1990; Cogger *et al.* 1993).

There have been no previous substantial reviews of the status of these groups in the rangelands, although there have been some compilations of herpetofaunal (reptile and frog) status for large areas within the rangelands, notably for the entire Northern Territory (Woinarski 1993b) and Western Australia (Burbidge and Wyre 1993), for western New South Wales (Tyler 1994; Sadlier 1994; Sadlier and Pressey 1994), for central Queensland (Covacevich *et al.* 1998), and for central Australia (Kerle and Fleming 1996). There have also been more detailed reviews of the status and management of individual rangeland species (including Ehmann 1992; Reid *et al.* 1993; Griffiths and Christian 1996), and research studies examining the response of herpetofauna to land use, management or other threatening processes (including Fyfe 1980; Friend and Cellier 1990; Ehmann 1992; Masters 1996; Griffiths 1998; Fisher 1999; Woinarski *et al.* 1999).

### *Examples of changing status in rangeland herpetofauna – western New South Wales.*

Compared to the frog fauna of non-rangeland eastern Australia, so little is known about the 20 species comprising the frog fauna of the New South Wales rangeland that it is almost impossible to discern any historical changes in abundance or distribution (Tyler 1994), although Robertson *et al.* (1989) reported anecdotal evidence of decline in the green and golden bell frog *Litoria raniformis* in the Murray River system, on the periphery of the rangelands.

There is reasonable evidence for decline in abundance or range for six (the fossorial skink *Anomalopus mackayi*, western blue-tongued lizard *Tiliqua occipitalis*, the woma (a python) *Aspidites ramsayi*, carpet python *Morelia spilota*, tiger snake *Notechis scutatus* and red-bellied black snake *Pseudechis porphyriacus*) of the 112 reptile species in this area (Sadlier and Pressey 1994). Additionally, extensive land clearance, especially in mallee environments around the margins of the rangeland area, has led to many local extinctions or to marked local population reductions of many other reptile species (Cogger 1984, 1989; Ehmann and Cogger 1985).

While there are some idiosyncracies among species in the factors contributing to these declines, four main factors have been generally implicated – changed fire regimes, land clearance, pastoralism and feral predators (Cogger 1989; Sadlier and Pressey 1994).

For the skink *Anomalopus mackayi*, Sadlier and Pressey (1994) considered:

*“The decline ... may be due, at least partly, to the impacts of stock grazing and clearing. There are several aspects of the ecology ... that make them particularly susceptible to habitat disturbance. They tend to burrow in loose soil or use existing soil hollows ... Loss of soil structure could ... isolate populations in unaffected areas. Fallen timber ... appears to provide important sheltering sites so the removal of such refuges could influence population densities ... The species could also be susceptible to reduction in the humidity of the subsurface soil layer after removal of surface debris”.*

Most declines of herpetofauna in western New South Wales have been in riparian strips and in habitats with *Triodia* (spinifex, or hummock grass) understorey, reflecting concentrated impacts upon these environments by livestock, and deliberate habitat modification (for riparian systems), clearing and changed fire regimes (for *Triodia* environments) (Sadler and Pressey 1994).

Given the extremely imprecise and fragmentary historical record, it is impossible to date the onset of declines, but Sadler and Pressey (1994) suggest that declines are ongoing and probably have not yet peaked.

#### *Examples of changing status in rangeland herpetofauna – central Queensland.*

Land clearance, principally for pasture improvement, has also been the major factor causing declines in the herpetofaunal communities of central Queensland. Of 148 reptile species recorded from the Brigalow Belt, one is probably now extinct, 12 others have undergone substantial declines and another five species have suffered at least major local losses (Covacevich *et al.* 1996, 1998), with these losses occurring over the last 60 years, and continuing.

#### *Examples of changing status in rangeland herpetofauna – Northern Territory.*

Woinarski (1993b) listed 44 frog species and 281 reptile species from the Northern Territory. Of this high tally, no frogs are reported to have declined, and only two reptile species (the bronzeback legless lizard *Ophiodiocephalus taeniatus* and great desert skink *Egernia kintorei*) show unequivocal evidence of decline (probably due to the impacts of pastoralism (Ehmann 1992), and to changed fire regimes and/or feral predators, respectively), although again the historical record is especially meagre.

Morton (1990) similarly reported that the reptile fauna of central Australia appeared to be remarkably resilient, claiming that no species had undergone any decline. Subsequently, Kerle and Fleming (1996) provided a more direct comparison between the current central Australian reptile fauna and that described (with 56 species) by the Horn Expedition of 1894. They concluded that four species (the painted dragon *Ctenophorus pictus*, and

three skinks *Egernia margaretae*, *E. stokesii* and *Ctenotus strauchii*) from this fauna “have suffered considerable reduction in range and may become regionally extinct”.

Notwithstanding the limited evidence of overall decline, many studies in the rangelands of the Northern Territory have demonstrated local changes in the abundance of individual reptile species, the species composition of reptile assemblages, and/or the local-scale species richness, in response to different fire regimes (Fyfe 1980; Braithwaite 1987; Reid *et al.* 1993; Trainor and Woinarski 1994; Masters 1996; Griffiths and Christian 1996; Woinarski *et al.* 1999), and to different levels of grazing by livestock or feral herbivores (Braithwaite *et al.* 1984; Friend and Cellier 1990; Ehmann 1992; Fisher 1999; Schlesinger 1999; Woinarski and Ash ms). The often marked response to these factors, and the landscape-wide imposition of grazing and fire regimes, suggests that many of these changes in the herpetofauna are probably extensive and sustained, but have not been evident because of the thin historical record and lack of extensive current monitoring.

The most conspicuous and well documented change in status for rangeland herpetofauna has been the extraordinary spread and superabundance of the cane toad *Bufo marinus* across much of the rangelands of northeastern Australia, with subsequent ongoing expansion into the Top End of the Northern Territory and towards the rangelands of southeastern Australia (e.g. Freeland 1986), with possible (although contested) impacts on native fauna (Freeland and Kerin 1988; Jacklyn 1992; Burnett 1997)

### Synopsis

It is clearly very difficult to be sure how herpetofaunal biodiversity is faring in the rangelands. The historical record is poor and current surveys are generally far from systematic. Much of the evidence of changing status is anecdotal (Pearson 1993; Sadlier and Pressey 1994). Nonetheless, marked responses of many reptile species to land management changes suggest that there may be large undescribed impacts operating over very extensive areas of the rangelands.

There is only little consistent pattern among the species reported to be in decline. Some are litter-dwelling species, which are disadvantaged when litter or other ground cover is reduced (by frequent fire and/or grazing). Some are relatively large slow-moving sedentary species (*Egernia* spp and *Tiliqua occipitalis*) which may be especially vulnerable to feral predators and/or which may not disperse well to new areas when their immediate environment is degraded. Several are relatively large snakes, which may be affected by feral predators, changed fire regimes, clearing, loss of normal prey species, or removal of cover.

While change in the rangeland herpetofauna is evident, the extent of this change is clearly far less than that affecting the frog fauna of the non-rangeland eastern coast of Australia, where there has been a major episode of extinctions over the last three decades (Ingram and McDonald 1993).

It also seems likely that change in the rangeland reptile fauna is less than that in the more extensively cleared lands around the rangeland perimeter (Sadler and Pressey 1994; Covacevich *et al.* 1996, 1998), with evidence that rangeland populations of particular species appear stable whereas populations of the same species in the agricultural zone have declined sharply or have suffered regional extinctions (Pearson 1993).

## Invertebrates

Much more so than for vertebrates, assessment of change in the invertebrate fauna of Australia's rangelands is hampered by a very poor historical foundation - and, across large areas, not appreciably better current information (Yen and Butcher 1997). Probably the most substantial consideration of change is that of Yen (1996), who compared the invertebrate fauna collected on the Horn Expedition to central Australia in 1894, with that occurring now. Unfortunately, there were major biases in the methodology of the early collecting, and the total collection of invertebrates on the Horn Expedition was not large, reducing Yen (1996) to conclude that "*with regard to invertebrates, there is simply not enough information to determine whether any species have declined or become extinct over the last 100 years*", but he did note that one earthworm species had not been collected subsequent to the Horn Expedition, and one beetle species collected then was still known only from the type locality.

Lacking then an adequate historical baseline from which to measure departure, it is possible to infer change in the invertebrate communities of rangelands through studies of the impacts of land use or management regimes. There are now a number of important studies within the rangelands which demonstrate such responses, and a well-established recognition that invertebrates may be useful indicators of environmental change (Majer and Beeston 1996; Andersen 1997, 1999; Andersen and Sparling 1997).

With respect to effects of *pastoralism*:

- across a series of sites in *Acacia* shrublands and chenopod shrublands, Landsberg *et al.* (1999) noted that 26% of ant species showed significant decreases in abundance and 24% a significant increase in abundance, with increasing grazing pressure; but that responses were far more muted for Coleoptera (beetles; 2% of species with clear relationship with grazing intensity), Collembola (12%) and Orthoptera (grasshoppers; 5%).
- mygalomorph (trapdoor) spiders require a stable litter layer around their burrows, and studies in south-western Australia have demonstrated that this is readily disturbed by grazing, leading to declines and local extinctions (Main 1987).
- in savanna woodlands in northern Queensland, the composition of ant and spider assemblages differed markedly between grazed areas and adjacent ungrazed areas (Woinarski *et al.* ms), and ant species richness was significantly less in grazed areas.
- in savanna woodlands in the Victoria River District of the Northern Territory, the number of grasshopper species declined significantly in association with increasing intensity of grazing by cattle, with the response most marked within 500 m of an artificial water sources (Ludwig *et al.* 1999).
- also in the tropical savannas, the species composition of spiders changed markedly across a gradient of grazing intensity (Churchill 1998), with some taxonomic groups notably less abundant in heavily grazed areas.

- based on two studies in tropical savanna woodlands and *Acacia* shrublands, Majer and Beeston (1996) noted that grazed areas may have higher species richness for ants than ungrazed (or lightly grazed) areas, but that there was a change in ant species composition (although they noted that this was less substantial than for more extreme disturbances such as mining and clearing).
- in chenopod shrublands in northern South Australia, Read (1999) reported increases in ant abundance, but decrease in the abundance of all other invertebrates, immediately after the introduction of heavy grazing, with inconsistent or negligible longer-term responses for most invertebrate groups considered separately.
- in chenopod shrublands in Western Australia, the species richness of soil mites (Acarina) was substantially less in a “good condition” site exposed to 10 years of heavy grazing compared with one exposed to light grazing (Kinnear and Tongway 1999); and a “poor condition” site (exposed to a far longer history of intense grazing) had both reduced richness and abundance.

With respect to effects of *fire regimes*:

- in eucalypt forests of the north Australian rangelands, ant assemblages diverge substantially in species richness, abundance and species composition when exposed to different fire regimes, typically with more species and individuals of ants (especially of *Iridomyrmex* spp.) in frequently burnt areas, but also with some species (and species-groups) favouring infrequently burnt areas (Andersen 1991b).
- in the sandstone heathlands of the Pine Creek- Arnhem bioregion of northern Australian rangelands, the distinctive Leichhardt’s grasshopper *Petasida ephippigera* is locally extirpated when the interval between fires is less than about 4-5 years (the period required for regeneration of its obligate-seeder host plant *Pityrodia* spp.) (Lowe 1995).
- in mallee eucalypt vegetation of the southeastern rangelands, there are major differences in the composition of foliage invertebrates between mature trees and post-fire coppice (Yen 1989).
- in mallee eucalypt vegetation of the southeastern rangelands, there is a very marked change in ant species composition following wildfire, with dominant ground-foraging ants severely depleted (Andersen and Yen 1985).
- some invertebrate species (characteristically including the mygalomorph (trapdoor) spiders and camaenid landsnails) are poor dispersers and have highly restricted ranges, typically associated with patches of microclimate moderation and stability; and these patches, and hence their associated invertebrate fauna, are presumed to be very non-resilient to increases in fire intensity or frequency, although this conclusion is largely inferential from the few studies which have been done in areas near rangelands, and from snippets of anecdotal notes (Main 1987; Solem 1993; Yen 1995; Friend 1995).

With respect to effects of *mining*:

- several studies in rangeland areas have shown a marked reduction in diversity of invertebrate assemblages in mined areas, and only gradual and incomplete re-assembly with vegetation regeneration after mining (Andersen 1993; Read 1996; Andersen and Sparling 1997; Griffiths 1998; Read and Pickering 1999).

Taken together, these studies suggest that the invertebrate fauna of the rangelands may have been subject to substantial re-arrangement at localities subject to major disturbance (e.g. mines and “sacrifice areas” around bores), and more subdued changes extensively across the rangelands. Some groups of species (e.g. spiders, scorpions, grasshoppers, ants) have been more susceptible than others (e.g. termites) (Read 1999).

In contrast to most vertebrates, many invertebrate species have particularly narrow ranges. This is most notable in “relict” species with poor dispersal abilities, such as land snails and mygalomorph (trapdoor) spiders (Solem and McKenzie 1991; Stanistic 1999; Main 1999). Many species in these groups have a tenuous hold in the rangelands (typically occurring in unusually fertile, moist or sheltered habitats), and are readily eliminated by disturbance (fire, grazing, clearing) to the few sites of their occurrence (Stanistic 1999; Main 1999). As an extreme example, Solem (1993) records that the total known distribution of one central Australian land snail species is restricted to the litter under a single fig tree, with many more examples for other species of ranges which are nearly as localised.

As with other parts of Australia, the rangelands have been colonised by a large number of introduced invertebrate species (around 500 species of exotic insects, Collembola and arachnids have become established across Australia: Yen and Butcher 1997), in some cases to the detriment of native invertebrates and other components of biodiversity (e.g. Hoffman *et al.* 1999). Spessa (1999) lists some introduced invertebrates considered worthy of monitoring (largely because of their pest status).

## Plants

Unlike mammals, it is difficult to generalise about which plants have been most depleted under European management in Australian rangelands. One estimate of the extent of loss is given by Briggs and Leigh (1995), who considered that about 17 species of plants had become extinct in the Australian rangelands since European settlement. This tally is appreciably less than that given for non-rangeland areas of south-western and eastern Australia, but the authors caution that their assessment “reflects the lack of detailed plant distribution knowledge ... particularly for Western Australia, the Northern Territory and Cape York Peninsula”. Many more rangeland plant species have declined significantly, with around 200-300 rangeland plant species now recognised as endangered or vulnerable (Briggs and Leigh 1995). Most of the listed threatened plant species in rangelands occupy specialised habitats, such as mountain ranges and granite outcrops, which are currently sheltered from habitat destruction. For example, of the four species and one variety listed as threatened in the Pilbara and Kimberley regions of Western Australia, all are either from specialised or rugged habitats. However, other threatened species with localised and specialised habitats happen to occur at sites which are now subject to substantial impacts, such as mound springs in the rangelands of northern South Australia and western Queensland.

The effects of pastoralism on the vegetation of Australia’s rangelands will never be fully understood. Clearly the species that are most palatable, most sensitive to trampling or grazing and have limited capacity for regeneration will have been the most seriously affected. For example, there is a suite of daisies with a particular morphology that may be especially vulnerable: *Microseris lanceolata* (McIntyre 1995), *Picris evae* (Fensham 1998), *Stemmacantha australis* (Fensham 1998) and *Trioncinia retroflexa* (Fensham 1999a) have all become extremely rare in pastoral landscapes, and now are largely confined to roadsides. They are all daisies with a fleshy carrot-like tap root. Anecdotal evidence would suggest that these species may not be especially preferred as fodder by stock. However, the breaking of this fleshy tap root as a result of grazing or trampling may be the factor that renders them especially vulnerable to stock grazing. In Appendix B, we have compiled a list of the reported responses of plant species to grazing, from studies across the Australian rangelands. The species have been categorised using agronomic terminology as either “increasers” or “decreasers” under grazing.

It is evident that individual species can behave in contrary ways in different environments. For example, the important fodder grass *Dichanthium sericeum* is recorded as an increaser in the eucalypt woodlands of north Queensland (Fensham and Skull 1999) and a decreaser on the grasslands of central and northern Queensland (Hall and Lee 1980; Fensham *et al.* 1999). In this instance this paradox can be explained because in the former case the species has an annual lifeform and in the latter case a perennial lifeform. In other cases the reason is obscure and may sometimes be an artefact of inefficient sampling. The response of some of the species listed in Appendix B have not been statistically tested. Most species have a bell-shaped response along

environmental gradients and Fensham *et al.* (1999) demonstrated that this general rule applies to grazing intensity. Thus some species have a favoured position that is intermediate along a grazing gradient and decrease under both light and heavy grazing. Thus contradictory responses of species may result from sampling of different portions of the gradient. In other cases contradictory responses may well represent the fact that species behave differently in different environments, under different seasonal conditions or under grazing by different animals. McIvor (1998) listed the daisy *Camptactra barbata* as both an increaser and a decreaser within the same study at two different locations.

**Table 5. Percentage of increasers and decreasers, by plant family.** Data compiled from Appendix B; for only those families with a sample size greater than 9.

<b>Family</b>	<b>Decreasers</b>	<b>Increasers</b>	<b>Grand Total</b>
Nyctaginaceae	30.0	70.0	10
Convolvulaceae	54.5	45.5	11
Cyperaceae	36.4	63.6	11
Zygophyllaceae	41.7	58.3	12
Goodeniaceae	53.8	46.2	13
Myoporaceae	60.0	40.0	15
Portulacaceae	53.3	46.7	15
Brassicaceae	68.8	31.3	16
Caesalpiniaceae	70.6	29.4	17
Euphorbiaceae	58.8	41.2	17
Solanaceae	52.6	47.4	19
Amaranthaceae	54.5	45.5	22
Mimosaceae	54.5	45.5	22
Fabaceae	38.2	61.8	34
Malvaceae	53.2	46.8	47
Asteraceae	64.5	35.5	107
Chenopodiaceae	55.3	44.7	114
Poaceae	58.0	42.0	119
<b>Grand Total</b>	<b>56.9</b>	<b>43.1</b>	<b>731</b>

An analysis of the data compiled in Appendix B reveals that there are few clear patterns (Tables 5,6). The species recorded as known decreasers include 51 plant families and the increasers include 43 plant families. Overall 57% of species are decreasers and 43% increasers. There are no plant families that are obviously prevalent in either the increaser or decreaser category, although the Asteraceae have a greater proportion of decreasers than average and Fabaceae a greater proportion of increasers. As has been generalised from world-wide studies (Milchunas and Lauenroth 1993), annual grasses are generally

favoured by grazing. This is weakly evidenced in this analysis where in total 55% of annual grasses are increasers and 45% are decreasers, although the pattern is reversed in Chenopod/Acacia shrublands. Perennial grasses are clearly and perennial herbs and shrubs generally disfavoured by grazing. Apart from these general patterns there do not appear to be any obvious general rules about the vulnerability of plant species to grazing, with species representing a wide range of lifeforms showing increases or decreases.

Tree clearing has an obvious adverse effect on trees and shrubs. For example, Brigalow *Acacia harpophylla* and its associated tree species have been drastically reduced in abundance. The species was, however, so abundant in the forest and woodlands that it dominated, that it is still widespread with stands on roadsides, National Parks, as an infrequent species in more marginal vegetation types and as regrowth in grazing paddocks. To a lesser extent many of the constituent species are also relatively secure at the species level. Some of the species associated with brigalow which have a patchy distribution, such as ooline (*Cadellia pentastylis*), have been extensively cleared and the species is now classified as vulnerable. Fairfax and Fensham (2000) document a marked reduction in species diversity across a wide range of plant lifeforms in response to clearing and subsequent establishment of buffel grass pasture.

**Table 6. Percentage of increasers and decreasers, by plant broad functional group and environment.** Data compiled from Appendix B; for only those families with a sample size greater than 9. "inc"=number of increaser species. "dec"=number of decreaser species.

Plant life form	Environment								Total	
	Chenopod /Acacia shrubland		Grassland		Mulga		Tropical eucalypt woodland			
	inc	dec	inc	dec	inc	dec	inc	dec	inc	dec
<i>annual grasses</i>	10	15	10	3	5	7	6	0	31	25
<i>annual herbs</i>	55	92	25	10	17	25	19	3	116	130
<i>perennial grasses</i>	6	14	2	8	5	15	6	9	19	46
<i>perennial herbs</i>	54	77	16	8	14	34	12	11	96	130
<i>shrubs</i>	29	46	0	2	7	13	0	1	36	62
<i>trees</i>	9	17	-	-	3	1	-	-	12	18

## **Ecological processes**

Biodiversity is more than a static listing of species and environments. It also includes, indeed depends upon, ecological processes which link and maintain individual living entities. Such processes include pollination, seed dispersal, fire, and cycling of nutrients and water. Some of these processes provide a major link between biodiversity and environmental health and productivity. We consider some of these processes below. This treatment is brief, as, for most ecological processes, little is known of their workings or of the extent to which they may have been disrupted. We do not discuss fire here, as we consider it instead in a subsequent section on threatening processes. We do not consider hydrological processes, as these are largely outside our brief. We also do not attempt any assessment of the economic benefit of these processes, or of the biodiversity contributing to them. A substantial project on such ecological services, including a detailed case study in one rangeland area, is being undertaken elsewhere (refer to <http://www.dwe.csiro.au/ecoservices>).

### **Nutrient cycling**

Nutrient cycling involves a complex relationship between soils, vegetation, mycorrhizal fungi, consumer fauna, and land use. The maintenance of this relationship is critical for long-term sustainability, especially in the generally nutrient-poor Australian rangelands. Over-stocking can lead to the rapid and irreversible loss of protective soil crusts, decline in cover of perennial grasses, change in composition of functionally important fauna, and hence diminution of soil fertility (e.g. Ludwig and Tongway 1995; Ragupathy *et al.* 1999; Noble *et al.* 1999; Northup and Brown 1999*a,b*).

Termites are one important component of the nutrient cycling pathway, and are a critical component of biodiversity structuring in the rangelands generally (Morton and James 1988). On the rangeland margins of south-western Australia, harvester termites were found to be less abundant in areas exposed to grazing by sheep (Abensperg-Traun *et al.* 1996); however another study in the same area found no difference in termite richness between grazed and ungrazed areas, but an increase in mound density in the grazed areas (Abensperg-Traun 1992). Livestock-grazing has also been associated with increases in the abundance of at least some termite species or groups in mulga woodlands (Watson *et al.* 1973) and in tropical eucalypt forests and savanna woodlands (Braithwaite *et al.* 1988).

Soil mites (Acarina) are another component of biodiversity to provide important ecological services, in the decomposition of soil organic matter and the maintenance of soil structure. The decline in total abundance and richness of the soil mite fauna with increasing grazing pressure (Kinnear and Tongway 1999) will result in diminished soil fertility and structure, and contribute towards desertification.

The loss of many rangeland mammal species over the last century may have had some pervasive flow-on impacts to environmental health and function. One of the most abundant macropods across almost all rangeland areas was the burrowing bettong (*Bettongia lesueur*), which constructed elaborate and extensive warrens. Noble (1999) considered that these burrow systems “served to maintain a high degree of mesoscale (50 m up to 1 km) surface heterogeneity that, in turn, structured both plant and animal communities ... This landscape heterogeneity has undoubtedly influenced a number of fundamental landscape processes and vegetation dynamics ... (they) clearly act as major obstruction elements trapping large quantities of runoff-transported topsoil and organic matter above, and within, the warren.” Based on the ecological impacts of this burrowing activity, Noble *et al.* (1999) considered that burrowing bettongs were “landscape engineers”, and played a critical role in soil biodiversity and fertility, and vegetation patterning in at least Mulga (*Acacia aneura*) woodlands. Similar roles may have been played by mallee fowl (*Leipoa ocellata*) and some of the larger megafauna which became extinct in the Pleistocene (Noble 1993; Noble *et al.* 1999). Noble *et al.* (1999) considered that the regional extinction of this native fauna “has been a neglected facet of desertification that has yet to register its full impact.”

To some extent, the widespread loss of these species and their function may have been counterbalanced by the spread of rabbits, but the burrows constructed by these newcomers are generally far less substantial than those of the native fauna they have replaced: rabbit warrens are also characteristically convex and denuded of perennial plant cover, with the result that they typically become resource-poor patches (Eldridge and Myers 1999).

## Pollination

Some invertebrate and vertebrate groups play crucial roles in the pollination of plants. With few exceptions, nectarivorous birds and bats have shown little change in distribution or abundance across most of the rangelands. The exceptions are mostly around the more modified fringes of the rangelands: the localised extinction of some honeyeater species from southwestern New South Wales because of habitat loss (Smith and Smith 1994), regional declines in some honeyeaters and lorikeets in central Queensland, probably due to extensive tree clearing (Nix 1993; Woinarski and Catterall *unpubl.*), and changes in the distributions of some partly nectarivorous flying-foxes in north-eastern Australia, again probably due to habitat fragmentation (Webb and Tidemann 1995). As many nectarivores must move over large areas and across a range of habitats to maintain year-round access to resources (Palmer and Woinarski 1999), the loss of these species may reverberate through many environments. While losses of the vertebrate nectarivores currently appear to be localised and restricted mainly to those rangeland areas subject to wholesale environmental modification, decline may also be occurring more subtly across the extensive core of the rangelands, as land management practices (such as imposed fire regimes and grazing) affect the timing and productivity of flowering events and the species composition of vegetation (e.g. Setterfield and Williams 1996; Setterfield 1997; Williams 1997).

A rich fauna of native bees and other insects plays a pivotal role in the pollination of very many Australian plants (Schwarz and Hogendoorn 1999). In some rangeland areas, elements of this fauna are in decline. In some cases (although the evidence is generally relatively thin: Schwarz and Hurst 1997), this may be related directly to the spread of exotic pollinator species, which may have different preferences among flowering plants, and may be far less effective at pollination, to the detriment of at least some native plant species (Taylor and Whelan 1988; Paton 1997).

## Seed dispersal

Many vertebrate and invertebrate species in Australia's rangelands are specialised granivores (seed-eaters) or frugivores (fruit-eaters) (Morton 1985; Andersen 1991; Franklin *in press*). These species play a critical role in vegetation dynamics, influencing plant recruitment, dispersal and floristic composition. In some cases, these interactions are extraordinarily specific. For example, in the tropical savanna rangelands of northern Australia, ants in the *Meranoplus diversus* group are reliant on only one or few plant species (typically *Sorghum* spp.), and will become locally extinct with loss of such plants. Conversely, the ants may play a key role in the local-scale distribution and germination pattern of the *Sorghum*, through hoarding much of the available seed in underground chambers (Andrew 1986). At a far coarser ecological scale, large-seeded plants may achieve long-distance dispersal only through passage with highly mobile vertebrates. The decline of some of these vertebrates across rangeland areas (such as the Emu *Dromaius novaehollandiae* in parts of northern Australia) may prevent emigration of plants for distant recruitment, and hence distort local and regional plant dynamics and responses to disturbance (Noble 1975; McGrath and Bass 1999). Conversely, management practices (such as the imposition of changed fire regimes and grazing) may alter seeding and fruiting patterns and productivity, to the detriment of the dependent fauna (Williams 1997; Crowley and Garnett 1999).

As with nectarivory, the availability of seeds or fruits may fluctuate asynchronously across the landscape, forcing species dependent upon them to undertake landscape-wide movements to track resources. The inter-dependence of these diverse elements of the biota may render the linkages particularly susceptible, and require conservation management responses which consider and protect all integrated components simultaneously, across scales from the local to the broad regional (Price *et al.* 1999).

## **Ecosystems**

The rangelands comprise a distinctive array of ecosystems, of varying resilience and level of use. Here, we provide a brief review of the main or most significant of these environments, largely seeking to indicate the extent to which they have been (or are being) modified, their susceptibility to disturbance, and their conservation status. This review is limited by the current unsatisfactory level of consistency and resolution of vegetation mapping across Australia, a problem which may soon be resolved through the Audit's Theme 3. This treatment builds upon reviews of the reservation status of vegetation types compiled for most rangeland jurisdictions, notably for Queensland by Sattler and Williams (1999), for the Northern Territory by Connors *et al.* (1996), for Western Australia by Hopkins *et al.* (1996) and for New South Wales by Benson (1999).

### **Chenopod shrublands**

Chenopod shrublands (typically dominated by bluebush and saltbush, mostly *Chenopodium*, *Maireana*, *Sclerolaena* and *Atriplex* species) extend across between 400,000 to 500,000 km<sup>2</sup> (or 8% of the rangeland area) (Leigh 1981; Graetz and Wilson 1984). Although diversity is generally relatively low in chenopod shrublands, many of the constituent species are highly restricted to this environment (Leigh 1981).

The chenopod shrublands have fared badly since European settlement of the rangelands. Newman and Condon (1969) considered that 25% of the chenopod shrublands were "severely degenerated", 40% were "moderately deteriorated", 25% showed minor degeneration and only 10% showed little or no deterioration. Of 280 endemic chenopod plant species, 3% are now considered endangered and 6% are vulnerable (Hartley and Leigh 1979).

Relatively little of this environment is represented in conservation reserves. For example, in the Northern Territory only 0.1% of the area of chenopod shrublands is protected (Woinarski *et al.* 1996). Of 19 chenopod shrubland communities defined by Specht *et al.* (1995) for the Australian rangelands, three were not represented at all in conservation reserves and one other had "poor" representation. Instead, most chenopod shrublands are used for pastoralism, and are regarded as one of the most favoured grazing areas, for sheep in the southern rangelands and cattle in the northern rangelands.

The impacts of grazing have been substantial, most notably in floristic and structural changes over extensive areas (Lay 1979; Graetz and Wilson 1984; Lange *et al.* 1984; Graetz *et al.* 1995). Grazing by feral animals (principally rabbits and camels) has exacerbated damage on pastoral lands, and had major impacts on chenopod shrublands in non-pastoral areas (Beard 1975; McKenzie and Burbidge 1987). Much of the degradation now evident occurred soon after pastoral occupation (Williams and Oxley

1979; Lange *et al.* 1984), but impact is continuing (Lay 1979). The major impacts have been loss of the preferred grasses and forbs and, under heavy grazing, loss of the chenopod shrubs themselves (Graetz and Wilson 1984; Eldridge *et al.* 1990; Read 1999). When the shrubs are removed by overgrazing, weedy species may invade, soils become unstable (especially for duplex soils), litter and lichen cover is reduced, rainfall infiltration is affected and there is increased loss of soil nutrients (Graetz and Wilson 1984; Graetz and Tongway 1986). Alternatively, in some cases, a stable subclimax grassland may replace the shrublands, and this grassland may be favoured by pastoralists because of its greater productivity (Leigh 1981; Benson *et al.* 1997). Under moderate but continuous grazing, recruitment strategies for some major plant species are stymied, leading to senescence and eventual loss of those species (Graetz and Wilson 1984). This resulting floristic change may be positive, negative or neutral for productivity, but undoubtedly has ecological impacts. Absence of grazing may also lead to senescence, for at least some plant species in some areas (Graetz and Wilson 1984).

Because of the major floristic and structural vegetation responses to grazing pressure in chenopod shrublands, there is typically a pronounced piosphere effect. A high proportion of “decreaser” plant species typically occurs only or mostly at sites remote from water (Landsberg *et al.* 1997). As only about 25% of chenopod shrublands in the pastoral rangelands now lie more than 10 km from a water point (Landsberg and Gillieson 1996), the distribution and abundance of such decreaser species is probably markedly reduced across the entire rangelands.

Although less clearcut than impacts upon plant biodiversity, the fauna of chenopod shrublands has also suffered some pronounced impacts. This is most apparent for birds, where a number of species are closely associated with this environment (e.g. white-winged fairy-wren *Malurus leucopterus*, thick-billed grass-wren *Amytornis textilis*, redthroat *Pyrholaemus brunneus*, slender-billed thornbill *Acanthiza iredalei* and rufous fieldwren *Calamanthus campestris*), and a high proportion of these has declined, in some cases to local and regional extinction (Reid and Fleming 1992; Burbidge and Pedler 1993). Chenopod shrublands were also the preferred habitat for some rangeland mammal species, such as the desert rat-kangaroo *Caloprymnus campestris*. Degradation of chenopod shrublands most probably contributed to the extinction of these species (Carr and Robinson 1997). There is little evidence about the status of other faunal groups, except that Read (1999) recorded some changes in ant species composition between grazed and ungrazed chenopod shrublands and Landsberg *et al.* (1999) recorded a high proportion (10-38%) of ant species (but not beetle, grasshopper or collembolan species) showing significant associations with either more or less grazing intensity across several chenopod shrubland piosphere sites.

Although typically a rare event, fire may have a severe effect on chenopod shrublands, particularly when fire is followed by heavy grazing (Hodgkinson and Griffin 1982), or when a second fire follows soon after the first.

In higher rainfall areas of southern Australia, some chenopod shrublands have been cleared for cropping (Leigh 19981), with obviously acute local biodiversity consequences.

At least in the Nullabor bioregion, chenopod shrublands have also been degraded by weed invasion (McKenzie and Robinson 1987; Burbidge and Pedler 1993), especially by wards weed *Carrichtera annua*.

## Tussock grasslands

Native tussock grasslands in the rangelands largely comprise mitchell grasslands (dominated by *Astrebla* species) in arid and semi-arid areas of northern and northeastern Australia, with a total extent of about 320,000 km<sup>2</sup> (or 6% of the rangeland area) (Orr and Holmes 1984). More restricted tussock grassland types also include floodplains (dominated by the grasses and sedges *Oryza rufipogon*, *Eleocharis*, *Pseudoraphis*, *Hymenachne* and *Cyperus* spp.) in higher rainfall areas of northern and northeastern Australia, bluegrass grasslands (dominated mainly by *Dichanthium* species) in northeastern Australia, and relatively scattered semi-arid grasslands (dominated variably by *Stipa*, *Eragrostis*, *Aristida*, *Enneapogon* and *Sporobolus* species) in southeastern Australia (Groves and Williams 1981).

Of 20 rangeland grassland communities defined by Specht *et al.* (1995), six were unreserved, with three others “very poorly conserved” and one other “poorly conserved”. In the Northern Territory, only 0.3% of the extent of tussock grasslands is included within conservation reserves (Woinarski *et al.* 1996). Tussock grasslands comprise a high proportion of the “regional ecosystems” regarded as “at risk” in the Queensland rangelands (Sattler and Williams 1999).

Reflecting their value for grazing, most of the tussock grasslands in the rangelands are in pastoral tenure (e.g., more than 90% for the Northern Territory: Woinarski *et al.* 1996). The proliferation of artificial water sources across most tussock grassland areas of the rangelands has ensured that few grassland areas are ungrazed. For example, Fisher (1999) reported that less than 0.2% of the mitchell grasslands in the Northern Territory was now more than 10 km distant (and only 20% of their extent was more than 5 km distant) from a permanent water source, a vastly different distribution to that preceding pastoral development. Infrastructure development is even more intensive in the Queensland component of the mitchell grasslands, suggesting further deviation from pre-pastoral conditions. The impact of closer water points is evident from piosphere studies, which demonstrate that some animal and plant species are significantly more abundant (or occur only) in areas remote from water (i.e. relatively little grazed) (James *et al.* 1999). Based on these studies, Fisher (1999) estimated that at least four animal species (the lizards *Delma tincta* and *Ctenotus pulchellus*, the snake *Demansia torquata* and the little button-quail *Turnix velox*) and two plant species (*Astrebla squarrosa* and *A. pectinata*) had declined in total population by at least 50% since the advent of pastoralism, with declines expected to continue with increasing intensity of pastoral use. Pastoral impacts have also included an increase in the relative abundance of annual grasses (e.g. *Brachyachne convergens*, *Panicum laevinode*) at the expense of perennial grasses. A range of animal species favoured by short grass, open areas and increased occurrence of permanent water (e.g. galah, crested pigeon, Australian pratincole) have increased in distribution and abundance (Fisher 1999). However, these relationships with distance from water are generally less pronounced, and involve a smaller proportion of species, than comparable relationships recently found in mulga woodlands and chenopod

shrublands (Landsberg *et al.* 1997), possibly suggesting that the biota of mitchell grasslands is generally resilient, at least to the levels of grazing now operating. This apparent resilience of mitchell grassland to moderate levels of grazing has been claimed by a number of other authors (Orr and Holmes 1994), and is also evident in the lack of any strong relationship between grazing intensity and patterning in floristics across a very large number of mitchell grassland sites in Queensland (Fensham *et al.* 2000). However, deterioration of mitchell grass grasslands (principally loss of the perennial *Astrebla* species, with replacement by less palatable annual grasses) is reported under prolonged heavy stocking (Orr 1980; Bowman *et al.* 1996).

In a review of pasture condition across northern Australia, Tohill and Gillies (1992) noted that only a very small proportion of mitchell grasslands was in the poorest (“degraded”) condition, whereas 70-90% of mitchell grasslands in the Northern Territory and north Queensland, 45-60% in Western Australia, and 25% of rolling downs mitchell grasslands in southern Queensland was in the best category (“sustainable”) condition. In contrast, Bowman *et al.* (1996) reported that “*the Mitchell grass ecosystem of north-western New South Wales is fragile, and therefore prone to deterioration on the more alkaline, saline, flood prone and waterlogged clay soils ... the decline of native pasture in these situations is a complex phenomenon, in which loss of perennial vegetation is linked to grazing pressure and soil processes, leading to possible increase in soil salinity.*”

The woody weed *Acacia nilotica* has invaded around 66,500 km<sup>2</sup> mainly of mitchell grasslands in Queensland (Mackey 1997), but the consequences for biodiversity of this infestation are unclear.

Extensive areas of mitchell grasslands at the southern edge of its range have been cleared for cultivation (Rhodes 1981; Bellotti 1987).

Floodplain grasslands in northern Australia are generally more likely to be reserved (e.g., around 15% of their extent are in reserves in the Northern Territory: Woinarski *et al.* 1996) than the semi-arid mitchell grasslands, possibly because the floodplains have far more conspicuous conservation values, and because they present some formidable pastoral problems (Whitehead *et al.* 1990). Nonetheless, there are major management problems for this environment within conservation reserves, and major threats to biodiversity in unreserved floodplain grasslands. The floodplain grasslands are subject to invasion by a range of weed species, notably including the woody *Mimosa pigra* which now forms almost a monoculture across 800 km<sup>2</sup> of floodplain grasslands in the Northern Territory (Lonsdale 1993; Cook *et al.* 1996). There are also deliberate introductions of a range of aggressive exotic plant species to pastoral areas (principally para grass *Brachiaria mutica*), with subsequent spread to conservation reserves, to the detriment of a range of native plant and animal species (Whitehead 1999; Whitehead and Dawson 2000). Especially in northeastern Queensland, there is also deliberate major change to the floodplain environments, through manipulation of hydrology and landform and the introduction of exotic plants (principally *Brachiaria mutica*, *Echinochloa polystachya* and *Hymenachne amplexicaulis*), to form ponded pastures (Clarkson 1995; Neldner *et al.* 1997).

The tropical floodplains are also threatened by saltwater intrusion, which is likely to be exacerbated by climate change and rises in sea level (Whitehead 1999). High densities of farmed and feral water buffalo on floodplain grasslands have contributed to saltwater intrusion and its impacts (Whitehead 1999).

Burning regimes on floodplain and other tropical lowland grasslands have undoubtedly changed since cessation of traditional Aboriginal land management (Russell-Smith *et al.* 1997b; Crowley and Garnett 2000). One result appears to be the invasion of native woody species (principally *Melaleuca* spp.), changing the floristics and structure of these environments over very large areas, at the expense of a range of grassland-favouring plant and animal species (Stanton 1992; Garnett and Crowley 1997; Crowley and Garnett 1998).

There is less information on the status of native grasslands in the rangelands of southern Australia. Benson *et al.* (1997) provided an overview of the conservation status of native grasslands (largely secondarily derived from chenopod shrublands and open *Acacia* woodlands) in the Riverina area on the rangeland fringe. They noted that palatable species such as kangaroo grass *Themeda australis* and the native yam *Microseris lanceolata* had been severely depleted by grazing, and are now rare in the region. Sites which have best retained native plant species are mainly in travelling stock routes, roadsides and some privately-owned paddocks which have been only lightly grazed. They considered that main threats to these grasslands were increased extent of cultivation, rising soil salinity, pasture improvement, inappropriate grazing, and small-scale urban development. These threats were also largely identified as those affecting native fauna in these grasslands, and in particular the vulnerable plains-wanderer *Pedionomus torquatus* (Baker-Gabb 1993).

## Hummock grasslands

Hummock grasslands (dominated by *Triodia* spp.) extend over 2,010,000 km<sup>2</sup> of continental Australia, comprising 37% of the rangelands, mostly in arid and semi-arid areas (Groves and Williams 1981; Griffin 1984b; Allan and Southgate in press). Hummock grasses also make up the understorey of a range of other habitat types (including woodlands dominated by eucalypt, *Acacia*, *Allocasuarina* and *Callitris* spp.) and occur in rocky ranges in higher rainfall rangeland areas.

Although they are generally not prime pastoral lands (Griffin 1984b), the reservation status of hummock grasslands is highly variable. Specht *et al.* (1995) classified 10 hummock grassland communities across the rangelands: of these, four were completely unreserved and two were poorly reserved. In the Northern Territory rangelands, only 0.9% of their extent is reserved (Woinarski *et al.* 1996). The tenure of most hummock grasslands is Aboriginal freehold lands or “unproductive” vacant crown land (or jurisdiction-specific variations on these themes) (Griffin 1984b).

Physical characteristics of the hummock grasses underwrite an extraordinary diversity of reptiles and, to a lesser extent, small mammals in these environments (Pianka 1969; Cogger and Cameron 1984; Morton and James 1988), with correspondingly high diversity also of some invertebrate groups (Watson 1982).

The hummock grasslands are affected by grazing from livestock and feral animals (principally rabbits and camels, but also goats, donkeys, horses and cattle). The hummock grasses themselves have generally been relatively resilient to such grazing (and certainly more so than the dominant plant species from most other arid and semi-arid environments; Scattini *et al.* 1988), but some plant species associated with the environment have been far less robust, and have suffered major changes in population structure, and declines in range and abundance (Craig 1999).

The biota of hummock grasslands has suffered substantial impacts from feral cats and foxes, with loss of many mammal and some bird species across vast areas of this environment (Burbidge *et al.* 1988; Burbidge and McKenzie 1989; Morton 1990; Reid and Fleming 1992).

However, the major land management factor affecting hummock grasslands is fire. Much of the hummock grassland biota shows a strong variation with time since fire, and can be locally and regionally eliminated with inappropriate fire regimes (Suijendorp 1981; Reid *et al.* 1993; Masters 1993, 1996; Latz 1995; Allan and Southgate in press). As the relationships vary among different species, the maintenance of the full complement of the hummock grassland biota depends upon the maintenance of a range of ages since fire, within a fine-scale mosaic (Bolton and Latz 1978; Griffin 1984a; Burbidge 1985; Reid *et al.* 1993).

In many rangeland areas, current fire regimes are favouring the expansion of hummock grasslands, at the expense of more fire-sensitive vegetation types, such as mulga woodlands (Bowman *et al.* 1994; Latz 1995; Allan and Southgate in press) and heathlands (Russell-Smith *et al.* 1998). Interactions between fire and grazing may further complicate impacts, with some hummock grassland communities being apparently resilient to either factor singly, but readily degraded by combination of impacts (Craig 1999).

## Mulga and other *Acacia* woodlands

Woodlands and shrublands dominated by mulga (*Acacia aneura*) extend over about 1,500,000 km<sup>2</sup>, or about 27% of the rangelands (Johnson and Burrows 1981; Neldner 1986), mostly in areas with 200 to 500 mm annual rainfall other than semi-arid areas with a regular summer or winter drought (Nix and Austin 1973). Across different climatic and edaphic profiles, these are complemented by functionally similar woodlands and shrublands dominated by other *Acacia* species, most notably (Beadle 1981; Johnson and Burrows 1981; Hnatiuk *et al.* 1982; Hnatiuk and Maslin 1988):

- **lancewood** *Acacia shirleyi* (occupying around 60,000 km<sup>2</sup>, mostly in areas with 500-750 mm annual rainfall, in the northern and northeastern rangelands: Woinarski and Fisher 1995);
- **bendee** *A. catenulata* (on deeper and loamier soils than *A. shirleyi*, in central Queensland);
- **brigalow** *A. harpophylla* (formerly occupying around 60,000 km<sup>2</sup>, on clay soils within 500-750 mm annual rainfall areas on the rangeland fringe of central southern Queensland, but now much diminished by clearing);
- **gidgee** *A. cambagei* and *A. georginae* (on light clays and loams of northeastern Australia);
- **western myall** *A. sowdenii* (dominating or co-dominating around 80,000 km<sup>2</sup>, in the arid rangelands around the Nullabor Plain: Lange and Purdie 1976);
- **myall** *A. pendula* (on clays in areas of the southeastern rangelands with annual rainfall of 375-550 mm); and
- **pindan** *A. eriopoda*, *A. tumida* and *A. platycarpa* (on sandy soils in the Dampierland bioregion of northwestern Australia)

but with many other *Acacia* species dominant over more localised areas, or subdominant over more extensive areas.

The conservation status of these communities is highly variable, but generally poor (Purdie *et al.* 1986). In the Northern Territory, only 0.6% of the extent of *Acacia* woodlands and shrublands is included within conservation reserves (Woinarski *et al.* 1996). Of 104 *Acacia* dominated communities described across the rangelands by Specht *et al.* (1995), 38 are unreserved, 8 others are “very poorly” reserved, and 14 others are “poorly” represented in reserves.

The main threatening processes vary somewhat among the different *Acacia* communities, but generally include inappropriate fire regimes, grazing by livestock and feral animals, and clearing. While careful use of fire may maintain diversity within most rangeland *Acacia* communities (Fox 1985; Griffin and Hodgkinson 1986), many *Acacia* communities are susceptible to the impact of frequent and/or intense fires, with reduction in range, or degradation, reported for at least mulga (Griffin *et al.* 1983; Griffin and Friedel 1984; Morrissey 1984; Fox 1985, 1986; Griffin and Hodgkinson 1986; Start 1986;

Reid *et al.* 1993; Bowman *et al.* 1994; Latz 1995) and lancewood (Woinarski and Fisher 1995) communities.

Grazing by livestock and feral animals (especially rabbits: Robertshaw 1995) has led to substantial changes in the plant species composition in many rangeland *Acacia* shrublands and woodlands, to degradation of soils, and to lack of recruitment of the dominant *Acacia* and other overstorey species (Lange and Purdie 1976; Foran 1984; Morrissey 1984; Mills 1986; Beale *et al.* 1986; Passmore and Brown 1992; Lunney 1994; Tongway and Ludwig 1995; Wilson 1997; Witt *et al.* 2000). In an assessment of the condition of mulga lands in southwestern Queensland, Mills *et al.* (1989) showed that more than two-thirds of properties west of the Warrego River showed signs of serious land degradation (as indicated by soil erosion, increase in cover of unpalatable shrubs, and decrease in cover of palatable perennial grasses). A relatively high proportion of plants and animals in mulga communities shows a marked relationship with grazing intensity, as demonstrated by recent piosphere studies (Landsberg *et al.* 1997). Mulga regeneration can be completely blocked where exposed to rabbits, with less severe impact from sheep, and the cattle grazing (Williams *et al.* 1995).

In some semi-arid rangelands and the rangeland fringes, extensive areas of *Acacia* communities have been cleared, for intensive horticulture, for improved pasture for grazing, or for fodder during drought (Webb 1984; Sattler and Webster 1984; Silcock 1986; Pressey 1990; Wilson 1997; Fensham *et al.* 1998). For 11 brigalow and gidgee communities in central Queensland, only 9-38% of the original (pre-European) extent now remains (Fensham *et al.* 1998).

Many *Acacia* communities have been degraded by weed invasion. Pastoral managers have planted buffel grass *Cenchrus ciliaris* widely in mulga and other *Acacia* communities, to the detriment of native plant and animal species (Silcock 1986; Wilson 1997; Craig 1999). There has been an increase in “woody weeds” in many *Acacia* communities over the last century, in response to pastoralism and fire exclusion (Anderson 1984; Morrissey 1984; Holmes 1986; Booth 1986; Harrington 1986).

Some *Acacia* communities have been affected by timber extraction, particularly for fence-posts (especially in the pioneering pastoral phases). In some regions, this use has been so intensive that vegetation structure has changed over large areas and with long-lasting impact (Lange and Purdie 1976; Denny 1983).

In response to these environmental changes, and to the spread of feral predators, the fauna of *Acacia* woodlands and shrublands has suffered substantial losses, with mammals particularly affected (Denny 1983; Gordon 1984; Burbidge *et al.* 1988), but with losses also being reported for birds (Reid and Fleming 1992) and reptiles (Covacevich *et al.* 1996, 1998). Concomitant with the destruction of much of the brigalow woodlands, some wildlife species are now restricted to a minute proportion of their former range (Gordon and Lawrie 1980; Covacevich *et al.* 1998).

## Tropical and subtropical eucalypt woodlands

Tropical and subtropical eucalypt woodlands extend over 960,000 km<sup>2</sup> of the rangelands of northern Australia (or about 15% of the Australian rangelands area) (Young *et al.* 1984). Although some tree (e.g. *Eucalyptus miniata* and *E. tetradonta* in the higher rainfall north; *E. melanophloia*, *E. populnea* and *E. crebra* in northeastern Australia) and grass species (e.g. *Sorghum* spp, *Heteropogon contortus*) are dominant over large areas (Mott and Tothill 1984), there is also marked variation in floristics and structure in response to regional climate and soils (e.g. Williams *et al.* 1996).

The eucalypt woodlands are generally relatively well reserved (e.g. in the Northern Territory, Woinarski *et al.* (1996) recorded the three major variants of these woodlands and open forests as having 2.2%, 6.6% and 11% of their area reserved). Of 51 floristic communities of these eucalypt-dominated tropical and subtropical woodlands recognised by Specht *et al.* (1995), only two are unreserved, six are “very poorly” reserved and an additional two are “poorly” reserved.

Biodiversity in these woodlands is affected mainly by fire regimes, pastoralism, weed invasion, feral animals and clearing. Relative to most other rangeland ecosystems, there has been relatively little loss of biodiversity (as measured by regional extinctions) in the tropical and subtropical eucalypt woodlands (Woinarski and Braithwaite 1990), but this “healthy” state appears to be now decaying (Kitchener 1978; McKenzie 1981; Johnson and Southgate 1990; Johnson and Kerle 1991; Franklin 1999; Woinarski *in press*).

Russell-Smith (*in press*) and Williams *et al.* (*in press*) provide the most recent of many reviews of the occurrence and impact of fire regimes in the tropical and subtropical eucalypt woodlands. Both note that there has been a major divergence from pre-European fire regimes, generally with a now greatly reduced incidence of fire in pastoral lands (especially those in the lower rainfall areas), but a generally higher incidence of hot extensive late dry season fires in the higher rainfall and non-pastoral areas.

In response to fire exclusion (or low incidence) on pastoral lands (a regime imposed either deliberately or as an inevitable consequence of the reduction in fuel through grazing), there has been a marked increase in cover of a range of shrubs and trees over extensive areas (Dyer *et al.* 1997; Burrows 1998), and some changes in grass species composition (e.g., Stocker and Stutz 1966; Mott and Andrew 1985; Crowley and Garnett 1998). In turn, these vegetation changes have advantaged and disadvantaged different components of the fauna (Woinarski *et al.* 1998b), and as relatively uniform fire management is practised over very large areas of pastoral lands (Russell-Smith *in press*), the resulting changes in habitat suitability for these components of the fauna have been played out over very extensive scales. Critical fire-caused environmental changes are not necessarily conspicuous, as even subtle changes in grass species composition may affect the continuity in seed resources needed for a distinctive component of the savanna woodland biota, granivorous birds (Crowley and Garnett 1999). Reluctance to use fire in these lands has also limited capacity to control some invasive weed species (Noble 1997).

In the taller open forests of higher rainfall areas, fire regimes affect particularly the structural composition of the vegetation, with tall shrubby understoreys developing in fire excluded areas, but more simple grassy understoreys dominating frequently burnt areas, with floristic changes generally less evident (Bowman *et al.* 1988b; Fensham 1990; Bowman and Panton 1995; Williams *et al.* 1999). Extensive late dry season wildfires are generally regarded as relatively destructive, and anticipatory early dry season burning is widely used on many conservation reserves and Aboriginal lands as a mechanism for minimising the incidence and impact of these (Williams *et al.* in press). However, a relatively recent apparent extensive spread of annual spear-grass *Sorghum* (which contributes a very high fuel load) may have reduced the preventative capacity of early dry season burns, and also increased their intensity and impact (Russell-Smith *in press*). This problem is likely to be even more magnified with ongoing invasion of tropical open forests by exotic grasses, such as mission grass *Pennisetum polystachion* and gamba grass *Andropogon gayanus*, which contribute even far higher fuel loads. Responses of fauna to the range of fire regimes are less clearcut. At least several mammal species (such as the brushtail possum *Trichosurus vulpecula* and black-footed tree-rat *Mesembriomys gouldii*) occur only or preferentially in areas which have had little fire impact (because such areas have a higher incidence of hollow logs, large hollow-bearing trees and fleshy-fruit in a shrubby understorey), and the diminution of such areas is likely to have contributed substantially to their regional decline (Kerle 1985, 1998; Friend 1987). At least several bird species appear to track fire across the landscape, moving locally or regionally to forage in recently-burnt areas (Woinarski 1990). Many invertebrate and vertebrate species have been shown to respond distinctly and idiosyncratically to different fire regimes (Woinarski 1990; Andersen 1991b; Griffiths and Christian 1996; Williams *et al.* in press), such that the maintenance of a full complement of the tropical open forest fauna within a property or region may be dependent upon the long-term maintenance of a range of fire histories, maintained in an intricate mosaic (Woinarski 1999). Fire dictates not only the environmental characteristics *within* these tall open forests, but also their boundaries and extent relative to adjacent vegetation types, in some areas leading to expansion into rainforest patches with high fire frequency, and vice-versa with long-term fire exclusion (McKenzie and Belbin 1991; Woinarski *et al.* 2000; Russell-Smith *in press*).

Pastoralism has affected (and is continuing to affect) biodiversity in tropical and sub-tropical woodlands through ongoing environmental change, often relating to, or compounding, the impacts of changed fire regimes. In these environments, grazing typically affects the species composition and structure of the understorey, with the magnitude of the effect determined largely by stocking rate and seasonality/climate (Winter 1990; Gardener *et al.* 1990). Some palatable grass and herb species of the tropical and subtropical woodlands have declined extensively across pastoral lands (Fensham and Skull 1999). In turn, these vegetation changes influence the abundance of some fauna, particularly those dependent upon grass seeds, litter layer, and dense understoreys (Franklin 1999). For example, the quadrat-scale richness of reptiles and ants in an ungrazed eucalypt woodland in the Einasleigh Uplands bioregion was about 300% and 170% higher respectively than in adjacent grazed lands, and the species

composition was distinctly different (Woinarski and Ash *ms*; Woinarski et al. *ms*). The impacts of pastoralism in tropical and sub-tropical woodlands have been compounded by feral animals, and in particular pigs, donkeys, horses, cattle and water buffalo (Freeland 1990).

Weeds such as rubber vine *Cryptostegia grandiflora*, para grass, mission grass and buffel grass have invaded large areas of tropical and subtropical eucalypt woodlands (Lonsdale 1994). While a few studies have considered the impacts of some of these weed species upon aspects of biodiversity (Fairfax and Fensham 2000), there has been no more comprehensive assessment.

Vegetation clearance, for horticulture, pastoral improvement and timber plantation, has affected tropical eucalypt open forest, particularly in the Daly Basin, where 1500 km<sup>2</sup> has been cleared (Price *et al* in press) mostly for failed agricultural schemes (Woinarski and Dawson *in press*) and Victoria-Bonaparte bioregions. Vegetation clearance has been far more extensive in the subtropical eucalypt woodlands of central Queensland, where around 1,000,000 ha was cleared over the period 1990-95 (Barson *et al.* 1999), with this rate of clearing persisting or increasing over the last few years.

## Semi-arid eucalypt woodlands

Semi-arid eucalypt woodlands form a miscellaneous collection of vegetation types, including the rangeland components of mallee vegetation (along the southern fringes of the rangelands, particularly in the Great Victoria Desert, Coolgardie and Murray-Darling Depression bioregions), and taller more open woodlands, mostly extending southwards from the subtropical woodlands of inland southern central Queensland, but also across parts of Western Australia, the Northern Territory and South Australia (Gillison and Walker 1981; Harrington *et al.* 1984; Noble 1984). Together, these occupy an area of around 650,000 km<sup>2</sup>, or around 12% of the rangelands.

The reservation status of the semi-arid eucalypt woodlands is highly variable. Of 48 floristic communities recognised nationally by Specht *et al.* (1995), 11 were classified as unreserved and a further five as “poorly” represented in reserves.

The semi-arid woodlands have generally suffered more degradation than tropical and sub-tropical woodlands, with substantial losses of mammal and bird species, extensive clearing, and major impacts from clearance, pastoralism, feral animals, weeds and changed fire regimes, with these factors often operating synergistically (Noble 1997).

Most of the mallee vegetation lies outside the rangeland area as used in this paper (Noble 1984), but major reviews of the management problems and conservation status of mallee environments (Noble and Bradstock 1989; Noble *et al.* 1990; Stephens 1992) are generally applicable to those mallee areas within the rangeland sector. Based on compilation of expert opinion across a range of threatened species, Stephens (1992) considered the major threats to biodiversity in mallee environments to be clearing, fragmentation, inappropriate fire regimes, grazing and trampling (stock, rabbits), dieback, weeds, altered hydrological regimes, predators, competitors (e.g. herbivores, honey bees), collecting and hunting, illegal take, vehicle use, and genetic deterioration. The loss of biodiversity associated with these processes has been very substantial: for example, Bennett *et al.* (1989) noted that, of 45 native mammal species (other than bats) present before European contact in the mallee of southeastern Australia, 20 were now regionally extinct.

Other semi-arid eucalypt woodlands are more disparate, and their management and conservation status has not been comprehensively reviewed. Threatening processes are generally similar to those of mallee, and especially include widespread clearing, fragmentation, pastoralism and changed fire regimes, and feral animals (especially rabbit and goat). Especially along the eastern fringe of the rangelands, large areas of semi-arid woodlands have been cleared, and across very large areas the structure and floristics of uncleared areas has undergone a dramatic change since European settlement (Harrington *et al.* 1984; Noble 1997). The extent of loss of mammal species has been comparable to that in mallee (Dickman 1994), and there has been a major decline affecting many bird species (Smith and Smith 1994; Reid 1999).

## Rainforests

Rainforests constitute a minute proportion of the rangelands, mostly occurring in the higher rainfall areas of the Kimberley (where they comprise a total of 68km<sup>2</sup>, or less than 0.1% of the land surface), the Top End of the Northern Territory (where they comprise about 2,700 km<sup>2</sup>, or 0.4% of the land surface), and Cape York Peninsula (where they comprise about 2,600 km<sup>2</sup>, or 2.3% of the land surface) (Bell *et al.* 1987; Russell-Smith *et al.* 1992). Even more scattered rainforest patches occur in the Einasleigh Uplands and Brigalow Belt North (Kahn and Lawrie 1987; Gasteen 1987; Horsup *et al.* 1993; Fensham 1995), and outlying components of the rainforest biota also occur in fire-protected refugia in other rangeland bioregions (Morton *et al.* 1995).

Rainforests have a significance for rangeland biodiversity far beyond this meagre extent, largely because their biota is so distinct from that of other rangeland environments, but also because they typically have relatively high local-scale richness and high turnover of species between different patches (Kikkawa *et al.* 1981; Russell-Smith 1991; McKenzie *et al.* 1991; Price *et al.* 1995).

Compared with other rangeland environments, rainforests are relatively well represented in the current conservation reserve system (e.g. 26% of their extent in the Northern Territory is within conservation reserves: Woinarski *et al.* 1996). However, this does not provide sufficient conservation security for their biota because (i) the high turnover of species between patches and the highly restricted range of many species means that not all rainforest species will be protected even with this high proportion of reservation (McKenzie and Belbin 1991; Price *et al.* 1995), and (ii) many of the threats facing rainforests operate at a landscape-wide scale, almost impervious to land tenures (Russell-Smith and Bowman 1992). The potency of these threats is particularly heightened by the typically small size of rainforest patches, with most less than 20 ha (Bowman and Woinarski 1994; Price *et al.* 1995).

The main threats facing rainforests are:

- inappropriate fire regimes (typically too frequent, too late in the year and too extensive);
- feral animals, especially pigs, but also cattle, water buffalo and some invertebrates (e.g. the exotic ant *Pheidole megacephala*: Hoffman 1998; Hoffman *et al.* 1999); and
- weed infestation,

with all three frequently compounded to have greater cumulative impacts (McKenzie and Belbin 1991; Russell-Smith and Bowman 1992; Fensham 1993, 1996). Other processes (climate change, impacts of water use in nearby areas, and mining) also provide less immediate or more localised threats.

In some, typically pastoral, areas, fire regimes have changed to be probably less frequent than those operating before European settlement (Stanton 1992; Crowley and Garnett

2000), and in these areas some rainforests have expanded into tropical savannas and open forests (Stanton 1992; Harrington and Sanderson 1994).

## Wetlands and riparian areas

Naturally occurring wetlands and water sources comprise a very small proportion of the rangelands, but are pivotal in rangeland ecology and exploitation. The distribution of many rangeland plant and animal species is concentrated around natural water sources, and many rangeland species are restricted entirely or mainly to these environments (Greenslade *et al.* 1985; Ponder 1986; Reid and Gillen 1988; Zeidler and Ponder 1989; Bunn *et al.* 1993; Rowley 1993; Williams 1994; Woinarski *et al.* in press). Pastoral activity, tourism, town development, horticulture and other human use is also concentrated around natural water sources. This conjunction of restricted area, ecological significance and relatively high levels of use, has meant that these environments have suffered probably the most profound deterioration in condition and loss of biodiversity of any rangeland environment. This damage may be especially pronounced when this concentration of both ecological significance and use of wetland areas is accentuated during periods of drought (Barnard 1917, 1925; Morton 1990).

Before the widespread use of artificial water sources for rangeland pastoralism, the riparian areas and wetlands bore the brunt of pastoral impacts, with much of the vegetation in water edges destroyed within a few decades of pastoral settlement (Barnard 1925; Bauer 1959; Riddett 1990). In many rangeland areas, this has never recovered. In rangeland areas where pastoral and other activity continues to use natural water sources, the degradation has continued. For example, Rowley (1993) described the condition of riparian lands along the Fitzroy River system: “*the riverside vegetation of the lower Fitzroy has suffered extreme degradation. The tall “cane-grass” ... has almost all been grazed to the ground.*” In turn, this loss of vegetation has substantially affected animal biodiversity, in this case explicitly causing at least local extinction of the riparian purple-crowned fairy-wren *Malurus coronatus*.

Not only has native vegetation been lost from many riparian and wetland areas in the rangelands, but in many places heavy utilisation has promoted the spread of exotic weed species (notably including rubber vine *Cryptostegia grandifolia* in northern Queensland, athel pine *Tamarix aphylla* in central Australia, parkinsonia *Parkinsonia aculeata* across much of the northern rangelands, prickly mimosa *Acacia nilotica* in drainage lines on mitchell grasslands, and mimosa *Mimosa pigra* in northern floodplains), such that these now comprise a large proportion of the vegetative cover or plant species richness in wetland and riparian areas (Boekel 1980a,b; Griffin *et al.* 1989; Humphries *et al.* 1991; Grice 1996; Abbott *et al.* 1999; Woinarski *et al.* in press). More deliberately, in some rangeland areas, exotic plant species have been used to create ponded pastures in wetland areas, and these exotic species (principally *Brachiaria mutica*, *Echinochloa polystachya* and *Hymenachne amplexicaulis*) are invading beyond the artificial wetlands to dominate natural wetlands (Wilson *et al.* 1990; Clarkson 1995; Neldner *et al.* 1997; Whitehead 1999; Whitehead and Dawson 2000).

Wetland and riparian vegetation has been affected not only by trampling and grazing by stock, but also by changes in water quality and regimes (e.g., in water flow and flooding

periodicity and extent) caused by impoundment, allocation of waters for irrigation and other uses, altered run-off, infiltration and sediment loads due to changes in vegetation cover across the landscape, pollution, and impacts of use of groundwater for artificial water sources. These hydrological changes have had considerable impacts on rangeland riparian and swampland vegetation (Bren 1988, 1991, 1992; Close 1990; Craig *et al.* 1991; Bacon *et al.* 1993; Jolly *et al.* 1993; Walker and Thoms 1993; Walker *et al.* 1994; Casanova 1999; Kingsford 2000) and aquatic biota (Briggs *et al.* 1994; Lake 1995; Gehrke *et al.* 1995, 1999; Kingsford and Thomas 1995; Kingsford 1999a,b; Kingsford and Johnson 1999; Puckridge 1999).

Mound springs are probably the most susceptible rangeland wetland system. These highly localised features are associated with seepage of artesian waters. They are an important focus for biodiversity in arid and semi-arid regions of south-western Queensland and north-eastern South Australia, notable particularly for narrowly endemic plant (e.g. *Eriocaulon carsonii*, *Eryngium fontanum*, *Sporobolus pamela*, two undescribed species of *Peplidium* and undescribed species of *Dimeria*, *Eragrostis* and *Fimbristylis*) and animal species (e.g. Ponder and Clark 1990), and for highly disjunct populations of some species more characteristic of higher rainfall areas (Greenslade *et al.* 1985; Ponder 1986; Reid and Gillen 1988; Zeidler and Ponder 1989; Cox and Barron 1998; Fensham *unpubl.*). Many of these springs have been severely degraded. Fensham's unpublished survey demonstrates that the habitat for the wetland biota has been eradicated around a high proportion of springs by excavation, damming or converting the spring into a concrete tank exceeding the pressure head of the spring such that the spring no longer flows. Another major threat to the springs of the great artesian basin is the diminution of flow following water extraction from the basin by bores. The current estimated rate of extraction is 569 GL/annum and this is following estimated savings of 85 GL/annum following an extensive program to cap and pipe bores (Cox and Barron 1998). Declines in extraction are not only a result of capping as declines in pressure resulting from excessive extraction have also been substantial with average flows per bore having decreased from about 1.3ML/day 1915 to 0.4ML/day in 1995 (Noble *et al.* 1998). Monitoring of flow rates is occurring for some springs in South Australia, but is urgently required for a cross section of springs throughout the basin. Cattle and sheep trampling and grazing is another destructive process in artesian spring wetlands (Greenslade *et al.* 1985; Ponder 1986; Reid and Gillen 1988; Zeidler and Ponder 1989). However, there is mounting evidence that total protection from disturbance can have deleterious effects. In South Australia, fencing has resulted in total dominance by *Phragmites australis* at the expense of other plants and animals (Niejalke 1998) and the same phenomenon has been observed in Queensland (Fensham pers. obs.). There is no doubt that many populations of plants from artesian wetlands have been eradicated by the gamut of threatening processes affecting these rare habitats. Only one of the endemic species (*Myriophyllum* sp.) has been observed colonising artificial wetlands associated with bores and bore drains.

Extraction of water from calcrete aquifers is also recognised as a serious threat to aquatic subterranean ecosystems associated with limestone areas, which have a highly distinctive,

specialised and localised stygofauna and troglobitic fauna (W. Humphreys, WA Museum, pers. comm.).

There are other examples where combinations of specialised wetland habitat and threatening processes may be leading to the endangerment of rangeland species. Pig and buffalo rooting and damage are threatening the populations of the palm *Ptychosperma bleeseri* which is restricted to spring-fed rainforest habitat in the Top End Coastal bioregion. An undescribed species of *Burmannia* is similarly imperilled by pig rooting in spring-fed rainforest on Bathurst Island (Fensham 1993). The endangered *Pandanus spiralis* var. *flammeus* has also been threatened by cattle in its spring habitat in the Edgar Range in the Kimberley region of Western Australia (Brown *et al.* 1998). Drainage, use and modification of wetlands is also a major threatening process for some threatened rangeland animal species, such as the Australian painted snipe *Rostratula (benghalensis) australis* (Lane and Rogers 2000).

Despite their ecological and economic significance, and concern about their condition, wetland and riparian areas are very poorly served by current pastoral monitoring schemes, partly because riparian areas are narrow and linear and springs are typically very small in area, and partly because most monitoring plots are deliberately positioned to sample moderate levels of grazing (and hence are placed away from water sources). Hence there is little systematic information across the rangeland extent about their current condition and trends in that condition.

## THREATENING PROCESSES

Rangeland biodiversity is affected by many factors, with the intensity of this impact varying between different regions and different components of biodiversity. As monitoring schemes may find it easier to report on the threatening processes rather than the responses of biodiversity *per se* (Saunders *et al.* 1998), it is important that linkages can be established between measures of the threat and responses of the biota, and that an assessment of the relative impacts of different threatening processes can be made and used to prioritise monitoring actions. In this section, we briefly review the main threats considered to affect rangeland biodiversity. While each potentially threatening process is considered separately, it is important to recognise that, at any site, typically a number of processes may be operating together, with compounded or complex interactive effects.

### Changed fire regimes

While fire clearly has a major role in structuring rangeland ecology, the derivation, designation and imposition of appropriate fire regimes has been a deeply vexed and controversial issue (Flannery 1994; Langton 1998). It is indisputable that Australian rangeland environments were shaped by the use of fire by Aboriginal land managers over tens of thousands of years, and that when this management was usurped, many of those rangeland environments were remoulded (Hallam 1985). Change from the traditional fire regimes will have benefitted some components of biodiversity and disadvantaged others. The extent of the resulting change in biodiversity is probably loosely correlated with the extent of change in the fire regime. Such a relationship may allow the extent of change in fire regime to provide some surrogate of likely changes to biodiversity (Saunders *et al.* 1998), especially where the current regime can be well described and mapped (Press 1988; Russell-Smith *et al.* 1997; Crowley and Garnett 2000). However, the utility of the index is substantially compromised if there is a problem in determining what the pre-European fire regimes were for many rangeland areas (Fensham 1997; Bowman 1998).

Very coarsely, most evidence supports the notion that Aboriginal management involved relatively frequent fine-scale patchwork burning (Bolton and Latz 1978; Kimber 1983; Griffin 1984; Braithwaite and Estbergs 1985; Burbidge 1985; Low 1986; Bowman and Panton 1993; Price and Bowman 1994), with fires deliberately lit for a broad range of reasons (Hallam 1985; Lewis 1985). The intricacy of the mosaic, and the return time between fire events, is likely to have varied across rangeland areas, in relation to underlying topographic complexity, human population density, abundance and type of food resources, climate and rainfall events pre- and post-fire (Hopkins 1985; Hallam 1985). This patterning has now been replaced across much of the rangelands by one of two regimes: either attempted fire exclusion (particularly in areas of greatest intensity of pastoral use, where livestock may reduce fuel loads so much that fires cannot be established anyway), or abandonment of any attempt to maintain a fire regime (particularly in remote and sparsely inhabited areas, where relatively long periods without

fire may be interrupted by occasional extensive hot fires, usually caused by lightning strike) (Lewis 1985; Dyer *et al.* 1997; Noble 1997; Bowman 1998). These regimes each have distinct ecological consequences.

Deliberate exclusion of fire has led to extensive changes in vegetation in the grazing lands of western New South Wales, western Queensland and in parts of northern Australia. The most marked manifestation of this change in fire regime is a major increase in the density of trees and woody shrubs (the “woody weed” problem), typically at the expense of grasslands and open woodlands with savanna understoreys (Rolls 1981; Wright 1981; Noble 1997). Less conspicuously, fire exclusion has probably also reduced local- and regional-scale environmental heterogeneity. The impacts upon fauna of this broad-scale change in vegetation structure and patchiness have not been comprehensively assessed, and may be difficult to disentangle from those due to other factors which typically coincide with fire exclusion (most notably intensive pastoralism). Nonetheless, such extensive and substantial changes in vegetation would clearly be detrimental to at least the fauna reliant on grassy understoreys and open woodlands. The limited available evidence suggests that birds associated with such habitats have suffered substantial declines across these rangeland areas (Franklin 1999).

Reversion to the “natural” fire regime (no or scarce ignitions by humans, but occasional extensive wildfire) has had major impacts on the environments of some remote rangeland areas (Griffin *et al.* 1983). In the Tanami Desert, this regime has led to loss of intricacy in the environmental mosaic, probably to the cost of the mala *Lagorchestes hirsutus*, now extinct on the mainland (Bolton and Latz 1978; Lunding-Jenkins 1993). Across much of the rangelands, it has led to decline in the extent of fire-sensitive vegetation. In central Australia, the most conspicuous response has been the contraction in range of mulga communities and the expansion of hummock grasslands (Fox 1985; Bowman *et al.* 1995).

Increasingly severe fire regimes (greater frequency of extensive hot fires) have been reported for parts of the wet-dry tropics of northern Australia, where frequent fire is almost inevitable. For current fire regimes, intervals between fires are now insufficient to maintain some plant species, particularly those which reproduce as obligate seeders (Bowman and Panton 1993; Price and Bowman 1994; Russell-Smith *et al.* 1998), animals associated with those plant species (e.g. the Leichhardt grasshopper *Petasida ephippigera*: Lowe 1995), or animals requiring large hollow trees, fallen hollow logs and/or a dense tall shrubby understorey (e.g. brushtail possum *Trichosurus vulpecula* and black-footed tree-rat *Mesembriomys gouldii*: Kerle 1985, 1998; Friend 1987). In these areas, current fire regimes are now also leading to the diminution and deterioration of patches of fire-sensitive environments, most notably monsoon rainforests (Clayton-Greene and Beard 1985; McKenzie and Belbin 1981; Russell-Smith and Bowman 1992).

Hence, across broad landscapes, the current fire regimes have led to changing juxtaposition and relative extent of contrasting vegetation types. They have also changed characteristics *within* most vegetation types, altering vegetation structure and floristic composition over very large areas.

In the open forests dominating the landscape of higher rainfall areas of northern Australia, several studies have examined the response to fire exclusion, in locations where there has been a massive effort spent on ensuring fires do not enter small selected areas. These studies have documented floristic and more marked structural changes (Bowman *et al.* 1988; Fensham 1990) but they are not of sufficient magnitude nor do they represent clearly desirable or readily achieved management options. However, impacts of fire will be far more pronounced following the proliferation and spread of improved pasture grasses, such as mission grass *Pennisetum polystachion* and gamba grass *Andropogon gayanus*, which provide far greater fuel loads, and which cure far later in the dry season (Panton 1993; Bowman 1999).

Because many plant and animal species have distinct associations with, or dependence upon, resources available only at particular time spans after fire, the full range of biodiversity potentially present within any environment will only be maintained when a broad range of fire histories are locally or regionally available. For example, for hummock grasslands, some *Triodia* species are obligate seeders while others are resprouters. There seems to be little consistent geographic pattern in these response categories, although the patterns of abundance of the obligate seeders can be profoundly affected by fire regime (Rice and Westoby 1999). Many ephemeral plant species are abundant only in the first few years after fire in spinifex (Bowman *et al.* 1990; Griffin 1991) and the longterm persistence of these species may well be dependent on at least occasional fire. On the other hand for the sandstone ranges and plateau in the monsoonal regions of the Northern Territory, there are a number of species that are clearly sensitive to very frequent fire regimes. Shrub species such as *Regelia punicea* are poorly dispersed obligate seeders and may take up to 10 years to become fertile. Clearly frequent fires will have a detrimental affect on the abundance of this species, and others with similar life histories (Russell-Smith *et al.* 1998).

In mallee vegetation in the southern rangelands, there is a well-established “succession” of bird species with increasing time since fire, from widespread opportunists (including nankeen kestrel *Falco cenchroides*, australian magpie *Gymnorhina tibicen*) less than 1 year post-fire, to species associated with low patchy heathlands (including chestnut quail-thrush *Cinclosoma castanotus*, white-fronted honeyeater *Phylidonyris albifrons*, shy heathwren *Hylacola cauta*) at 1-10 years post-fire, to species associated with taller, denser heaths and low woodlands (including red-lored whistler *Pachycephala rufogularis*, southern scrub-robin *Drymodes brunneopygia*, and crested bellbird *Oreoica gutturalis*) at 10-30 years post-fire, to species associated with taller woodlands, denser litter layers and hollows (including malleefowl *Leipoa ocellata*, white-browed babbler *Pomatostomus superciliosus*, black-eared miner *Manorina melanotis*, and regent parrot *Polytelis anthopeplus*) at more than 30 years post-fire (Woinarski 1999). Analogous sequences have been described for mammals and reptiles in hummock grasslands in the arid and semi-arid rangelands (Masters 1993, 1996; Reid *et al.* 1993), and are undoubtedly replicated by other groups of biota in other environments. A regional fire regime which results in the retention of only one of these stages will eliminate many species and homogenise the landscape (Woinarski and Recher 1997; Woinarski *et al.* 1999b).

But the retention of all types of regimes will not necessarily maintain all species, as connectivity between patches of particular ages may also need to be maintained, and the juxtaposition of patches of contrasting history may also be critical for some species which prefer old vegetation for breeding but younger regrowth for foraging, or which require access to patches of different ages in order to maintain resource availability across different seasons (Carpenter and Mathew 1986; Priddel 1989, 1990).

Fire regimes are more than simply about frequency or periodicity. Other components such as the extent of fire, the seasonal timing of fire, the amount of rainfall before and after fire, and the intensity of fire may be critical in determining their ecological effects (Gill *et al.* 1981; Gill 1999).

The maintenance of a particular fire regime is a critical requirement for the persistence of many rangeland plant and animal species. For example, Garnett (1992) listed inappropriate fire regime as a threatening process for 51 threatened birds in Australia, of which 35 are rangeland taxa.

## Pastoralism

There have been a series of reviews which consider the broad environmental impacts of pastoralism across the rangelands, or for large regions of it (e.g., Winter 1990; Wilson 1990; Morton *et al.* 1995; Landsberg *et al.* 1997, 1999b; James *et al.* 1999), based generally on the collation of a series of generally short-term local-scale studies of vegetation responses (e.g., Orr 1980; Scanlan *et al.* 1996; Ludwig *et al.* 1999), and some longer-term studies of extent of vegetation change at a property or regional scale (e.g., Crowley and Garnett 1998; Landsberg *et al.* 1998; Fensham and Holman 1998; Witt *et al.* 2000). Together, these reviews indicate that pastoralism has produced some extensive and major changes in the rangeland environments, but the extent of this change is highly dependent upon stocking rates, seasons, environment, geographic location, management regimes, and interactions with other threatening factors.

There has been a massive influx of herbivores with European management of rangelands. In the pre-European landscape, lack of reliable water limited the density of herbivores, and there were no ungulates, the largest herbivore being the red kangaroo. Megafauna in the form of giant marsupials did exist as recently as around 30,000 years ago, but their densities and hence impacts would probably have been limited by the sparsity of natural water points. Herds of sheep and cattle, feral exotic animals and the large macropods have all increased with the proliferation of artificial water sources (James *et al.* 1999). Landsberg and Gillieson (1996) demonstrated that outside the deserts, areas more than 10 km from permanent water are extremely rare. James *et al.* (1999) reviewed the impact of this landscape transformation including the findings of a large study conducted across a number of rangeland bioregions. The study examined the response of species along the grazing gradients with decreasing distance to water. Their broad conclusions were that:

- 1) grazing at moderate densities leads to higher within habitat species richness compared with grazing at low or high densities; and
- 2) heavy grazing results in a decline in the number of native species, a reduction in abundance of the remaining species and dominance by a few species.

State and transition models are founded in successional theory and are based on the premise that vegetation changes between multiple steady states as a result of combinations of management and climate. They have been developed for many of Australia's rangeland environments through discussions between researchers and land managers. These models have been published in a special issue Volume 28 (1994) of *Tropical Grasslands*. The models largely relate to shifts in the major lifeforms or dominant species. In general they are not verified by empirical data. Furthermore the discussion of these models in the relevant papers suggests that many of the causes of transitions between various states are not universally accepted. Despite these shortcomings, the state and transition models provide a useful framework and are certainly the most comprehensive assessments of the effects of grazing management in

Australia's rangelands. Some extremely broad generalisations are encompassed by state and transition models. For example in general, palatable perennial grasses are replaced by less palatable perennial grasses with grazing. As grazing intensifies the non-palatable perennial grasses are replaced by annual grasses. The combination of grazing and lack of fire can result in invasion by "woody weeds" (native or exotic shrubs). The existing state and transition models will no doubt be refined as further empirical evidence is developed, although the underlying premise of predictability and equilibrium is questionable. Non-equilibrium models of plant dynamics (based on the foundation of a highly variable climate for the Australian rangelands) are now widely used for understanding responses to pastoralism, and for the development of sustainable management practices.

A clear theme from quantified research is that the longer-term grazing impacts are partially obscured beneath the generally more important influence of climate history. This has been demonstrated in relation to calcareous shrubby grassland in central Australia (Foran *et al.* 1985; Foran 1986) and Mitchell grasslands of north-eastern Australia (Orr and Phelps 1994; Bowman 1997). However, many of the studies in rangelands have had an agronomic focus, recording changes in the relative abundance of dominant species, and often ignoring ephemeral herbs and other components of plant biodiversity. Some grazing sensitive species may have been largely eradicated with European management over large areas. This may well be the case in the derived native pastures of central western New South Wales that were studied by Leigh *et al.* (1989). Fensham and Skull (1999) had the opportunity to compare savanna woodland that had never been grazed by cattle with comparable areas currently managed with cattle. They record that one of the most abundant grasses in the ungrazed savanna *Capillipedium parviflorum* was completely absent from the grazed savanna. Within the grazed pasture group there was a broad spectrum of floristic composition. With the exception of eradicating *Capillipedium* which is relatively common in other landscape settings the results indicated that most native species were maintained by conservative stock grazing. However, declines in native plant species and dramatic changes in plant species composition clearly occur if grazing is heavy. This result is corroborated by McIvor (1998) working in similar savannas who concluded that where the dominants have been reduced to unpalatable species, "overgrazing reinforces their dominance and reduces diversity". Given that the majority of our rangelands have been assessed as having a degraded condition because of overgrazing (Tothill and Gillies 1992) then we can assume a massive impact on vegetation biodiversity.

There have been some studies that compare the feeding behaviour of macropods, domestic stock and other herbivores in rangeland environments. The evidence suggests that the diet of kangaroos tends to have a greater proportion of grasses than for sheep (Ellis *et al.* 1977) although some herb species may be preferentially selected, such as *Portulaca oleracea* by red kangaroos *Macropus rufus* (Griffiths and Barker 1966; Griffiths *et al.* 1974). Grey kangaroos *Macropus fuliginosus* and *M. giganteus* seem to have a penchant for soft grasses of low fibre, when these are available, relative to other native and domestic animals (Bell 1973; Taylor 1983). In semi-arid western New South Wales, as pasture deteriorates, sheep tend to rely most heavily on flat leaved chenopods whereas red kangaroos continue to rely most heavily on grass with supplementation from a variety of other species. Cattle also

favour grasses over other herbage and the evidence of Chippendale (1968), Low *et al.* (1973) and Ellis *et al.* (1992) suggests that the dietary preferences of bovines is relatively specialised. Rabbits also preferentially graze grass (Bhadresa 1977; Leigh *et al.* 1989). The most likely order of the major herbivores in rangelands from generalist to specialist is sheep, red kangaroo, grey kangaroo, rabbits, cattle. When feed becomes extremely scarce all animals become less selective and can be sustained by feed components that would be avoided during times of abundance (e.g. Bailey *et al.* 1971).

The intermediate position of the native grazer species in terms of their dietary preferences would suggest that, in qualitative terms, the grazing disturbance of native herbivores will not be vastly aberrant from the influence of domestic stock. However, in Mitchell grasslands, there are some features of kangaroo grazing that are likely to result in lower impacts than those due to domestic stock (Edwards 1989):

- 1) they have a broad soft foot resulting in less soil compaction and trampling of vegetation than sheep or cattle; and
- 2) they have a lesser tendency to congregate around watering points thus reducing the likelihood of degraded scalds with little or no plant cover.

In relatively high rainfall regions grazing can enhance plant species diversity because it precludes dominance by robust perennial grasses. In rangelands however, this mechanism may be less important because the climate is sufficiently sporadic to maintain intertussock space.

There is some evidence that sustained grazing by stock has led to the endangerment of some plant species. The following is based on the account of the native daisy *Trioncinia retroflexa* published by Fensham (1999a). This species was first collected by Ferdinand von Mueller at 'Peak Downs' (Clermont region), central Queensland in 1856. This was prior to European settlement, when Mueller, a member of the Gregory expedition, was nearing completion of an arduous trek from Victoria River in the Northern Territory to Brisbane. The visit was extremely brief and despite the exceptional capacities of Mueller as a field botanist its discovery suggests that the species must have been at least locally common. The species was not formally recorded again until 1935 when Stan Blake collected it at Blair Athol on "black soil". *Trioncinia* was listed as Presumed Extinct on the Queensland Herbarium rare plant database up until 1996 when it was relocated on a roadside near Clermont (Clermont population) presumably in close proximity to the 1935 collection. Extensive survey (Fensham 1999b) has revealed three populations containing about 1500 individuals. All populations are on roadsides and the species is clearly absent from the grazed paddocks. The roadside habitat of the known populations of *Trioncinia* suggests sensitivity to the continuous grazing regimes of paddocks. After the application of IUCN criteria it is proposed that *Trioncinia retroflexa* be listed as Endangered (criteria B, C).

Another species that could well have a similar story is the recently rediscovered *Acanthocladium dockeri*. It is also restricted now to roadsides, in rangeland areas of South Australia. Another species that is listed as Vulnerable on National lists is *Dichanthium queenslandicum*. It was one of the original dominants of the basalt grasslands in central Queensland. It extended as far south as the Darling Downs where it is now presumed to be extinct after considerable survey. It is confined to roadsides and conservatively managed paddocks where it can still form a dominant sward. We know less about species that may have become endangered because of grazing in the more arid rangelands. *Sclerolaena walkeri* is a potential candidate. It is known only from areas with less than 300 mm mean annual rainfall and the only recent collection held at the Queensland Herbarium germinated from a soil seed bank experiment on Currawinya National Park that had recently been destocked.

There is concern that the recruitment and regeneration of native trees and shrubs is impacted by grazing. For example Tiver and Andrew (1997) showed negative relationships between either regeneration or recruitment in relation to grazing in South Australian rangeland. Their study suggested that grazing by sheep seemed to be more deleterious for this plant recruitment than that by kangaroos, rabbits or goats.

Foran *et al.* (1985) note that negative impacts must exceed the rate of natural thinning to have a major impact on overstorey structure. One environment where this would seem to be the case is in the mulgalands of Queensland and New South Wales where there appears to be an almost total absence of regenerating *Eucalyptus intertexta* (A. Franks pers. comm.). There is an obvious interaction between grazing and lack of fire with regards tree regeneration in rangelands and these interactions are poorly understood. There is also an interaction between clearing and grazing, because small remnants become stock camps and are preferentially grazed (Ludwig *et al.* submitted).

The impacts of pastoralism upon native vegetation may reverberate through native animal communities, with the abundance of many vertebrate and invertebrate species shown to be strongly related to grazing intensity (Landsberg *et al.* 1997a; James *et al.* 1999). Even floristic changes which are minor and inconspicuous to us may be profound and fatal to some animal consumers. For example, granivorous birds need a continuous availability of grass seeds, typically maintained by switching from one grass species to the next as they produce seed serially: where one (even non-dominant) grass species is lost or greatly reduced in abundance because of grazing, this continuous chain of seed availability may be broken (Woinarski 1993a; Garnett and Crowley 1994), and such dependent species may decline to extinction over vast areas (Franklin 1999). Grazing may also alter food availability to native wildlife species not necessarily by change in floristic composition, but rather by changing the timing and amount of grass seed set, again with potentially major impacts upon granivorous species (Crowley and Garnett 1999). Grazing also clearly affects the total amount of herbage, causing changes to other herbivorous species such as grasshoppers (Ludwig *et al.* 1999).

Loss of vegetative cover also reduces the availability of shelter to species such as bandicoots and rodents, making them more susceptible to increased predation pressure,

and hence compounding the impacts of the spread of feral predators. Trampling by livestock may also cause the collapse of the shallow burrows used by some native mammal species and reduce the nesting success of ground-nesting birds, such as quail, bustards and flock bronzewing. Trampling may also break down the margins of rainforests, riverbanks and riparian areas.

Some management practices associated with pastoralism may also have major impacts upon native biodiversity. In most rangeland bioregions, pastoralism has been associated (deliberately or by default) with changes in fire regime, and environmental change associated with the now prevalent fire regimes have favoured some native wildlife species, but been detrimental to others, typically across very extensive areas (e.g. Noble 1997; Woinarski *et al.* 1999). The degradation of many natural water sources, but provision of many artificial water sources, has also benefitted some species (most obviously including galah and crested pigeon), but disadvantaged others (James *et al.* 1999). The widespread cutting of timber for fence posts and building material has altered vegetation structure and hence suitability for some native wildlife, in many pastoral areas. Barbed-wire fencing has resulted in the direct mortality of many native animals, although the numbers have never been adequately quantified. Soil erosion associated with over-grazing has had major local effects, and more extensive effects on water quality and hence suitability to aquatic organisms in many rangeland areas (Winter 1990).

## Exotic predators

There is some (inevitably speculative) evidence that the arrival and subsequent spread of the dingo *Canis lupus dingo* (around 4000 years ago) led to major changes in the abundance and distribution of some rangeland mammal and bird species, possibly including the extinction from the mainland of the thylacine *Thylacinus cynocephalus* and Tasmanian native-hen *Gallinula mortieri* (Archer 1984; Baird 1991a).

Far more recently, feral cats and foxes have spread across all and most (respectively) of the rangelands. The impact, distribution and management of feral cats has been reviewed by Potter (1991) and Dickman (1996), and of foxes by Saunders *et al.* (1995). While both species eat a broad range of invertebrate, reptile, mammal and bird prey (Catling 1988; Martin *et al.* 1996; Paltridge 1998; Molsher *et al.* 1999; Risbey *et al.* 1999), their main conservation impacts have been on medium-sized terrestrial mammals (especially bandicoots, small macropods, larger rodents and the numbat) (Johnson *et al.* 1989; Johnson and Southgate 1990; Friend 1990a,b; Short *et al.* 1992; Dickman 1993; Robinson *et al.* in press; Morris *in press*). There is strong evidence of their local impact on threatened mammals (Kinnear *et al.* 1988, 1998; Horsup and Evans 1992), leading to at least local extinctions (Kerle *et al.* 1992; Gibson *et al.* 1994). As both feral predators are now widely dispersed across the rangelands, it is highly plausible that these local extinctions build to regional and national extinctions for the most susceptible native fauna, as has been suggested by many authors (e.g. Finlayson 1961; Burbidge and McKenzie 1989; Friend 1990a,b; Southgate 1990; Dickman *et al.* 1993; Smith and Quin 1996; Short 1998; Copley 1999). As an example, the disappearance of the night parrot in northern South Australia in the 1880s, and around Alice Springs in the 1890s, was reported to have coincided with the arrival of large numbers of cats from New South Wales (Whitlock 1924), and that of the numbat *Myrmecobius fasciatus* in central Australia with the arrival of the fox (Friend 1990a).

The recovery of native fauna following exclusion or other control of feral cats and foxes provides strong evidence of their impact (Kinnear *et al.* 1988, 1998; Short *et al.* 1992, 1994; Christensen and Burrows 1994; Gibson *et al.* 1994). To some extent, this suggests that monitoring the abundance of feral cats and foxes may provide a measure of trends in the populations of at least some native fauna. However, this simple relationship may be complicated by the availability of alternative prey (e.g. rabbits: Sandell and Start 1999), compensatory responses by other predators to the control of one predator species (Newsome 1993; Christensen and Burrows 1994; Dickman 1996), and variation in diet among individuals of a particular predator species (for example, there are several incidences of single cats or foxes consuming a high proportion of prey animals in a colony: e.g., Spencer 1990; Hone 1999).

Introduced predators may have impacts on native biodiversity not only through predation, but also through spread of exotic diseases and parasites, most notably toxoplasmosis. Some studies have reported at least localised occurrences of such diseases (Attwood *et al.*

1975; Johnson *et al.* 1988), but there is no compelling evidence that transmission of this disease has had extensive major impacts, and other studies (e.g., Oakwood and Pritchard 1999) have reported no evidence of exotic disease accompanying decline in native mammals in the rangelands.

## Other feral animals

European settlement has coincided with the deliberate or inadvertent introduction of many exotic animals to the rangelands (Preece and van Oosterzee 1997). Many of these species have thrived and expanded their range well beyond the point of introduction or the area of major human modification of these landscapes. Many of these species now have a substantial impact on the rangeland environments and their native biota. In some cases, this deleterious impact on native biodiversity is matched by a negative impact on human livelihood, but, in other cases, exotic species which diminish native biodiversity may be neutral or even beneficial to (at least some) human residents of the rangelands. Such discordance of values may substantially affect the management resources available, and willingness, to control particular feral animals. For example:

- rabbits and cats are widely recognised as harmful to native biodiversity, but now have substantial values as food for Aboriginal residents of the central and western deserts to the extent that control mechanisms may not be supported (Rose 1995; Bomford and Caughley 1996).
- commercial harvesting of other feral animals (e.g., camels, pigs, goats) usually is based on the goal of maintaining industry sustainability through the retention of at least a base population, rather than on the goal of eradication.
- the inadvertent escape to the wild of some honey bee populations is generally regarded as an acceptable price to pay for the maintenance of a honey industry in parts of the rangelands.

For many feral animal species in the rangelands, there have been major reviews of distribution, abundance, management control mechanisms, and environmental impacts: for example, for rabbits (Williams *et al.* 1995), rodents (Caughley *et al.* 1998); goats (Parkes *et al.* 1996); pigs (Choquenot *et al.* 1996), horses (Dobbie *et al.* 1993), cane toads (Freeland 1984; Jacklyn 1992) and honey bees (Paton 1995). We do not aim to repeat the detail of these reviews here, but instead consider only some of the main impacts related to biodiversity.

The impacts of several feral herbivorous mammal species are partly cumulative, in that they may combine with those of livestock (and often increased abundance of macropods) to together greatly inflate total grazing pressure, placing even more stress on plants favoured by livestock, extending the range of plant species affected, and reducing fuel loads even further than any decrease due to livestock alone.

### Rabbit

Rabbits *Oryctolagus cuniculus* invaded the rangelands of south-eastern Australia around 1870, then spread remarkably rapidly to occupy almost all of the arid and semi-arid rangelands by around 1910 (Williams *et al.* 1995). Their extraordinary numbers were checked by the introduction and establishment of myxomatosis in the early 1950s.

Population increases subsequently have been checked (for many rangeland areas) by the spread of rabbit calicivirus disease in the late 1990s (Neave 1999).

Rabbits have had a devastating impact on Australian rangelands, causing marked changes in floristics, structure and age composition over very large areas for most vegetation communities other than the high rainfall monsoonal tropics. The impacts of rabbits have often been far more severe than those due to livestock, and may be even more exacerbated when grazing by rabbits and livestock together act synergistically. Anecdotal evidence cited in Williams *et al.* (1995) reports that many short-lived palatable plant species became locally extinct in the first few decades after rabbit invasion. Subsequently, longer-lived plants (including many *Acacia*, *Callitris*, *Allocasuarina* and other woody shrub and tree species) have failed to regenerate over large areas, because all seedlings or suckers have been consumed. When this period without recruitment extends beyond the lifespan of the plants, these species too will suffer local and more extensive extinctions. The lack of recruitment is often exacerbated as many rangeland plants reproduce episodically in response to good rainfall events, conditions which also favour build-up in rabbit numbers (Lay 1986). In drier times, rabbit impacts will expand to include foraging on plant species not normally favoured, and on older woody plants, which may be ringbarked and killed.

In turn, these environmental changes caused by rabbits have contributed to the decline of many rangeland animal species. The diets of many herbivorous native mammals (and some birds, reptiles and invertebrates) overlap with rabbits. Decline in the availability of these food resources because of rabbits is likely to have contributed to decline in the native animals, with effects most pronounced during drought periods when food was particularly limiting (e.g., Dawson and Ellis 1979; Foulkes and Kerle 1989; Ingleby and Westoby 1992).

Vegetation structural and floristic changes caused by rabbits affect many native wildlife species other than herbivorous competitors: for example, the principal habitat feature of the Eyrean grasswren *Amytornis goyderi* is canegrass *Zygochloa* spp., and this can be eliminated by rabbit grazing (Williams *et al.* 1995).

Rabbits may also aggressively displace native species, with some evidence that they evict native macropods and other small and medium-sized mammals from burrows and preferred feeding areas (Williams *et al.* 1995).

Rabbits also affect native wildlife indirectly, through supporting an inflated density of native and feral predators – although this effect may also be partly beneficial (e.g. because some predators may switch from native prey to rabbits). Where rabbit numbers are now reduced by control management, there may be an at least short-term compensatory impost upon native species from predators (Sandell and Start 1999). The control of rabbits has some biodiversity costs as well as benefits. Many native animals have been the unintended victims of rabbit-trapping, poisoning, and warren-ripping. Native predators (notably raptors) may decline with decreasing rabbit populations following the spread of RCD (Sandell and Start 1999).

## Horse and donkey

There are an estimated 300,000 to 600,000 feral horses *Equus caballus* in the Australian rangelands, by far the largest population in the world (Dobbie *et al.* 1993). Donkeys *E. asinus* are at least as abundant, with densities of greater than 10/km<sup>2</sup> reported over hundreds of square kilometres in the Northern Territory alone (Freeland and Choquenot 1990). Both species are extremely patchily distributed in the rangelands, with high numbers particularly in the Cape York Peninsula, Gulf Plains, Gulf Falls and Uplands, Sturt Plateau, Ord-Victoria Plains, MacDonnell Ranges, Burt Plains, Channel Country, Finke, Central Ranges, Central Kimberley, Dampier Land, Gascoyne, Pilbara, and Murchison bioregions (Dobbie *et al.* 1993; State of the Environment Advisory Council 1996).

Dobbie *et al.* (1993) summarise the environmental impacts of feral horses, but recognise that few studies have considered such impacts, and none in appreciable detail. They noted that feral horses may deplete and foul waterholes (which may be a problem especially in arid and semi-arid areas during drought); contribute to localised erosion; damage vegetation and alter floristic composition; disperse weed species; contribute to changed fire regimes (through decreasing fuel loads) and collapse wildlife burrows. As feral horses (and donkeys) may move further from waterholes than cattle, their impacts may be more extensive; and as feral animals may be less likely than livestock to be removed by managers during drought, their impact during bad times may be particularly significant. The impact of donkeys is generally similar, but may be more pronounced as they are more likely to occur in larger groups, and they can typically survive longer into drought or degradation than cattle or horses (Letts *et al.* 1979; Freeland and Choquenot 1990).

## Camel

The first major importation of camels *Camelus dromedarius* to Australia was in 1860, but major feral populations did not really establish in the rangelands until the 1920s, when the rapid development of rail and car transport led to the redundancy of camel transport, and the broad-scale release of camels (Wurst 1995). The current population of feral camels in the rangelands is estimated at around 200,000, and to be rapidly expanding (Wurst 1995). These occur mainly in the Channel Country, Simpson-Strzelecki Dunefields, Stony Plains, Tanami, Finke, Central Ranges, Great Sandy Desert, Pilbara, Great Victoria Desert, Gibson Desert, Little Sandy Desert and Nullabor bioregions (Short *et al.* 1988). The environmental impacts of camels have not been documented in detail, but are generally similar to those reported above for horses and donkeys, except that camels consume an unusually broad range of plant species and browse more from shrubs and trees (Doerges and Heucke 1989).

### Feral cattle and water buffalo

Feral cattle *Bos taurus* and *B. indicus* are widespread across the rangelands, other than the most settled areas of the southern, eastern and western rangeland margins, and the central and western deserts. In general the impacts of feral cattle are the same as those of managed cattle, except that they are more likely to drink from (and degrade) natural water sources, and the lack of management for them may mean that they are less likely to be regulated during drought (and hence may have more impacts then).

Feral water buffalo *Bubalus bubalis* are locally abundant in the higher rainfall areas of northern Australia, and especially so in the Pine Creek-Arnhem, Central Arnhem and Top End Coastal bioregions. Feral (and managed) water buffalo may have substantial environmental effects, due mostly to the impact of their pathways and wallowing on hydrological regimes in the topographically simple floodplain wetlands. This disturbance is a primary contributor to saltwater intrusion, which then may cause wholesale vegetation change (Letts *et al.* 1979; Whitehead *et al.* 1990; Whitehead 1990). Feral water buffalo have also been implicated in the spread of weeds, particularly *Mimosa pigra*, and mortality of rainforest and paperbark trees (through rubbing), and their pugging and wallowing has been associated with local-scale increases and decreases in a range of frog, reptile, mammal and bird species (Braithwaite *et al.* 1984; Friend and Taylor 1984).

### Goat

Goats *Capra hircus* were introduced to the rangelands in the 1840s, and have since spread widely in the semi-arid rangelands of eastern and western Australia (particularly the Carnarvon, Gascoyne, Murchison, Geraldton Sandplains, Stony Plains, Flinders and Olary Ranges, Gawler, Murray-Darling Depression, Broken Hill Complex, Cobar Penepplain and Mulga Lands bioregions). The total Australian population was estimated in 1993 as 2,600,000, of which most were in the rangelands (Parkes *et al.* 1996). As with other herbivorous mammals, goats may change the plant species composition, and decrease the abundance of favoured species. Vegetation change and damage due to goats may be locally intensive, because of their sociability (particularly around camp sites), and different to that due to other herbivores (because of a more generalised diet and their agility). Their ability to use steep rocky areas may introduce grazing pressure into some environments not grazed by other animals, and particularly to “refuge” areas which may contain plant species with unusually localised ranges (Davies 1990; Morton *et al.* 1995). As with other large feral herbivores, goats may also deplete and foul waterholes and contribute to localised erosion. Goats may also compete with native wildlife for shelter (e.g. caves in rocky areas), and have a substantial overlap in diet with some native herbivores (Dawson and Ellis 1979). This competition has been implicated in the decline (and/or lack of recovery) of rock wallabies in rangeland areas of South Australia and New South Wales (Lim *et al.* 1992; Parkes *et al.* 1996), although this may be a relatively minor threat (Dovey *et al.* 1997).

## Pig

Feral pigs *Sus scrofa* are widespread and abundant in the higher rainfall rangeland areas, and around wetlands and riparian areas in the lower rainfall rangelands of eastern Australia (particularly in the Top End Coastal, Victoria-Bonaparte, Gulf Plains, Cape York Peninsula, Darling Riverine Plains, and Channel Country bioregions). Their total Australian population is roughly estimated as between 3.5 and 23.5 million, most of which is in the rangelands (Choquenot *et al.* 1996); and they reach densities of up to 10-20 animals/km<sup>2</sup> in preferred rangelands environments (such as floodplains and swamplands). The environmental damage due to pigs differs from that of the other feral mammals described above, largely because their diet range is so much greater, including fruits and seeds, foliage and stems, rhizomes, bulbs and tubers, fungi, and animal material (including earthworms, snails, arthropods, crustaceans, shellfish, frogs, fish, reptiles (and their eggs), birds (and their eggs), and small mammals. Pigs may have substantial local impacts due to rooting up ground, especially in rainforest and swamp areas (Russell-Smith and Bowman 1992), which may affect local hydrology, nutrient cycling and erosion, and plant recruitment, with one reported case of such rooting directly endangering one highly localised plant species in rainforests of the northern Australian rangelands (Fensham 1993). There is anecdotal evidence that predation by pigs may have major local impacts on some wildlife species, including earthworms and nesting birds (Choquenot *et al.* 1996).

## Feral rodents

The house mouse *Mus domesticus* is widespread across much of the rangelands, particularly on the semi-arid fringes of eastern and south-western Australia, and in arid central Australia (Caughley *et al.* 1998). In contrast, it is relatively localised and uncommon in the higher rainfall rangelands of northern Australia (Woinarski *in press*). Numbers fluctuate enormously, in response to rainfall and fire patterns (Newsome and Corbett 1975; Masters 1993), and when conditions are suitable, the house mouse may be by far the most abundant mammal species across extensive areas of the rangelands, with an unusually broad habitat range. House mice may affect native biodiversity through competition with native species (rodents, birds and invertebrates), impact upon vegetation recruitment and floristic composition, and changes in predator abundance. However, these effects have not been quantified for the rangelands, and are probably generally minor (Caughley *et al.* 1998). Potentially more serious is that house mice may serve as the dispersal mechanism to native fauna for many exotic viruses, parasites and other diseases (Singleton *et al.* 1993; Caughley *et al.* 1998), although again, there is little evidence to bear on this possible impact. In contrast, there is much anecdotal evidence and some published studies which indicate substantial local impacts upon native fauna of the attempts at chemical control of mice plagues (Caughley *et al.* 1998; Brown and Lundie-Jenkins 1999).

The black rat *Rattus rattus* has a far more restricted occurrence in the rangelands, mostly in coastal areas of high rainfall northern Australia (Caughley *et al.* 1998; Woinarski *in press*). It may have been introduced by Macassan sailors several centuries before European settlement, with records suggesting that it was common and widespread in northern Australia in the 1890s (Dahl 1897). On the mainland, its impacts upon biodiversity have been very limited, but on some islands off northern and western Australia, it occurs at very high densities and may eliminate seabird breeding, alter vegetation patterning and, through competition, threaten important island populations of vulnerable and endangered native mammals (Burbidge 1989; Woinarski *et al.* 1999a).

### Feral birds

The broad expanse of the rangelands has generally not been colonised by exotic birds, although some rangeland towns and more disturbed areas have populations of feral birds, most notably of starling *Sturnus vulgaris* (mostly in the semi-arid rangeland fringe of southern and eastern Australia), feral pigeon *Columba livia*, laughing turtledove *Streptopelia senegalensis* (in Alice Springs), house sparrow *Passer domesticus* (across most of the semi-arid rangelands of eastern Australia), nutmeg mannikin *Lonchura punctulata* (in coastal northeastern Australia) and Indian mynah *Acridotheres tristis* (in northeastern Australia). Notably, most of these are increasing in range and/or abundance (Blakers *et al.* 1984; Franklin 1999), or are being held in check only through vigilant management control. The environmental impacts of these feral populations are poorly quantified. The most significant is probably that of competition for nest hollows between starlings, mynahs and native species, especially in areas where clearing or other land management activities has reduced the availability of these.

### Cane toad

The cane toad *Bufo marinus* was introduced to northeastern Australia in 1935, and is rapidly expanding across the rangelands of northern and eastern Australia. It has colonised an extraordinary range of habitats, and reaches phenomenal densities at some locations (Freeland 1984, 1986). Opinion about its impacts on biodiversity ranges very widely. All stages of its life cycle are toxic, presenting a major hazard to potential predators (including fish, other frogs, some invertebrates, birds, goannas, snakes, turtles, crocodiles and dasyurid mammals), and it is a voracious consumer with a broad diet of invertebrates and small vertebrates. There is a substantial body of largely anecdotal evidence that invasion by cane toads has coincided with local declines (and possible local extinctions) of a range of dasyurid, turtle, goanna and snake species (e.g. Covacevich and Archer 1975; Burnett 1997), but more rigorous studies have generally failed to detect long-term impacts upon biodiversity (Freeland and Kerin 1988; Catling *et al.* 1999).

### Invertebrates

Many invertebrates have been introduced intentionally or accidentally to the Australian rangelands. For most species, there is remarkably little information on the consequences of this introduction for native biodiversity. Exceptional are a series of studies on the environmental impacts of the big-headed ant *Pheidole megacephala*, which have shown a substantial reduction in the local diversity of native invertebrates in rainforest sites in northern Australia (Hoffman 1998; Hoffman *et al.* 1999), and in open forest and coastal dunes elsewhere in Australia (Venderwoude 2000). In turn, the simplification of invertebrate assemblages at colonised sites may then affect vertebrate assemblages and vegetation dynamics.

The more deliberate introduction and subsequent feral spread of the honey bee *Apis mellifera* to rangeland areas may have less localised but more extensive environmental impacts, potentially including disruption of plant-pollinator systems (especially where native plants have pollination systems involving only a restricted number of specialised native pollinator species), resource use and/or monopolisation leading to decline in native pollinator species, and usurpation from nesting and roosting hollows of native invertebrate and vertebrate species. However, the assessment of this impact is difficult to quantify and generally not yet conclusive (New 1997; Schwarz and Hurst 1997; Manning 1997; Paton 1997). Apiculture may also cause some native species (notably the rainbow bee-eater *Merops ornatus*) to become regarded as pests, and hence at least locally destroyed.

### Parasites

Many parasites, pathogens and other disease-carrying organisms have been introduced to Australia, through human immigration, and the introduction of livestock, pets and food. The previous isolation of Australia rendered much of the fauna unprepared for the diseases associated with such exotic organisms. Where the native fauna can be colonised by such diseases and parasites, the impacts may be unusually potent and extensive (Murray and Snowdon 1976; Spratt and Presidente 1981; Freeland 1993). Disease episodes causing very high incidences of mortality and/or morbidity have been reported (mostly anecdotally) for many rangeland vertebrate species, with some of these events likely to be caused by newly invading organisms, and some of these cases probably contributing substantially to local and regional extinctions (Finlayson 1961; Freeland 1993). For example, the protozoan parasite *Toxoplasma gondii* is now a well recognised cause of disease in many Australian marsupials, but the only species known to act as the final or definitive host in its life-cycle is the cat *Felis catus* (Obendorf and Munday 1990).

## Weeds

Humphries *et al.* (1991) reviewed the occurrence and impact of introduced plants in Australia. Some bioregions have been subject to particularly marked landscape change caused by weeds, but all rangeland areas are affected to some extent by exotic plant species. In some cases, this marked change is desired by some landholders (e.g. the planting of improved pasture grasses). Adverse environmental impacts may be associated with exotic plants which are economically beneficial, neutral or detrimental. In some cases, there may be substantial environmental impact even when weeds are relatively inconspicuous. The following discussion briefly considers impacts of some of the most significant weeds upon native vegetation biodiversity in Australia's rangelands.

*Acacia nilotica*. Over 500,000 ha of mitchell grasslands are infested by this old world tree. Frost may limit its spread to the south but its range is certainly expanding. Nothing has been documented regarding its impact on biodiversity. It undoubtedly eradicates native grassland under its canopy. Fensham *et al.* (2000) demonstrated that the botanical composition of mitchell grassland varies over its geographic range in concert with macro-climatic gradients. Thus the invasion of prickly acacia may be having a significant impact on the abundance of some native plant species. However, because the mitchell grasslands are so extensive the invasion of prickly acacia or other species in this environment is unlikely to be the cause of species endangerment.

Ponded pastures (principally based on the exotic *Brachiaria mutica*, *Echinochloa polystachya* and *Hymenachne amplexicaulis*) pose a severe threat to floodplains and wetlands throughout tropical Northern Territory and Queensland. These species have the capability to completely displace native vegetation (Wilson *et al.* 1990; Neldner *et al.* 1997), and thence native animals (Whitehead 1999; Whitehead and Dawson 2000). Given that wetland habitats are naturally restricted, these exotic perennial grasses pose a serious threat to native species. There has been no quantification of the rates of spread, although some of these species are still being actively promoted by government agencies (Neldner *et al.* 1997).

*Cenchrus ciliaris*: Buffel grass is listed by Humphries *et al.* (1991) chiefly because of its impact to the mesic habitats in semi-arid and arid regions. The pervasiveness and continuing invasion of these habitats by this species is confirmed by more recent studies and observations (Griffin 1993; Best 1998; J. Machonichie, D. Albrecht *pers. comm.*). However, while the species is recognised as a serious threat by Humphries *et al.* (1991), its impact and distribution there may be understated. The Queensland herbarium has collections in every pastoral district and it is the dominant grass over nearly all of the improved pastures in the Brigalow Belt to the east of its range as represented by Humphries *et al.* (1991). Fairfax and Fensham (2000) estimate that it is capable of proliferating throughout semi-arid Queensland on most soil types, except the heavy clays and extremely infertile soils. It is spreading through woodlands naturally dominated by spinifex with phosphorus levels well below 5ppm and there are genotypes that have

proliferated on extremely acid soils formed on granite in Mexico (R. Silcock *pers. comm.*). Fairfax and Fensham (2000) report observations of the rapid spread of this species through remnant vegetation. The vast majority of land clearance in Queensland is accompanied by the establishment of buffel grass. This spread is exacerbated by the deliberate sowing of the species following land clearance. It is estimated that the species is currently abundant or dominant over 30-50 million ha in Queensland alone. The species develops far greater bulk than native species and is generally viewed as a highly valued pasture species. Fairfax and Fensham (2000) documented 56% higher native plant species richness in uncleared native pasture compared to cleared buffel grass pasture for gidgee lands, 69% higher for eucalypt lands and 295% for brigalow lands. Buffel grass also causes changes in the species composition of invertebrate communities and reduction in invertebrate biodiversity (Best 1998), and may have similar effects on vertebrates, although there are no substantial studies yet published (Ludwig *et al.* submitted). Given this local-scale change and reduction in biodiversity, and the vast extent of this alien species, buffel grass has certainly had a very major impact on Australia's biodiversity. This impact has spread beyond pastoral tenure to incorporate conservation lands, with understoreys dominated by buffel grass causing severe problems for conservation management of fire sensitive vegetation in many National Parks.

*Cryptostegia grandiflora*. Rubber vine poses a severe threat to dry rainforest and riverine vegetation in north Queensland. There have been no quantified studies of its effects on native vegetation. Fensham (1996) discusses the impact of the species on dry rainforest in north Queensland, but ranks the species at a lower level of threat for that vegetation type than *Lantana camara*. The species can be effectively managed with fire in non-rainforest vegetation and a rust has resulted in diminished vigour throughout much of the species range.

*Mimosa pigra*. This invasive shrub poses a severe threat to the floodplains of northern Australia, with around 10% of the floodplain grasslands and sedgeland in the Northern Territory now transformed to mimosa thickets (Lonsdale *et al.* 1995), with consequential change in suitability for many native plant and animal species (Whitehead *et al.* 1990; Whitehead 1999). The species continues to spread rapidly, with dispersal aided by transport by feral animals, flotation and waterfowl (Lonsdale 1993; Cook *et al.* 1996).

*Parkinsonia aculeata*. This tree provides a severe threat to wetlands and watercourses throughout semi-arid tropical Australia and is particularly severe on the black soils where it can also invade drylands. There is no systematic research addressing its rate of spread or impacts on biodiversity.

*Pennisetum polystachion*. This grass is invading eucalypt savanna woodland. It has the capacity to build up large fuel loads and hence increase fire intensity. It is abundant around Darwin and is also becoming a problem in Kakadu and Cape York Peninsula. Its potential for further spread would seem to be high. Panton (1993) has documented the role of this species in causing the almost total destruction of a monsoon rainforest near Darwin.

*Andropogon gayanus*. This grass has a similar pattern of invasion as mission grass, although it would appear to have the capacity to build even higher fuel loads (see Bowman 1999).

*Prosopis* spp. Tree taxa within the genus *Prosopis* would appear to have a similar ecology to *Acacia nilotica* but with a broader climatic and edaphic range. They have penetrated further into temperate zones, becoming a pest in New South Wales. They are not yet as abundant as prickly acacia but would appear to pose a particularly insidious threat.

*Lantana camara* has severe ecological impacts within relatively high rainfall areas along the eastern seaboard. However, the species penetrates inland to rangeland bioregions with greater than 700mm mean annual rainfall (e.g. Einasleigh Uplands). Its disastrous impact in dry rainforest at Forty Mile Scrub in combination with fire and pigs is documented by Fensham *et al.* (1994). This provides a clear example of interacting threatening processes combining to form a particularly serious impact. They surmised that pig rooting has broken up the overstorey canopy and that this has then provided enough light to allow for the establishment of lantana. Lantana has proliferated and now forms a dense fuel layer that renders a normally inflammable vegetation type highly flammable. Fires from the surrounding savanna have penetrated destroying the overstorey. The resulting vegetation is an extremely dense lantana shrubland with native species regenerating from basal sprouts. This shrubland is more flammable than ever and provides a wick for extremely hot fires to penetrate further and further into the previously unburnt forest. Since Fensham *et al.*'s (1994) study the situation at Forty Mile Scrub has become even more parlous. Lantana has spread almost throughout the forest and fires have now effectively destroyed a large proportion of this forest. The situation is not confined to Forty Mile Scrub with many other scrubs in the Einasleigh Uplands being particularly severely affected. Clearing for mineral exploration lines is another agent for providing the initial penetration of lantana into dry rainforest. Several of the most pristine dry rainforests in the Einasleigh Uplands at least have been severely degraded by this means since the study of Fensham in 1992/93.

The arid and semi-arid rangelands of southern Australia have been less disturbed by weed infestation than the northern rangelands and the settled eastern and south-western margins of the continent (Humphries *et al.* 1993). However, there are some severe localised problems (especially in areas currently or previously exposed to over-grazing, such as parts of the Flinders and Olary Ranges bioregion: Greenwood *et al.* 1989) and more extensive (albeit relatively minor) problems in these southern rangelands, including *Prosopis* spp., African boxthorn *Lycium ferocissimum*, warden weed *Carrichtera annua*, salvation jane *Echium plantagineum* and rosy dock *Rumex vesicarius* (Muir 1992; Humphries *et al.* 1993; Department of Conservation and Land Management, WA 1999).

## Mining

Mining has been a major industry across the rangelands for more than a century, and continues to be the most economically important industry for many rangeland bioregions, such as the Mount Isa Inlier, Coolgardie, Pilbara, Stony Plains, Pine Creek-Arnhem, and Broken Hill Complex.

This economic significance far outweighs the proportion of rangeland lands directly affected by mining activities. However impacts of mining upon biodiversity can be locally devastating, and can extend considerably off-site. The impacts of mining activities can be broadly divided into:

- pre-mining exploration;
- direct impacts of excavation;
- use of natural resources for mine construction;
- use, or disposal, of toxic materials and waste waters;
- changed hydrological processes.

The most notable feature of mining exploration in at least some rangeland areas is the construction of a network of seismic lines, often in areas remote from any other disturbance (e.g., Greenslade *et al.* 1986; Gibson *et al.* 1997; Gibson and Lyons 1998*a,b*). In most cases, these have been left with no attempt at restoration. There is little documentation of their impact, but they may lead to localised erosion, spread of weeds into otherwise relatively weed-free areas, and may favour the spread or hunting efficiency of feral predators. Uncapped cores left from test drillings may form lethal pitfall traps for local fauna (Ehmann 1993). Some mining exploration activities include the use of extensive hot fires to clear vegetation in order to provide more detailed resolution of imagery, to improve access and to view geological features better. Frequent use of such fires may be detrimental to biodiversity.

Most rangeland mining operations are relatively localised, and hence their direct impacts relatively circumscribed. However, where the target of mining and exploration provide specialised habitat such as ultramafic rocks as a source of nickel impacts can be substantial on localized endemic species or communities. For example, in the Parker Range of southern Western Australia, the occurrence of the endemic plants *Euromyrtus ciliata* and *Isopogon robusta* coincides closely with mining activities (Gibson and Lyons 1998*b*).

Strip-mining activities, such as for manganese on Groote Eylandt (Central Arnhem bioregion) and bauxite at Nhulunbuy (Top End Coastal bioregion) and Weipa (Cape York Peninsula bioregion), are exceptional in transforming far more extensive areas. In such cases, direct impacts may extend to tens or hundreds of square kilometres. While regeneration may allow much of the biota to recolonise strip-mined areas, the few studies

have shown that some components of biota do not recolonise, due to the absence or reduction of some habitat features, such as hollow-bearing trees and suitable ground cover (Winter and Atherton 1985; Cameron and Cogger 1992). For more point-based mining activities, impacts upon biodiversity depend upon the conservation values of the localised area at the mine site. In exceptional cases, the mine-site may coincide with much of the distribution of highly restricted species, or coincide with significant aggregation sites for colonial species. The most notable of the latter case is that of the important ghost bat *Macroderma gigas* maternity colony at Mount Etna, in the Brigalow Belt North bioregion (Jolly 1987). Improved regeneration practices developed over the last few decades have led to increased proportions of the pre-mine biota recolonising mined areas, and have included some notable long-term monitoring of biodiversity in such areas (Majer 1983; Read 1992; 1999b; Andersen 1993, 1997; Andersen and Sparling 1997).

Mining activity may also directly benefit some elements of biodiversity. The most well-documented case is for those bat species which are normally obligate cave-dwellers. Hall *et al.* (1997) noted that 29 bat species were known to use abandoned mines as roost sites, with 21 of these species using them for maternity roosts. They considered that mining activity had contributed to the extension in distribution and/or increase in population for at least 10 species. However, this benefit is not necessarily enduring, as old mines may collapse more readily than natural caves, may be exposed to re-development, and/or may contain toxic substances.

Particularly with the earliest phases of mining in the rangelands, more extensive biodiversity consequences followed from the use of natural resources for mine construction and operation, particularly of timber for structural support of mine tunnels and, in some cases, for fuel. For example, in 1880s, mining smelters at Cobar were consuming 70,000 tons of wood per year, and timber supplies near the town had become so sparse that an 18km railway was constructed away from the town to search for and haul wood (Pressey 1990). Borrow pits for gravel for road-making may also impact particular environments selectively (Atkins *et al.* 1999), and where suitable material is regionally limited, these impacts may cumulatively be substantial.

Many mining ventures use or produce a range of chemicals which may be harmful to biodiversity. Cyanide is used in many gold mines, and is typically then stored in tailings dams. As the rangelands are characterised by relatively few water resources, such dams may attract many aquatic species (and especially waterfowl), resulting in substantial mortality when dams have high concentrations of cyanide (most notably including about 2700 birds killed in a single incident at one tailings pond: Sinclair *et al.* 1997). Waterfowl mortality is a relatively common occurrence at tailings dams across the rangeland area, although industry standards seeking to reduce WAD cyanide concentrations to below 50mg/L have generally been successful in reducing mortalities (Donato 1999). Where retention is inadequate, chemical pollution due to mining by-products may extend well beyond the mine site. For example, following discharge of high concentrations of copper, manganese and zinc from the Rum Jungle uranium mine (Top End Coastal bioregion) into the east branch of the Finnis River, Jeffree and

Williams (1980) noted that very few fish and macroinvertebrates survived, with major fish kills for at least 15 km downstream. Discharge of sulphur dioxide from the Mt Isa silver-lead-zinc mine (Mount Isa Inlier bioregion) into the atmosphere has led to considerable change in environments and biota within 15 km downwind of the source, with major reduction in species richness and change in species composition of plants, ants, reptiles and birds (Griffiths 1998). A similar, but more localised, response of reptiles (particularly geckoes) to sulphur dioxide and salt-spray emissions was reported by Read (1998,1999*b*) from the Olympic Dam Corporation mine at Roxby Downs (Stony Plains bioregion).

Some mining developments use very large quantities of water, and this use may affect environments relatively distant from the mine. For example, water extraction from the Great Artesian Basin for the Roxby Downs Olympic Dam mine may lead to substantial degradation of mound springs also dependent upon that water source, although the extent of this impact has been contested (Seidel 1980; Kinhill Stearns 1984; Boyd 1990).

## Hunting and harvesting of native species

There has obviously been a long history of use of native plant and animal resources by Aboriginal people across all rangeland areas. In some cases, this use was accompanied by land management practices which helped to sustain and enhance populations of preferred species (Williams and Hunn 1982; Bomford and Caughley 1996). In other cases, harvest may have been less sustainable, and there is a widely-disputed suggestion that hunting by Aboriginal people caused the extinction of at least some of the marsupial megafauna (Merrilees 1968; Horton 1980; Flannery 1990, 1994), but a more compelling case for hunting by Aboriginal people as a major cause of the loss of some mammals (especially macropods) from some islands of northern and western Australia (Abbott 1980).

Aboriginal use of native plant and animal resources continues in many rangeland areas, and has been enshrined in legislation (e.g., Haigh and Coleman 1995). In rangeland areas where the Aboriginal population density is relatively high and traditional practices are largely maintained, the harvest of native species may be considerable (e.g. around 290,000 magpie geese *Anseranas semipalmata*, 240,000 goannas *Varanus* spp., and 180,000 flying-foxes *Pteropus alecto* and *P. scapulatus* consumed by Aboriginal people in the Top End of the Northern Territory annually: Vardon *et al.* 1997). Where take is for local (i.e. non-commercial) consumption and closely monitored by landowners, this rate is probably generally sustainable (Vardon *et al.* 1999), at least in the absence of novel threatening processes also affecting populations. Sustainability may be a more challenging outcome when harvesting by Aboriginal landholders is for economic gain (Vardon and Tidemann 1995; Tidemann and Vardon 1997), as evident in the recent largely unregulated expansion of the didgeridoo market across many parts of the rangelands.

Hunting of wildlife accompanied much of the entry of Europeans to the rangelands, initially as a necessary part of subsistence, but subsequently for sport (for some species in some areas), for commerce (especially for furs and meat) and for “pest control”. Rangeland animals hunted for sport have included waterfowl, macropods and quail. At various stages in the rangelands, there has been at least locally substantial commercial harvesting of possums, koalas, macropods, water-rats, pythons, crocodiles and emu for pelts; of possums, macropods, crocodiles, waterfowl, pigeons, bustards, emu and fish for meat; and of finches, parrots, various reptiles and fish for the pet trade. Native animals harvested as pests in the rangelands have included the dingo, macropods, possums, quolls, bandicoots, flying-foxes, emu, cockatoos and parrots, magpie geese and finches. This take has included very large numbers for some specifically targetted species, such as nearly three million rat-kangaroos in New South Wales between 1883 and 1920 (Short 1998), and over 280,000 finches in the period 1974-1986 in the Kimberley (Franklin *et al.* 1999). Current levels of exploitation may also be considerable, especially for macropods – for example, the annual quota for kangaroos in Queensland and New South

Wales combined is between 3 and 4 million animals (Eveleigh 1995; Lundie-Jenkins *et al.* 1999; Gilroy 1999).

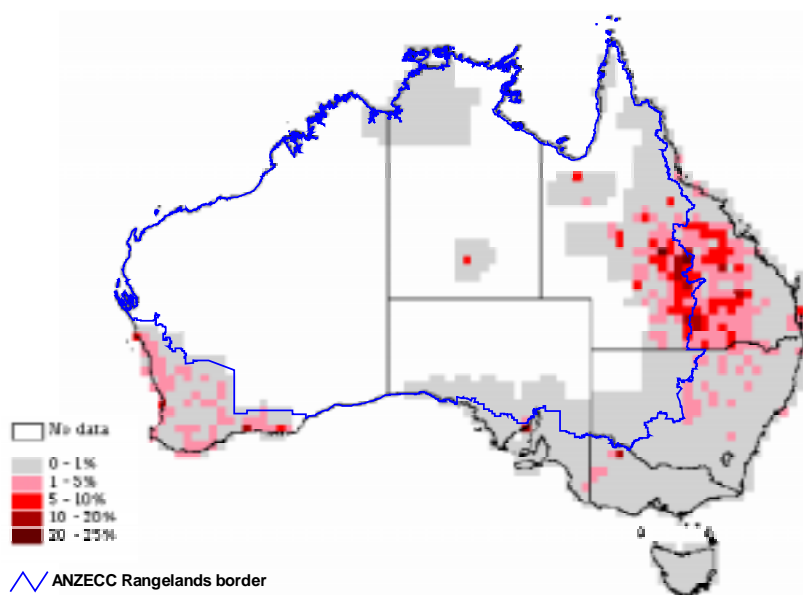
In some cases, there is reasonable, though largely anecdotal, evidence that this historic harvesting led to, or contributed to, major decline, local or regional extinctions for some rangeland species, such as for the koala *Phascolarctos cinereus* in central Queensland (Finlayson 1934). These losses were a result of unregulated commercial hunting, generally associated with industries operating before responsibility for sustainability was accepted. A more substantial regulatory framework is now established for most commercial and recreational hunting and harvesting of native fauna in the rangelands, especially for waterfowl and kangaroo-harvesting, with focus now much more on “resilient” species. This regulation and sustainability is based largely on a reasonably sound knowledge of population numbers and life history parameters (Caughley *et al.* 1987; Pole and Cairns 1995; Grigg *et al.* 1999), with ongoing monitoring of harvest and response (Caughley *et al.* 1977). Indeed, the population monitoring associated with harvested waterfowl and macropods is the best available broad-scale monitoring for rangeland biodiversity (e.g. Caughley 1977; Cairns *et al.* 1991; Grigg *et al.* 1999).

In at least some rangeland areas, there is a compelling argument that the development of a kangaroo-harvesting industry (and others based on the sustainable use of native biodiversity) may have substantial biodiversity benefits, if the expansion of this industry leads to reduced grazing pressure from exotic herbivores (Grigg 1987, 1989, 1995; Lunney 1995; Armstrong and Abbott 1995; Sattler 1995; Alexander 1997); however, the benefits of such replacement are not necessarily achieved so simply (Freudenberger 1995).

There has also been intensive commercial harvesting of some rangeland plant species, including sandalwoods *Santalum* spp., cypress-pines *Callitris* spp., ironwood *Erythrophleum chlorostachys*, river red gum *Eucalyptus camaldulensis* (mainly for railway sleepers), and mallee eucalypts (for roots and oils); and substantial levels of harvest of some timbers for local use as fence-posts and building material. Again, where poorly regulated, this has led to major declines of targeted species in at least some rangeland areas (Hanssen and Wigston 1989).

## Clearing

The rangeland area is defined in part by the absence of intensive agriculture, and hence, by perversity of definition, vegetation clearance is not a conservation management issue across most of the rangelands. Nonetheless, vegetation clearance for improved pasture for grazing and for crops has been substantial in parts of the rangeland margins (notably in the Murray-Darling Depression, Cobar Peneplain, Brigalow Belt North, Desert Uplands, Mulga Lands and Coolgardie bioregions) and in parts of the higher rainfall rangelands proper (notably in the Victoria-Bonaparte, Daly Basin and Top End Coastal bioregions), and clearing is continuing at or around an unprecedented pace in some of these rangeland bioregions (Figure 7; Table 7). For the purposes of this review we include the ploughing of grasslands for crops or improved pasture as vegetation clearance. Both the Brigalow Belt South and Brigalow Belt North bioregions are now more than half cleared and thus no longer strictly defined here as rangelands. However, large areas of these bioregions are still extensively rather than intensively grazed by stock and they are still subject to substantial clearing. The results from recent studies by the Bureau of Resource Sciences indicate the breakdown of clearing by state. The vast majority of clearing is occurring in Queensland and is replacing rangeland with exotic pasture (chiefly buffel grass).



**Figure 7. Clearing rates in Australia, 1990-95**, expressed as a percentage of native vegetation remaining in 1990. (Modified from Barson *et al.* 1999). The rangeland area is indicated by the blue line.

**Table 7. Status and clearing rates of remnant vegetation by bioregion, conservation status and tenure in Queensland.** The Regional Ecosystems are categorised into their conservation status: *Endangered* is defined as remnant vegetation less than 10% of its original area, or 10-30% of the original area and remnant area less than 10,000ha; *Of Concern* is 10-30% of original area or remnant area less than 10,000ha; and *No Concern* is >30% of original area and remnant area greater than 10,000ha.

Biogeographic Region (no. of Regional Ecosystems)	Total area (ha)	EN-DANGERED		OF CONCERN		NO CONCERN		REMNANT AREA BY STATUS BY TENURE (ha)		
		remnant 2000 <sup>1</sup> (ha)	ann. clearing 1995-1997 (ha)	remnant 2000 (ha)	ann. clearing 1995-1997 (ha)	remnant 2000 (ha)	ann. clearing 1995-1997 (ha)	En-dangered Freehold Lease- Reserves <sup>9</sup> hold		
North West Highlands (41)	7,314,309	0		0	0	7,281,484	420	0	0	0
Gulf Plains (83)	22,000,704	0		0	0	21,948,715	3,399	0	0	0
Cape York Peninsula (211)	12,167,306	1		297,000	0	11,799,162	1,730	0	1	0
Mitchell Grass Downs (53)	24,183,350	0	0	304,390	9,810	19,896,906	5,807	0	0	0
Channel Country (56)	23,814,608	0	0	5,000	0	23,764,590	1,886	0	0	0
Mulgalands (66)	18,500,259	49,295	1,887	274,799	9,070	14,986,630	29,202	40,422	6,408	2,465
Wet Tropics (105)	1,983,855	8,000		100,000	0	1,045,200	1,778	4,000	1,600	2,400
Cent Mackay Coast (37)	1,443,561	51,410	1,268	104,993	1,687	882,961	1,344	22,620	20,564	8,226

**Table 7 (continued)**

Einasleigh Uplands (46)	11,861,477	0	104	100,000	122	10,981,880	4,051	0	0	0
Desert Uplands (58)	7,030,753	941	34	100,290	2,107	5,498,735	27,127	423	518	0
Brigalow Belt (163)	36,417,956	696,178	20,370	2,179,809	49,284	11,518,978	58,252	320,242	313,280	62,656
Southeast Qld (145)	6,212,667	151,849	141	440,140	3,336	1,853,100	7,472	92,628	15,185	44,036
New England Tableland (21)	775,157	22,989	177	67,203	620	175,817	353	20,000	1,379	1,609
<b>Totals</b>	<b>173,705,962</b>	<b>980,663</b>	<b>23,981</b>	<b>3,973,624</b>	<b>76,036</b>	<b>131,634,158</b>	<b>142,821</b>	<b>500,336</b>	<b>358,935</b>	<b>121,392</b>

<sup>1</sup> 2000 remnant area extrapolated from 1995 using clearing rates for 1995-1997

<sup>8</sup> From SLATS 1995-97 report

<sup>9</sup> Reserves includes state forest conservation and other reserves

**Table 7 (continued)**

Biogeographic Region  (Regional Ecosystems)	Of Concern			DATA SOURCE	
	Freehold	Leasehold	Reserves	Data reliability	Mapping source
					<b>Note figures derived from incomplete mapping of a region are plus or minus 20%</b>
North West Highlands (41)	0	0	0	Intermediate	Limited spatial data, and only limited threatened REs affected by clearance. Remnant clearing rates estimated.
Gulf Plains (83)	0	0	0	Robust	Limited spatial data, and only limited threatened REs affected by clearance. Remnant clearing rates estimated.
Cape York Peninsula (211)	44,550	213,840	38,610	Robust	Remnant clearing rates estimated.
Mitchell Grass Downs (53)	152,195	133,932	18,263	Robust	Mapping for majority of region heavily impacted by tree clearing.
Channel Country (56)	50	4,600	350	Robust	Remnant from mapping data for most of region. Remnant clearing rates estimated.
Mulgalands (66)	208,847	43,968	21,984	Robust	Mapping for all of the 15% of the region heavily impacted by tree clearing. Remnant clearing rates estimated for the remainder.
Wet Tropics (105)	31,000	24,000	45,000	Unreliable	Mapping data from lowlands between Townsville and Tully, extrapolated for remainder of region
Central Mackay Coast (37)	9,449	95,544	0	Intermediate	Mapping for 80% of region. Extrapolated over the remainder where figures are less reliable

**Table 7 (continued)**

Einiasleigh Uplands (46)	8,000	89,000	3,000	Intermediate	Limited spatial data, and only limited threatened REs affected by clearance. Remnant area and remnant clearing rates estimated
Desert Uplands (58)	21,061	77,223	2,006	Intermediate	Mapping for 100% of region; Subject to minor review.
Brigalow Belt (163)	937,318	1,046,308	196,183	Robust	Mapping for 98% of region. Subject to minor review.
Southeast Qld (145)	237,676	35,211	167,253	Robust	Mapping data for most of region. Subject to minor review.
New England Tableland (21)	59,139	1,344	6,720	Robust	Mapping data for most of region. Subject to minor review.
Totals	1,709,285	1,764,970	499,369		

**Table 8: Clearing rates in Australia by cause of change between 1990-1995.** (source Bureau of Resource Sciences (<http://www.brs.gov.au>). NA not available. Note that these figures include non-rangeland areas. Values given are in ha.

<b>State</b>	<b>Agriculture</b>	<b>Grazing</b>	<b>Other</b>	<b>Total</b>
Australian Capital Territory	0	0	0	0
New South Wales	51,860	NA	8,070	59,930 +
Northern Territory	5,870	1,070	9,550	16,490
Queensland	40,510	924,410	27,380	992,300
South Australia	6,130	NA	360	6,490 +
Tasmania	230	3,790	440	4,460
Victoria	9,240	NA	3,480	12,720 +
Western Australia	106,220	NA	19,400	125,620 +
<b>Total</b>	<b>220060</b>	<b>929,270 +</b>	<b>68,680</b>	<b>1,218,010 +</b>

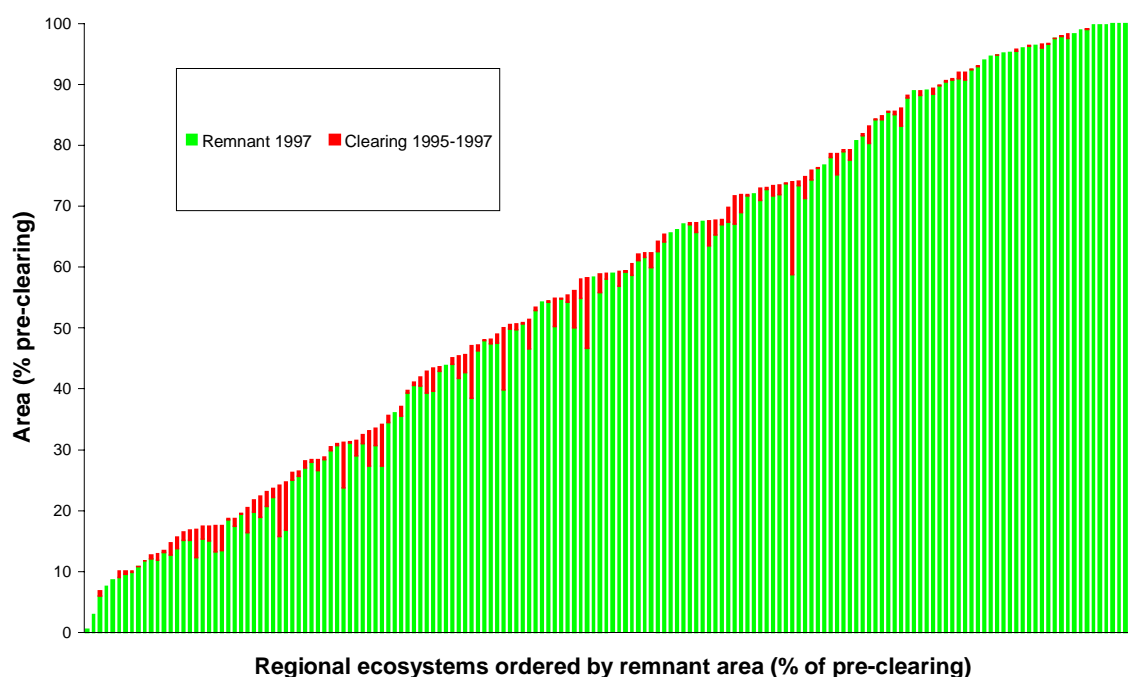
The status of the rangeland bioregions within Queensland and recent rates of clearance are available from the data compiled as part of the Queensland Herbarium mapping program and the Statewide Landcover and Trees Study (SLATS) (Statewide Landcover and Trees Study 1999) (Table 8). The majority of current clearance is occurring in the Brigalow Belt bioregions, although the frontier of clearing is still moving west-ward into the more marginal lands of the Mulgalands, Mitchell Grass Downs and Desert Uplands regions.

In the past, clearing has generally been selective, affecting especially relatively fertile or potentially fertile lands, and relatively flat lands (Pressey 1990; Fensham *et al.* 1998). Hence particular rangeland environments most affected include those on clays (including some grasslands, brigalow woodlands and *Eucalyptus microtheca* – *Bauhinia cunninghamii* woodlands), mallee vegetation in the southern rangelands and some eucalypt woodlands in central Queensland. Clearing has greatly diminished some of these environments, rendering some now close to ecologically non-viable (Pressey 1990; Benson *et al.* 1997; Fensham *et al.* 1998). Clearing is now occurring apace in marginal lands that were almost intact as little as a decade ago. The eucalypt woodlands with a spinifex understorey that typify the Desert Uplands are a case in point. Land management manuals recommend against the clearing of this land type (Turner 1978). However, it has been the clearing of this land type that has resulted in the extremely high rates of recent clearance in the Desert Uplands over recent years (Statewide Landcover and Trees Study 1999). An analysis of the Queensland Herbarium data that superimposes recent clearing rates reveals a number of issues with regards land clearing in Queensland (Fig. 8). The status of each Regional Ecosystem is a relatively reliable inverse surrogate

of its productive potential i.e. the most arable land has been most cleared and the least arable is most intact.

- For many ecosystems it is occurring at a greater rate than 1% per annum
- Clearing is occurring at high rates in the parts of the landscape that are of moderate productivity, but are sufficiently intact to ensure that the current patterns of clearing will continue for many decades

**Figure 8. Recent clearing rates for Queensland regional ecosystems** (see Sattler and Williams 1999). Clearing rates are indicated by the length of the red bar; the status (proportion remaining in 1997) of each regional ecosystem is represented by the length of the green bar.



Clearing may also be less absolute, but selective for particular tree species, with extensive cutting of individual species with favoured timbers for fenceposts and housing being a commonplace accompaniment of pastoral expansion throughout the rangelands, leading to at least localised marked changes in woodland structure and species composition (Pressey 1990). At various rangeland regions and various times, there has also been substantial amounts of selective timber cutting to fuel mining smelters, riverboat furnaces, for drought forage, and for structural support in mines.

The impacts of clearing on biodiversity include:

- spread of invasive exotic plants, which then have the capacity to invade non-cleared lands;
- localised loss of populations, which, if the total species range is limited, may mean extinction;
- reduced viability of remnant patches, which may be too small to support individual species with extensive home range requirements;
- reduced scope for management flexibility, most notably in seeking the maintenance of a range of fire regimes;
- substantially reduced total populations of some species, which may render them more prone to other threatening factors.

Clearing, or the destruction of habitat, is widely recognised as the most serious process threatening plant species. In States such as Western Australia, where broadscale habitat destruction has dwindled, its importance may have diminished, but nonetheless Brown *et al.* (1998) declared for that State that “*land clearing has undoubtedly been the major threatening process in the past, particularly for plants*”. In many cases the consequences of clearance as a threatening process may be realised only when it is too late. The Brigalow Belt North and South have been dramatically transformed, with about 58% of habitat cleared, with particularly high rates for the brigalow communities themselves. Johnson (1997) asserted that the flora of this vegetation type is sufficiently generalist that there are few if any rare taxa. However, more recent analysis of the flora suggests that the clearance of brigalow has imperilled some taxa. The collections of *Solanum dissectum* at the Queensland Herbarium suggest that it only occurred in Brigalow scrubs in the Dululu-Baralaba-Moura-Biloela area. It has not been collected since 1989 and there are very few remnants surviving of its habitat within the area. It may well survive somewhere but must at least be regarded as either extinct or critically endangered until re-survey of potential habitat is conducted. *Xerothamnella herbacea* is also only known from brigalow and related vegetation types and while extant populations are known it is rightly regarded as endangered.

There are few studies of the effects of vegetation clearing on rangeland faunal biodiversity. The most detailed is for mallee reptiles, for which Cogger (1989) noted that “*The effects of mallee clearing are unequivocal, resulting in the permanent loss of 70-95% of the original herpetofauna*”. Of 37 reptile species found at a site with mallee vegetation in western New South Wales, only 10 of these species (27%) were regularly found in an adjacent area cleared and grazed about 20 years previously, and only 4 species (11%) in another adjacent area cleared and used for cropping. The species remaining in the cleared areas were a biased subset, comprising fossorial (foraging in the soil) species or generalist species known to have very broad habitat tolerances (Cogger 1984). Ehmann and Cogger (1985) estimated that clearing of mallee lands in South Australia and New South Wales since the mid-1960s would have conservatively led to

the death of at least 39 million reptiles, and the permanent removal of 26 species from those cleared lands.

Obviously clearing has a substantial impact on vegetation structure. It has been argued that clearing provides a new age surrogate for the burning of aborigines, and hence a balance to “natural” increases in tree cover (Burrows 1998). The effects of fire on the Australian rangelands, and the extent to which changed fire regimes have contributed to increase in vegetation cover, is the subject of vigorous debate. Certainly vegetation structure of rangelands in Australia has great relevance to the pastoral industry because of the inverse relationship between the abundance of trees and grass (Burrows *et al.* 1990). The real possibility that the density of the woody component of woodlands is increasing is thus of grave concern for that industry and has been a prime motivation for the mechanical clearance of vegetation. There is widespread perception that the Australian landscape was substantially more open at the time of European settlement than it is today (Rolls 1981; Flannery 1994). This view is corroborated by international examples, with those from mesquite rangelands in Texas having been the subject of the most sophisticated study (Boutton *et al.* 1998). The evidence of this purported dramatic change in Australia is dependent on the interpretation of the historical record. Careful scrutiny of the historical scenario has concluded that while some dense vegetation has undoubtedly advanced (such as rainforest in north Queensland), the view that broadscale areas have converted from grassland and open woodland to forest is undoubtedly overstated (Benson and Redpath 1997; Fensham and Holman 1998). There is little doubt that stocks of woody vegetation can undergo considerable flux. The open question is whether these fluxes are the result of normal climatic cycles, the result of cattle grazing through the inverse relationship between wood and grass, changes in fire regime or are the result of feedback resulting from the primary cause of the greenhouse effect, namely CO<sub>2</sub> fertilisation. Fensham and Holman (1999) have recently demonstrated that dieback collapses can undoubtedly result from extreme drought events. The disentanglement of climate and other influences on woodland structure must await the findings of further research.

Clearing may also have substantial impacts off-site, notably in the consequence of increased levels of run-off, erosion and siltation of waterways (Scanlan and Turner 1995).

Not all species are equally susceptible to the impacts of clearing, and indeed some may be advantaged. The groups most likely to be affected are trees and woody shrubs (obviously), and arboreal fauna, or those terrestrial species dependent upon the microclimate or litter characteristics determined by the original vegetation.

Clearing is not necessarily terminal, as, without further modification, some species in some environments may regrow or re-invade regrowth areas. For example, Johnson (1997) refers to unpublished data suggesting that rehabilitated brigalow regrowth can have the full complement of plant species found in uncleared vegetation, and Panton (1993) provides an example of rehabilitation of a rainforest patch which was previously largely destroyed by fire, cyclones and weeds. Such recovery cannot be assumed for all

components of the biota, and the rate and extent of recovery may be very dependent upon management of the regrowth and its surrounds.

## Horticulture

The rangelands are defined to exclude areas of intensive horticulture. But horticulture is a significant land use for bioregions around the southeastern and southwestern margins of the rangelands (e.g., Murray-Darling Depression, Carnarvon, Cobar Peneplain, Darling Riverine Plains, Coolgardie: principally involving wheat and, in some places, rice), the eastern margins (e.g., Brigalow Belt North, Mulga Lands: mainly cotton), and some northern bioregions (e.g., Victoria-Bonaparte, Daly Basin, Top End Coastal: for tree crops, sugar cane and mixed crops). Advances in dryland cropping techniques, genetic modification of crops, and greater use of water resources are likely to lead to an increased extent of horticulture in many other rangeland bioregions. So it is pedantic and short-sighted to consider trends in rangeland biodiversity without some consideration of the extent and impacts of horticultural development.

Horticultural development may have a number of direct and indirect impacts upon rangeland biodiversity. Typically, there is an acute localised impact through the replacement of native vegetation typically with monocultures of exotic plant species. This leads to marked reduction in species richness and localised loss of many native plants and animals (e.g., Abbott *et al.*, 1979; Majer and Beeston 1996). These local losses accumulate to regional losses when horticultural development is extensive, when particular environments are devoted mostly to horticulture, and where little attempt is made to maintain some fragments of natural areas within the horticultural enterprise.

Horticultural development may also simplify the environmental matrix which underpins ecological functioning and biodiversity at regional and larger scales (Woinarski 1999b), hence having impacts which percolate beyond the fenceline of horticultural properties.

More diffuse impacts also occur through use of water and pesticides. When horticultural development is reliant on use of natural water sources, this may result in reduction in biodiversity well beyond the horticultural enterprise (Kingsford 2000). Application of pesticides within horticultural areas may affect natural biodiversity within and around the horticultural development or even, in cases such as the control of plague locusts, in far removed rangeland areas.

Horticultural development may also result in a broad range of native species (such as macropods, magpie geese, cockatoos, rodents, flying foxes, herbivorous insects) being treated as pests, and hence targeted for population reduction. In some cases, this may simply reverse an “artificial” increase in abundance (for species which benefit from the resources offered by horticulture); but in other cases, localised destruction of these native species may produce a net range-wide population decline (Vardon and Tidemann 1995; Tidemann *et al.* 1997).

## Pesticides

There is remarkably little information available about the current extent of use of pesticides in the rangelands, nor of the unintended impacts upon biodiversity of this use. The use of pesticides in the Australian rangelands generally varies more or less inversely with intensity of land use. In the agricultural fringes of the rangelands (such as in central Queensland and the Murray-Darling Depression), there is a relatively high rate of application of pesticides, particularly for the control of horticultural weeds, insects and occasional plagues of house mice (e.g., Eldershaw 1996). In the more remote rangelands, pesticide use is far less intensive and restricted mainly to the use of baiting for dingoes (on pastoral lands) and other feral predators (on some conservation lands: Short *et al.* 1997), with intermittent broad-scale application of pesticides for plague locusts. In some other rangeland areas, a range of herbicides is used to control woody weeds.

The impacts upon non-target species are rarely recorded following application, but some research studies have documented likely responses (McIlroy 1981*a*, 1981*b*; 1982, 1984; King 1989; McIlroy and Gifford 1992; Martin *et al.* 1994; Arthington 1996), and there have been occasional forensic studies of unintended casualties or assessments of the extent of this mortality (Bird 1996; Water and Rivers Commission 1998; Brown and Lundie-Jenkins 1999). The inadvertent destruction of non-target species has been recognised, at least anecdotally, as a problem since the application of poisons (such as strychnine) in the early years of pastoral settlement in the rangelands (e.g., Barnard 1925).

There is a little, but mostly anecdotal, evidence that pesticide use has led to the decline in abundance and distribution of some non-target species in some rangeland areas, for example for the western blue-tongued lizard *Tiliqua occipitalis* in the Murray-Darling Depression bioregion (Robertson *et al.* 1989) the plains-wanderer *Pedionomus torquatus* in the Riverina and Murray-Darling Depression bioregions (Baker-Gabb *et al.* 1990), and owls in north-eastern Queensland (Young and De Lai 1997).

Conversely, where pesticides have been used to control feral predators, there are many examples of recovery of native faunal species (Kinnear *et al.* 1988, 1998; Short *et al.* 1997). However, this relationship may be complicated, as the use of pesticides to control dingo numbers may result in increases in the numbers of feral foxes and cats, to the detriment of biodiversity (Newsome 1993).

There are also direct intended impacts of pesticides on native rangeland biodiversity, through the broad-acre treatment with herbicides of native “woody weeds” (Burrows 1973; Moore 1973; Anderson 1984), baiting for native “pest” species, such as wallabies and cockatoos (Gooding and Harrison 1952; Crossman 1984), and aerial spraying of plague locusts.

The use of some of the more pernicious, persistent and broad-spectrum chemicals previously used for pest control in the rangelands (e.g. Luci-jet, arsenic, phosphate) is now generally regarded as less acceptable, and many agencies and landholders are using more specific and/or less enduring poisons (Allen *et al.* 1984). Nonetheless, known broad-spectrum pesticides (e.g. pindone, 1080, fenitrothion, strychnine, glyphosate) continue to be used in many rangeland areas, undoubtedly with substantial at least local unintended losses to native biodiversity.

## Changed water regimes

Availability of water has been a major preoccupation of, and prerequisite for, human activity in the rangelands. Pastoralism and, to a lesser degree, mining enterprises, have demanded reliable access to water supplies. Initially, this access was limited to naturally occurring sources, such as rivers, lakes and mound springs. Concentration of pastoral activity around these natural water sources led to major degradation of water quality, the condition of aquatic and riparian vegetation, and availability of the water itself (Barnard 1925; Bauer 1959; Riddett 1990). In many rangeland regions, this unrestricted use by livestock of naturally occurring water sources continues, with ongoing degradation.

However, across much of the pastoral rangelands, artificial water sources (bores) have replaced natural sources (James *et al.* 1999). The proliferation of these artificial water sources has led to:

- some respite for natural water sources and their margins;
- expansion in range or abundance of some water-dependent fauna (e.g. crested pigeon, galah) (Williams and Wells 1986), possibly with resultant decline in competing species;
- more pervasive use of the landscape by livestock and feral stock, with resultant decline in range and abundance of a substantial proportion of native biota;
- (in some rangeland areas), changes in the water table, resulting in reduced flow to natural water sources;
- (for calcrete aquifers) increased vulnerability of the specialised and localised stygofauna;
- (in some rangeland areas), changes in the water table, leading to salinisation and broad-scale vegetation change (Landsberg *et al.* 1997; Cox and Barron 1998; Noble *et al.* 1998).

On the mitchell grasslands of the Barkly Tablelands, Fisher (1999) calculated that 80% of land is now within 5 km of water points, and only 1.5% is beyond 8 km from water points. Based on biosphere relationships of abundance of individual species, he calculated that this increase in water sources had led to declines in regional population of 36-85% for “decreaser” animal species (including singing bushlark *Mirafra javanica*, little button-quail *Turnix velox*, long-tailed planigale *Planigale ingrami*, the lizards *Delma tincta* and *Ctenotus pulchellus* and the snake *Demansia torquata*) and 42-84% for “decreaser” plant species (including *Astrebla squarrosa*, *A. pectinata* and *A. elymoides*).

In some rangeland bioregions (notably Victoria-Bonaparte, Brigalow Belt North, Murray-Darling Depression, Cobar Peneplain and Darling Riverine Plains), natural water sources have been used to provide irrigation for cropping. This has vastly altered the flow characteristics of many rangeland waterways, typically leading to lower minimum flow levels, fewer flood events, decreased water quality and/or changes in the riparian

vegetation (Thoms and Cullen 1998; Kingsford 2000), although in some cases also including more permanent impoundments (Briggs 1994). Further major irrigation schemes have been proposed for additional rangeland river systems, in the Channel Country and Dampier Lands bioregions (Kingsford 2000).

In part due to increased water allocation and other changed water regimes, to grazing and to cropping, swamps of some rangeland bioregions have been drained or degraded (Briggs 1994; Seddon and Briggs 1998; Kingsford 2000), at the expense of some animal and plant species (Briggs 1997), including some waterfowl and the painted snipe (Briggs 1994; Kingsford and Thomas 1995; Lane and Rogers *in press*). The deliberate modification of natural wetlands to form artificial ponded pastures is recognised as a major threat to aquatic biodiversity, and is discussed above in the sections on weeds and on wetlands.

The extent and effects of salinisation in the rangelands is addressed in detail elsewhere in the Audit (Theme 2), and hence will not be considered here. Theme 1 of the Audit also includes a far more comprehensive treatment of water use in the rangelands and its environmental consequences.

## Climate change

The Australian rangelands have suffered through periods of rapidly shifting climatic conditions over the course of the Plio-Pleistocene, with a general trend towards increased aridification, at least in central Australia. This climatic dynamism continued through the Holocene, with generally wetter climates developing in northern Australia over the period 20,000 to 10,000 years before present (Nix and Kalma 1972). This erratic climate sifted much of the biodiversity, leading to extinctions for much of the least resilient biota, substantial changes in the distribution and characteristics of many environments, and marooning many species in refugia which provided some relief from generally hostile climates now prevalent across much of their former range. The rangeland climate is still characterised as much by its variability as by any other feature (McKeon *et al.* 1998), with this variability in part associated with contrasts in the ENSO cycles (e.g., Nicholls *et al.* 1996).

Notwithstanding the climatic variability in the rangelands, there is ample evidence that climatic patterning is a paramount influence on the structure and composition of the rangeland biota. For example, a recent study of Mitchell grassland throughout its environmental range in Queensland demonstrated that the primary and secondary floristic gradients were clearly aligned with the macroclimatic gradients related to mean annual rainfall and temperature (Fensham 2000). Hence, it follows that shifts in global climate will have major impacts on species distributions.

The rate of climate change is now expected to accelerate substantially, due to increased rate of greenhouse gas emissions. Rangeland scenarios vary and are very imprecise predictions (Bouma *et al.* 1996; Howden *et al.* 1998), but plausible scenarios include doubling of atmospheric CO<sub>2</sub>, increases in mean temperatures of around 1 to 3°C, marginal decreases in rainfall, and increases in extreme weather events (Climate Impact Group 1992; Bouma *et al.* 1996; Hennessy *et al.* 1997; Hall *et al.* 1998, 1999). Such changes are likely to lead to increased vegetation growth, at least for some plant species and some rangeland areas (Hall *et al.* 1999). Such changes will clearly benefit some plant species (potentially including many weed species) but disadvantage others (Howden *et al.* 1999a), and will probably lead to changes in fire regimes, vegetation structure and phenological patterning. In turn, these vegetation changes will affect faunal biodiversity. Both plant and animal distributions may also be affected by changed availability of water across the rangelands, due to climate change. These hydrological changes are likely to include changes in river flow, flood frequency, and nutrient and sediment inputs, and the extent of this change is likely to be greatest in arid and semi-arid rangeland areas (Watson *et al.* 1997).

But climate change will also have a direct impact upon individual species, particularly for those operating with relatively narrow envelopes of thermal tolerance (Howden *et al.*

1999b). For some species such as *Eucalyptus* that have poor dispersal abilities rapid climate change could be cataclysmic (Hughes *et al.* 1996). Twenty-five percent of *Eucalyptus* species have a current mean annual temperature range of less than 1<sup>0</sup>C, and 53% have a range spanning less than 3<sup>0</sup>C. While the climatic tolerances of some species may be wider than their current range, it seems likely that many species will not be able to shift as rapidly as the climate. Chapman and Milne (1998) predicted changes in the distribution of several rangeland plant and animal species under a range of climate change scenarios, based on models of the climatic characteristics associated with their current distributions. For some species, such as the Kowari *Dasyuroides byrnei*, an endangered dasyurid mammal, almost all climate change scenarios considered led to catastrophic declines in abundance and distribution. Of the five other rangeland species considered, they predicted substantial and significant contractions for the northern small-eyed snake *Rhinoplocephalus pallidiceps* and endangered golden-shouldered parrot *Psephotus chrysopterygius*, smaller contractions for the Ulysses Butterfly *Papilio ulysses*, and curly mitchell grass *Astrebla lappacea* and little change for the laughing kookaburra *Dacelo novaeguineae*.

The rangelands may be particularly vulnerable to the effects of climate change because they offer relatively little topographical relief across very extensive areas, and hence provide little options for species to move locally or regionally to select more preferred climatic conditions. The obvious exceptions to this general flatness are the rocky ranges of central Australia, the Kimberley and Pilbara, western Arnhem Land and around Mount Isa. Such landscapes will have an increasingly important role as refugia for rangeland biota (Morton *et al.* 1995a). However, there are many rangeland species for which these rocky areas will offer no succour as they provide no suitable habitats.

## CONCLUSIONS

Much of the natural essence of Australia is within the rangelands. The rangeland environments and biota make up a large proportion of what is most distinctive about this continent. A relative lack of intensive development has served to help retain much of the character of these areas. But more insidious factors have operated over almost all of the rangelands, re-working the ecological processes which underpin their environments. Some of the resulting changes are conspicuous; many others are more subtle. The nature of change is often difficult to detect, because we have an almost derisively imperfect baseline description of how things were before European settlement, because in many areas our contemporary information base is little better, and because much of the rangeland biota is inconstant and fluctuates naturally in abundance or distribution in response to climatic pulses operating over at least decadal scales.

In this review, we recapitulate the dismal fate of native mammals in the semi-arid and arid rangelands, the most odious biodiversity blight inflicting Australia, and probably the world, in modern times. But this is not simply an event of the past, to be now regretted. While the rate of species loss has declined in the arid and semi-arid rangelands, this is largely because the residue of species still present now includes a high proportion of relatively resilient species. Some mammal species continue to decline in these areas, and the phenomenon of mammal decline has spread across an increasing proportion of the rangelands, now including tropical northern Australia. While the great shock of mammal decline occurred several decades ago, at a time when few cared about conservation and when ecological knowledge was probably insufficient to prevent the losses, such negligence and ignorance cannot now be used as an excuse to accept the ongoing declines.

There is increasing evidence to suggest that many rangeland bird species are also declining, and some rangeland bioregions have undergone major shifts in bird species composition. The argument has been presented (Recher and Lim 1990; Reid and Fleming 1992; Recher 1999) that the relatively subdued rate of bird decline recognised to date is a false hope, masking out a high rate of local extinctions which will soon coalesce to regional and absolute extinctions. To this argument of increasing rate of decline of birds, may be added the argument that longevity of much of the woody rangeland vegetation has also temporarily hidden the reality of inexorable major vegetation change due to lack of plant recruitment (caused mainly by grazing, especially by rabbits and sheep).

This wholesale vegetation change (and its linkage to other biota) for the rangelands of at least the southern arid zone was described by Reid and Fleming (1992) as

*“the lack of effective regeneration of perennial shrubs and trees ... is like a time bomb quietly ticking away; as this aging generation of chenopods, mulga, western*

*myall and native pines thins and dies out, we will witness unprecedented changes in bird community composition”,*

and by Lay (1986) as

*“in many areas we are now witnessing the dramatic transformation of woodlands to degraded annual pastures and inedible shrubs”.*

To some extent, this problem has been at least temporarily halted by the recent control of rabbits with RCD (Neave 1999), but this may be only a short-term respite, and the combined effects of livestock and other feral animals may continue to drive environmental change across the rangelands.

This review demonstrates that it is possible to describe very broadly those components of the rangeland biota which are most susceptible to change, and those factors which are most likely to cause change in biodiversity. But such generalisation overstates some of the patterning. For example, there remains a series of competing hypotheses which have been used to explain decline in the mammal fauna of the semi-arid and arid rangelands, and none of these hypotheses adequately predicts the current decline in the rangelands of northern Australia. A multiplicity of factors is involved, with many of these operating synergistically or idiosyncratically affecting different species or in different bioregions.

There are few studies which have clearly related particular putatively threatening factors to responses by biodiversity, an inadequacy which is magnified by the variability in impact and response due to the climatic inconstancy of the rangelands. Without knowledge of how different factors contribute to change, any monitoring program for rangeland biodiversity will be blunt. And any monitoring program which relies mostly on measurement of either the putatively threatening factors (“pressures”: OECD 1993) or the management “responses” to these factors, rather than the components of the biodiversity itself (“condition” or “state”), will be rooted in a fairly illusory and infirm foundation.

Rather, a more balanced monitoring mechanism for the state of rangeland biodiversity will need to report on, at least, the most symptomatic components of the biota, including:

- the species composition, abundance and distribution of native mammals (particularly bandicoots, small macropods, large dasyurids, possums, and large rodents);
- the species composition, abundance and distribution of native birds (particularly granivores, but also including a broad range of other foraging groups, especially those that nest, roost or forage on the ground or in low understorey); and
- the floristic composition and age structure of most rangeland vegetation types,

with more specific targeted reporting for exploited species. To make sense of any trends, this reporting of aspects of the biota needs to be linked directly with information on the occurrence of “pressures” at all monitoring sites, and backed up by carefully designed research which seeks to investigate how the biota responds to particular threatening processes.

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