GRAZING IMPACTS IN RANGELANDS:
Assessment of two contrasting landscape types in arid Western Australia from different land management perspectives

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DECLARATION

The work presented in this thesis is my own. Specific contributions by other have been referred to in the text and acknowledgments.

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ABSTRACT

Regional patterns of land use and systems for assessing the 'health' of rangelands have not changed their focus from production of meat and fibre, despite ecologically sustainable development becoming the dominant environmental paradigm. In this study I evaluated i) traditional range condition assessment as practised in Western Australia, and explored its capacity to predict changes in: ii) biodiversity; and iii) landscape patterns and processes along grazing gradients in two contrasting landscape types.

I estimated grazing history using a commercially available model and a model based on sheep track density. Although distance from water was a more simple proxy for grazing history, it did not account for as much of the systematic variation in environmental response as either model.

Range condition declined with increased grazing in both landscape types, and while the decline was more marked in mulga landscapes, range condition assessments may overestimate degradation in this landscape.

Approximately one quarter of plant species were 'decreasers' that declined in abundance with increased grazing, compared to 16 per cent that were 'increasers'. The number of decreaser species also declined with grazing, more rapidly in mulga landscapes than chenopod landscapes.

Over a third of ant species (not including singletons) were decreasers in chenopod landscapes. Increaser species in chenopod landscapes, and both decreasers and increasers in mulga landscapes, comprised a relatively minor component of the ant fauna. The sensitivity of the ant fauna to grazing in chenopod landscapes contrasts strongly with conclusions from previous studies in North American rangelands.

Overall, grazing suppressed the capacity for landscapes to redistribute and conserve resources in fertile patches. Chenopod landscapes were most profoundly altered: with increased grazing fertile bush mounds decreased in density and proportional area as open areas expanded and became more saline.

While range condition was correlated with some variables that were important from perspectives of nature conservation and natural resource management, it did not reveal contrasting responses to grazing within and between landscape types. Thus it does not convey the specific nature of impacts on aspects of biodiversity and ecosystem function. Nor

does it help pastoralists manage different landscapes. I present an alternative, hierarchical system of assessing rangelands that integrates concepts of non-equilibrium ecosystem dynamics, ecosystem drivers and engineers, and focal life forms. It also allows for the incorporation of emergent properties for broader scale (i.e. remotely sensed) assessment and monitoring.

On the basis of which model of grazing history best predicted response variables, I was able to interpret grazing impacts spatially. The RANGEPACK/Paddock model estimated a near linear decline in grazing with increasing distance from watering points in my region, while the track-density based model predicted very intense grazing that dissipated rapidly away from watering points.

The spatial distribution of impacts on range condition and floristic diversity may be widespread, while impacts on ants and landscape pattern and process may be more concentrated near watering points. The interests of pastoralists and natural resource managers might be best served by an increase in the density of watering points, while the interests of nature conservationists might be better served by exactly the opposite. The most practical reconciliation may be to improve the reserve network, rather than trying to integrate production and general conservation objectives at a paddock scale, particularly for mulga landscapes.

Finally, I see an urgent need for more cohesive land use planning and rangeland assessment, involving a blurring of institutional boundaries in regionally nested adaptive environmental management systems.

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CHAPTER 1. MANAGEMENT OF RANGELANDS IN THE ARID SHRUBLANDS OF WESTERN AUSTRALIA

1.1 Introduction

Rangelands are areas where primary productivity is too low to support a more intense use of land than the extensive production from livestock at low densities on overwhelmingly native pastures (Wilson and Harrington 1984). Rangelands are 'station' or 'ranch' country, although there may be other land uses within the region. This thesis is about how station management affects the biophysical environment as judged from traditional and more recent perspectives.

In this chapter I set the scene for subsequent chapters. I commence with a brief review of management of rangelands in the arid shrublands region of Western Australia. I then summarise some of the major concepts that have recently driven policy concerning environmental management. I do so within a rangeland context, so that I can then compare current management of rangelands with the objectives implicit in these more recent policies.

I have adopted a Pressure-State-Response approach in this study, which is consistent with state of environment reporting in Australia (State Of Environment Advisory Council 1997). In Chapter 2 I describe my survey design and methods of data analysis. Chapter 3 describes how I derived estimates of grazing history at study sites (Pressure). Subsequent Chapters 4 to 8 describe grazing impacts on the biophysical environment from different perspectives of land management (State), while Chapter 9 is a synthesis which includes suggestions as to how assessment of rangeland 'health' and rangeland management might be improved (Response).

1.2 Management of rangelands in the arid shrublands of Western Australia

1.2.1 The eco-region

The arid shrublands of Western Australia extend from the Exmouth Gulf in the north-west, in a generally south-easterly band through to the Nullarbor Plain and the South Australian border. The climate is semi-arid in coastal and southern parts, and arid in the remaining major part of the region (Burnside *et al.* 1995). The region is bounded to the south and west of the State by cropped lands and to the east by largely uninhabited desert (Figure 1.1). Three broad vegetation formations predominate. Spinifex (most commonly *Triodia*

basedowii) hummock grasslands occur on sand plains, acacia shrublands and low woodlands on plains with red earth soils and chenopod low shrublands on variously saline and calcareous soils, most extensively in lowest parts of landscapes. On the karst (limestone) landscapes of the Nullarbor Plain chenopod shrublands predominate throughout (Beard 1990).

1.2.2 Land use

1.2.2.1 'Conquering the land'

Pastoralism has been the most extensive European land use in most areas for more than a century. It has usually involved the grazing of native vegetation by sheep. Pasture improvement with exotic species has not become widely established, although an exotic grass, *Cenchrus ciliaris* (buffel grass) has become established in river and coastal systems in the north-west of the region.

Pastoralism began as shepherding from the west coast, moving inland along river frontages in the 1860s and has been dominated by sheep rather than cattle. Further inland, the industry developed initially to support food requirements of prosperous mining centres around the turn of the twentieth century, expanding away from such centres to include almost all lands in the region during the 1920s and 1930s (Burnside 1979).

Pastoralism has been a relatively stable land use in contrast with episodic mineral booms characterised by scattered outbreaks of intense mining activity. Areas on the periphery of the pastoral zone and extending into the more arid interior have been designated Aboriginal reserves. Reservation for nature conservation has also been a residual land use for similarly unproductive areas (Pringle 1995a). The overwhelming pattern has been development of station country with intense mining activity in scattered and small areas along greenstone belts.

Nature conservation has historically been a low priority. Pastoralists were a wealthy and politically influential part of Western Australian society. Conservation may have been dismissed as an unfortunate idiosyncrasy of a spoiling minority who failed to share the frontier 'conquer the land' philosophy that has characterised the development of Western Australian since European settlement (Webb 1993).

1.2.2.2 Current land use

Pastoral leases still occupy approximately 85 per cent of the region. Individual leases vary in size, but most are between 100 000 and 250 000 ha. Leases are fenced into smaller paddocks, which are usually square or rectangular management units with fences running north-south and east-west, and watering points supplied by bores or wells in corners. A typical paddock is approximately 8 km square (6 400 hectares), though this may vary due to topographic features such as hills or salt lakes (Pringle 1994a).

Sheep remain the main livestock in the region, though some cattle are also run. Sheep are generally mustered only once a year, when they are shorn and returned to paddocks. There is little tactical management (i.e. movement of stock in response to seasons), partly because of the high costs involved in mustering large paddocks with few sheep, and poor commodity prices. However, old holding paddocks that were used to move stock in response to seasonal conditions are scattered through many stations, and bear testament to a more intensive and tactical form of stock management in the past (Lod Hurst, personal communication).

Livestock grazing is no longer the main land use on some pastoral leases. An increasingly common occurrence is for 'stations' to be purchased by non-traditional stakeholders. Considering the declining terms of trade in the pastoral wool industry, it is not surprising that stations are being acquired for reasons other than pastoral production (Anon. 1993b; 1995). At the forefront are mining companies and Aboriginal groups. While some of these more recently purchased leases continue to be run as traditional stations, many are not. Mining companies may acquire pastoral leases in an effort to increase resource security and to minimise conflicts with local pastoral enterprises. Aboriginal groups have often attempted to continue pastoral enterprises, but have had varied success, possibly because pastoralism was not the intended use of acquired stations in some cases.

Mining is now the greatest contributor to the economy in the arid shrubland rangelands. In 1991/1992 the estimated value of pastoral production in the study area was $5M, compared with $2 356M from the mining industry (Holm and Friedel, M. 1996).

The nature reserve system in this region is inadequate (Specht, Roe and Boughton 1974; CALM 1992; Pringle 1995a). No systematic approach has been implemented to develop a comprehensive nature reserve system, although methods to do so have been proposed and surveys have been conducted to support implementation (Biological Surveys Committee 1984; McKenzie 1984; McKenzie *et al.* 1989; Pringle, Van Vreeswyk and Gilligan 1994;

Keighery, Milewski and Hall 1995). Appreciation of the inadequacy of the reserve system is not a recent revelation (Specht, Roe and Boughton 1974). Rather, the inadequacy may reflect the political influence of competing stakeholders; namely the pastoral and (more recently) mining industries and their concerns over access to resources.

1.2.2.3 Future issues

Mining companies and Aboriginal interests are likely to become increasingly major holders of pastoral leases, not necessarily with a view to running traditional station enterprises. Traditional pastoralists are likely to become increasingly interested in enterprises such as ecotourism and horticulture, in an effort to diversify to a more secure economic future. Conservationists are also turning their attention to the outback, usually commenting on the inadequacy of the nature reserve system and the extent of degradation of pastoral lands (Morton 1993; Siewart 1993; Auty 1994). Thus, the social landscape is a rapidly changing one and forces that were not present in previous economic downturns will test the culture of resilience that has historically characterised the pastoral industry.

There are indicative features of major social change in the study area. Firstly, few stations have effective programs to renew ageing infrastructure. Most stations were fully developed in a few years, and 60-80 years on, many fences and other infrastructure are nearing the end of their design life (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993). This is happening at a time when pastoralists cannot afford to re-invest in infrastructure. Secondly, there is a trend for pastoralists' children to seek careers other than pastoralism, so that when pastoralists retire they sell their leases (personal observations). Not only is pastoral management becoming more difficult due to ageing infrastructure, long-established pastoral families with decades of experience are leaving the industry and the district. The traditionally strong social fabric of the pastoral industry is unravelling (Abel and Ryan 1996).

The challenges facing the pastoral industry have not gone unnoticed. In recent years, Commonwealth and State Governments have invested heavily in land-use planning in the region. Two costly land-use projects have commenced in my study region. 'RANGEWAYS' is a collaborative program between researchers and local stakeholders that is investigating participative approaches to land-use planning. One of its tools is 'LUPIS', a software package that allows for ongoing multi-stakeholder input to planning land use scenarios (Ive, Cocks and Parvey 1989). More recently, the Gascoyne Murchison Strategy has seen industry-led re-planning commence over much of the region. As yet, changes have not

flowed through to lease adjustments. With a budget running into tens of millions of dollars, this will surely happen soon (Roger Jaensch, personal communication). Up to twenty pastoral leases ('stations') are likely to be purchased by the Department of Conservation and Land Management to develop the first systematic, regional network of nature reserves in Australian rangelands (Tony Brandis personal communication).

The public investment in these regional programs is quite different to traditional Government responses to downturns in the pastoral industry, when pastoralists were generally subsidised in order to survive (Wilcox and Burnside 1994). Now the Government is funding change, including rationalising the pastoral industry through lease re-adjustment and acquisition, rather than maintaining the *status quo*. This represents a radical shift in policy. The shift is from trying to re-enforce stability in systems of rangeland use by subsidy (i.e. investing in resistance to change), to building resilience based on the capacity of these systems of rangeland use to adapt to external forces (Gunderson, Holling and Light 1995; Abel 2000).

Native Title is another major issue in contemporary rangelands. Under current Government policy and the 1993 Native Title Act, local indigenous people have right of access and can apply for resource use on stations, so long as these actions do not interfere with station management (the legally superior land use). What this actually means on a paddock or station basis is still unclear and adds to pastoralists' uncertainty (Brian Money personal communication). On the other hand, many indigenous groups are dissatisfied with the subservient status given to their rights in favour of pastoralism (Bachelard1997). However, there is an ongoing program of pastoral lease acquisition on behalf of indigenous groups, adding to the changing social and cultural landscape.

In summary, the management of the rangelands is fast changing, partly driven by gloomy terms of trade in the traditional pastoral industry, and partly due to social and cultural changes. The rangelands are currently experiencing strong external forces unlike those that have gone before.

1.2.3 Environmental assessment of stations

1.2.3.1 Early assessments: using the land

Environmental assessment from the late nineteenth century through to the 1970s was primarily concerned with rates of pastoral development; the state of the biophysical environment was less important. Pastoralists were considered to be meeting the terms of their leases if they were following their development plans. This often meant maintaining and

expanding infrastructure to open up and exploit more of the resource on the lease. Original lease conditions imposed from London referred to responsible resource use and questions of overgrazing were sometimes asked (Webb 1993). The *Land Act (1933)* and the *Soil and Land Conservation Act (1945)* also included stipulations regarding the maintenance of the resource on stations. At the same time however, minimum stocking rates and limitations on de-stocking are integral to the former Act, and they were enforced more rigorously than were restrictions aimed at maintaining environmental standards (Anon. 1991).

The pastoral industry developed without inventories of the resources or assessments of land capability, with the full support of Government. Any failure to exploit a resource was frowned upon, and was certainly in breach of lease conditions. Government must therefore carry some responsibility for any historical degradation of rangelands by pastoralists in the development of the industry (Anon. 1991). Government encouraged an extractive approach to management of renewable resources. The financial benefits of 'pastoral mining' were clearly visible in terms of pastoralist-owned prime real estate in Perth and an increasing demand for private boarding school education (Webb 1993). It is only recently that society has questioned the environmental costs of this extractive approach (Anon. 1991; Heathcote 1994).

1.2.3.2 Range condition: the first ecological perspective

In the 1960s, and more particularly in the 1970s, district advisers employed by Agriculture Western Australia began promoting concepts of conservative pastoral management and maintenance of range condition, with varied success. The aim of this extension was to encourage pastoralists to look more closely at the condition of the land rather than relying on the condition of their stock to make management decisions. In particular, pastoralists were encouraged to assess perennial pasture plants and potential 'woody weed' species. The aim was to discriminate between these species to assess the condition of the land and to use the prevailing abundance of ephemeral plant species to assess seasonal condition. It was hoped that such an approach would help reverse the history of overstocking that many areas had experienced (David Wilcox, personal communication).

Wilcox published the first systematically conducted regional assessment of range condition and overgrazing in Western Australia, in the Gascoyne River catchment (Wilcox and McKinnon 1972). He recommended stock reductions by half, was publicly vilified, and did not receive much support from the Pastoral Board or Agriculture Western Australia (Pringle 1991). It is ironic that the survey was conducted to investigate whether overgrazing was

causing increasingly frequent flooding of Carnarvon townsite by the Gascoyne River in the 1960s, rather than concerns about range resources *per se* (Pringle 1991).

However, the survey highlighted the need to assess grazing impacts and provide better information to pastoralists regarding the range resources and their limitations. Range condition was subsequently adopted for environmental assessment in regular pastoral lease reporting, and the developing rangeland surveying program undertaken by Department of Land Administration and Agriculture Western Australia (Pringle 1991).

The Pastoral Board and Commissioner for Soil and Land Conservation have increasingly demanded and acted on range condition assessments, and pastoralists have responded positively to extension efforts promoting conservative management and the maintenance of palatable perennial species and soil health (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Wilcox and Burnside 1994; Pringle and Riches 1996).

Agriculture Western Australia has also conducted rangeland monitoring on many stations in the region for at least a decade. Sites are generally revisited every five to seven years. Originally developed as an aid for pastoral management, the Western Australian Rangeland Monitoring System (WARMS) is now part of a formal Government system to report on range conditions at a regional level (Holm 1993). Assessment involves counting and measuring perennial plant species. More recently, assessment of soil surface condition and landscape function analysis has been incorporated into WARMS (Tongway 1994; Tongway and Hindley 1995). However, the system has not been running for long enough for any conclusive assessments to be made of long-term trends. Thus separating grazing from other sources of variation (e.g. rainfall and soil) has been problematic (Alec Holm, personal communication).

1.2.3.3 Recent initiatives in rangeland assessment

The need for assessments of rangeland conditions beyond their pastoral value has recently been recognised (Wilcox and Burnside 1994; Pringle, Payne and Burnside 1995; Pringle and Burnside 1996). This has been addressed in two ways. Firstly, rangeland inventory is done in two stages. It is commenced without any presumption of a particular land use. So for instance, ‘pasture types’ are now termed ‘site types’ or ‘habitat types’ (Pringle 1994b; 1998b). This has been a relatively easy transition because the hierarchical criteria used in classifying rangelands have, from the outset, been physiographic, edaphic and botanic; rather than pastoral. Interpretive reports are then produced for specific clients, such as pastoralists or the mining industry. These are much briefer than the full inventory, and styled and

interpreted to facilitate uptake by the targeted audience (Pringle 1994a; Van Vreeswyk and Godden 1998).

The second major change has been in the way impacts of grazing are assessed. Given the paucity of reserved areas and in recognition of Government policies to improve nature conservation (Anon. 1987), it was decided to make assessments of grazing impacts relevant beyond pastoralism (Pringle, Payne and Burnside 1995; Pringle and Burnside 1996). While traditional, pastorally focussed assessments are still conducted, other, more ecological observations are now included. These include the presence of exotic species and whether perennial plant cover has been reduced or increased beyond what might be expected in analogous, undeveloped landscapes. This can avoid misleading interpretations where exotic species have invaded and maintained or improved pastoral productivity, but perhaps compromised nature conservation values (Pringle 1998a). A conservation biologist would certainly want to know whether an area was in 'excellent range condition' because it was little modified, or because it was a chenopod shrubland transformed into an exotic grassland. The rangeland survey of the Pilbara region (now in progress) is the first adoption of this expanded approach.

1.3 New perspectives in environmental management

The two main conceptual developments in environmental management are increasing recognition of ecological complexity (particularly with respect to scale and ecological hierarchy (Allen and Starr 1982; 1986; Allen, O'Neill and Hoekstra 1984; O'Neill *et al.* 1986; Brown and Allen 1989; Allen and Hoekstra 1992) and multiple values (Cohen 1992; Brunson 1992; Kessler *et al.* 1992; Abel and Ryan 1996). Management of rangelands has a tradition of focus at one scale; the plant community, and from one perspective; pastoralism. Newer approaches can be recognised in the current dominant paradigm of environmental management, ecologically sustainable development (ESD) (State Of Environment Advisory Council 1997; Anon. 1999). The process of implementing the principles of ESD has been termed ecosystem management (Kessler *et al.* 1992; Grumbine 1997; Lackey 1999). Below I address some of the key perspectives in this recently established paradigm of environmental management.

1.3.1 Ecological sustainability and ecosystem management

Ecologically sustainable development (ESD) principles have been widely endorsed at international, national, State and regional levels as a framework for integrating sustainable resource use with nature conservation. They were developed to reconcile development with nature conservation in the face of increasing societal demands to both exploit and protect natural resources (Anon.1980). Five important objectives in the State Conservation Strategy for Western Australia (Anon. 1987) are:

(1) to maintain essential ecological processes and life-support systems;

(2) to preserve genetic diversity;

(3) to ensure the sustainable utilisation of species and ecosystems;

(4) to maintain and enhance environmental qualities; and

(%) to optimise the quality of life for Western Australians

The first three objectives have been adapted from the World Conservation Strategy (Anon. 1980), and the fourth from the (Australian) National Conservation Strategy (Anon.1984). The fifth was adopted specifically for Western Australia. While these principles have been widely accepted, issues of defining appropriate land management objectives, and scales of assessment and implementation have dogged their implementation. In rangelands these issues give rise to a number of specific questions. What objectives should apply within paddocks, stations, districts, regions, States and so on? Must every plant population be preserved, or can compromises be allowed to accommodate resource use? How do we integrate expectations into assessable land management objectives and standards?

Morton and Stafford Smith (1994) discriminate between sets of management objectives appropriate at the level of enterprise (station) and those appropriate at regional scales. Regional objectives include regional maintenance of biodiversity. At the enterprise level, objectives should focus on maintenance of resource condition; range condition in this instance.

The art and science of ecosystem management includes integrating different value systems and their hierarchical structures (Grumbine 1997; Lackey 1998; Lackey 1999). The traditional sustainable yield paradigm emphasised local inputs (subsidy) and levels of production (output), as if activities on land management units (e.g. a station) could not influence the capacity for objectives to be met on neighbouring management units. Sustainable yield was focussed within local human boundaries, and nature conservation was

something that happened in nature reserves. Fences were mental as well as physical boundaries in discrete, self-contained mosaics of human endeavour.

Ecosystem management emphasises fences as boundaries to property rights, not boundaries to landscape processes such as overland flow or soil erosion. Ecosystem management also recognises the need for parts (e.g. stations) to contribute to the whole (e.g. maintenance of viable populations of rare species within a region). It is the blurring of boundaries and the intrusion of external influences that most concerns some traditional rangeland ecologists (e.g. West and Smith 1997). However, blurring the boundaries may offer as many opportunities as it presents threats to traditional pastoralism, particularly if formal off-reserve stewardships can be formalised (Morton *et al.* 1995; Beilin and Fleming 1997; Shea *et al.* 1997).

ESD allows for directing and assessing environmental management against sustainability goals. This is an evolving process, for as ecosystems and human activities change, so will the considerations for sustainability need to be revised in an iterative learning process. For instance, sheep may be replaced by goats, fires may allow exotics to invade nature reserves, facilitated by global climatic change. New land uses may emerge. Any such changes will introduce new issues, aspirations and priorities, which may change expectations, objectives and hence requirements for environmental assessment. There is no 'fixed target' in pursuing ESD.

The important proposition of Morton and Stafford Smith (1994) is that different objectives can be developed and applied for different areas within a region, with a bottom line requirement that biological diversity is maintained at a regional level. While this does not provide an immediate panacea to the problem of implementing ESD principles, it represents one avenue that might promote progress. The question then becomes which objectives are appropriate for which areas?

In Australia's inaugural State of Environment Report (State Of Environment Advisory Council 1997), the following conceptual perspectives were identified as being critical for environmental management of Australia's ecosystems:

(1) That indigenous biota and associated life supporting processes are conserved (i.e. conservation of biological diversity).

(2) That ecosystem patterns and processes are maintained so as to preserve ecosystem function and absorb the effects of human activities (i.e. the maintenance of ecological integrity).

(3) That the long-term productivity of natural resources (especially soils) is maintained (i.e. the maintenance of natural capital).

(4) That the long-term capacity of rangelands to support pastoral production is maintained (i.e. the maintenance of range/resource condition).

The report also emphasised the importance of scale and hierarchical issues: that off-site impacts of activities need to be addressed and that all activities should be considered in terms of their influence on issues operative at both higher (e.g. global) and lower levels (e.g. rare plant populations).

I will briefly discuss each of these key perspectives below.

1.3.2 Biological integrity/diversity

Biological integrity may be defined as the full complement of naturally evolved organisms and the ecosystem processes that sustain them. In this respect *biological integrity* includes both ecosystem processes and their living elements: organisms, communities and so forth (Angermeier and Karr 1994). This definition of *biological integrity* is similar to that of *biological diversity* proposed by Noss (1990). However I am inclined to agree with the former authors, who contend that such a broad definition of *biological diversity* adds ambiguity without utility, and does not overtly embody the idea of natural evolution. For instance, the addition of exotic species may increase diversity, while comparison with naturally evolved conditions would lead to an assessment of reduced integrity. Although a 'natural' prefix is often assumed or implicit in use of the term *biodiversity*, it is not explicit, and is therefore open to misinterpretation.

Biological integrity is very useful in the context of *ecological sustainability* in that it provides a conceptual approach to evaluating how human activity modifies naturally evolved systems. If *ecological sustainability* is taken to include the regional maintenance of biodiversity (Morton and Stafford Smith 1994; Anon. 1999), alterations of naturally evolved conditions can be used to assess the compatibility of various land uses with nature conservation. Such considerations can guide land use planning and the setting of land management goals commensurate with both the land use and regional conservation objectives. For instance,

one might ask whether grazing management represents a major threat to the integrity of a particularly sensitive plant community or to the capacity of ground dwelling birds to breed successfully (i.e. to their breeding habitat). If the answer is 'yes', then questions might need to be asked about the adequacy of reserve systems and the integration of conservation and resource-use goals outside nature reserves. In other words, conservation objectives will probably need to be met in specific areas outside the reserve system in order to meet regional conservation objectives (Morton *et al.* 1995).

The degree to which land use may compromise biological integrity is highly relevant in regions such as the one discussed here, where reserve systems provide an inadequate insurance policy for nature conservation across the region (Specht, Roe and Boughton 1974; Thackway and Cresswell 1995; Pringle 1995a). For instance, in this region there are types of ecosystems in which biological integrity is greatly compromised by pastoralism and yet they are not represented in the reserve system (Pringle 1995a). Examples of all such ecosystems may never be reserved. Planning reserve systems typically involves compromises with competing land uses (Margules, Pressey and Nicholls 1991).

New uses of rangelands may challenge the concept of biological integrity. How do we assess whether a decision to introduce a new land use will compromise future land use opportunities? The answer is probably that we may never really know with confidence. It has been argued that trying to optimise biological integrity on grazed lands will represent the most prudent safeguard in this region (Curry and Hacker 1990; Allison, Blyth and Halleen 1990; Pringle 1995b).

Historical pastoral management has compromised the natural attributes of many landscapes in this region (Wilcox and McKinnon 1972; Pringle 1994a; Pringle 1995a). The usual (rather than rare) occurrence of widespread environmental problems cannot forever be dismissed as a legacy of the past in Western Australia's rangelands (Anon. 1991).

Despite its conceptual clarity, there are some fundamental problems with basing management on the concept of *biological integrity*. For instance, do naturally evolved conditions (the pristine state) still exist for comparison? Do we consider naturally evolved conditions to have prevailed until European settlement, or was the arrival of Australia's earliest indigenous peoples the commencement of 'future eating' (Flannery1995)?

Some naturally evolved ecosystems are highly dynamic, such as those frequently subjected to wildfires. The dynamics become the focus of assessment, rather than the state. It would

be quite difficult to compare patterns of dynamics in exploited and reference areas in such circumstances. This would involve recognising different phases within cycles of disturbance and renewal (Holling 1992; Hansell and Bass 1998) and then trying to assess whether they were in an acceptable domain. Indeed this task was considered too complex for *Triodia basedowii* (hard spinifex) hummock grasslands (which are subject to recurrent fires) during a rangeland survey of part of the study region (Pringle, Van Vreeswyk and Gilligan 1994).

However, it is essential to include considerations of naturally evolved conditions if one is to integrate sustainable use of resources with regional conservation of nature. Where truly pristine benchmarks are not available, much may still be gleaned from the best available (least modified) representative areas. Little modified benchmarks are perhaps more common in arid rangelands than in many other agro ecosystems (e.g. regions dominated by cropping) which have been substantially transformed into different ecosystems (Holzman and Isaacs 1999). In such transformed ecosystems, there may be little value in making comparisons with natural benchmarks (Wicklum and Davies 1995). Even in rangelands, areas beyond the normal influence of grazing management are few and scattered (Fleischner 1994; Landsberg and Gillieson 1996; Holzman and Isaacs 1999); however they provide valuable benchmarks against which to assess grazed lands (Landsberg *et al.* 1999).

1.3.3 Ecological integrity

'Ecological integrity' (or '*ecosystem integrity'*) lacks normative meaning. It has been used as an all-encompassing concept similar to biological integrity (Grumbine 1997). I prefer to adapt the approach of West and colleagues, in which emphasis is placed on the capacity of the soil 'to furnish nutrients and water for plant growth' (West *et al.* 1994). If one integrates this point-based concept with Noy-Meir's ideas about sources and sinks within arid landscapes, (Noy-Meir 1973; 1981; 1985), then a workable definition might be 'the capacity of landscapes to redistribute and conserve resources'. This is analogous to the CSIRO Wildlife and Ecology concept of Landscape Function Analysis (Ludwig and Tongway 1995b; Ludwig and Tongway 2000; Tongway and Hindley 2000). In this respect, the identity of species involved in ecosystem patterns and processes is secondary to the patterns and processes themselves. This is consistent with the use of the concept in the State Of Environment Report Australia 1996 (State Of Environment Advisory Council 1997), but is tailored for rangelands.

As with *biological integrity*, *ecosystem integrity* can be used to refer to naturally evolved processes and be assessed in terms of deviation from naturally evolved conditions.

Alternatively, the term could be interpreted with the understanding that substantial alteration of naturally evolved conditions does not preclude an ecosystem from retaining its capacity to redistribute and conserve resources (West *et al.* 1994; Busby, Ruyle and Joyce 1996; Shachack, Sachs and Moshe 1998).

This is a more pragmatic interpretation: why would one want to preserve a naturally preserved ecosystem's ecological processes while simultaneously removing or modifying its biota in order to implement a land use? Why should the totality of naturally evolved processes be optimal for a new land use? Altered ecosystem processes might still be sustainable, if the transition does not degrade the systems ability to conserve scarce resources and retain resilience (ability to assimilate disturbance) (Holling 1973). This more flexible interpretation of integrity would presumably be similar to 'sustainability', as recommended by Wicklum and Davies (1995).

In the context of rangeland environments, ecological integrity has much in common with the concept of 'rangeland health' proposed by the U.S.A. Committee on Rangeland

Classification as a new, integrative approach to 'classify, inventory and monitor' rangelands (National Research Council 1994). The approach assumes no specific land-use objectives and focuses on ecological processes:

> *'Rangeland health should be defined as the degree to which the integrity of the soil and the ecological processes of rangeland ecosystems are sustained'* (National Research Council 1994 page 34).

Rangeland health has much in common with the concept of ecosystem health (Rapport 1992; 1995) that has also emerged largely in the USA. However, it lacks the integration of human value systems integral to ecosystem health. It can be argued that the human health analogy is unnecessary and may cause confusion when interpreted in the ecosystem health context, which lacks normative definitions (Wicklum and Davies 1995). The health context, as defined by the NRC is very similar to my use of the term ecological integrity, including the acknowledgment of ecosystem complexity, comparison with intact examples, acceptable ecological domains of use and the exclusion of land use specific assessment criteria.

'Rangeland health' might imply a far broader context than ecological functioning simply by association with 'ecosystem health'. However, that is not intended; it is meant to be a strictly biophysical environmental concept in which an acceptable domain of resource use is defined (National Research Council 1994; Whitford 1995; Busby, Ruyle and Joyce 1996).

Rangeland health also does not include biological integrity in overall, regional environmental management of rangelands.

Despite the appeal of a system that assesses the health of rangelands, such approaches were criticised at the Vth International Rangeland Congress held in Salt Lake City in 1995. An integrative assessment based on minimum ecosystem function was proposed (Busby, Ruyle and Joyce 1996), developed largely from ideas in the NRC rangeland health report (National Research Council 1994). The basis of much of the criticism was that land-use objectives were a prerequisite to assessment, and that setting limits on what constitutes 'healthy' relies upon value judgements (Friedel and Laycock 1996).

Alternatively, 'ecosystem health' has its reliance on analogy with human health, and is a somewhat nebulous concept, often including biological diversity (Rapport 1989). It seems to be published as part of 'the new paradigm' mostly in recently established journals associated with it. It is not the underlying paradigm in many articles in mainstream journals that focus on environmental management such as the Journals of Environmental Management, Applied Ecology, Ecology and so forth. Many of its positive aspects, such as recognising values and scale issues are already acknowledged as important in more mainstream concepts such as ESD.

The use of the health metaphor has psychological value: it encourages people to acknowledge that by their actions, they can make ecosystems 'sick'. Ecosystem 'sickness' is far less controversial than ecosystem 'health', because of the convergence of multiple value systems at the sick end of the scale. We generally agree what sickness means in ecosystems: erosion, pollution, algal outbreaks in estuaries, rising saline water tables and so forth. This psychological approach begs a new philosophy of environmental management, based on the idea of humans nested within ecosystems, rather than being outsiders managing ecosystems (Norton 1988; Costanza, Kessler *et al.* 1992; Norton and Haskell 1992). However, these positive aspects still do not distinguish ecosystem health apart from recent developments in more mainstream environmental management. I have therefore seen its inclusion in my study as potentially confusing, rather than being a new paradigm or sufficiently unique concept.

While sustainable production can conceivably be achieved in stable, but degraded rangelands, application of an approach such as ecological integrity would highlight the degradation by comparing degraded rangelands with benchmark conditions. Ecological integrity might then provide a solution to a common criticism of sustainability: 'Sustainability,

but at what capital cost and level of output?' In other words, ecological integrity may be useful as a means of evaluating sustainability of production within the context of future use options (natural resource conservation). This approach emphasises the retention of natural capital; for example the structure, stability and ability of soil to absorb and store water, and the conservation and distribution of nutrients, and the assimilation and conservation of energy. In the context of ecological integrity, indigenous biota and biotic processes are only important in their influence on the 'natural capital' and landscape processes.

On the basis that ecological integrity allows for modifications resulting from anthropogenic activity, it is independent of value judgement about 'naturalness'. A challenge for ecologists is to help develop sustainable management systems that fall within the capacity of rangelands to meet current societal demands of ecological sustainability, and to articulate the likely ecological boundaries within which ecosystems can be exploited in this context.

In other words, *ecosystem integrity* relates to sustaining natural resources rather than sustaining either the 'naturalness' of a system, or the production of a particular land use. In this respect it provides a generic conceptual basis for assessment of all exploited lands. It is also quantifiable, in terms of the state of nominated ecosystem functions in relation to defined (reference) areas. Assessment of multiple ecosystem properties need not produce a single index. Indeed any attempt to do so is likely to obscure ecological issues and rely on the choice of variables to consider, and their relative weighting.

Assessment of numerous attributes independently can produce attribute profiles that will provide information on how each attribute is affected by a particular activity or group of activities in a place or type of ecosystem (Odum 1995; Rapport and Whitford 1999). This will not only provide detail on the nature of human impact, it may also provide insights into management of ecosystems within their capability.

Ecosystem integrity does not have to be classified into 'healthy', 'at risk' and 'unhealthy' as proposed in the rangeland health context (National Research Council 1994). Ecosystem integrity can be restricted to assessment of impacts on stated patterns and processes related to ecosystem function. This is clearly the domain of ecologists. The use of health classes, particularly ones with connotations of ethical value, is not helpful and disguises the complexity of ecosystem dynamics. A challenge to ecologists is to synthesise various types of ecological information into descriptive categories, which can then be assessed according to various value systems (Odum 1995; Pringle and Burnside 1996).

Although ecological integrity is an important concept, it has rarely been systematically applied in assessing rangeland ecosystems. Reference ecological profiles for assessing ecosystem function are frequently lacking (MacDonald 1989; Odum 1995). In reviewing human impacts on rangelands in southern Africa, MacDonald (1989) could only look at impacts on the elements of nature, not their supporting ecological processes. This was because there was almost no process information available.

As well as a paucity of reference data, there is also a deficiency in currently applied methods of assessment to accommodate variations in scale, as is widely acknowledged (Brown and Allen 1989; Friedel 1991; National Research Council 1994; West *et al.* 1994). In Western Australia's arid rangelands, assessment has been focussed on the counting and measuring of plants, often with cruder approximations of soil stability (Holm, Burnside and Mitchell 1987; Pringle 1991; Holm 1994).

A further challenge for ecologists is to link new processes of site-based assessment of ecological integrity with remotely sensed systems of assessment through emergent properties. Methods to do just this are currently being developed in northern Australia by CSIRO Wildlife and Ecology (John Ludwig, personal communication).

1.3.4 Natural capital

Natural capital in a rangeland context refers mainly to the maintenance of the soil resources that support primary productivity (in an ecological sense as well as a pastoral sense). Natural capital is the basis of renewable resources. While changes to vegetation may reduce pastoral productivity - at least in the medium term - the resource remains essentially renewable until physico-chemical deterioration and / or loss of soil takes place (Harrington, Wilson and Young 1984; Milton *et al.* 1994).

In this respect, pastoral productivity may be diminished by a change in vegetation due to grazing, but there has not necessarily been a loss of natural capital if the soil has remained stable and biologically productive. As with ecological integrity, natural capital is not constrained or defined by land-use. Quite clearly there is considerable overlap between ecological integrity and natural capital, and natural capital provides one basis for assessing ecological integrity. In some respects, any loss of natural capital can be traced by monitoring ecological integrity (Milton *et al.* 1994). However, losses of ecological integrity are not necessarily permanent, particularly if soils are minimally affected in the process. In contrast, loss of soil in arid rangelands is generally considered to be irreversible in time frames

relevant for human management (Charley and Cowling 1968; Thurow, Blackburn and Taylor 1988; Fanning 1994).

In this work, I treat ecological integrity and natural capital as one. If a distinction had to be made, natural capital could be seen as the physico-chemical status of the soil and ecological integrity as the suite of patterns and processes that maintain it. One reason why natural capital and ecological integrity are difficult to differentiate between in arid rangelands is that there are few instances of more intense human stresses such as pollution and fertilisation/run-off, that are focal to ecological integrity in more intensely used biomes. Another is that landscape processes and soil fertility are so tightly coupled in arid and semi-arid rangelands (Ludwig and Tongway 1995b; Schlesinger *et al.* 1996; Reynolds *et al.* 1999), that it is not helpful to separate them.

1.3.5 Resource (range) condition

Biophysical attributes become resources when an instrumental value is placed on them (Conacher 1978). Within this context, biological integrity, ecological integrity and natural capital might be classified as components of (biophysical) 'environmental quality' and range condition as a resource quality. The former components are broad concepts without any overt utilitarian character, while range condition reflects the capacity of land to satisfy specified needs (producing meat and fibre). 'Environmental quality' is closely allied to 'deep ecology', while range condition is more closely aligned with 'shallow ecology' (Naess 1973).

These conceptual differences are important. When 'Ecological' practitioners of range condition (Sampson 1919; Dyksterhuis 1949; 1958) have tried to combine them, confusion and philosophical debate have often ensued, particularly in the USA (Smith 1989; Joyce 1993).

In Australia, a different approach has been proposed. It focuses on the specific objectives of land management; and the long-term capacity of the land to produce meat and fibre (Wilson and Tupper 1982). Although this approach has intuitive appeal, its implementation has been problematic. The most obvious problem has been in deciding what best reflects the long-term capacity of the rangeland to produce meat and fibre (Wilson and Harrington 1984; Wilson and MacLeod 1991; Ash *et al.* 1995; Pringle 1995b; Ash and Stafford-Smith 1996).

I discuss these concepts further in introductory sections of relevant chapters in the rest of this thesis. Here I have merely presented a brief review of the fast-changing paradigm of environmental management.

1.4 Management of rangelands beyond pastoral resources

My discussion of traditional (pastorally-focussed) and contemporary management of rangelands in the arid shrublands of Western Australia, and recent paradigm shifts in environmental management, highlights the need for research to support evolution of management of rangelands towards contemporary values and policies.

Two major deficiencies in contemporary management of rangelands exist:

(i) current patterns of land use are residual from the 'conquer the land' philosophy that existed at the time of pastoral development, and

(ii) assessment of grazing impacts on the biophysical environment remains focussed on a single (pastoral) land use and hence 'shallow' (Pringle 1998a).

1.4.1 Re-planning land use

Given that nature reserves are inadequate, what is the best way to improve nature conservation? Some have suggested that closing down the pastoral industry would be the best solution (e.g. Auty 1994), others that pastoralists should become conservation stewards first and livestock managers second (e.g. Siewart 1993). These approaches assume the political will to take such drastic action and the willingness of pastoralists to embrace fundamental change in management objectives. These prescriptions overlook pastoralists' values, for example, the pride some of them have in the genetic improvements they have overseen in their flocks.

A less drastic and more ecological approach has been proposed by scientists in Australia's CSIRO Division of Wildlife and Ecology (Morton and Stafford Smith 1994; Morton *et al.* 1995). It is based on four classes of land:

(1) Reserves: areas set aside for nature conservation.

(2) Excised Management Units (EMUs): areas outside nature reserves that have high conservation value and require complete protection from stock.

(3) Restricted Use Units (RUUs): areas of high conservation value outside reserves, which need particularly sensitive management of grazing.

(4) Sustainable Use Living Areas (SULAs): areas in which sustainable management of the pastoral resource occurs.

A feature of this approach is that it recognises different requirements for nature conservation: in introducing some structural complexity (as opposed to 'reserves' and 'not reserves'), it offers flexibility. The hierarchical nature of concepts of biodiversity and conservation readily lends itself to such an hierarchical, but flexible approach (Noss and Harris 1986; Noss 1990).

For instance, representative fragile landscapes that are susceptible to accelerated erosion and are favoured by stock might be reserved, if they typically occur over a large area. Chenopod shrublands on fragile sedimentary rocks (Jimba land system in the Carnarvon Basin) are an example (Payne, Curry and Spencer 1987). Alternatively, if a similarly grazing-intolerant landscape occurs typically as small, scattered areas, then excision from pastoral use might be more appropriate. Chenopod shrublands on breakaway footslopes (Sherwood land system in the Goldfields region) are good examples of potential Excised Management Units (Pringle 1994b; 1998b). The inland salt lakes of the arid shrublands hold water for more than a few weeks at a time once or twice per decade. When they do, water birds migrate to breed in vast numbers (Chapman, Kealley and Williamson 1994). Perhaps this is an occurrence when pastoralists might block causeways leading onto islands where birds are breeding and help conservation officers control fox numbers. Stock numbers might also need to be temporally reduced. This would become a Restricted Use Unit.

The CSIRO stewardship model described above was an attempt to encourage creative thought on how pastoralism and nature conservation could co-exist with mutual benefit. It is 'naïve' in the literal sense of the word: it is a start. It has an emphasis on mobile fauna and fertile parts of rangeland landscapes. This overlooks the likelihood that many native species and ecosystems beyond these subsets may be grazing-intolerant, as this research group's more recent research has indicated (Landsberg, James and Morton 1997; Landsberg *et al.* 1999).

The other limitation I perceive in the CSIRO stewardship model is that it suggests that 'deep' ecological concepts should not apply to sustainable resource areas (SULAs). This may be a pragmatic compromise in order not to repel pastoral stakeholders from the outset. It suggests 'business as usual' beyond the hierarchy of conservation categories. This is contradictory to endorsed Government policies discussed earlier. For instance, the use of declared environmental weeds such as buffel grass (*Cenchrus ciliaris*) is still advocated for pasture improvement by some pastoralists. However, while such species might improve production of meat and fibre, they may also jeopardise other legitimate values both on and off-site.

The requirements for exclusive conservation lands will be influenced by the perceived 'hostility' of the pastoral matrix. If conservative approaches to pastoral management (ones that aim to maintain the general vegetation composition, structure and cover) are practised, it is likely that much of the native biota will be able to persist off reserves (Curry and Hacker 1990; CALM 1992). This type of approach will enhance landscape connectivity; providing a 'friendly' rather than 'hostile' matrix.

1.4.2 Assessing the state of the rangeland environment

If conservation and resource objectives are integrated into a hierarchical system such as that advocated by Morton and his colleagues (e.g. Morton *et al.* 1995), then a range of land use objectives may apply to each piece of land. The challenge is to allocate these objectives to a manageable number of rangeland use categories.

These categories will need to be hierarchical, in order to reconcile them with the hierarchies of pastoral organisation (watering points within paddocks within stations within Land Conservation Districts and so on) and ecosystem organisation (individuals within populations within communities and habitats within landscapes within catchments and so on). The result is ecosystem management, and it is far more complex than traditional management of rangelands. The challenge is to integrate the various hierarchies into a system of assessment that can be successfully implemented.

Ecosystem management has the potential to link land use planning and assessment of the state of rangelands as never before. Despite documented degradation of range condition and loss of biodiversity, little re-planning of land use has occurred to date. However, with an integrated system of planning and assessment aimed at meeting regional goals - as advocated in recent rangelands management policy (Anon. 1999) - the onus may shift onto individual land managers to make the required contribution to the regional effort.

The challenge to ecologists is to develop an integrated framework that can provide useful feedback between local land users and regional community representatives. This will require creative thinking to accommodate complexity, and to introduce genuine environmental accountability into the system. Pastoralists will then be able to demonstrate that they are environmentally responsible, and that they have addressed deficiencies in pastoral management of the past.

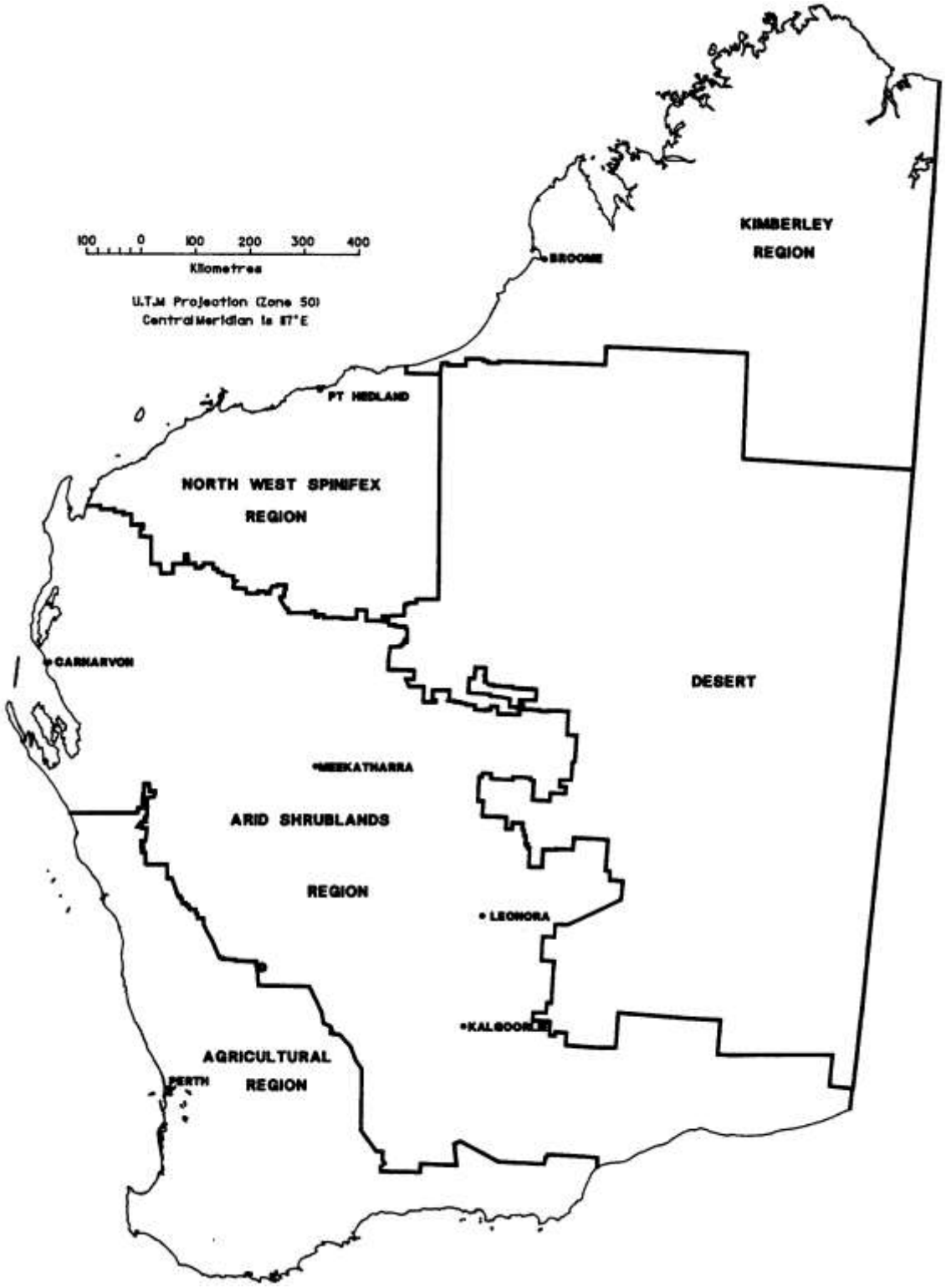


Figure 1.1 Principal agricultural regions of Western Australia.

CHAPTER 2. SURVEY DESIGN AND ANALYSIS

2.1 Introduction

2.1.1 The scope of this chapter

In the previous chapter, I reviewed environmental management in rangelands, with particular respect to assessing the state of the environment in Western Australian rangelands. In this chapter I describe a survey design with which to assess rangelands in the Goldfields of Western Australia according to what may collectively be termed 'new perspectives in environmental management'. This is sometimes referred to as 'ecosystem management' (Kessler *et al.* 1992; Noble and Dirzo 1997; Pringle 1998a). I also include range condition to compare traditional with new perspectives.

I start by briefly reviewing my selection of 'new perspectives' and associated attributes. I then describe the study region and the design of the survey I undertook, in terms of paddock and site layouts in different landscapes. Finally, I describe my approach to statistical modelling.

2.1.2 Questions of interest

The primary questions of interest in designing the survey were:

(1) Does grazing impact differently when viewed from the various key management perspectives?

(2) Does grazing impact from a particular management perspective differ according to type of landscape?

(3) Does type of landscape affect the interrelationships between management perspectives and grazing?

The motivation for these questions is my interest in whether any single system of site-based rangeland assessment (for instance, the current system of range condition assessment) is likely to suffice in reporting on the state of the environment according to different perspectives. A universally applicable approach has been proposed in the United States under the name 'rangeland health' (e.g. National Research Council 1994; West *et al.* 1994; Busby, Ruyle and Joyce 1996). My focus on landscape scale (land systems, mapped at 1: 100 000 scale) and lower should facilitate development of a system for assessing 'rangeland health' if this is at all feasible.

2.2 Choosing perspectives of 'health' to assess

My primary interest is in assessing attributes of landscapes according to a small number of clearly recognisable constituencies or perspectives. These constituencies must be prominent in environmental debate and at the forefront of rangeland debates, in particular (see the previous chapter). I am interested in where assessments according to these constituencies converge and where they diverge, and in whether any limited and practical suite of indicators can accommodate these different management perspectives.

The perspectives I selected are: (1) range condition; (2) biodiversity; (3) natural capital; and (4) ecological integrity, on the basis that they are four key 'criteria clusters' for environmental assessment identified in the State of Environment Report Australia 1996 (State Of Environment Advisory Council 1997). They also represent respectively;

- traditional sustainable (pastoral) yield,
- nature conservation,
- the resource base (soil stability and chemistry), and
- landscape function (the processes supporting ecosystem heterogeneity and resource regulation).

Clearly, these categories are not mutually exclusive, far from it. However, there are discernible differences. For instance, exotic, palatable species might be considered 'good' (useful) according to range condition and 'bad' (threatening or harmful) according to biodiversity (Pringle 1998a).

Below I briefly outline how I selected attributes to represent these perspectives. The individual attributes are discussed in more detail in individual chapters allocated to each perspective.

2.2.1 Range condition

I chose a system of range condition assessment that has been widely used in Western Australian rangelands, through the rangeland survey program (Pringle 1991). I did so for two main reasons. Firstly, these assessments have provided a considerable amount of spatially captured information throughout the study region. If these assessments are closely related to other attributes (say, aspects of biodiversity) cautious predictions could be made about one from the other. Secondly, this project is primarily concerned with representing different perspectives and since local pastoralists have endorsed this system of assessment (North

Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Pringle 1994a) it represents their perspective.

The system is described more fully in Chapter 4. It is essentially based on visual assessment according to the cover and composition of perennial plants. Pastoralists have interpreted assessment results in terms of pastoral productivity for the landscapes studied (Pringle *et al.* 1992). I used both the visual assessments, and their inferred productivity value, as attributes of range condition.

2.2.2 Biodiversity

Biodiversity is literally every living organism, its genes and the communities and ecosystems of which it is a part (State Of Environment Advisory Council 1997). Clearly, it is not feasible to measure it in a direct sense. It is therefore necessary to identify aspects of biodiversity that can be measured, and use these to indicate to some extent the welfare of other groups of organisms.

I was constrained by logistics and my own previous experience to sampling a limited number of aspects of biodiversity. I chose plants, because they are on the 'frontline' in grazing systems and I am familiar with the flora of the region. Other rangeland studies have shown them to be the biotic group most directly affected by grazing, and most responsive to it (de Soyza *et al.* 1998; Whitford *et al.* 1998; Landsberg, Morton and James 1999; Landsberg *et al.* 1999). I also selected ground dwelling invertebrates, because they have been shown to respond to grazing in arid Australian rangelands (Landsberg, James and Morton 1997; Landsberg *et al.* 1999) and were amenable to sampling in the field within my timeframe.

2.2.3 Natural capital

Natural capital in terrestrial ecosystems refers principally to the soil resources that support plant growth and soil biota (State Of Environment Advisory Council 1997)). To address natural capital, I assessed the fertility of topsoil (0-2cm layer) in terms of available nitrogen (AN), available phosphorus (AP), organic carbon (C), electrical conductivity (EC) and cation exchange capacity (CEC). I also assessed Caesium-137 concentrations in the soil, primarily to assess ecological integrity, as indicative of soil erosion and deposition (Gillieson, Cochrane and Murray 1994; Gillieson *et al.* 1996).

2.2.4 Ecological integrity

Ecological integrity is defined for this study as the capacity of ecosystems to maintain patterns of heterogeneity and retain scarce water and nutrients. Noy-Meir (1973) highlighted the importance of these patterns and processes in maintaining the productivity of arid ecosystems. While biological integrity (Angermeier and Karr 1994) is primarily concerned with conserving biota and maintaining the processes responsible for doing so, ecological integrity focuses on maintaining system function, regardless of the identities of the species contributing to this.

To assess ecological integrity, I used two complementary approaches: landscape function analysis and assessment of fallout radionuclide (Caesium 137) distribution.

2.2.4.1 Landscape function analysis

I adopted the approach of David Tongway and his colleagues at CSIRO Wildlife and Ecology (Tongway 1994; Tongway and Hindley 1995; 2000). This approach is fundamentally concerned with identifying landscape patterns and assessing processes within them.

There are two key aspects. Firstly, transects are used to assess landscape patchiness: the types of patches, and the proportion of site occupied by each type of patch. I term this procedure landscape pattern analysis. Secondly, assessments of soil surface features are used to define the qualities of the different types of patches in terms of indicators of water infiltration, nutrient cycling and soil surface stability. This is termed soil surface assessment. Patch qualities can be aggregated to indicate site quality by weighting patch types according to the proportion of a site they occupy (Tongway and Hindley 1995). This is termed landscape function analysis. Landscape function analysis is more of a heuristic interpretation than a statistical modelling procedure.

2.2.4.2 Caesium 137 enrichment

The second approach focussed on measuring soil enrichment by Caesium 137, a fallout radionuclide from atomic testing in the 1950s. Its concentration reflects patterns of sediment distribution and hence the capacity for landscapes to maintain patterns of heterogeneity and regulation of resources (e.g. water and nutrients). While the approach has been extended to measuring absolute rates of soil erosion in arid Western Australia (Gillieson, Cochrane and Murray 1994; Gillieson *et al.* 1996), I focus on assessment of patterns of resource regulation within landscapes. This assumes that rain-borne Caesium-137 has fallen uniformly within a landscape over the last five decades, and that differences in enrichment therefore result from

the landscape's capacity to concentrate resources into fertile patches. Patterns of resource concentration reflect the capacity of the biota (particularly perennial vegetation) to exploit landscape processes (wind and water transport of resources) to develop and maintain fertile patches and hence conserve resources.

2.3 The study region

2.3.1 A brief overview

The survey was conducted in the north-eastern Goldfields of Western Australia, an area that was recently the subject of a biophysical resource survey conducted by Agriculture Western Australia and the Department of Land Administration (Western Australian))(Figure 2.1). The following information has been extracted from these sources.

The climate is arid (rainfall at Leonora averages 221 mm per annum, over 40 rain days) and non-seasonal. Mean monthly maximum/minimum temperatures at Leonora for the months of January (mid summer) and July (mid winter) are $37.1^{o}/18.2^{o}$ C and $21.6^{o}/6.0^{o}$ C respectively.

While rainfall is non-seasonal, high summer temperatures result in rapid drying of the soil and Leonora experiences an average of only 2 pentads (5 day periods) of continuous growth in summer, as compared to 10 pentads (50 days) in winter. Similarly, Leonora has received an 'effective' winter season (at least four continuous pentads) in 71 per cent of years, compared with only 19 per cent of summers (four continuous pentads) (Gilligan 1994).

The vegetation of the region consists largely of three major formations: mulga tall shrublands/ low woodlands, chenopod succulent steppe and spinifex (*Triodia* spp.) hummock grasslands. The region is noted for its occurrences of ephemeral plants ('wildflowers'), which can be quite spectacular in good seasons.

2.3.2 Landscapes of this region

The north–eastern Goldfields are underlain by granite bedrock, with near linear greenstone hill systems trending NNW-SSE, and salt lake systems linked in a tributary palaeodrainage system. Relief is subdued, with hills rarely attaining a relief of over 30 metres above surrounding mulga plains. Plains are generally level to very gently inclined; slopes are rarely as much as 1 per cent. The monotony of sheet-flood plains dominated by mulga (*Acacia aneura*) is occasionally broken by creek systems supporting woodlands of river red gum (*Eucalyptus camaldulensis*) and fringing chenopod flood out plains. The latter are generally

severely degraded and eroded, a consequence of poor grazing management in the early decades of the pastoral industry. Downslope from mulga sheet-flood plains are alluvial plains with saline soils supporting chenopod shrublands, fringing bare salt-lake beds (Pringle, Van Vreeswyk and Gilligan 1994). A typical landscape toposequence is presented in Figure 2.2, based on the dominant granite domain. The vegetation of breakaways and stony plains (depicted in Figure 2.2) varies according to the salinity of the soil. In saline areas, halophytic low shrublands predominate, whereas on less saline soils scattered mulga and other sclerophyll species occur as sparse mixed shrublands.

2.3.3 Landscapes studied

The landscapes studied were the mulga-dominated sheet-flood plains ('mulga landscapes') and the chenopod-dominated depositional plains ('chenopod landscapes'). The mulga landscapes occur immediately downslope from erosional terrain while the chenopod landscapes flank salt lakes within palaeodrainage systems.

Their definition follows the land system approach, which was developed by CSIRO (Christian 1958; Christian and Stewart 1968). A land system is defined as an area with a characteristic pattern of geology, physiography, soils and vegetation that is discernible from air photos. However, because land systems often contain a number of minor units and may grade with topography, I have defined landscapes as major features of land systems, rather than attempting to sample all of the features within a land system. Both land systems and landscapes are mental constructs, so this discrimination is, at least to some extent, one of semantics.

Below I refer to 'gradients', which form the basis of my sampling strategy. A gradient is a succession of sites in a paddock at progressively greater distance from a watering point (a bore or well and the focus of livestock activity), with site 1 nearest to the watering point. Each gradient is located in a particular landscape type. For instance, 'Gradient 2GM' refers to Gradient 2, which is in a groved mulga landscape. The survey design is discussed in more detail later in this chapter.

2.3.3.1 Mulga landscapes

Mulga landscapes have been extensively developed by the provision of artificial watering points (bores or wells), which are located in most corners of most paddocks. This reflects the ease of finding shallow, good quality groundwater in these landscapes. Although the landscapes provide little stock feed in poor seasons, growth of palatable, nutritious annual species in good seasons is prolific. Little of these landscapes remain undeveloped in the

region. There are two major forms of mulga landscapes: groved mulga plains and sandy bank/mulga drainage lanes.

Groved mulga plains

Two of the four mulga study gradients (Gradients 2GM and 3GM) were located in groved mulga plains. These plains are described as Yanganoo or Rainbow land systems in the rangeland survey of the region (Pringle, Van Vreeswyk and Gilligan 1994). Gradient 2GM occurs in the Yanganoo land system. It is described as hardpan plains and sandy tracts with groved mulga shrublands, hard spinifex and wanderrie grasses; and as distributory, alluvial plains commonly with central drainage tracts, receiving run-on from adjacent granite uplands. It grades downslope and laterally into sandplain. It is very similar to Rainbow, differing in that it grades into sandplain rather than extending downslope into palaeodrainage systems. The transitional, sandy units were avoided. Gradient 3GM was incorrectly mapped as Desdemona land system in the rangeland survey, but it is typical of the Rainbow land system. The Rainbow land system is described as hardpan plains supporting mulga shrublands and as alluvial plains subject to sheet flow with sparse, generally narrow and unincised concentrated drainage tracts.

In both land systems, the mulga upper storey is concentrated in groves aligned along the slopes. These are rarely more than two trees deep, but may extend along-slope for up to 100 metres. The inter-grove areas, from which water runs off, support little perennial vegetation.

Sandy bank/ mulga drainage lanes

The other two mulga gradients (Gradients 4SM and 6SM) were located in landscapes dominated by sandy banks and mulga drainage lanes(Mabbutt 1963b). This landscape is described as the Monk land system in the rangeland survey of the area (Pringle, Van Vreeswyk and Gilligan 1994). It consists of hardpan plains with occasional sandy banks, supporting mulga tall shrublands and wanderrie grasses; and has extensive, level to gently inclined plains subject to sheet flow with generally sparse, sub-parallel unincised drainage zones with sandy tracts and banks in lower areas.

2.3.3.2 Chenopod landscapes

Three gradients (Gradients 1CH, 5CH and 7CH) were located in chenopod landscapes. They represent some of the most valuable and reliable (i.e. drought-durable) grazing country in the region (Pringle and Riches 1996). However, chenopod landscapes have not been as

intensively developed with artificial watering points as mulga landscapes. Groundwater is plentiful, but is usually too salty for stock consumption (Allen 1994). Some extension of water supplies into these landscapes has occurred since 'poly-pipe' reticulation became possible in the 1970s. However, large areas remain sparsely watered.

These landscapes are described as the Carnegie or Darlot land systems in the rangeland survey of the area (Pringle, Van Vreeswyk and Gilligan 1994). They are very similar land systems, differing principally in that Darlot land system tends to have more prominent sandy banks. Their landforms, soils and vegetation are very similar, as are the landscape processes responsible for their development.

The Carnegie land system includes salt lakes with fringing saline flats and dunes; salt lakes and fringing level to very gently inclined plains with saline alluvium, sandy banks and low sand dunes above surrounding saline plains. Less extensively, there are undulating (mainly gypsiferous sediments) dunes, gently undulating plains with calcrete rubble, areas of sandsheet, alluvial plains subject to sheet flow on the margins of the system, drainage foci and claypans. Carnegie was the dominant land system at Gradient 1CH and the sole land system at Gradients 5CH and 7CH.

Part of Gradient 1GM was located in the Darlot land system but land units and their patterning was similar at all sites. Darlot land system is described as salt lakes and fringing saline plains, with extensive, regularly arranged, sandy banks and numerous claypans and swamps, supporting halophytic shrublands and spinifex and wanderrie grasses.

2.4 Survey design for assessing grazing

2.4.1 Gradients away from water

The approach I used for assessing grazing impacts is based on the 'piosphere' concept first described by Osborn *et al.* (Osborn, Wood and Paltridge 1932) and later refined by Lange (1969). From it has developed an approach with which to research grazing impacts in arid rangelands based on gradients in grazing pressure out from livestock watering points (Graetz and Ludwig 1978; Andrew and Lange 1986a; 1986b; Curry *et al.* 1994; Jeltsch *et al.* 1997; Landsberg *et al.* 1997). The approach is based on the need for livestock to return to watering points to drink. As a result, grazing pressure is highest in the immediate vicinity of watering points and declines with increasing distance away (see Chapter 3 for more detailed discussion).

The approach is particularly applicable in this study area as persistent sources of natural surface water are rare and temperatures are often high. This means stock and other large herbivores must rely on artificial watering points for extended periods every year. Additionally, paddocks are generally large and stock rarely accesses all parts of them; distance from water constrains their distribution. Gradients are less obviously developed in smaller, more intensely managed paddocks such as are common in the Western Division of New South Wales (Knight 1995).

The pattern of decreasing grazing pressure away from water occurs because (i) the need to drink limits the time that stock can spend away from water; (ii) there is a linear increase in the area of land available in successive annuli (rings bounded by specific distances from water) away from the watering point. The increase in the area of successive annuli is a consequence of geometry. The area of each ring is proportional to πr^2 and the increase in area compared to the previous one (say a 5km radius versus a 4km radius) by the first differential: $2\pi r$. Grazing patterns around watering points are also affected by paddock design. This is particularly so when fences opposite watering points effectively reduce the arc of distant rings and hence reduces their area**.**

In reality, grazing pressure seldom occurs in neat concentric circles (or sectors of circles), because of a range of other factors that influence spatial patterns of stock behaviour. These include water quality (particularly salinity), paddock design (e.g. arrangement of fences and watering points), vegetation types (pasture preferences) and prevailing wind direction (Lange 1969; Hacker 1979; Stafford Smith 1988; Cridland and Stafford Smith 1993; Landsberg *et al.* 1997).

Minor confounding influences may produce warping in the rings radiating away from water (e.g. Lange 1969; Pickup, Bastin and Chewings 1998). However, stronger influences can sometimes exert a greater influence on distribution of grazing pressure than distance from water, as happens when a particularly preferred pasture type occupies a relatively small part of a paddock, to which herbivores are disproportionately attracted (e.g. Landsberg and Stol 1996). These potentially confounding influences were taken into account in both selection of gradients and subsequent location of individual sites along gradients, so as to ensure that distance from water was the primary driving influence on variation between sites. The criteria used for selection of sites are presented in section 2.4.3.

2.4.2 Why grazing gradients?

I was faced with two choices:

(i) to use a grazed versus reference ungrazed comparison and include several major types of landscape in the region, or

(ii) to look at a number of sites along gradients and restrict the number of landscapes investigated.

I chose the second approach as I was primarily interested in comparing assessments according to value systems and hence needed some variability in conditions across which to compare. In effect, I wanted to compare response patterns in the variables reflecting different perspectives, rather than simply scanning these variables for the presence or absence of a grazing effect.

The benefit of using gradients is that one can derive patterns of response that provide substantially greater insights into ecosystem dynamics and their implications. Grazing gradients can also be used in developing decision support systems for spatial planning, be it for improving the efficiency of use of pastoral resources (Cridland and Stafford Smith 1993) or off-reserve nature conservation (Landsberg *et al.* 1999). I have considered the spatial implications of my results with regard to land use planning in the synthesis of this thesis.

Nevertheless, there is an underlying potential for 'natural' gradients to influence my results as my sites were generally located at successively greater distances from water within the same quadrant in sampled paddocks. While it was beyond the resources of this study to attempt to partition these sources of variation, I was careful to avoid obvious gradients such as slopes, distances from salt lakes and so forth in locating sites. Given that my sites were generally kilometres apart and in almost flat landscapes of recurring subtle landform patterning, it is unlikely that 'natural' gradients were strong influences in this study. My use of multiple paddocks for each land type also reduced the risk of underlying 'natural' gradients from being a strong influence.

2.4.3 Gradient designs

I initially intended to have three gradients for each of two major types of landscapes (mulga and chenopod). Each gradient was to include five, approximately evenly distributed sites (an area selected for sampling) at successively greater distances from water in the same general direction, to a targeted furthest distance of at least eight kilometres. Sites were to have been approximately 2 km apart, with the first site as close to water as possible.

Land system maps at 1:250 000 scale (Pringle, Van Vreeswyk and Gilligan 1994) and 1:100 000 scale LANDSAT TM images and pastoral lease plans were used to identify potential gradients. I then inspected 1:50 000 black and white aerial photographs to check photo patterns for potentially confounding influences.

The criteria guiding selection of potential gradients included:

- the land system of interest had to occupy a major portion of the paddock and be continuous from the watering point of interest to the furthest site;
- other potentially confounding land systems of preferred pasture types should not be distributed in such a way as to interfere with the influence of increasing distance from water on historical grazing activity;
- sites should be located in land units showing the same pattern on 1:50 000 black and white aerial photographs, unless the site nearest water appeared to differ due to grazing impact (e.g. Gradient 1GM);
- other sources of permanent water should be no closer to any site than the watering point from which the gradient was anchored;
- there should be no obvious systematic variation in the distribution of rabbit activity (warrens and dung) with respect to distance from water;
- other potentially confounding disturbances (e.g. fire scars) should be avoided;
- there should be no obvious difference in the pedogeomorphic setting along the gradient; and
- the furthest (reference) site should be at least eight km from the anchoring watering point .

I was not able to find gradients that matched the criteria perfectly. The intensity of developed infrastructure in mulga landscapes, due to the ready availability of good quality groundwater (Allen 1994), has resulted in paddocks too small to provide negligibly grazed reference sites. Even in chenopod landscapes, I struggled to find reference sites further than eight kilometres from water. I compromised by selecting seven gradients, four in mulga landscapes and three in chenopod landscapes, varying the layout of sites in each. Their locations are shown in Figure 2.3.

2.4.3.1 Mulga landscapes design

In coming to this compromise I located four gradients in the mulga type, two in each of the major sub-types (i) groved mulga plains (section 2.3.3.1.1) and (ii) sandy bank/mulga drainage lanes (section 2.3.3.1.2). This was because I could not find three acceptable gradients in either sub-type.

In a further compromise, the design for each mulga gradient consisted of 3 sites at increasing distances from water in a paddock, and a reference site in nearby undeveloped, but otherwise similar country (Figure 2.4). The undeveloped country had not been used for grazing livestock and was not near any watering point. While I aimed to find paddocks in which site 3 would be at least 4km from water, this was not possible at Gradient 3GM, where the furthest site is only 3.7 km from the anchoring watering point. It was the last gradient chosen and remains problematic.

All mulga reference sites were in landscapes that had not been developed for grazing, even if a nearby watering point was supporting stock in another paddock (e.g. at Gradient Four). In such cases, I inspected the site carefully to check that there were no obvious signs of grazing on species that are particularly attractive to stock (and feral goats), such as *Santalum spicatum* (sandalwood).

2.4.3.2 Chenopod landscapes design

Because chenopod landscapes are less intensely developed, I was able to follow my original design more closely in selecting chenopod gradients. This involved selecting five sites at successively greater distances from a watering point within the same quadrant (90-degree sector) in each paddock (Figure 2.5). Sandy banks were prominent at Gradients 1CH and 5CH, but not at Gradient 7CH. However I chose not to sample sandy banks because their morphology varied substantially within sites and between sites. This was reflected in major differences in vegetation unrelated to management history (Pringle 1994b). In addition, the morphology of these sandy banks is primarily a result of sediment dynamics unrelated to grazing (Mabbutt 1963b; Churchward 1977) and they are a subordinate component of these land systems (Pringle, Van Vreeswyk and Gilligan 1994).

The reference site at Gradient 5CH was only 6.9 km from water. However, I judged it to be an adequate reference since obvious signs of grazing activity (e.g. tracks, dung, and defoliated shrubs) were lacking, possibly because much grazing may have occurred in more densely watered parts of the paddock. The compromises in design and other problems are summarised for each gradient in Table 2.1.

2.4.4 Landscape pattern and scale issues affecting sampling

Patterning within landscapes varies according to landscape type, particularly with respect to the scale or fractal dimension of the patterns. Given limited resources, I could not afford to employ a fully randomised sampling strategy at my sites. I therefore stratified each site according to clearly identifiable internal patterns and adopted a strategy of maintaining uniform sampling effort across landscape types where feasible.

In chenopod and sandy mulga (sandy banks/ mulga drainage lanes) landscapes, there are two distinct land units in terms of landform/soil/vegetation associations (Christian 1958). These may be up to a kilometre across, but are usually measured in tens of metres (in chenopod landscapes) to hundreds of metres (in sandy mulga landscapes). These are chenopod flats and sandy banks in chenopod landscapes and sandy banks and mulga drainage lanes in sandy mulga landscapes.

Groved mulga plains are extensive land units, extending up to 10 km across and downslope. Groves are generally only one or two trees wide (< 10 m), but may exceed 50 m long across-slope. There is also distinct fine scale patterning within the chenopod flats land unit: bush mounds and inter-bush areas. These patches generally range from less than a metre (bush mounds) to tens of metres across (in the case of inter-bush areas at heavily grazed sites).

2.4.4.1 Chenopod sites

At each site, five sampling loci (within-site replicates) were located at least forty paces apart within a single large chenopod flat. Where large enough flats did not occur, two adjacent flats were selected. I kept constant the distance of loci from adjacent sandy banks (where they were present), at about 20 paces from the transition zone (see Figure 2.6). Sampling was stratified according to bush mounds and inter-bush areas for assessment of most attributes; detailed sampling protocols are presented in relevant chapters.

2.4.4.2 Mulga sites

At sandy mulga sites, five sampling loci were located along the central axis of a mulga drainage lane, forty paces apart. Associated sandy bank loci were selected at intervals of forty paces in the adjacent, sandy unit, at least 20 paces from the transition zone (Figure 2.7). The sandy banks in these landscapes are often referred to as ‘wanderrie banks’ after the perennial wanderrie grasses (e.g. *Eragrostis eriopoda* and *Monachather paradoxa*) that are common in them.

In groved mulga landscapes I stratified my sites into groves and inter-groves (Figure 2.8), selecting five groves and five inter-grove areas downslope from a predetermined starting point identified on air photos. Where groves were less than forty paces apart, I skipped a grove and chose the next one.

2.5 Analysing unbalanced, multi-level data

I have confined my analysis to the linear component of responses to estimates of grazing history (see Chapter 3 for more detail). Non-linear responses to grazing are well documented (Graetz and Ludwig 1978; Cridland and Stafford Smith 1993). However, to have addressed non-linear components of responses would have required considerably more intense sampling within individual paddocks (to avoid 'overfitting' the data), at the cost of number of paddocks studied. Detailed local study is something of a tradition in rangeland research (e.g. Lange 1969; Holm 1994; Ludwig *et al.* 1997). However, I wanted to achieve some measure of regional overview. To have restricted my sampling effort at higher levels of my survey design in order to address non-linear responses in local ('case') studies would have been at the expense of my ability to generalise regionally.

The sampling design I used is both multi-levelled and unbalanced. In such instances, conventional analysis of variance procedures are inappropriate for statistical modelling. This is because between-level variances cannot be accurately estimated, with the result that errors are under-estimated. This causes artificial inflation in estimates of the strength of effects, making them more likely to be found statistically significant when they are not (Lawes Agricultural Trust 1997).

Instead of conventional analysis of variance procedures, I used 'residual' or 'restricted' maximum likelihood (REML) (Patterson and Thompson 1971; Robinson 1987). REML is a type of generalised mixed model that estimates sources of variation in the response variable being investigated. These sources may result from the underlying experimental design (random effects, such as differences between sampling loci within a site, or gradients within a landscape) or the 'treatments' of biological interest (fixed effects, in this case variation resulting from differences in landscapes and grazing).

2.5.1 Random effects

Random effects are those that are a consequence of the experimental design, but are not of fundamental interest with regard to the question being addressed. Variance is allocated to terms in the random model, commencing with the lowest order effects. The main sources of random variation in my study were between loci and strata within a site (lowest order effects), between sites within gradients and between gradients within types of landscapes.

My design was not neatly nested at the lowest level. In mulga landscapes, sampling loci were nested within landscape strata (such as a mulga grove or a sandy bank), but in chenopod landscapes the opposite was the case, with landscape strata (bush mounds and inter-bush areas) nested within loci. The random effects model was therefore expressed accordingly:

- gradient (between gradient variation);
- gradient*site (within gradient variation); and
- gradient*site*locus**.**stratum (within site variation).

Variance components and standard errors were estimated for each term. Variances of individual model components can also be estimated as an option within REML. These are:

- gradient;
- site;
- stratum; and
- locus.

Where sampling was conducted above the locus level in the design (e.g. Caesium-137 cores), the locus term was dropped from the random model.

2.5.2 Fixed effects

These effects are the imposed treatments of interest, in my case the variation in effects of grazing between landscape types and within landscapes. To investigate this, I developed the following terms within a fixed model, to address the associated questions:

- landscape type: Does the mean of a response variable differ between mulga and chenopod landscapes?

- main grazing effect: Does grazing affect the response variable, regardless of landscapes or landscape stratum? (I used three alternative estimates of grazing; their derivation is described in the next chapter).
- landscape type.grazing effect: Does the nature of any response to grazing differ between landscape types?
- stratum.landscape type.grazing effect: Does the response to grazing differ between landscape strata? (Landscape strata effects are compared among landscapes, comparisons are not confined to types of landscapes).

REML uses one of two approaches to test significance. The first and more conservative assesses change in deviance, based on the effect of dropping or adding terms to a model. It is time consuming, however. Because of the amount of data I had to analyse, I chose the second, less conservative approach, but used caution in interpreting the results. Significance is tested against the Wald statistic, which approximates a Chi squared distribution; I disregarded effects where probability values were greater than 0.045.

I commenced each analysis by fitting the full fixed model and then eliminated non-significant terms, starting with lowest order effects. This approach approximates to backwards elimination (Crawley 1993).

The complex hierarchical nature of my mixed models places constraints on the level of statistical inference that is valid. Standard errors are generalised, rather than being specific to individual values or classes within a model term. It is not appropriate therefore to do multiple pair-wise comparisons within model terms. It is more appropriate to produce schematic graphic presentations and interpret them heuristically (Ross Cunningham, personal communication). This is the approach I adopted.

As previously stated, I analysed only the linear component of responses to grazing. Linear approximation is valid for the range of values investigated in my study. However, caution is needed in extrapolating beyond the range. In particular, response variables that indicate a cumulative response to grazing are likely to asymptote in the long term. Response variables that indicate cumulative grazing effects will eventually reach zero or infinite values in the long term (as grazing years accumulate). Thus my modelled estimates of grazing effects should be treated with caution, particularly with respect to making inferences about long term sustainability or regional land use planning.

2.5.3 Presenting model results

I ran full statistical models for each of the three estimates of grazing history to investigate grazing 'treatment'. These were Stocking History (SH), Stocking Rate (SR) and Distance From Water (DW). I present model results for all three estimates, but focus discussion on Stocking History (SH) models if significant fixed effects occur.

In subsequent chapters I present the results of statistical modelling graphically. I use bar charts with an approximation of the least significant difference (lsd) when graphically presenting landscape effects on a variable (e.g. differences between mulga grove and inter-groves in available phosphorus in the topsoil). The least significant difference is estimated using two times the standard error of the difference between means in REML models. It provides a robust measure with which to compare means at a 5 per cent confidence level. I present only lines from fitted values where grazing effects were found to be significant. This is because of the large number of data points involved (usually 310) and their possible differentiation according to landscape context (e.g. mulga grove versus inter-grove). The graphs would be virtually impossible to interpret with observed values superimposed on fitted lines; such would be the density of information. However, I present a measure of goodness of fit (p-value, derived from a chi-squared approximation of Wald Statistics) for graphed effects. I usually present goodness of fit estimates for interaction terms only (where they exist), unless main effects are an order of magnitude greater than the interaction. In these cases, I present both estimates. The fitted lines show differences in the sampling range across the independent variable where there are both landscape and grazing effects. This is standard graphical output for REML models (Ross Cunningham, personal communication).

2.6 Summary

(1) Attributes of range condition, biodiversity, ecological integrity and natural capital were assessed along seven grazing gradients in two major landscapes in a large rangeland region (Figures 2.1 and 2.2). Each gradient was separated by at least twenty kilometres from its nearest neighbour (Figure 2.3).

(2) My key interests were to compare and contrast the impacts of grazing on attributes representing various management perspectives and to consider how much these interrelationships were influenced by the landscape in which attributes were assessed.

(3) In mulga landscapes, three sites were located at increasing distance from a watering point in each of four paddocks. A further reference site was located in nearby, undeveloped country for each of these paddocks (Figure 2.4). Two gradients were located in groved mulga landscapes and a further two in sandy mulga landscapes.

(4) Three gradients in different paddocks were located in chenopod landscapes. Each consisted of five sites at successively greater distance from water (Figure 2.5).

(5) Statistical analysis of the effects of grazing and landscape treatments used restricted maximum likelihood modelling (REML). This is a recent procedure designed for linear regression modelling of data from unbalanced experimental designs, using a generalised mixed model.

(6) Regression analysis was limited to the linear component of responses to grazing. Thus extrapolation of results requires some caution, particularly with respect to inferences about long term sustainability and implications for land-use planning.

Table 2.1 Design issues affecting study gradients

Gradient	Landscape type	Issues
1CH	chenopod	• not stocked for last 5 years • rabbits throughout, areas close to warrens avoided • failed bore 5 km from site 5 • site 1 severely eroded into a new landform • cattle occasionally run with sheep (major stock type) • minor wind erosion from a recent dry cyclone at all sites except site 1
2GM	groved mulga	• mulga plains subordinate to spinifex sand plains in total area (but not grazing value) • site 1 is near the mulga-spinifex transition
3GM	groved mulga	• furthest site in paddock is only 3.7 km from water • influence of another watering point modelled to be considerable at site 3 • reference site similar, but higher in the landscape than paddock sites
4SM	sandy mulga	• chenopod landscapes prominent in paddock • reference site lightly stocked for one year 4 years before sampling commenced, 4.2 km from the watering point
5CH	chenopod	• reference site only 6.9 km from water • rabbits present, warrens discovered at site three during sampling • exotic plant (*Centaurea melitensis*) observed near reference site (5)
6SM	sandy mulga	• minor hail damage at site 3, one mulga locus affected
7CH	chenopod	• site 4 adjacent to a saline depression • watering point only 15 years old, but heavily stocked • rabbit activity affects half of site 5 (reference site), little evidence of activity elsewhere

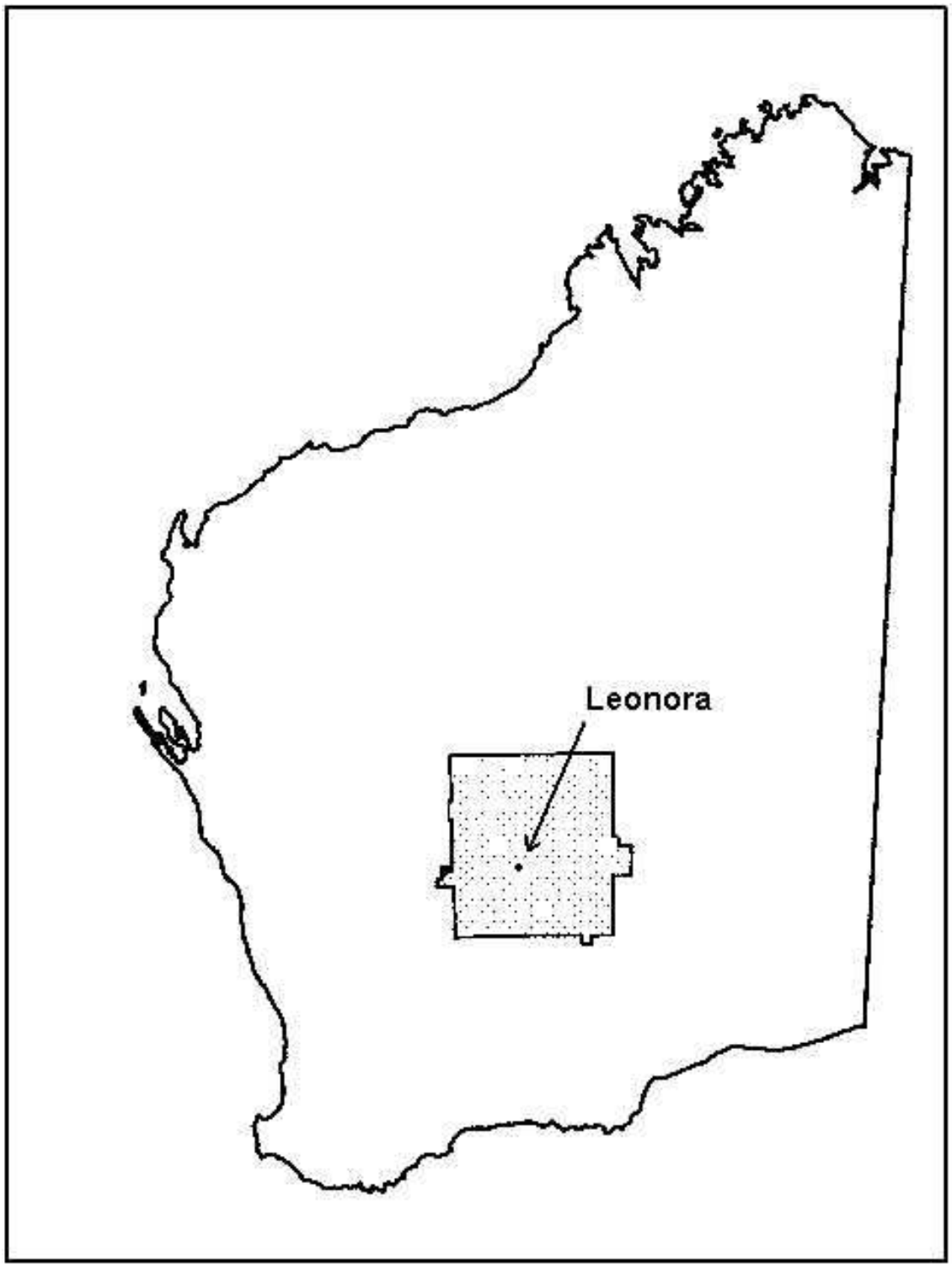


Figure 2.1. Location of the study area.

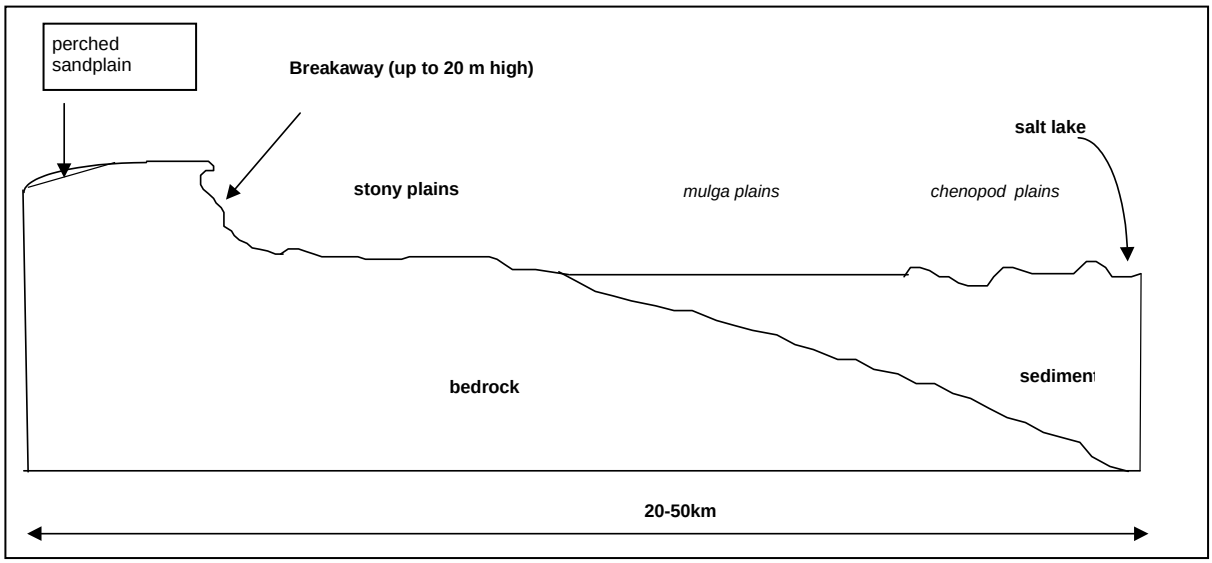


Figure 2.2 A typical sequence of landscapes in the study region.

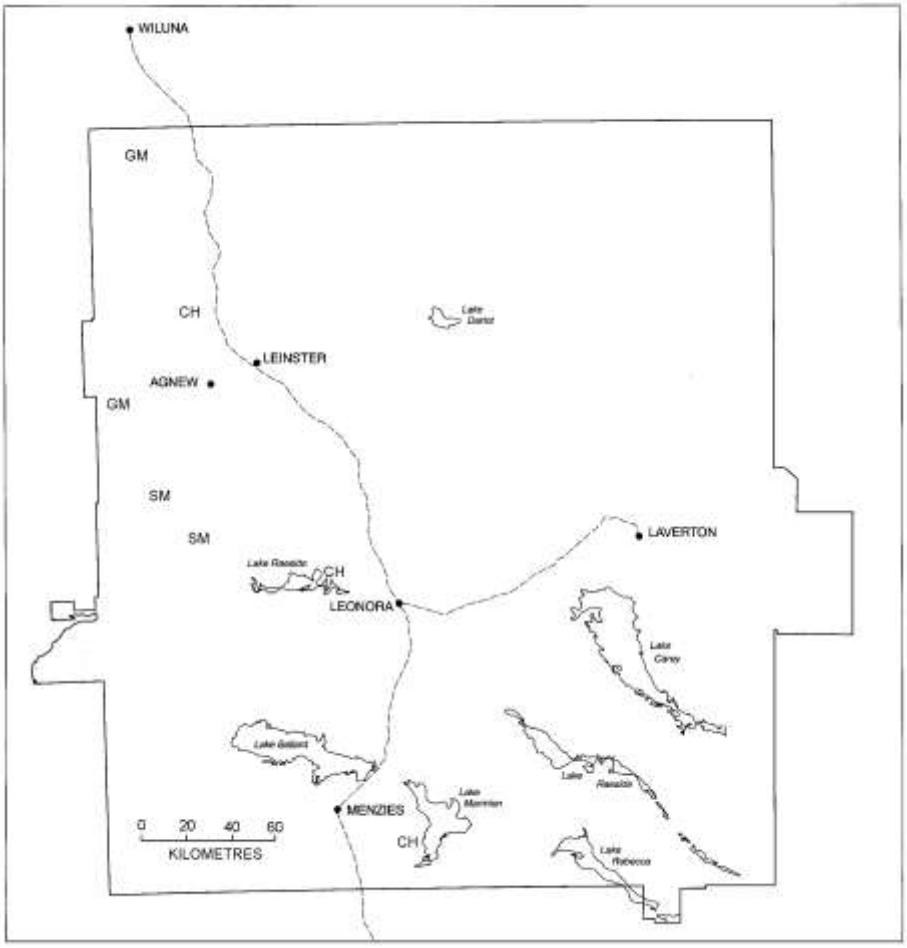


Figure 2.3 Location of gradients within the study area.

For reasons of confidentiality, I have not identified individual gradients.

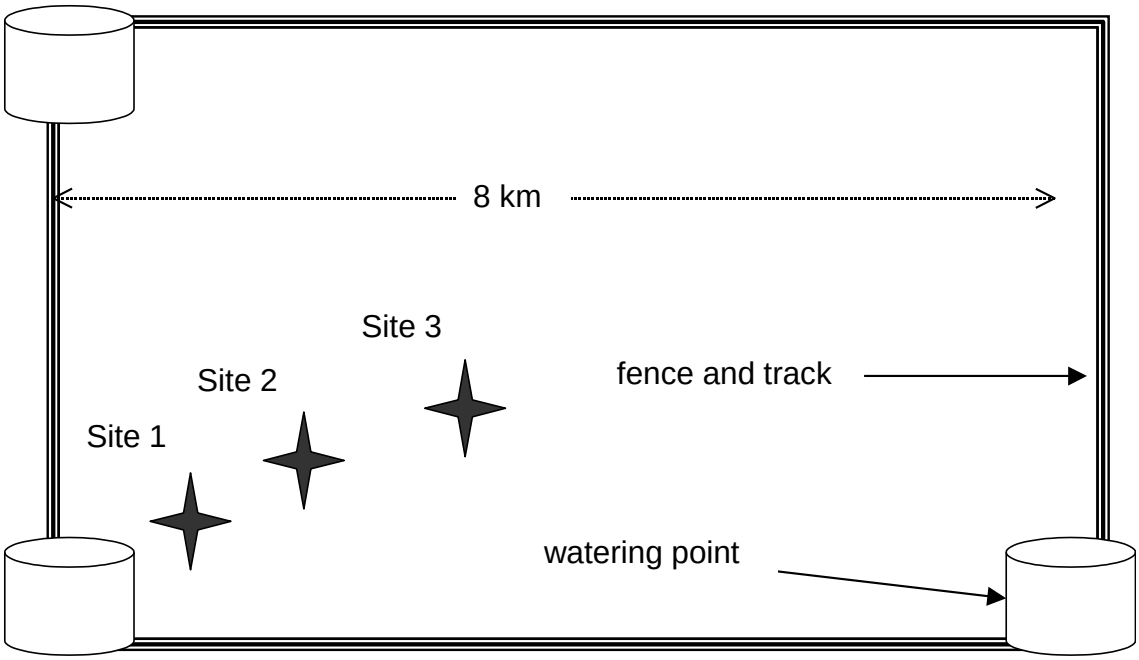


Figure 2.4 Typical layout of a study paddock in mulga landscapes.

Note that the reference site is located in an ungrazed area outside the paddock, but within the same sub-catchment. This layout is for both sandy and groved mulga landscapes.

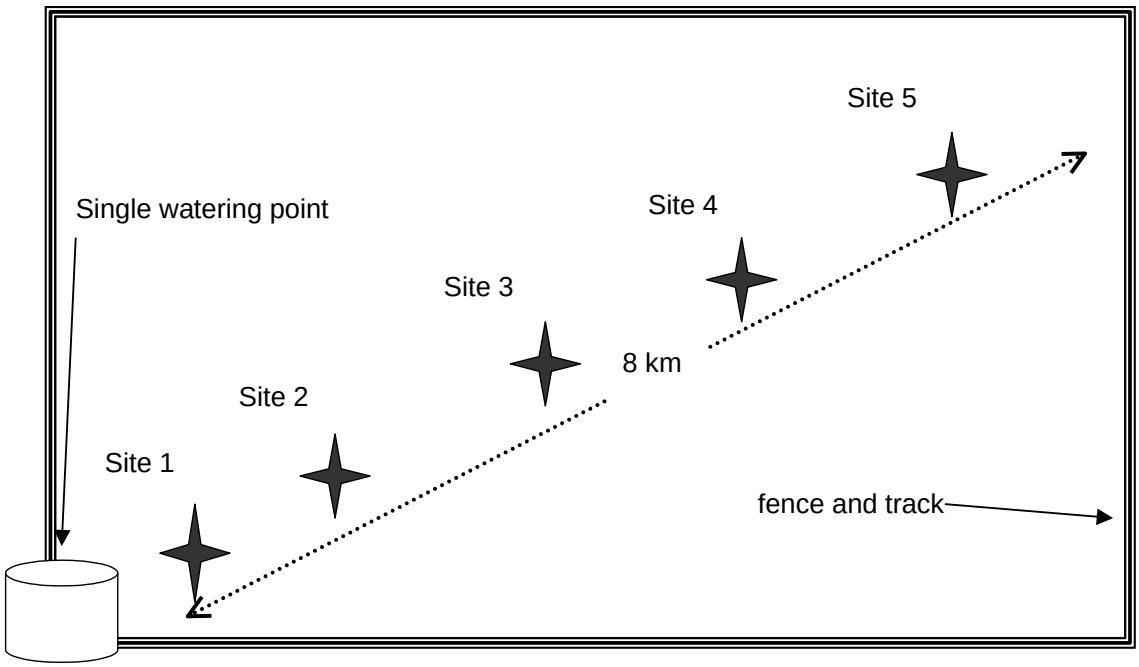


Figure 2.5 Typical layout of a study paddock in chenopod landscapes.

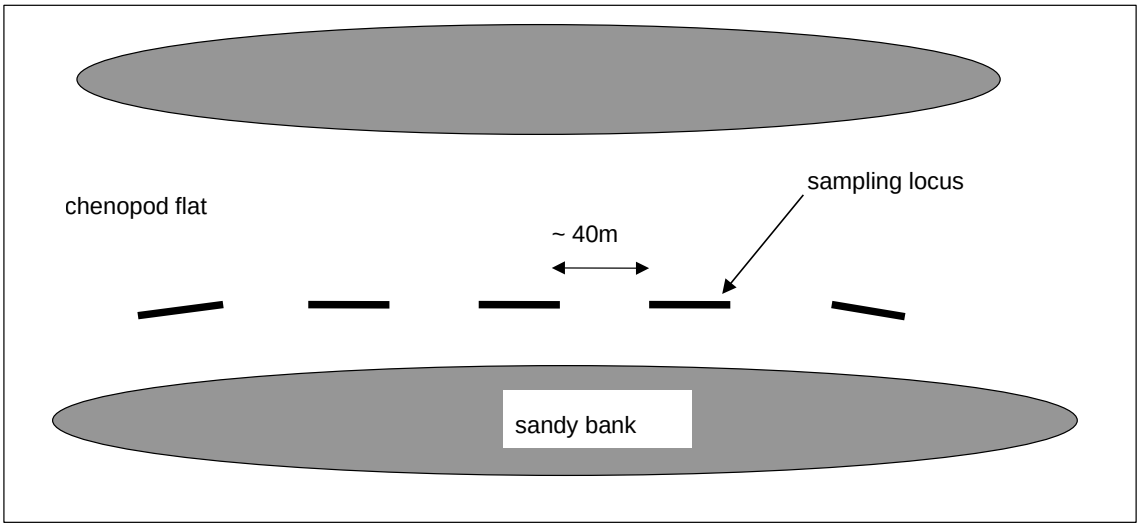


Figure 2.6 Layout of a typical chenopod site.

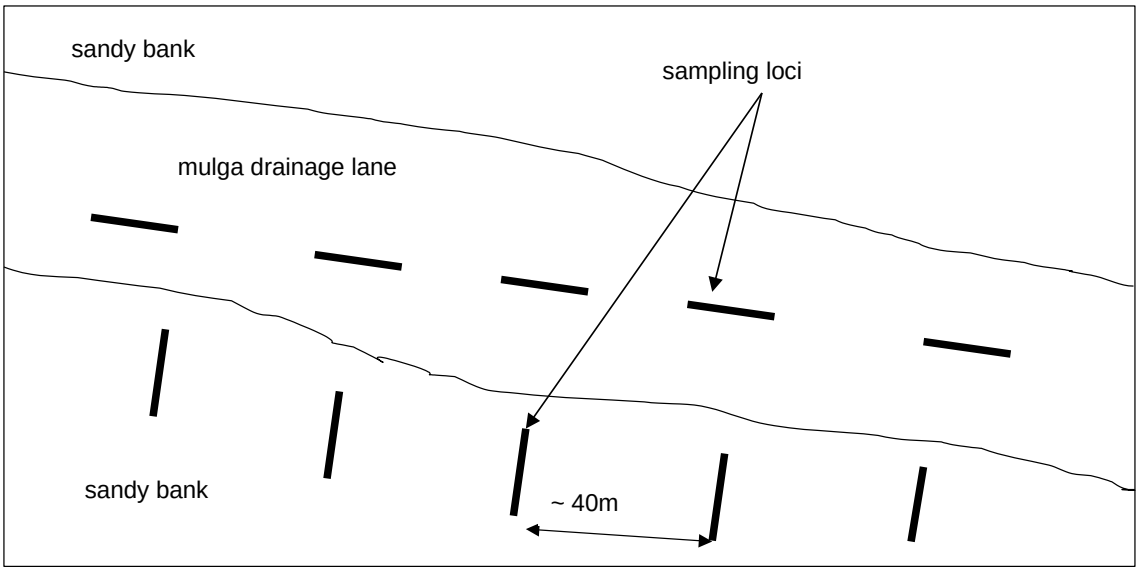


Figure 2.7 Layout of a typical sandy mulga site.

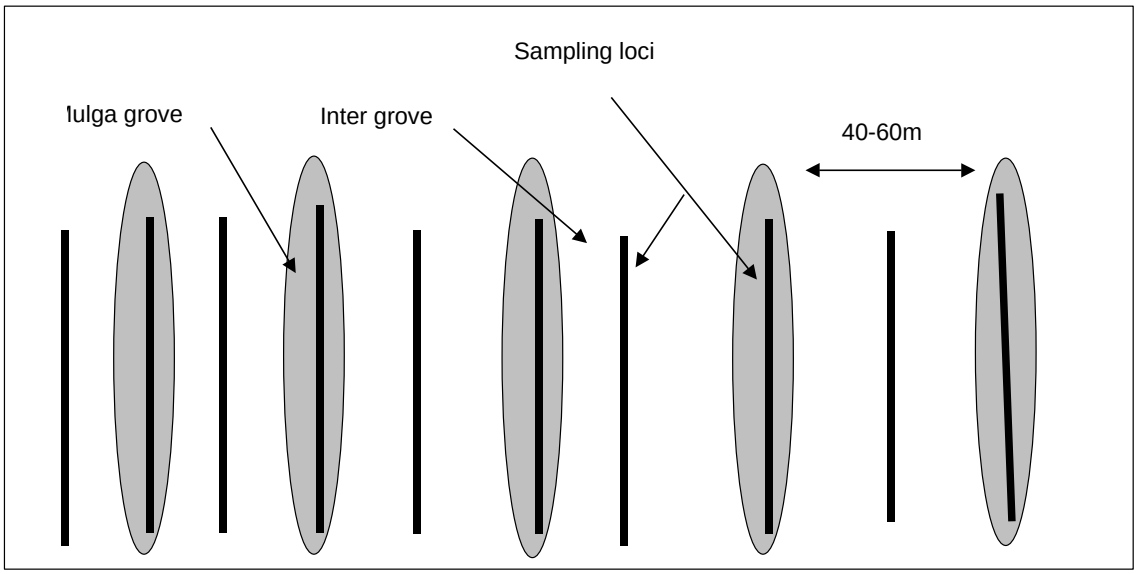


Figure 2.8 Layout of a typical groved mulga site.

CHAPTER 3. MODELLING GRAZING HISTORY

In this chapter I describe how I developed three estimates of grazing history: a simple surrogate and two predictive models. The estimates were developed to include the effects of grazing in my statistical modelling of response variables representing different perspectives in rangeland management.

I commence with a brief discussion of grazing history as a conceptual basis for this work. I then review ways in which it has been estimated, before describing the process of developing the estimates based on collection of data in the field, statistical modelling and application of models spatially using G.I.S.

3.1 Grazing history in rangeland paddocks

Grazing has considerable potential to modify environments (Ratcliffe 1936; Milchunas and Lauenroth 1993; West 1993; Pringle, Van Vreeswyk and Gilligan 1994) but is surprisingly more difficult to quantify than is the consequent ecosystem response. This is particularly so in large rangeland paddocks where low densities of livestock are allowed to range widely. In arid Australian rangelands, sheep densities of less than 10 animals per square kilometre are common, in paddocks tens to hundreds of square kilometres in area (Pringle 1994a).

Managers usually know (or have a reasonable idea) about the average numbers of domesticated livestock in each paddock. The standard grazing estimates derived from this information are *stocking density* (number of animals per unit land area) and *stocking rate* (number of animals per unit land area per management cycle) (Heitscmidt and Taylor 1991). In the context of long-term impacts, the total time the land has been exposed to stock is also important. This is incorporated into an expression called *stocking history*, which is defined as the average *stocking rate* multiplied by the total time the land has been in use, expressed in units of sheep years per hectare (see Glossary: Appendix 1).

However, these stocking estimates are invariably calculated as averages across whole paddocks, and sheep do not use large paddocks evenly (Lange 1969; Wilson and Harrington 1984; Lange 1985; Stafford Smith 1988; Burnside, Williams and Curry 1990).

Stocking density, rate and history in a particular part of a paddock depend not only on the total number of animals in the paddock, but also on their distribution within it. A number of static and dynamic factors influence stock distribution. Static factors include distance from drinking water, the salinity of the water, location of fences and the direction of prevailing

winds. There is also a range of dynamic factors, including season and the availability of forage (which varies with season and stocking rate and history) (Dudzinski, Pahl and Arnold 1969; Lynch 1974; Squires 1974; 1976; Noble 1979; Stafford Smith 1988; Landsberg and Stol 1996). There are various ways grazing distribution can be modelled to incorporate factors such as these, but all have their limitations (see reviews by (Stafford Smith 1988; Coughenour 1991; Bailey *et al.* 1996).

Coughenour (1991) recognised three main types of models:

- multiple regressions, which seek to explain a large portion of the variance in animal distribution by combinations of variables associated with the grazing landscape;
- models using habitat suitability indices, where a suitability index is constructed for each of the different habitats in a grazing landscape, by combining explanatory variables chosen from regression modelling; and
- individual-based models of animal foraging behaviour.

The individual-based approach avoids many of the simplifying assumptions of the other two approaches, which are based on the distribution of aggregate populations. However its use is severely limited, as it requires a high level of understanding of the mechanisms that determine foraging behaviour. These mechanisms are both dynamic and poorly understood (Bailey *et al.* 1996). Thus the aggregate-based approaches are generally more amenable to modelling long-term distributions, particularly across broad scales. Of these, the first approach is the simplest, and forms the basis of this work. Habitat suitability is also incorporated in the models.

Regardless of approach, validation of models of long-term grazing distribution remains a problem, because the average distribution of grazing animals over long periods is seldom known. Empirical ‘snapshots’ of animal densities (counts of animals) do not necessarily reflect long-term aggregated densities. Mismatches of current density and average long-term density of animals at a site can readily arise. During droughts, a lack of forage in areas where animals usually graze may force them to graze in less preferred areas (Hodder and Low 1978; Low, Dudzinski and Muller 1981; Stafford Smith 1988). In theory, long-term pictures can be constructed by aggregating large numbers of snapshots, but snapshots are rarely available.

I commence with a brief review of major factors affecting the distribution of grazing, how some of these have been incorporated into an existing commercial decision support system,

and how I decided to test this system. I then describe the process of obtaining the surrogate measures and developing the models I tested.

3.2 Review of ways of estimating grazing history

3.2.1 Surrogate measures of grazing history

I discriminate between 'surrogates' and 'models' of grazing history in this chapter. I treat surrogates as direct co-variates with measures of grazing history in space and time, while models represent integrated mathematical predictors (see Glossary: Appendix 1).

The use of surrogates has been adopted widely in the face of difficulties in directly measuring the distribution of long term grazing activity (stocking history). Four surrogates are commonly used: distance from water, accumulation of dung, track density and vegetation cover. They are reviewed briefly below.

3.2.1.1 Distance from water

Some broad, general relationship between grazing distribution and distance from water in large, arid zone paddocks is inevitable with water-dependent animals. In the arid zone large herbivores such as sheep need to drink regularly, particularly in warm and dry weather, when they need to drink at least once per day (Stafford Smith, Noble and Jones 1985; Stafford Smith 1988; Burnside *et al.* 1990). Natural surface water is rarely reliably available in Australian rangelands (Landsberg and Gillieson 1996) and my study region is no exception. Artificial sources of water have therefore become the focus of animal activity. However, the nature of the relationship between grazing distribution and water is likely to vary considerably between watering points, especially in heterogeneous paddocks (Stafford Smith 1988; Cridland and Stafford Smith 1993; Landsberg and Stol 1996).

The relationship between grazing distribution and distance from water is seldom linear (e.g. Arnold and Dudzinski 1978; Lange 1985; Stafford Smith 1988). This means that distance alone does not provide an adequate explanatory measure of grazing distribution within a paddock. At the very least, distance needs to be transformed, but the nature of the transformation is likely to differ between paddocks (Stafford Smith 1988; Cridland and Stafford Smith 1993).

In addition, the influence of distance from water can be misleading when other local factors become influential in a paddock. These factors include grazing preference for different types of vegetation(Landsberg and Stol 1996) and proximity to a fence line (Hacker 1978).

However, the paddock and site selection criteria I used were designed to minimise the influence of such local factors (see Chapter 2). Multiple watering points and variations in paddock design, vegetation and management history may also influence the distribution of grazing activity. These complications make it inappropriate to rely solely on distance from water as a surrogate for stocking history when making comparisons between paddocks. In other words, two sites at one kilometre from water in paddocks with different infrastructure, vegetation and/or management history might be *expected* to have experienced different stocking histories.

Despite these shortcomings, distance from water can provide a useful first approximation for indicative regional assessments of grazing history and impacts (Landsberg and Gillieson 1996). However, the limitations (as discussed) in developing more accurate or precise relationships must be acknowledged.

3.2.1.2 Dung

Accumulation of animal dung can be a useful surrogate for recent grazing activity, particularly for sheep (Rawes and Welch 1969 in Lange 1969; Lange and Willcocks 1978; Lange 1985; Stafford Smith 1988; Landsberg and Stol 1996). However, it is similar to instantaneous density in its limitations as a surrogate for long-term activity. Under typical rangeland conditions, dung seldom persists for more than a few months (Landsberg and Stol 1996). Therefore, although dung accumulation provides a more integrated picture of grazing activity than animal density, it is essentially a 'snap-shot' with a slightly longer time exposure.

Given the time taken for forage to be digested, it is highly unlikely that a sheep will deposit dung in the same spot as the forage was grazed. Dung accumulates around watering points (which are often devoid of forage) and campsites; its highest densities are probably in areas of little forage intake. However, if these areas are avoided, dung can be a useful relative measure of recent grazing distribution within paddocks (Landsberg and Stol 1996). Dung may also be transported from run-off zones and accumulate in relatively fertile, depositional parts of landscapes and catchments. Dung has therefore limited use in the specific context of my study.

3.2.1.3 Track density

The density of well-developed tracks has a long history of use as a surrogate of long-term grazing activity, e.g. (Lange 1969; Squires 1974; Hacker 1978) although it appears to have been neglected recently. Tracks strictly reflect walking rather than grazing - or at least

'disturbance'. But they have been shown to relate to vegetation patterns which can be reasonably assumed to have resulted from grazing (Hacker 1978).

The logic is that areas of higher track density should reflect areas that have been used more heavily and for longer periods of time. In rangeland paddocks, sheep tracks branch out from water and become less distinct as animals fan out to graze (Arnold and Dudzinski 1978). Areas of high track density reflect areas where grazing activity is likely to have been heavy in the past, but more recent grazing activity may be occurring in areas where well-defined tracks have not yet developed. Therefore, track density is at best a relative, rather than an absolute measure of long-term grazing activity. However, it has other advantages: it reflects a much longer time-frame than animal counts or dung accumulation and it is also relatively easy to sample.

Unlike vegetation cover, track density is less likely to be both a cause and a consequence of grazing activity. However, as all stock grazing patterns partially reflect previous grazing impacts (e.g. favouring or avoiding degraded areas), a totally independent surrogate may be illusory.

3.2.1.4 Vegetation cover

The problem with using impact on vegetation as a surrogate for grazing distribution is the implicit circularity that results if modelled levels of grazing are used subsequently to infer impacts on vegetation. Nevertheless changes in vegetation cover undoubtedly occur in response to grazing (Foran 1980; Foran, Bastin and Shaw 1986; Bastin *et al.* 1993; Pickup, Bastin and Chewings 1994; 1998; Pickup and Chewings 1994; Pickup and Bastin 1997). Furthermore, remotely sensed data can provide an appropriately long time frame for assessing aggregate grazing impacts on vegetation. This approach has been successfully applied in central Australia (e.g. Bastin, Sparrow and Pearce 1993; Bastin *et al.* 1993; Pickup, Chewings and Nelson 1993; Pickup and Chewings 1994; Pickup, Bastin and Chewings 1994; Pickup, Tynan and Chewings 1998; Pickup, Bastin and Chewings 1998). Cridland and Stafford Smith (1993) used this approach to develop predictors of long-term sheep activity used in the 'Paddock' module of RANGEPACK, a commercially available software program which is used in pastoral extension (e.g. in designing paddock infrastructure) (Stafford Smith and Foran 1990). In the Sahel region of Africa, Hannan *et al.*(1991)and Hellenden (1991) have also used remotely sensed approaches to assess patterns of grazing-induced change based on vegetation cover around watering points (Jeltsch *et al.* 1997).

3.2.2 Other influences on grazing history

3.2.2.1 Paddock age

Paddock age can influence grazing distribution because the zone of severe impact (or 'piosphere') (*sensu* Lange 1969) is not necessarily static or in equilibrium (Andrew and Lange 1986a; Hunt 1995; Jeltsch *et al.* 1997). Rather, it may expand over time, as areas close to water are 'eaten out'. However, the rate of expansion is likely to decline over time as the length of the arc or 'degradation front' increases (a consequence of the geometric relationship between the radius and circumference of a circle) and the effort required by sheep to walk out to it increases. However, an equilibrium model is often assumed (e.g. Cridland and Stafford Smith 1993) and may be a reasonable approximation in lightly stocked or very old paddocks.

3.2.2.2 Stocking rate

In an analogous fashion to paddock age, higher numbers of stock can accelerate the spread of the 'piosphere' by reducing the amount of forage near the watering point more rapidly.

3.2.2.3 Paddock configuration

The size and shape of paddocks and the location of watering points can have a strong influence on the distribution of grazing (Stafford Smith 1984; 1988). For instance, watering points in the corners of rectangular paddocks have a quarter of the angle of approach of open watering points (located away from fences). If similar numbers of stock access waters, the former might then be expected to have a zone of severe impact extending further from water than that from the centrally located watering point. This is because sheep are 'funnelled' in to water by the fences. The result is that any effective 'stocking rate' will occur further from water in a corner compared with an open watering point and the stocking rate near a corner will be higher than near an open watering point. Sheep will therefore remove forage more quickly in the confined area and to a greater distance from the watering point than would occur around an open watering point (Cridland and Stafford Smith 1993).

3.2.2.4 Salinity of water

Sheep have to drink more regularly when using salty water as they expend some of the water expelling salt in their urine. They are therefore constrained to graze less far from water (Wilson 1974; Noble 1975; North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Cridland and Stafford Smith 1993). Salty stock water therefore tends to concentrate grazing activity in the vicinity of the watering point, and to limit the distance

grazed out from water, particularly in poor seasons when fresh, natural water and succulent ephemeral plants are not available.

3.2.2.5 Preference for different range types

'Pasture preference' is a well- established concept in rangeland management e.g. (Low, Dudzinski and Muller 1981; North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Landsberg and Stol 1996). It is analogous to the habitat 'suitability' or 'quality' concepts of Coughenour (1991) and Bailey *et al.* (1996). Differences in pasture preference lead to uneven distribution of grazing activity among vegetation types. The decline in grazing activity and impact with increasing distance from water may also be influenced by pasture preference. (Pickup and Bastin 1997; Pickup, Tynan and Chewings 1998).

3.2.3 Models of grazing history

3.2.3.1 RANGEPACK/Paddock

RANGEPACK/Paddock is a decision support model developed by CSIRO Wildlife and Ecology and Agriculture Western Australia to help design pastoral paddocks that minimise degradation and maximise production (Cridland and Stafford Smith 1993). It was parameterised mainly on the chenopod shrublands of the Nullarbor Plain, Western Australia using a remotely sensed index of vegetation cover that has been validated in central Australia (Bastin *et al.* 1993; Pickup 1994). There was little other ground validation of this model, although pastoralists were asked about model outputs and commented favourably (Cridland and Stafford Smith 1993). However, it was more difficult to parameterise the RANGEPACK/Paddock model in more heterogeneous landscapes such as my study region (Cridland personal communication).

RANGEPACK/Paddock is based around a negative exponential function of impact incorporating distance from water and watering point salinity. This underlying decay function is then modified according to the influence of pasture preference and wind direction at any point in a paddock. Cridland and Stafford Smith note that wind direction is most likely to be influential in relatively homogenous landscapes where other influences are less likely to overwhelm its influence.

The formulation of the model and its application are described in more detail in section 3.4 of this chapter.

3.2.3.2 Track density model

I decided to develop a locally parameterised model based on track density. This was in the light of the problems in parameterising RANGEPACK/Paddock in landscapes such as those in my study region and also the potential logical circularity of using a model parameterised using one of the response variables of interest in this study (vegetation cover). The independent variables I included and the summarised reasons for their consideration are:

D = distance from water: chosen because of its paramount importance in arid rangelands

S = salinity of water: chosen because of its potential to constrain the distance sheep can travel between drinks

A = age of watering point: chosen because of its potential to influence accumulated impact on forage availability and therefore distance animals must travel along tracks before spreading out to graze

N = average number of sheep using the watering point, which was chosen for the same reason as age

I chose not to include a measure of the location of water with respect to fence-lines because five of my seven watering points were located in a corner of their paddocks, one was a few hundred metres from the corner and the other was along a fence-line (180° access). Angle of access was accounted for in applying models spatially (see below).

3.3 Data collected in the current study

3.3.1 Distance from water

Distance from water was assessed from 1:50 000 scale black and white aerial photographs, marked up with fence lines, tracks and watering points. Distance was estimated to the nearest 100 metres from the nearest watering point.

The layout of gradients varied between mulga and chenopod gradients (see Chapter 2) resulting in some systematic differences in site distances from water (Figure 3.1). In particular, the values for site four on mulga gradients (Gradients 2GM, 3GM, 4MS and 6MS) may be misleading as these sites occur in areas that have not been stocked (in 'reference' or 'benchmark' areas). The discrepancy in survey design between chenopod and mulga landscape types comes about because watering points have been developed close together in mulga landscapes. No continuous stretches from a watering point to a water-remote (> 8 km) site within developed areas of mulga landscape were found (Chapter 2).

3.3.2 Dung

Dung was sampled at one gradient only, where it soon became apparent that its distribution was not significantly related to distance from the watering point. This was because sampling was conducted during an excellent run of seasons and there was abundant growth of pasture with high moisture content. As a result stock were not frequenting the watering point, but instead were grazing in areas further from water where more feed was available. Dung was sampled by counting pellet groups along three transects of 45 paces per land unit (e.g. wanderrie bank). All pellets within a metre either side of the centre line were counted.

3.3.3 Track density

I estimated a track density by counting tracks along transects at 42 locations across six of the seven grazing gradients. Transects were located at the study sites and at intermediate sampling locations equidistant between them. The transects were orientated perpendicular to the direction of the nearest watering point, with each transect approximately 45 m in length. At each sampling location three transects were walked in each land unit present. Thus I conducted three transects at sampling locations in groved mulga and chenopod landscapes, and six in sandy mulga landscapes (three each on sandy banks and in mulga drainage lanes, though the densities differed little between units). Occasional local topographic features e.g. low rises or drainage foci), which can influence track patterns (Lange 1969), were avoided and minor, local anastomoses were considered single tracks. Only those tracks with clearly recognisable courses were counted. Track counting was conducted in November 1998. The procedure was rapid, with each site frequently completed in less than ten minutes.

Track density declined exponentially with increasing distance from water (Figure 3.2). This indicates extremely intense activity (grazing, trampling, defecation and so forth) near water and a rapid decline in activity away from water. Track definition also declined with increasing distance from water: tracks became shallower and had less clearly defined margins.

3.3.4 Paddock age

I ascertained paddock age by asking pastoralists. Where the pastoralist was new to the area, I contacted previous managers.

There was a wide range of paddock ages (Table 3.2). Gradients 3GM and 5CH represent some of the oldest pastoral country in the district, being major paddocks near homesteads built in the first years of the local pastoral industry. In contrast, the youngest paddocks tend

to be those furthest away from their respective homesteads. Gradient 7CH is about 50 km from its homestead and was first stocked only 15 years ago.

3.3.5 Stocking rate

I also acquired this information from pastoralists and expressed it as the long term average number of dry sheep equivalents (d.s.e.) that had been held during most years in that paddock. In two out of seven gradients stock record books were available from the time when the paddock was developed. For the other paddocks I sent letters to past and present managers. I also interviewed all present managers and a few previous managers to check that my figures were correct and to discuss the history of my study paddocks.

The long-term average number of sheep (dry sheep equivalents) in the study paddocks ranged from 300 to 1000 (Table 3.2). Generally speaking, paddocks with considerable areas of chenopod shrublands had carried more sheep each year.

3.3.6 Paddock configuration

Digital plans of the stations were supplied by the Spatial Resources Information Group, Department of Agriculture, Western Australia (DAWA). A member of the Group, Emma Raff, extracted the paddock information for each gradient. This included fence lines, watering points and major salt lake beds. Pastoralists confirmed that these lake beds represent effective barriers to movement of sheep and feral goats. Thus I treated them as equivalent to fences. I also treated cliff faces as fences where they ran continuously from one fence to another. Again, pastoralists confirmed that this was realistic.

Paddocks were generally rectangular and most watering points were located in corners of paddocks, a pattern developed on the basis that such placement allows for troughs to be placed in four paddocks from one well or bore. Diagrams of typical paddock layouts are included in Chapter Two.

3.3.7 Salinity of water

The concentrations of total dissolved salts (TDS) in water from bores and wells in my study were obtained from the Water and Rivers Commission (WA), courtesy of Dr. Tony Laws. Water supplied to stock in the paddocks sampled was generally fresh, i.e. had low concentrations of TDS. (Table 3.2).

3.3.8 Sheep preference for different range types

Range types were designated from land system maps of the area (Pringle, Van Vreeswyk and Gilligan 1994), which had been captured digitally during the rangeland survey of the region. They were allocated sheep preference ratings adapted from those of Cridland and Stafford Smith (1993), who had used grazing trial data augmented by the opinions of local pastoralists and range ecologists. For those land systems that were not rated by Cridland and Stafford Smith, I compared pastoralists' assessments of productivity with similar assessments of rated land systems (Pringle *et al.* 1992; Pringle 1994a) to assign a preference rating.

Chenopod depositional plains tended to be rated as most preferred by sheep, followed by up-land chenopod plains, then mulga sheet flood plains, stony plains with sclerophyll vegetation, and sclerophyll hill systems and sandplain systems (Table 3.3). Lakebeds are generally devoid of vegetation and were rated zero.

3.4 Modeling grazing history of my study sites

3.4.1 Applying RANGEPACK/Paddock

A slightly modified version of the relative grazing activity model of Cridland and Stafford Smith (1993) was developed in consultation with these authors.

Their original function is:

$$RGA = e^{(C*D)} * W_o * P_v$$

Where:

RGA = Relative grazing activity

C = -0.392-0.741*salinity of water supply (%)

D = distance from water supply in kilometres

W_o = proportion of days wind blows from the direction (octant) of the water supply

P_v = relative grazing preference values for different types of rangeland mapped in paddocks.

This function is calculated from each watering point in a paddock, assuming that they support equal numbers of the paddock flock. When the sum of scores for any place (pixel) is added up and then divided by the sum of these scores for the whole paddock, a 'total relative activity index' is calculated for that place. When sheep numbers for a paddock are multiplied

by this index, the number of sheep using that area is estimated and can be expressed as a stocking density if the area of that place (pixel size) is known. I used pixels of 100m by 100m, so stocking densities were automatically expressed as sheep per hectare (or part thereof).

I introduced two modifications. One was to drop the wind term. It did not appear to have much influence on animal behaviour in the study from which the model was developed (Cridland and Stafford Smith 1993). In addition, Curry *et al.* (1994) failed to distinguish a consistent wind effect in a regional survey of the adjacent, ecologically comparable Murchison River catchment and surrounds.

My second modification was to incorporate differential use of water supplies. This was done on the basis of local managers' estimations. For example, if a paddock had two watering points, but the local manager estimated that the sheep used one of them 75 per cent of the time or that on average, 75 per cent of the sheep watered there, then I assumed that watering point supported 75 per cent of the flock in that paddock. Otherwise, the modelling was identical to that developed by Cridland and Stafford Smith.

I consulted both Cridland and Stafford Smith about these modifications and they supported them.

A raster-based geographic information system was used to apply RANGEPACK/Paddock algorithms. I specified the components of the RANGEPACK/Paddock model as individual grid operations as described above, but the GIS modelling was conducted by Emma Raaff of DAWA, South Perth.

The model produced a curvilinear decline in the average long term Stocking Rates (SR) along each gradient, with successive approximately equidistant sites away from water (Figure 3.3).

As a form of local validation I determined how closely these modelled Stocking Rates (SR) were associated with assessed track densities using generalised linear modelling (Crawley 1993). When I used raw track densities there was strong systematic variation in residuals, which was successfully overcome using their natural logarithms. The RANGEPACK/Paddock model of stocking rate (Cridland and Stafford Smith 1993) accounted for 41 per cent of the variance ($P < 0.001$) in natural logarithms of track density across all paddocks. The regression was:

$$Ln(TD) = 1042*RP - 36.6$$

Where:

Ln(TD) = Natural log of track density

RP = RANGEPACK/Paddock model of stocking rate

When different y-intercepts ('constant' terms in the linear model) were allowed for each gradient the fit improved to 68 per cent of variation ($P < 0.001$). Thus while there was definitely a significant relationship between the RANGEPACK/Paddock estimates and track density, RANGEPACK/Paddock was not accounting for major differences between paddocks in the relationship between track densities and distance from water. Landscape type was not a significant effect in accounting for variation among paddocks.

In part, this may have been because several sites included in the analysis had track densities of zero, where RANGEPACK/Paddock would (realistically) have estimated a small positive Stocking Rate. However, the most obvious variation in track densities between paddocks was close to water where track densities were high. This variation was not reflected in the RANGEPACK/Paddock estimates of stocking rate.

3.4.2 Modelling track density

A non-spatial model for predicting track densities was developed using multiple linear regression and a range of independent variables with the potential to modify grazing history (see earlier sections) (Senft, Rittenhouse and Woodmansee 1983; Stafford Smith 1988). Track density (T), the response variable, was transformed to its natural logarithm to meet underlying assumptions in the statistical modelling, including avoidance of systematic variation in residuals. The models fitted were of the general form:

$$\ln TD = a + bD + cS + dA + eN$$

Where:

TD = track density (tracks/km)

D = distance from water (km)

S = salinity of supplied water (%)

A = age of paddock (years)

N = average stock numbers (dry sheep equivalents).

Inverse values were also considered (e.g. 1/A). The significance of each term was tested by backward elimination (Crawley 1993). The statistical modelling employed the Generalised Linear Modelling module in Genstat 5 release 4.1.(Lawes Agricultural Trust 1997).

The minimal model (*sensu* Crawley 1993) that best reflected variation in track density within and among paddocks was:

$$Ln(TD) = 4.6 - 0.9D - 30.0A^{-1}$$

Where:

Ln(TD) = natural logarithm of track density (tracks/km)

D = distance from water (km)

A = age of paddock (years)

The percentage of variation accounted for was 83.1 per cent and all terms were highly significant (Table 3.4).

3.4.3 Applying the modified model across my study paddocks

I transformed the logarithmic function back to 'raw' track density for use as a decay function in subsequent spatial modelling. The resultant function used to describe decline in relative grazing activity with increasing distance from water in study paddocks was:

$$RA = e^{4.6 - 0.9D - 30.0/A}$$

Where:

RA = relative grazing activity (estimated tracks/km)

D = distance from water (km)

A = age of paddock (years).

Firstly, each paddock was displayed spatially, as a series of pixels 100 x 100 m (one hectare) in size. Next, a score for the 'unadjusted' relative grazing activity was estimated for each pixel, by modelling the track density for the pixel mid-point. For paddocks with multiple watering points, separate unadjusted scores were calculated for each watering point independently, scaled according to the proportion of use attributed to that water. The resultant scores from each watering point were summed to give an accumulated grazing score for each pixel; these were then multiplied through by the grazing preference rating for the land system assigned to each pixel. The adjusted, accumulated pixel scores were then

standardised to a proportion of the paddock total, by dividing the total score for each pixel by the total sum of scores for all pixels in that paddock. These steps follow procedures described in RANGEPACK/Paddock (Cridland and Stafford Smith 1993).

These standardised estimates of relative grazing activity were converted to stocking rates (in units of sheep per hectare per year) by multiplying the average number of sheep in the paddock per year over its pastoral history by the standardised pixel score. In effect, this spreads sheep across the paddock according to the relative grazing activity of each pixel.

Finally, the Stocking History (SH) of each pixel (in units of sheep years per hectare) was estimated by multiplying the average long-term stocking rate by the number of years the paddock had been in use. The use of paddock age twice in this model is addressed in the Discussion section. I also transformed the RANGEPACK/Paddock estimate of Stocking Rate (SR) into a stocking history in the same way to compare the predictions of the two approaches to modelling in a relative sense.

In summary, for each hectare pixel within a paddock, the process involved:

- estimating the proportion of total grazing activity in the paddock it had received (relative grazing activity),
- multiplying this value by the average number of sheep run per year in the paddock to estimate stocking rate (sheep per hectare per year), and
- multiplying the stocking rate by number of years of grazing to estimate stocking history (sheep years per hectare).

3.4.4 Comparing model outputs

Substituting the RANGEPACK/Paddock decay function with the track density-based distance/ inverse age function resulted in a considerably more exponential decline in modelled grazing history at successive sites out from watering points at each gradient (Figure 3.4). In several gradients there was a ‘crossing over’ in estimated values.

RANGEPACK/Paddock tends to predict slightly higher relative grazing activity than the modified model at greater distances from water, while the track density model predicts substantially higher relative grazing activity near water.

3.5 Discussion

3.5.1 Influences on grazing history

3.5.1.1 Distance from water

The deficiencies in distance from water as a surrogate measure of grazing history were clearly highlighted by the track density data (see below). Not only do track densities suggest that decline in activity with distance from water is non-linear; the non-linearity differs between gradients. This has been found elsewhere (Cridland and Stafford Smith 1993). Nonetheless, distance from water remains a strong influence on grazing distribution, as the models show.

3.5.1.2 Track density

The strong non-linearity in declining track density away from water probably reflects sheep behaviour. At successively greater distances from water, increasing numbers of sheep may have peeled away from tracks to graze, leaving ever decreasing numbers of sheep still in tracking mode. The gradual loss of track definition with increasing distance from water further supports the idea that sheep track together through degraded country, then begin to spread out as they encounter land which has retained its capacity to produce stock feed (Stafford Smith 1988).

More particularly, one might quite easily envisage this general pattern of sheep behaviour resulting in the observed pattern of successively fewer, less clearly defined tracks with increasing distance from water. Around troughs there is no evidence of tracks. Congestion is such that sheep are behaving as individuals: trying to get in to drink and away to feed or camp. Trampling of the soil surface is extreme. However moving away from the trough, tracks quickly become apparent: they are dense and clearly incised into the soil surface. Moving further away, track density and definition decline, congestion dissipates, forage availability increases and sheep begin once again to behave increasingly as individuals, this time to leave tracks in order to feed.

However, the considerable variation in the densities of tracks and their rate of decline with distance from water in different paddocks indicate that they are best used as a measure of relative stocking history within individual paddocks. It is also known that animals travel considerably further to graze than the track density data indicate. Cridland and Stafford Smith (1993) found evidence of grazing off-take (reduction in vegetation cover by grazing) beyond 10 kilometres from watering points over a summer period using comparisons of

remotely sensed cover at the beginning and end of a generally warm, dry summer. However, this activity did not leave a permanent impact on vegetation cover.

Track densities therefore underestimate both Stocking Rate (SR) and Stocking History (SH) at sites beyond approximately four kilometres from water (see Figure 3.2). This distance is consistent with the recommendations for watering point spacing for the region (Burnside *et al.* 1990; Cridland and Stafford Smith 1993); areas beyond more than three or four kilometres from a watering point are considered 'under-utilised' in terms of pastoral management.

My use of track densities in modelling Stocking History (SH) means that I am focussing on walking, grazing, defecation and urination behaviours of sheep. I am not, for instance, addressing camping behaviour (campsites were avoided).

3.5.1.3 Paddock age

My use of paddock age (or age of watering point, to be more precise) as a simple scaling variable to derive stocking histories (accumulated sheep years per hectare) from modelled stocking rates (sheep per hectare) was a compromise. It was clearly desirable to include it in some way, since it would incorporate influences such as development of patterns of impact (e.g. Andrew and Lange 1986a; Hunt 1995; submitted), growing environmental awareness among local pastoralists (e.g. North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993) and the accumulation of management mistakes.

Including paddock age in the decay function and as a scaling variable in developing the Stocking History (SH) model might appear at first to be a case of 'double dipping'. However, this is not so. In the first instance, paddock age is used to model the 'shape' of variation within individual paddocks. This has intuitive appeal and some empirical basis (Andrew and Lange 1986a). The second use is to account for variation between paddocks in the total number of stock-years of use. It seems quite reasonable that the pattern of relative grazing activity will change over time, and that the period over which a paddock has been grazed is pivotal to estimated accumulated grazing.

The approach I used was simplistic because it assumes paddock age exerts the same, linear influence across all paddocks. It also assumes a regular, consistent pattern of grazing pressure and impact, rather than punctuated, episodic change such as might occur during drought years, for instance. By not using paddock age as a separate independent variable in modelling responses to grazing, I have foregone some capacity to account for systematic

variation. However, grazing models such as RANGEPACK/Paddock do not include paddock age at all.

Thus my somewhat simplistic incorporation of paddock age was a compromise between leaving it out all together, and adding another level of complexity to an already complex and multi-facetted experimental design.

3.5.1.4 Stocking rate

It is highly likely that the best predictor of grazing impacts might be stock numbers during and immediately after droughts, when damage and suppression of recovery might be more likely, but these data are not available. My use of long term average stocking rate was because annual stock records were not available for most paddocks.

3.5.1.5 Salinity of water

Ground water in the region is generally of good quality (low salinity) (Allen 1994), and is unlikely to have been a major factor influencing differences in patterns of grazing distribution among paddocks studied. This contrasts with the findings of Cridland and Stafford Smith (1993) on the Nullarbor Plain, where the range is greater and levels of water salinity are generally higher. This issue is discussed further in terms of influence on modelled grazing history.

3.5.1.6 Sheep preference for different range types

The use of grazing preference ratings is analogous to the habitat 'suitability' or 'quality' approaches discussed by Coughenour (1991) and Bailey *et al.* (1996). Without this term, modelled grazing distributions would have been neatly radial out from watering points. The inclusion of a preference term mimics to some extent the 'star patterning' in remotely sensed data detected by Pickup, Bastin and Chewings (1998). They attributed this patterning to different grazing preferences among landscape types.

The preference ratings I used had limited empirical basis (Cridland and Stafford Smith 1993) and ground validation by strategic dung and track sampling was planned. Unfortunately, this work was abandoned after I was 'rained out' on two successive field trips. However, even without empirical validation there is good reason to believe that the ratings are realistic. Cridland and Stafford Smith (1993) spent considerable time discussing preferences with pastoralists and experienced range ecologists in this pastoral region.

The grazing preference term has a scalar effect on modelled stocking densities. This is consistent with recent work on cattle in central Australia (Pickup and Bastin 1997; Pickup,

Bastin and Chewings 1998). A simple linear, scaling co-efficient for range type (land systems in my study) is apparently sufficient.

Use of land managers' suggestions (Pringle 1994a; Pringle and Riches 1996) as to pastoral productivity of land systems has considerable merit. This approach is consistent with the view of Bailey *et al.* (1996), that forage quality and supply are principal determinants of habitat selection for grazing. As Stafford Smith (1988) pointed out, sometimes many years of local knowledge is more useful than limited research findings regarding a complex grazing system.

My use of a land system-based preference rating system was another compromise in the interests of simplicity. Pickup, Bastin and Chewings (1994) found that finer scale stratification using Multi-Spectral Scanner (MSS) data was superior to land system information in identifying grazing impacts in heterogeneous paddocks. Their result reflects the aggregation of different remotely sensed signatures at a land system level and perhaps the hierarchical nature of grazing behaviour (e.g. Senft *et al.* 1987). I did not have the resources to model grazing history at a finer scale. Furthermore, had I allocated different estimates of grazing history within sites (e.g. separately to each wanderrie bank or mulga drainage lane in sandy mulga landscapes), I would have lost the capacity to investigate systematic variation at the landscape type level of my survey design. I would have uncoupled complementary landscape strata in terms of their 'grazing treatment'.

The failure to discriminate grazing estimates between strata of each landscape may have prevented me from unravelling more closely the relationship between the way animals use landscapes and how the components of the landscapes respond.

I used land systems to allocate preference ratings as digitally-captured data were available. I recognise the hierarchical nature of grazing distribution (Senft *et al.* 1987; Coughenour 1991; Bailey *et al.* 1996) and that grazing is unlikely to have been uniform across each study site. However, the landscape scale at which I have applied these ratings is also the main level of interest of this work. Heterogeneity in the distribution of grazing within my sites is at least partly a landscape effect, which is included in statistical models of response variables.

3.5.2 Performance of models

3.5.2.1 RANGEPACK/Paddock model

While the RANGEPACK/Paddock model of grazing distribution is based on a negative exponential function, it predicts a weak exponential decay in grazing history with distance from water in my study region. This is because of the minimal influence of the salinity of stock drinking water in this region, as compared with the Nullarbor Plain where RANGEPACK/Paddock was parameterised (Cridland and Stafford Smith 1993). It is also in contrast to the strongly exponential decline in track densities with increasing distance from water.

Nevertheless, the modelling framework in RANGEPACK/Paddock was easily modified to suit local conditions and was a reasonable predictor of variation in track densities. It's clearly laid out structure and procedures make it an excellent tool for modelling grazing impacts from the paddock to regional scales.

3.5.2.2 Track density model

The strong predictive capacity of the simple model developed for predicting track density (83.1 per cent of variation accounted for) indicated that an alternative, locally parameterised model of grazing history was indeed worth developing. It was no surprise that distance from water was the strongest influence, based on the strong patterning in the raw track data. The significance of the inverse of paddock age as a strong discriminator between paddocks also makes intuitive sense.

Mathematically, the negative exponential of inverse paddock age has the effect of concentrating relative grazing activity near watering points in young paddocks and spreading it out as paddocks age. This is a reasonable pattern of response to declining quality and quantity of feed near water. The non-linear deceleration with time also makes sense; it is in response to increasing area with distance from water, which is proportional to Πr^2. At some distance, areas must be too far from water for sheep to leave a permanent imprint (Hunt 1995; submitted). This process may be partly captured by the inverse of paddock age in the decay function for track densities.

3.5.2.3 Comparing models

Although the track-based model may more accurately reflect relative grazing activity near water, it undoubtedly underestimates it further away, where tracks are very sparse and undeveloped. Conversely, the RANGEPACK/Paddock model based on remotely sensed change in plant cover appears to underestimate relative grazing activity near water, but provides a more realistic estimation of its extension out from watering points. However, remotely sensed evidence of grazing activity beyond 10 kilometres on the Nullarbor Plain indicates that even RANGEPACK/Paddock underestimates grazing very far from water (Cridland and Stafford Smith 1993). It was developed to predict grazing impact, rather than stocking history. Discussions with local pastoralists in my study region and observations of sheep and feral goat activity at some of my water-remote sites confirms to me that I have underestimated grazing history with both RANGEPACK/Paddock and the track density based models at sites beyond about four kilometres from water.

However this underestimation is not a major problem for my study because; (i) the relative discrepancies are small, (ii) they are not critical to the objectives of this study and (iii) I investigated only the linear responses to grazing activity. In other words, fine-tuning grazing models at the lower end of the range of estimated values is not needed in order to compare the linear component of interrelationships at different levels of grazing.

In subsequent chapters I will concentrate on interpreting the relationship between response variables and three estimates of grazing history:

(1) track-based model of Stocking History (SH: sheep years per hectare);

(2) RANGEPACK/Paddock model of (long-term average) Stocking Rate (SR: sheep per hectare per year); and

(3) Distance from Water (DW: measured in kilometres).

The Stocking Rate (SR) model was parameterised from data reflecting impact on vegetation cover, which suggests that it already includes an influence of historical management. It is not intuitively sensible therefore to scale its estimates by years grazed; it is already a model of impact. Conversely, the track density-based model is less dependent on impact (but clearly not totally so) and is developed as a model of where sheep have grazed in the past. It therefore seems appropriate in this case to scale its estimate of stocking rate by years grazed to better estimate stocking history. I will also include Distance from Water (DW) in statistical modelling of response variables given the ease of measuring it and its potential to

provide some cautious predictive capacity at a regional level, as well as the preliminary findings of Cridland and Stafford Smith (1993) regarding patterns of off-take.

I place most emphasis on the Stocking History (SH) model in subsequent chapters relating to responses to grazing, as it is locally-tailored and less dependent on grazing impacts in its derivation than the Stocking Rate (SR) model.

3.6 Summary

(1) It is almost impossible to measure directly the accumulated historical distribution of large herbivore grazing activity in large, old, arid zone paddocks, where grazing patterns are strongly non-random and dynamic. Despite this difficulty, this type of information is needed in order to understand patterns of ecosystem responses to grazing. It is also critical for planning ecologically systems of sustainable resource use. Indirect ways of estimating grazing history are therefore investigated in this chapter.

(2) Distance from water can be a reliable method of ranking sites within a paddock, but it does not accommodate non-linearity in grazing distribution patterns and has negligible value in comparing sites in different paddocks. Track density is a useful surrogate of relative grazing distribution within an individual paddock, but similarly has little value for comparing sites in different paddocks. It also tends to underestimate grazing distribution at distances greater than four kilometres from water.

(3) The RANGEPACK/Paddock model is commercially available software developed to model long term grazing impacts. RANGEPACK/Paddock was developed in a region with a wide range in groundwater salt content and, not surprisingly, this variable was a significant factor in modelling grazing impacts in that study. However, groundwater in the paddocks that I studied was generally fresh. This may account for the failure of this model to predict more than 41 per cent of variation in track densities across the six sampled paddocks. Other factors were more influential on grazing distribution.

(4) Better predictions were obtained using a track-based model within the general RANGEPACK/Paddock framework.

(5) A simple linear regression model involving only the distance from water and the inverse of that watering point's age accounted for 83 per cent of variation in track densities.

(6) The mathematical relationship between track density and distance from water was incorporated with spatial information (including paddock infrastructure, sheep grazing preference ratings for range types and water point use) and historical paddock stocking information to derive spatial distributions of average stocking rate and history.

(7) The track-based model predicts higher, more realistic relative grazing activity near watering points and underestimates relative grazing activity further out from water in comparison with RANGEPACK/Paddock. There is evidence to suggest both RANGEPACK/Paddock and the track-based model underestimate the extent of grazing activity very far from water (more so in the latter model).

(8) I multiplied modelled stocking rates by years grazed to derive stocking histories. This was done in order to account for variation in accumulated stock-years among my study paddocks, acknowledging in the process that it is a statistically restrictive approach.

(9) Unfortunately it is impossible to assess rigorously the quality of any of these models since no direct measures of long term grazing are available. However, the use of a combination of them should suffice in finding systematic variation in response variables in relation to pastoral management. The models seem intuitively more realistic and widely applicable for estimating accumulated, long term grazing activity than the two surrogates, whose relationships with grazing are paddock specific.

(10) In analysing the impacts of grazing in subsequent chapters I will focus on the track-based model of Stocking History (SH: sheep years per hectare), but will also include the RANGEPACK/Paddock model of Stocking Rate (SR: sheep per hectare per year) and Distance from Water (DW: kilometres).

Table 3.1 A summary of the three estimates of grazing history

Variable and its code in statistical modelling	Type	Description
Distance from Water (DW)	surrogate	Distance from the nearest watering point
Stocking Rate (SR)	model	RANGEPACK/Paddock estimate of long term average stocking rate expressed in sheep/ha
Stocking History (SH)	model	Locally tailored model of stocking history based on track densities expressed in sheep years per hectare

Table 3.2 Pastoral information of paddocks studied

Gradient	Year first stocked	Years destocked	Last year of grazing	Years of grazing	Average no. of sheep per year	Water quality (TDS in mg/l)
1CH	1926	8	1993	63	1000	5250
2GM	1938	12	1997	47	335	2270
3GM	1902	0	1997	95	600	650
4SM	1926	0	1997	71	600	5000
5CH	1900	0	1997	97	650	4400
6SM	1930	0	1995	65	350	1800
7CH	1982	0	1997	15	570	2720

Table 3.3 Grazing preferences assigned to land systems.

Preferences range from 0 for least preferred to 40 for most preferred (adapted from Cridland and Stafford Smith 1993)

Land system	Preference rating
Carnegie	40
Cyclops	40
Darlot	40
Mileura	40
Wilson	36
Gumbreak	30
Gransal	24
Cunyu	20
Leonora	20
Deadman	15
Desdemona	12
Melaleuca	12
Monk	12
Rainbow	12
Sherwood	12
Yanganoo	12
Pan	9
Waguin	9
Hamilton	8
Bandy	6
Challenge	6
Sunrise	6
Windarra	6
Yowie	5
Laverton	4
Bullimore	2
Marmion	2
Wyarri	2
Lake bed	0

Table 3.4 The minimal model predicting natural logs of track densities

Model terms	Significance (P) of effect of dropping term	Cumulative variance accounted for(%)
Distance from water	<0.001	73.0
+ Inverse age of water	<0.001	83.1

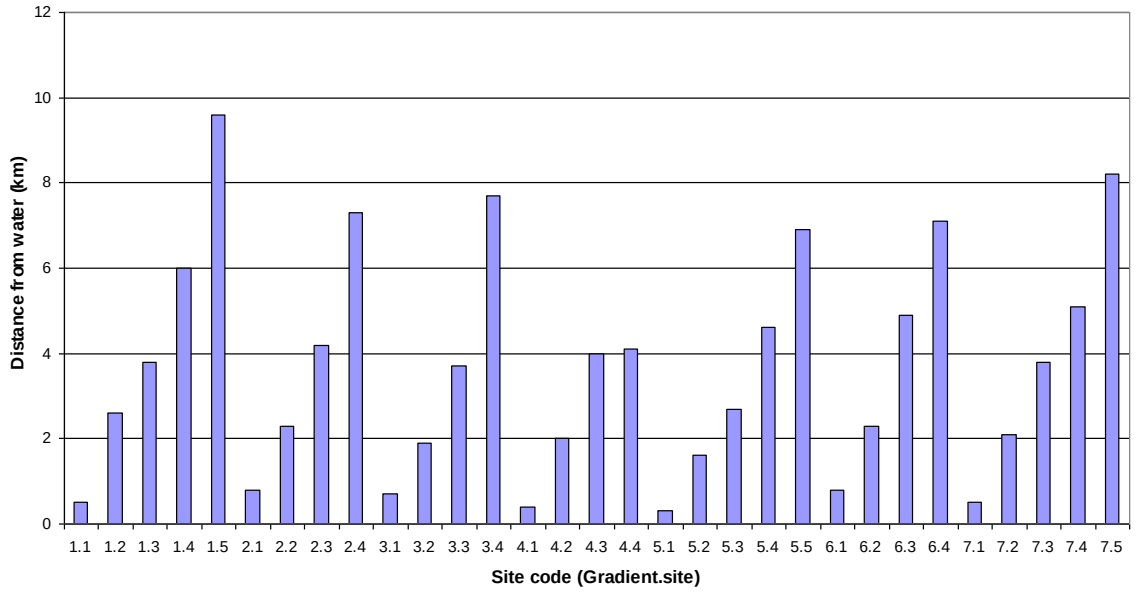


Figure 3.1 Site distances from water.

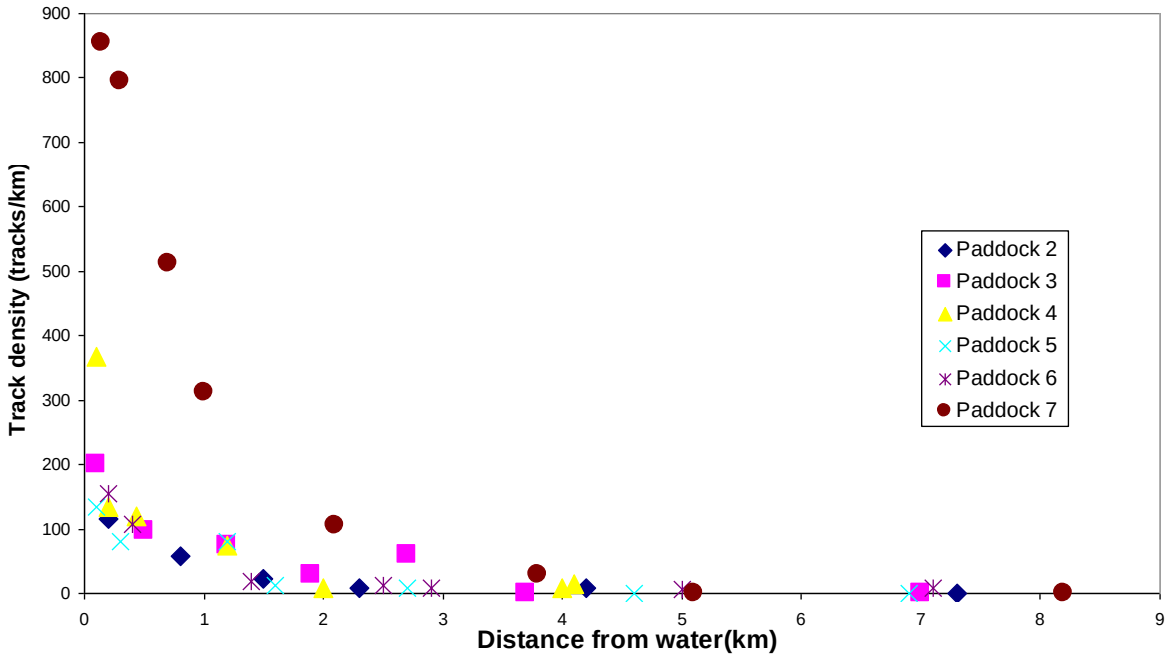


Figure 3.2 Influence of distance from water on track density.

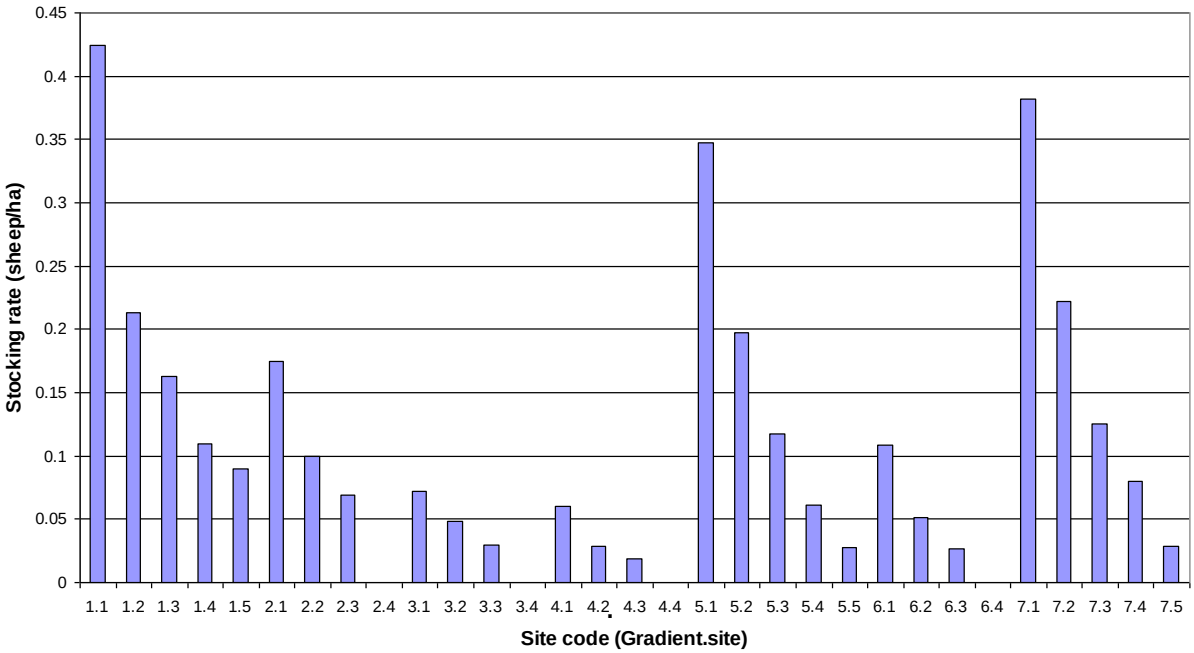


Figure 3.3 RANGEPACK/Paddock model of stocking rate at all sites.

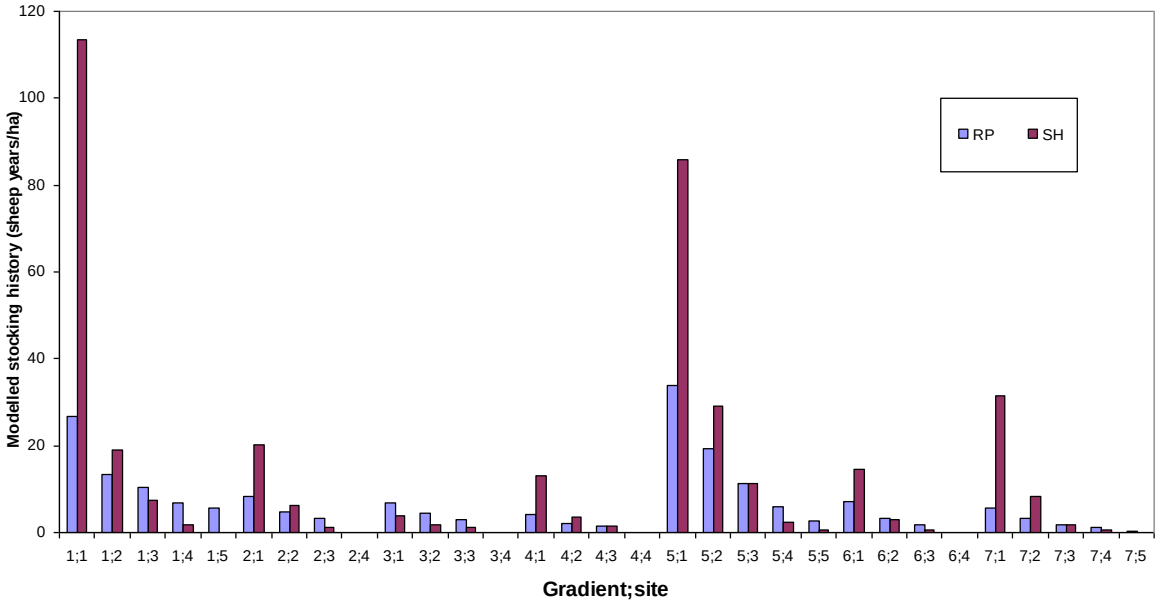


Figure 3.4 Comparison of RANGEPACK and the track density models of stocking rate history.

CHAPTER 4. RANGE CONDITION

4.1 Scope of this chapter

My principal aim in this chapter is to present an assessment of the impacts of grazing history on range condition as assessed using the formal approach of Agriculture Western Australia. I also interpret these principal findings according to local pastoralists' understanding of how range condition affects the capacity of different 'types of country' to support livestock in absolute and relative terms.

Thus the chapter is essentially an assessment according to traditional methods and objectives associated with the pastoral industry. My specific aims are to investigate how grazing history affects:

- range condition (based on the cover and species composition of perennial vegetation),
- Sustainable Carrying Capacity (pastoralists' interpretation of range condition classes in terms of the capacity of land to support livestock for a particular 'type of country') and,
- Pastoral Integrity (Sustainable Carrying Capacity relative to the undeveloped state).

Sustainable Carrying Capacity refers to the long term sustainable stocking rate: the stocking rate that is 'safe' in normal years (say 7 out of 10 years). A normal year in my study area consists of a reasonable or better winter season and a hot, dry summer (i.e. little ephemeral forage) (Pringle 1994a).

I commence with a discussion of factors that influence the capacity of land to support livestock and a brief discussion of traditional methods of range condition assessment in Western Australia. I then describe how range condition was assessed at my study sites, and how these scores were converted to estimates of Sustainable Carrying Capacity.

4.2 Brief overview of concepts

4.2.1 Factors influencing range condition

What influences the capacity of the rangeland to support sheep? Access to reasonably fresh water is critical, as I have discussed in the previous chapter. The next most important factors are seasonal conditions (which influence the amount of green, short-lived 'ground feed' or 'herbage' and landscape characteristics ('country types' in pastoralist parlance and

'landscape type' in this study) (Pringle 1994a; Pringle and Riches 1996). Palatable perennial plants are also important as they provide persistent forage that sustains sheep through dry times (Holm 1994; Ash *et al.* 1995). Impacts of past grazing management (i.e. grazing history) affect perennial plants and are the main focus of systems of range condition assessment currently used by many State government agencies across Australia, including Agriculture Western Australia (Tothill and Gillies 1992; Lange, Lay and Tynan 1994; State Of Environment Advisory Council 1997). There are many other influences on the amount and quality of feed available to sheep in time and space, some of which I refer to below.

4.2.1.1 Season

The study area, like most arid rangelands, is one of extremes in seasonal conditions (Gilligan 1994). During prolonged dry spells, sheep die from lack of food, even when fresh water is available. During particularly good seasons, there is far more food than sheep can eat and they generally prosper in whatever type of landscapes they graze. Climatic variation is the major determinant of the quantity and quality of feed available in this arid region (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; 1994; Pringle and Riches 1996).

Distance from markets and the substantial costs associated with mustering and transporting livestock make it impractical and uneconomic to 'track' forage availability by manipulating the number of sheep carried on stations in response to seasonal conditions (Foran and Stafford Smith 1991; Stafford Smith 1992). The viability of agistment strategies is strongly influenced by wool prices (Hodgkinson, Marsden and Hacker 1998), which have been very low since the late 1980s. Instead, pastoralists generally vary stock numbers according to longer-term variability: runs of good or poor seasons (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993).

Short-lived species are not included in the system of range condition assessment used by Agriculture Western Australia. I acknowledge the contribution that short-lived species make to pastoral production for much of the time (Wilson and Tupper 1982; Wilson and MacLeod 1991; Holm 1994; Ash *et al.* 1995). However, range condition assessments might be confounded by seasonal conditions if short-lived species were included in assessments. Range condition considers a longer time frame required for sustainable management based on perennial species.

Periods when forage availability becomes critically limiting occur approximately once per decade in my study area, during runs of prolonged poor seasons (Pringle 1994a; Gilligan

1994). At these times, perennial species provide the critically persistent forage in arid environments (Norton 1986; North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Roshier and Barcia 1993; Holm 1994) that may help pastoralists survive economically (Boykin, Gray and Caton 1962; Stafford-Smith and Foran 1992; Buxton and Stafford Smith 1996; Holecheck 1996). Recent economic modelling of economic performance of range condition treatments from a ten year study at Boolathana station in my study region confirm the economic importance of palatable perennial shrubs (David Atkins, unpublished report for Agriculture Western Australia).

4.2.1.2 Range type

Spatial heterogeneity is a long-recognised factor in rangeland management (Renner and Johnson 1942). Different types of country produce forage of varying quality and quantity, reflecting differences in moisture availability and nutrient status within and among landscapes (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Pringle 1994a; Pringle and Riches 1996). Pastoralists in my study region have suggested that long-term carrying capacity (i.e. sustainable stocking rates) varies by an order of magnitude according to range type (Pringle *et al.* 1992; Pringle and Riches 1996). Not only does range type influence carrying capacity, it also influences grazing distribution due to preferences shown by livestock at the landscape scale. Generally speaking, there is a strong positive relationship between carrying capacity and grazing preference (Landsberg and Stol 1996; Bailey *et al.* 1996).

4.2.1.3 Past grazing management

As with any system of resource use, excessive exploitation can degrade the capacity of the resource to sustain production. In rangelands across the world, the impacts of grazing on the capacity of the land to sustain pastoral production has traditionally been addressed under the concept of range condition (Sampson 1919; Dyksterhuis 1949; 1958; Wilson and Tupper 1982; Friedel 1991; Payne *et al.* 1998), by assessing the level of damage ('degradation') caused by past management. However, interventionist management can enhance resource values above the undeveloped (or 'pristine') state. Indeed, the introduction of the exotic *Cenchrus ciliaris* (buffel grass) to the Carnarvon Basin has purportedly increased the pastoral productivity of landscapes previously dominated by indigenous chenopod shrubs (Payne *et al.* 1987). In other words, pastoral management can improve the carrying capacity of the land as well as degrade it (Wilson and MacLeod 1991; Behnke and Abel 1996a; b; c; Abel 1997).

4.2.1.4 Other factors

While season, range type and history of grazing management are key influences on carrying capacity, there are many others. Fire is a common disturbance in grasslands in my study region, particularly spinifex hummock grasslands on extensive sand plains (Pringle 1994b).

These landscapes often support increased levels of production for a few years following fire (Tauss 1991). Hail damage associated with incursions inland of low-pressure systems from the north-west is most obvious on mulga plains. In extreme cases, it can cause soil erosion and consequent reduction in carrying capacity. Bladder saltbush (*Atriplex vesicarius*) plains are susceptible to occasional outbreaks of borers, leaving large tracts of dead low shrubs. Recovery may take a few years, but is usually complete. Gale-force winds from 'dry' (rain-less) cyclones can also do considerable damage, particularly in chenopod shrublands without a protective upper storey to dissipate near-ground wind velocities.

Grazing by large herbivores other than sheep and rabbits can also have a substantial impact on carrying capacity by increasing competition for forage (Wilson 1991a; 1991b; North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993). Feral goats, rabbits and kangaroos are common in my study region (Stevens 1994).

4.2.2 Assessment of range condition

4.2.2.1 Underlying theories and approaches

United States

Traditionally there have been two approaches to range condition assessment; both emanating from the United States of America. Agencies conducting assessments have often favoured one or other approach, though the approaches might best be used complementarily. 'Ecological' approaches are grounded in Clementsian succession theory (Clements 1916) and floristic similarity to a perceived 'climax' or 'pristine' state (Dyksterhuis 1949; 1958; Whitford *et al.* 1983). 'Production'-oriented approaches have developed on the basis of instantaneous assessment of forage abundance and quality (Humphrey 1945; 1949), but have not been as widely used (Smith 1989; Society for Range Management 1995). The ecological approaches have a longer-time frame and might best be used to guide strategic management, while instantaneous forage assessments might guide tactical management (e.g. seasonal manipulation of grazing).

Following widespread dissatisfaction with the theoretical basis of 'ecological' assessments (Smith 1978; Joyce 1993), there has been a move in the United States towards a more

complex and landscape-oriented (as opposed to botanical) approach to assessing rangelands (National Research Council 1994; West *et al.* 1994; Society for Range Management 1995; Busby, Ruyle and Joyce 1996). This approach is based on assessing 'ecological sites' (equivalent to land units in this work) in terms of the capacity of vegetation to protect the soil from 'accelerated' (land use-induced) erosion. This is assessed in terms of a Site Conservation Rating, for which a threshold zone separating 'satisfactory' from 'unsatisfactory' condition is determined. This system has been expanded to include 'healthy' and 'at risk' classes within the 'satisfactory' class of the Society for Range Management under the 'rangeland health' concept of the National Research Council (National Research Council 1994; Whitford 1995; Busby, Ruyle and Joyce 1996). This primary assessment according to the status of a site relative to the Site Conservation Threshold is conducted without regard to land use. It indicates the domain in which land uses should operate.

Management within the range of Site Conservation Ratings is according to identified 'Desired Plant Communities' which are essentially semi-stable states or domains of attraction. They are bounded by threshold conditions and separated by transition zones (Holling 1973; Westoby, Walker and Noy-Meir 1989b; Friedel 1991). In adopting the Desired Plant Community concept, the National Research Council and Society for Range Management are responding to long established criticisms of Clementsian Plant Succession Theory. These criticisms include the potential irreversibility of changes, the presence of semi-stable states rather than domains of gradual and predictable change, hysteresis in recovery trajectories and the limited capacity of manipulations of stocking rate to bring about desired changes (Connell and Slatyer 1977; Mentis *et al.* 1989; Smith 1989; Westoby, Walker and Noy-Meir 1989b; Friedel 1991; Laycock 1991).

Australia

There was some support for the use of the climax community-based approaches developed in the United States (Lendon and Lamacraft 1976). However, this approach has been used mostly by researchers as evidence of a grazing effect (Friedel, Cellier and Nicolson 1980; Friedel 1981), rather than in systematic assessments by State governments.

The controversy regarding 'ecological' and 'production' approaches to assessing range condition did not occur in Australia to the same extent as it did in the United States. This is partly due to the work of Allan Wilson and his colleagues (Wilson, Tupper and Tongway 1982; Wilson and Tupper 1982; Wilson 1984). They diverted attention from philosophical concepts towards clearly defined objectives. They proposed that range condition reflects the

consequences of grazing history (past grazing management) for the long-term capacity of land to support the production of meat and fibre, relative to the potential for that type of land to do so (Wilson and Tupper 1982). This definition recognises a clear land use objective and encourages a focus on qualities that influence this objective. In focusing so explicitly on a specific objective, the approach bypasses the philosophical debates that typify the history of the range condition concept in the United States of America (Smith 1989; Joyce 1993).

Determining how well the land can support grazing relative to its potential is philosophically 'cut and dried', but methodological problems remain (Wilson and MacLeod 1991; Pringle 1992; Roshier and Barcia 1993; Ash *et al.* 1995). None-the-less in Australia at least, this has been the conceptual basis for assessing range condition for some time (Wilcox and McKinnon 1972; Pringle 1991; Lange, Lay and Tynan 1994).

However, while the conceptual basis of assessment has been focussed on objectives, implementation has not been rigorous, partly because of the large research effort required to identify and calibrate indicators related to the objectives. Instead, assessors have tended to assess what is readily assessable and then attempt to interpret the implications of what they produce in terms of carrying capacity (Pringle 1991; Lange, Lay and Tynan 1994). There has been little effort to correlate assessed variables with direct measures such as forage production and persistence. That is not to say however, that these assessments have been without theoretical basis (Hacker 1984a; Gardiner 1986b; Morrissey and O'Connor 1988; Pringle 1995b).

An ecological approach based on perennial plant cover and composition has been used in range condition assessment for decades in Western Australia (Wilcox and McKinnon 1972; Mitchell, McCarthy and Hacker 1979). This approach was adopted in the belief that the degree of similarity in mixes and abundances/cover of palatable perennial plant species with ungrazed sites would correlate closely with the capacity of the land to support livestock (David Wilcox and Ron Hacker, personal communications).

It is relatively recently that these assumptions have been tested in this region, through grazing trials at Boolathana and Coodardy Stations (Holm 1994; Yan, Holm and Mitchell 1996). It appears that the assumptions do not hold in some circumstances and that the relationship between perennial species composition and forage and animal production are not as simple as previously perceived. Differences in production among 'better' classes of range condition may be overestimated. Differences in productivity among range condition classes may also be affected substantially by seasonal variations and the capacity of

pastoralists to respond to them without overgrazing during dry times (Norton 1986; Wilson and MacLeod 1991; Foran and Stafford Smith 1991; Roshier and Barcia 1993; Ash *et al.* 1995).

In order to investigate how well range condition reflects changes in pastoral productivity of my study area and to compare how grazing affects both attributes, I have translated range condition scores into estimates of Sustainable Carrying Capacity. To do so, I have used pastoralists' suggestions as to Suggested Carrying Capacities for combinations of types of landscapes and range condition, based on workshops held during the rangeland survey of my study area. I used pastoralists' suggestions on the basis that local knowledge accumulated over many years is probably a reasonable substitute for costly long-term research. It was suggested in the inaugural national state of environment report that the capacity to make inferences from disparate regional surveys of range condition was very limited (State Of Environment Advisory Council 1997). I will investigate whether these limitations also exist within a region (i.e. between types of landscape).

4.2.2.2 The evolution of range condition assessment in Western Australia

Formal assessment of the condition of pastoral resources by pastoral inspectors has occurred in Western Australia at least since the pastoral provisions of the Land Act were enacted in 1933. However the extent and severity of degradation were only realised once regional rangeland surveys of pastoral resources commenced in the late 1960s (Pringle 1991). These surveys systematically assessed and mapped range condition as part of the rangeland survey programme (Wilcox and McKinnon 1972; Mitchell, McCarthy and Hacker 1979; Pringle 1991; 1994a; Pringle, Payne and Burnside 1995).

Systematic range condition assessment in Western Australia commenced in the Gascoyne River catchment. This was essentially an investigation into possible links between overgrazing in the catchment and the frequency with which Carnarvon townsite was being flooded (Wilcox and McKinnon 1972). However, the widespread degradation that was reported led to the development of a State-wide rangeland survey programme, based on the cover and composition of perennial plants within their landscape context and inferences about their consequences for carrying capacity and pastoral production. Therefore the procedure blends ecological foundations (recognising range type and indicator species) with (inferred) production outcomes (Curry and Payne 1989; Pringle 1991).

The focus on perennial species is reasonable in this arid study region. They represent the ‘haystack’ for the inevitable runs of poor seasons. Costs of manipulating stock numbers in

response to seasonal conditions prohibit tracking seasonal availability of forage (David Atkins, unpublished report for Agriculture Western Australia involving economic modelling of data from a 10 year grazing trial). However, a strategy of destocking and agistment has been advocated for eastern Australian rangelands on the basis of simulation studies using wool prices from before their collapse in the late 1980s (Hodgkinson, Marsden and Hacker 1998). In arid rangeland regions, commercial pastoral enterprises are dependent on drought proofing that perennial forage species provide (Roshier and Barcia 1993; Holecheck 1996; Buxton and Stafford Smith 1996). Furthermore, unlike short-lived species, perennial species are less likely to reflect prevailing seasonal conditions and more likely to reflect the impacts of historical grazing. Nevertheless, ignoring short-lived plants underestimates their substantial contribution to pastoral production and may lead to over estimation of the extent of degradation (Wilson and Tupper 1982; Wilson and MacLeod 1991; Holm 1994).

In recent years there has been a move towards augmenting pastorally focussed assessments with more observation-based, ecological assessments (Pringle, Payne and Burnside 1995; Pringle and Burnside 1996). The criteria leading to a range condition assessment are recorded as well as the final assessment and some ancillary observations that might alter a non-pastoral perspective (e.g. nature conservation and mining impacts) of an assessed site. This involves recording observations such as the relative abundance of grazing-sensitive and grazing-favoured species (e.g. ‘woody weeds’), changes in cover and so forth, the presence of exotic species and their abundance and any non-grazing impacts (e.g. alteration of sheet flows by roads or railway lines). The current rangeland survey of the Pilbara region of Western Australia will be the first broad-scale use of this expanded system of range assessment.

The complexities of different pastoral management systems such as different management strategies (e.g. agistment, rotational grazing), types of livestock (e.g. ewes versus wethers, sheep versus cattle or goats) and types of product (e.g. wool versus mutton or sale lambs) have not been incorporated into range condition assessment in Western Australia. However, I recognise that perception of the productivity of any area of land may be influenced by the variation in management systems and objectives (Fletcher 1991; Behnke and Abel 1996a;b). These complexities are beyond the capacity of the system of regional range condition assessment used by Agriculture Western Australia. The current system is used in surveys, each covering thousands of square kilometres (Curry and Payne 1989) and it is not feasible to categorise each paddock assessed according to a complex combination of management attributes before making assessments. Furthermore, if the management system changed, such context-specific assessments would become redundant.

4.3 Methods used in the current study

In this section I described how the range condition was assessed and how the assessments were interpreted in terms of Sustainable Carrying Capacity and Pastoral Integrity.

4.3.1 Assessing range condition

I have adopted the current system of range condition assessment applied by Agriculture Western Australia and endorsed by local pastoralists in the study region (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Pringle 1994a; Pringle and Riches 1996). In doing this, I aim to represent these traditional approaches and values and test them within the wider context of ecosystem management.

An experienced member of the rangeland survey team (Alan Payne) assessed range condition. Alan Payne has been the leader of the rangeland survey programme since the early 1980s and has led several surveys in the arid shrublands of Western Australia (Payne *et al.* 1987; 1994; 1998). He is the most experienced range ecologist in Agriculture Western Australia.

Assessments were undertaken in November and December 1998. In order to avoid bias in assessment, I took the assessor to sites in random order, rather than starting at or working towards the watering points anchoring each gradient.

The rangeland survey system of assessment was adapted slightly to suit my survey design. Instead of driving past sites in order to make assessments, the assessor walked thirty paces up then down the central axis of each sampling locus. He scrutinised a band ten metres either side of the axis, which covers the sampling area for the other variables I assessed in this study. The guidelines used to assess range condition classes are presented in Table 4.1 (Pringle 1994b).

4.3.2 Estimating sustainable carrying capacities

I have linked range condition classes within each of the range types (at a land unit level) to Sustainable Carrying Capacities, using suggestions provided by local pastoralists at workshops organised during the rangeland survey of the area (Pringle *et al.* 1992) (Table 4.2).

The workshops took the form of two field days, one each with the Kalgoorlie and North Eastern Goldfields Land Conservation Districts. We took 25 pastoralists (eight from the

Kalgoorlie Land Conservation District Committee and 17 from the North Eastern Goldfields Land Conservation District Committee) to pre-selected examples of range types within their Districts. We also showed slides of these range types in different degrees of perceived modification. Pastoralists volunteered estimates of what they felt were stocking rates that would not degrade the resource over a normal year (a reasonable winter and a dry summer) for the various range types in different condition (Pringle *et al.* 1992).

Mean values were calculated for combinations of range types and range condition classes, from all estimates falling within one standard deviation of the original mean. Thus this analysis is an indicative data summary, rather than a formal statistical test or model. The results are presented in Table 4.2.

I have set mulga drainage lanes to the same values as groved mulga plains (which were assessed at the workshops) on the basis of their similar landforms, soils and vegetation. They differ principally in the level of contagion of mulga trees (see Chapter 2). We only asked pastoralists for their assessments of 'good', 'fair' and 'poor'. I have given 'very good' and 'good' the same values. One of the reasons we did not ask pastoralists to assess 'very good' range condition is that the two better classes were not consistently assessed during the rangeland survey (Pringle 1994b). Furthermore, many pastoralists stated that the difference between 'very good' and 'good', if it existed, had little to do with the capacity of the land to support stock. They indicated that the loss of the most grazing-sensitive species at a site would not have a significant impact on its carrying capacity. David Blood (personal communication), an Agriculture Western Australia range ecologist with a background in pastoral management, endorsed this view. I also set the 'very poor' values to zero, consistent with Agriculture Western Australia policy. Rangelands in this condition lack persistent forage in dry times and the circumstances for which we asked for assessments included a dry summer. Therefore this assignation is also reasonable purely in terms of the number of stock the land can support at such times.

4.3.3 Pastoral integrity

I have transformed the Suggested Carrying Capacities for each condition class into an index of Pastoral Integrity (Table 4.2). This involved expressing carrying capacities as a percentage of the suggested Sustainable Carrying Capacity for undeveloped country of that range type. The undeveloped state is synonymous with 'very good' range condition in this region (Pringle 1994b).

4.4 Results

4.4.1 Range condition

Range condition declines with Stocking History (SH) more rapidly in mulga landscapes than it does in chenopod landscapes (Figure 4.1). In other words, it requires less grazing to transform mulga landscapes into a severely degraded state (class 5: 'very poor' range condition).

Universal degradation (given enough time) is not as inevitable as the graph suggests. This analysis considers only the linear component of the response to grazing (i.e. I used linear regression, see Chapter 2.5) and 'very poor' range condition is largely confined to sites nearest water, which occupy a small proportion of paddock area. These degraded sites do not represent general paddock conditions.

The Stocking Rate (SR) model produced more significant and higher order interactions. There was a significant ($p=0.005$) interaction between landscape stratum and Stocking Rate (SR) (sheep/ha). All mulga landscape strata appear to be more easily degraded (i.e. have steeper slopes than the chenopod landscape) in this model. The interaction between landscape type and Stocking Rate (SR) was similar to that presented for Stocking History (SH), but it was more significant ($p<0.001$). While range condition improved with increasing distance from water (DW) ($p <0.001$), there were no interactions with landscape effects.

4.4.2 Sustainable carrying capacities

There is a considerably greater range of Sustainable Carrying Capacities in the chenopod landscape than in mulga landscape strata (Table 4.2). Sustainable carrying capacities decline with increased Stocking History (SH) and pastoralists clearly indicated that chenopod landscapes have greater capacity to support livestock ($p < 0.001$ for both grazing and landscape type, see Figure 4.2 and Table 4.4). Although there is no significant interaction between Stocking History (SH) and landscape strata, the degree of resource degradation is far greater in the chenopod landscape as a consequence of its considerably greater Stocking History (SH).

A similar pattern of highly significant effects was found using Stocking Rate (SR) derived from RANGEPACK/Paddock, although the grazing effect was more significant for Stocking Rate (SR) (Wald Statistic of 80.9 versus 29.0, see Table 4.4). Sustainable Carrying Capacities increase significantly faster with increasing distance from water (DW) in chenopod landscapes than in mulga landscapes ($p=0.004$).

4.4.3. Pastoral integrity

Pastoral Integrity declines more rapidly with increased Stocking History (SH) in mulga landscapes than in chenopod landscapes ($p = 0.007$) although chenopod landscapes can lose more Pastoral Integrity overall due to greater Stocking History (SH) (see Table 4.5 and Figure 4.3). Pastoral integrity is greater in chenopod landscapes than in mulga landscapes at all but the lightest levels of Stocking History (SH).

A highly significant interaction ($p < 0.001$) between landscape strata and grazing was found using the Stocking Rate (SR) model. Pastoral Integrity declines most rapidly with increasing Stocking Rate (SR) in mulga drainage lanes and least quickly in the chenopod landscape stratum. The range of Pastoral Integrity recorded was similar and large in these two landscape strata. The groved mulga landscape and sandy banks were intermediate in terms of rate of loss in Pastoral Integrity and they had a narrower range of Pastoral Integrity values (higher lower limit). Pastoral Integrity also increases with increasing distance from water (DW) ($p<0.001$), but without any significant landscape effects.

4.5 Discussion

All three estimates of grazing history accounted for systematic variation in the three measures of condition of pastoral resources. However, in terms of comparable Wald Statistics, the Stocking Rate (SR) model produced the most significant main effect (SR) in all three measures and provided the highest order interaction effects in two of three cases (effects were generally highly significant in terms of probability values). This is not surprising: the model was parameterised using grazing impacts on perennial vegetation to predict the distribution of grazing impacts in large arid zone paddocks such as my study paddocks (Cridland and Stafford Smith 1993).

Stocking History (SH) declines considerably more quickly with increasing distance from water than Stocking Rate (SR) (see Chapter 3) and the latter model was superior in accounting for variation in response variables. This suggests that while Stocking History (SH; my primary grazing estimate) may decline rapidly with increasing distance from water, its impact declines less rapidly (more linearly) away from water.

4.5.1 Range condition

Range condition declines more than five times quicker with increased Stocking History (SH) in mulga landscapes compared with chenopod landscapes and its highest Sustainable Carrying Capacity is 43 per cent of that assigned to chenopod landscapes. Thus the rate of

decline in range condition with increased grazing is faster than might be expected on the basis of its Sustainable Carrying Capacity relative to the rate of decline and Sustainable Carrying Capacity of chenopod landscapes. The vegetation structure and composition in mulga landscapes, and the system of assessment can explain the sensitivity of range condition to accumulated grazing. Palatable shrubs within the reach of sheep are generally sparse and hence for a given level of grazing pressure, they are more easily removed from the landscape than their more abundant chenopod counterparts. Furthermore, by focussing on palatable perennial plants in the understorey, the system assesses a minor component of these landscapes' vegetation. The majority of the forage produced in mulga landscapes is from short-lived species (Wilcox 1963; Pringle and Riches 1996; Noble, Greene and Muller 1998). In this respect, claims that assessments based on grazing-sensitive perennial plants (an 'ecological' approach) tend to exaggerate the extent and severity of overgrazing, and consequent degradation, have some basis (Wilson and MacLeod 1991; Holm 1994; Behnke and Abel 1996a; b; c). Losing most or all of a little can only ever entail losing a little.

This may explain why I have experienced problems persuading pastoralists to stock their mulga landscapes more conservatively. In hindsight, this may reflect their more realistic understanding of exactly what represents the grazing resource base: short-lived plants in fair or better seasons. The link between sparse perennial shrubs and pastoral production in mulga landscapes is therefore tenuous. To a lesser extent this may also apply to perennial ('wanderrie') grasses on sandy banks: these grasses provide abundant feed in fair or better seasons, but hay-off as seasons deteriorate (Freudenberger, Wilson and Palmer 1999).

The difference in management focus between production-based goals (i.e. treating mulga landscapes as seasonal pastures) and ecological goals (setting stocking rates to maintain palatable perennial species in the understorey) reflects a difference between private and public interests (Stafford Smith and Morton 1997). Pastoralists' private interests are in maximising sustainable production and profit, while an important public interest is the maintenance of regional populations of all species.

Can private and public interests be reconciled in my study landscapes? In chenopod landscapes both pastoral and public interests involve the maintenance of palatable perennial species. Pastoralists need palatable shrubs to provide the essential 'haystack' to survive droughts. It has been proposed that the biggest contribution pastoralists can make to regional nature conservation is to maintain the overall cover and composition of native vegetation (CALM 1992). Therefore in chenopod landscapes, there is considerable overlap in private and public interests as far as desirable management outcomes are concerned. It is

likely that pastoralism can be ecologically sustainable in chenopod landscapes if pastoral management goals are achieved.

The low densities of palatable perennial plants and their vulnerability to removal with grazing indicate that there is little overlap between goals associated with private and public interests in mulga landscapes. Pastoralists probably develop mulga landscapes to take advantage of their abundant seasonal forage, rather than to maintain what little drought durability (i.e. the sparse palatable perennial plants) these landscapes might posses. Indeed this outlook towards mulga landscapes has been proposed by pastoral advisers (e.g. Wilcox 1963). An approach that regards sparse palatable perennial species as dispensable is not in the public interest, so what might be done to reconcile private and public interests?

The simplest option would be to apply 'ecological' stocking rates; these are rates that are low enough for sparse palatable perennial species to persist. However, this is unlikely to be a viable option for pastoralists. The costs of development and maintenance of infrastructure might exceed returns from low numbers of sheep (particularly with prevailing low wool prices). In addition, pastoralists cannot afford to forgo the use of mulga landscapes. They may need to stock mulga landscapes heavily in good seasons to allow palatable perennial species in chenopod landscapes to recover vigour, particularly if they have recently been used as a 'haystack' (as recommended by Wilcox). Thus while mulga landscapes may have considerably lower pastoral potential compared to chenopod landscapes, they are an essential complement to these more productive landscapes at a station (and hence regional) scale.

It would appear that the solution lies in setting aside areas of mulga landscapes to be protected from grazing. Regional goals may be met at a paddock level in this instance. In chenopod landscapes, there is more chance of meeting conservation goals within well-managed paddocks.

4.5.2 Sustainable carrying capacities

The translation of range condition scores into Sustainable Carrying Capacities using local pastoralists' input produced similar trends to those based on field research in the south-western United States (Workman and Tanaka 1991; Workman 1995). The differences in productivity between successive range condition classes declines with improving range condition: the difference in productivity between poor and fair range condition is greater than the difference between fair and good. Differences between range condition classes (e.g. fair versus good) are also greatest in most productive landscapes. It is possible then that

pastoralists manage productive landscapes most carefully, and consider some degree of degradation of less productive landscapes as a partial trade off for maintaining their best pastures in good range condition. Results from a survey of my study region support this proposition (Pringle 1994a).

The translation of range condition scores into locally suggested Sustainable Carrying Capacities illustrated misgivings that were highlighted in the State of Environment Report Australia 1996 (State Of Environment Advisory Council 1997). The criticism was that range condition scores cannot be interpreted into absolute estimates of the resource, such as carrying capacities, for comparison between regions. Range condition assessments indicated that the major issue in natural resource management occurs in mulga landscapes (on the basis of the slopes of the fitted effects). However, the opposite became apparent when local pastoralists interpreted these assessments in terms of the capacity of the land to support livestock. Thus, the concerns regarding the interpretability of range condition scores between regions also apply between types of landscapes within regions.

Range condition assessment was designed to assess the condition of rangelands and facilitate their better management (Dyksterhuis 1949). The concept was never meant to provide absolute estimates of loss of pastoral productivity to contemporary regional and national assessments such as State of Environment reporting. However, I have illustrated an approach with which this could be done, substituting decades of accumulated local experience for detailed long term research in interpreting range condition assessments in different range types.

4.5.3 Pastoral integrity

Similar problems with interpretation emerged when the suggested Sustainable Carrying Capacities were translated into an index of Pastoral Integrity. Like range condition, Pastoral Integrity is a relative measure and is specific to each landscape or parts of landscapes. Therefore it has limited use for comparing directly impacts of grazing on pastoral productivity between landscapes or parts of landscapes.

Never the less, the Pastoral Integrity index can be useful as a summary measure. It could be used at a local level (i.e. as a measure of what proportion of pastoral value has been retained in a certain landscape). It could also be useful for a regional summary (by considering the impact of grazing on different types of landscapes within a region and then deriving a summary measure by weighting for the area of each landscape type assessed). However, problems with interpretation would arise if one then compared across regions. A

solution is to consider Sustainable Carrying Capacity and Pastoral Integrity together. One might then conclude for instance that one region has lost a greater proportion of its potential to support stock than another, but the latter may have lost more (absolute) capacity to do so because it is an inherently more fertile region.

4.5.4 Synthesis of the response variables

That grazing effects are so pronounced on all three measures analysed indicates that pastoral resources decline with increasing grazing history. More specifically, I have shown that range condition declines with increasing grazing history and pastoralists have confirmed that this translates into a loss in the capacity of the land to support livestock. This is particularly relevant in chenopod landscapes. In mulga landscapes, 'range condition' may not correlate well with actual carrying capacity or animal production until a state of deterioration is reached in which the productivity of the soil is degraded. At least in chenopod landscapes that are managed by year-round continuous grazing (as is the norm in my study region), low stocking rates will best conserve pastoral resources. This means maintaining the essential composition and cover of grazing-sensitive perennial plants (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Pringle 1995b).

There is considerable evidence that palatable perennial species need periods of rest form grazing in order to recruit new individuals and recover vigour (and so reduce rates of mortality) (Hacker *et al.* 1991; Hodgkinson 1993; Hodgkinson, Marsden and Hacker 1998; Hunt submitted). If agistment is economically unfeasible, there may be a need for pastoralists to practice a system of rotational resting of paddocks, such that paddocks are relieved of grazing pressure every few years, or at times of vulnerability to degradation (Hacker and Hodgkinson 1996).

Although the process of transforming range condition scores into Sustainable Carrying Capacities was based on local pastoralists' input, this could equally have been done more scientifically, had the necessary information been available. I chose to use pastoralists input for two reasons. Firstly, I wanted to represent them as a constituency or stakeholder group. Secondly, local, anecdotal evidence developed over decades of management can often be a more reliable model of complex system behaviour than a limited amount of research (Stafford Smith 1988). It might take decades of research to determine what levels of grazing are 'sustainable' under different conditions in an arid, highly variable climate. Similar approaches employing local knowledge have been used elsewhere in Australia (Forge 1994; Scanlan *et al.* 1994; Johnston, McKeon and Day 1996; Johnston, Tannock and Beale 1996) and overseas (Bosch and Booysen 1992; Pulina *et al.* 1999).

4.6 Summary

(1) A professional rangeland survey officer using standard assessment techniques assessed range condition at a land unit level at all of my study sites.

(2) Range condition scores were translated into Sustainable Carrying Capacities suggested by local pastoralists.

(3) Stocking History (SH), Stocking Rate (SR) and distance from water (DW) were all useful predictors of grazing impacts, with each producing interactions with landscape type or strata. I have focussed on the effects of Stocking History (SH).

(4) Range condition declines with increasing grazing history and increasing proximity to watering points. It declines more quickly in mulga landscapes due to the vulnerability of sparse palatable perennial plants.

(5) The achievement of private and public goals in paddocks characterised by mulga landscapes may prove difficult. Palatable perennial plants appear to be dispensable from a pastoral perspective and 'ecological' stocking rates to retain most vulnerable species may require more intense management such as rotational spelling. Whether increased management costs under prevailing wool prices in such low carrying capacity country can be justified is questionable. Ecologically sustainable pastoralism may therefore require a network of protected areas to overcome the conflict between private and public interests.

(6) The translation of range condition classes into estimates of Sustainable Carrying Capacity shows that range condition assessment can obscure the relative impacts of grazing on the capacity of the land to support livestock in different landscapes.

(7) While loss of absolute pastoral value with increasing Stocking History (SH) occurs at the same rate across all landscape strata; the dynamic range of Sustainable Carrying Capacities is far greater in chenopod landscapes. This reflects the inherently higher pastoral value of chenopod landscapes and their capacity to lose all pastoral value under stocking histories that far exceed those experienced in mulga landscapes.

(8) In the chenopod landscapes of my study region in particular, the strong influence of grazing on range condition, coupled with pastoralists' interpretation that this severely diminishes land capability, indicates that conservative pastoral management (light stocking rates to conserve palatable perennial plants) is needed for sustaining pastoral resources. Some form of resting strategy may also be needed.

(9) Local pastoralists' suggestions about Sustainable Carrying Capacities may have general applicability for any inter-regional or national interpretation of range condition assessments (in terms of pastoral productivity) and they represent a timely substitute for long-term research.

Table 4.1 Range condition classes used in assessment

(From Pringle, Van Vreeswyk and Gilligan 1994)

Class	Condition indicators
1	Excellent or very good. For the land unit-vegetation type, the site's cover and composition of shrubs, perennial herbs and grasses is near optimal, free of obvious reductions in palatable species or increases in unpalatable species
2	Good. Perennials present include all or most of the palatable species expected; some less palatable or unpalatable species may have increased, but total perennial cover is not very different from the optimal.
3	Fair. Moderate losses of palatable perennials and/or increases in unpalatable shrubs or grasses, but most palatable species and stability desirables still present; foliar cover is less than on comparable sites rated 1 or 2 unless unpalatable species have increased.
4	Poor. Conspicuous losses of palatable perennials; foliar cover is either reduced through a general loss of perennials or is increased by invasion of unpalatable species.
5	Very poor. Few palatable perennials remain; cover is either greatly reduced, with much bare ground arising from loss of stability desirables, or has become dominated by a proliferation of unpalatable species.

Table 4.2 Translation of range condition classes

Land unit	Range condition class	Sustainable carrying capacity (sheep/hectare)	Pastoral integrity (%)
Chenopod flat	1	0.167	100
	2	0.167	100
	3	0.091	55
	4	0.05	30
	5	0	0
Mulga drainage lane	1	0.067	100
	2	0.067	100
	3	0.059	88
	4	0.037	55
	5	0	0
Groved mulga plain	1	0.067	100
	2	0.067	100
	3	0.059	88
	4	0.037	55
	5	0	0
Wanderrie bank	1	0.083	100
	2	0.083	100
	3	0.05	60
	4	0.029	34
	5	0	0

Table 4.3 Effects of grazing history on range condition

(See Glossary: Appendix 1 for terminology and Chapter 2 for explanation of modelling terms.)

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking History, derived from modelling track densities and expressed in sheep years per hectare	landscape type	0.1	1	n.s.
	SH	17.1	1	< 0.001
	stratum.landscape type	56.9	3	< 0.001
	landscape type.SH	8.7	1	0.003
	stratum.landscape type.SH	0.9	3	n.s.
SR: Stocking Rate; average long term stocking rate developed from RANGEPACK/Paddock model and expressed in sheep per hectare	landscape type	0.0	1	n.s.
	SR	62.4	1	< 0.001
	stratum.landscape type	58.8	3	< 0.001
	landscape type.SR	14.4	1	< 0.001
	stratum.landscape type.SR	12.7	3	0.005
DW: Distance from Water, measured in kilometres	landscape type	0.1	1	n.s.
	DW	60.2	1	< 0.001
	stratum.landscape type	58.0	3	< 0.001
	landscape type.DW	1.0	0	n.s.
	stratum.landscape type.DW	4.9	3	n.s.

Table 4.4 Effects of grazing history on sustainable carrying capacities

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking History	landscape type	21.9	1	< 0.001
	SH	29.0	1	< 0.001
	stratum.landscape type	18.9	3	< 0.001
	landscape type.SH	0.5	1	n.s.
	stratum.landscape type.SH	0.5	3	n.s.
SR: Stocking Rate	landscape type	23.5	1	< 0.001
	SR	80.9	1	< 0.001
	stratum.landscape type	18.5	3	< 0.001
	landscape type.SR	0.6	1	n.s.
	stratum.landscape type.SR	3.6	3	n.s.
DW: distance from water	landscape type	10.3	1	0.001
	DW	42.5	1	< 0.001
	stratum.landscape type	18.8	3	< 0.001
	landscape type.DW	8.5	1	0.004
	stratum.landscape type.DW	0.3	3	n.s.

Table 4.5 Effects of grazing history on pastoral integrity

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking History	landscape type	1.1	1	n.s.
	SH	23.0	1	< 0.001
	stratum.landscape type	2.3	3	n.s.
	landscape type.SH	7.3	1	0.007
	stratum.landscape type.SH	4.8	3	n.s.
SR: Stocking Rate	landscape type	0.8	1	n.s.
	SR	55.4	1	< 0.001
	stratum.landscape type	2.0	3	n.s.
	landscape type.SR	4.1	1	0.04
	stratum.landscape type.SR	21.0	3	< 0.001
DW: distance from water	landscape type	0.9	1	n.s.
	DW	39.9	1	< 0.001
	stratum.landscape type	2.4	3	n.s.
	landscape type.DW	0.2	1	n.s.
	stratum.landscape type.DW	5.3	3	n.s.

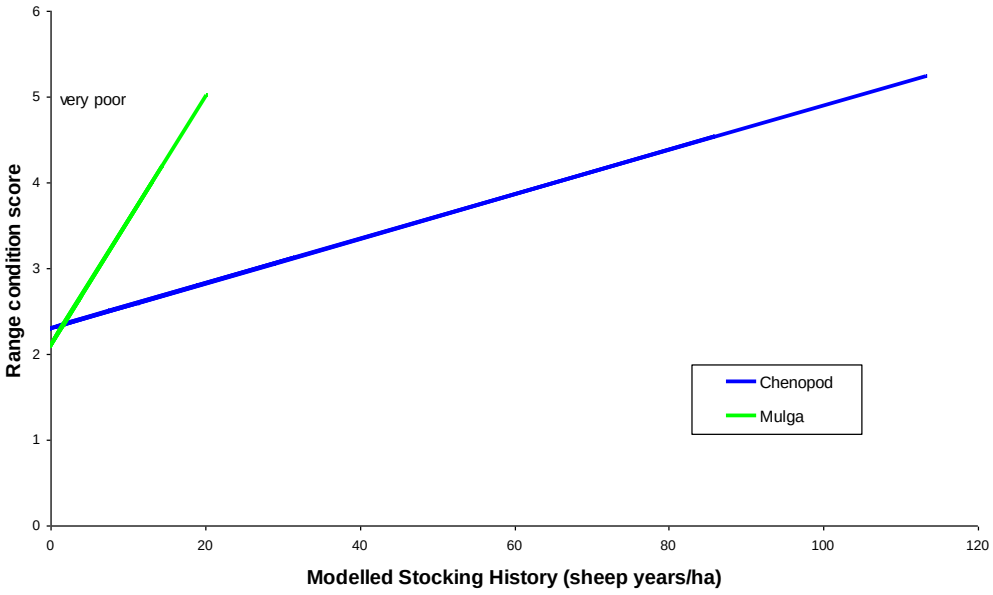


Figure 4.1 Impacts of stocking history on range condition.

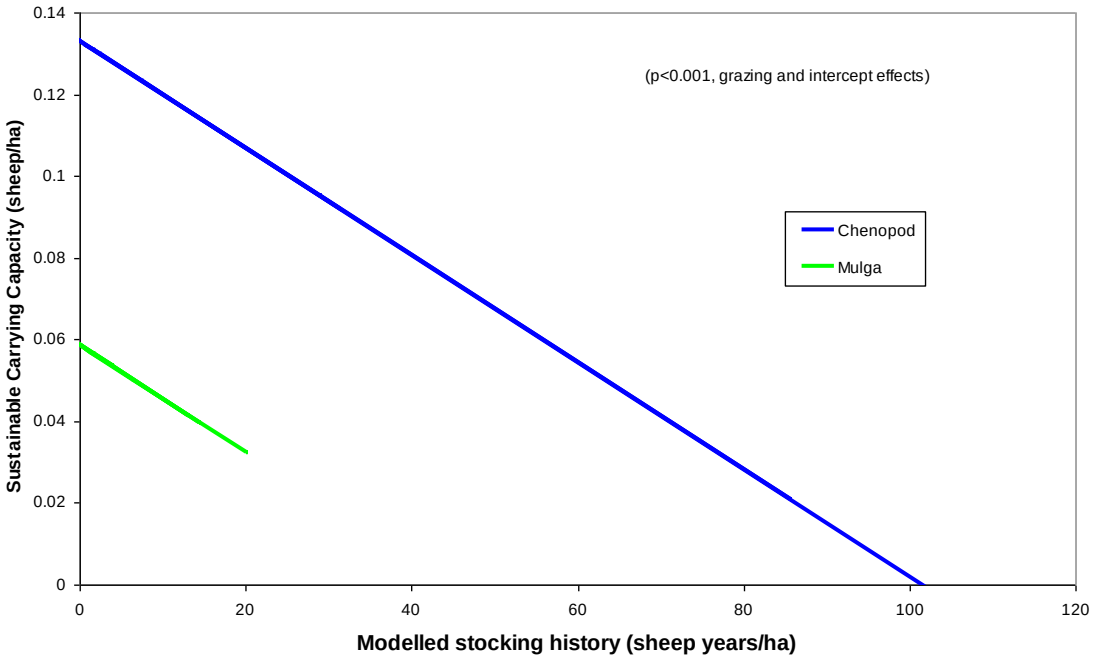


Figure 4.2 Impacts of stocking history on sustainable carrying capacity.

Figure 4.3 Impacts of Stocking History on Pastoral Integrity

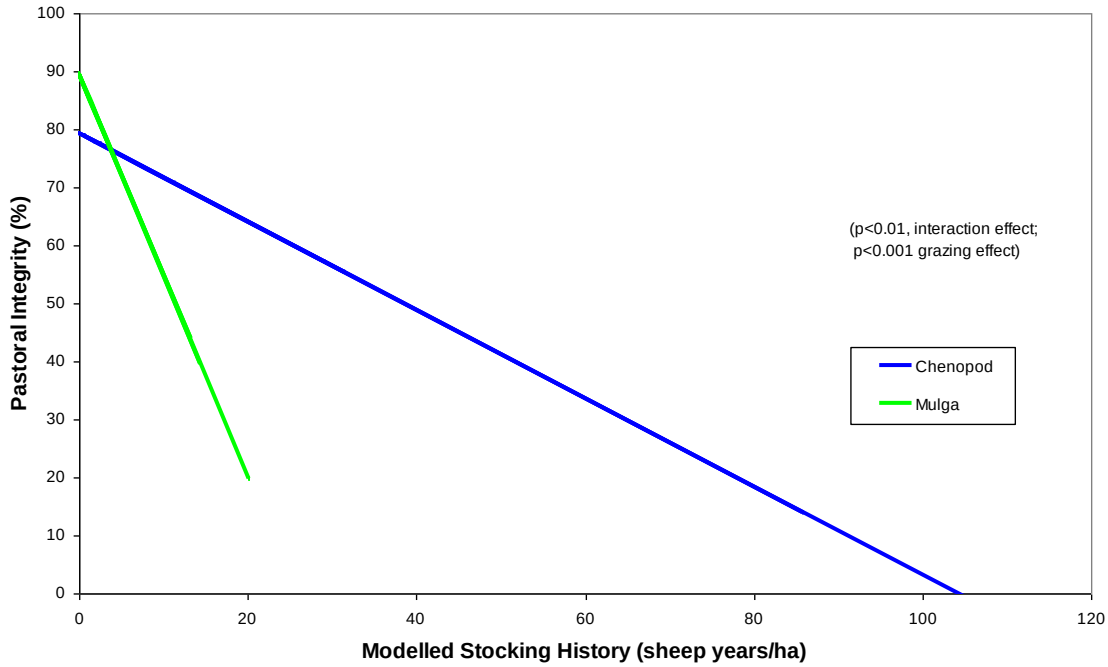


Figure 4.3 Impacts of stocking history on pastoral integrity.

CHAPTER 5. PLANT DIVERSITY

5.1 Introduction

In this and the following chapter, I investigate the impact of grazing on aspects of biodiversity within rangeland paddocks. I include a brief overview of biodiversity, including the focus of this study and its relevance to nature conservation. I discuss some major factors affecting the distribution of plant diversity in rangelands, before describing what I did in the field and how I have analysed and interpreted the data and results.

5.1.1 What is biodiversity?

Biodiversity (or 'biological diversity' in its fuller form) gained eminence in the World Conservation Strategy (Anon. 1980) in the context of ecologically sustainable development, which seeks to balance the interests of development with nature conservation. Biodiversity became a collective term representing nature conservation concerns. It is now the focus of an international convention and in Australia has become a key concept in national policy (e.g. The National Biodiversity Strategy) and legislation (e.g. the recently enacted Environment Protection and Biodiversity Conservation Act 1999) in Australia.

A widely accepted definition of biodiversity is:

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

(State Of Environment Advisory Council 1997; Saunders, Margules and Hill 1998).

Biodiversity therefore includes every living organism and where and with what other organisms it lives. Biodiversity is all encompassing and so is also impossible to assess in total over even the smallest of areas. The best we can do is focus on subsets of biodiversity, selected for their relevance to specific interests or concerns.

5.1.2 Two major avenues of research on biodiversity

5.1.2.1 Planning protected areas

Early use of the concept of biodiversity was based on the idea that areas of high biodiversity (many species) were of particularly high conservation value (World Conservation Monitoring Centre1992). This approach tended to overlook the fate of individual species and was as

such flawed for assessing anthropogenic disturbances. It focussed on 'high numbers', almost regardless of the identities of the species and levels of fragmentation and extirpation of their populations (Landsberg, O'Connor and Freudenberger 1999). In addition, species are also only one aspect of biodiversity.

More recently, the idea of complementarity has become established in planning networks of protected areas (or 'reserve systems'). In essence, complementarity relates to the efficiency of allocating areas for protection within an overall region based on complementary suites of biodiversity (mainly species, but also habitats, landforms and so forth). Instead of focussing on 'hotspots' alone, complementarity looks to protect the majority of a region's biodiversity in complementary areas. In this case, the members or units of biodiversity matter as much as their 'richness' (Margules, Nicholls and Pressey 1988; Margules and Stein 1989; Rebelo and Siegfried 1990; 1992; Margules, Pressey and Nicholls 1991; Nicholls and Margules 1993; Pressey *et al.* 1993; Margules and Redhead 1995; Pressey and Logan 1995; Walker 1995).

5.1.2.2 Ecosystem function

Given that biodiversity emerged as a collective term to represent nature conservation, it is not surprising that value-laden interpretations have been implicit in its use. While conservation biologists talked about conserving biodiversity, they usually inferred or directly stipulated native or local biodiversity (e.g. Noss 1990). For instance, exotic, invasive species were undesirable and not to be conserved beyond their natural ranges. This seems quite rational, but confusion arises if the sole measure of biodiversity is number of species regardless of their identities and distribution.

An example of such confusion is the recent research interest in the relationship between biodiversity and ecosystem function. Initially, much emphasis was on the influence of biodiversity measured in terms of numbers of species, regardless of their identity (Ehrlich and Ehrlich 1981; Walker 1992; Lawton 1994; Tilman and Downing 1994; Tilman, Wedin and Knops 1996). There is growing acceptance that species identity matters and that the influences of biodiversity are more likely to reflect functional composition than a simple arithmetic measure such as species richness (Naeem *et al.* 1996; Chapin *et al.* 1997; Grime 1997; Huston 1997; Walker, Kinzig and Langridge 1999).

It appears therefore, that from both perspectives of biodiversity discussed, arithmetic measures are useful but insufficient. Consideration of species identity (or functionality) is a necessary extension of studies into reserve planning or ecosystem function.

5.1.3 'Biodiversity' concerns in rangeland paddocks

Since biodiversity includes all living things, one is forced to choose aspects of biodiversity within the hierarchy extending from genes to continents. It is also important to choose a scale of sampling that is meaningful to both the aspects of biodiversity under study, and the fundamental questions of interest (Allen and Starr 1982; Noss 1990; Ahl and Allen 1996).

There has been considerable assessment and theorising about nature conservation (mostly at broad scales) in Australian rangelands (Pressey and Nicholls 1991; Morton *et al.* 1995; Pressey and Logan 1995; Pringle 1995a; James, Landsberg and Morton 1999), but little practical application (State Of Environment Advisory Council 1997).

It has been suggested that ecologically sustainable development, including implementation of systematic nature conservation, should have a regional focus for implementation in Australia's rangelands (Morton *et al.* 1995; Noble *et al.* 1998). Persistence of native species in rangeland paddocks is an important contribution to the regional maintenance of biodiversity, particularly in light of the general lack of systematically planned and representative reserve systems (Specht, Roe and Boughton 1974; Pressey and Logan 1995; Pringle 1995a; Thackway and Cresswell 1995).

My study is concerned with conserving species threatened by increasingly widespread anthropogenic disturbances. I am interested in investigating 'winners' and 'losers' (*sensu* McKinney and Lockwood 1999) under pastoral management. Minimising numbers of losers is a critical issue in redressing the balance in environmental policy between pastoralism and nature conservation. On the basis of a review of trends in conservation biology, Caughley (1993) has argued that the major concern for species conservation is the decline in abundance and distribution of widespread, rather than small and restricted, species.

The focus in this chapter is vascular plants in the standing vegetation. Recent work in the arid rangelands of Australia clearly indicates that, in dry seasons, as much as half of the flora may be present only as seed at a site (Landsberg *et al.* 1999). However, I was not able to include this component of the flora because of time constraints, although I was fortunate in undertaking sampling during a succession of wet seasons, when much of the seed bank flora was likely to be present in the standing vegetation.

My approach is to analyse distribution and abundance of species along spatially arranged gradients of grazing history at paddock scales (Lange 1985) in order to identify increaser and

decreaser species (Landsberg, James and Morton 1997) (see Chapter 3). An increaser species is one whose abundance tends to increase as levels of grazing increase; decreasers show an opposite trend (Dyksterhuis 1949; Wilson and Tupper 1982; Pringle 1994b).

The number (and proportion) of native decreaser and increaser species within any landscape type is an indication of how well that flora has adjusted to the novel pressures associated with pastoral development. Numerous decreaser species would indicate that the regional flora is not well adjusted to the imposed management regime. Populations of decreaser species presumably struggle to persist near watering points and are restricted (to varying degrees) to less-heavily grazed parts of paddock landscapes. A preponderance of increaser species might also indicate that a large proportion of the flora has not adjusted well to grazing. However this would not necessarily preclude a similar number of decreaser species from occurring, unless many had become locally extinct (McIvor 1998), which is not believed to be the case in my study region (Curry and Hacker 1990; Pringle 1998b).

I also consider exotic species, but separately. Their prominence reflects the availability of propagules and the invasibility of a landscape type; its predisposition to become dominated by exotic species. Exotic perennial species are a minor component of the flora of my study region (Dell *et al.* 1988; Dell, How and Milewski 1992; Pringle 1994c; 1998c). Little is known of the richness and invasive characteristics of exotic ephemeral plant species that may occur, beyond those that are 'declared' agricultural weeds (Pringle, Van Vreeswyk and Gilligan 1994; Payne *et al.* 1998).

In summary, my interest is in identifying the relative proportions of native decreaser and increaser species (they are the native 'winners' and 'losers' respectively) as well as exotic species. The approach can only be indicative of trends because it is unlikely that I could sample the local flora in my study paddocks, let alone the regional flora.

5.1.4 Key biological questions

I pose the following key questions:

(1) Does grazing affect plant species richness?

(2) Does grazing affect floristic composition?

(3) What proportions of the flora appear to be either advantaged or disadvantaged by grazing?

(4) Do the proportions of increaser and decreaser species differ among parts and types of landscapes?

(5) Does grazing affect increaser and decreaser response groups to differing degrees in different types or parts of landscapes?

(6) What are the implications of patterns in paddocks to regional nature conservation?

5.2 Plant diversity in the north-eastern Goldfields

There are many factors that affect plant species composition in the rangelands. I deal with some of the more obvious ones briefly, with an emphasis on my study area. Clearly, there are interactions among these factors, but I do not discuss them in any detail. Much of this information is taken from a recent rangeland survey of the study region (Pringle, Van Vreeswyk and Gilligan 1994), with some reference to the Vegetation of Western Australia series (Beard 1990) and the Biological Survey of the Eastern Goldfields series (Dell *et al.* 1988; 1992).

5.2.1 Natural environmental factors

By 'natural' I mean factors that are related to inherent biogeographical and environmental variation and factors, as opposed to factors that derive principally from land use, in this case pastoralism. Clearly, there will be interaction between natural and pastoral factors.

5.2.1.1 Landscape factors

Soil type is a major influence on vegetation in this region (Beard 1976; Dell *et al.* 1992; Pringle 1994c). In general, chenopod shrubland steppe occurs on alkaline, saline texture contrasting soils. These shrublands are most prominent on plains flanking salt lake systems, but are also found downslope of granite breakaways and on weathered greenstone footslopes and stony plains. Acacia low woodlands/tall shrublands are usually dominated by mulga (*Acacia aneura*) and occur on neutral to acidic red earths. This vegetation is most extensive on sheet-flood plains, which are underlain by a (natural) impervious hardpan (Van Vreeswyk 1994). On sandier soils with similar pH, tussock 'wanderrie' grasses (e.g. *Eragrostis eriopoda* and *Monachather paradoxa*) are found with variously developed shrub layers. Finally, spinifex (*Triodia* spp.) hummock grasslands are found on soils that are even coarser in texture.

5.2.1.2 Climate factors

My study gradients are aligned in a sequence running approximately north – south across the region, resulting in only minor systematic variation in climate. Temperatures and rainfall are often slightly higher in the north during summer. However, increased evaporation of soil water tends to balance out the higher rainfall in terms of likelihood of a successful summer

season. There tends to be slightly more winter rainfall in the south of the study area (Gilligan 1994). These patterns may be reflected in more prolific tussock grass growth in good summers in the north of the survey area, and slightly denser shrub growth in the south. These patterns are aligned approximately south-west to north-east and are described in the rangeland study of the area immediately west of the survey area (Pringle 1998b; 1998c).

There is considerable temporal variation in climatic influences. In good winter seasons, daisies (Asteraceae) and other succulent forbs are common, while good summer seasons often produce grasses and cotton bushes (*Ptilotus* spp.). Every once or twice a decade there is a good summer season and tussock grasses become established. The abundance of tussock grasses is strongly influenced by number of years since the last good summer rains; mortality of C_3 grasses is high in dry periods (Mitchell and Wilcox 1984; Pringle 1994b). My sampling was undertaken toward the end of two years of almost continuous good seasons. I suspect that grazing activity at the time of sampling was diffuse and not focussed around watering points, as forage availability had outstripped demand for a prolonged period prior to it.

Rainfall events can also be extremely patchy, particularly in summer, when very localised thunderstorms are common (Gilligan 1994). Fortunately, rainfalls around the time of my sampling appeared to have been widespread and there was no sign of a variation in seasonal conditions that might have confounded my sampling strategy.

5.2.1.3 Natural disturbance regimes

Fire is the most obvious natural disturbance regime in the study area, but it is more usually associated with spinifex hummock grasslands than the landscapes studied. Fire may occur on wanderrie sandy banks after good summer rains and prolific grass growth, but no evidence of fire was found. Fire is not regularly used for pasture management in the landscapes I studied.

Outbreaks of herbivorous insects are rare, but can be severe in both mulga and chenopod landscapes. My study sites did not appear to have been affected recently. Hail storms can also have major influences on vegetation. One sampling locus (at site three at Gradient 6SM) appeared to have been affected by hail several years ago.

5.2.2 Pastoral impacts

Pastoral impacts in addition to those caused by livestock include the introduction of other exotic biota (feral grazing animals and predators), control or eradication of native herbivores (kangaroos) and carnivores (dingoes and wedge-tailed eagles), and replacement of indigenous land management (particularly involving alteration of fire regimes). Impacts of livestock and feral and native herbivores include not only grazing, but also trampling, defecation, urination, camping and track development. I focus on those aspects relevant to the objectives and design of this study in the brief review below. I briefly discuss impacts of grazing and associated activities in general terms, then focus on spatial patterns within paddocks resulting from provision of artificial sources of water. I have not attempted a comprehensive review of the literature, but have instead focussed on those studies of most relevance to arid Western Australian rangelands.

Pastoral infrastructure can also affect rangeland ecosystems. For instance, the development of management access tracks can have profound effects on water-starved vegetation downslope. However, I have avoided these effects in selecting sites.

5.2.2.1 Impacts of grazing activity

Range condition assessments based upon responses of perennial plant species to grazing have been conducted for decades in the region (Wilcox and McKinnon 1972; Wilcox 1973; Mitchell, McCarthy and Hacker 1979; Curry *et al.* 1994). Assessment includes classification of indicator species based on locally accumulated wisdom augmented by a small number of research trials and grazing demonstrations (Hacker 1984a; 1984b; 1986; Gardiner 1986a; 1986b; Fletcher 1991; Holm 1994; Yan, Holm and Mitchell 1996; Watson 1997; Watson, Westoby and Holm 1997a; 1997b). However, most of the species studied and subsequently classified are common perennial pasture species or perennial increaser species (‘undesirables’). Less common perennial species and ephemeral species have largely been overlooked, leaving a large gap in our understanding of the total flora's response to grazing.

One exception is the PhD research of Ron Hacker, who assessed responses of 20 ground storey species to range condition within the same region as my study sites (Hacker 1984b). Uncommon species were excluded from that study on the basis of their potential to distort the analysis which used reciprocal averaging. The remaining species were classified into increaser, decreaser and equivocal groups according to their response to ‘range condition’. Among the shorter-lived ground storey species, *Podolepis canescenscens* and *Swainsonia* spp. were missing only from most degraded sites and *Brunonia australis* and *Podolepis*

lessonii were found only at most degraded sites. The two increaser species were rare (frequency < 5 per cent) and Hacker suggested that sampling error and local factors, as well as range condition may have influenced their distributions.

General patterns of grazing impact in this region are similar to those reviewed across the Australian arid zone by James, Landsberg and Morton (1999). Heavy grazing can either reduce the cover and density of perennial species, or result in infestation by 'increaser' species that replace more palatable perennial species. However, as in other parts of Australia (Osborn, Wood and Paltridge 1932; Andrew and Lange 1986b; Wilson 1990), heavy grazing tends to reduce perennial species richness. This decline in species richness occurs because there are usually fewer generalist 'increaser' species than 'decreaser' species (Pringle 1994b). This may reflect the contrast between pastoral and evolutionary regimes of herbivory (Milchunas, Sala and Lauenroth 1988; Milchunas and Lauenroth 1993; Landsberg, O'Connor and Freudenberger 1999); most Australian perennial plants have not evolved under the elevated and continuous grazing activity found in contemporary arid rangelands. This may also explain why an intermediate disturbance hypothesis is seldom evoked to explain grazing impacts on biodiversity in Australia (James, Landsberg and Morton 1999), in contrast to rangelands in other parts of the world (Archer, Garrett and Detling 1987; Chaneton and Facelli 1991; West 1993).

Rarity and genetic isolation have proved to be good predictors of the likelihood of extirpation (Pimm, Jones and Diamond 1988; Saunders, Hobbs and Margules 1991; Reid and Fleming 1992; Hobbs and Saunders 1993; Keith 1998; Cofre and Marquet 1999). By focussing on widespread and common species, traditional rangeland science has possibly overlooked those species most likely to become locally extinct due to grazing and so be a major concern for nature conservation. This is not surprising though: the research was done under the traditional paradigm of sustainable yield.

Recent research across arid Australian rangelands has shown that ephemeral species may constitute the majority of the flora at any site (Landsberg *et al.* 1997). There has been little research into the grazing ecology of individual ephemeral species in my study region (Mitchell and Wilcox 1984; Hacker 1984b). As is common elsewhere in Australia (Harrington, Wilson and Young 1984; Ludwig *et al.*), those studies that have considered ephemeral species have usually assessed them as a single amalgamated group or by plant form (e.g. forb or grass) (e.g. Holm 1994). Therefore this study represents part of a new, wider focus that includes the responses of overlooked, but potentially important aspects of rangeland flora to grazing management.

'Vital attributes' or 'plant trait syndromes' have been used in attempts to classify types of ephemeral species most likely to persist or be lost under heavy grazing (Noble and Slatyer 1980; Wilson and Harrington 1984; Landsberg, Lavorel and Stol 1999). For example, one trait syndrome that might allow a species to persist under heavy grazing would be to couple a very short life span with the ability to set abundant small seeds. Species with these traits could emerge, set seed and desiccate during periods when available forage far outstrips forage demand. Many daisies (Asteraceae) conform to this description. In contrast, a slower growing, longer-lived species, may invest heavily in a small number of large seeds. Species with these traits may outlast the shorter-lived species in the seasonal cycle (*sensu*Westoby 1980), but may also be subject to heavy grazing before setting seed. Some of the longer-lived *Sclerolaena* species and shorter-lived *Maireana* species of the chenopod family conform to this description (Mitchell and Wilcox 1984).

Variations in palatability confound the capacity for predicting responses to grazing based on easily recognised vital attributes or plant trait syndromes. No formal, easy system for visually assessing palatability has been developed. Human taste was found to quite closely resemble sheep preferences (Ian Noble, personal communication regarding a test using students at Adelaide University as 'guinea pigs'). However, Landsberg, Lavorel and Stol (1999) did not find it a useful trait for differentiating between increaser and decreaser species in south-west Queensland.

The impacts of grazing animals on physico-chemical properties of the soil further confound classification of species responses to grazing. For instance, intense grazing may be associated with nutrient enrichment, soil compaction or salinisation of the soil (Andrew and Lange 1986a). Therefore species may be removed or invade due to these edaphic changes, rather than the direct impacts of herbivory.

These concepts are critical to developing some predictive capacity about grazing impacts on whole flora of different regions. This is an active area of current area of research by others (Diaz, Acosta and Cabido 1992; Diaz and Cabido 1997; Lavorel *et al.* 1997; Landsberg, Lavorel and Stol 1999; Weiher *et al.* 1999). Although this is not the focus of my current study, I hope to return to the data to explore these ideas.

5.2.2.2 The provision of artificial sources of water

As with rangelands in the western United States (Fleischner 1994), there are few areas of arid Australian rangelands that are not within easy access to livestock as a result of the widespread provision of artificial sources of water (Landsberg and Gillieson 1996). The

location of artificial watering points gives spatial expression to the often widespread and severe modification wrought by pastoral development.

That few areas remain beyond normal access of livestock is consistent with traditional extension advice on infrastructure development given to pastoralists (Trumble and Woodroffe 1954; Burnside *et al.* 1990; Stafford Smith and Foran 1990; Cridland and Stafford Smith 1993). Pastoralists have been advised to spread watering points at regular intervals so that available resources can be used optimally, while reducing grazing pressure (and hence degradation) near watering points. This advice was aimed at reducing the extent of rangeland degradation around watering points; little attention was paid to the consequences for nature conservation if pastoralists were to graze everywhere.

However, an unintended outcome of this advice has been the elimination or fragmentation of 'water-remote areas', where species sensitive to grazing activity might persist.

Recent advice regarding distribution of watering points has incorporated principles of ecosystem management (Cohen 1992; Morton *et al.* 1995; Pringle 1995a; 1998b Landsberg *et al.* 1997). Water-remote areas are less often seen primarily as under-utilised; they are increasingly being recognised as potential benchmarks for assessing land management (Pringle 1995a) and as critical off-reserve conservation areas complementing reserves in regional conservation (Morton *et al.* 1995; Pringle 1995a; 1998b; Noy-Meir 1996; Noble *et al.* 1998; James, Landsberg and Morton 1999). To some extent, 'water remoteness' is emerging as a rangeland analogy to fragmentation in more intensely managed agro ecosystems. For rangeland species that require substantial protection from grazing, regions under widespread pastoral management may well appear highly fragmented.

Within rangeland paddocks, there are usually marked patterns of changing vegetation structure, cover and composition related to distance from water (James, Landsberg and Morton 1999). There is often a 'sacrifice zone' (Valentine 1947) in the immediate vicinity of watering points due to the extreme levels of herbivory and trampling. In inherently stable landscapes, this zone may support abundant ephemeral plants when seasons permit. Soil erosion in less inherently stable landscapes often leaves a compacted land surface with ephemeral plants restricted to patchy residuals or accumulations of variously stabilised topsoil (Charley and Cowling 1968; Pringle 1994b).

Immediately beyond the sacrifice zone there may be a range of vegetation conditions. In some cases there may be a dramatic increase in unpalatable shrubs ('woody weeds') (Burrows 1972; Noble 1998). In my study area, woody weed infestation usually occurs on

sandy 'wanderrie grass' banks and more fertile areas receiving run-on in chenopod landscapes. In these circumstances there is also usually a decline in the species richness of perennial plants (Pringle 1994b). Increases in woody plants are not usually as severe in arid mulga rangelands of Western Australia (Pringle 1994b) as in parts of eastern Australia (Noble 1998), nor as disruptive to landscape processes as occurs in North American grasslands (Archer and Smiens 1991; Whitford, Martinez-Turanzas and Martinez-Mesa 1995; Reynolds *et al.* 1999).

Heavily grazed areas beyond the sacrifice zone that are not infested with woody weeds generally have reduced cover, density and richness of perennial plants. In some cases the changes in species composition may involve replacement within similar life forms. However, it is common for open grasslands to become low shrublands (*Eremophila* spp.*)* and for low (<1m tall) chenopod shrublands to be replaced by taller shrubs (especially *Acacia victoriae* and *Hakea preissii*) (Curry *et al.* 1994; Pringle 1994b; 1998b). Thus in my study region, the heavily grazed area beyond the sacrifice zone may variously become sparser, denser or structurally modified.

As distance from water increases, vegetation gradually more closely resembles its state(s) prior to pastoral development. Graetz and Ludwig (1978) found a sigmoidal response curve best described the pattern of increase in vegetation cover with distance from water in chenopod shrublands in eastern Australia. A similar pattern has been observed near my study area (Ron Hacker, unpublished data from dams in wooded saltbush landscapes).

Changes in perennial plant cover beyond sacrifice zones are not characteristic of mulga (*Acacia aneura*) dominated landscapes of Western Australia (Pringle 1994b; 1998b). This reflects the dominance of tall shrubs and trees whose foliage is mostly out of reach of local livestock (sheep, feral goats and kangaroos). However, sheep grazing has been incriminated in the failure of mulga regeneration in parts of my study area, where it has been suggested that sheep grazing on mulga seedpods has severely reduced the soil seed bank (Moriarty 1972).

Several overseas studies have reported an elevation of species richness at intermediate levels of grazing (Archer, Garrett and Detling 1987; Andresen *et al.* 1990; West 1993). This is not the usual case in Australia (James, Landsberg and Morton 1999), although a spatially consistent pattern has been reported in South Australian chenopod shrublands (Barker and Lange 1970). In Australia there is usually either no obvious spatial pattern in species richness (Landsberg, James and Morton 1997), or an increase with increasing distance from

water until one reaches a zone of no impact. The latter appears to be the more common pattern as far as perennial plants are concerned in my study area (Pringle 1994b; 1998b) (see previous discussion).

5.3 Methods

In this section I briefly outline my field sampling and then describe the analytical pathway I developed to consider various approaches to assessing plant diversity. Specific terms such as sampling locus, site, landscape stratum and so forth were described in Chapter 2 and included in the Glossary (Appendix 1).

5.3.1 Sampling

Sampling was conducted in September/October 1997 over a period of four weeks. This was a month later than I had intended, but the delay was unavoidable due to unusually wet conditions. Late August through September is the 'wildflower' season in the study area and is the most reliable season for identifying ephemeral species. However, the good preceding seasons meant that ephemeral species were still identifiable at the end of my sampling period.

5.3.1.1 Protocols for sampling with quadrats

I adopted my sampling protocols from the biodiversity surveys conducted along grazing gradients by CSIRO Wildlife and Ecology (Landsberg, James and Morton 1997). I used a metre square quadrat for recording species presence/absence. All species rooted within the quadrat and not exceeding the height of my belt (approximately 1 metre) were included. Where a species was in a condition too poor to warrant collecting for future identification, it was assumed to be the same as the nearest similar but identifiable species within the sampling area. I also recorded species above 1m high in 10 x 10 m quadrats, but do not address these data in this thesis.

5.3.1.2 Site stratification

Arrays of 10 quadrats, two metres apart were aligned along the central axis of any sampling locus. In chenopod shrublands the grain of spatial heterogeneity is finer than the quadrats I used, so no stratification within sites was applied. Otherwise sampling was according to normal within-site landscape strata: groves and inter-groves, and sandy banks and drainage lanes (see figures at the end of Chapter 2 for site layouts). This resulted in a sample size of 100 quadrats per site in mulga landscapes and 50 quadrats per site in chenopod landscapes,

which is 20 per cent greater sampling effort per landscape stratum than that successfully employed in a similar grazing gradient study (Landsberg *et al.* 1999). I did not sample under the dense bowgada/wanyu bushes (*Acacia ramulosa/linophylla*) on sandy banks as they have never been shown to vary in prominence due to grazing and represent uncommon patches that I could not hope to sample adequately.

For multivariate analysis, I aggregated all quadrat data for each landscape stratum, giving species frequencies out of a maximum possible of 50. For all other analysis, I aggregated quadrat data at the sampling locus level for each landscape stratum, giving a species frequency out of a maximum possible of 10.

5.3.1.3 Species identification

I collected specimens of all species identified in the field and additional specimens where I was unsure of the identification. Field identifications were made using a range of plant keys (Jessop 1981; Simon 1983; Mitchell and Wilcox 1984; Wilson 1984; Cunningham *et al.* 1992; Pringle and Cranfield 1995). Ray Cranfield (Western Australian Herbarium) identified the pressed plant specimens, which are now lodged at the Western Australian Herbarium.

Databases were built by cross-referencing formal identifications and field codes. I returned some specimens for re-identification and in a small number of cases this resulted in minor changes to the databases.

5.3.2 Analysis

Analysis commenced with an investigation of simple species richness at the quadrat and sampling locus level. I then looked for variation in species composition related to landscape and grazing effects using multivariate techniques. Finally I narrowed my focus down to responses of individual species to grazing within their landscape strata and then aggregated these results to assess the responsiveness of grazing response groups in different landscapes and their strata (Figure 5.1).

5.3.2.1 Species richness

I analysed species richness by first counting the number of different species that were recorded in the 10 quadrats at each sampling locus. I also derived the average number of species recorded per quadrat for each sampling locus. This I call fine-scale species richness. These data were then analysed using the standard REML fixed and random models, but without the inter-bush mound stratum (see Chapter 2 for details of statistical modelling). This allowed for the absence of within-site stratification in chenopod landscapes.

5.3.2.2 Multi-variate species composition

I undertook exploratory multivariate analysis to check whether my landscape stratification was reflected in overall patterns of species composition. This involved aggregating individual species presence data at a site level within each landscape stratum to obtain a frequency score for each species across 50 quadrats. Analysis was conducted using PATN (Belbin 1995).

I commenced by generating a Bray and Curtis matrix of dissimilarity between 'objects', where an object comprises the data from all five representative sampling loci of a landscape stratum (e.g. mulga grove) within a site. I chose the Bray and Curtis measures as they have proven to be robust in a wide range of situations (Faith, Minchin and Belbin 1987). I used an unweighted, polythetic, agglomerative fusion, with recommended default options (Belbin, Faith and Milligan 1992) to cluster all objects into one final group, starting with all objects as individuals to investigate how they agglomerate (come together). I produced a dendrogram of these objects in order to display the process of agglomeration. Outcomes using log transformed species frequency data showed little difference.

5.3.2.3 Species response groups

The aim of this analysis was to classify species firstly according to their occurrence and secondly according to their response to grazing, and then to determine the sensitivity of the groups so formed to variation in grazing history.

Classification by occurrence

I commenced by treating exotic and native species separately. I then classified them according to whether or not they were 'singletons'. 'Singletons' were identified as species that never occurred at more than one site at any gradient, for a particular landscape stratum (e.g. mulga groves). Subsequent analysis focussed on non-singleton native species. There were too few species classified as exotics to conduct statistical modelling.

Classification by responses to grazing

I analysed each non-singleton species using generalised linear modelling in Genstat (Lawes Agricultural Trust 1997). The model terms included landscape stratum, estimated grazing history and their interaction. Grazing history was estimated using the commercially available RANGEPACK/Paddock model of average long-term Stocking Rate (SR) (Cridland and Stafford Smith 1993). I recorded those species for which frequency was significantly related to grazing (t-value probability <0.05). Few species occurred in both mulga and chenopod landscapes and the single chenopod landscape stratum was assessed first. Thus problems

in species significance being assessed against another stratum's data rather than against zero was fortuitously circumvented in almost all cases. I examined the standard error of the interaction effect to assess species where they occurred in both mulga and chenopod landscapes. Species for which frequency was significantly related to grazing were categorised as 'increaser' or 'decreaser' response types within landscape strata, on the basis of their direction of trend with increasing grazing history. Remaining species were classified as 'neutral' response type. As the classification procedure does not constitute formal statistical modelling, I have not evaluated whether 'neutral' species were not responsive to increased grazing, or whether their frequencies were insufficient for them to be classified.

Sensitivity of response groups to grazing

Finally, each set of ten quadrats within a landscape stratum was scored according to the number of 'decreaser' and 'increaser' species present. These data were then analysed using the normal REML model, with the inter-bush stratum excluded (see Chapter 2 for details on statistical modelling).

The sorting of individual species by grazing response, aggregation into numbers of species per response group at the sampling locus level, and then REML analysis was done on the full data set. However, I then repeated the process excluding the data from Gradient 3GM, where problems with site selection were reflected in anomalous results (next section).

There is superficial circularity in classifying species by grazing response, then analysing changes in response with grazing, but it is more apparent than real. Species were classified as increasers or decreasers according to how their frequency changed with grazing, not according to their presence or absence. Some increaser species were able to persist at the sites furthest from water, although they were significantly less frequent there. Conversely, some decreaser species were recorded at most grazed sites, albeit with significantly reduced frequency. Classifying the species in this way provided information on trends for individual species, and proportions of the regional flora showing similar trends.

The second stage of the analysis focussed on the consequences of these trends for regional patterns of distribution, by determining to what extent decreaser species were restricted to lightly grazed sites, or increaser species to heavily grazed sites. (It is conceivable for instance, that while frequencies decline or increase, presence does not).

I used the RANGEPACK/Paddock estimate of Stocking Rate (SR) to sort species, but used additional models to estimate grazing history for analysing aggregated responses. However,

for the process of sorting data to assign species to response groups I used only one model (Stocking Rate) so as to provide a quantitative, consistent, pre-analysis, procedure.

5.4 Results

5.4.1 A taxonomic summary

A total of 218 clearly differentiable understorey species were identified (Appendix 2), concentrated into four main families: Asteraceae (59 species), Chenopodiaceae and Poaceae (33 species each) and Myoporaceae (11 species) (Table 5.1). The most diverse genera were *Maireana* (Chenopodiaceae; 14 species), *Eremophila* (Myoporaceae; 11 species), *Sclerolaena* (Chenopodiaceae; 9 species), *Rhodanthe* (Asteraceae; 8 species), *Gnephosis* (Asteraceae; 7 species) and *Ptilotus* (Amaranthaceae; 7 species). While *Maireana* species ranged from short-lived ephemerals (e.g. *M. carnosa*) to long-lived perennials (e.g. M. *pyramidata*), most of the *Rhodanthe* and *Gnephosis* species were relatively short-lived species. This floristic pattern is typical of arid Australia (Beard 1976; Cunningham *et al.* 1992; Landsberg *et al.* 1999) and is consistent with findings from previous work in the region (Dell *et al.* 1988; 1992; Pringle 1994c; 1998c).

5.4.2 Species richness

5.4.2.1 Sampling locus level

Sandy banks had noticeably lower species richness than the other landscape strata ($p < 0.001$), with a mean of 15.7 species per sampling locus. The means of other landscape strata were in the range of 20-26 species (Figure 5.2). Estimated grazing history did not significantly affect the number of species recorded at the sampling locus level. Surprisingly, distance from water did have a significant effect on species numbers:

$$SPP = 21.56+0.9D \quad (p = 0.001)$$

Where;

SPP = number of species per sampling locus

D = distance from water (km).

In other words, there is approximately one more species per sampling locus with every successive kilometre travelled away from water.

5.4.2.2 Quadrat level

There was a clear effect of landscape stratum on species richness at this scale ($p < 0.001$, Table 5.2, Figure 5.3), with fewer species occurring in quadrats on sandy banks. This effect is consistent with results from the sampling locus level (i.e. species recorded per set of 10 quadrats).

Fine scale species richness at the quadrat level declined significantly with increasing Stocking History (SH) estimated from the track density-based model. The rate of decline (slope of the fitted line) is more pronounced in mulga landscapes, particularly mulga groves where the rate and range of decline is greatest ($p=0.02$; see figure 5.4. Grazing did not appear to affect species richness in inter-groves.

Using RANGEPACK/Paddock estimates of Stocking Rate (SR) instead of the track density-based estimate of Stocking History (SH), the main grazing effect ($p = 0.02$) and the landscape stratum effect ($p < 0.001$) were apparent, but not the interaction. Species richness per quadrat declined at a rate of 6.1 species per sheep/ha. Species recorded per quadrat also decreased at a rate of 0.4 species per kilometre with proximity to water ($p < 0.001$)

5.4.3 Floristic pattern and the underlying stratification

I then addressed the critical question of whether my hierarchical classification of landscapes was consistent with variation in species composition (i.e. whether I had adequately 'sorted the apples from the oranges'). I chose a simple, visual approach to assess the analysis, as I was mostly concerned with whether there were any major anomalies in my stratification of landscapes as reflected in species composition.

Variation in species composition among landscape stratum/site combinations ('objects' in the analysis) was best explained by paddock with site and landscape effects secondary (Figure 5.5). In groved mulga landscapes, the most similar objects were sometimes in the same landscape stratum (e.g. 21G and 22G) and at other times the same site, with pairs of grove and inter-grove sites (e.g. 34G and 34IG). In sandy mulga landscapes, the wanderrie banks and mulga drainage lanes appeared first to cluster within landscape stratum and paddock, then with objects in the same landscape stratum from another gradient rather than the other landscape stratum within the same paddock.

Agglomeration according to gradient was very strong in the chenopod landscape. Notable exceptions were the sites nearest to water (objects 11B and 51B) at two chenopod gradients (1CH and 5CH). Both sites are severely degraded and eroded.

A striking feature of the dendrogram is the apparent dissimilarity of the grazed sites at Gradient 3GM to all of the other data. The reference site for this gradient agglomerates with the other groved mulga site-landscape strata, but the paddock itself is nearly as different to the other mulga landscapes as it is to the chenopod sites. This was a forewarning of problems I encountered in the next phase of analysis. The possibility of problems was anticipated; however, from the difficulties I experienced choosing this (last) gradient (see Chapter 2).

5.4.4 Species response groups

Aggregated results from this process highlighted the fact that Gradient 2GM and 3GM were floristically quite dissimilar and it was inappropriate to analyse them as a single floristic landscape (see Box 5.1 for further explanation). On that basis I decided to drop Gradient 3GM and present findings from the remaining six gradients. The influence of this decision on the statistical appropriateness of the sorting process was minor, given the already very low degrees of freedom at this level (Ross Cunningham, personal communication). However, the decision was slightly anti-conservative (in a strict statistical sense) for the landscape type concerned (groved mulga). That is, the decision improved the likelihood of finding significant effects.

5.4.4.1 Groups by species occurrence

Exotic species

Only nine exotic species were recorded (Table 5.3) and they were predominantly short-lived forbs and grasses. None of these species is a ‘declared’ agricultural weed under the Agriculture and Related Resources Protection Act 1976 (Stevens 1994). All were minor components of the vegetation, never dominating communities to any extent. Exotic species were disproportionately more common in chenopod landscapes (all nine species), compared to groved mulga (one species) and sandy mulga (three species). There was little discernible pattern in the distribution of exotic species along grazing gradients: there were not more species close to water except at Gradient 7CH.

Native species

There were consistently more non-singleton species (species recorded at least twice along a gradient) than singletons and this was most evident in the mulga grove landscape (Table 5.4). Mulga drainage lanes tended to have slightly higher average numbers of singletons per sampling locus (2.3 singleton species per sampling locus, compared with 1.25 to 1.45 in the other landscape strata means ranged from, with an overall average standard error of the mean of 0.4 ($p = 0.01$, Figure 5.6). There was no effect of grazing on the number of singletons recorded at a sampling locus.

5.4.4.2 Responses to grazing in occurrence groups

Apart from mulga groves, numbers of decreaser species per landscape stratum outnumbered increaser species by approximately 2:1 (Table 5.5), and together these response groups represented between a third to a half of all non-singleton species. Mulga groves had approximately double the proportion of species sorted into the increaser group compared with the other mulga landscape strata and had a similar proportion of decreaser species. Chenopod landscapes had the largest number of decreaser species.

5.4.4.3 Landscape effects on sensitivity of response groups to grazing

Native decreaser species

Decreaser species were generally more numerous at sampling loci in groved mulga landscapes (groves and inter-groves) than elsewhere. Sandy banks had fewest decreaser species on average ($p < 0.001$, Figure 5.7).

Stocking History (SH) estimated from track densities had a more profound effect on landscape strata in mulga landscapes than it did in chenopod landscapes ($p = 0.002$, Figure 5.8, Table 5.6). This suggests that less accumulated grazing is required to remove species sensitive to grazing from mulga landscapes than from chenopod landscapes. This pattern is particularly strong in mulga groves, where the slope is greatest and the range of fitted values is comparable to the more heavily grazed chenopod landscapes. Sandy banks had fewer decreaser species ($p < 0.001$, landscape stratum effect).

These patterns were consistent with those found using the RANGEPACK/Paddock estimate of Stocking Rate (SR). The grazing-landscape stratum interaction was even more significant in this model ($p < 0.001$). The interaction effect was also significant using distance from water ($p = 0.02$).

Native increaser species

On average, sandy banks had fewest increaser plant species per sampling locus and groved mulga landscapes had most (p=<0.001, Figure 5.9).

The effect of Stocking History (SH) (estimated from track densities) on the number of native increaser species also showed strongly a significant interaction with landscape strata. Mulga groves were the most affected, with the steepest slope and greatest range of fitted values (Figure 5. 10, Table 5.10). Furthermore, the groved mulga landscape was more affected than either the sandy mulga or chenopod landscapes. This is reflected in the strong significance of the landscape type-grazing interaction (p=0.003), which averages the two mulga landscape sub-types.

A similar pattern was apparent with RANGEPACK/Paddock estimates of Stocking Rate (SR)s, but with a slightly stronger interaction between Stocking Rate (SR) and landscape stratum (Wald statistic of 25.3 versus 17.5; d.f. = 3). The interaction was even stronger when distance from water was used as a surrogate for grazing (Wald statistic of 29.1).

5.5 Discussion

5.5.1 Species richness

5.5.1.1 Species richness at sampling loci

The lack of any systematic variation in species richness related to modelled grazing (Distance from Water: DW was a significant effect) at the level of sampling locus is consistent with similar work by Landsberg and colleagues (James, Landsberg and Morton 1999) in arid Australia. However, they amalgamated data from individual sampling loci within sites for their analysis and were hence assessing a level higher. Traditionally patterns like this have been interpreted as evidence that grazing is a benign land use as far as plant diversity is concerned (Curry and Hacker 1990; West 1993); the numbers do not raise alarm.

5.5.1.2 Fine-scale species richness

It is not surprising that finer scale species richness - or 'species density' (as coined by McIvor 1998) - declines under successively heavier grazing, as plant mortality rates would be expected to increase under such conditions (Watson, Westoby and Holm 1997a). If fewer individuals survive it follows that the chance of a species being recorded in quadrats would decline (O'Connor 1991; Pandey and Singh 1991). However, if previously less common species increase their rates of recruitment there could be some buffering of mortality-driven decline in species density (Chaneton and Facelli 1991; Walker, Kinzig and Langridge 1999).

Contrasting results were found in the Pampas of Argentina, where point-level floristic diversity was enhanced by grazing, but stand-level diversity was reduced (Facelli 1988; Chaneton and Facelli 1991). The authors attributed the increase in fine-scale species density to grazing reducing the dominance of some grasses, and allowing invasion by other species. They attributed the decline in diversity at coarser scales to a reduction in patchiness within stands.

The pattern of declining fine-scale species density with grazing that I found might reflect lower levels of inter-specific competition than in the Argentinean Pampas and density independence typical of high stress environments (Slatyer 1975). My results are more consistent with a model of negligible suppression of diversity by dominant but palatable species, which may be removed and not replaced. It appears that relatively few species in my study region have adjusted well to current grazing regimes and grazing thus reduces the average density of many more species than it increases: at this scale there are 'few winners and many losers' (McKinney and Lockwood 1999).

There is an apparent contradiction in my results, however, in that although species richness declines with grazing at the lowest level (i.e. the quadrat, or species density scale), there is no decline in species richness apparent at the next level (the scale of sampling locus). It appears that although species may become more sparsely distributed under grazing, this has not led to a reduction in the numbers of species surviving. However, it is only by looking at species composition (taking account of the identities of species) that one can really assess whether grazing is benign or not. It is possible, for example, for some species to be replaced by others without any change in total numbers. Even if there has been no loss of species, a decline in the density of species could affect habitat quality for other biota (Morton, James and Landsberg 1996).

These considerations all suggest that the level of analysis needs to go deeper than simple arithmetical measures in order to assess the full impacts of grazing on the persistence of native plants.

5.5.2 Multivariate species composition

The floristic classification of landscape strata confirmed that my imposed landscape stratification captured real floristic differences and therefore represents a useful primary stratification for this work. However, it also confirmed my impression that Gradient 3GM is floristically quite different. Thus I did not use these data for the regional (aggregated) analysis. The other data showed quite neat agglomerations reflecting combinations of

paddock and landscape influences. That paddocks seemed to have such a major influence on species composition is hardly surprising, since some of the paddocks are more than 100km apart.

The degree of floristic difference between objects was consistent with the degree of edaphic contrast between them. For instance, adjacent sandy banks and mulga drainage lanes tended not to cluster floristically, mulga drainage lanes tended to cluster first with the groved mulga landscape places with similar, clay loam soils. Chenopod places were floristically distinct from the mulga places and have very different soils (saline texture contrasting soil profiles) (Van Vreeswyk 1994).

That grazing impacts were partly obscured by more profound influences was not surprising considering the contrasting physical environments sampled and the degree of their geographic separation. In a similar, hierarchically structured project in Spanish arid and semi-arid shrublands, Bergkamp (1995) produced a matching result. He used multivariate cluster analysis of floristic diversity to confirm that his 'holons' (components of the wider system under study) were distinctive, but found floristic diversity to be a poor indicator of position along a degradation gradient.

Overall, this phase of analysis indicated that I had a workable ecological classification within which to examine grazing impacts and their interactions with parts and types of landscapes.

5.5.3 Exotic species

Exotic species are generally rare and do not usually dominate arid rangeland plant communities in Australia (Humphries *et al.* 1992; Landsberg *et al.* 1999). My study region was no exception. However, the occurrence of exotic species at many of my sites (and not just close to watering points) suggests that they may be widespread and a potential future threat should disturbances become more widespread, frequent or severe. For instance, this might occur with predicted climate change (Allen-Diaz *et al.* 1996; Kirschbaum *et al.* 1996; Howden *et al.* 1999). Currently however, the threat posed by exotic plants seems less serious in my study region than has been reported elsewhere. This may be in part because the exotics recorded in my study area are generally small and short-lived, rather than the longer-lived, aggressive invaders that have replaced native communities elsewhere (Cox and Ruyle 1986; Payne *et al.* 1987; Anable *et al.* 1992; Young and Allen 1997; Whitford *et al.* 1999).

5.5.4 Species response groups

5.5.4.1 Landscape effects

There is little overlap between chenopod and mulga landscapes in the identities of the species in either increaser or decreaser groups, although there is some overlap between similar soil types (e.g. mulga drainage lanes and mulga groves and inter-groves). This differentiation highlights the need to use appropriate landscape reference sites against which to assess 'biodiversity condition'.

5.5.4.2 Landscape and grazing interactions

Decreaser species are clearly more prominent than increaser species in most landscape strata. This is consistent with a flora that has not adjusted well to current grazing regimes: more species appear to be sensitive to grazing than are advantaged by it. That this is the case and yet there is no significant effect of modelled grazing activity (there was for Distance from Water: DW) on total species richness at the level of this analysis (section 5.5.1) may reflect intermediate responses of some species and the statistical 'noise' of within-landscape heterogeneity. It might also reflect a relatively early stage of decline where many species are showing declines in abundance but few have been totally lost.

These results illustrate the limitations of the 'arithmetic measures' approach of assessing biodiversity. Rather they are consistent with a model of biotic homogenisation: 'few winners and many losers' (McKinney and Lockwood 1999) and a flora with an evolutionary history in marked contrast with current land use (Milchunas, Sala and Lauenroth 1988; Milchunas *et al.* 1989). While species numbers at the stand level are not currently affected by grazing, there is an underlying (but previously undetected) potential threat to floristic diversity.

Increaser and decreaser species respond more strongly to grazing in mulga landscapes than they do in chenopod landscapes. This was most apparent in mulga groves. However, this conclusion should be treated with caution, as there was only one gradient representing this landscape.

While decreaser species in chenopod landscapes appear to be more tolerant of grazing (i.e. it requires more grazing on average to remove them), the flora in these landscapes is threatened to a similar extent to that of mulga landscapes. This is because chenopod landscapes have been far more heavily grazed; pastoralists tend to stock paddocks with these landscapes far more heavily and sheep preferentially graze in them in paddocks with mixes of landscape types (Pringle *et al.* 1992; North Eastern Goldfields and Kalgoorlie Land

Conservation Districts 1993; Pringle 1994a; Pringle and Riches 1996). Far from being least affected; it may be that chenopod landscapes require equal protection from grazing. This is not because of any greater sensitivity of their flora but because their greater pastoral productivity has resulted in greater intensity of use.

5.5.4.3 Paddock patterns and their regional implications

The combination of changes in species composition due to grazing and the considerably greater number of native species that are sensitive to grazing rather than favoured by it suggest that grazing is not a benign land use from the perspective of nature conservation. Furthermore, regional impacts may be more severe than in my study, because the configurations of my study paddocks were regionally abnormal. Most were large and included areas further from watering points than is usual in the region.

Indications of detrimental regional impacts of grazing on nature conservation are as follows. Firstly, there is a decline in abundance of a significant proportion of the flora away from watering points in my study paddocks, which were very large. Secondly, in large paddocks there is likely to be a more obvious decline in relative grazing activity than in smaller paddocks where area available for grazing is substantially more bounded and therefore more uniformly distributed (Knight 1995). Integrating these two factors in the context of the smaller paddocks that are more usual in the region leads to the strong possibility that the area of lightly grazed landscapes where some decreaser species can persist is very small.

Thus in contrast to a more traditional view that the 'numbers look fine', there is a reasonable probability that a significant part of the flora may have been extirpated or be surviving as isolated fragments in 'normal' pastoral paddocks in this region.

These indications warrant further investigation. Questions of particular concern include: which species are involved, are their responses in this study consistent with existing knowledge (e.g. Mitchell and Wilcox 1984; Pringle 1994b) and do they share functional or morphological attributes that can help predict responses to grazing in other species?

There is also an urgent need for a register of 'water remote' sites in this region. These sites need to be surveyed and described in terms of landscapes, so as to provide benchmarks against which to assess land use impacts and to systematically identify threatened aspects of biodiversity. This is planned for the region under the Gascoyne Murchison Strategy. Similar initiatives are already under way in the Northern Territory and South Australia (Steve Morton, personal communication).

5.5.4.4 Implications beyond my study region

The study region has had a relatively recent and light pastoral history compared to the river systems flowing to the west coast. Pastoralism in the study region only became extensive in the 1920s and 30s, when it became an export industry, rather than a supply of produce to local mining operations. Therefore the region was spared the major degradation associated with the earlier shepherding of huge flocks that has damaged vast areas of prime pastoral landscape in the Murchison and Gascoyne River catchments (Wilcox and McKinnon 1972; Curry *et al.* 1994). Similar histories of early extensive and severe degradation have occurred in eastern Australia (Tothill and Gillies 1992; Abel and Ryan 1996; State Of Environment Advisory Council 1997).

Much of the area was not stocked until polypipe reticulation became available in the 1970s because suitably fresh groundwater for stock was difficult to find in the chenopod landscapes (Pringle 1994a). Thus this region has been more recently, lightly and less extensively grazed than many other parts of arid Australia.

Fragmentation of water-remote areas sufficient for the persistence of some species may be far more serious elsewhere. Therefore, it seems sensible to adopt a systematic, hierarchical approach to addressing off-reserve conservation across several regions or biomes (Noss 1983; Noss and Harris 1986; Morton *et al.* 1995; Pringle 1998a). This might involve starting with worst affected regions, presuming that all is not lost already. Within regions one might focus on the most extensively degraded landscapes, in the hope that native species can persist a bit longer in landscapes that are more intact. Concurrently, special projects can be implemented for particular species or habitats known to be threatened (Morton *et al.* 1995; Pringle 1995a).

Whatever approach is eventually adopted for improving nature conservation in the rangelands, an important first step is quantifying the extent of change. Although I found no evidence of species loss in my study region my results indicate that changes in species composition and a continuing decline in some species are likely to be extremely widespread and pervasive.

5.6 Summary

(1) All vascular plant species were recorded in each of series of quadrats nested within sampling loci within various landscape strata at sites in mulga and chenopod landscapes. In most cases the data were then aggregated at the sampling locus level to give frequencies of occurrence for each species.

(2) Grazing history had a significant negative effect on fine scale species richness (i.e. the number of species recorded per quadrat), particularly in mulga landscapes. Mulga groves appeared most affected. However, grazing appeared to have no significant effects on species richness at the level of sampling locus, except using the Distance from Water (DW) model.

(3) Floristic composition closely resembled the ecological classification used for stratifying sites in this study, with the exception of the three grazed sites on the mulga gradient 3GM.

(4) A substantial proportion of the flora appeared to be significantly advantaged or disadvantaged by grazing, with disadvantaged species (i.e. decreasers) clearly outnumbering advantaged species (increasers) in all but one landscape stratum. This suggests that a substantial part of the flora has not adjusted well to current grazing regimes.

(5) Decreaser species are considerably more sensitive to (modelled) grazing history in mulga landscapes than in chenopod landscapes, particularly in mulga groves. However, because chenopod paddocks are grazed more heavily, the number of disadvantaged species may be similar in both landscapes.

(6) Increaser species are most prominent in groved mulga landscapes; they appear to be a minor component of sandy mulga and chenopod landscapes.

(7) The sequence of analyses highlights the importance of assessing the responses of individual species, as opposed to relying on a simple metric such as richness to assess land use impacts on biodiversity.

(8) The influence of landscape heterogeneity on species composition, both in its own right and as an interactive influence with grazing, was considerable. This highlights the importance of having a conceptual model of underlying landscape pattern and an understanding of the grazing ecology of species as precursors to assessing ‘biodiversity condition’.

(9) The sites I chose to represent extremely light grazing are not common in the study region. Thus pastoral management may have had a more severe impact on the region as a whole than was apparent in my study. Additionally, this region has had a relatively recent and light grazing history compared to many similar arid regions in other parts of Australia, where the situation may be considerably worse.

Table 5.1 A taxonomic summary

Family	Number of species	Prominent genera
Asteraceae	59	*Rhodanthe* (8)1, *Gnephosis* (7), *Brachyscome* (4), *Angianthus* (4)
Poaceae	33	*Eragrostis* (5), *Austroatipa* (5)
Chenopodiaceae	33	*Maireana* (14), *Sclerolaena* (9), *Atriplex* (4)
Myoporaceae	11	*Eremophila* (11)
Amaranthaceae	7	*Ptilotus* (7)
Malvaceae	7	*Sida* (4)
Portulacaceae	7	*Calandrinia* (6)

[1]Figures in parentheses are numbers of species.

Table 5.2 Effects of grazing history on fine-scale species richness

(i.e. average number of species per quadrat at each sampling locus.) See Glossary (Appendix 1) for terminology and Chapter 2 for explanation of modelling terms.

Grazing estimate	Term	Wald statistic	d.f.	p-value
SH: Stocking History, derived from modelling track densities and expressed in sheep years per hectare	landscape type	0.9	1	n.s.
	SH	5.8	1	0.02
	stratum.landscape type	110.5	3	<0.001
	landscape type.SH	2.2	1	n.s.
	stratum.landscape type.SH	9.4	3	.02
SR: Stocking Rate; average long term stocking rate developed from RANGEPACK/Paddock model and expressed in sheep per hectare	landscape type	0.7	1	n.s.
	SR	5.5	1	0.02
	stratum.landscape type	109.0	3	<0.001
	landscape type.SR	3.3	1	n.s.
	stratum.landscape type.SR	6.6	3	n.s.
DW: Distance from Water, measured in kilometres	landscape type	0.9	1	n.s.
	DW	15.6	1	<0.001
	stratum.landscape type	105.5	3	<0.001
	landscape type.DW	2.2	1	n.s.
	stratum.landscape type.DW	0.7	3	n.s.

Table 5.3 Occurrence of exotic species

Species	Family	Life form	Chenopod	Groved mulga		Sandy mulga	
			flat	mulga grove	inter-grove	mulga drainage	sandy bank
Anagallis arvensis		forb	2/3, 3/15				
Cuscuta epithymum	Cuscutaceae	fine ectoparasite	3/3, 8/15	1/1, 2/4	1/1, 2/4	2/2, 4/8	1/2, 1/8
Emex australis	Polygonaceae	forb	1/3, 1/15				
Hordeum leporinum	Poaceae	short-lived grass	1/3, 1/15				
Pentascistis airoides	Poaceae	short-lived grass	2/3, 8/15			2/2, 3/8	2/2, 4/8
Rostraria cristata	Poaceae	short-lived grass	3/3, 11/15			1/2, 2/8	
Sonchus oleraceus	Asteraceae	forb	2/3, 5/15				
Sonchus tenerrimus	Asteraceae	forb	1/3, 1/15				
Vulpia myuros	Poaceae	short-lived grass	1/3, 1/15				

Pairs of fractions are firstly numbers of gradients occupied and total numbers of gradients, and secondly relative numbers of sites occupied within a landscape.

Table 5.4 Occurrence of all species by landscape stratum

Landscape	Landscape stratum	Number of singletons	Number of non-singletons	Total
Chenopod	flat	55	85	140
Groved mulga	mulga grove	18	63	81
	inter-grove	16	46	62
	both1	27	70	97
Sandy mulga	mulga drainage	46	60	106
	sandy bank	31	43	74
	both	70	72	144

[1,]'both' refers to total number of different species in each category for that landscape, regardless of landscape stratification.

Table 5.5 Number of species per response group

Landscape	Landscape stratum	Increaser species	Decreaser species	Neutral species	Total
Chenopod	flat	11 (13%)	24 (28%)	50 (59%)	85 (100%)
Groved mulga	mulga grove	15 (24%)	17 (27%)	31 (49%)	63 (100%)
	inter-grove	8 (17%)	14 (30%)	24 (52%)	46 (100%)
	Both1	20 (25%)	20 (25%)	41 (50%)	70 (100%)
Sandy mulga	mulga drainage	7 (12%)	13 (22%)	40 (67%)	60 (100%)
	sandy bank	5 (12%)	9 (21%)	29 (67%)	43 (100%)
	Both	10 (11%)	20 (23%)	55 (67%)	72 (100%)
	Stratum averages2	9.2 +/ 1.7 (15.6% +/- 2.3)	15.4 +/-2.5 (25.6% +/- 1.7)		

[1] 'Both' refers to the number of different species in that response group, regardless of landscape stratification. The 'Total' column for 'Both' accommodates the non-mutual exclusivity of component groups between landscape strata within mulga landscapes (hence its values are lower than might be expected).

[2] Stratum averages are presented as means and standard errors of the mean across the five landscape strata, for number of species (top) and percentages of flora (below).

Table 5.6 Effects of grazing history on native decreaser species richness

Grazing estimate	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	0.0	1	n.s.
	SH	33.5	1	< 0.001
	stratum.landscape type	33.2	3	< 0.001
	landscape type.SH	10.3	1	0.001
	stratum.landscape type.SH	14.4	3	0.002
SR: Stocking rate (sheep/ha)	landscape type	0.0	1	n.s.
	SR	67.1	1	< 0.001
	stratum.landscape type	41.6	3	< 0.001
	landscape type.SR	11.8	1	< 0.001
	stratum.landscape type.SR	17.8	3	< 0.001
DW: Distance from water (km)	landscape type	0.0	1	n.s.
	DW	41.6	1	< 0.001
	stratum.landscape type	31.4	3	< 0.001
	landscape type.DW	0.5	1	n.s.
	stratum.landscape type.DW	10.0	3	0.02

Table 5.7 Effects of grazing history on native increaser species richness

Grazing estimate	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	0.4	1	n.s.
	SH	10.5	1	0.001
	stratum.landscape type	22.8	3	< 0.001
	landscape type.SH	8.4	1	0.004
	stratum.landscape type.SH	17.5	3	< 0.001
SR: Stocking rate (sheep/ha)	landscape type	0.8	1	n.s.
	SR	47.0	1	< 0.001
	stratum.landscape type	27.0	3	< 0.001
	landscape type.SR	20.1	1	< 0.001
	stratum.landscape type.SR	25.3	3	< 0.001
DW: Distance from water (km)	landscape type	0.3	1	n.s.
	DW	62.0	1	< 0.001
	stratum.landscape type	21.0	3	< 0.001
	landscape type.DW	3.9	1	0.048(n.s.)
	stratum.landscape type.DW	29.1	3	< 0.001

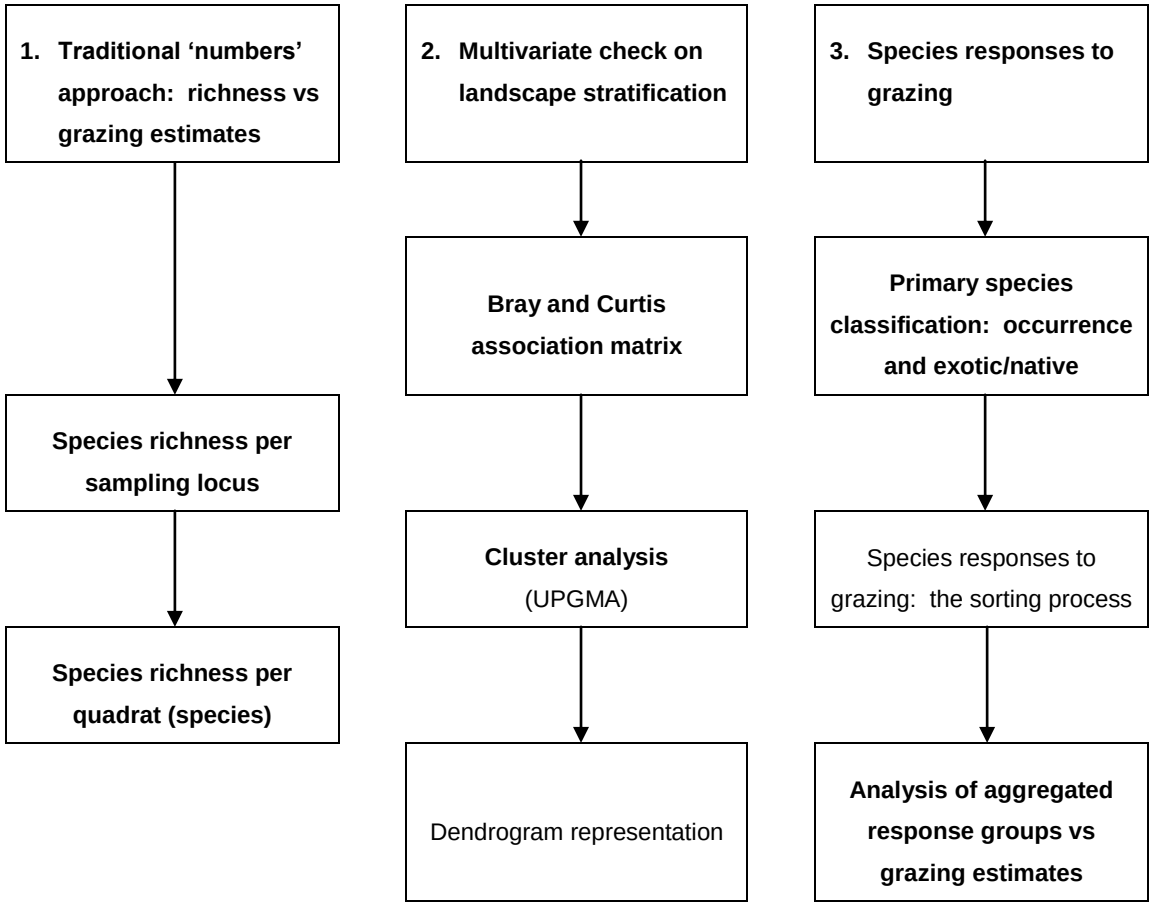


Figure 5.1 Analytical framework.

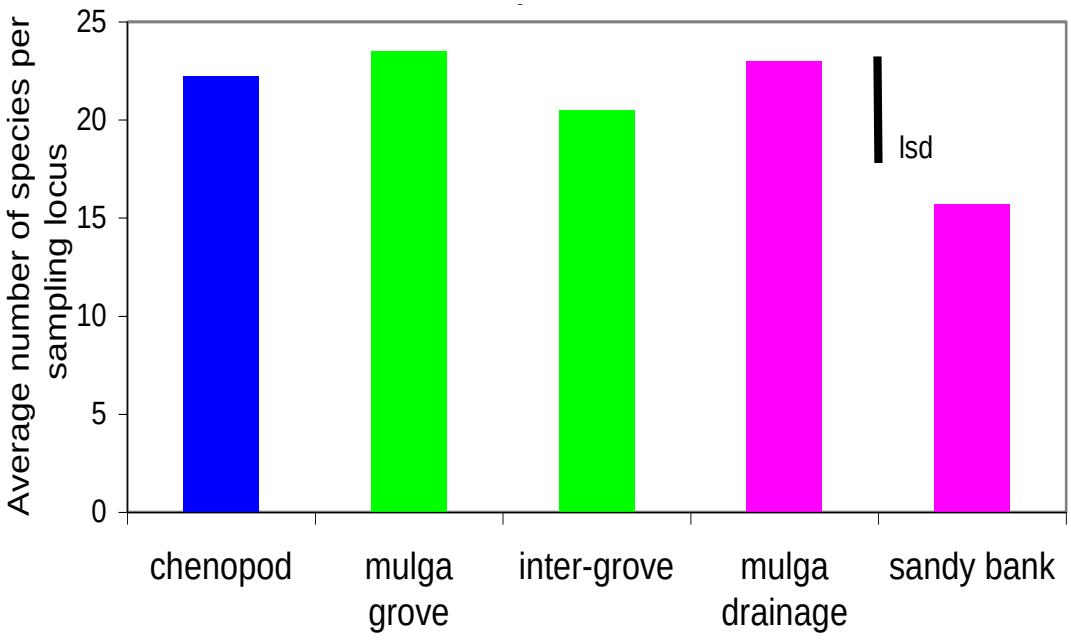


Figure 5.2 Variation in plant species richness with landscape stratum.

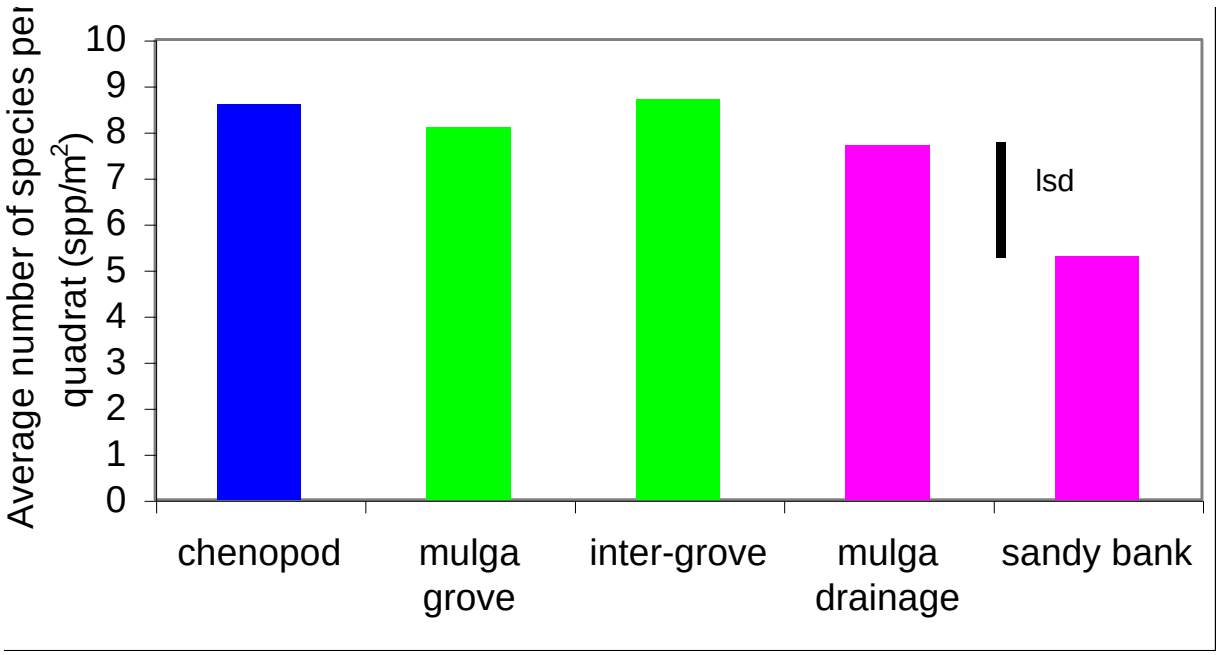


Figure 5.3 **Variation in fine-scale plant species richness with landscape stratum.**

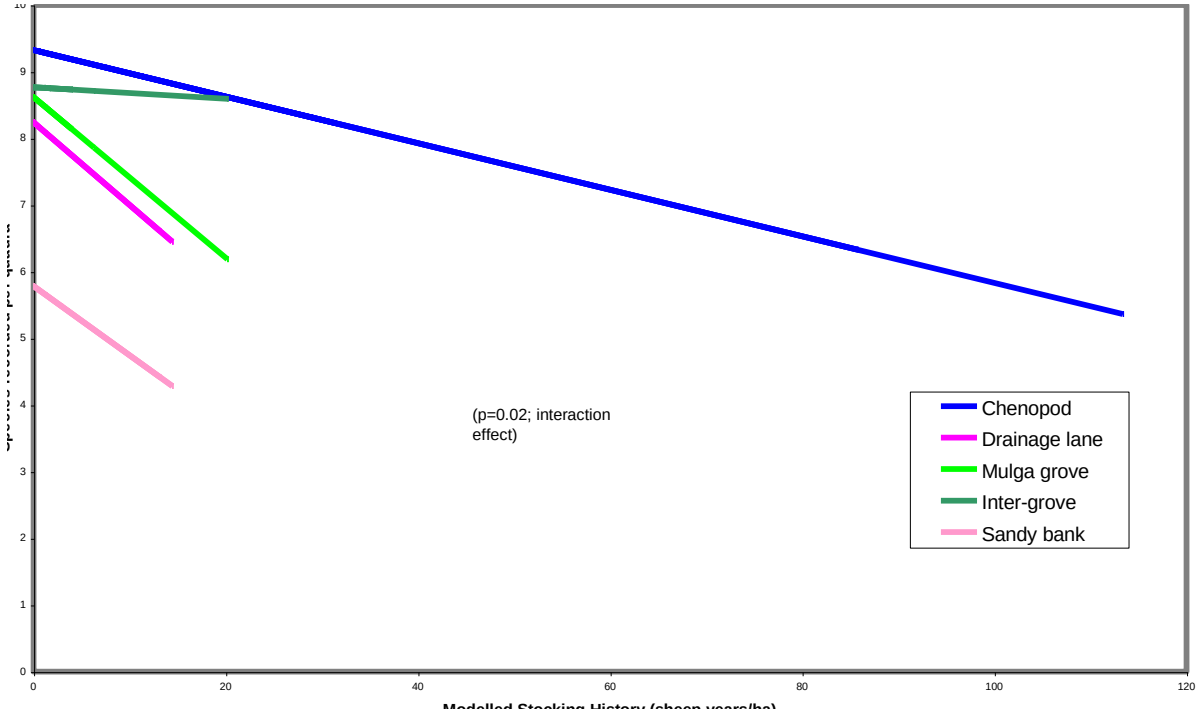
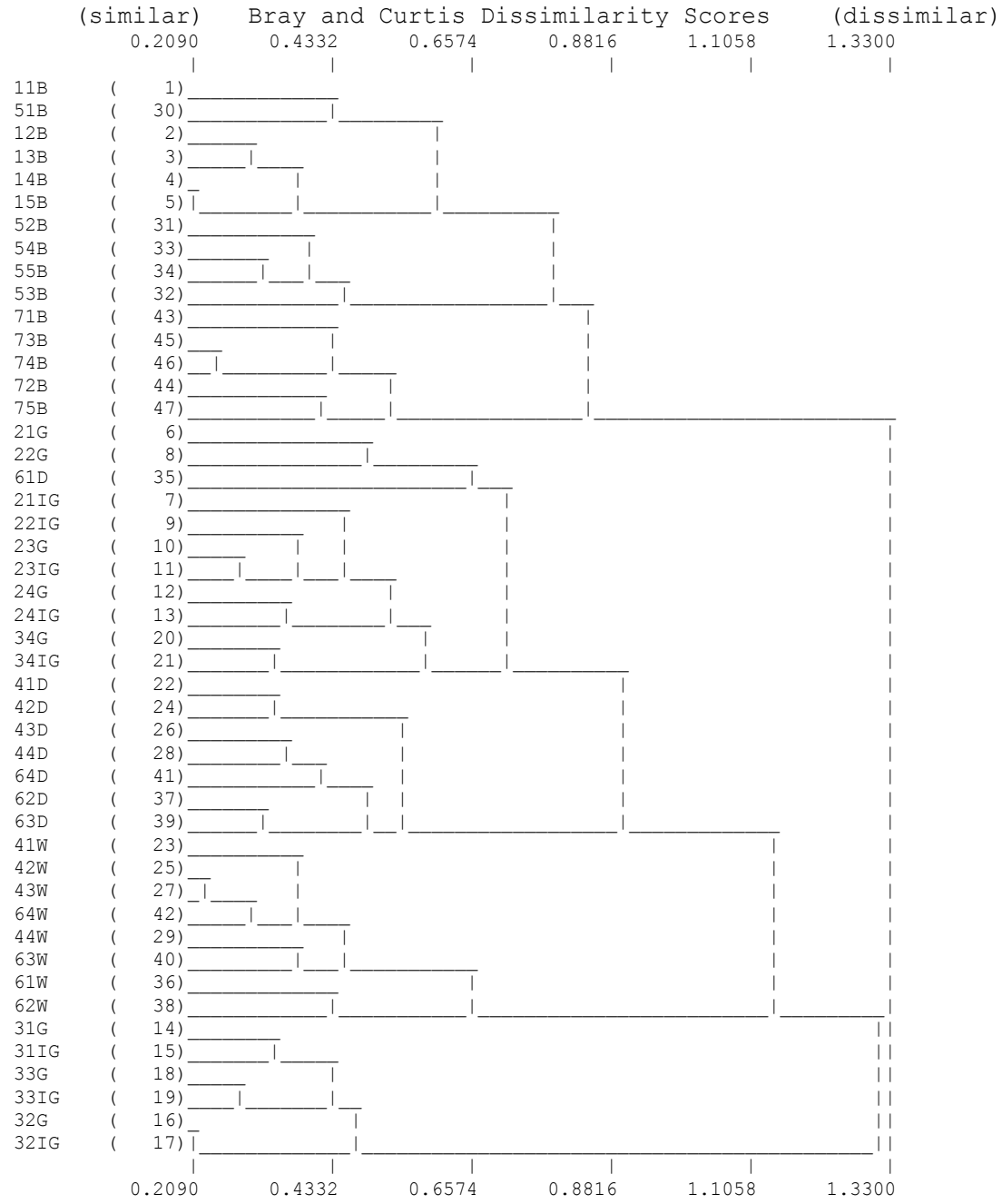


Figure 5.4 **Impacts of stocking history on fine-scale plant species richness.**



Key to object codes
"21IG" = Gradient 2, site 1, intergrove
(B=chenopod, D=mulga drainage, G=mulga grove, IG= inter-grove and W=Sandy bank)

Note, these codes are not to be confused with gradient codes in the main text, e.g. 1CH = Gradient 1, which is in a chenopod landscape.

Figure 5.5 Dendrogram of objects by understorey species composition.

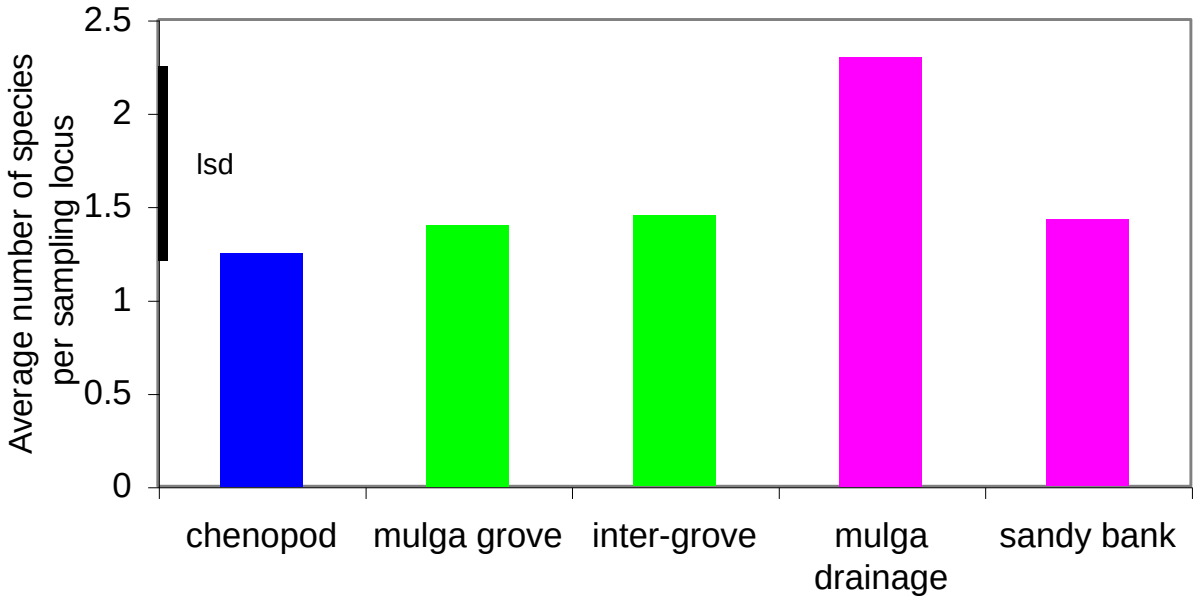


Figure 5.6 **Variation in number of singleton plant species with landscape stratum.**

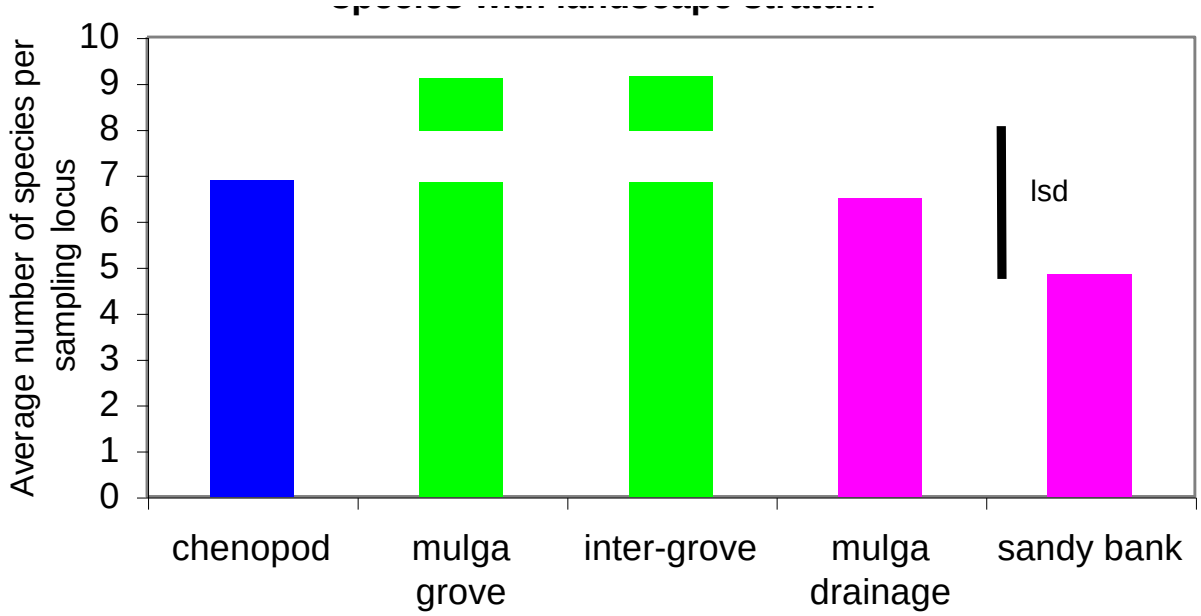


Figure 5.7 **Variation in number of decreaser plant species with landscape stratum.**

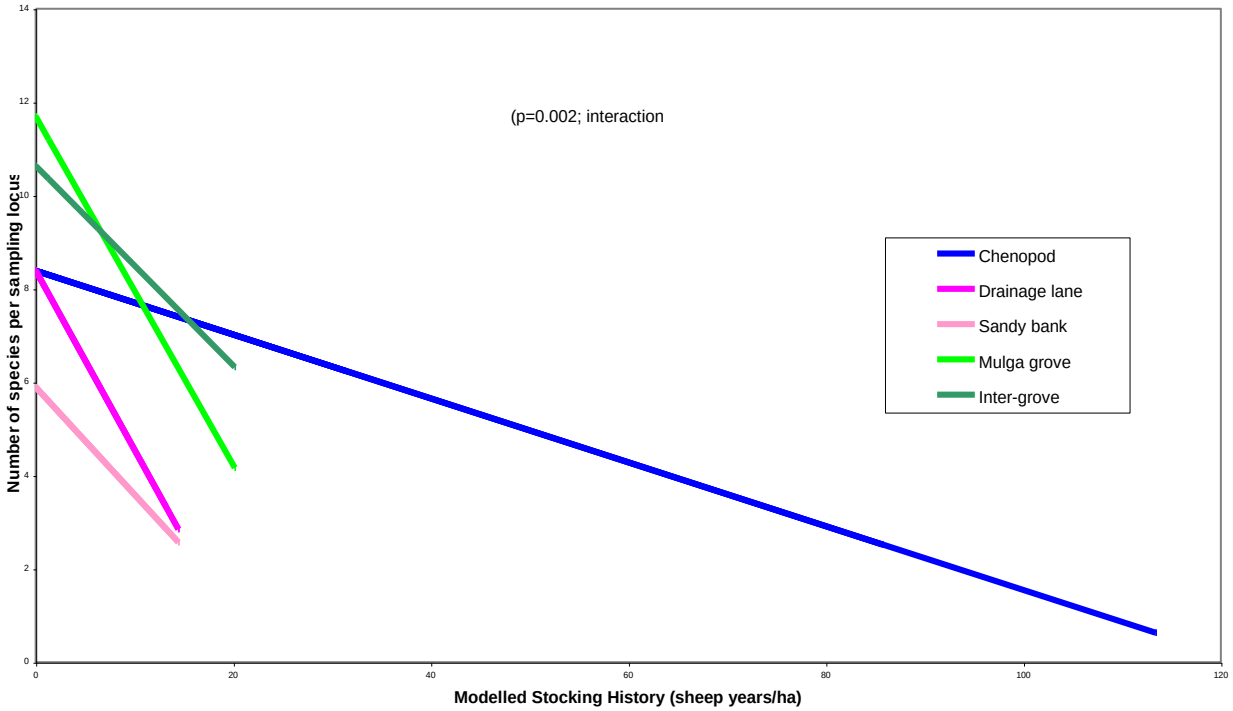


Figure 5.8 Impacts of stocking history on native decreaser plant species richness.

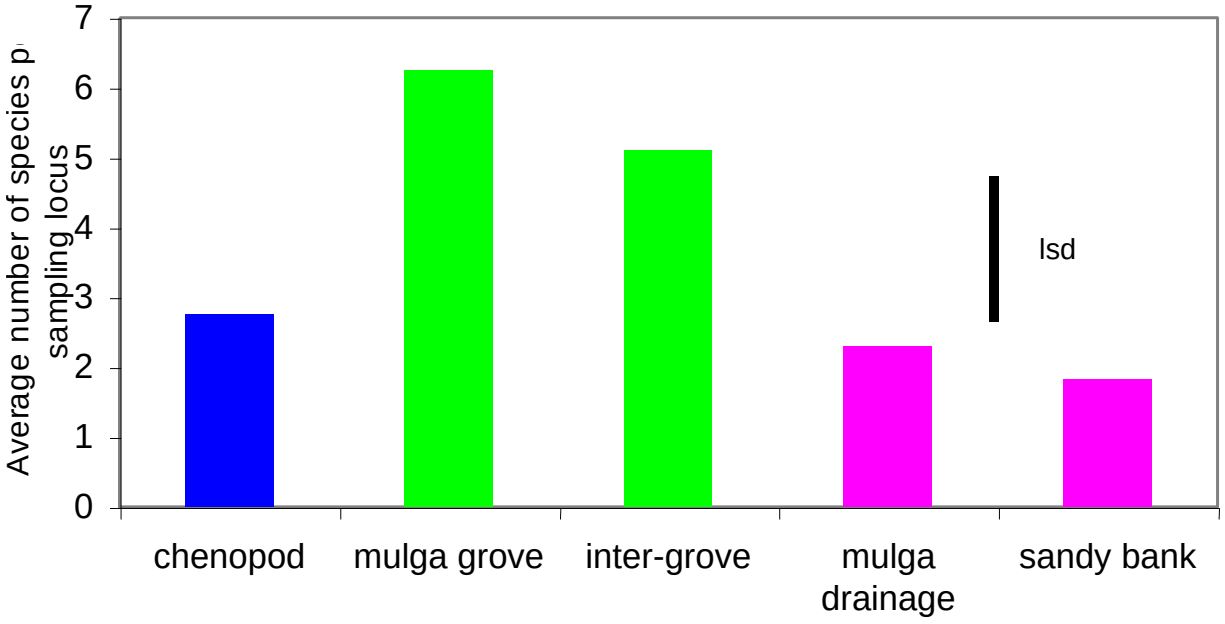


Figure 5.9 Variation in number of increaser plant species richness with landscape stratum.

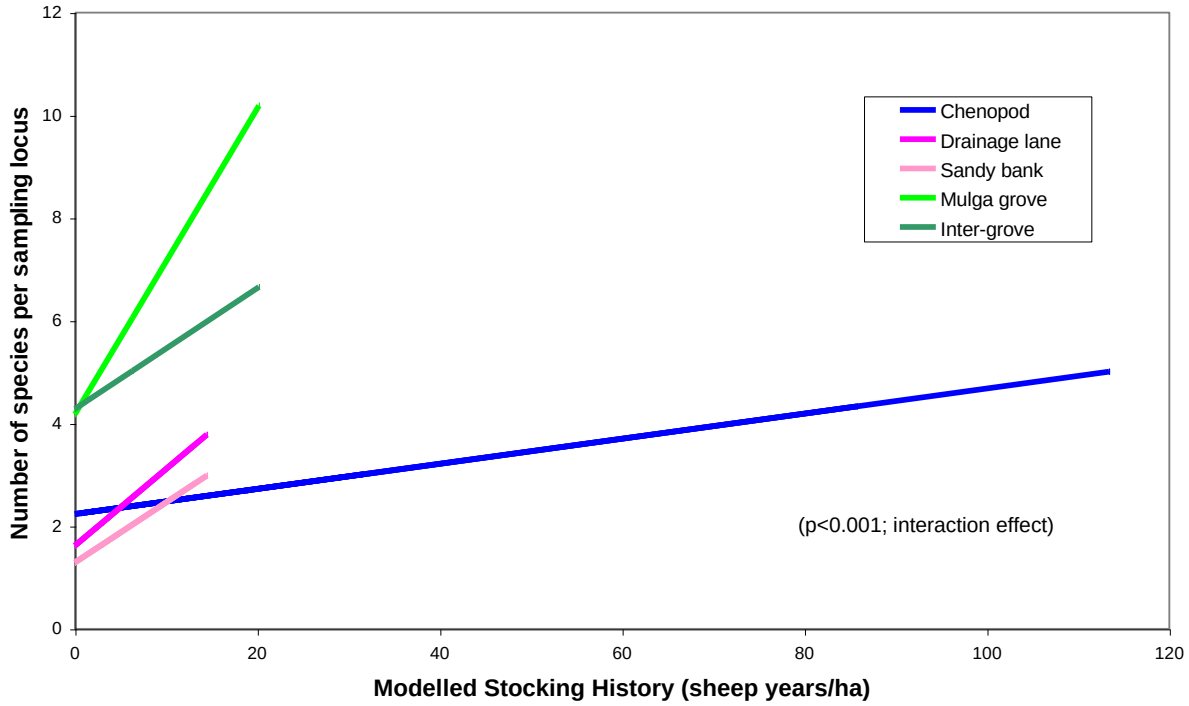


Figure 5.10 Impacts of grazing on richness of native increaser plant species richness.

Box 5.1 Variability within Groved Mulga landscape

The issue:

Sites within the grazed paddock on Gradient 3GM appeared to be floristically dissimilar to all other sites, irrespective of type of landscape. This is illustrated by the dendrogram of sites based on Bray Curtis measures of dissimilarity (Figure 5.5). The reference site associated with this gradient agglomerates with the sites from Gradient 2GM, the other Groved Mulga gradient.

It was therefore not surprising that the systematic sorting of species into indicator groups produced strange results using aggregated data for the two gradients. For example, increaser species decreased with increasing grazing at gradient three, but followed a more predictable, increasing pattern at gradient two. Furthermore, there were fewer decreaser species at the reference site than at the three grazed sites for gradient three.

This outcome is consistent with looking for consistencies between gradients in response to grazing when dealing with two fundamentally floristically different sets of sites. So, how should I respond to this contrasting floristic difference?

Possible responses

Response one: I accept that the failure to identify consistent responses to grazing means that this landscape is little affected by it. This seems flawed on the basis of above discussion and application of the precautionary principle to decisions reflecting sampling design problems rather than the biology of the system. I reject this scenario.

Response two: I accept that I sampled two quite different gradients within the Groved Mulga landscape type for this region, and that both are equally legitimate representatives of the type. If so my problem is that I have sampled this highly variable landscape inadequately (N=2 is not enough). This would result in me disregarding further pursuit of my underlying biological questions in this type of landscape. I prefer to determine whether one gradient can be rejected rather than both. This would be valid if the rejected gradient is unrepresentative of its type. The regional context, rather than the variation within the data is critical in determining representativeness in this context.

Response three: Accept that Gradient 2GM is more similar to other landscapes because it represents a transitional state: it is an outlier of Groved Mulga and Gradient 3GM is a better (more representative) gradient of this landscape type. However, the fact that the reference site at Gradient 3GM clusters with Gradient 2GM, rather than with the rest of Gradient 3GM suggests that Gradient 3GM is the outlier. Furthermore, several of the plant species recorded at Gradient 3GM was unfamiliar to me, despite my decade of botanical survey experience in this region. Some of the species with which I was not familiar and which only occurred at gradient three are: E*remophila* species D22, *E. hydrophana, Euphorbia* species D11, *Podolepis* speciesM8, and unknown taxon D34. Their regional rarity and occurrence at Gradient 3GM further suggests that it is atypical of its landscape type. In addition, one would expect strong floristic similarity between mulga drainage lanes and groved mulga communities on the basis of the similarity between their landscape processes, soils and position in the landscape (Pringle 1994b; Pringle 1998b). They should cluster together, as the reference site for gradient three and the sites for gradient two have done. Thus the floristic evidence suggests that it is more likely that the grazed sites at Gradient 3GM are unrepresentative of Groved Mulga landscapes.

Response four: Accept that the grazed sites at gradient three are outliers of Groved Mulga landscape type and that Gradient 2GM is a representative set of sites for this landscape. Without actually collecting a large amount of additional floristic data, a rigorously quantitative assessment is not possible. However, on the basis of preceding discussions, I propose that this appears to be the case.

CHAPTER 6. ANT DIVERSITY

6.1 Introduction

6.1.1 Scope of this chapter

This is the second chapter on aspects of biodiversity. In the previous chapter I addressed many of the general issues regarding biodiversity. In this chapter I focus on ant diversity and its relationship to variations in landscape and grazing history (see Chapters 2 and 3).

My primary interest in ant diversity is as one indication of faunal diversity from a perspective of nature conservation. Specifically, I am interested in how well ant species in my study area have adjusted to pastoral development, as represented by their species assemblages at different levels of estimated grazing history. I am also interested in whether assemblages of species collectively exhibit different sensitivity to grazing according to types of landscapes (e.g. groved mulga landscapes) and their landscape strata (e.g. groves and inter-groves).

I commence with a brief review of previous work on ant diversity in arid rangelands, with particular reference to current understanding of its relationship to grazing. I then describe how I sampled ant diversity, identified samples and analysed the data. I conclude with a discussion of the results of the analyses in the context of contemporary literature and emerging principles in 'ecosystem management' of rangelands (Abel and Ryan 1996; Pringle 1998a).

6.1.2 Key biological questions

The issues I address in this chapter can be translated into the following key biological questions:

(1) Does grazing affect the number of ant species sampled at a site (species 'richness')?

(2) Does grazing affect the composition of ant assemblages?

(3) Is there consistency of response to grazing within genera? (Is genus taxonomically sufficient?)

(4) What proportions of ant species appear to have adjusted well ('inceasers') or poorly ('decreasers') to pastoral grazing management?

(5) Do the proportions of increaser and decreaser species differ among parts and types of landscapes?

(6) Does grazing affect these response groups to differing degrees in different types or parts of landscapes?

(7) What are the implications of paddock scale patterns for regional nature conservation?

The third question is not directly related to my primary research objectives in this study. However, I have included it as it may contribute to developing much needed invertebrate indicators of grazing impacts in rangelands (Whitford *et al.* 1999).

6.2 Ant diversity in arid rangelands

6.2.1 Why select ants?

Ants play critical roles in arid Australian ecosystems, including biopedturbation and seed dispersal (Greenslade 1978; Andersen 1990; 1995). They have also been shown to exhibit considerable functional and trophic diversity that is broadly consistent within genera. For instance, *Iridomyrmex* species are generalist opportunist species that are often found in disturbed landscapes (Greenslade and Greenslade 1984; Andersen 1990; Abensperg-Traun *et al.* 1996). Furthermore, ants have proven sensitive to grazing in an extensive research project across Australia's arid zone (Landsberg, James and Morton 1997; Landsberg *et al.* 1997).

While ants are sampled reasonably efficiently using pitfall traps, their subsequent sorting and identification to species or morphospecies can be time consuming and expensive (New 1996; Landsberg, Morton and James 1999). Fortunately, I had funds to pay parataxonomists to undertake this sorting, courtesy of WMC Resources. Ant taxonomy is also poorly developed compared to vertebrates and vascular plants (Andersen 1990), but Alan Andersen (CSIRO Wildlife and Ecology, Darwin) kindly offered to provide expert identification of my type specimens representing morphospecies.

6.2.2 Factors affecting ant diversity

The strongest influence on the distribution of ant diversity within any region appears to be soil type. Other factors such as soil surface cover (litter), level of exposure to solar radiation and invasion by exotic plant species have also been reported as influential (Greenslade and Greenslade 1984; Holldobler and Wilson 1990; Morton, James and Landsberg 1996; Whitford *et al.* 1999).

I hypothesised that impacts of grazing, if they occur, are therefore likely to be nested within different soil-landscapes. This seems to be a logical and efficient underlying framework upon which to conduct a study of grazing impact.

6.2.3 Ant diversity and grazing

Comparatively little work has been done on the impacts of grazing on invertebrates in general, including ants (Morton, James and Landsberg 1996; James, Landsberg and Morton 1999). What evidence exists is contradictory.

In some cases grazing appears to have negligible effects on ant diversity (Heske and Campbell 1991; Abensperg-Traun *et al.* 1996; Whitford *et al.* 1999). Indeed, Whitford *et al.* (1999) state that 'ants cannot be used as indicators of exposure to stress….on rangeland ecosystems', on the basis of their research of responses to grazing in the south western United States.

Conversely, Landsberg and colleagues (Landsberg, James and Morton 1997; Landsberg *et al.* 1997; Landsberg *et al.* 1999) found that approximately half of the ant species in their study showed a response to different levels of grazing (distance from water) across arid southern and central Australia and they advocate ants as potentially useful arthropod indicators of grazing impacts in Australian rangelands. However, they suggest that understorey plants and birds may be more efficient indicators of grazing impact (Landsberg, Morton and James 1999). Others have also reported impacts of grazing on ant diversity (Bestelmyer and Wiens 1996; Cristo and Wiens 1996; Majer and Beeston 1996; Bromham *et al.* 1999) or dramatic responses in ant communities on removal of grazing (Putman *et al.* 1989).

There is also evidence that grazing can result in shifts in the composition of ant assemblages in terms of their functional groups (Bestelmyer and Wiens 1996), though Landsberg *et al.* found little consistent evidence of this in their data (Landsberg *et al.* 1999).

The absence of any universally applicable generalisation about impacts of grazing on ant diversity reflects (at least) two underlying factors. Firstly, the effects of grazing on ants are almost entirely indirect. The impacts are felt through changes to habitat (e.g. vegetation structure and food supply) and competitive interactions that may result (Bestelmyer and Wiens 1996; Morton, James and Landsberg 1996; James, Landsberg and Morton 1999).

Simple models such as 'increasers' and 'decreasers' (Dyksterhuis 1949; 1958) that have been applied successfully to rangeland plants (Mitchell and Wilcox 1984; McIntyre and

Lavorel 1994; Pringle 1994b; Landsberg, James and Morton 1997; Landsberg *et al.* 1997) may not apply as consistently therefore to ant species. The response of an ant species to grazing may depend on a multitude of complex interactions that may vary substantially according to context (Morton, James and Landsberg 1996). This contextual variation might include the influences and interactions between type of stock and landscape (Milchunas *et al.* 1989; West 1993), seasonal history (Bestelmyer and Wiens 1996), evolutionary grazing regime (Milchunas, Sala and Lauenroth 1988), current grazing regime (West 1993), presence of invasive exotic plants (Whitford *et al.* 1999) or competing ant species (Andersen 1992).

This complexity leads on to the second factor: the paucity of research on grazing impacts beyond pasture production, and on native invertebrate fauna in particular. Given the limited historical research on these complex systems involving indirect, interactive and cascading effects, it is predictable that we have little understanding as to how a species or similar group of species may respond to grazing in any given situation.

Perspectives in environmental management and community concerns about rangeland use are currently broadening and intensifying (Morton and Price 1994; Abel and Ryan 1996; Pringle 1998a). Yet we are struggling to develop ways of implementing and monitoring ecologically sustainable development in Australian rangelands because of a traditional research focus on agricultural development (James, Landsberg and Morton 1995; Morton *et al.* 1995; Pringle 1995a).

Some distinctive patterns of the impacts of traditional rangeland use on native fauna are emerging. For instance, some granivorous ant species appear to be dependent on a particular plant species or similar groups of plant species. The disappearance or invasion of such plants may lead to a major decline or increase in the ant species involved (Andrew 1986; Whitford *et al.* 1999). Nevertheless, there is a long way to go before predictions can be made about more general impacts on ant fauna. A study such as this one is therefore particularly timely.

6.2.4 Sampling bias in pitfall trapping ants

Sampling bias can readily occur if the influence of habitat structure on the 'trap-ability' of ant species is not considered. This can lead to erroneous inferences regarding changes in ant species composition. Trap-ability may differ considerably between species. Some species may become easier to trap under certain habitat conditions, while others may become harder. Even within these broad groupings, there may be substantial differences in the

strength of the underlying bias and several different sources of bias (e.g. weather, trap avoidance and capacity to escape traps). Therefore, it has been suggested that any underlying bias should be evaluated as part of the experimental and analytical design of ant surveys (Melbourne 1999).

While I acknowledge the potential for bias in trap-ability, I attempted to minimise its impact. For instance I completed all sampling in the same season. I also installed and retrieved all the pitfall traps at a gradient on the same day and in random order as far as distance from water was concerned. I also used many traps in order to even out trap-to-trap variation.

Furthermore, the excellent seasonal conditions and profuse groundcover of ephemeral plants at the time of sampling effectively homogenised the level of 'landscape obstruction' that ants would be likely to encounter within any landscape stratum. I am therefore reasonably confident that variation in ant diversity in my study is due primarily to variations within landscapes and grazing history, and their interaction.

6.3 Methods

6.3.1 Sampling

Sampling was conducted from late November through December 1997. Gradient 6SM was re-sampled in February 1998 due to repeated flooding of this gradient. The weather was generally dry during trapping at all other gradients. The only exceptions were light rainfall (< 2 mm) recorded the night before retrieval of pitfall traps on Gradients 5CH and 7CH.

6.3.1.1 Pitfall trapping

I used plastic 30 ml medical urine vials of 30 mm diameter to trap ground dwelling invertebrates. The traps were filled with a mixture of 80 per cent water, 19 per cent propylene glycol, 1 per cent glacial acetic acid and a squirt of detergent; most traps had to be re-filled once due to evaporative loss. Pit fall traps were installed within one day at each gradient, making sure there was no bias in any sequence with respect to distance from water. The traps were left open for four nights and retrieved on the fifth day, in the same order as they were installed, thus keeping trapping time approximately even and timing of sampling (e.g. early morning installation and retrieval) independent of distance from water.

6.3.1.2 Site stratification

Sampling within sites was stratified in a similar manner to sampling for floristic diversity, with no stratification within chenopod sites and the usual stratification in groved and sandy mulga landscapes. Five pitfall traps were located at 5 m intervals along the central axis of each

sampling locus: in chenopod flats in chenopod landscapes, in mulga drainage lanes and sandy banks in sandy mulga landscapes, and in groves and adjacent inter-grove areas in groved mulga landscapes. I used five pitfall traps per landscape stratum (e.g. mulga grove or inter-grove) at each of five sampling loci within a site. Thus I used 25 pitfall traps per chenopod site and 50 traps per mulga site. See figures in Chapter 2 for diagrams showing the layouts of sites.

This differed in the size and number of pitfall traps to a similar research project conducted by CSIRO Wildlife and Ecology. I used a larger number of smaller traps over a larger trapping area to minimise the influence of mass captures of *Iridomyrmex* species that made sorting extremely difficult. This approach may also have reduced the bias in 'trap-ability' of other species by reducing the chance of individual traps being infested by *Iridomyrmex* to the exclusion of other species.

6.3.1.3 Sample processing

All five pitfall traps at any sampling locus were combined into one sample and placed in pure alcohol for transport back to the Canberra laboratory of CSIRO Wildlife and Ecology. There, the chenopod samples were sorted to morphospecies. The mulga ants were sent to Queensland Department of Primary Industry laboratories at Gympie, Queensland, for sorting.

Two technicians with considerable prior experience in sorting ants sorted the samples. Helen Tongway sorted chenopod samples at Canberra, while Cas van de Woude sorted the mulga samples at Gympie. Alan Andersen, CSIRO Wildlife and Ecology Darwin, identified all representative specimens of sorted morphospecies to species level although many of these species have not yet been formally described. There were four composite *Melophorus* species and three composite *Tetramorium* species because morphospecies were split into non-mutually exclusive species and I did not have the resources to have associated samples resorted.

Representative specimens are lodged at CSIRO Wildlife and Ecology's Darwin Laboratory.

6.3.1.4 Analysis

I followed the same analytical framework as that used for floristic diversity (see Chapter 5), except that I used species abundance scores instead of frequencies and did not undertake analysis at the finest scale ('species density') because I used pitfall traps rather than quadrats. The framework is summarised in Figure 5.1 of the previous chapter.

In brief, analysis involved assessment of patterns in total ant abundance and generic and species richness, followed by multivariate analysis to check landscape stratification. Finally individual species were sorted into response groups and their aggregated responses to grazing were analysed.

I assessed generic richness (i.e. number of genera represented in a sample) because membership of functional groups is usually consistent within genera and ant genera have been shown to be reasonable surrogates of species richness (Andersen 1995b; Pik, Oliver and Beattie 1999).

I conducted multivariate analysis of species abundance scores aggregated at the site-by-landscape-stratum level. Each abundance score represented a landscape stratum at a site, being the sum of individuals of a species caught at 25 pitfall traps (5 per sampling locus x 5 loci). This was done as a check on my landscape stratification; to determine whether the strata supported different ant assemblages. Because I had very high abundance scores in a few samples, I standardised the data matrix (places by species) by the sum of each column and row (Belbin 1995).

In the next phase of analysis I investigated individual and aggregated species responses to grazing. I commenced by determining for each species whether it showed a positive, negative or neutral trend in abundance with increasing grazing history (represented by modelled long-term average Stocking Rate). Trends in each landscape stratum (e.g. mulga groves and adjacent inter-groves) were treated separately. The analysis was conducted with abundance scores for individual species per sampling locus (i.e. not aggregated up to site level scores). The analysis was that used for floristic data. I assessed whether species abundance was significantly related to Stocking Rate (SR), using generalised linear modelling in Genstat 4.1 (Lawes Agricultural Trust 1997). Subsequently, each sampling locus was characterised in terms of the number of decreaser and increaser species present. I then used a standard REML model to investigate landscape and grazing effects on the number of increaser and decreaser species recorded at each sampling locus.

6.4 Results

6.4.1 Overall abundance

The mean number of ants caught at each sampling locus (set of 5 pitfall traps) did not differ significantly among landscape types ('landscape type' and 'stratum.type' model terms in Table 6.1). This was also reflected in a landscape type-grazing interaction for all three estimates of grazing history: Stocking History (SH) (Figure 6.1), Stocking Rate (SR) and distance from water (DW) (Table 6.1). Overall, ant abundance increased with increasing grazing history in mulga landscapes and decreased (slightly) in chenopod landscapes.

Exploratory regression analysis of the influence of Stocking History (SH) on abundance scores for different genera indicated that much of the increase in overall abundance in sandy mulga landscapes was due to increase in abundance of *Iridomyrmex* spp. ($p < 0.001$; interaction effect). Abundance of *Monomorium* spp. increased with increased Stocking History (SH) in inter-grove areas and declined in chenopod plains, mulga drainage lanes and sandy banks ($p=0.007$; interaction effect). In both cases the abundance scores were transformed to their natural logarithms in order to overcome systematic variation in residuals. Several other genera appeared to favour particular landscapes or parts of them (i.e. had significantly high abundance), but did not appear to be affected by grazing.

6.4.2 Richness

6.4.2.1 Generic richness

Alan Andersen distinguished 241 clearly different species in 26 genera. The most species-rich genera were *Monomorium* (25 species), *Melophorus* (24 species), *Iridomyrmex* (23 species), *Pheidole* (14 species) and *Tetramorium* (13 species). Just over half of all species recorded (133/241) occurred at least twice in at least one gradient (i.e. they were not 'singletons'). I include a species list of ants in Appendix 3.

The average number of genera trapped at each sampling locus was slightly higher in mulga groves than anywhere else (Figure 6.2). There were no effects of grazing history on generic richness. Exploratory analysis of the influence of Stocking History (SH) on species richness within genera indicated that species richness declined with grazing in *Melophorus* ($p<0.001$). No other genera showed significant relationships with grazing, though in some there was variation in species richness among and within landscapes. For instance, *Pheidole* species richness was higher in chenopod landscapes than mulga landscapes ($p=0.004$) and *Tetramorium* had higher species richness in mulga groves than anywhere else ($p<0.001$).

6.4.2.2 Species richness

The highest average number of ant species recorded per sampling locus occurred in the chenopod plains (Figure 6.3). Species richness per sampling locus declined with increasing grazing history, but only in the chenopod plains did the fitted range of species richness extend beyond one species (Stocking History (SH) model, Figure 6.4; Table 6.2). No significant effects of grazing were detected using the other two estimates of grazing history.

The most species rich genera in chenopod plains were all less rich in mulga landscapes. They include *Iridomyrmex* (13 species in chenopod plains), *Melophorus* (19 species), *Monomorium* (17 species) and *Pheidole* (9 species).

6.4.3 Influence of landscape stratification on species composition

Multivariate analysis of species composition showed strong landscape influences on the order and level of similarity in which objects were clustered (Figure 6.5). As with similar floristic analysis, each 'object' was a landscape stratum (e.g. mulga groves) within a site (i.e. distance from water at which sampling was centred) within a gradient/paddock. As with the floristic analysis, chenopod places tended to segregate according to gradient/paddock rather than order of sites away from watering points.

In the mulga landscapes, objects showed a strong tendency to cluster according to type of landscape stratum within landscape type or according to soil type (e.g. groves, inter-groves and mulga drainage lanes) rather than spatial proximity. In the floristic analysis there was more variable clustering, with strong similarities among different, but adjacent landscape strata (e.g. groves and inter-groves) within individual sites.

Another notable difference between the clustering according to floristic and ant composition was the integration of Gradient 3GM objects with other mulga objects in terms of ant composition. In contrast, the grazed sites at this gradient appeared floristically quite unlike their reference site or any other sites.

The high similarity between sandy bank places from sites closest to water at Gradients 4SM and 6SM (places 41W and 61W) was evidence of an underlying, but weaker influence of proximity to water (and its associated grazing) for this landscape unit.

6.4.4 Species response groups

All species sampled were native (Alan Andersen, personal communication). The number of singleton species recorded per sampling locus was not correlated with grazing history. On this basis I excluded singletons from any further analysis.

6.4.4.1 Species classification by response to grazing history

Chenopod plains had the greatest number of non-singleton species, followed by mulga groves. The number and proportion of decreaser species were considerably higher in chenopod plains, while increaser species were more evenly abundant among landscape strata (Table 6.3). Considering that I applied a 5 per cent significance threshold in the sorting process, proportions of response groups are not very much larger than might be expected if ant species were distributed randomly among the uniformly spaced sites (decreaser species in chenopod plains excepted). There was considerably less overlap in increaser and decreaser species composition compared with neutral species among landscape strata.

In general terms, *Melophorus* and *Mononomorium* provided most decreaser species (11/26 and 7/47 species respectively), while *Iridomyrmex* (9/40 species), *Monomorium* (6/47 species) and *Rhytidoponera* (6/16 species) comprised most of the increaser species. (The species ratios indicate designations of increaser, decreaser or neutral species within a landscape stratum). In only one instance did a species have contradictory designations in different landscape strata (i.e. was an increaser in one landscape stratum and a decreaser in another).

Within landscape strata, the following genera showed rather consistent trends and may therefore have indicator potential:

- *Camponotus* (2/3 species were decreasers on sandy banks),
- *Iridomyrmex* (3/7 species were increasers in mulga drainage lanes),
- *Melophorus* (10/19 species) and *Pheidole* (4/9 species) were decreasers in chenopod plains, and
- *Rhytidoponera* (3/5 species) and *Stigmacros* (2/2 species) were increasers in mulga inter-groves.

In summary, there were relatively large numbers of decreaser species in chenopod landscapes and low numbers of decreasers in mulga landscapes. Increaser species generally comprised between 10 and 20 per cent of the regional ant fauna sampled. I did not formally test the significance of these proportions and their differences because this was essentially a quantitative sorting process rather than a formal exercise in statistical modelling.

6.4.4.2 Variation in sensitivity to grazing within responses

The average number of decreaser species recorded per sampling locus in chenopod landscapes was significantly higher than that in mulga landscapes (Figure 6.6, Table 6.4), which is consistent with the higher proportion of decreaser species also sampled in the chenopod flora than in mulga flora (see last section).

There is an interaction effect between Stocking History (SH) and landscape stratum on the numbers of decreaser species at a sampling locus (Figure 6.7). The range of fitted values attributable to grazing is about four times greater in chenopod plains than it is in mulga landscape strata, where the range is less than or approximately one species. Thus it is only in chenopod landscapes that decreasers are a prominent component of the ant fauna.

While main effects of grazing were also statistically significant using the estimates of average long-term Stocking Rate (SR) and distance from water (DW), they were weaker and there were no significant interaction effects with landscape (Table 6.4).

Average numbers of increaser ant species recorded per sampling locus were highest in inter-grove areas, but did not vary substantially among strata (Figure 6.8, Table 6.5). While the influence of Stocking History (SH) was also statistically significant, it did not vary with landscape stratum and was also slight (Figure 6.9, Table 6.5). Stocking Rate (SR) had a less significant effect on numbers of increaser species and the effect of distance from water (DW) was not significant at all (Table 6.5).

6.5 Discussion

6.5.1 Summary measures of abundance and richness

6.5.1.1 Abundance

Ant abundance increased dramatically with increasing grazing history in sandy mulga landscapes, where the dominant dolichoderine genus *Iridomyrmex* was influential. A similar pattern has been reported elsewhere in arid Australian rangelands (Greenslade 1978; Landsberg *et al.* 1999) and abroad (Putman *et al.* 1989; Baines, Sage and Baines 1994).

While dominant dolichoderines appear to be favoured by disturbance elsewhere in the world, in Australia, *Iridomyrmex* appears to have an unusually strong influence on ant faunal composition. Members of this genus have the capacity to out compete other ant species that are otherwise common in disturbed habitats (Greenslade 1978; Scougal, Majer and Hobbs 1993; Bestelmyer and Wiens 1996).

In my study area the responses to grazing among species of *Iridomyrmex* were inconsistent however. It appears that only some species in this genus increase massively in abundance in disturbed areas. Other species were classified 'decreasers', which is consistent with the findings of Landsberg *et al.* (1999) and is the subject of ongoing research (Alan Andersen, personal communication). Therefore, the abundance of ants in sandy mulga landscapes might represent a co-incidence of grazing disturbance and landscape preference of some *Iridomyrmex* species in this region. The level of disturbance was certainly not as profound in this landscape as it was in chenopod landscapes (see Chapters 7 and 8). However, the nature of the disturbance was slightly different in this landscape compared to the other two. Grazing appeared to increase potentially mineralisable nitrogen, available phosphorus and nutrient cycling in this landscape (Chapter 8). Conceivably, the *Iridomyrmex* spp. were responding to this increased availability of resources.

6.5.1.2 Richness

There was little variation in the numbers of genera recorded among and within types of landscapes and no apparent influence of grazing history. Mulga groves had slightly higher numbers of genera than elsewhere, which is consistent with this landscape stratum being a particularly fertile patch (see patterns of Caesium-137 enrichment in Chapter 7 and topsoil fertility in Chapter 8).

The most species rich genera in this region belong to a range of functional groups and are similar to those recorded by Landsberg *et al.* (1999). They include dominant Dolichoderinae, hot climate specialists, opportunists and generalised myrmicines. Few species recorded belonged to the functional groups described as 'cold climate specialists', 'cryptic species' or 'large, solitary foragers and/or specialist predators' (Greenslade 1978; Andersen 1995a).

Chenopod plains had the highest species richness on average, consisting of a range of genera and associated functional groups. This may reflect strong fine-scale heterogeneity in bush mounds and exposed inter-bush areas and their influence on microclimate (Holldobler and Wilson 1990; Cristo and Wiens 1996). It may also reflect the absence of a strong *Iridomyrmex* influence in terms of competitive exclusion in grazed areas. Ant abundance tended to decline slightly with increased grazing history in chenopod landscapes.

While the strength of the effect of grazing history on different landscapes did not vary, there was a significantly greater loss of ant species attributable to grazing in chenopod landscapes than in mulga landscapes. This pattern is consistent with grazing-induced homogenisation of landscapes and habitats at the scale of individual ant distributions; co-incident with the

denudation of perennial vegetation and loss of fertile soil patches (see Chapter 8). These preliminary results are contrary to some of those of some other studies in Australia (Majer and Beeston 1996) and the United States (Heske and Campbell 1991; Whitford *et al.* 1999). Contrary to the contention of Whitford *et al.* (1999), the ant fauna in my study region does not exhibit 'remarkable resistance to human-induced disturbances', particularly not in chenopod landscapes. Indeed, the results of the analyses I have discussed so far indicate that ants may be at least as strongly affected by grazing as understorey plants in my study landscapes (see Chapter 5).

The use of number of genera, rather than number of species, would not have been 'taxonomically sufficient' (*sensu* Pik, Oliver and Beattie 1999) for assessing the impacts of grazing on richness within the ant fauna. Patterns of richness among landscape strata also varied between these taxonomic levels.

6.5.2 Species composition and landscape stratification

The multivariate analyses confirmed that my stratification within landscapes was also an appropriate stratification of ant assemblages. It also supported my proposition that components of the flora, rather than the landscape type itself, were anomalous at Gradient 3GM (see Chapter 5). This suggests that ant species composition may correlate more reliably than floristic composition with landscape differentiation in this region, as has been reported elsewhere (Whitford *et al.* 1999)

In several instances, similarity between 'places' more closely reflected edaphic and landscapes characteristics than spatial proximity or paddock memberships. There were also clusters indicative of an underlying grazing influence. For instance, the sandy banks closest to water at Gradients 4SM and 6SM had ant fauna more similar to each other than to other sandy banks in their paddocks.

6.5.3 Species response groups

6.5.3.1 Species classification by response to grazing

There was no significant effect of grazing on the number of singleton species (species recorded at one site only in a gradient of a particular landscape type), which is consistent with previous findings in this biome (Landsberg *et al.* 1999). This result suggests that neither invasive species nor grazing-refuge species are a major part of this region's ant fauna. This suggestion is based on the notion that invasive species are likely to be restricted to only the sites most heavily impacted near watering points and grazing-refuge species are likely to survive only at sites farthest from watering points. The results are indicative rather than

conclusive; my low sampling intensity within the region does not warrant a bold statement regarding rare or restricted species. The lack of evidence for an ant fauna associated with grazing-refuges could also reflect a relatively weak (indirect) effect of grazing because of the inherent stability of these landscapes in most areas (see Chapter 8).

The most striking result in terms of putative increaser and decreaser species is the considerably greater number and proportion of decreaser species in chenopod landscapes than in mulga landscapes. *Melophorus* (a hot-climate specialist genus) provided over a third (10 species) of chenopod decreaser species and more decreaser species than there were increaser species (8) in this landscape type. In contrast, Landsberg et al (1999) recorded more increaser than decreaser species for this genus at all four of their chenopod grazing gradients. *Pheidole* was the other genus that provided clearly more decreaser species (4) than increaser species (1) in this landscape. This is consistent with the findings of Landsberg *et al.* (1999) at two Western Australian gradients, but contrary to their findings at the other two gradients they sampled. (Note that the same sorter and identifier of the ants was used at all gradients; Jill Landsberg, personal communication).

There was a remarkable lack of contradiction of responses to grazing by individual species; in only one case was a species classified as an increaser in one landscape stratum and a decreaser in another. This lack of contradictory species response at a regional level contrasts with the apparent contradictions reported within genera at a biome or continental level. Although differences among studies might be partly methodological, they also highlight the need to develop indicators of response to grazing or 'biodiversity condition' that are specific to landscapes within regions.

That the majority of responsive species in different landscape types and parts of landscapes belonged to different genera indicates that a generalised, regional set of ant indicator species may not exist. This situation is no different for plants in this region (Mitchell and Wilcox 1984; Pringle 1994b; 1998b) and highlights the need for a more comprehensive understanding of the factors that influence ant (and plant) species patterns of distribution.

6.5.3.2 Variation in sensitivity to grazing within response groups

There were approximately five times as many decreaser species recorded on average in chenopod landscapes than in mulga landscapes. Not surprisingly, the major decline in decreaser species also occurred in chenopod landscapes. This pattern is also consistent with the notion of fine-scale homogenisation of this landscape due to grazing, as previously discussed.

While there were statistically significant increases in number of increaser species sampled with increased grazing, they were ecologically far less significant than losses of decreaser species. This is illustrated by the co-incidental declines in decreaser and total species richness in chenopod landscapes. Increaser species did not effectively 'make up the numbers'.

It has been suggested that a network of heavily grazed areas within a region may be advantageous in maintaining regional biodiversity. The idea is that these heavily impacted areas provide a scattering of habitats ideal for that part of the fauna that relies on disturbed areas for persistence (Bestelmyer and Wiens 1996). However, this might not be so important in my study region. The contemporary ant fauna has evolved in conditions quite different to the disturbance imposed by pastoral development; they are not evolutionarily adapted to massive grazing-induced disturbance (Milchunas, Sala and Lauenroth 1988). Thus my grazing increaser species may be fortuitously co-adapted to recent, anthropogenic regimes, rather than being dependent upon them to persist in these ancient and stable landscapes (Pringle 1994c).

My results contrast strongly with some previous work (Heske and Campbell 1991; Whitford *et al.* 1999). Contradictory findings seem to be the norm in this area of research, be they within studies (such as my landscape-grazing interactions), within biomes (Landsberg *et al.* 1999) or beyond (Putman *et al.* 1989; Bestelmyer and Wiens 1996; Landsberg, James and Morton 1997; Bromham *et al.* 1999; James, Landsberg and Morton 1999). While some of the variation may reflect different research methods, it also seems likely that responses of ant fauna to grazing vary considerably among and within environments.

In this region at least, grazing-induced changes to the diversity of ant habitats and the capacities of individuals to persist in the face of competition from other individuals and species seem to be profound. I have not untangled the influences of habitat change, intra-specific and inter-specific competition and the complex interactions that they involve. However, this will be a necessary precursor to predicting the spatial and temporal effects of pastoral development and systematically pursuing ecologically sustainable development.

6.5.3.3 Regional and wider implications

Walker (1996) proposed a framework for assessing options for future rangeland use in the context of four quadrants derived from combinations of 'high' and 'low' pastoral (economic) value versus nature conservation value. In this study on ants, landscapes and grazing, Walker's framework illustrates some interesting contrasts. Mulga landscapes have moderate to low pastoral value (Pringle 1994a) and grazing seems to have little effect on their ant fauna. Thus mulga landscapes belong to the 'low'-'low' quadrant, where there is little tension between the two perspectives on value. In contrast, chenopod landscapes have relatively high pastoral value across the southern Australian rangelands (Harrington, Wilson and Young 1984) and a significant component of the ant fauna in this landscape seems unable to adjust to heavy grazing in my study region. These chenopod landscapes thus fit in the 'high'-'high' quadrant of Walker's value space and are therefore likely to be the landscapes in which current patterns of land use and the future maintenance of biodiversity seem least compatible.

This pattern of variation in intensity of potential conflict among the different types of landscapes is mirrored in the degree to which the landscapes have been severely degraded and the levels of inadequacy of their reservation in this region. There appears to be a co-incidence of high pastoral value, considerable historical degradation, ongoing threat from grazing and particularly poor representation in nature reserves (Wilcox and McKinnon 1972; Pringle 1994a; 1995b; Morton *et al.* 1995).

As I discussed in the previous chapter, the study region is less intensely managed and has a shorter history of pastoral management than many other rangeland areas. It was largely spared what are today seen as the excessively high stocking rates used in the early days of Australian pastoral development. It is quite possible that the ant faunas of other rangeland regions have been disrupted considerably more by grazing than is the case in my study.

6.6 Summary

(1) The influences of landscape variation and grazing history on ant faunas were investigated by pitfall trapping.

(2) Grazing had a positive effect on the abundance of ants trapped at a local level, particularly in sandy mulga landscapes. In contrast, species richness declined with increasing grazing history, particularly in chenopod landscapes.

(3) Variation in the composition of ant species mirrored my landscape stratification and indicated that my dropping of floristic data from Gradient 3GM was justified. There was evidence that ant assemblages may reflect variations in the physical landscape more closely than plants.

(4) Over a third of the (non-singleton) ant species in chenopod landscapes were classified as decreasers. The percentages of ant species classified as increaser species were far lower in all landscape types and strata. This suggests that a substantial proportion of the ant fauna in chenopod landscapes has not adjusted well to heavy grazing in my study region.

(5) There was a major decline in the number of decreaser species recorded as grazing increased in chenopod landscapes. A decline also occurred in mulga landscapes, but was much lower. Numbers of increaser species increased as expected in all landscapes, but the range was low and small.

(6) Ants appear to be more sensitive to grazing than plants in chenopod landscapes. Patterns in ant diversity with regard to landscape and grazing interactions were at least as distinct as patterns in floristic diversity. The responsiveness of ants to grazing in my study landscapes is in marked contrast to results from some North American studies.

(7) The genera and species comprising increaser and decreaser groups varied considerably among and within landscapes. There were also numerous landscape effects on measures of ant diversity, both among and within landscape types. This highlights the need for sound landscape stratification before the effects of land use can be assessed.

(8) These findings indicate considerable tension between nature conservation and pastoralism, particularly in chenopod landscapes.

(9) Ants proved to be a responsive group for including when assessing the ecological sustainability of rangeland management in this region.

Table 6.1 Effects of grazing history on ant abundance

(See Glossary: Appendix 1 for terminology and Chapter 2 for explanation of modelling terms.)

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history, derived from modelling track densities and expressed in sheep years per hectare	landscape type	0.2	1	n.s.
	SH	0.1	1	n.s.
	stratum.landscape type	3.4	3	n.s.
	landscape type.SH	10.0	1	0.002
	stratum.landscape type.SH	11.8	3	0.008
SR: Stocking rate, average long term stocking rate developed from RANGEPACK/Paddock model and expressed in sheep per hectare	landscape type	0.2	1	n.s.
	SR	0.6	1	n.s.
	stratum.landscape type	3.4	3	n.s.
	landscape type.SR	8.0	1	0.005
	stratum.landscape type.SR	10.5	3	0.02
DW: Distance from water, measured in kilometres	landscape type	0.2	1	n.s.
	DW	3.7	1	n.s.
	stratum.landscape type	3.4	3	n.s.
	landscape type.DW	8.7	1	0.003
	stratum.landscape type.DW	4.9	3	n.s.

Table 6.2 Effects of grazing history on ant species richness

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	4.6	1	0.03
	SH	6.3	1	0.01
	stratum.landscape type	14.2	3	0.003
	landscape type.SH	0.1	1	n.s.
	stratum.landscape type.SH	2.7	3	n.s.
SR: Stocking rate (sheep/ha)	landscape type	5.2	1	0.02
	SR	2.7	1	n.s.
	stratum.landscape type	14.2	3	0.003
	landscape type.SR	0.1	1	n.s.
	stratum.landscape type.SR	1.0	3	n.s.
DW: Distance from water (km)	landscape type	4.5	1	0.03
	DW	1.3	1	n.s.
	stratum.landscape type	14.1	3	0.003
	landscape type.DW	0.4	1	n.s.
	stratum.landscape type.DW	1.8	3	n.s.

Table 6.3 Summary of species responses to grazing

(non-singletons only)

Landscape stratum	Decreaser species	Increaser species	Neutral species	Total
Chenopod plains	26 (35%)[1]	8 (11%)	40 (54%)	74
Mulga drainages	1 (3%)	6 (17%)	29 (80%)	36
Sandy banks	5 (13%)	4 (10%)	30 (77%)	39
Mulga groves	4 (8%)	8 (15%)	40 (77%)	52
Inter-groves	1 (3%)	8 (21%)	29 (76%)	38
Total[2]	35 (22%)	29 (18%)	97 (60%)	133[3]

[1] Percentages are calculated across rows.

[2] Totals are corrected for species that occur in more than one landscape stratum.

[3] The total number of non-singleton species, corrected for species occurring in more than one landscape stratum or response group.

Table 6.4 Effects of grazing history on richness of decreaser ant species

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	10.5	1	0.001
	SH	19.3	1	< 0.001
	stratum.landscape type	70.5	3	< 0.001
	landscape type.SH	0.0	1	n.s.
	stratum.landscape type.SH	10.0	3	0.02
SR: Stocking rate (sheep/ha)	landscape type	13.0	1	< 0.001
	SR	17.8	1	< 0.001
	stratum.landscape type	56.2	3	< 0.001
	landscape type.SR	0.4	1	n.s.
	stratum.landscape type.SR	5.3	3	n.s.
DW: Distance from water (km)	landscape type	9.4	1	0.002
	DW	5.2	1	0.02
	stratum.landscape type	56.1	3	< 0.001
	landscape type.DW	1.6	1	n.s.
	stratum.landscape type.DW	4.7	3	n.s.

Table 6.5 Effects of grazing history on richness of increaser ant species

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	0.5	1	n.s.
	SH	14.9	1	< 0.001
	stratum.landscape type	29.2	3	< 0.001
	landscape type.SH	3.6	1	n.s.
	stratum.landscape type.SH	4.1	3	n.s.
SR: Stocking rate (sheep/ha)	landscape type	0.6	1	n.s.
	SR	9.5	1	0.002
	stratum.landscape type	28.7	3	< 0.001
	landscape type.SR	3.6	1	n.s.
	stratum.landscape type.SR	2.9	3	n.s.
DW: Distance from water (km)	landscape type	0.3	1	n.s.
	DW	3.2	1	n.s.
	stratum.landscape type	28.4	3	< 0.001
	landscape type.DW	1.5	1	n.s.
	stratum.landscape type.DW	3.6	3	n.s.

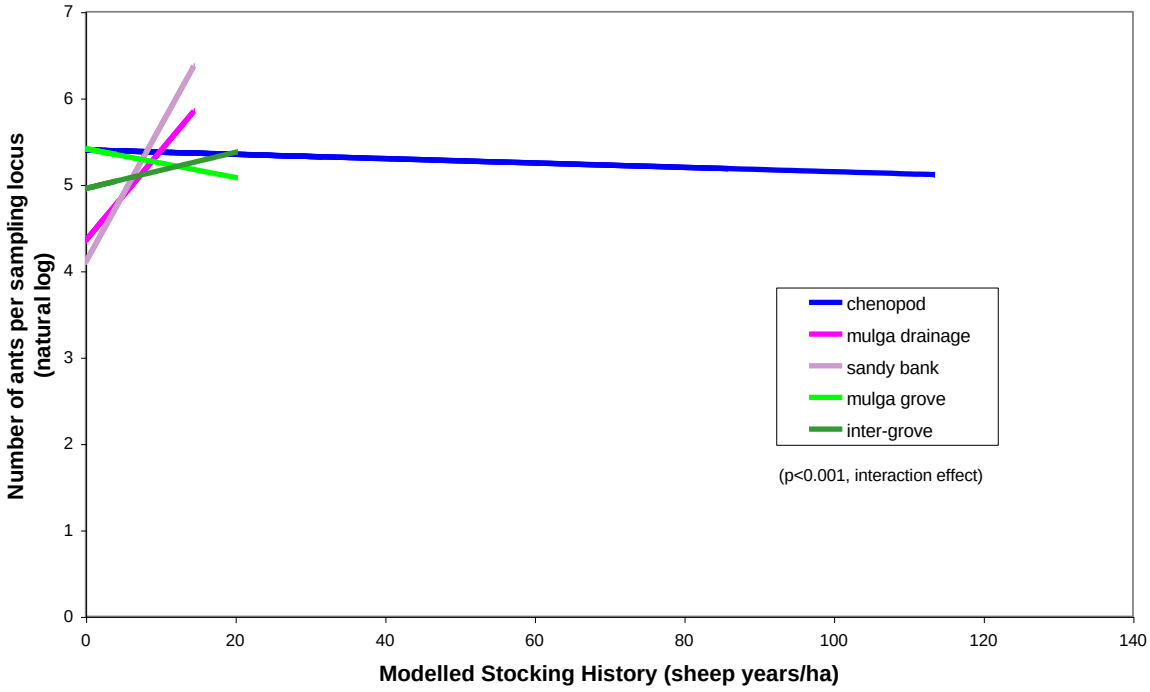


Figure 6.1 Impacts of stocking history on ant abundance.

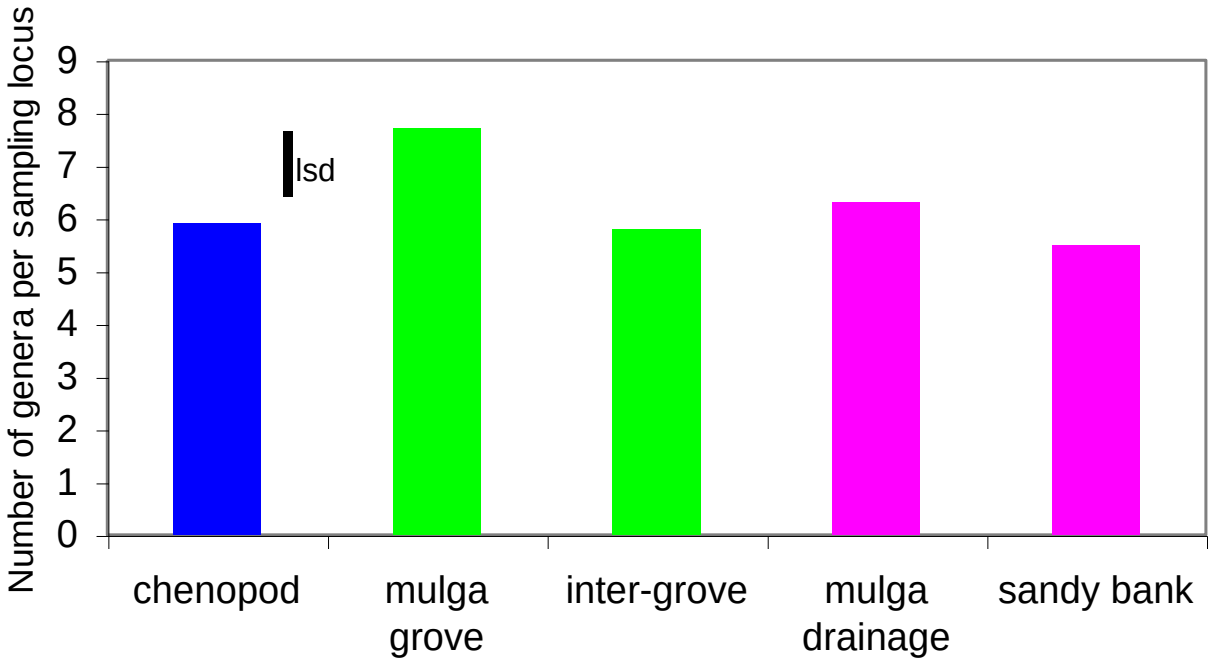


Figure 6.2 Variation in ant generic richness with landscape stratum.

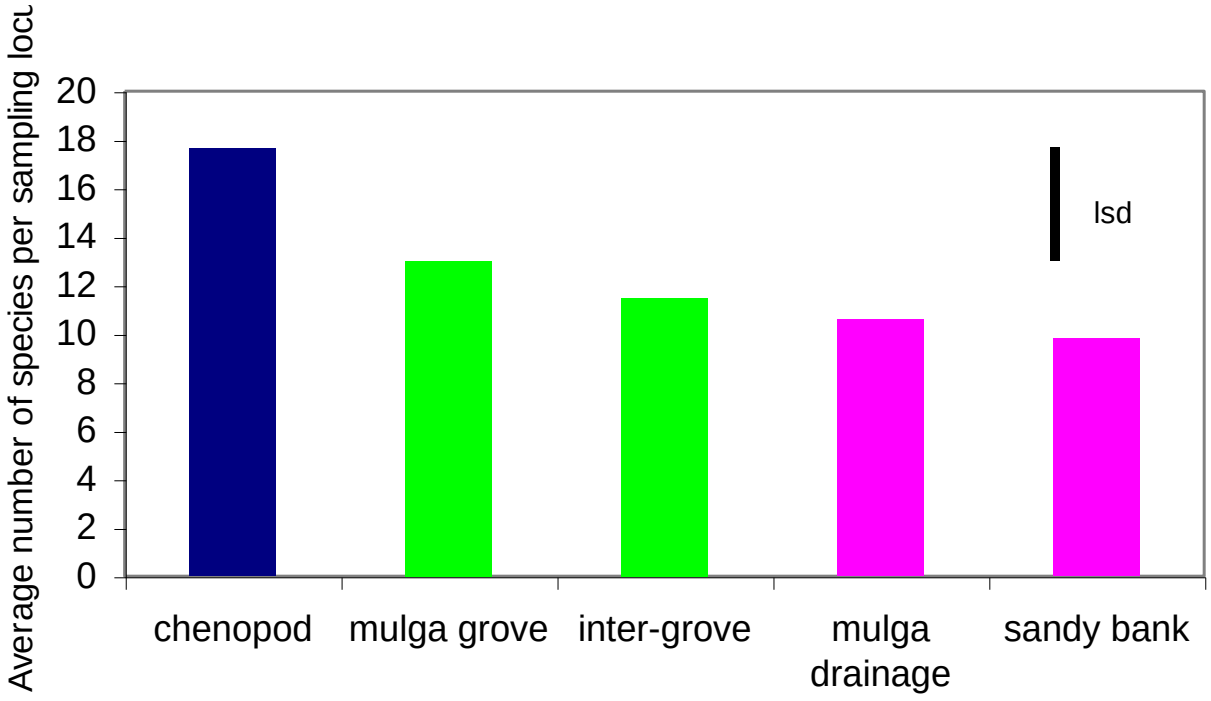


Figure 6.3 Variation in ants' species richness with landscape stratum.

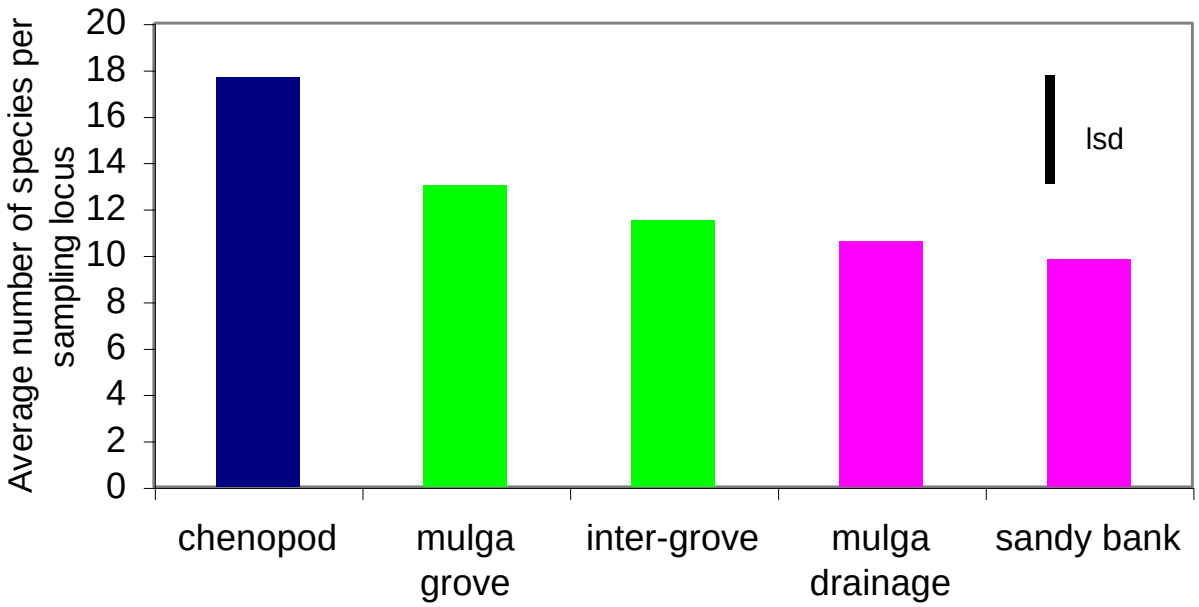
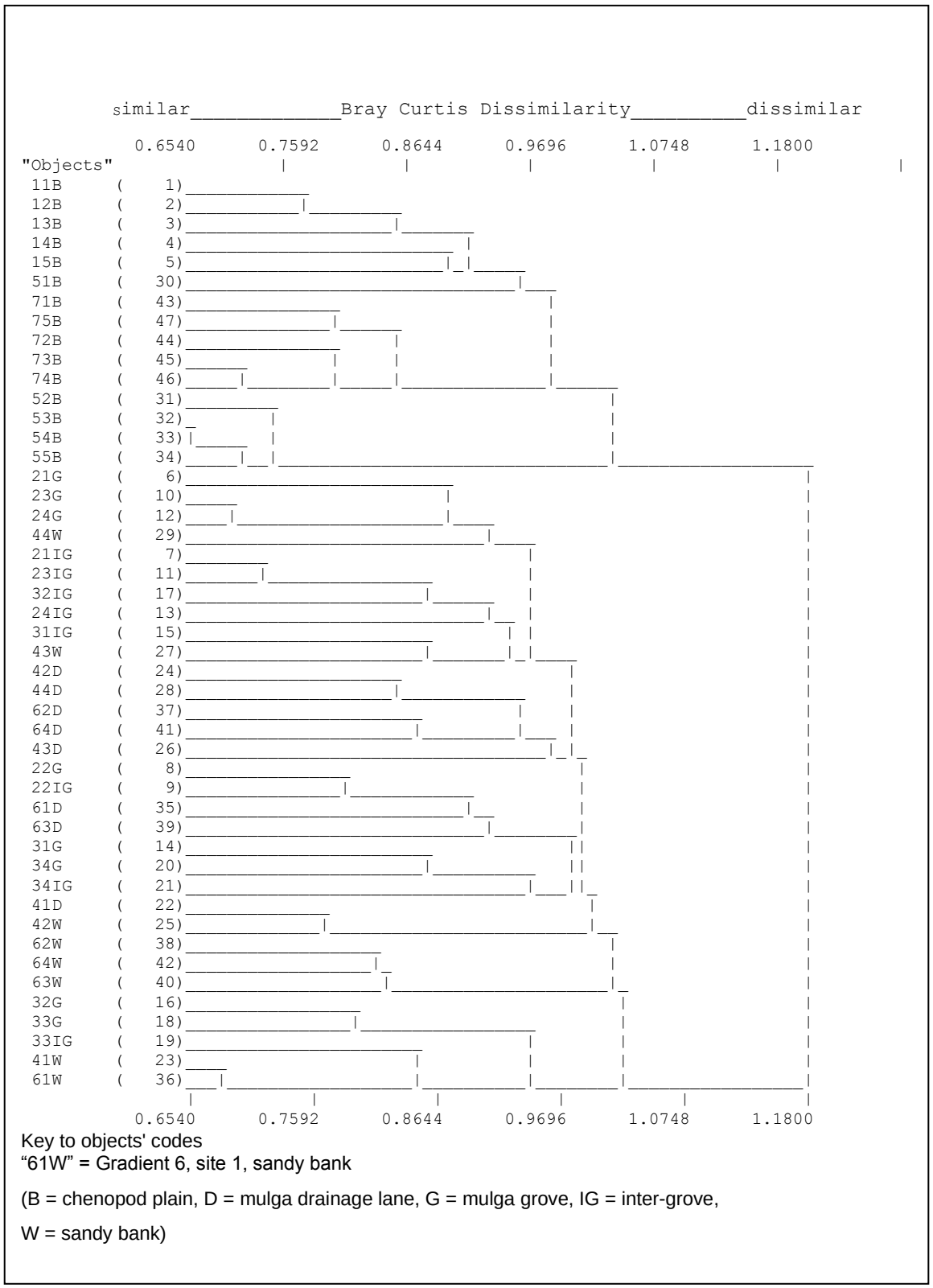


Figure 6.4 Impacts of stocking history on ant species richness.



Key to objects' codes
"61W" = Gradient 6, site 1, sandy bank

(B = chenopod plain, D = mulga drainage lane, G = mulga grove, IG = inter-grove,

W = sandy bank)

Figure 6.5 Dendrogram of places by ant species composition.

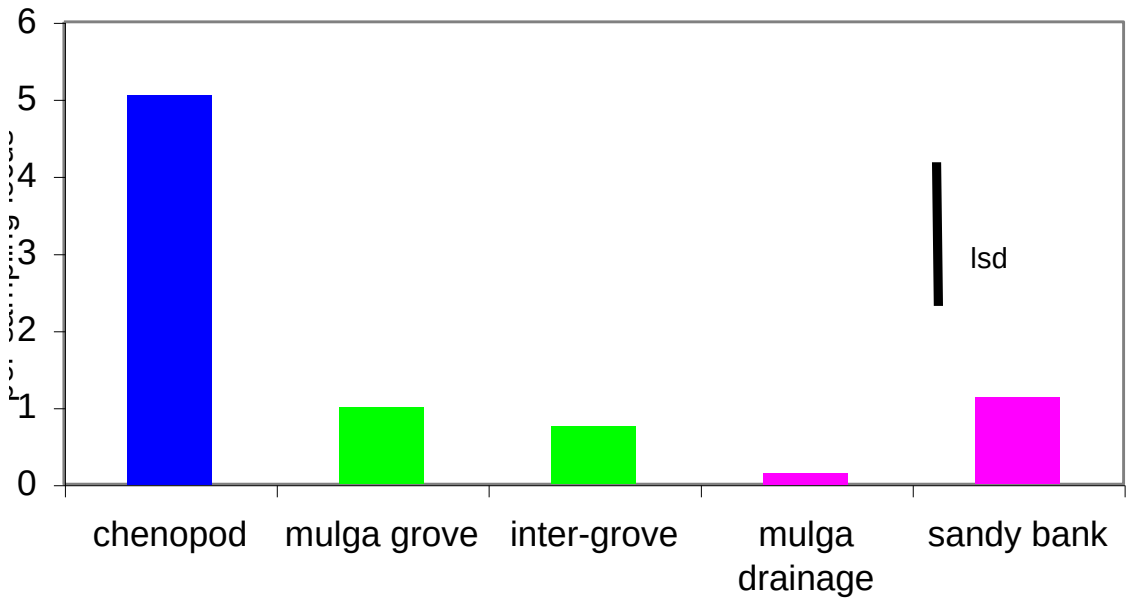


Figure 6.6 Variation in decreaser ant species richness with landscape stratum.

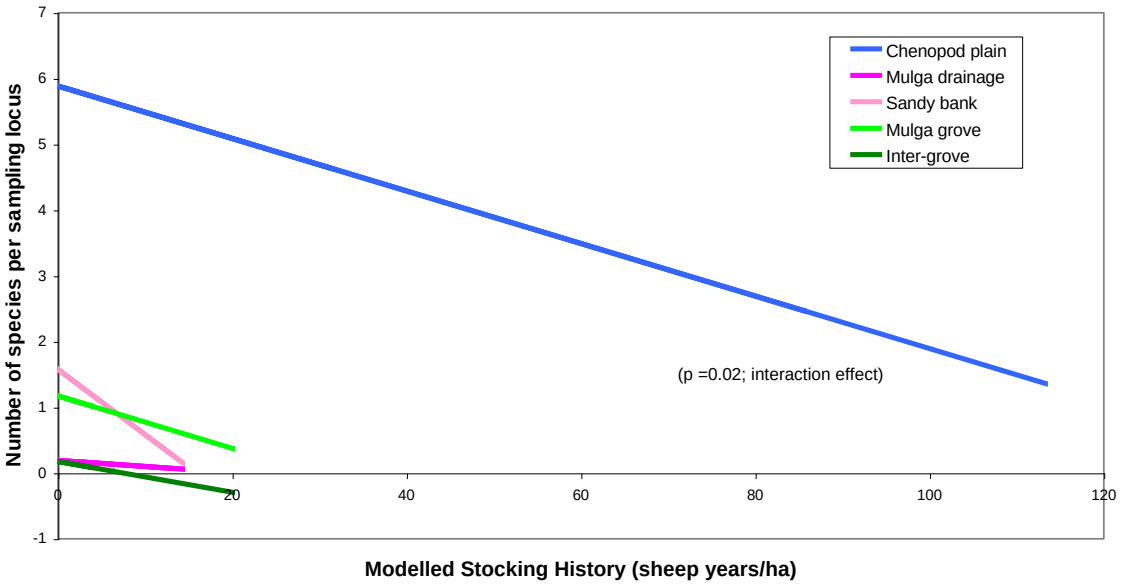


Figure 6.7 Impacts of stocking history on decreaser ant species richness.

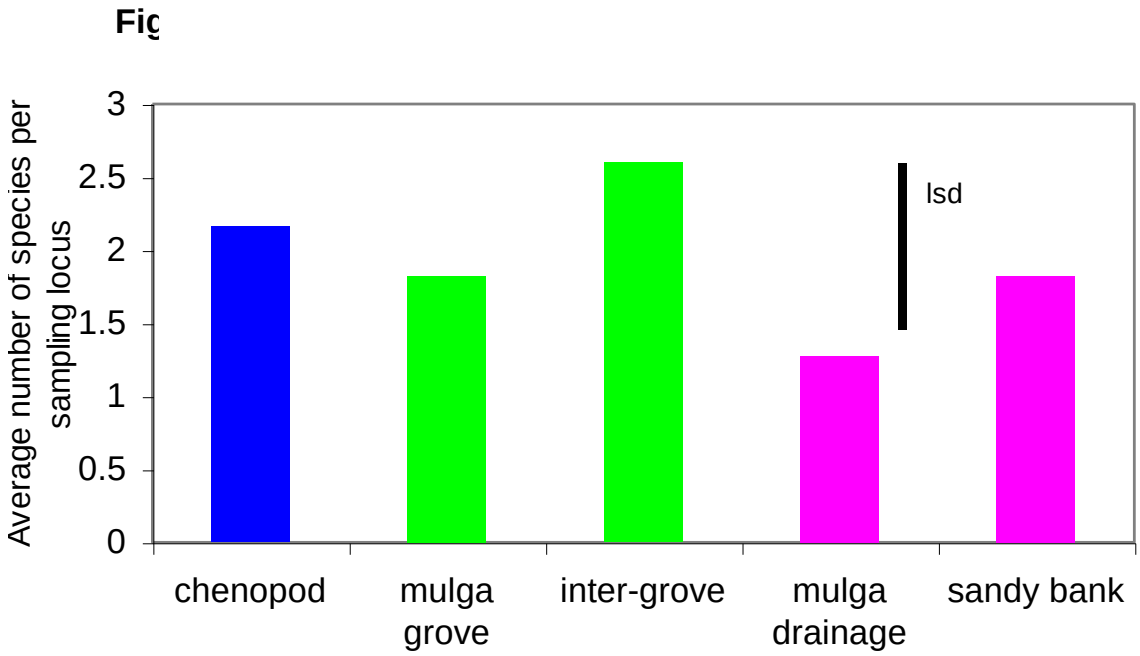


Figure 6.8 Variation in increaser ant species richness with landscape stratum.

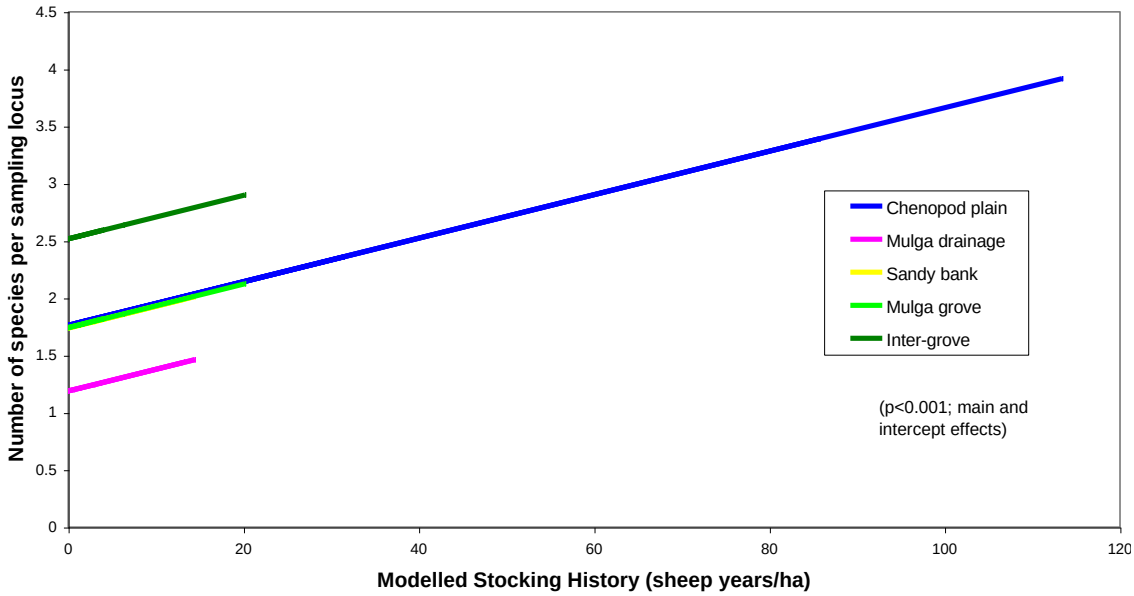


Figure 6.9 Impacts of stocking history on increaser ant species richness.

CHAPTER 7. LANDSCAPE ORGANISATION

7.1 Introduction

In this chapter I investigate landscape patterns and the processes supporting them within the landscapes I am studying, with particular emphasis on the influence of grazing on these patterns and processes. This is the first of two inter-related chapters: in it I investigate the impacts of grazing on the way landscapes are organised. In the next chapter I investigate properties of topsoils within landscape strata. My perspective in these chapters is ecological integrity: how the ecosystems studied are organised and function when grazed, in relation to their least disturbed state, regardless of the species involved. This perspective contrasts with *biological* integrity (see Chapter 1 for further discussion of these concepts). Species composition - or representation of different functional groups of similar species - undoubtedly also influences ecosystem patterns and processes (Slatyer 1975; Archer 1995; Walker, Kinzig and Langridge 1999), but I have chosen to assess *biological* and *ecological* integrity perspectives separately in this study. However, I will explore some of the links between species, their functional groups and landscape patterns and processes in my synthesis chapter.

In this chapter I integrate two independently derived approaches to assessing landscapes. Firstly, I use the Landscape Function Analysis approach of David Tongway and John Ludwig (Tongway and Hindley 1995; Ludwig and Tongway 1995a; b) to identify recurring landscape 'compartments' (*sensu* Noy-Meir 1981). I then use the concentration of Caesium-137, a fallout radionuclide to assess water, detritus and sediment distribution patterns. Together these approaches provide an assessment of patterning within landscapes and the strength of redistribution processes between landscape compartments. I begin with a review of these and other approaches.

7.2 Organisation of arid landscapes

It has been hypothesised that the most efficient organisation of arid landscapes with limited resources may be strongly patterned (Noy-Meir 1981) or, contradictorily, regular (Woodell, Mooney and Hill 1968). Most of the arid landscapes described to date appear to be strongly patterned, or clumped (Williams 1955; Charley and Cowling 1968; Burrows and Beale 1969; Slatyer 1975; Stafford Smith and Pickup, G. 1990; Cornet *et al.* 1992; Montana 1992; Whitford, Martinez-Turanzas and Martinez-Mesa 1995; Ludwig *et al.* 1997) and this seems also to apply to the landscapes in my study region (Mabbutt *et al.* 1963; Hacker 1978; Pringle, Van Vreeswyk and Gilligan 1994).

Landscape patterning is nested at different levels of ecological organisation, and these levels are typically expressed at different spatial scales (e.g. Pickup 1985; Bergkamp 1995). However, only recently have frameworks been developed to accommodate describing the complexity of nested hierarchies of spatial pattern; (Allen *et al.* 1984; O'Neill *et al.* 1986; Allen and Hoekstra 1992; Lavorel, Gardner and O'Neill 1993; Ahl and Allen 1996). Specific techniques have also recently been developed to apply these frameworks to landscape assessment (Tongway 1994; Bergkamp 1995; Tongway and Hindley 1995; Pickup 1996).

The processes involved in developing and maintaining landscapes also appear to differ markedly with scale and level of organisation (Pringle 1994c; Bergkamp 1995; Ludwig and Tongway 1995a; 1995b). Broad-scale patterns (involving distances of tens of kilometres and more) are often controlled by terrain and regolith history, reflecting characteristic patterns of regional landform development (Jennings and Mabbutt ; Mabbutt 1963a; Ollier *et al.* 1988; Wyrwol and Glover 1988; Gillieson 1993; Pringle 1994c). Nested within broad-scale, physiographic regions and catchments are repeating landscape-scale patterns often referred to as land systems (Christian and Stewart 1968; Pringle, Van Vreeswyk and Gilligan 1994) land units are nested in these (Mabbutt 1963b; Christian and Stewart 1968; Mabbutt and Fanning 1987; Pringle 1994c). In arid Australia, these physiographic regions, land systems and units are mostly of Tertiary age (Mabbutt 1963b; Pringle 1994c). It is only in the meso-scale (tens of metres) and fine-scale (metres to less than a metre) that observed patterns can mostly be attributed to contemporary processes. Thus there is a coupling of land shaping processes and the scales at which they operate, ranging from broad scale Tertiary and Quaternary patterns reflected in regional and land system patterns to contemporary, fine-scale processes which are most readily affected by land management.

In chenopod landscapes such as those along some of my grazing gradients, wind is the major process responsible for the development of the fine-scale pattern of bush mounds and inter-bush areas (Charley and Cowling 1968; Tongway and Ludwig 1994). Perennial shrubs trap wind-born sediments and gradually a mound of relatively coarse-textured material accumulates. The coarse texture of sediments may reflect aggregation of entrapped fines as well as saltating coarse particles.

In groved mulga landscapes of my study region, terrain-modulated fluvial processes are dominant (Slatyer 1962; Mabbutt and Fanning 1987; Tongway and Ludwig 1990). Bands of vegetation (‘groves’) occur on level areas separated by larger, slightly inclined, open areas. Resources are redistributed by diffuse sheet flow from the gentle, open slopes to the groves.

The primary patterning in the sandy mulga landscapes of my study region consists of sandy bank land units interspersed with mulga drainage lane units. This pattern is attributed to the interaction of aeolian and fluvial processes acting over many thousands of years (Mabbutt 1963b). The patterns and processes occurring in the mulga drainage lanes are similar to, but less clearly defined than those in groved mulga landscapes. On sandy banks, there is very little surface flow of water as the soils are coarse and highly porous. Nevertheless, after heavy rains when the soil profile becomes saturated or flow within drainage lanes becomes quite deep (say >20cm), there may be diffuse surface flows on sandy banks (Tongway and Ludwig 1994; 1995a; Ludwig and Tongway 1995b). More commonly though, aeolian process operating at fine scales (grass butts or between groups of grass butts) predominate (Tongway and Ludwig 1994).

It would appear then that fluvial processes dominate at the meso-scale level and aeolian processes at fine scales in my study area, as occurs elsewhere in Australia (Ludwig and Tongway 1995b). This should result in recognisable forms of patterning in perennial vegetation (Cornet *et al.* 1992; Archer 1995; Bergkamp 1995; Gillieson, Wallbrink and Cochrane 1996). This assumption is tested in this and the next chapter.

7.3 Models of grazing impact on landscape organisation

7.3.1 The process of change

There is considerable debate as to which models of change best represent rangeland dynamics within management time frames (Heady1975; Mentis *et al.* 1989; Smith 1989; Westoby, Walker and Noy-Meir 1989a; b). The crux of the debate is about whether change is orderly, continuous and reversible or unpredictable, episodic and not necessarily reversible in time scales of land management. The first view is represented by the traditional North American paradigm developed by range ecologists from the theory of plant succession (Clements 1916). It is a relatively simple view of how rangelands are affected by grazing and is based on the idea that changes may be reversed in an orderly manner by manipulating grazing pressure.

The second view is extremely different; it emphasises unpredictability, non-linear rates of change, thresholds, hysteresis and multiple semi-stable domains of attraction. It is a complex view of rangeland dynamics and emphasises the importance of events and irreversibility in human management time frames. Parallel debate occurs in geomorphology, relating to the contrast between stochastic, steady state processes and episodic, event driven processes and thresholds. For instance, it has been proposed that models of erosion developed in croplands are inappropriate for predicting erosion in rangelands (Pierson 2000).

In recent times there has been a perception that the pendulum of rangeland management has swung too far from the simple, continuous view to the complex, event-driven view (Watson, Burnside and Holm 1996). In reality, rates of change occur as a continuum, from highly episodic to gradual. Perception of the continuity of change may be strongly influenced by the time frame at which it is assessed (Allen and Starr 1982) and appropriate timeframes need to be selected to address particular questions or objectives. In this chapter I am interested in how grazing affects landscape patterning, a process that generally takes years to decades to occur in arid environments, due to considerable inertia (Noble 1986; Archer and Smiens 1991). In contrast, a considerable smaller time frame might be more appropriate for changes in matching ephemeral pasture and tactical stocking decisions (Hacker and Tunbridge 1991; Hodgkinson 1993; Watson, Burnside and Holm 1996).

Timeframe of assessment also influences perceptions of irreversibility. For instance, infestation of grasslands by shrubs may seem irreversible to the current land manager, but perhaps an intense wildfire with a periodicity of over a century might reverse the change.

7.3.1.1 Gradual (reversible) change: single climax communities

Plant Succession Theory was developed by Frederick Clements from studies on landform and vegetation evolution away from the edges of the Michigan Lakes (Clements 1916). The theory was adapted by range ecologists to develop an ecological approach to range management (Sampson 1919; Dyksterhuis 1949; 1958). There appears to have been some over-extension or misinterpretation of the original theory in this process, and a paradigm of range management became established based on the dubious assumption that manipulation of grazing pressure alone can shift vegetation towards or away from a single climax state in perpetuity. Single climaxes and reversibility have been shown not to apply in many arid rangelands around the world (Heady 1975; Connell and Slatyer 1977; Smith 1978; Mentis *et al.* 1989; Westoby, Walker and Noy-Meir 1989a; b; Whitford, Martinez-Turanzas and Martinez-Mesa 1995). It is possible that some of the severe degradation that has occurred since the establishment of this model as the paradigm for range management might be due to the failure of managers to acknowledge the risk of permanent (irreversible) change to both vegetation and the landscape as a whole.

Clements paid close attention to the coupling of landscape and vegetation patterning and process in an eco-region of low variability and high lability. It seems that many of the interpreters of his work may have failed to do the same in more stable, but unpredictable rangeland environments under a land use whose disturbance regime bore little semblance to

that in which the environments had evolved. Critically the relationship between grazing management and landscape patterns and processes was often overlooked.

Although the simple, traditional model appears inappropriate for many range landscapes, it may apply to some low energy chenopod landscapes; where areas almost entirely denuded by drought and grazing in the early 1970s are no longer obviously different to analogous landscapes that escaped the denudation (Pringle 1994b).

7.3.1.2 Punctuated (irreversible) change

Evidence contradictory to the traditional climax ecological paradigm accumulated in the 1960s and 1970s, leading to published criticisms of this paradigm (Noy-Meir 1973; Slatyer 1975; Connell and Slatyer 1977; Westoby 1980). These criticisms included observations of apparent irreversibility of change despite reversal of management steps, and the apparent existence of thresholds, which when crossed led to large and often unpredictable changes.

Two complementary approaches developed from this period of re-assessment of range ecology. The first was the 'vital attributes' approach; in which functional groups of plant species were described and their response to disturbance (including grazing) was predicted (Noble and Slatyer 1980). This approach has not been widely used in rangelands, partly because it is essentially quantitative and requires considerable input of data and autecological knowledge of species within the system to be modelled. While this complexity may have detracted from its uptake by range ecologists, it is a considerable simplification of complex ecosystem dynamics.

The second, more conceptual approach is at the forefront of contemporary range ecology. The State and Transition model (Westoby, Walker and Noy-Meir 1989a; b) which forms the conceptual basis of the Society for Range Management's 'Rangeland Health' approach to assessing rangelands (National Research Council 1994). There are also parallels with the development of 'ecological integrity' for assessing rangelands in the USA (West *et al.* 1994). The critical feature of this model is the recognition of multiple semi-stable states and the transitional pathways between them. The concept is inherently allied with stable limit cycle theory (May 1972; Holling 1973); which states that systems vary within a domain of attraction until conditions are met for a sudden change to a new domain of attraction. A State and Transition approach to the art and science of range management is seen as recognising and directing or avoiding sudden changes between states; or managing small changes within the stable limits of desirable states (Slatyer 1975; Hacker and Tunbridge 1991).

By the early 1990s, most range ecologists had adopted the State and Transition model as their operational paradigm (Stafford-Smith 1992; Joyce 1993). More recently however, there has been growing realisation that some changes within a system are episodic and others may be subtler, more easily overlooked and yet lead to considerable change (Watson, Burnside and Holm 1996). A more inclusive approach may be to recognise that any system may be subject to a range of changes that can occur at different time-scales and with degrees of continuity (Archer and Smiens 1991). This view recognises that a system's capacity to self-organise may be considerable for much of the time, when energy transfer levels are low, but the system may also exhibit quasi-chaotic behaviour at higher (low frequency) energy conditions (Gunderson, Holling and Light 1995).

7.3.2 Patterns of change

Concerns that inappropriate grazing regimes can degrade rangeland landscapes were raised many decades ago (Dixon 1892; Ratcliffe 1936; Trumble and Woodroffe 1954) and several models have been proposed to predict or explain different patterns of degradation.

In this section I have classified some of these models in a hierarchical fashion (Figure 7.1). The first split is on the basis of whether degradation is expressed in terms of the scale (grain) of landscape patterning. Where degradation does affect grain size, I differentiate between whether patterning becomes coarser (2A, Figure 7.1) or finer (2B). In degraded landscapes, I discriminate between those that have been invaded by shrubs or trees ('woody weeds')(3A) and those from which perennial vegetation has been denuded (3B). These last two classes differ principally in that in the 'woody weed' scenario, a new broader scale of heterogeneity evolves; while in the denudation scenario, a level of heterogeneity is removed without replacement. Finally, I discriminate between denuded landscapes that are stable and seasonally productive (4A), and those that are variously eroded, salinised and much less biologically productive (4B).

7.3.2.1 No change in landscape pattern: successions within vegetation types (1A)

Some landscapes are prone to infestation by similar life forms. For instance, the perennial exotic grass *Eragrostis lehmaniana* appears to be actively spreading in the southern United States, out-competing native grass species of similar form (e.g. black gramma grass *Bouteloua eriopoda*) (Cox and Ruyle 1986; Anable *et al.* 1992; Whitford 1995; Rapport and Whitford 1999). Although there is homogenisation of species composition (McKinney and Lockwood 1999), major changes in landscape function are not inevitable.

Similar shifts are common in low energy chenopod landscapes of inland Australia, where preferentially grazed sub-shrub species may be replaced by more grazing-tolerant sub-shrub species without noticeable change in vegetation pattern or soil deterioration. The replacement of *Atriplex vesiacria* by *Maireana pyramidata* is a common example in the study region (Pringle 1994b).

7.3.2.2 Decreasing grain size: conversion of shrubland to grassland (2B)

In parts of north-western Australia low shrublands dominated by bladder saltbush (*Atriplex vesicaria*) and other chenopod shrubs have been replaced by the exotic perennial buffel grass (*Cenchrus ciliaris*). The replacement grasslands are thought to have enhanced value as pastures and their successful infestation is closely related to heavy grazing disturbance and denudation of native shrubs (Payne *et al.* 1987). This represents a shift from fine-grain patterning (shrub-scale) to even finer-grain patterning (grass-butt scale). Buffel grasslands appear to be quite stable, representing a system flip to a new stable domain of attraction (*sensu* Holling 1973). Native grasses such as *Stipa* species and the exotic woody herb *Carrichterra annua* have replaced large areas of chenopod shrubland in southern Australia (Mitchell, McCarthy and Hacker 1979). Fires started by coals from steam-driven trains, rather than grazing, have wrought these changes.

The pattern of reduced grain-size is the opposite of many reported system flips in the USA, where degradation appears more commonly associated with flips from grassland to shrubland (Archer and Smiens 1991). This common American pattern of change may reflect overall steeper landscape gradients which facilitate change from self-organising fine scale systems to broader scale, fluvially integrated systems.

7.3.2.3 Increasing concentration of resources in fertile woody islands (3A)

Increasing dominance of landscapes by shrubs (and less frequently trees) appears particularly widespread in the south-western United States, where grasslands have been replaced by a number of woody species (Schlesinger *et al.* 1990; Archer 1995; Whitford, Martinez-Turanzas and Martinez-Mesa 1995). Similar patterns have been reported further north (Reynolds *et al.* 1999; Schuman *et al.* 1999), in southern Africa (Acocks 1964; Van Vegten 1983; Skarpe 1990; Scholes and Walker 1993), Australia (Burrows 1972; Harrington, Oxley and Tongway 1979; Lonsdale and Braithwaite 1988; Noble 1998) and elsewhere (Archer 1994). Shrub infestation does not necessarily result in degradation (i.e. a loss of primary productivity, or a loss of resource conservation), although there may be some soil redistribution in the transition process.

In landscapes where this model is invoked, homogeneity (at the relevant scale) is sometimes considered to reflect minimal anthropogenic disturbance, or good 'health' (e.g. Schlesinger *et al.* 1990; Whitford 1995). However, the process of change may progress to a stage where fertile islands eventually coalesce. Although the process of 're-homogenisation' from grasslands/savannahs to patchy shrublands and then to woodlands may be so slow as to defy normal systems of observation (Archer 1995), coalescence may eventually result in secondarily homogenous vegetation, albeit at a coarser grain. This re-homogenised state appears analogous to the 'noxious scrub' distinctive of overgrazed areas in much of arid eastern Australia (Noble 1998). One might propose that the speed with which re-homogenisation occurs may be influenced by the degree to which sediments have been redistributed and the resulting contrast in soil fertility in largely bare patches between fertile islands (Whitford, Martinez-Turanzas and Martinez-Mesa 1995).

In arid Australia, there is often a tendency for shrubs to establish as single plants at varying densities, rather than as discrete island thickets (Burrows 1972; Pringle 1994b; Noble 1998). This may reflect weak patterns of redistribution on coarse-textured soils subject only to very diffuse sheet-flow, leading to fine-scale redistribution about individual shrubs. Sediments in arid Australia are frequently more stable and fluvial processes may also be weaker, resulting in this weaker, finer-grained pattern of shrub encroachment.

7.3.2.4 Simplification of pattern: formation of stable productive herblands (4A)

This is a model proposed in Australia, partly as a response to the perception that Australian range ecologists were over-emphasising the importance of palatable perennial shrubs and grasses and overlooking the contribution of the ephemeral flora in range production (Wilson and Tupper 1982; Wilson and MacLeod 1991; Holm 1994). In the riverine plains of south-eastern Australia the removal of bladder saltbush (*Atriplex vesicaria*) and its replacement with forbs and grasses has been shown to maintain forage and animal production (Wilson and MacLeod 1991). A similar outcome was found in coastal chenopod plains of north-western Australia (Holm 1994), though subsequent economic modelling suggests that shrublands may be more productive in the long-run (Ian Watson, personal communication). Similar findings emphasising the retention of perennial forage species have been reported in North America (Holecheck 1996).

Some models describing patterns of change in sub-Saharan Africa are contradictory. In particular, losing grazing-sensitive perennial forage species did not necessarily represent degradation of the land's productivity in communal areas (versus commercial ranches). Communal lands have traditionally been regarded by range ecologists as degraded because

they lack the perennial decreaser species associated with more conservatively stocked commercial ranches. However, it has been argued that these communal lands may be quite productive and sustainable despite the loss of palatable perennial species in the understorey when the full range of benefits to communities is assessed on an area basis (e.g. per hectare) (Roe 1984; Hellden 1988; Hellden 1991; Dodd 1994; Behnke and Abel 1996a; b; c; Abel 1997).

The key to the maintenance of productivity in this pattern of change is stable soils: it would be unlikely to occur in areas with high energy gradients that might exacerbate soil loss, or in areas where any soil loss has serious consequences. For example, arid Australian soils are often shallow and infertile and usually have disproportionately high concentrations of nutrients in the top few centimetres of soil (Charley and Cowling 1968; Burrows 1972; Tongway and Ludwig 1994). The loss of soil and mineral capital from landscapes with soils like these is usually associated with a collapse in the capacity of the landscape to support any reliable form of vegetation cover (see below).

7.3.2.5 Denudation and loss of productivity

'Natural resource mining' best sums up this pattern of change. Vegetation is gradually removed without replacement in the face of sustained heavy grazing, leaving soils exposed to the elements. In worst cases, massive soil loss, sometimes accompanied by secondary salinisation result in desertification, that is, degradation of the landscape to a degree where its capacity to support life is virtually eliminated (Mainguet 1991; Milton *et al.* 1994).

Decertified landscapes are the end point in this model: the situation may be reversible before a threshold is crossed, or may become irreversible as degradation continues. What distinguishes this model is the lack of replacement of lost vegetation, a concomitant loss of mineral capital in the soil and a progressive decline in biological productivity.

This model appears to describe best some landscapes in my study region, namely those with fragile soils that are lost from the landscape on becoming detached from the soil mass. Breakaway footslopes and flood out plains supporting chenopod shrublands are good examples of such fragile landscapes identified in a recent rangeland survey of the study region (Pringle, Van Vreeswyk and Gilligan 1994).

7.3.3 Integrating pattern and process: Source-sink and erosion cell models

7.3.3.1 Conceptual frameworks

Noy-Meir is generally attributed with formalising the concept that arid ecosystems overcome scarcity of resources by concentrating and conserving them in relatively rich compartments (Noy-Meir 1973; 1981; 1985). Tongway and Ludwig (Tongway and Ludwig 1990; 1994; Ludwig and Tongway 1995b) further developed these ideas into a conceptual framework that might best be described as a 'Source-Sink' model (Ludwig, Tongway and Marsden 1994; Ludwig and Tongway 2000). The framework envisages that areas that shed resources (water, nutrients and detritus) are interspersed with smaller areas that trap and sequester resources, and these trapping areas are generally associated with patterns of perennial vegetation and other obstructions (e.g. logs). This is so because these structures represent obstacles to flows of water. By slowing the flow, reduce water's capacity to entrain and transport sediment, thereby facilitating infiltration and deposition of sediments and detritus. In contrast, source zones are largely free of obstructions and generally have relatively high hydraulic energy, while sinks are filters that slow flows.

This Source-Sink model provides a process-view of strongly patterned fluvial systems. Slopes and impedance to flows are critical in landscape function in this view of arid landscape patterns and processes.

Erosion cells have long been an integral part of conceptual models of landscape function within drainage basins (Schumm and Hadley 1957; Leopold, Wolman and Miller 1964; Schumm 1977). Pickup (1985) quantified the concept in an elaborate, hierarchical manner that was particularly apt for many rangeland landscapes in Australia. The Pickup model is based on the interaction of nested patterns of source, transfer and sink zones. Source zones are areas of high energy gradients that are erosive, transfer zones are intermediate and sediments progress across them in time, while sink zones are of low relative relief and nutrients and sediments tend to accumulate within them.

Erosion cells may be identified from catchment scales to catenary, or finer, scales. In my study region, the pallid breakaway footslopes are catchment level source zones. Their fragile, deeply weathered saprolitic material is easily eroded naturally, and more so when disturbed by livestock trampling. Downslope, the extensive, almost level mulga sheetflood plains are transitional plains, in which sediments are both deposited and gradually transported further downslope as the catchment works to minimise its energy gradient. Finally, the mulga plains drain into palaeodrainage systems, the sink zones where sediments

accumulate. Occasionally, after extreme rainfalls, the palaeodrainage systems flow in a tributary manner to Lake Boonderoo on the edge of the Nullarbor Plain (a pattern representing a higher level of integrated erosion cell). My study catchments themselves are nested in the western salinaland system (Jennings and Mabbutt 1982; Pringle 1994c).

At a finer scale, erosion cells can be seen in upland areas supporting chenopod shrublands, particularly breakaway footslopes. There are often terracettes, which may be less than a centimetre high, downslope of which is a zone in which sediments are being transported downslope. Below that is a depositional zone, where sediments come to rest and microbiotic crusts start to colonise them. At the same time as this colonisation process is occurring, a terracette is etching away at the crusted, stable surface further downslope.

The erosion cell model is a dynamic one, in which nested scales of cells migrate upslope, reducing hydraulic energy and working towards a low-energy, equilibrium state, until the next disturbance occurs, new elevated hydraulic gradients are developed, and the process begins again. Grazing can accelerate and increase the size of erosion cells by accelerating rates of soil erosion out of the catchment as a result of preferential grazing of low-lying, resource-enriched pastures (Pickup 1985; Fanning 1994). This is particularly relevant in exoreic (outward flowing) catchments, where base-line terrain can be lowered.

In my study area, fluvial processes are strong only in scattered uplands and their major drainage systems. However, even where gradients are low and fluvial processes are weak, there is distinct patterning in vegetation suggesting that resource redistribution driven by fluvial processes still occurs (Mabbutt 1963a; Pringle 1994c). Despite the occurrence of fluvial patterns, aeolian processes may dominate in many landscapes and the fluvial-based models may need modification in such circumstances.

On the Nullarbor Plain of Western Australia, for example, sediments have been moved upslope due to grazing-induced loss of shrub cover, from preferentially grazed claypans to adjacent ridges (Gillieson, Wallbrink and Cochrane 1996). This work used fallout radionuclides, to provide unequivocal evidence that sediments had moved up onto the ridges. Thus I decided to integrate the CSIRO model of Landscape Function Analysis (Ludwig *et al.* 1997 ; Ludwig and Tongway 2000; Tongway and Hindley 2000) with sediment tracer techniques employing fallout radionuclides in order to gain a comprehensive view of how grazing affected landscape patterns and processes in my study area.

7.3.3.2 Assessing landscape organisation at my study sites

The models of rangeland landscape dynamics discussed form the basis of a method of assessment termed Landscape Function Analysis or LFA (Ludwig, Tongway and Marsden 1994; Tongway 1994; 1995; Tongway and Hindley 1995). Following the LFA approach, I used transects to assess patterning within landscapes, from which processes of resource redistribution and conservation are inferred, based on obstructions to flows. Patch types were identified (e.g. mulga groves and inter-groves) and designated 'source' (runoff) zones, 'sink' (run-on) zones, or 'interception' (transitional) from visual interpretation of patterns of flow and sediment movement.

I also used measurements of Caesium-137, a radionuclide fallout, to assess resource redistribution more directly. The two approaches were integrated in a complementary manner. Firstly, patterning was assessed using transects and then Caesium -137 within patches, which indicates how water and sediments are moved within the landscape (Ritchie and McHenry 1990; Gillieson, Wallbrink and Cochrane 1996).

My use of transects to assess landscape patterning followed standard techniques described in the Methods section. However radionuclides have not been widely used in rangelands, so I elaborate on this technique below.

7.3.3.3 The use of Caesium-137 fallout radionuclides

Caesium-137 is part of the fallout from atomic weapons testing. Once released after a nuclear explosion, it circulates globally in the stratosphere (Longmore 1982; Ritchie and McHenry 1990). Return to the earth surface is via precipitation, with amount of fallout closely linked to levels of precipitation. Caesium-137 has a half-life of between 30 and 40 years and reached a peak rate of fallout in Australia in 1964. I have measured the residual Caesium-137 that has been redistributed in my study landscapes since then (Ritchie and McHenry 1990). I have therefore not been able to assess sediment redistribution that took place prior to atomic testing with this technique.

When it reaches the land surface, Caesium-137 quickly and irreversibly attaches to oxide coatings on sediments and organic matter. Subsequent redistribution is therefore predominantly by sediment transport processes; biological and chemical processes of redistribution amongst vegetation occur, but are relatively minor influences on Caesium-137 distribution in the landscape (Ritchie and McHenry 1990). Thus the patterns of Caesium-137 distribution reflect primarily the patterns of movement of sediments to which it has bound since its fallout.

Within any region, cumulative Caesium input can be assumed uniform; particularly in areas where orographic influences on wind and rain are minimal. Coupled with its propensity to rapidly bind to surface sediments, this makes Caesium-137 an excellent tracer of sediment redistribution within landscapes such as those in my study area.

Caesium-137 has been used widely to estimate rates of soil erosion and deposition (Campbell, Elliott and Loughran 1986). This is generally accomplished by comparing Caesium enrichment per unit area (radionuclide activity) against a reference profile sampled in an analogous area representing the best available 'natural' benchmark (Walling and Quine 1990; Gillieson *et al.* 1996). A recent evaluation of this technique includes a concise account of some of the more serious limitations (Harper and Gilkes 1994). I faced problems related particularly to the adequacy of sampling that I could achieve in the face of high costs of collecting, processing and 'counting' soil samples. Another problem was a paucity of reference profiles analogous to the range of soil profiles at grazed sites. Although I recognised that my sampling of reference profiles was not as comprehensive as desirable, I sampled as intensively as my resources allowed. I used the reference profiles principally to confirm that I sampled deeply enough into the soil profile to account for almost all Caesium at my sites. However, I did not use the profiles to calculate soil loss / gain directly because I did not wish to place too much reliance on how accurately the shape of the reference profiles reflected the distribution of Caesium-137 concentration with depth. Although my sampling of reference profiles was similar to that used to successfully calculate sediment budgets on the Nullarbor Plain to the south-east of my study region, this is a much more uniform environment (Gillieson, Cochrane and Murray 1994; Gillieson *et al.* 1996).

Because the environments in my study area are more heterogeneous, I used a more conservative approach, using the compartmentalisation of Caesium-137 within landscape strata to investigate the impact of grazing on landscape function, rather than on absolute sediment budgets. This approach is sufficient to provide an independent assessment of the CSIRO paradigm of Landscape Function Analysis: that landscapes are highly patterned and that this pattern reflects the capacity of the system to maintain fertile patches. Previous assessments have used nutrient distributions, particularly nitrogen. These are indicative rather than conclusive, as the nutrients might have been fixed *in situ* or concentrated by root uptake from afar and subsequent *in situ* litter-fall. Caesium overcomes this problem because it is so quickly and tightly bound to sediments.

7.3.4 Applicability of models to this study region

7.3.4.1 The region as a whole

I refer now to the models of grazing impact on landscape patterning discussed previously (see Figure 7.1), but with some integration of processes operating within my study region. The first model, involving species changes without noticeable change in landscape grain appears to best apply to chenopod shrublands in low energy, depositional landscapes. It might also apply to grasslands on deep, well-drained sandy soils, though encroachment of shrubs is also frequent in such landscapes (Pringle 1994b).

The model involving the development of fertile shrubby islands was invoked occasionally in the rangeland survey of the study area, in areas receiving concentrated run-on and on the margins of flood out plains flanking major creeks, where chenopod steppe had been replaced by needlebush (*Hakea preissii*) and *Senna* (cassia) thickets (Pringle 1994b).

Grazing-induced change in landscape grain resulting in stable productive herblands was described rarely in the regional survey (Pringle 1994b). Stable herblands were occasionally seen in upland 'greenstone' chenopod plains that had heavy stone mantles protecting soils from erosion after perennial shrubs had been removed. In similar areas lacking a protective stone mantle, removal of vegetation usually leads to accelerated sheet erosion in the form of migrating terracettes (Pickup 1985). Pastoralists have stated that loss of palatable shrubs in such areas leads to a decline in the pastoral value of the land (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Pringle and Riches 1996). However it is not known what effect the transition from shrubland to herbland has on biological productivity *per se*.

Denudation of perennial vegetation (mostly chenopod shrublands) more frequently leads to accelerated soil erosion, secondary salinity and exposure of subsoil pans. This was described as a particularly common occurrence in broad flood out plains flanking major creeks and in depositional fans adjacent to major greenstone hill systems (Pringle 1994b).

Thus all of the degradation models appear to apply to different types of landscapes and under particular conditions in my study region. They are not necessarily mutually exclusive models, but do have contrasting distinctive characteristics. Different models or combinations of models are likely to be necessary for describing responses to heavy grazing in different types of landscapes in my study region.

7.3.4.2 My study landscapes

I did not sample within 200m of watering points, which is the zone of extreme trampling (Valentine 1947; Andrew and Lange 1986a). Grazing impacts on landscape pattern beyond this zone are more likely to highlight contrasts among landscapes, in contrast to the severely trampled zone, where impacts tend to converge.

In chenopod plains such as those in my study landscapes, grazing usually leads to some loss of species richness and projected foliar cover, with a shift in species dominance to more grazing-tolerant shrub species. Rarely does this lead to complete denudation of perennial vegetation (Hacker 1979; Pringle 1994b). I would therefore expect the model of little change in landscape grain to apply at most of my sites in chenopod landscapes, though with some expansion of inter-shrub spaces.

In groved mulga landscapes, I would expect grazing to affect the competitive balance between sparse palatable shrubs and more grazing-tolerant shrubs with a propensity to proliferate, at least in the understorey of groves (Hacker 1984a). I would expect a similar pattern in mulga drainage lanes of sandy mulga landscapes, but to a lesser extent, as the focal sites for understorey establishment are smaller and more scattered.

On sandy banks in sandy mulga landscapes, I would expect a reduction in palatable perennial grass cover to be followed by either shrub encroachment or a proliferation of unpalatable C_4 grasses, depending on seasonal conditions. Should normal seasons (mostly winter effective rainfall) prevail, shrub encroachment would be expected (Pringle 1994a), but should a sequence of good summer rains occur, the unpalatable C_4 grasses *Eragrostis lanipes* and *Eriachne helmsii* might proliferate. Given the regional climate and the capacity for shrubs to concentrate nutrients within their above-ground biomass and litter zone, one might suspect the long-term trend to favour shrub-dominance as nutrients in inter-shrub zones are gradually sequestered by shrubs (Burrows 1972; Reynolds *et al.* 1999).

These then are the conceptual models against which I assessed the effect of grazing on patterns and processes in my study landscapes.

7.4 Methods

7.4.1 Assessing patterns within landscapes: LFA

I used aerial photography and recent information from a biophysical survey of the region (Pringle, Van Vreeswyk and Gilligan 1994) to identify and delineate landscape strata apparent along my gradients. This is described more completely in Chapter 2, where the landscape strata studied are described and represented diagrammatically.

In order to assess how the landscape strata were arranged, I adopted the transect component of the Landscape Function Analysis approach (Ludwig and Tongway 1995b; Tongway and Hindley 1995; Ludwig *et al.* 1997). I did not include any features that occupied less than 5% of the area of most sites within any landscape type. For instance, logs were ignored and scalds were amalgamated with micro topographically similar zones in which they occurred.

In the mulga landscapes, I ran single transects downslope for at least 100m to include a minimum of three recurrences of the pattern of landscape strata. This equated to grove/inter-grove sequences in groved mulga landscapes and clump/inter clump sequences in mulga drainage lanes of sandy mulga landscapes. In mulga drainage lanes, transects were aligned along the central axes of sampling loci, whereas in groved mulga landscapes they were perpendicular to the central axes of sampling loci (see Figures 2.7 and 2.8 in Chapter 2). I recorded boundaries between open or clumped patches; with clumps being differentiated by canopy cover (Slatyer 1962). I also recorded the position of obstructions to flow (e.g. logs). On the sandy banks within sandy mulga landscapes, I ran transects along the central axes of sampling loci for 25 metres and recorded grass sward cover obstructions to flow. I did not analyse the sward cover data as grass cover was extraordinarily high following a run of good seasons, stock were not using watering points frequently and preliminary analysis of cover assessed within quadrats for soil surface assessment showed no systematic variation with distance to water.

In the chenopod landscapes I ran 25m transects down the central axis of sampling loci. Sandy banks were prominent across gradients 1CH and 5CH and I kept transects at a constant distance from them (see Figure 2.6 at the end of Chapter 2). I recorded the positions of bush mounds and shrubs, but analysed only the shrub data on the basis that there was little difference between the two (usually only a few centimetres either way). While shrub dimensions may exceed those of bush mounds, the opposite may also apply under very heavy grazing of shrubs and/or as a shrub senesces. However, no such trends were obvious in this study.

7.4.2 Assessing redistribution within landscapes: Caesium-137

The aim of these measures was to determine how much soil has moved between the various landscape strata at sites subject to grazing (since atomic testing commenced in the 1950s). I adapted a sampling procedure successfully used on the Nullarbor Plain to produce sediment budgets and evaluate the impact of grazing on patterns of sediment redistribution (Gillieson *et al.* 1996; Gillieson, Wallbrink and Cochrane 1996). The approach involved sampling reference soil profiles and soil cores at grazed sites, which I processed in Canberra at CSIRO Land and Water laboratories.

7.4.2.1 Reference profiles

I sampled each of the three major soil types encountered in my study at a representative reference site. The clay loam and sandy soil profiles were sampled at site 4 of Gradient 6SM and the saline texture contrast soil profile was sampled at site 5 of Gradient 7CH. The sites were selected on the basis of their ease of vehicular access. This was necessary because the substantial amount of soil sampled required transport back to the laboratory.

From three representative excavations at each site, I took 20 cm x 20 cm sample slices at every centimetre to a depth of 20 cm. Each slice was weighed and bagged separately. At the reference site of Gradient 6SM (the mulga drainage lane and sandy bank reference profile site), I sampled representative open areas, avoiding mulga trees. At site 5 of Gradient 1CH (the chenopod reference profile site) I sampled inter-bush areas. Replicate samples were then thoroughly mixed and sub-sampled at the WMC Resources Environment Centre in Leinster before being air-dried and transported for laboratory processing and analysis in the laboratory at CSIRO Land and Water in Canberra.

7.4.2.2 Site cores

Again following the Nullarbor Plain sampling protocol (Gillieson *et al.* 1996; Gillieson, Wallbrink and Cochrane 1996), I used steel tubes of 54mm diameter and 15cm depth to sample soil cores along the central axis at each sampling locus (see figures at the end of Chapter 2). Three cores at 5m intervals were collected and weighed at each of the five sampling loci at each site. The 15 samples were then thoroughly mixed and sub-sampled for processing and analysis in Canberra.

7.4.2.3 Laboratory processing and analysis

Samples were dried at 105°C for 60 hours, weighed and then placed in a furnace for 40 hours at 400°C. They were then made into fibreglass casts for counting of radionuclides. Counting took 48hours. I used the count data of Caesium enrichment (in Bq/kg of soil) and

the area sampled to produce a measure of enrichment per unit area (Bq/m^2) for each landscape stratum (e.g. bush mound) at each site.

Statistical modelling of these values was conducted using standard REML procedures described in section 2.5 of Chapter 2.

7.5 Results

7.5.1 Landscape patterns: LFA

7.5.1.1 Landscape differences

The LFA methodology refers to the distance between obstructions as 'intercept lengths', but to avoid confusion with statistically modelled 'intercept effects', I will call average intercept lengths 'width'. Woody landscape strata (groves or bush mounds) occupied minor proportions of sites in both landscape types. Mulga groves occupied approximately one-third of groved mulga landscapes (33.1 per cent +/- 0.09), while bush mounds occupied less than one-sixth of chenopod landscapes (14.8 per cent +/- 0.09).

The average dimensions of the landscape strata also varied markedly. Mulga groves were around 4 metres wide (4.35 +/- 1.36m), while intervening inter-groves were over 7 metres wide (7.44 +/- 1.95m). Bush mounds were much smaller (0.2 +/- 0.1m), while inter-bush areas were 2.9 +/- 4.5m. The discrepancy in widths and proportional cover of woody patches indicates that groved mulga landscapes are characterised by few, large patches, while the opposite applies in chenopod landscapes. Chenopod landscapes are also characterised by variability in the average size of inter-bush areas (open patches).

7.5.1.2 Grazing effects

Increased grazing activity led to declines in the proportional area occupied by bush mounds in chenopod landscapes and mulga groves in groved mulga landscapes, for all models tested (Figure 7.2, Table 7.1).

Although the rate of decline was the same (no interaction effect), there were strong differences in the average patch between the two landscapes (Figure 7.3, Table 7.2). Two significant effects dominate the three different models of grazing history: the differences in patch widths among landscape strata, and the different influences of grazing upon them. The most striking grazing effect is the increase in the width of inter-bush (open) patches. This indicates that the very high variability in this stratum (mean width 2.9 m, standard error 4.5 m) may be due largely to grazing impact.

While the proportional area and widths of mulga groves appear to be affected by grazing, some caution is warranted in interpreting these data. I had over 70 data points for chenopod landscape strata, but only 8 points for groved mulga strata. The history of use of mulga landscapes in my study region severely constrained my choice of sites within them. My primary interest in collecting these data was to describe the internal heterogeneity *per se* in groved mulga landscapes, rather than to explore the impacts of grazing upon them (see Chapter 2).

7.5.2 Redistribution processes: Caesium-137

7.5.2.1 Differences between landscapes

The data from the reference profiles indicate that the depth of core sampling was sufficient to account for virtually all of the Caesium within the soil profile (Figure 7.4). There is a rapid decline in the Caesium content of the soil with depth down the soil profile in all three cases. A high proportion of the Caesium-137 was contained in the top 3-4 cm and there was no sign of accumulation lower in any of the soil profiles. Thus at 15 cm depth, my site cores were likely to capture all the Caesium-137 within limits of detection. The concentration of Caesium-137 near the surface makes this approach sensitive to losses or accumulations of topsoil.

The two woody landscape strata (mulga groves and bush mounds) had the highest concentrations of Caesium-137 (Figure 7.5). Furthermore, they were more enriched than their respective inter-grove (not statistically significant) and inter-mound (significant) strata. Differences between sandy banks and mulga drainage lanes were not significant in the sandy mulga landscape, though the average enrichment was higher in the mulga drainage lane stratum.

The most striking difference was the much greater degree of contrast in Caesium-137 enrichment within the finely patterned chenopod landscape than in either of the mulga landscapes: bush mounds and inter-bush areas in chenopod landscapes had average Caesium enrichment of 414 and 161 Bq/m^2, while mulga groves and inter-groves had 274 and 217 Bq/m^2 (least significant difference of 90 Bq/m$^{2)}$.

7.5.2.2 Grazing effects

The woody strata (bush mounds and mulga groves) had declining Caesium-137 enrichment with increasing levels of grazing history, while the opposite was the case for their open, more extensive partnering strata. Sandy banks showed the greatest rate of decline in Caesium-137 enrichment with increased grazing history, consistent with having less material with which to bind aggregates and hence having more labile sediments.

In all three landscapes, the smaller stratum lost sediments with increased grazing history while its counterpart appeared to experience very slight deposition (Figure 7.5, Table 7.3). When the net Caesium-137 for each site was estimated by weighting the stratum means by their proportional area, grazing had no significant effect on net Caesium-137 in any landscape. This indicates that one stratum's loss is the other's gain. Fitted lines tend to converge. For instance, bush mounds lose sediments (Caesium-137) as they break down and their proportional area also declines. However there is a concomitant increase in the proportional area of inter-bushes, where Caesium-137 enrichment increases slightly (although this is not statistically significant). Thus there is no evidence to suggest that sediments have been lost from the landscape due to grazing, at least since atomic weapons testing commenced in the 1950s. This is consistent with results from the Nullarbor Plain, where grazing led to sediment redistribution within landscapes without any net loss (Gillieson *et al.* 1996).

There were obvious signs of historic soil erosion (as observed by Alan Payne during range condition assessment of study sites, Chapter 4), including remnant bush mounds and salt scalds at sites closest to water at Gradients 1CH and 5CH. However, these areas appeared to have stabilised and become sink zones. Both the paddocks in which the gradients were located were developed in the early part of the twentieth century, decades before atomic weapons testing (and Caesium-137 fallout).

7.6 Discussion

7.6.1 Landscape patterning

The strong internal organisation of the landscapes evident from the landscape transects is consistent with that described in the recent rangeland survey of the region (Pringle, Van Vreeswyk and Gilligan 1994). The grain of internal heterogeneity is fine in chenopod landscapes (where aeolian processes dominate) and much coarser ('meso-scale') in the groved mulga landscapes, where fluvial processes dominate. This matching of grain and dominant processes is consistent with the findings of Tongway and Ludwig (1994). Similarly strong landscape patterning has been reported in rangelands around the World (Belsky 1989; Cornet *et al.* 1992; Montana 1992; Bergkamp 1995). The evidence overwhelmingly supports Noy-Meir's view that these arid landscapes tend to consist of large source areas and smaller sink areas, rather than being uniformly patterned.

In the chenopod landscapes, grazing history has strongly reduced the size and proportional area covered by woody patches and has led to expansion of open areas. Similar results have been reported in other parts of the study region (Hacker 1979; Pringle 1994b; 1998b) and in regions nearby (Curry *et al.* 1994; Gillieson, Cochrane and Murray 1994).

These changes in patterning within chenopod landscapes suggest a progressive denudation of perennial vegetation (shrubs), but whether this leads to a stable productive herbland (model 4A in Figure 7.1) or a major loss of productivity (4B) will be assessed in the following chapter on topsoil properties.

Although my results show a similarly significant influence of grazing on mulga groves, these findings refer to so few data points they should be treated very cautiously. There is insufficient evidence to suggest that a change in grain occurs, so a model of succession within vegetation types (1A in Figure 7.1) may be most applicable. This is consistent with the findings of Hacker (1979; 1984a), that major changes in mulga woodlands are likely to occur in the understorey of mulga trees and the clumps they form. That is, the long-lived tree patterning provides the spatial structure beneath which more rapid changes in vegetation composition occur.

7.6.2 Redistribution within landscapes

7.6.2.1 Differences between landscapes

The contrasts in Caesium-137 indicate that resource redistribution is strongest and most tightly coupled in the chenopod landscapes, with their fine grain size and aeolian modulation. Mulga groves had a higher (though not significant) average level of Caesium-137 enrichment compared with inter-groves. In the sandy mulga landscape, Caesium-137 enrichment was quite similar in sandy banks and mulga drainage lanes. This is consistent with the notion that sediment transfer between strata in sandy mulga landscapes is only loosely coupled because their boundaries are very permeable.

These data provide convincing, independent evidence that patterns of landscape heterogeneity reflect the capacity of these landscapes to redistribute scarce resources. In particular, they indicate marked concentration of resources in woody ‘islands’ that occupy small fractions of the total landscape area.

The grain of patterning and strength of contrast in Caesium-137 are entirely consistent with my predicted models of vegetation-landscape interaction. In the chenopod landscapes, wind dominates sediment transport and obstructions to wind (shrubs) are near the ground level.

Hence resource patches form at fine scales below individual shrubs. Conversely, in the groved mulga landscapes, gentle (low hydraulic energy) processes dominate and there is only minor vegetation-modulated resistance to flows. Therefore, patterns tend to evolve that reflect subtle variations in terrain topography and contrasts in Caesium-137 enrichment are far less profound than in chenopod landscapes. However, patterning in the groved mulga landscapes of the study region is far simpler than in similar banded vegetation studied elsewhere (Mabbutt and Fanning 1987; Tongway and Ludwig 1990; Cornet *et al.* 1992; Bergkamp 1995). In these other regions topographic gradients are stronger and there are often clear zonations of different vegetation running parallel to the band (and converse to flow); thus modulation of landscape processes that influence spatial and temporal dynamics of these banded landscapes may be far stronger.

7.6.2.2 Grazing effects

As grazing history increased in the chenopod and groved mulga landscapes, there was a consistent pattern of declining Caesium-137 enrichment in woody patches and a lesser increase in adjacent open spaces, with no significant net changes. The apparent retention and redistribution of Caesium in the grazed endoreic (internally draining) landscapes of my study region are consistent with findings on the nearby Nullarbor Plain (Gillieson *et al.* 1996; Gillieson, Wallbrink and Cochrane 1996). It is also entirely consistent with a model of increased leakage of resources from fertile patches to adjacent run-off/open patches under heavy grazing (Tongway and Ludwig 1990; 1994; Ludwig, Tongway and Marsden 1994; Ludwig and Tongway 1995b; Ludwig *et al.* 1997).

These grazing impacts on redistribution within landscapes are further evidence that models of degradation 4A or 4b (Figure 7.1) might apply in the chenopod landscapes. The changes do not conform to the pattern of re-organisation into coarser-grained 'fertile islands' as has been observed in grasslands overseas (Archer *et al.* 1988; Brown and Archer 1989; Schlesinger *et al.* 1990; Archer 1990; 1994; 1995; Whitford, Martinez-Turanzas and Martinez-Mesa 1995; Eve, Whitford and Havstad 1999). Rather, there seems to be denudation of woody patches as physical factors gain control of landscape pattern and processes. However, this 'smearing' of the landscape need not represent desertification (irreversible loss of biological productivity). Indeed the overall apparent retention of Caesium-137 within the landscapes indicates that the stable, productive herbland model (Model 4B) (Wilson and MacLeod 1991; Holm 1994) might apply across much of the historically heavily areas of these landscapes. However, soil loss that may have occurred pre-atomic testing was apparent at sites closest to watering points at two of three gradients in chenopod landscapes.

In sandy mulga landscapes, Caesium-137 enrichment declined strongly with increased grazing history on sandy banks and increased in adjacent mulga drainage lanes. These losses and gains within a roughly balanced landscape budget are again consistent with local redistribution. Perennial grasses on sandy soils are prone to removal under heavy grazing within this region (Hacker 1979; 1984a; Pringle 1994b). Unless infestation by poverty bushes (e.g. *Eremophila gilesii, E. margarethae* and *E. spectabilis*) occurs, fine sediments (with bound Caesium-137) in surface soil would be prone to winnowing away by occasional strong summer winds. The taller, denser vegetation in adjacent mulga drainage lanes would trap many of these fines. The process of entrapment would most likely involve physical contact of sediments with vegetation and vegetation-modulated reduction in wind energy and subsequent deposition (Marshall, Hirvonen and Wiken 1993). The effectively entrapped Caesium-137 in mulga drainage lanes might be partly due to increased development of understorey vegetation in the form of poverty bushes (*Eremophila spectabilis* and *E. gilesii*) at the most heavily grazed sites (see Chapter 5). These findings, and those of Hacker (1986) regarding reduced sorptivity of soils subjected to heavy grazing provide preliminary evidence that sandy banks may trend consistently with Model 4B: i.e. denudation and loss of productivity, under heavy grazing (see Figure 7.1). This is in contrast to the development of very stable shrub-encroached dune systems that replace heavily grazed grasslands in southern New Mexico (Model 3A, Walt Whitford personal communications, personal observations).

7.7 Summary

(1) Landscape patterning and the distribution of Caesium-137, a fallout radionuclide, were investigated to determine patterns of landscape organisation and its interaction with grazing history.

(2) Fine-scale patterns driven by aeolian processes are prominent in chenopod landscapes. The recurrent pattern consists of bush mounds and open, inter-bush areas. Coarser patterning is prominent in the groved mulga landscapes, reflecting the dominance of subtle variations in terrain topography and its interaction with gentle sheet flows. The development of sandy banks and mulga drainage lanes in sandy mulga landscapes reflects the influence of Quaternary landscape processes on sediments and drainage patterns super-imposed on geologically old, bevelled palaeocatchments.

(3) Resource redistribution indicated by Caesium-137 is considerably stronger, though at finer-scales, in chenopod landscapes compared to groved mulga landscapes. This reflects the interaction of different driving processes of sediment transport and vegetation patterning (capacity to entrap and retain resources). Differences between sandy mulga strata were negligible.

(4) In chenopod and groved mulga landscapes, the proportion of landscapes occupied by woody and presumably fertile patches (bush mounds and mulga groves) declined with increasing grazing history. The size of open inter-bush areas increased markedly with increased grazing history.

(5) In these strongly patterned landscapes the woody patches had decreasing Caesium-137 enrichment with increasing grazing history. This is consistent with a model of declining capacity of these patches to entrap and retain resources and a general homogenisation - 'smearing' - of the landscape at that level of organisation.

(6) In the less strongly patterned sandy mulga landscapes, sandy banks may lose sediments to adjacent mulga drainage lanes by wind erosion in areas that have received historically heavy grazing.

(7) All three landscapes appear to behave in a manner consistent with a model of landscape homogenisation (but to quite different degrees), rather than a model of 'heterogenisation' into coarser-scaled fertile islands.

Table 7.1 Effects of grazing history on proportional areas of woody patches

Grazing model	Model terms	Wald statistic	d.f.	p-value
SH: Stocking history, derived from modelling track densities and expressed in sheep years per hectare	SH	13.1	1	< 0.001
	stratum.type	11.1	1	< 0.001
	stratum.type.SH	0	1	n.s.
SR: Stocking rate, average long term stocking rate developed from RANGEPACK/Paddock model and expressed in sheep per hectare	SR	33.6	1	< 0.001
	stratum.type	23.0	1	< 0.001
	stratum.type.SR	0.7	1	n.s.
DW: Distance from water, measured in kilometres	Dist	9.6	1	0.002
	stratum.type	13.1	1	< 0.001
	stratum.type.Dist	3.3	1	n.s.

Note: Because only two landscape strata were analysed, two model terms have been omitted ('landscape type' and 'landscape type'.grazing estimate)

Table 7.2 Effects of grazing history on lengths of transect intercepts

Grazing model (estimate of grazing history)	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	15.1	1	< 0.001
	SH	17.3	1	< 0.001
	stratum.landscape type	111.0	2	< 0.001
	landscape type.SH	0.2	1	n.s.
	stratum.landscape type.SH	214.0	2	< 0.001
SR: Stocking rate (sheep/ha)	landscape type	13.7	1	< 0.001
	SR	6.1	1	0.01
	stratum.landscape type	73.8	2	< 0.001
	landscape type.SR	0.1	1	n.s.
	stratum.landscape type.SR	75.7	2	< 0.001
DW: Distance from water (km)	landscape type	9.8	1	0.002
	DW	2.4	1	n.s.
	stratum.landscape type	58.8	2	< 0.001
	landscape type.DW	0.0	1	n.s.
	stratum.landscape type.DW	19.9	2	< 0.001

Table 7.3 Effects of grazing history on Caesium-137 enrichment

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history (sheep years/ha)	landscape type	32.1	1	< 0.001
	SH	0.0	1	n.s.
	stratum.landscape type	67.5	4	< 0.001
	landscape type.SH	.3	1	n.s.
	stratum.landscape type.SH	92.2	4	< 0.001
SR: Stocking rate (sheep/ha)	landscape type	13.2	1	< 0.001
	SR	0.0	1	n.s.
	stratum.landscape type	27.4	4	< 0.001
	landscape type.SR	0.5	1	n.s.
	stratum.landscape type.SR	25.2	4	< 0.001
DW: Distance from water (km)	landscape type	7.5	1	0.006
	DW	0.0	1	n.s.
	stratum.landscape type	17.7	4	0.001
	landscape type.DW	0.7	1	n.s.
	stratum.landscape type.DW	8.3	4	n.s.

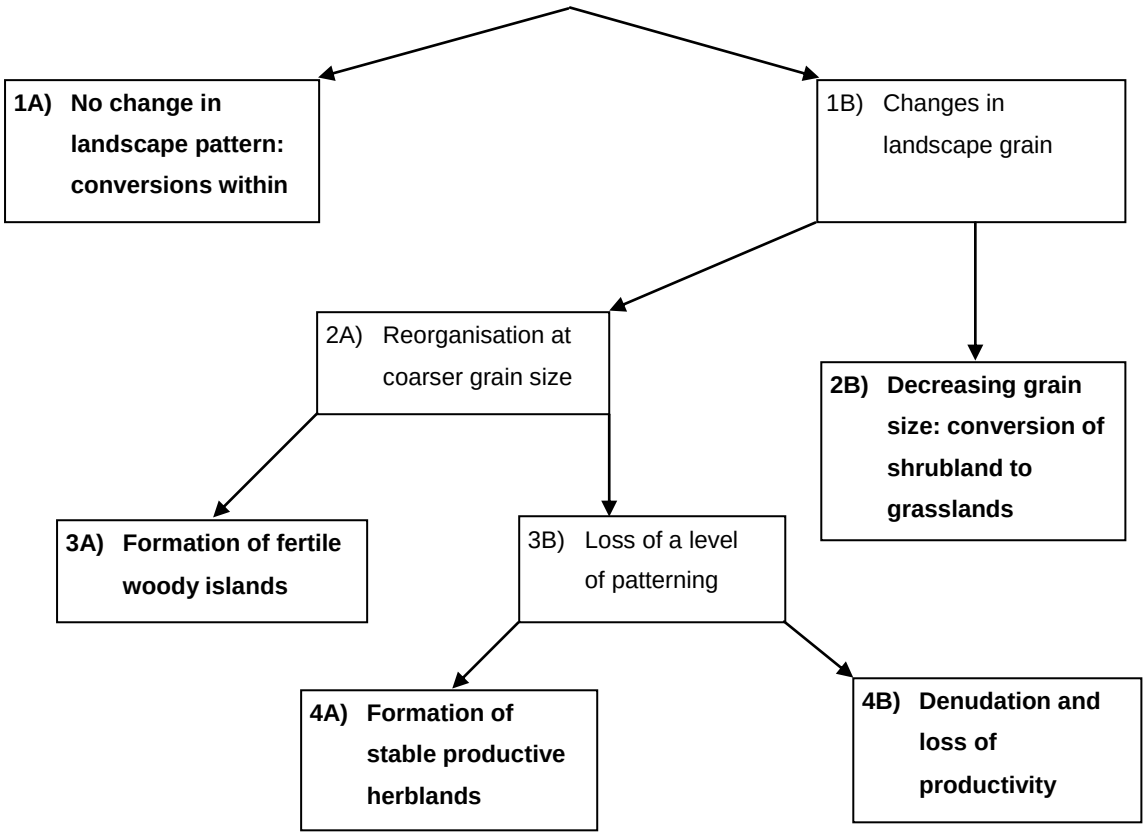


Figure 7.1 A classification of degradation models.

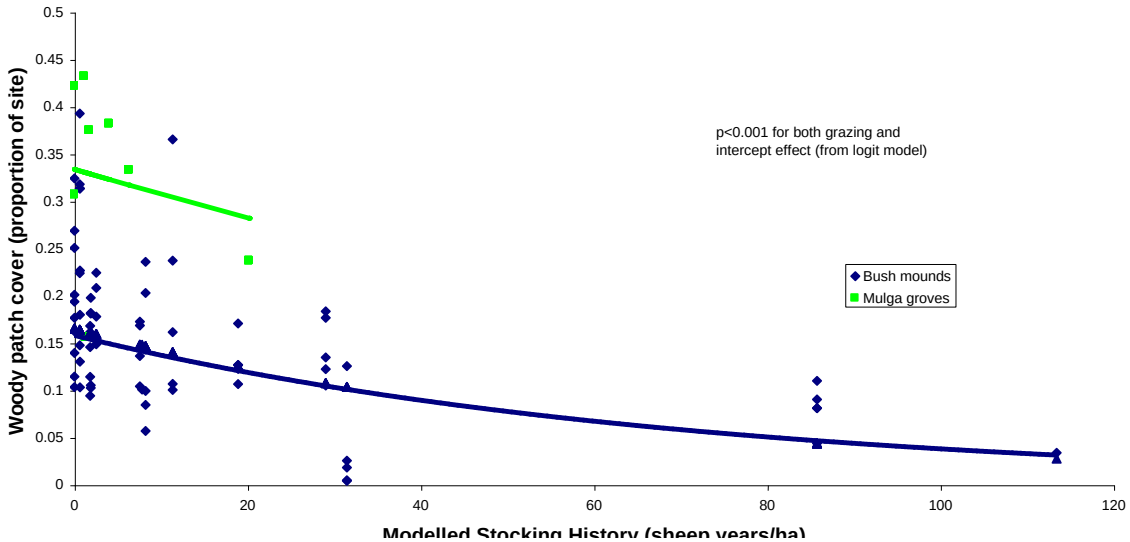


Figure 7.2 Impacts of stocking history on areas of woody patches (with fitted trend-lines).

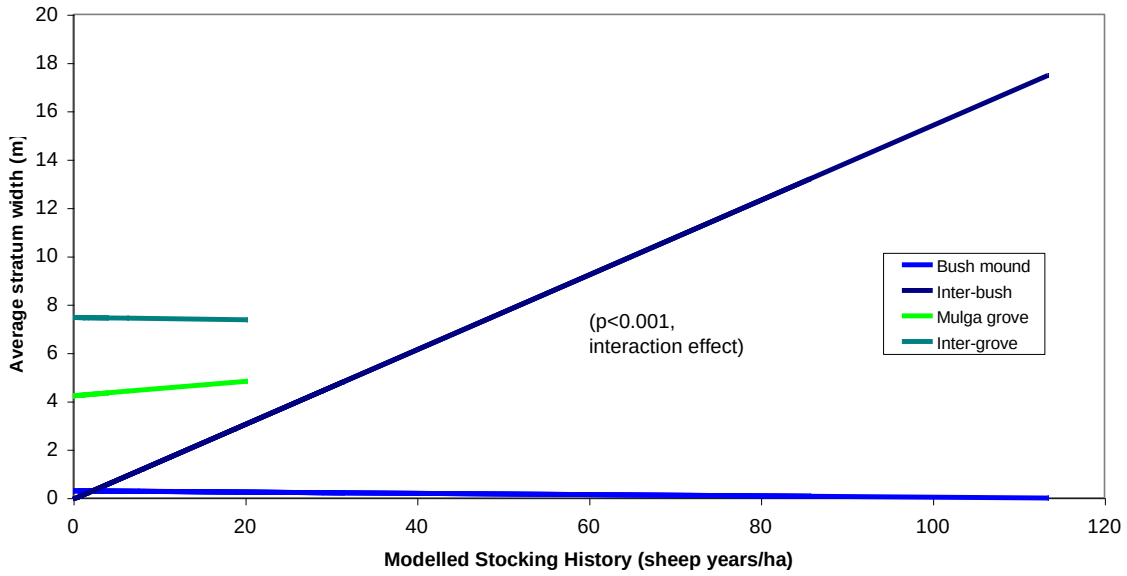


Figure 7.3 Impacts of stocking history on sizes of landscape strata.

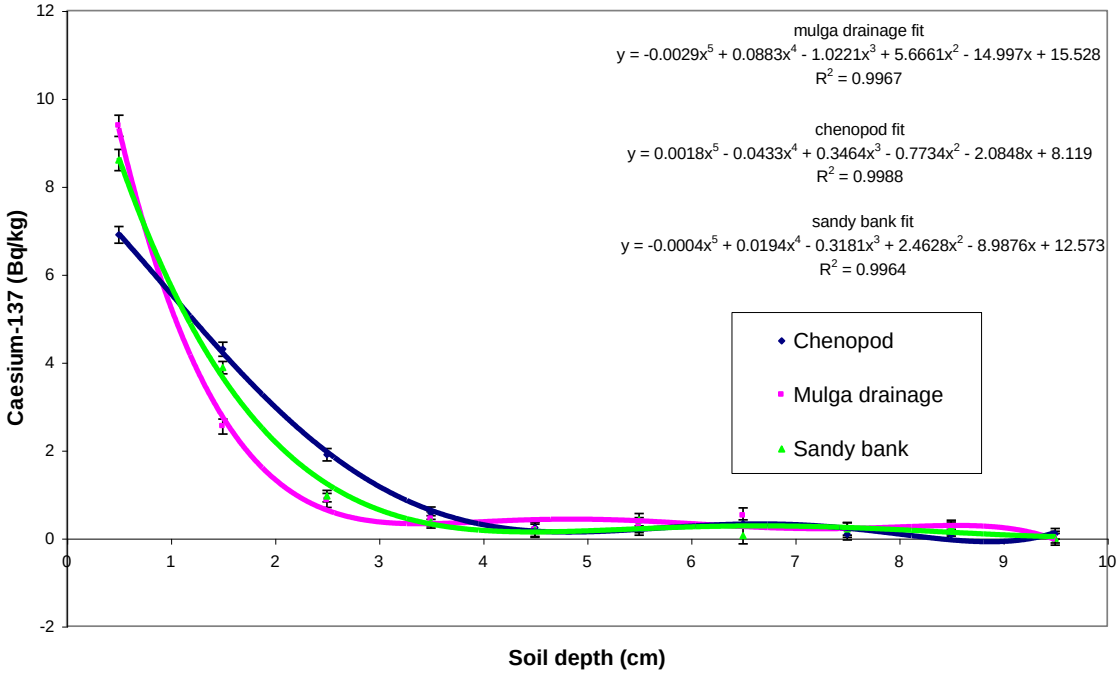


Figure 7.4 Effects of soil depth on Caesium-137 enrichment.

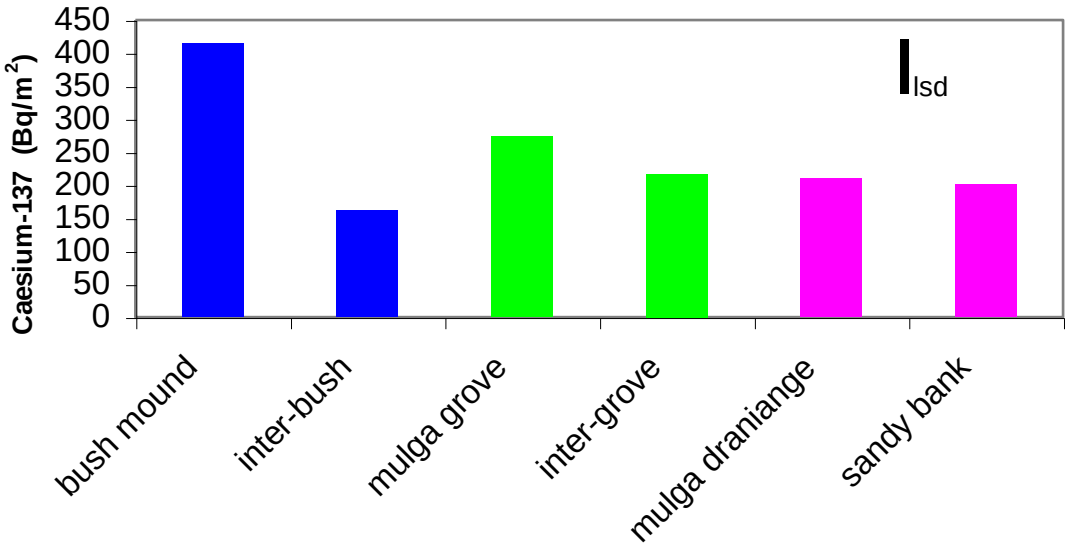


Figure 7.5 Variation in Caesium-137 enrichment with landscape strata.

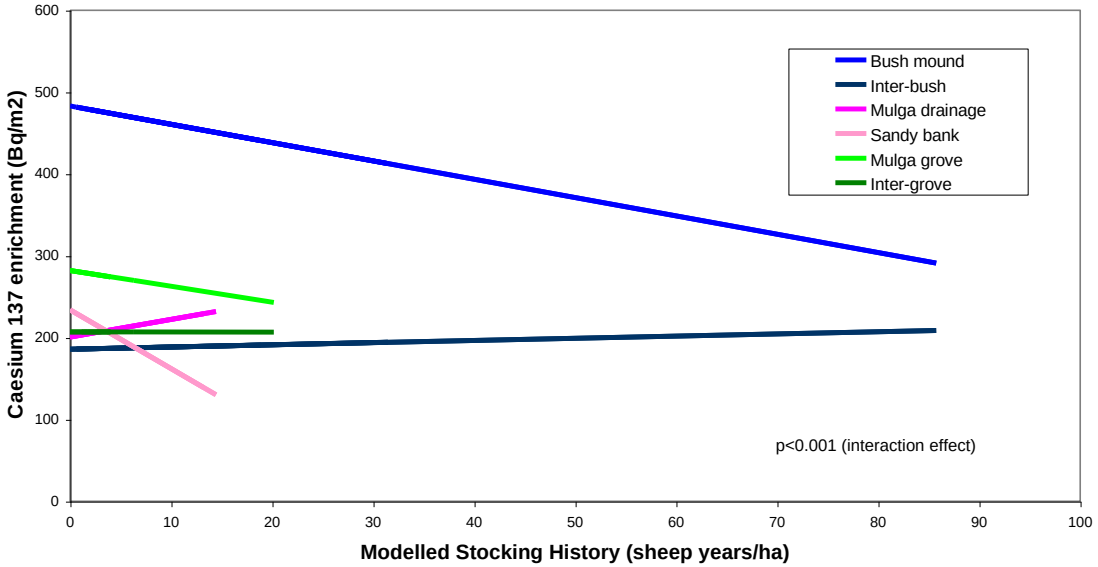


Figure 7.6 Impacts of stocking history on Caesium-137 enrichment.

CHAPTER 8. PROPERTIES OF TOPSOILS WITHIN LANDSCAPES

8.1 Scope of this chapter

This is the second chapter considering the impacts of grazing on the integrity (similarity to least disturbed conditions) of ecosystems in my study region. In the previous chapter I described landscape organisation and how it was affected by grazing. I now look within the landscape compartments to investigate patterning within them and how this is affected by grazing.

My emphasis is on the impacts of grazing history on topsoil properties. Generally, the impacts of grazing on subsoils are minimal (Charley and Cowling 1968; Hacker 1986) and therefore are not discussed. I commence with a brief overview of the importance of topsoil, with an emphasis on Australia's arid zone, followed by a brief account of the topsoil properties I have selected to investigate. I include a brief review of previous research on these soil properties and their susceptibility to change under grazing management, using examples from Australia and overseas. I then describe my sampling strategy and the techniques used to analyse the soil samples and subsequent data. Next I present results for each topsoil property, followed by a discussion in terms of the existing literature and my specific research objectives.

I conclude with a brief section bringing together the findings from this chapter within the framework provided by Chapter 7 on landscape organisation. My objectives for this chapter can be expressed in the following questions:

(1) Which of topsoil properties are affected by grazing?

(2) Are the effects of grazing history on soil properties consistent within and among landscape types?

(3) How does the effect of grazing history vary among soil properties?

8.2 An overview of topsoils in arid Australia

8.2.1 The ecological importance of topsoil

Nutrients and biological processes are disproportionately concentrated in topsoils in arid zones (and elsewhere). This is particularly so over most of the Australian arid zone, where soils are geologically ancient and leached (Mulcahy 1960; Beckmann 1983) and nutrients are highly concentrated in the top few centimetres of soil profiles (Charley and Cowling 1968; Whitford 1986a). My study region is no different in these respects (Churchward 1977;

Tongway 1994; Van Vreeswyk 1994; Pringle 1994c). I use the term 'topsoil' to refer to the top 2cm of the soil profile (which is what I sampled).

One might envisage arid topsoils as habitats for episodic, but vigorous biological activity in otherwise largely inert sediment profiles. While biological activity extends below topsoils, biopedturbation is not a major characteristic of arid Australian soil profiles (Churchward 1977). That is not to say that biopedturbation does not have strong impacts on soils locally (e.g. termitaria, bettong mounds and rabbit warrens).

Topsoil is the substrate that supports germination of plants and the growth of sometimes spectacular assemblages of ephemeral 'wildflowers'. Perennial plants' fine root systems also take up nutrients in the topsoil. Topsoil is also habitat for the microbiota that play important roles in ecological processes such as detritivory and nutrient cycling (Greenslade and Greenslade 1984; Solange, MacKay and Whitford 1985; Whitford 1986b; Moorhead and Reynolds 1989; Andersen 1990; Majer and Beeston 1996; Kay, Sobhy and Whitford 1999; Whitford and Sobhy 1999). Therefore, topsoil is critically important as the substrate upon which much biodiversity depends and in which key ecosystem functions occur in my study region.

Topsoil is also intimately involved in spatial processes of redistribution of resources (e.g. water, sediments and nutrients) into 'environmental resource patches' (*sensu* Forman and Godron 1981) or 'compartments' (*sensu* Noy-Meir 1985). These patches are critical in arid ecosystem functioning (Slatyer 1975; Noy-Meir 1985; Thurow 1991; Ludwig and Tongway 1995b; Thurow 2000). For instance, differences in topsoil characteristics affect rates of infiltration, facilitating elevated primary production in relatively fertile patches within arid landscapes (Greene 1992; Ludwig and Tongway 1995b).

Thus arid topsoils play critical roles in ecosystem functioning and as habitat for biodiversity. Grazing impacts are considered in terms of their effects on soil chemical, physical and biological properties.

8.2.2 Impacts on soil chemical properties resulting from grazing

Few studies have been conducted on the long-term impacts of grazing on soil nutrient levels and on rates of nutrient cycling. The limited data available are inconclusive (Briske and Heitschmidt 1991). In arid Australia, progress since the groundbreaking work of Charley and Cowling (1968) has been fragmented and minor in comparison (David Tongway personal communication).

Grazing tends to accelerate cycling of nutrients by reducing their above-ground residence time and increasing the rate of mineralisation of organic matter. Mineralisation may be accelerated directly, by herbivory and urination of mineral forms of essential nutrients, or indirectly by defecation of organic, physically decomposed forms (Wilkinson and Lowrey 1973). Trampling can also accelerate decomposition (Manley *et al.* 1995; Schuman *et al.* 1999).

Studies to date indicate that losses from ecosystems due to grazing are generally minor by comparison with other system fluxes (Lauenroth and Milchunas 1991). Furthermore, large proportions of nutrient pools are often inorganic and hence relatively stable, offering some short-term buffering against losses (Woodmansee 1978). However the relative sizes of nutrient pools are likely to vary considerably among types of landscape and in relation to grazing (Archer and Smiens 1991).

There may be some redistribution of nutrients as a result of grazing. This may be between pools (e.g. from plants to soil) (Reynolds *et al.* 1999; Schuman *et al.* 1999) or spatially, between landscape patches (Charley and West 1975; Archer and Smiens 1991; Archer 1994). Such redistribution is not necessarily coincident with accelerated leakage, particularly if new spatial patterns evolve that are as effective in conserving nutrients. In some cases biological productivity may be conserved or enhanced by the formation of 'fertile islands' (Burrows 1972; Archer 1994; Reynolds *et al.* 1999).

Noy-Meir theorised that because arid ecosystems are water-limited, productivity per unit area would be higher with patchy distribution of soil water than it would be if the same amount of soil water was distributed evenly (Noy-Meir 1981; 1985). Tongway and Ludwig (1990) have shown that, in rangeland ecosystems that have retained their functional integrity, nutrient redistribution into fertile patches is coupled with redistribution of water.

Soil erosion is likely to be the major cause of any substantial nutrient loss from arid rangelands because nutrients are concentrated very near the soil surface (Charley and Cowling 1968; Marshall 1970; Fletcher, Sorensen and Porcella 1978; Tongway and Ludwig 1990). My study landscapes have very low topographic gradients, and hence fluvially induced erosion is not likely to be a major effect of grazing. However, wind may erode exposed soils in those landscapes susceptible to reduction in plant cover (grasslands and low shrublands).

8.2.3 Grazing impacts on soil physical properties

The two physical soil processes that are of extreme importance in arid rangelands are soil erosion and infiltration (Johnston 1962; Greene and Tongway 1987; Thurow 1991). Each of these is reviewed below.

8.2.3.1 Soil erosion

Removal of surface sediments is possibly the most damaging impact of overgrazing in arid Australian rangelands. This is because nutrients, soil biota, organic matter and biological activity are concentrated in topsoil. Once topsoil is lost, the process of recovery can be extremely slow, if it occurs at all (Charley and Cowling 1968; Hacker 1989; Thurow 2000). Additionally, once soil erosion is initiated, it may be self-perpetuating (thus beyond management control) due to energy gradients and other hydrological characteristics that can prevent effective intervention (Thurow, Blackburn and Taylor 1988; Fanning 1994).

Grazing-induced soil erosion in my study area is most severe in landscapes with texture contrasting soil profiles that are subject to run-on. Such situations exist on breakaway footslopes (e.g. Sherwood land system), on concave slopes on weathered greenstone rocks (e.g. Gundockerta land system) and in concentrated drainage areas and flood outs associated with creeklines (e.g. Wilson land system). However, grazing-induced soil erosion is not generally a feature of my study landscapes (Pringle 1994c).

The initiation of grazing-induced soil erosion may be direct or indirect. In areas of intense stock activity, such as in the immediate vicinity of watering points, trampling by the hard hooves of stock (and feral goats) causes pulverisation of the soil. Pulverised soil is easily entrained by wind or water and lost from the system (Noble and Tongway 1986; Greene, Kinnell and Wood 1994).

Incised stock tracks are obvious examples of soil compaction and erosion, but comprise a minuscule part of the landscape beyond the immediate vicinity of watering points in large, arid zone paddocks (Lange 1969; Andrew and Lange 1986a). The extent to which stock tracks maintain intensity/density away from water appears to be a consequence of the age of the watering point and the number of stock using the watering point (Andrew and Lange 1986a) (see also Chapter 3).

There is growing recognition of the role played by microbiotic, cryptogam soil crusts in preventing soil erosion. These are biological rather than physical crusts and occur on all but the coarsest textured topsoils (e.g. as occur on sand dunes) in my study region (Van Vreeswyk 1994). Cryptogams appear to play a critical role in maintaining soil surface

coherence and structure against the impact of rain splash and sheet flows. They prevent dispersion and slaking, which can lead to entrainment of sediments in water flows and result in sediment loss from the landscape (Fletcher and Martin 1948; Greene and Tongway 1989; Belnap 1992; Eldridge and Greene 1994). This protection is particularly important in my study region, where clays are generally dispersive, having been derived from granitic rocks in the main (Churchward 1977; Pringle, Van Vreeswyk and Gilligan 1994). Physical contact by stock hooves appears to be the most serious cause of decline in soil crust cover and is generally restricted to the sacrifice zone around watering points (Rogers and Lange 1971; Andrew and Lange 1986a).

Beyond the area of intense physical disturbance around watering points, grazing effects on soil erosion are likely to be indirect, due to loss of protective soil cover. Plants and litter obstruct flows of wind and water, reducing their energy and hence their capacity to entrain sediment. They also intercept rain drops, reducing the severity of soil detachment due to rain splash (Marshall 1970; Thurow, Blackburn and Taylor 1988; Greene, Kinnell and Wood 1994). The rate of soil erosion is likely to increase rapidly below a threshold of soil cover, rather than be linearly related to soil cover. The nature of the response curve is likely to vary with soil landscape, types of vegetation architecture present and intensity of rainfall events (Thurow, Blackburn and Taylor 1988; Thurow 2000).

8.2.3.2 Infiltration

Infiltration is the process by which surface water enters the soil profile. This happens in two ways. Firstly, water enters the soil profile by filling pores. This is sorptivity. Secondly, when the soil profile becomes saturated, further infiltration occurs due to hydraulic conductivity. In water-limited arid ecosystems, infiltration can be regarded as the key determinant of primary productivity (Noy-Meir 1973; 1981; 1985). Key determinants of infiltration are soil texture, soil cover, soil surface conditions and soil structure.

Soil texture

Coarse textured soils usually have a higher infiltration rate than finer textured soils because they have more, larger air spaces through which water can infiltrate. Soil texture is typically affected by grazing when accelerated erosion removes coarse sediments, leaving a finer textured, less porous and less easily infiltrated soil surface (Beadle 1948; Marshall 1970; Hacker 1989). However, the reverse can also occur, when fines are ‘winnowed’ out of the soil profile by wind, leaving a more porous, coarser-textured topsoil. This has occurred over thousands of years in some sandy soil landscapes covered by a fine ferruginous lag in my

study region (Ararak land system) (Churchward 1977; Pringle, Van Vreeswyk and Gilligan 1994).

Increased bulk density has also been identified as a cause of reduced infiltration due to grazing activity (particularly trampling) (Johnston 1962; Crisp 1975; Bridge *et al.* 1983). However, in large, arid rangeland paddocks, this is likely to be confined to the sacrifice zone in the immediate vicinity of watering points (Andrew and Lange 1986a). Hacker (1979; 1986) found no evidence of soil compaction in my study area. However, loss of coarse surface soil in texture contrasting soil profiles can leave an exposed subsoil with a higher bulk density, resulting in a lower infiltration rate than the intact soil profile.

Vegetation cover

Vegetation cover is highly influential on infiltration within similar soil types. For instance, a gradient extending from tree and shrub cover (high infiltration), through bunchgrass and then shortgrass/sodgrass, to bare soil (low infiltration) has been proposed (Thurow, Blackburn and Taylor 1988). Tree and shrub cover may directly affect infiltration by funnelling water intercepted by tree canopies down the main branches to the trunk and then into the soil via relatively porous soils in the root zone (Slatyer 1962). In other cases the effect might be a combination of water funnelling and entrapment of highly porous sediments, as is the case with bush mounds in chenopod landscapes (Charley and Cowling 1968; Greene and Tongway 1989; Greene 1992).

Biopedturbation

Soil arthropods such as termites can also affect the porous nature of soils, acting as ‘ecosystem engineers’ as they move about the soil profile consuming organic matter and creating pores in the process (Whitford, Ludwig and Noble 1992; Lavelle 1997). There is a hierarchy of organisms involved in biopedturbation, ranging from microscopic organisms operating within individual peds, to mammals and reptiles (for instance) affecting topsoil properties within patches at a landscape or catchment level (Noble 1993; Whitford 1998; Whitford and Kay 1999). Biopedturbation is generally thought to increase soil porosity and hence infiltration.

Interactions

The determinants of infiltration interact closely. For instance, infiltration rates were 10 times greater in wooded groves and adjacent, grassy interception zones compared to open areas of bare soil in a semi-arid landscape in western New South Wales (Greene 1992). The higher infiltration rates in groves were attributed to the presence of macropores (> 0.75 mm

diameter), which in turn were maintained due to the capacity of the plant cover to intercept rain splash and prevent soils from slaking (which would destroy pores), as occurred in open areas. Similar interactive processes have been observed in grasslands (Greene, Kinnell and Wood 1994) and rehabilitating woodlands (Tongway and Ludwig 1993). Grassy areas generally have higher rates of infiltration than bare soil surfaces, except where soil surfaces have sandy accumulations (Hacker 1986).

Soil surface conditions may also directly affect infiltration. These conditions may be natural, such as cryptogamic crusts on red earths, or loose sand on sand dunes. However, there is unequivocal evidence that grazing can alter both the types of soil surfaces present at a site, and their proportions. Infiltration and primary productivity across a site may be strongly influenced by these dynamics (Johnston 1962; Hacker 1986; Tongway and Smith 1989; Noble, Greene and Muller 1998).

Cryptogamic crusts

Evidence of the role of cryptogamic crusts on infiltration is contradictory. For instance, infiltration rates were increased by three to four times when the crust was carefully removed from a chenopod landscape in north-western Australia (Greene and Tongway 1987). Presumable the cryptogams blocked soil pores at the soil surface and the removal of this biological crust thus facilitated infiltration. However, tests of infiltration rate against level of crusting without scalping provide strong, but contradictory evidence of crusting being beneficial for infiltration under these circumstances (Blackburn 1975; Greene and Tongway 1989; Belnap 1992).

The interaction of soil surface types and intensity of rainfall events can affect patterns of infiltration and run-off. For instance, cryptogamic crusts in open ('run-off') areas can lead to surface roughness features that effectively create very fine-scale water ponding structures. In such instances there may be little run-off from these bare, open areas to interception zones (e.g. mulga groves) after light rains, and run-on/run-off processes may operate only once the soil has become saturated and the 'ponds' have been filled (Noble, Greene and Muller 1998).

The influences of soil crusts and soil texture interact. Some soils without extensive cryptogamic crusts might have been subject to historical erosion, sealing and collapse in porosity; this would explain why infiltration is lower in some lightly crusted soils. However, biological crusts do not form readily on coarse textured soils, which usually have naturally high infiltration rates. In contrast, adjacent, finer-texture soils might readily form crusts but may have lower infiltration rates, because of having fewer, smaller pores. For instance, red

earths typically have a high content of fine sand, which leads to low porosity and lower infiltration rate (Greene 1992). The loss of the cryptogamic crust from such soils might not lead to increased infiltration.

Eldridge and Greene (1994) demonstrated the capacity of cryptogams to retard detachment of soil particles due to rain splash. Given this last finding and the potential for red earths in my study area to become sealed and less able to take up water, cryptogams could be regarded as essential conservers of soil stability, structure and biological activity (Eldridge and Greene 1994). Their tendency to spread across disturbed areas is probably a good thing.

In summary, for soil of a given texture, vegetated areas (e.g. groves and grass butts), areas covered by litter, and sandy accumulations are likely to have highest infiltration rates, cryptogamically crusted surfaces to be intermediate and sealed surfaces (often with collapsed soil structure) to have lowest infiltration rates. From this, it seems that longstanding prescriptions regarding the maintenance of perennial plant cover represents an appropriate strategy for managing the topsoil that is so vital to primary (including pasture) production in rangelands (Ratcliffe 1936; Packer 1951; Trumble and Woodroffe 1954; Copeland 1963). However, areas of low susceptibility to erosion may persist despite having low cover (Wilson and MacLeod 1991; Holm 1994); and soil deterioration may occur in overgrazed sites with minimal soil loss (Hacker 1979; 1986; Mott, Bridge and Arndt 1979). Pastoralists in my study region have endorsed the traditional principle of maintaining perennial plant cover (Anon.1993a; Pringle and Riches 1996); and it provides a good general approach that might be modified to suit specific local conditions.

Degradation of soils, including accelerated soil erosion and reduced infiltration, may occur in a step-wise fashion associated with episodic droughts. When cover is low going into a drought, erosion and consequent declines in infiltration may reduce the capacity for the soil to support the regeneration of protective vegetative cover with the advent of drought-breaking rains. This may exacerbate the situation in the following drought cycle (Thurow, Blackburn and Taylor 1988; Milton *et al.* 1994). The lessons to be learnt from these findings are that the maintenance of cover is critical for maintaining the soil's capacity to support primary productivity (Hacker and Tunbridge 1991). Managing for plant forms may be critical in this process, particularly in landscapes prone to erosion. Perennial shrubs and grasses are far more effective insurance against erosion than ephemeral plants, which have usually desiccated and disappeared by the time a drought is upon a manager, leaving the soil exposed to the full brunt of rain splash and wind and water flows.

8.2.4 The properties assessed

The chemical properties I assessed were:

- nitrogen and phosphorus, as essential nutrients,
- cation exchange capacity as an indication of attachment sites for nutrients,
- organic carbon, as a measure of the biological activity (or 'health') of the soil and as a further indicator of attachment sites for nutrients, and
- electrical conductivity, as a measure that reflects the capacity of plants to take up nutrients and water (osmotic effects).

Rather than measuring physical properties directly, I used the CSIRO Soil Surface Assessment system to provide indices of infiltration, soil stability and nutrient cycling (Greene and Tongway 1989; Tongway and Smith 1989; Tongway 1994; 1995; Tongway and Hindley 1995).

8.3 Methods used in the current study

8.3.1 Sampling topsoil

Topsoil was sampled within the landscape strata at each site over a six week period in November/December 1996 (see figures at the end of Chapter 2 for site layouts). At each sampling locus within a stratum, I took a 4cm by 4cm excavation of soil to 2cm depth using a paint scraper with a 2 cm depth mark on it. I took 5 samples at five metre intervals along the central axis of each sampling locus. These samples were bulked and stored in plastic bags for transport back to Canberra. Most soils were air dry, but where soils were slightly moist, they were left open in the WMC Resources laboratory at Leinster until they were air dry.

In chenopod landscapes I sampled half way up the edge of bush mounds and at least 30cm from any bush mound in adjacent inter-bush areas. In groved mulga landscapes I sampled half way between the base of a tree and the projected edge of its canopy, and then at least five metres away from the upper edge of the grove. In sandy mulga landscapes I sampled every five metres, but avoided *Acacia linophylla* and *A. ramulosa* shrubs on sandy banks. (These occur in uncommon patches that I could not hope to sample adequately, and there is no evidence that their abundance is related to grazing history). Topsoil sampling took me six weeks to complete in November and December 1996.

8.3.2 Laboratory analysis

Air-dry samples were sieved to < 2 mm for all chemical analyses. The following chemical properties were assessed using the following methods:

- potentially mineralisable nitrogen (ammonium-N released from organic soil nitrogen) by heating with 2 M KCl at 95°C for 16 hours (Gianello and Bremner 1986),
- available phosphorus by a sodium bicarbonate extraction (Colwell 1965),
- organic carbon by a modified Walkley-Black procedure (Colwell 1969),
- cation exchange capacity by extraction with silver thiourea (Pleysier and Juo 1980)
- electrical conductivity and pH by 1:5 soil-water solutions of suspended sediments (Loveday 1974).

I also assessed total carbon measured on different soil depth samples from the Caesium-137 reference profile pits (see Chapter 7 for details). I had three trenches on the reference sandy bank, mulga drainage lane and inter-bush mound. Samples were taken at centimetre depth intervals. Analysis was done by combustion in a stream of oxygen in a furnace.

8.3.3 Assessing soil surface condition

I used the Rangeland Soil Condition Assessment Manual, which was developed for this region (Tongway 1994; 1995). It is a quadrat-based system of assessment that includes visual observations, manual pressure on the soil surface, and immersion of surface soil crumbs. Quadrats were stratified within sites according to the landscape organisation described in Chapter 7. Five quadrats at 2 m intervals were assessed along the centre line of sampling loci at chenopod sites (bush mounds and inter-bush areas separately); groved mulga sites (groves and inter-groves separately) and sandy mulga sites (sandy banks and mulga drainage lanes separately). I avoided occasional large acacias on sandy banks, as explained previously. I assessed areas under trees and open areas separately in mulga drainage lanes. I did not analyse log mounds (5 quadrats) and shrubs (14 quadrats) for the sandy bank data on the basis that these sampling frequencies were inadequate to represent variation in the eight sandy bank sites sampled.

I summarise the hierarchical breakdown of sampling stratification and sampling effort in Table 8.1. I kept the sampling effort uniform at a land unit level (the level encompassing most anthropogenically modulated landscape processes), but doubled effort at chenopod sites as the sampling area was small and the major time spent was in accessing these sites, rather than sampling them. In all cases, I exceeded the sampling intensity per site

recommended in the manual (Tongway 1994). Given that my design was already inherently unbalanced, I tried to optimise sampling effort, rather than aim for perfect balance among all sites.

Each quadrat took approximately a minute to assess, as was suggested by the developer of the methodology (Tongway 1995). It is a simple and quick method. The most difficult (and important) part of the assessments lies in recognising underlying landscape organisation, before commencing sampling.

There are 11 soil surface features recorded at each quadrat, and the scores for these features are aggregated to derive indices of soil stability (S), infiltration/runoff (I) and nutrient cycling status (N). Scores are calculated as a percentage of the maximum possible score for each index:

- Soil cover (interception of raindrops, classes 1-4) (S)
- Soil cover (obstruction to overland flow, classes 1-4) (I)
- Crust broken-ness (classes 1-4) (S)
- Cryptogam cover (classes 1-4) (S,N)
- Erosion features (classes 1-4) (S)
- Eroded materials (classes 1-4) (S)
- Litter (*cover* with classes 1-6, *origin* with classes ‘local’ or ‘transported’ and *incorporation* with classes ‘loose’, ‘slight’ and ‘extensive’) (S, I -*cover* only, N-all 3)
- Soil microtopography (classes 1-5) (I,N)
- Slake test based on immersion of soil crumbs (classes 1-4) (S)
- Surface nature based on manual pressure (classes 1-4) (S,I)
- Soil texture (classes 1-4, from a bolus test) (I)

8.3.4 Data analysis

My data analysis followed the standard procedures described in Chapter 2, commencing with raw data. I followed the advice of Ross Cunningham (Statistical Consulting Unit at the Australian National University) as to appropriate data transformations, where there was systematic variation amongst the residuals of my statistical models.

For the soil surface data, where I had a different hierarchy of sampling units to the norm, my new random model was:

gradient/site/unit/locus/patch/rep.

This hierarchy allowed me to model the ('random') effects of :

- 'reps' (quadrats) within patches (e.g. bush mound, mulga clump or grove),
- patches within sampling loci (central transect axes at a site),
- sampling loci within units (e.g. sandy bank or mulga drainage lane),
- units within sites (distances from water), and
- sites within gradients (paddocks sampled).

See Chapter 2 for a more detailed account of my statistical modelling procedures.

8.4 Results

The results in this chapter differ from previous chapters in that they are derived from two landscape stratifications. The standard stratification was used for chemical analyses, but I split mulga drainage lanes in sandy mulga landscapes into mulga clumps and inter-clumps for the three indices of soil surface condition. There was no change for chenopod (bush mound and inter-bush) and groved mulga (mulga grove and inter-grove) landscapes.

8.4.1 Topsoil chemistry

8.4.1.1 Potentially mineralisable nitrogen

The effect of landscape stratum within each landscape type was by far the most significant influence on variation in potentially mineralisable nitrogen (Table 8.2). Landscape stratum differences all involved significantly higher potentially mineralisable nitrogen in run-on or 'fertile patches' within landscapes. This contrast within landscapes was strongest between mulga groves and inter-groves (Figure 8.1).

Potentially mineralisable nitrogen declined with increasing Stocking Rate (SR) in bush mounds (slightly) and mulga groves (more), while it increased strongly in mulga drainage lanes and sandy banks (Figure 8.2). Stocking History (SH) and Distance from water (DW) did not explain any significant variation in potentially mineralisable nitrogen (Table 8.2).

8.4.1.2 Available phosphorus

Differences between strata from different landscapes were the most prominent source of variation in available phosphorus ($p<0.001$), with no significant within-landscape differences and a weakly significant difference between landscape types ($p=0.02$) (Table 8.3, Figure 8.3). Inter-bush areas in chenopod landscapes had the lowest available phosphorus, significantly less than inter-groves and mulga drainage lanes. Interestingly, though differences were not statistically significant, mulga groves had a lower mean value than inter-groves, while bush mounds had a higher mean than inter-bush areas.

While modelled Stocking History (SH) and Distance from water (DW) had no significant effect on variation in available phosphorus, as with the results for potentially mineralisable nitrogen, a significant interaction effect was found between modelled Stocking Rate (SR) and landscape stratum (Table 8.3). Landscape strata showed different sensitivities to modelled Stocking Rate, but all strata except bush mounds showed an increase in available phosphorus with increasing Stocking Rate (Figure 8.4). The sandy mulga landscape strata were most responsive (i.e. had the steepest slopes).

8.4.1.3 Organic carbon

Total organic carbon declined rapidly with depth from the surface in all three soil profiles sampled for Causium-137 (Figure 8.5). Sandy banks profiles differed from the mulga drainage lane and inter-bush (chenopod landscape) profiles. While total carbon dissipates similarly rapidly with soil depth near the surface, the base 'unenriched' level is higher in sandy profiles than it is in the other, finer-textured and less freely draining soils.

As with nitrogen and phosphorus, differences between landscape strata proved to be the dominant influence on organic carbon concentrations, with significant differences within the two types of mulga landscape (Table 8.4, Figure 8.6). Mulga groves had more organic carbon than inter-groves, as did mulga drainage lanes compared to sandy banks. While the mean for bush mounds was higher than that for inter-bush areas, it was not significant at this level. The directions of differences for organic carbon are similar to potentially mineralisable nitrogen (and opposite to those for available phosphorus in groved mulga landscapes).

Modelled Stocking History (SH) had a significant effect on organic carbon, but the ranges of decline in organic carbon due to grazing were small in all landscape strata (Figure 8.7). This graph indicates three groups of landscape strata; mulga drainage lanes and groves with highest organic carbon, an intermediary group consisting of sandy banks, bush mounds and inter-groves, and then inter-bush areas in chenopod landscapes, which had very little organic

carbon. The effects of modelled Stocking Rate (ST) and distance from water (DW) were not significant.

8.4.1.4 Cation exchange capacity

Differences among landscape strata also proved most influential on variation in cation exchange capacity, with a significantly higher level in mulga drainage lanes compared to adjacent sandy banks in sandy mulga landscapes (Table 8.5, Figure 8.8). Cation exchange capacity was also significantly lower on sandy banks and in chenopod landscapes (bush mounds and inter-bush areas) compared with groved mulga landscapes (mulga groves and inter-groves).

None of the estimates of grazing had a significant effect on cation exchange capacity (Table 8.5).

8.4.1.5 Electrical conductivity

Differences among landscape strata were also apparent in electrical conductivity (Table 8.6, Figure 8.9). Sandy banks had lower electrical conductivity than all other landscape strata, while bush mounds had higher electrical conductivity than all other strata. Groves had higher electrical conductivity than inter-groves. Inverse transformations were used to overcome systematic variation in residuals in preliminary statistical models of the raw data.

Modelled Stocking History (SH) and landscape strata had a strong interactive effect on electrical conductivity (Figure 8.10). All landscape strata except for bush mounds showed some increase in electrical conductivity with increasing Stocking History (SH). Sandy banks seemed most responsive, while inter-bush areas had by far the greatest range of electrical conductivity attributable to grazing (see translated scores of inverse electrical conductivity next to the y-axis). Modelled Stocking Rate (SR) showed similar trend, but without an interactive effect with landscape strata, while there was no effect of Distance from Water (DW).

8.4.2 Soil surface assessments

8.4.2.1 Infiltration index

There was a very strong effect of patch type on the infiltration index within landscapes, with the three fertile patch types (bush mounds, mulga clumps and mulga groves) having the highest scores, followed by sandy banks. The open patches (inter-bush, inter-clump and inter-grove) had the lowest infiltration scores (Table 8.7, Figure 8.11).

Modelled Stocking History (SH) had a significant, but very weak negative effect on infiltration index scores; and this did not interact with landscape stratification (Figure 8.12). On the other hand, modelled Stocking Rate (SR) had a significant interactive effect on infiltration index scores with patch type and lesser interactive effects with land unit and landscape type (Table 8.7). While this latter model predicted declining infiltration index scores with increasing grazing for most patch types, the opposite was clearly the case for mulga groves. However, the grazing effects were very weak (nearly horizontal slopes of fitted lines). Similar interaction was found with Distance from Water (DW), though the nature of trends was opposite to Stocking Rate (SR).

8.4.2.2 Nutrient cycling index

A similar effect of landscape stratum on nutrient index score was found to that found for infiltration scores, with strong within-landscape contrasts in nutrient cycling index. However sandy banks were clustered with the lower group of inter-bush, inter-clump and inter-grove areas, rather than being intermediate (Figure 8.13); and bush mounds had significantly lower values to mulga clumps and groves.

Statistically significant interactive effects with patch type were found with all three estimates of grazing history. There were also significant interactive effects with higher levels of landscape stratification (Table 8.8). Scores for the nutrient cycling index declined with modelled Stocking History (SH) for bush mounds, mulga clumps and inter-groves while the opposite was the case for inter-clump areas in mulga drainage lanes and adjacent sandy banks (Figure 8.14). This general pattern of interaction was consistent for the Stocking Rate model and distance from water (DW). While these interactions were statistically significant, they are very weak effects compared to landscape differences and so account for a trivial amount of variation overall.

In order to investigate the validity of the nutrient cycling index, I compared its mean values with mean scores for potentially mineralisable nitrogen in each landscape strata (Figure 8.15). I derived the nutrient cycling value for mulga drainage lanes by weighting the respective means of mulga clumps and inter-clumps according to the average proportion of sites they occupied. I used the site transects to assess proportional area (see Chapter 7). The results indicate a significant positive relationship ($R^2 = 0.74$; $p = 0.03$), indicating that the index has considerable value as a correlate of potentially mineralisable nitrogen both within and between landscape types within this region.

When I removed the imputed data point (the one derived by area-weighting clump and inter-clump mean scores) for mulga drainage lanes, the correlation was increased to $R^2 = 0.82$, with an unchanged probability of significance ($p = 0.03$).

8.4.2.3 Soil stability index

A similar, but weaker pattern of contrasts within landscapes resulted for the index of soil stability (Figure 8.16). Inter-bush, inter-clump, inter-grove and sandy banks had lowest scores. The contrasts between landscape strata were weaker than for the indices of infiltration and nutrient cycling.

Weak, but statistically significant interactive effects on the soil stability index were found, involving all three estimates of grazing history and patch type (Table 8.9). Soil stability scores declined with increasing grazing for all three estimates, for most patch types. No contradictory trends were observed (Figure 8.17).

8.5 Discussion

8.5.1 Topsoil chemistry

8.5.1.1 Potentially mineralisable nitrogen

Landscape variation

The tendency for nitrogen to be concentrated in fertile patches is consistent with the hypothesis of Noy-Meir (1973; 1981; 1985) and patterns described in natural (Charley and Cowling 1968; Tongway and Ludwig 1990) and extremely disturbed arid and semi-arid ecosystems (Tiedemann and Klemmedson 1973; Schlesinger *et al.* 1990; 1996; Reynolds *et al.* 1999). The large difference between mulga groves and inter-groves is consistent with the pattern for Caesium-137 (see Chapter 7). It is also further evidence of an intimate coupling of water and nitrogen in space in arid and semi-arid ecosystems (Bremner and Shaw 1958; Tiedemann and Klemmedson 1973; Cowling 1978; Fletcher, Sorensen and Porcella 1978; Skujins and Trujillo Y Fulgham 1978).

This coupling is also indicative of the influence of water distribution on spatial patterns of nutrient cycling because the measure of nitrogen used represents the highly labile, organic pool (Gianello and Bremner 1986).The concentration of potentially mineralisable nitrogen in fertile patches highlights the spatial congruence of patches of perennial vegetation, nutrients, water availability, nutrient cycling and primary productivity (Slatyer 1975; Moorhead and Reynolds 1989; Schlesinger *et al.* 1990; Reynolds *et al.* 1999).

Grazing effects

The only grazing estimate that influenced potentially mineralisable nitrogen was Stocking Rate (SR). Different responses of nitrogen to grazing were apparent among mulga landscape types, in ways consistent with previous research findings.

The decline in potentially mineralisable nitrogen with increased Stocking Rate in mulga groves is consistent with studies in similar banded landscapes in semi-arid eastern Australia and conforms to a conceptual model of landscape leakage and homogenisation under heavy grazing (Tongway and Ludwig 1990; Milton *et al.* 1994; Ludwig *et al.* 1997). The effect detected was not particularly pronounced, which is consistent with the relatively weak patterning of strata in my groved mulga landscapes compared to the more strongly groved landscapes in semi-arid eastern Australia where the model was developed (Tongway and Ludwig 1990; Ludwig and Tongway 1995b) and abroad (Montana 1992; Cornet *et al.* 1992). The patterning is also weak compared with similar landscapes described in other parts of the arid shrublands of Western Australia (Mabbutt *et al.* 1963; Mabbutt and Fanning 1987; Curry *et al.* 1994). If under-storey vegetation plays a major role in the functioning of these landscapes (as suggested in several studies), it is not surprising that grazing has a weak effect in my study landscapes, where the understorey typically is weakly developed.

The decline in potentially mineralisable nitrogen with increased Stocking Rate in bush mounds is also consistent with a model of leakage and homogenisation (Ludwig and Tongway 1995b), rather than re-organisation at coarser grain (Whitford, Martinez-Turanzas and Martinez-Mesa 1995; Schlesinger *et al.* 1996; Reynolds *et al.* 1999).

Potentially mineralisable nitrogen increased strongly with increasing grazing in both landscape strata in the sandy mulga landscape. However, different explanations may be warranted to account for these similar responses in grassy sandy banks and wooded mulga drainage lanes.

Coincidental increases in soil nitrogen and/or its mineralisation with increasing grazing have been reported in some other grassland landscapes. These responses to increased grazing have been attributed to re-allocation from standing vegetation to soil by herbivory and trampling, accompanied by increased rates of cycling (Shariff, Biondini and Grygiel 1994; Manley *et al.* 1995; Schuman *et al.* 1999). Shariff *et al.* (1994) reported a ‘humped’ response to grazing. This could be consistent with my results since my heaviest grazed sites were not in the sacrifice zone and correspond more closely to their ‘moderate’ grazing treatment.

Similar results to these were observed by Tongway and colleagues (unpublished data) in experimentally grazed landscapes in south-east and north-east Queensland.

A different explanation may be more appropriate for the response in adjacent mulga drainage lanes. The most obvious impact of grazing at sites near water is the patchy proliferation of 'woody weeds' (*Eremophila spectabilis* and *E. gilesii*) in the under storey of larger clumps of mulga trees. That these are increaser species is well-documented (Burrows 1972; Mitchell and Wilcox 1984; Curry *et al.* 1994; Pringle 1994b). At the site nearest water in Gradient 6SM, litter in mulga clumps invaded by these under-storey shrubs was continuous and up to 4 centimetres thick in places. In some more open areas, there was a combination of dense sheep tracks, scalded soil surfaces and several long-dead mulga trees. These relatively open, disturbed gaps do not rehabilitate quickly. They have also been adopted as thoroughfares by sheep, further disturbing the soil, retarding any regenerative processes and facilitating establishment of coarser, more contrasting patterns of fertile patches and intervening inter-patches.

Less strongly clumped infestation of *Eremophila spectabilis* and *E. gilesii* was observed at the other sandy mulga gradient's site closest to water. There was not the same pattern of gap disturbance and open spaces ('inter-clumps') had intact, stable surfaces.

The response to grazing of these mulga drainage lanes is consistent with the 'resource island' model developed in the south-western United States (Tiedemann and Klemmedson 1973; Brown and Archer 1989; Archer 1990; 1994; Schlesinger *et al.* 1990). The key prediction of this model is that grazing can precipitate a 'flip' from one stable state to another. In contrast to the pattern described in North America, the transition in the mulga drainage lanes does not involve a shift from 'homogenous' grasslands to shrublands or woodlands, and does not involve any noticeable change in landscape-scale physiography (e.g. development of coppice dunes). However, the transition does appear to represent a 'flip' to another stable and apparently resilient domain of attraction (Holling 1973; Connell and Slatyer 1977; Westoby, Walker and Noy-Meir 1989b).

8.5.1.2 Available phosphorus

Landscape variation

There was a contradictory pattern of distribution of available phosphorus and potentially mineralisable nitrogen in groved mulga landscapes, where mean available phosphorus tended to be higher in inter-groves and potentially mineralisable nitrogen was more

concentrated in groves. This could reflect the capacity of mulga trees' ectomycorrhizal root associations to take up and store scarce phosphorus.

It is likely that all but the very lightest rainfalls would run off into groves, and consequently, the soil moisture would be shallow and short-lived in inter-grove areas (Noble, Greene and Muller 1998). It is possible that soil moisture is often too short-lived out in the open inter-grove areas for mulga trees established in groves to regenerate fine mycorrhizal roots (Went and Stark 1968; Khudairi 1969) near the soil surface. However, soil moisture in the inter-grove areas may be sufficient for soil flora to rejuvenate and take up nutrients. Thus it is possible that soil flora, the 'microbiotic crusts' (Eldridge and Greene 1994) can sequester a significant proportion of scarce available phosphorus in the top few millimetres of soil, maintaining it in a cycling system in the topsoil of the inter-grove areas.

Days after light rainfalls, dark (moist) halos can be seen on the soil surface around mulga groves, while open areas are lighter and dry. Light rainfalls may be far more effective in stimulating root activity in mulga groves than inter-groves. Groves are sinks for water and facilitate infiltration via stem flow into the root zone (Slatyer 1962; 1965; Pressland 1973; Anderson and Hodgkinson 1997). They also have higher soil organic content and litter cover (Walker 1974; Tongway and Smith 1989). Thus available phosphorus may be much more efficiently taken out of soil and sequestered in perennial plants in mulga groves than in inter-grove areas.

This contradictory distribution of available soil phosphorus compared with water and nitrogen has been observed in similar banded landscapes in semi-arid eastern Australia (Tongway and Ludwig 1990) and in grasslands invaded by mesquite in USA deserts (Tiedemann and Klemmedson 1973). However, coupled distributions of phosphorus and nitrogen have been reported from other areas (Schlesinger *et al.* 1996). The results from my study further highlight the complexity of phosphorus dynamics in arid ecosystems.

The patterns in chenopod shrublands (where proteoid root systems are common in shrubs) and sandy mulga landscapes might indicate a simpler, more predictable system of phosphorus cycling in these landscapes, with closer coupling of nitrogen and phosphorus.

Grazing effects

Available phosphorus increased with increasing long-term average Stocking Rate (SR), particularly on sandy banks, which is opposite to the trend found by Hacker in similar soil samples (0-2 cm) of the same landscape type in my study region. I can only speculate that Hacker's sites were near fence-lines (Hacker 1978; 1979), which would have meant they

were subject to higher trampling than might be expected at the distances from water that he sampled. Similar results to Hacker's were also found in central Australia, but follow-up work indicated that soil phosphorus was not a reliable indicator of 'range condition' (implied grazing impact) in this region (Friedel, Cellier and Nicolson 1980; Friedel 1981).

One explanation for increased phosphorus levels with increased grazing is that there is disproportionately higher defecation near watering points (Landsberg and Stol 1996). Phosphorus in inorganic form is readily available when leached from dung pellets (Barrow and Lambourne 1962; Barrow 1975), and being a scarce nutrient, would be rapidly sequestered on attachment sites in surface soil. As with nitrogen, trampling might also have been a contributing factor.

8.5.1.3 Organic carbon

Landscape variation

The reference soil profiles all showed enrichment of organic carbon in topsoil and a rapid decline to a base level with depth. This provides additional evidence to that previously collected in my study area by Hacker (1979; 1986) and Tongway (1994), that these soils conform to the typical profile patterns described by Charley and Cowling (1968). The lower concentration in topsoil and higher concentration deeper in the soil profile in sandy banks compared with mulga drainage lanes and chenopod flats are consistent with a coarse, sandy soil that is less able to sequester available phosphorous near the surface and hence may allow more diffuse enrichment below the topsoil. This may be because the capillary action that concentrates soil water (and biological activity) near the topsoil in these deep, sandy soil profiles is relatively weak, and water is effectively stored at depth. This has been described as the inverse-texture effect (Noy-Meir 1973). Root activity may be far better developed in moister, deeper parts of these coarse-textured soils, and root exudates and senescence/detritivory might account for relatively high levels of sub-surface carbon.

The relative concentration of organic carbon in bush mounds, inter-groves and mulga drainage lanes is further evidence of a spatial coupling of nutrients (especially nitrogen), water (presumably) and biological activity in these more fertile, vegetated areas. The large difference between groves and inter-groves is indicative of this being a nutrient-cycling system driven by litter and other organic matter. The under-storey is usually sparse and unlikely to have much 'trapping power'. However, litter cover is usually continuous or nearly so in groves in these landscapes of low hydraulic energy. Litter also appears to help trap detritus, including dung. Inter-groves are virtually devoid of litter, except when in association with a surface obstacle such as a fallen log (personal observations). Fine root growth/death

in response to light rainfalls in mulga groves is also likely to influence the differences in organic carbon between groves and inter-groves.

If litter decomposition is a major contributor to soil organic carbon, this suggests that the detritivory path is very influential (Whitford *et al.* 1983; Whitford 1986b; Whitford, Stinnett and Anderson 1988; Whitford, Ludwig and Noble 1992). Groves are usually quite shaded, at least from maximum radiation when the sun is at its zenith, which is consistent with a dominant detritivory pathway in litter decomposition. In open areas, radiation-driven oxidation of carbon would account for much decomposition via the inorganic pathway (Moorhead and Reynolds 1989).

Grazing effects

Grazing-related reductions in organic soil carbon are statistically significant, but small and may therefore have little ecological significance. The fact that little or no organic carbon and potentially mineralisable nitrogen appear to be lost in response to grazing indicates that soil profiles probably remain largely intact in my study landscapes. In their landmark publication, Charley and Cowling (1968) showed that soil erosion is the major influence on nutrient depletion in arid Australian soils.

The minor nutrient losses I found may have been due to slight leakage from these landscapes (Ludwig and Tongway 1995b). However, they may also reflect a partial switch in litter decomposition from the detritivory pathway to the inorganic oxidation pathway as shade declined with declining perennial plant cover in parts of these landscapes (Moorhead and Reynolds 1989).

8.5.1.4 Cation exchange capacity

Landscape variation

Differences in cation exchange capacity between and within landscape types mirrors variation in the clay content and organic matter of the soils concerned. The landscape strata with coarse-textured A-horizons have low cation exchange capacity, while the loamy soils in the mulga landscape strata have higher cation exchange capacity. Similarly, the strata with the highest organic carbon also had highest cation exchange capacities.

Grazing effects

There was no evidence of grazing impact, which further supports the idea that there has been little recent sediment loss from these landscapes.

8.5.1.5 Electrical conductivity

Landscape variation

The deep, coarse-textured, well-drained sandy bank soils had considerably lower electrical conductivity (i.e. were less salty) than soils in all other landscape strata. The contrast between bush mounds and inter-bush areas in chenopod areas reflects the capacity of halophytic shrubs to concentrate salts by litter fall within their bush mounds (Sharma and Tongway 1973).

Grazing effects

While electrical conductivity in sandy banks seemed most affected by grazing, the dynamic range (range of values for the fitted response) was small and likely to be within the tolerance ranges of the sandy bank flora.

The effect of grazing on inter-bush areas is of far more environmental concern, considering that this landscape stratum also expands with increased grazing history (see Chapter 7). The dynamic range of 45 *u*S/cm (ungrazed) to 200 *u*S/cm (most heavily grazed) would probably affect the capacity of many plant species to persist in these chenopod landscapes.

At the chenopod sites that had been historically more heavily grazed, the eroded, scalded and probably compacted soils near water would have had substantially reduced plant cover, litter, microbiotic crust and sandy A-horizons. Thus there would have been little check on upward capillary pressure in their soil profiles. This would allow salts to move in solution up the soil and be precipitated near the soil as evaporation increased their concentration. Hacker also noted increased salinity in expanding bare patches at more heavily grazed sites and an associated strong influence on patterns of species recruitment in this region (Hacker 1979).

8.5.1.6 A synthesis of influences on topsoil chemistry

Landscape effects

Landscape effects (particularly at the level of landscape stratum) tended to be the dominant source of variation in topsoil chemistry.

Potentially mineralisable nitrogen and organic carbon had similar spatial patterns of concentration, with higher levels in groves, bush mounds and mulga drainage lanes than in inter-groves, inter-bush areas and sandy banks. This pattern is consistent with the ideas of Noy-Meir (1973; 1981; 1985) and Tongway and Ludwig (1990): that nutrients (and water)

tend to be concentrated into sink zones, rather than be evenly spread across arid landscapes.

Available phosphorus followed a similar pattern, except in groved mulga landscapes where inter-groves had higher concentration than in groves. Efficient phosphorus uptake by ectomycorrhizal roots may have played a role in this apparent reversal of nutrient distribution in this landscape. Tongway and Ludwig (1990) reported a similar uncoupling of phosphorus and nitrogen and organic carbon in groved mulga landscapes in eastern Australian.

Cation exchange capacity was strongly influenced by landscape stratum (probably reflecting differences in clay content and levels of organic matter), but did not appear to be affected by grazing. A similar pattern was apparent for electrical conductivity, which was far lower in coarse textured soils of sandy banks than elsewhere, while bush mounds had the highest levels, consistent with accumulation and leaching of saline leaf litter (Sharma and Tongway 1973).

Grazing effects within landscapes

There were small decreases in organic carbon and potentially mineralisable nitrogen with increasing grazing history in chenopod and groved mulga landscapes. However, in sandy mulga landscapes there were much larger and stronger increases in potentially mineralisable nitrogen and also available phosphorus as grazing increased. Increases in nutrient concentration were strongest on sandy banks, where trampling, increased cycling of leaf material by grazing and perhaps increased dung and urine inputs may account for the response (Manley *et al.* 1995; Schuman *et al.* 1999). The small decreases in organic carbon and potentially mineralisable nitrogen in other landscapes may indicate slight erosional losses.

While electrical conductivity increased significantly with increased grazing on sandy banks, the ecologically more significant response was probably in inter-bush areas, where secondary salinisation appears to be a major impact of heavy grazing.

The relatively small dynamic ranges in variables other than electrical conductivity indicate that these are inherently stable landscapes: major losses of nutrients and organic carbon due to export of topsoil from landscapes does not appear to have occurred (see Charley and Cowling 1968). This region has experienced comparatively light grazing history (Faithful 1994), compared to river systems to the north-west that were far more heavily stocked and where soil erosion is a more widespread problem (Wilcox and McKinnon 1972; Curry *et al.* 1994). It is possible that the lighter grazing in my study region has not been sufficient to

cross a threshold at which resource regulation is lost and soil erosion accelerates (National Research Council 1994; Thurow 2000; Tongway and Hindley 2000).

The different landscape strata in my study appear to respond to grazing in different ways. A conceptual model of denudation and loss of productivity (see model 4B in Figure 7.1, Chapter 7) may be appropriate for the more heavily grazed chenopod landscapes, where secondary salinisation has occurred (e.g. Milton *et al.* 1994).

The pattern of response in mulga drainage lanes appears more congruent with a conceptual model of re-organisation at coarser scale (model 3A in Figure 7.1), where a new pattern of 'woody weed' infested mulga patches has developed, sometimes with disturbed, open areas in between (Archer *et al.* 1988; Schlesinger *et al.* 1990; Whitford, Martinez-Turanzas and Martinez-Mesa 1995). On sandy banks, grazing appears to be having little effect on landscape patterning, although floristic changes have occurred (see Chapter 5 for details of floristic responses to grazing). This response is most congruent with a model of conversions within vegetation types (model 1A in Figure 7.1).

In groved mulga landscapes the pattern is weak, but generally consistent with a model of fertile patch decline and homogenisation (model 3A in Figure 7.1) (Ludwig and Tongway 1995b; Ludwig et al 1997).

In general however, landscape differences affect topsoil chemistry far more profoundly than does grazing, particularly in mulga landscapes. Basic landscape functions remain dominant over the dissipative effects of grazing. The soils of these landscapes thus appear relatively resistant to grazing, compared to similar landscapes in eastern Australia and perhaps to the north and north-west of this study area in Western Australia. This is likely to reflect flatter, internally draining catchments, as well as a shorter, lighter grazing history.

8.5.2 Soil surface assessments

8.5.2.1 Infiltration index

The patterns of variation I assessed in infiltration indices are consistent with previous research findings, which suggest that infiltration within a particular soil-landscape is highest under shrubs and trees, is slightly lower in grassy areas and is lowest in open, largely bare areas (Greene and Tongway 1987; Thurow, Blackburn and Taylor 1988; Greene and Tongway 1989). Grazing appears to reduce infiltration (as indicated by the index), though the effect is generally weak. Hacker found similar trends on sandy soils in my study region: he recorded reduced infiltration only in very heavily grazed sites (Hacker 1986).

8.5.2.2 Nutrient cycling index

The indicated variation in nutrient cycling between landscape strata is consistent with the patterns of variation in mineralised nitrogen. Both patterns of distribution support the idea of natural resource concentration in fertile patches (Noy-Meir 1981; 1985; Ludwig and Tongway 1995b; Tongway and Ludwig 1997).

Grazing effects on the nutrient cycling index were in the same direction as those for potentially mineralisable nitrogen in bush mounds (decrease), mulga groves (decrease) and sandy banks (increase). The increase in potentially mineralisable nitrogen due to grazing was not mirrored in the nutrient cycling index (weak decrease). Overall though, there was weak, but general agreement between these two indicators of nutrient cycling.

The nutrient cycling index appears to have substantial transportability at least within this region, evidenced by the strong correlation between mineralised nitrogen and the nutrient cycling index across landscape types (Figure 8.15). That grazing effects on the index were quite weak might suggest that, rather than being site specific and sensitive, the index may be generalisable, but insensitive to management impacts. However, levels of potentially mineralisable nitrogen also varied far more due to differences in landscape strata than grazing. The responses of the index and potentially mineralisable nitrogen to grazing were also most closely correlated in this landscape stratum. Thus grazing may generally have little absolute effect on nutrient cycling in the landscapes studied, except perhaps in the most heavily grazed areas.

8.5.2.3 Soil stability index

The soil stability index scores were generally high, exhibiting far less variation within landscapes than the nutrient cycling and infiltration indices. This pattern is consistent with a region of subdued physiography and low hydraulic energy. The proposition of generally stable soils in all study landscapes is further supported by the very weak effect of grazing history.

8.5.2.4 Synthesis of soil surface assessments

Variation in all three indices was primarily influenced by landscape heterogeneity: grazing effects were statistically significant, but quite minor in comparison. This is what one might expect in arid landscapes in which topsoil properties and processes differ very strongly in space, reflecting the functional processes that concentrate resources into proportionally small areas of disproportionately high fertility and biological activity. These landscapes seem resilient to all but the highest of the levels of grazing I assessed.

A methodologically useful finding is that the nutrient cycling index appears to correlate quite well with potentially mineralisable nitrogen across landscape types of this region.

8.5.3 A synthesis of landscape patterns and grazing effects on them

In this section, I relate this chapter's findings regarding topsoil chemistry and processes with those of Chapter 7 on landscape organisation.

The landscapes I have studied show evidence of strong patterning reflecting processes of resource sequestering, storage and utilisation. The Caesium-137 data provide unequivocal evidence that this patterning reflects strong redistribution of scarce resources into disproportionately small, fertile patches; as Noy-Meir hypothesised should occur in arid landscapes (Noy-Meir 1973; 1981; 1985). Potentially mineralisable nitrogen, organic carbon, infiltration, nutrient cycling and soil stability all appear to be greatest in these fertile patches.

The effects of grazing on these patterns are relatively small in comparison to those of landscape heterogeneity. The effect of grazing likely to have most biological significance was the emergence of secondary salinisation in grazed open areas in chenopod landscapes. Given that these areas have also been shown to expand considerably with increased grazing history, secondary salinity may be a hitherto underestimated impact of pastoral development in this prized pastoral country type.

The expansion of open areas and decline in bush mounds in chenopod landscapes begs further heuristic interpretation. If nutrients, organic carbon, infiltration, nutrient cycling and soil stability are significantly higher in bush mounds than inter-bush areas, the spatial dynamics alone indicate that denudation to a state of lost productivity and salinisation (Milton *et al.* 1994) is a valid model for conceptualising the impact of grazing on this landscape type.

The implications of this heuristic interpretation extend beyond pastoralism and the specific regional context. In chenopod (and similar) landscapes, if shrubs are removed and areas of high soil carbon are lost, there may be serious implications for national and global carbon budgets. The situation is likely to be far worse in exoreic (externally draining) catchments such as those of the Murchison and Gascoyne Rivers to the north-west of the study area. There, grazing history is longer and has been heavier in landscapes that are not as inherently stable as the ones described here. Extensive denudation of vegetation, as well as severe soil erosion, has been reported in fluvial landscapes of these catchments (Wilcox and McKinnon 1972; Curry *et al.* 1994).

In contrast, the mulga landscapes I have studied appear to be remarkably resistant to grazing, at least in terms of effects on their landscape organisation and topsoil properties.

From a theoretical perspective, the following models of grazing impacts on landscape patterning (see Figure 7.1 in Chapter 7) appear most appropriate for the various landscape strata:

- denudation and loss of productivity (model 4B) in chenopod landscapes
- re-organisation at coarser scale (model 3A) in mulga drainage lanes in sandy mulga landscapes
- conversion within vegetation type (1A) for sandy banks in sandy mulga landscapes, though examples more congruent with model 3A have been reported in this region (Pringle 1994b)
- fertile patch leakage, an early phase (3B) of transition to productive herblands (model 4A) or denudation and loss of productivity (model 4B) might apply in groved mulga landscapes

These models are invoked to help conceptualise the impacts of grazing based on changes wrought at highest levels of impact. They do not provide a regional assessment of the impacts of grazing on these landscapes. 'Desertification' (*sensu* Mainguet 1991) applies only to the small areas in the immediate vicinity of watering points where trampling of the soil (possibly accompanied by tree die-back) is most severe.

8.6 Summary

(1) Topsoil chemistry and indices of soil surface condition were assessed for different levels of grazing within patches of chenopod, groved mulga and sandy mulga landscapes.

(2) Landscape effects were more influential than grazing on variation in potentially mineralisable nitrogen, available phosphorus, organic carbon, cation exchange capacity, electrical conductivity and indices of infiltration, nutrient cycling and soil stability.

(3) There were strong within-landscape contrasts in topsoil properties, providing convincing evidence in support of a landscape model of resource sources and sinks.

(4) Available phosphorus distribution proved to be anomalous in groved mulga landscapes in that inter-grove areas had higher concentrations than groves. However, the patterns were consistent with previous studies of patterned mulga landscapes. Sequestration of available phosphorus by mulga trees within groves might be the dominant process underlying the patterns.

(5) Those grazing impacts that were significant were generally weak. In sandy mulga landscapes, mineraliasable nitrogen increased with grazing history on sandy banks and in adjacent mulga drainage lanes, while in chenopod landscapes, electrical conductivity increased markedly with grazing in inter-bush areas. Infiltration scores tended to decline slightly with grazing, as did nutrient cycling scores for bush mounds, inter-bush areas, mulga clumps, mulga groves, and inter-groves. Weak increases with grazing occurred in inter-clump areas and sandy banks in sandy mulga landscapes. Similarly weak declines in most patch types with increased grazing were recorded for the soil stability index.

(6) My heuristic interpretation of the most biologically significant effect of grazing within-landscape contrasts is that grazing has the capacity to denude and desertify chenopod landscapes under very heavy levels of grazing because of the salinisation effect and the grazing-induced contraction of bush mound areas and expansion of inter-bush areas.

(7) Mulga landscapes seem far more resistant to grazing, and severe degradation seems limited to disproportionately small areas in the immediate vicinity of watering points, where trampling may be more influential than herbivory.

(8) A significant positive relationship was found between Tongway's index of nutrient cycling and potentially mineralisable nitrogen across landscape types suggesting that the methodology is generalisable across landscape types, at least within this region.

Table 8.1 The hierarchical landscape stratification and sampling effort for the three indices used in soil surface assessment

Landscape type	Sub-type	Land unit	Patch type	Sampling loci	Quadrats ('reps')
Chenopod	-	chenopod plain	bush mound	5	10
			inter-bush	5	10
Mulga	groved mulga	groved mulga plain	mulga grove	5	5
			inter-grove	5	5
	sandy mulga	mulga drainage lane	mulga clump	5	5
			inter-clump	5	5
		sandy bank	open (grassy) area	5	10

Table 8.2 Effects of grazing history on potentially mineralisable nitrogen (See Glossary: Appendix 1 for terminology and Chapter 2 for explanation of modelling terms)

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking History, derived from modelling track densities and expressed in sheep years per hectare	landscape type	2.7	1	n.s.
	SH	0.7	1	n.s.
	stratum.landscape type	268.2	4	< 0.001
	landscape type.SH	0.9	1	n.s.
	stratum.landscape type.SH	7.8	4	n.s.
SR: Stocking Rate; average long term stocking rate developed from RANGEPACK/Paddock model and expressed in sheep per hectare	landscape type	3.3	1	n.s.
	SR	0.0	1	n.s.
	stratum.landscape type	269.2	4	< 0.001
	landscape type.SR	0.9	1	n.s.
	stratum.landscape type.SR	10.6	4	0.03
DW: distance from water, measured in kilometres	landscape type	2.3	1	n.s.
	DW	0.5	1	n.s.
	stratum.landscape type	268.5	4	< 0.001
	landscape type.DW	0.8	1	n.s.
	stratum.landscape type.DW	9.4	4	n.s.

Table 8.3 Effects of grazing history on available phosphorus

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	4.9	1	0.03
	SH	1.3	1	n.s.
	stratum.landscape type	55.9	4	< 0.001
	landscape type.SH	2.2	1	n.s.
	stratum.landscape type.SH	4.0	4	n.s.
SR: Stocking rate	landscape type	5.8	1	0.02
	SR	4.2	1	0.04
	stratum.landscape type	57.5	4	< 0.001
	landscape type.SR	1.5	1	n.s.
	stratum.landscape type.SR	11.6	4	0.02
DW: Distance from water	landscape type	4.8	1	0.03
	DW	2.2	1	n.s.
	stratum.landscape type	55.7	4	< 0.001
	landscape type.DW	0.6	1	n.s.
	stratum.landscape type.DW	3.0	4	n.s.

Table 8.4 Effects of grazing history on (natural log) organic carbon

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	5.1	1	0.02
	SH	7.1	1	0.008
	stratum.landscape type	359.7	4	< 0.001
	landscape type.SH	0.0	1	n.s.
	stratum.landscape type.SH	0.4	4	n.s.
SR: Stocking rate	landscape type	4.5	1	0.03
	SR	1.7	1	n.s.
	stratum.landscape type	359.8	4	< 0.001
	landscape type.SR	0.0	1	n.s.
	stratum.landscape type.SR	0.7	4	n.s.
DW: Distance from water	landscape type	4.2	1	0.04
	DW	1.2	1	n.s.
	stratum.landscape type	362.0	4	< 0.001
	landscape type.DW	1.2	1	n.s.
	stratum.landscape type.DW	2.8	4	n.s.

Table 8.5 Effects of grazing history on cation exchange capacity

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	4.2	1	n.s.
	SH	1.0	1	n.s.
	stratum.landscape type	99.5	4	< 0.001
	landscape type.SH	1.3	1	n.s.
	stratum.landscape type.SH	5.1	4	n.s.
SR: Stocking rate	landscape type	3.9	1	0.045
	SR	1.9	1	n.s.
	stratum.landscape type	99.3	4	< 0.001
	landscape type.SR	1.0	1	n.s.
	stratum.landscape type.SR	6.9	4	n.s.
DW: Distance from water	landscape type	4.1	1	0.04
	DW	0.8	1	n.s.
	stratum.landscape type	100.7	4	< 0.001
	landscape type.DW	2.1	1	n.s.
	stratum.landscape type.DW	9.0	4	n.s.

Table 8.6 Effects of grazing history on (inverse) electrical conductivity

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	58.8	1	< 0.001
	SH	5.8	1	0.02
	stratum.landscape type	277.5	4	< 0.001
	landscape type.SH	0.7	1	n.s.
	stratum.landscape type.SH	15.4	4	0.004
SR: Stocking rate	landscape type	76.4	1	< 0.001
	SR	7.4	1	0.007
	stratum.landscape type	272.4	4	< 0.001
	landscape type.SR	0.0	1	n.s.
	stratum.landscape type.SR	6.4	4	n.s.
DW: Distance from water	landscape type	48.3	1	< 0.001
	DW	0.3	1	n.s.
	stratum.landscape type	261.0	1	< 0.001
	landscape type.DW	0.6	1	n.s.
	stratum.landscape type.DW	0.9	1	n.s.

Table 8.7 Effects of grazing history on infiltration index

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	158.8	1	< 0.001
	SH	213.1	1	< 0.001
	landscape type.unit	81.8	2	< 0.001
	landscape type.SH	0.4	1	n.s.
	landscape type.unit.patch	6326.8	3	< 0.001
	landscape type.unit.SH	0.1	2	n.s.
	landscape type.unit.patch.SH	4.5	3	n.s.
SR: Stocking rate	landscape type	167.0	1	< 0.001
	SR	229.1	1	< 0.001
	landscape type.unit	124.2	2	< 0.001
	landscape type.SR	8.8	1	0.003
	landscape type.unit.patch	6654.7	3	< 0.001
	landscape type.unit.SR	7.1	2	0.03
	landscape type.unit.patch.SR	38.5	3	< 0.001
DW: Distance from water	landscape type	161.1	1	< 0.001
	DW	103.5	1	< 0.001
	landscape type.unit	100.4	2	< 0.001
	landscape type. DW	34.3	1	< 0.001
	landscape type.unit.patch	6564.3	3	< 0.001
	landscape type.unit. DW	1.4	2	n.s.
	landscape type.unit.patch. DW	32.1	3	< 0.001

Table 8.8 Effects of grazing history on nutrient cycling index

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	59.6	1	< 0.001
	SH	110.7	1	< 0.001
	landscape type.unit	556.9	2	< 0.001
	landscape type.SH	1.0	1	n.s.
	landscape type.unit.patch	2960.1	3	< 0.001
	landscape type.unit.SH	9.7	2	0.008
	landscape type.unit.patch.SH	36.2	3	< 0.001
SR: Stocking rate	landscape type	61.5	1	< 0.001
	SR	74.9	1	< 0.001
	landscape type.unit	610.3	2	< 0.001
	landscape type.SR	15.6	1	< 0.001
	landscape type.unit.patch	3035.4	3	< 0.001
	landscape type.unit.SR	20.3	2	< 0.001
	landscape type.unit.patch.SR	54.7	3	< 0.001
DW: Distance from water	landscape type	61.3	1	< 0.001
	DW	44.9	1	< 0.001
	landscape type.unit	597.2	2	< 0.001
	landscape type. DW	49.2	1	< 0.001
	landscape type.unit.patch	3047.2	3	< 0.001
	landscape type.unit. DW	4.3	2	n.s.
	landscape type.unit.patch. DW	54.9	3	< 0.001

Table 8.9 Effects of grazing history on stability index

Grazing model	Term	Wald statistic	d.f.	p-value
SH: Stocking history	landscape type	134.1	1	< 0.001
	SH	297.0	1	< 0.001
	landscape type.unit	625.8	2	< 0.001
	landscape type.SH	24.6	1	< 0.001
	landscape type.unit.patch	2516.2	3	< 0.001
	landscape type.unit.SH	1.7	2	n.s.
	landscape type.unit.patch.SH	28.8	3	< 0.001
SR: Stocking rate	landscape type	135.0	1	< 0.001
	SR	225.4	1	< 0.001
	landscape type.unit	713.0	2	< 0.001
	landscape type.SR	0.1	1	n.s.
	landscape type.unit.patch	2532.6	3	< 0.001
	landscape type.unit.SR	2.4	2	n.s.
	landscape type.unit.patch.SR	36.6	3	< 0.001
DW: Distance from water	landscape type	134.9	1	< 0.001
	DW	225.7	1	< 0.001
	landscape type.unit	684.7	2	< 0.001
	landscape type. DW	16.1	1	< 0.001
	landscape type.unit.patch	2571.2	3	< 0.001
	landscape type.unit. DW	3.1	2	n.s.
	landscape type.unit.patch. DW	14.8	3	0.002

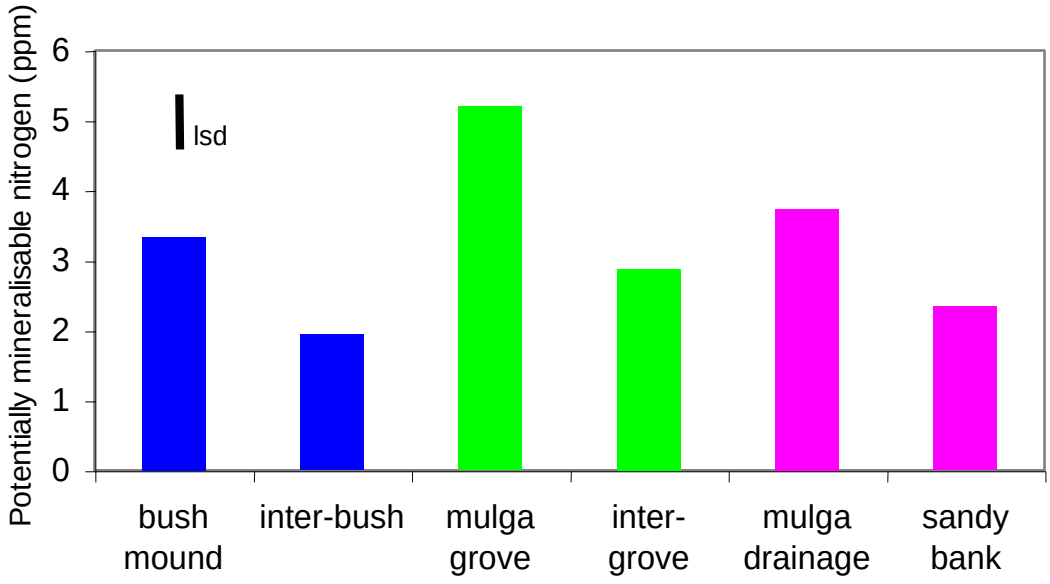


Figure 8.1 Variation in potentially mineralisable nitrogen with landscape stratum.

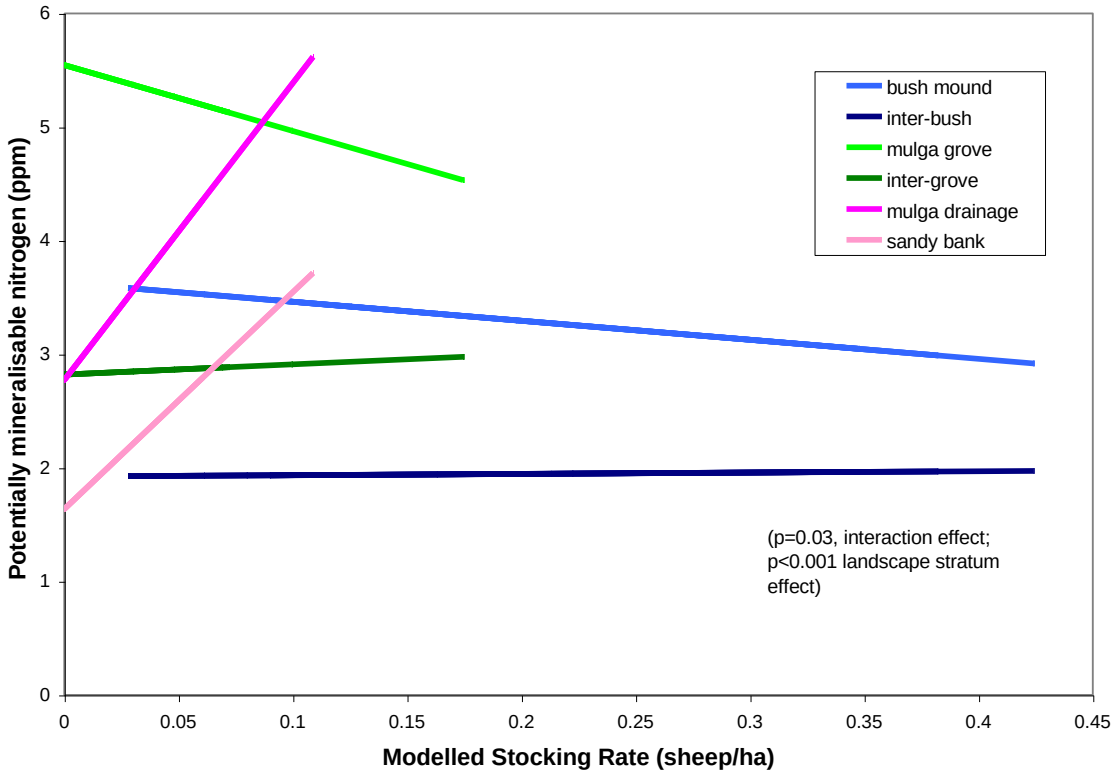


Figure 8.2 Impacts on stocking rate on potentially mineralisable nitrogen.

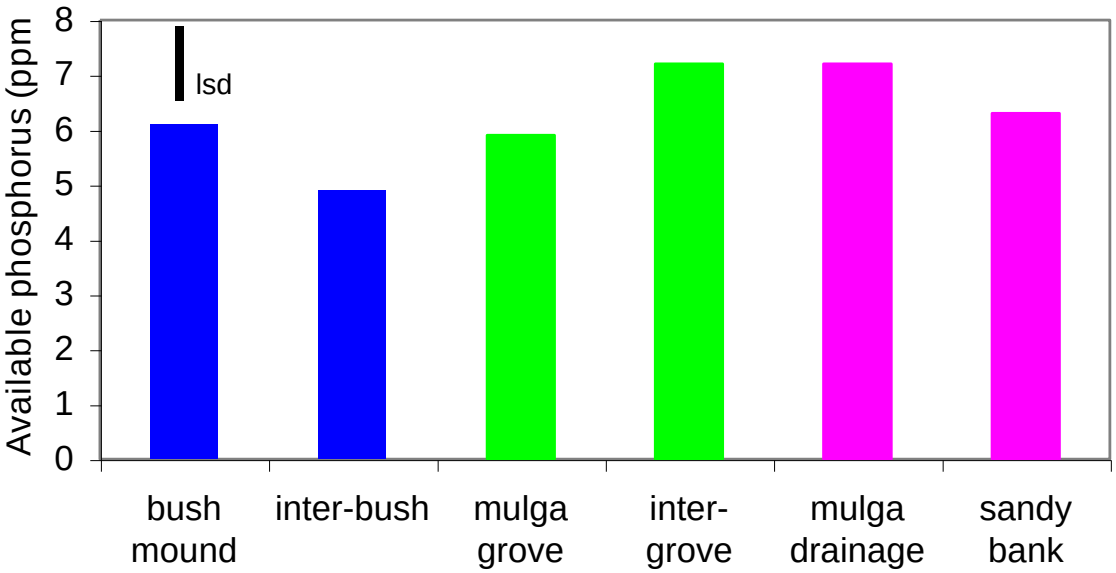


Figure 8.3 Variation in available phosphorus with landscape stratum.

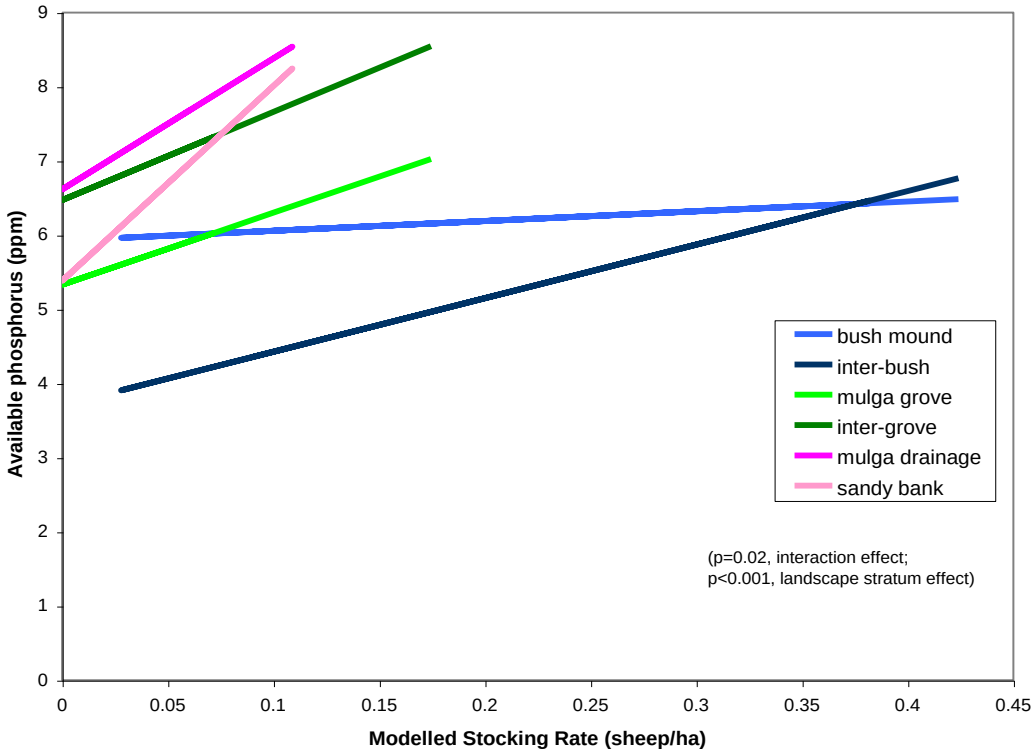


Figure 8.4 Impacts of stocking rate on available phosphorus.

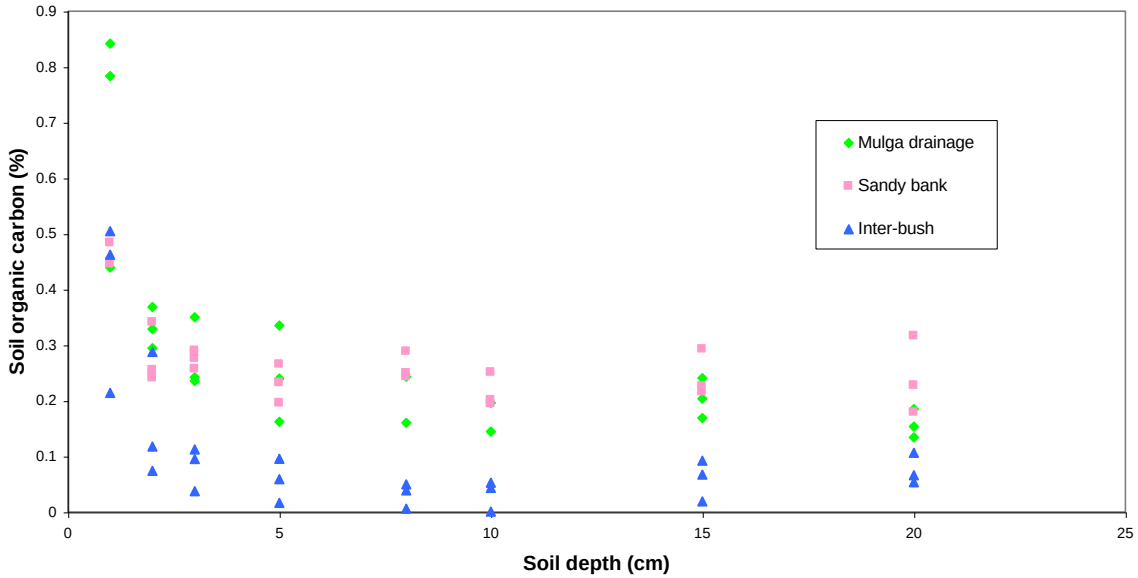


Figure 8.5 Variation in total organic carbon with soil depth.

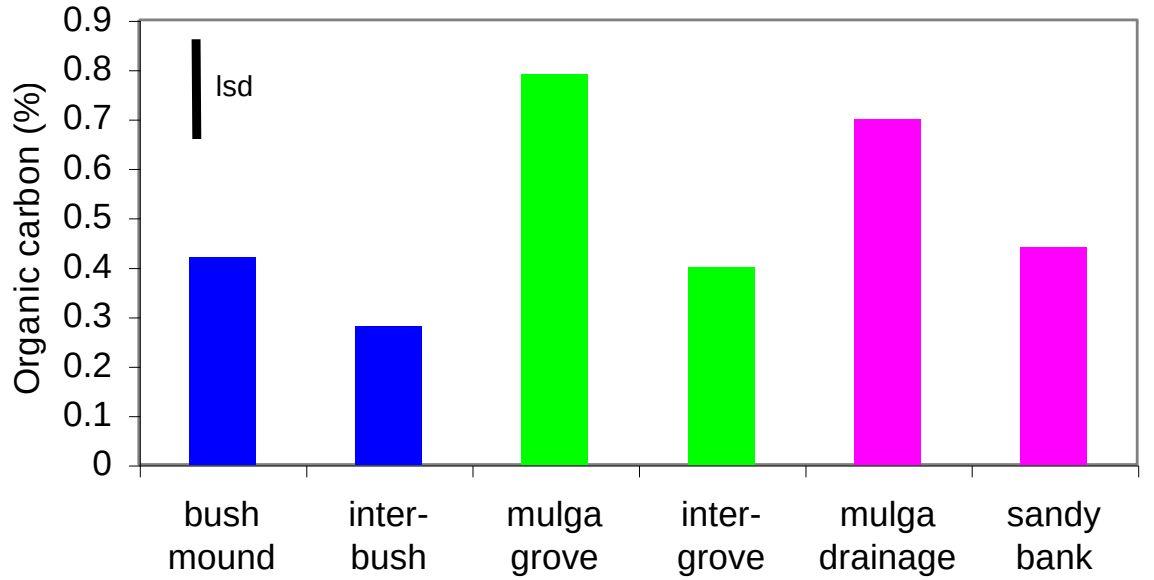


Figure 8.6 Variation in organic carbon with landscape stratum.

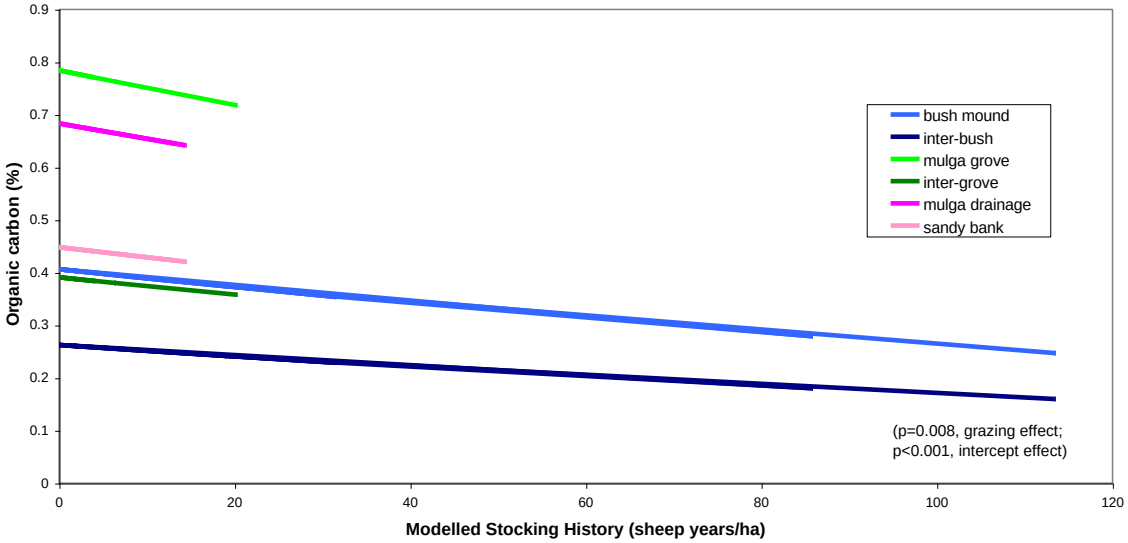


Figure 8.7 Impacts of stocking history on organic carbon.

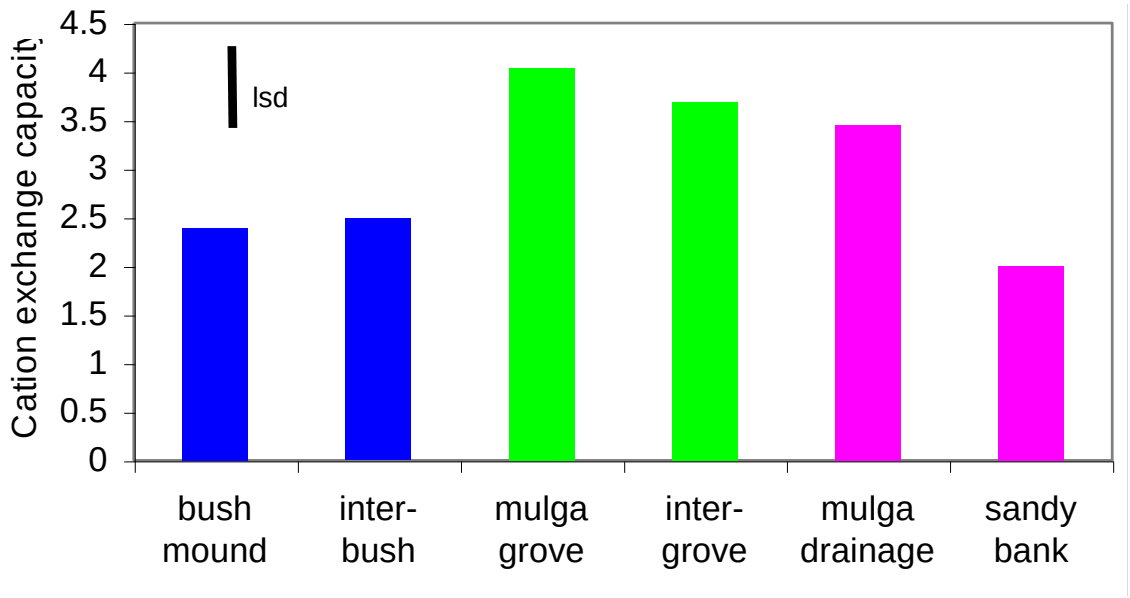


Figure 8.8 Variation in cation exchange capacity with landscape stratum.

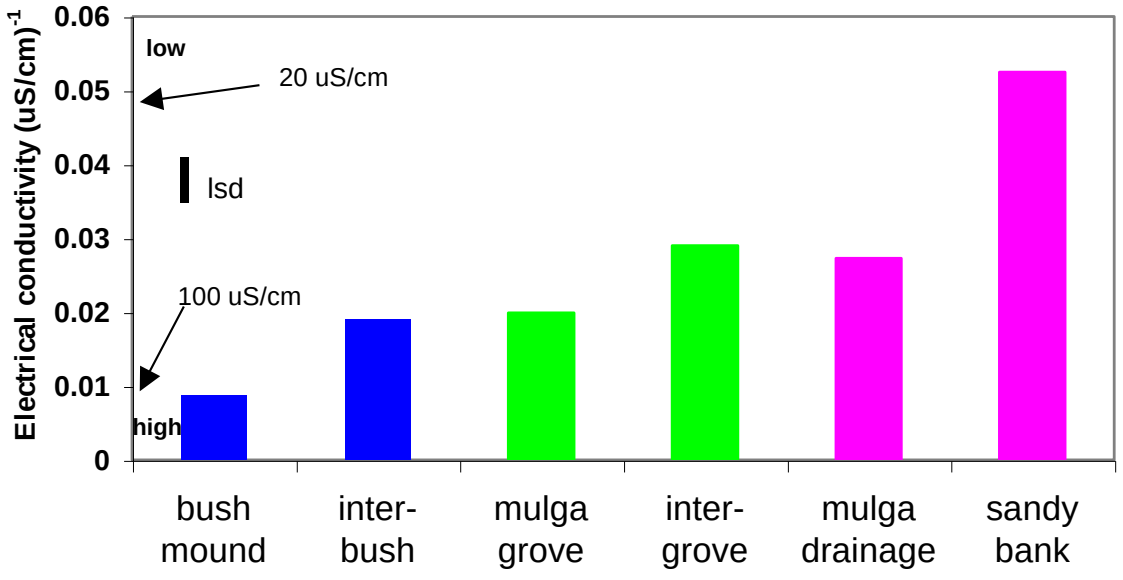


Figure 8.9 Variation in electrical conductivity with landscape stratum.

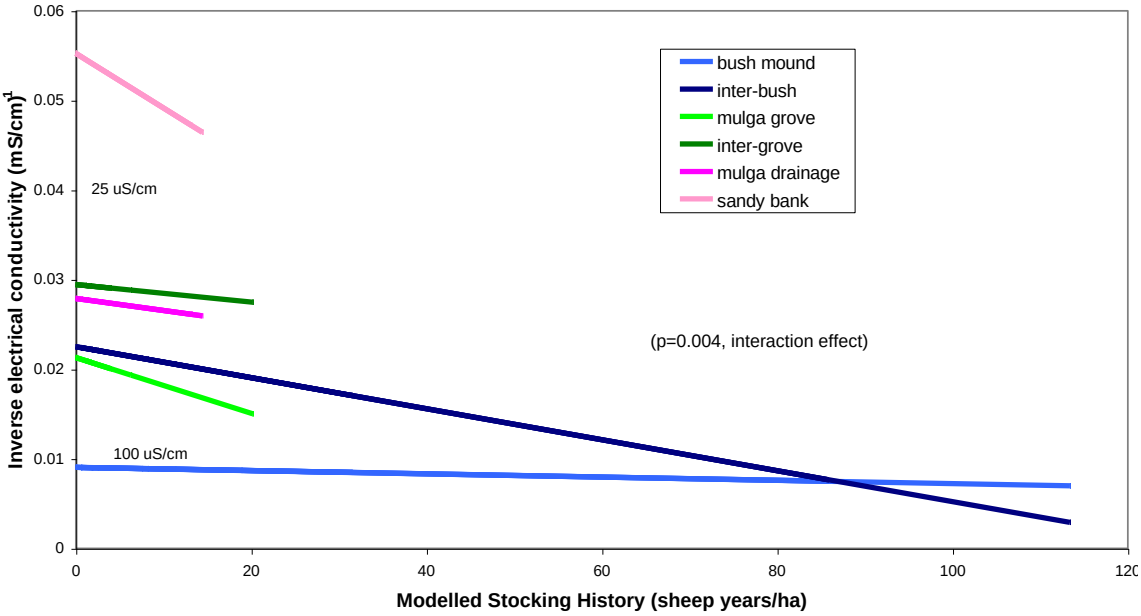


Figure 8.10 Impacts of stocking history on electrical conductivity.

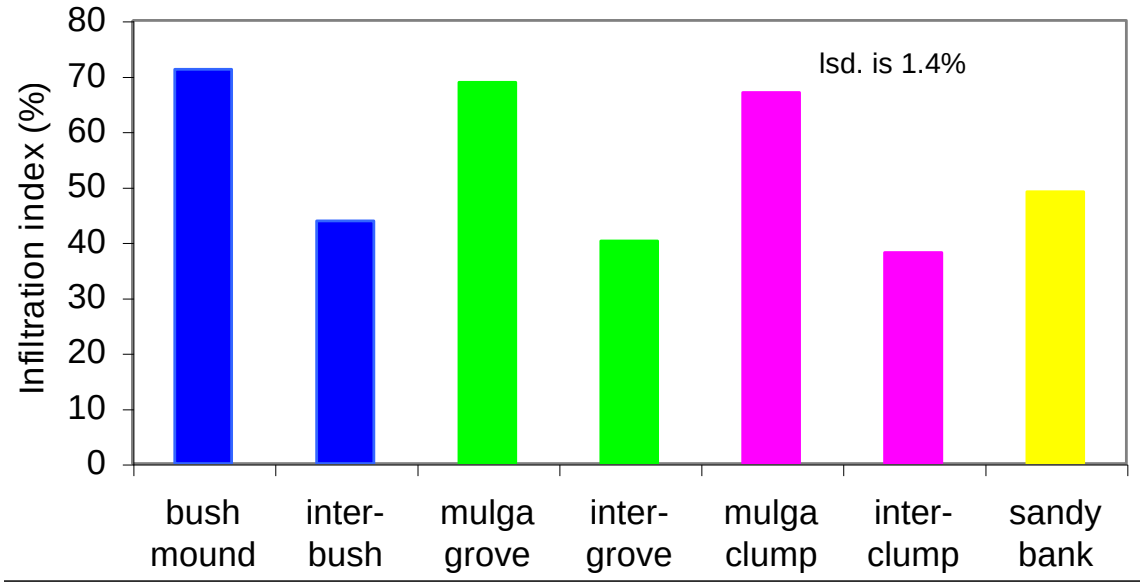


Figure 8.11 Variation in infiltration index with patch type.

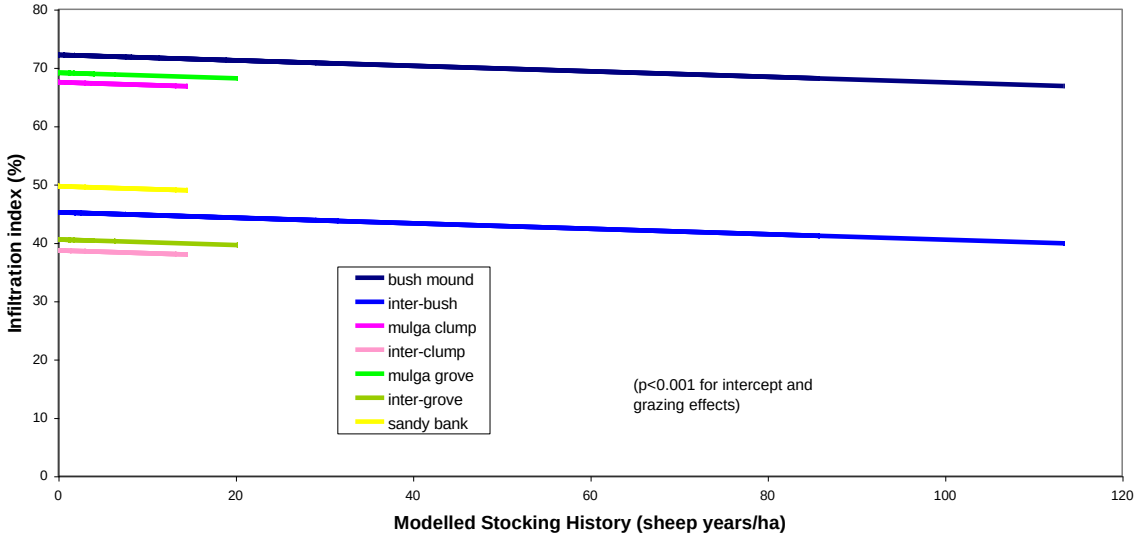


Figure 8.12 Impacts of stocking history on infiltration index.

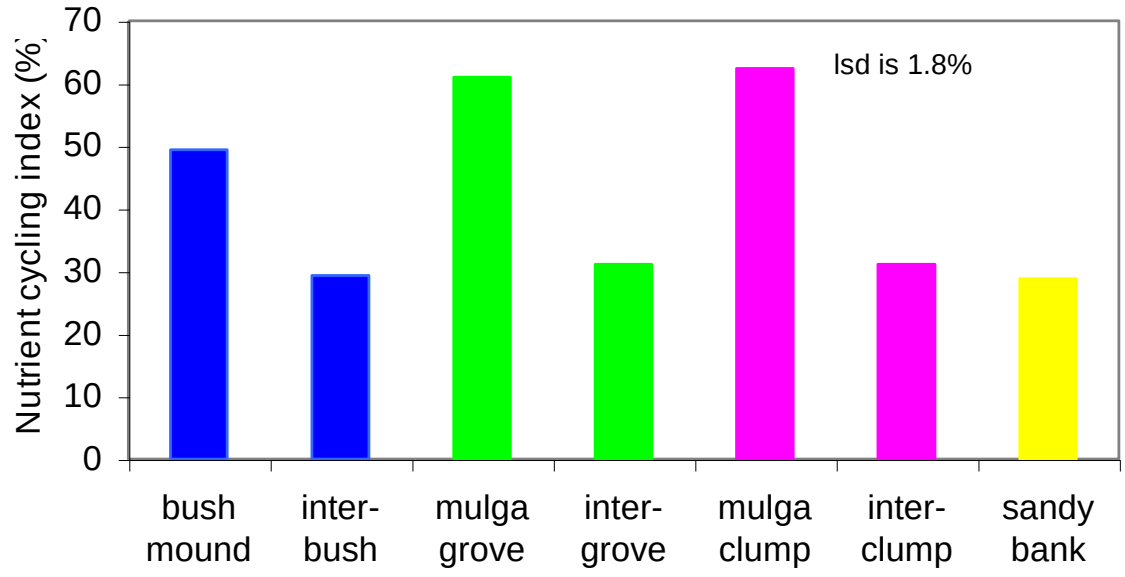


Figure 8.13 Variation in nutrient cycling index with patch type.

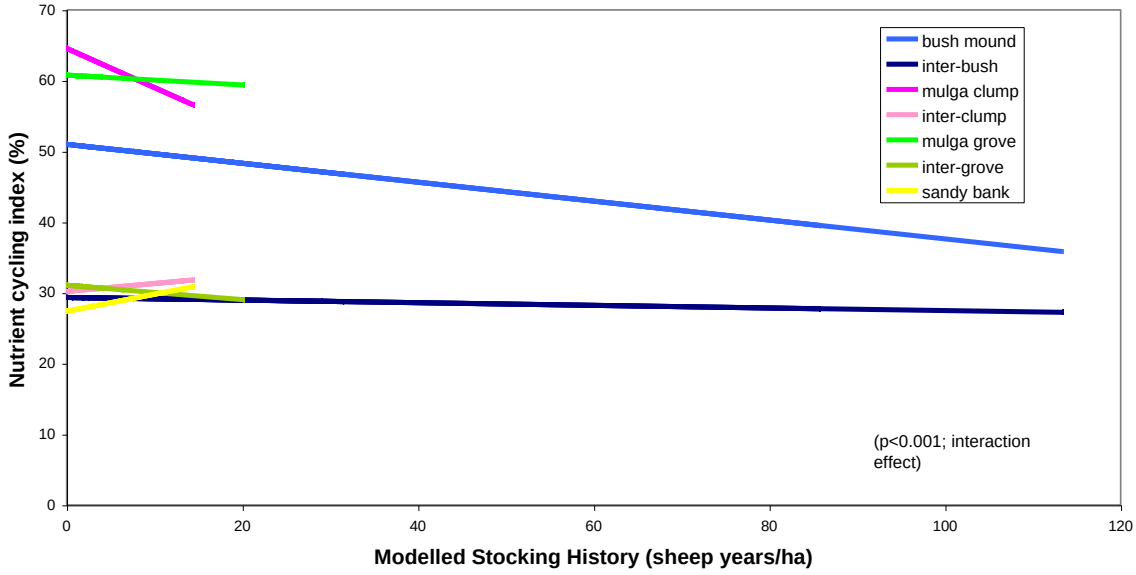


Figure 8.14 Impacts of stocking history on nutrient cycling index.

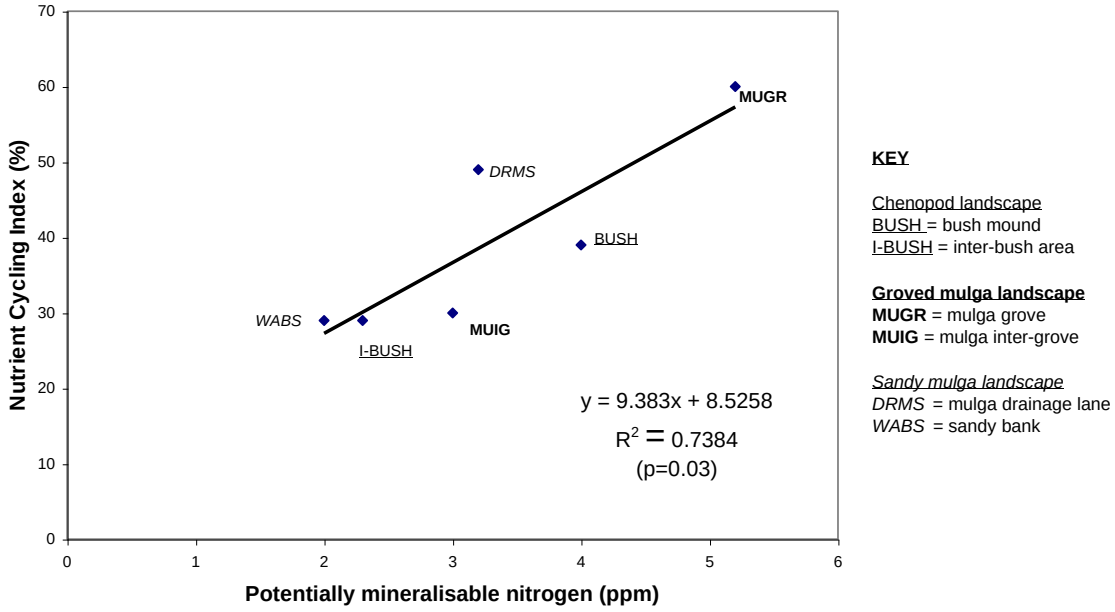


Figure 8.15 Comparison of nutrient cycling index with nitrogen.

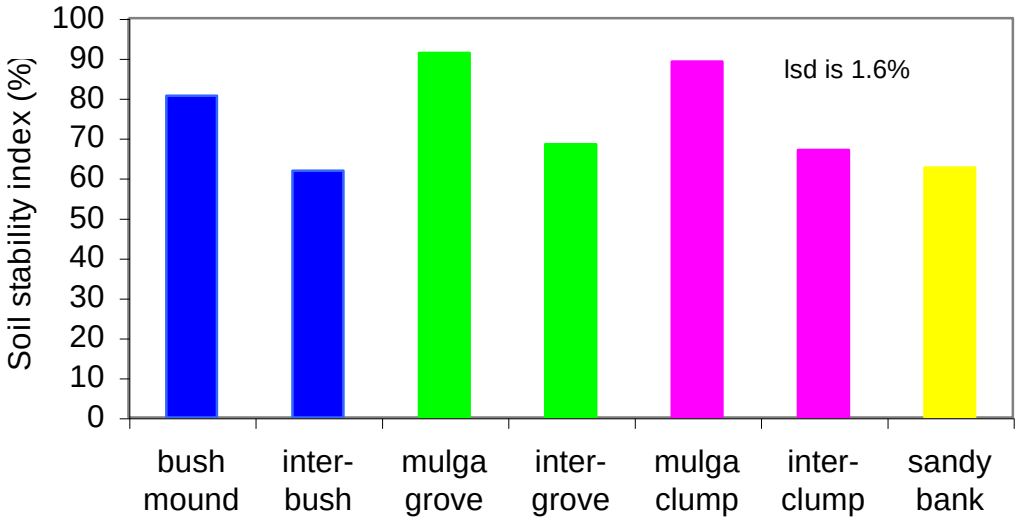


Figure 8.16 Variation in soil stability index with patch type.

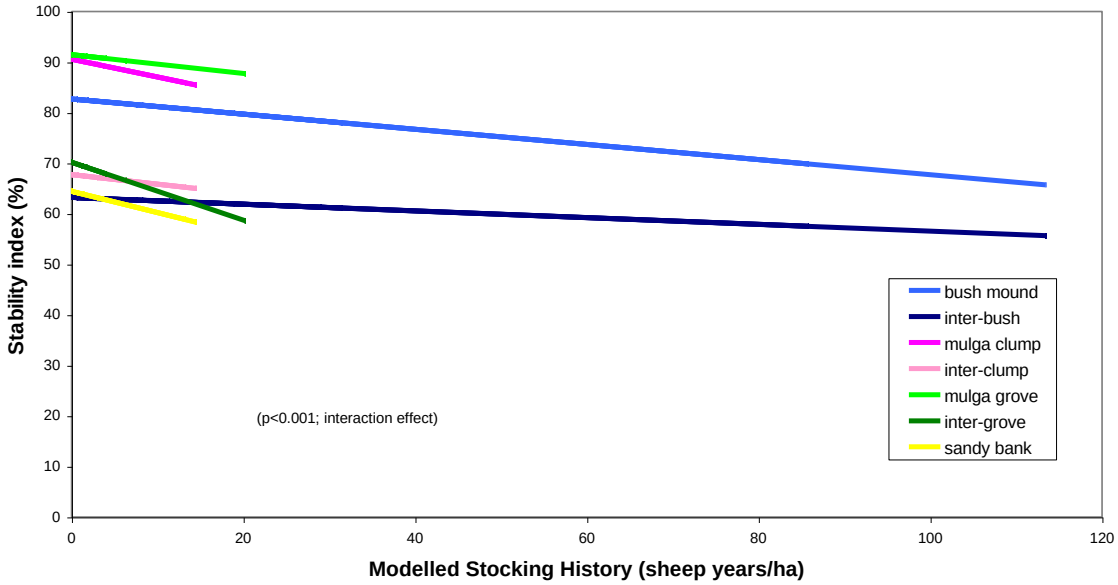


Figure 8.17 Impacts of stocking history on soil stability index.

CHAPTER 9. SYNTHESIS

9.1 Scope of this chapter

In this chapter I return to my original questions:

(1) Does grazing impact differently when viewed from the various key management perspectives?

(2) Does the impact of grazing from a particular management perspective differ according to type of landscape?

(3) Does type of landscape affect the interrelationships between management perspectives and grazing?

To address these questions, I summarise briefly major findings according to:

(A) key management perspectives of sustainable pastoralism (range condition), nature conservation (biodiversity) and natural resource management (ecological integrity),

and

(B) type of landscape. I then discuss the results within types of landscape as an ecological synthesis, before turning to the implications of the findings for future assessment of the 'health' (in a metaphorical sense) of arid rangelands.

To do this I firstly evaluate the performance of traditional range condition assessment, not only in its intended context of informing pastoral managers about livestock carrying capacity, but also from the broader perspectives of nature conservation and natural resource management. I then consider how site-based assessment might be improved beyond range condition. Finally, I consider my primary findings in a spatial context and in doing so develop a new framework for assessing impacts of grazing from different management perspectives. For this I adopt the 'pressure-state-response' approach used in state of the environment reporting around the world.

9.2 Summary of grazing impacts among key perspectives

9.2.1 Viewpoint of sustainable pastoralism

- Range Condition declined with increased grazing, more rapidly in mulga landscapes.
- Sustainable Carrying Capacity declined with increased grazing, most severely in henopod landscapes.
- Pastoral Integrity declined with increased grazing, more easily in mulga landscapes.

Summary: Grazing had a deleterious impact on sustainable pastoralism, but the magnitude of the effects differed between types of landscape and the attribute considered. The long-term carrying capacity (as estimated using Sustainable Carrying Capacity) was more severely depleted by grazing in chenopod landscapes.

9.2.2 Viewpoint of nature conservation

9.2.2.1 Floristic diversity

- Exotic species were more common in chenopod landscapes but appeared to be a minor problem only
- Species density (spp/m^2) declined with increasing grazing, particularly in mulga landscapes
- Species richness (spp/sampling locus) increased with distance from water.
- There were more decreaser than increaser species (i.e., more 'losers' than 'winners'), particularly in chenopod and sandy mulga landscapes.
- Decreaser species were lost more easily from mulga landscapes than chenopod landscapes with increasing grazing.
- Increaser species responded more readily to grazing in mulga landscapes, particularly groved mulga landscapes.

Summary: Grazing had a deleterious impact on most aspects of floristic diversity from the viewpoint of nature conservation, generally more so in mulga landscapes than chenopod landscapes (though impacts in chenopod landscapes were still substantial).

9.2.2.2 Ant diversity

- All species were native.
- Total abundance increased with increasing grazing, mostly in sandy mulga landscapes.
- Species richness (spp/sampling locus) decreased with increasing grazing (largest dynamic range in chenopod landscapes)
- Decreaser species outnumbered increaser species in chenopod landscapes
- Increasers outnumbered decreasers in mulga landscapes (but numbers were small compared to the number of chenopod decreasers)
- Declines in decreaser species with increasing grazing were much more prominent in chenopod landscapes than mulga landscapes
- Increases in increaser species with increased grazing were minor

Summary: Grazing had a deleterious affect on ant diversity from a viewpoint of nature conservation in chenopod landscapes, but impacts in mulga landscapes were minor in comparison.

9.2.3 Viewpoint of natural resource management

9.2.3.1 Landscape patterns

- The size and area covered by fertile patches declined with increasing grazing, particularly in chenopod landscapes

Summary: Grazing has a deleterious effect on landscape patterning in chenopod landscapes.

9.2.3.2 Redistribution processes

- Sandy banks in sandy mulga landscapes appeared to lose sediments with increasing erosion, as did bush mounds in chenopod landscapes and mulga groves in groved mulga landscapes (but far less strongly in the latter cases)
- There was no evidence of net sediment loss with increased grazing from landscapes (at least since atomic testing commenced in the 1950s)

Summary: Grazing appeared to suppress processes that concentrate and conserve resources in fertile patches in parts of all landscapes, but did not appear to cause loss of soil from the system.

9.2.3.4 Topsoil chemistry

- Potentially mineralisable nitrogen declined slightly with increasing grazing in bush mounds in chenopod landscapes and mulga groves in groved mulga landscapes, but increased in sandy mulga landscapes.
- Available phosphorus generally increased with increasing grazing, particularly in sandy mulga landscapes.
- There were minor decreases in organic carbon with increasing grazing.
- Electrical conductivity (indicative of salinity) increased with increasing grazing, most profoundly in (expanding) inter-bush areas in chenopod landscapes.

Summary: Grazing had mixed effects on topsoil properties from the viewpoint of natural resource management: salinity problems occurred in chenopod landscapes, soil erosion appeared to have occurred on sandy banks of sandy mulga landscapes and, nutrient availability increased in sandy mulga landscapes.

9.2.3.5 Soil surface assessment

- Increased grazing generally led to minor declines in all three indices of soil condition: infiltration, nutrient cycling and soil stability.

Summary: Grazing had slight negative impacts on soil surface condition.

9.2.4 Synthesis

Grazing generally had deleterious impacts from all three perspectives. This suggests that grazing reduces the capacity of the land to support livestock, that many native species have not adjusted well to pastoralism and that pastoral landscapes tend to lose the patterns and processes that maintain ecosystems in this arid region. These findings of the impacts of grazing are consistent with the introduction of pastoralism to a region in which perennial vegetation plays a major role in maintaining ecosystem patterns and processes, but is not co-adapted to pastoral grazing regimes. Such a lack of co-adaptation may have arisen because of the contrast between evolutionary and recent grazing regimes (Milchunas, Sala and Lauenroth 1988; Milchunas and Lauenroth 1993; Landsberg, O'Connor and Freudenberger 1999).

There appear to be more ‘losers’ than ‘winners’ among native plant and ant species, which is emerging as a globally recognisable trend associated with the (evolutionarily) recent introduction of anthropogenic disturbance (McKinney and Lockwood 1999). In this arid, marginal pastoral region, declines in aspects of plant diversity probably reflect direct impacts

of herbivory at least as much as alteration of competitive interactions (Hacker 1984a; Archer and Smiens 1991). Declines in aspects of ant diversity probably reflect the impacts of grazing on vegetation *per se* and the indirect effects of alteration to vegetation structure and cover that flow on to landscape patterns and process (particularly effects on the soil as ant habitat).

Over a third of ant species in chenopod landscapes were classified as decreaser species (i.e. species whose abundance declined with proximity to watering points). In contrast, over 70 per cent of species in mulga landscapes were classified as 'neutral'. There was generally more increaser than decreaser species in mulga landscapes. The contrasts in grazing impacts between mulga and chenopod landscapes beg a new model of grazing-ant community interactions that differs considerably from a model of little interaction derived principally in North America. (Heske and Campbell 1991; Whitford *et al.* 1999). Major modification of the physical environment occurs in chenopod shrublands, including loss of fine-scale pattern and increased topsoil salinity. Since most decreasers occurred in chenopod landscapes I hypothesize that ants are sensitive to modification of their physical habitat. In less physically and chemically altered landscapes one would expect most ants to be tolerant of changes to vegetation, but some ants to be favoured by minor changes to their physical environment. This is consistent with the mulga pattern. Under this model, ants could be regarded as good indicators of changes in the edaphic environment (Andersen 1990), although they are not as sensitive as plants to grazing *per se*.

These results do not necessarily mean that grazing will inevitably lead to degradation of entire rangeland landscapes, irrespective of what key management perspective is used in assessment. The degree to which different parts of pastoral landscapes are degraded needs to be addressed by spatial modelling of these results, since heavily impacted (i.e. heavily grazed) areas may cover relatively small parts of paddocks (Cridland and Stafford Smith 1993) (see section 9.5).

9.3 Summary of grazing impacts on each type of landscape

9.3.1 General findings

While there were few contradictory findings among types of landscapes for the various attributes assessed, there were frequently strong differences in the strengths of effects of grazing on different landscape types. I briefly summarise major differences below:

- Sustainable Carrying Capacity was substantially affected in both landscapes, but more strongly in chenopod landscapes than mulga landscapes.
- Floristic diversity was substantially affected in both landscapes, but species declined more readily in mulga landscapes.
- Ant diversity was most substantially affected in chenopod landscapes.
- Loss of fertile patches was severe in chenopod landscapes.
- Soil erosion affected sandy banks most noticeably.
- Potentially mineralisable nitrogen and available phosphorus were enhanced in sandy mulga landscapes.
- Topsoil salinity increased most notably in inter-bush areas in chenopod landscapes.

Only in the case of potentially mineralisable nitrogen was there a clear contradiction in the effect of grazing, which was associated with slight declines in mineralisable nitrogen in mulga groves and bush mounds, but increases in sandy mulga landscapes.

9.3.2 Synthesis for chenopod landscapes

Chenopod landscapes are predominantly wind-modulated systems consisting of individual or small clusters of shrubs with sandy accumulations underneath them ('bush mounds'). Nutrient concentrations and cycling, and infiltration are greatest on these mounds, which therefore function as fine-scale fertile patches (Charley and Cowling 1968; Charley 1972; Sharma and Tongway 1973; Charley and West 1975; Graetz and Tongway 1986; Greene and Tongway 1987).

The major impacts of grazing on chenopod landscapes were all deleterious regardless of the perspectives of management applied. The chenopod landscapes I have studied appear to collapse under heavy grazing .The major impacts of grazing included:

- declines in range condition and Sustainable Carrying Capacity
- declines in floristic and ant diversity
- loss of fertile patches (shrub cover)
- expansion of expanded open areas
- salinisation of open areas

If the proportional areas occupied by strata within a landscape are responsive to grazing and the mean values for an attribute (response variable) differ substantially between strata, then the whole landscape is likely to be responsive to grazing for that attribute whether or not there is any individual response within each stratum (Noy-Meir 1981; Tongway and Hindley 2000). Thus the impacts of grazing on shrubs and their bush mounds mean that grazing affects these chenopod rangelands at the whole landscape level, even though local grazing effects were minor for several variables. For instance, levels of potentially mineralisable nitrogen were considerably higher in bush mounds than in open areas. Combine these differences with the decline in 'fertile' bush mounds and expansion of 'infertile' open areas and one can heuristically interpret another level of grazing impacts. Bush mounds and inter-bush mounds also had contrasting high and low scores for the three indices of landscape processes: infiltration, nutrient cycling and soil stability. Since grazing reduced the proportion of the landscape occupied by bush mounds it most probably reduced the rates of infiltration and nutrient cycling, and soil stability across the whole landscape.

The decline of bush mounds and expansion of open areas is also probably important in terms of heterogeneity in habitat structure for fauna (Morton, James and Landsberg 1996), though I did not assess this directly. Thus, chenopod shrubs could be considered 'ecosystem engineers'(Jones, Lawton and Shachak 1994; Chapin *et al.* 1997) in two contexts; firstly in modulating physical landscape processes (particularly wind) and also in structuring habitat for a range of other organisms.

Chenopod shrubs are also particularly affected by grazing. Most shrubs are within grazing access (i.e. they rarely grow beyond the reach of sheep) and are readily grazed by stock, particularly in dry times (Mitchell and Wilcox 1984). The combination of grazing sensitivity of chenopod shrubs and their importance as ecosystem engineers predispose chenopod landscapes to degradation under heavy grazing. Compounding this is the preference sheep have for chenopod landscapes over others in paddocks that contain a mixture of landscape types (Burnside *et al.* 1995; Pringle and Riches 1996). Thus it was not surprising that I found the most severe impacts of grazing in chenopod landscapes.

I summarise these aspects of patterns and processes in chenopod landscapes and the effects of grazing on them in conceptual models (see Figures 9.1 and 9.2). The two conceptual models are based on my findings and previous research (Sharma and Tongway 1973; Hacker 1979; Graetz and Tongway 1986; Andrew and Lange 1986a; 1986b; Greene and Tongway 1987; Pringle 1994b; 1998b; Burnside *et al.* 1995).

Degradation of chenopod landscapes under heavy grazing follows my 'Denudation and loss of productivity' model (4B) in Chapter 7 (Figure 7.1). There are feedbacks that may tend to drive the degradation process once it commences (Thurow, Blackburn and Taylor 1988; Schlesinger *et al.* 1990; Milton *et al.* 1994). Mortality in chenopod shrubs is closely correlated with loss of size (Watson 1997), which in turn is associated with heavy grazing (and senescence). As shrubs are removed by grazing (assuming that grazing is not reduced), grazing pressure during dry times is concentrated on fewer and fewer shrubs, further reducing their likelihood of survival. This process of degradation is unlikely to be continuous. It is more likely to accelerate in prolonged dry times, when there is a convergence of physiological stress among perennial shrubs and increased grazing as ephemeral forage declines (Westoby 1980; Westoby, Walker and Noy-Meir 1989a; b; Watson, Westoby and Holm 1997a).

The degradation process is likely to be expressed spatio-temporally, with a spreading of impact away from watering points as sheep eliminate shrubs at successively greater distances from water (Noble 1979; Andrew and Lange 1986a; 1986b; Hunt 1995; submitted). Assuming similar numbers of sheep are run at watering points during this processes, one would expect two scales of expansion of degradation, reflecting different time-scales of the controlling processes. Over a number of seasons, one would expect expansion of the zone of degradation away from water to be at its maximum during and soon after prolonged dry spells, as already discussed. However, over longer time-scales (years to decades), one would expect the rate of expansion of the zone of degradation to decelerate as the area encapsulated at greater distances from water increases. Over these sorts of time spans, it is also likely that shrubs will have a chance of recovery between periods of heavy grazing. Therefore, the chance of survival further from water increases individually and collectively.

Thus the capacity for sheep to degraded areas at successively greater distances from watering points is physiologically constrained. The capacity of sheep to spend long periods grazing far from water is limited by factors such as heat balances and the need to drink regularly to expel salt ingested in halophytic vegetation (Dudzinski, Pahl and Arnold 1969; Squires and Wilson 1971; Wilson 1974; Arnold and Dudzinski 1978; Stafford Smith, Noble and Jones 1985; Stafford Smith 1988; Cridland and Stafford Smith 1993).

The profound changes to chenopod landscapes wrought by heavy grazing in this study beg a further question. Is extensive degradation of chenopod landscapes inevitable under pastoral management? Such claims have been made (e.g. Noy-Meir 1996). However, much of the developed chenopod landscapes in my study region may escape degradation. Paddocks are very large, often exceeding 6 000 hectares and with watering points generally 8 kilometres

apart or more. To degrade whole landscapes in this context would require the extraordinary stocking levels that characterised pastoral management in its formative years (Burnside 1979; Faithful 1994). However, these stocking rates are now acknowledged as being inappropriate by pastoralists (North Eastern Goldfields and Kalgoorlie Land Conservation Districts 1993; Pringle and Riches 1996).

Certainly, spreading watering points and grazing (Noy-Meir's key criticism of early 1990s trends in Australian range management) is likely to increase the aerial extent of chenopod landscapes that it is physiologically possible for sheep to degrade. Recent progress in remote sensing may allow any such ongoing degradation to be revealed (Hannan *et al.* 1991; Hellden 1991; Pickup, Bastin and Chewings 1998; Eve, Whitford and Havstad 1999).

9.3.3 Synthesis for mulga landscapes

9.3.3.1 Groved mulga landscapes

The major impact of grazing on groved mulga landscapes appears to occur in the groves, rather than in adjacent open areas. The major impacts include:

- rapid decline in range condition, Sustainable Carrying Capacity and Pastoral Integrity;
- decline in numbers of decreaser plant species; and
- increase in numbers of increaser plant species.

These impacts are manifested in the removal of palatable shrubs and their replacement by ‘woody weeds’ (*Eremophila gilesii* and *E. spectabilis*).

Impacts on landscape patterns and processes appear minor, and are reflected in relatively minor effects on ant communities. There are more ant species apparently favoured by increased grazing than species that decrease. This applies to both groves and inter-groves.

9.3.3.2 Sandy mulga landscapes

Grazing impacts in sandy mulga landscapes included some trends that were similar in both sandy banks and drainage lanes, and some that were contrasting. The major trends that were consistent as grazing increased were:

- increase in ant abundance;
- decline in range condition, Sustainable Carrying Capacity and Pastoral Integrity;
- decline in plant species density;
- decline in richness of decreaser plant species;

- increase in richness of increaser plant species; and
- increase in potentially mineralisable nitrogen and available phosphorus.

The most obvious contrast between strata was in Caesium-137 enrichment, with grazing being associated with significant losses from sandy banks and an insignificant increase in mulga drainage lanes. To some extent sandy banks appear similar to chenopod landscapes. They are finely patterned (individual or small clumps of grass butts) and resource redistribution is controlled locally, by grasses (Tongway and Ludwig 1994; Schlesinger *et al.* 1996). This means that sheep 'eat the architecture', predisposing the soil to wind erosion. Removal of perennial grasses can lead to invasion of unpalatable woody species in these sandy banks (Curry *et al.* 1994; Pringle 1994b). Shrubs could provide some protection against wind erosion, but shrub invasion had not occurred to any great extent at my study sites.

9.3.3.3 Mulga synthesis

In contrast to chenopod landscapes, mulga landscapes appear grazing-tolerant in many respects. Adverse impacts on landscape patterns and processes are relatively minor and the influence of grazing as measured by range condition may over-estimate the extent of lost pastoral productivity. Healthy soils support abundant ephemeral plant growth in good seasons, and this is the major contributor to pastoral production in these landscapes. In this respect, self-organising landscape processes (e.g. the formation of groves) predominate, rather than the external influence of grazing. This is likely to reduce the influence of historical dependency (grazing history) on contemporary ecosystem patterns and processes (Levin 1998).

The major impact of grazing in mulga landscapes is loss of plant diversity. It appears that substantial proportions of the plants in these landscapes are not well adapted to grazing. In terms of vital attributes (*sensu* Noble and Slatyer 1980); I predict that many of the poorly adapted species will be perennial understorey species that are preferentially grazed and unfit competitors under grazing. The strong correlation of range condition assessments (based on perennial grass and shrub species) with grazing history is consistent with this hypothesis. However, the decline in range condition is probably of less concern to pastoral managers in this 'seasonal country' than is the decline in grazing-sensitive species to nature conservationists. This is not a simple win-win situation; there may well be considerable underlying conflict between contemporary patterns of land use and the maintenance of regional biodiversity.

The decline in grazing-sensitive perennial plant species with increased grazing history indicates that, while pastoralists may have recognised the predominantly seasonal nature of forage availability in these landscapes, they have not adjusted their management in response to it. In other words, they appear not to have reduced stocking rates in poor seasons. Reduced workforces and declining terms of trade mean that it will become increasingly less practical and economically feasible for pastoralists to undertake this degree of tactical management in the future. Therefore, persistence of palatable perennial species in mulga landscapes may be increasingly threatened by pastoral management strategies that cannot afford to alter stocking rates between annual musters for shearing. Loss of sufficient perennial species may ultimately translate into more severe impacts on landscape patterns and processes.

9.4 Evaluating range condition

The cover and composition of perennial plant species are the focus of visual, site-based range condition assessment in Western Australia (Curry and Payne 1989) and elsewhere (Dyksterhuis 1949; Heady 1975). In this section I evaluate how well range condition assesses the capacity of the land to support livestock (see Chapter 4). I also consider the capacity of range condition assessments to reflect variation in attributes reflecting the interests of nature conservation and natural resource management. I acknowledge that like all site-based systems of assessment, range condition can only address limited parts of ecological hierarchy because of the scale of assessment (Allen *et al.* 1984; Pickup 1989; Noss 1990).

9.4.1 Pastoralism

How well does range condition reflect the long-term capacity of the land to support livestock production? Local knowledge accumulated over many decades seems a useful surrogate for detailed long-term research into patterns and causes of variation in forage availability (Stafford Smith 1988). For this reason I used pastoralists' suggestions of what can be inferred from different range condition classes among my study landscapes to help answer this question (see Chapter 4) (Pringle *et al.* 1992; Pringle and Riches 1996).

Chenopod shrubs provide the only forage that persists during the prolonged dry spells (‘droughts’) that characterise my study region. Since in this region it is unlikely to be economically feasible to avoid droughts by temporarily removing stock from stations the persistence of the palatable shrubs is critical to avoiding major stock losses during droughts. The condition of the chenopod shrubs is therefore a good metaphor for range condition in these chenopod landscapes under the prevalent strategy of continuous stocking.

However, range condition is probably not as good a metaphor for the long-term capacity of mulga landscapes to support livestock, because forage production in these landscapes is seasonally-driven (i.e. provided mostly by ephemeral species) and little forage persists through droughts (Wilcox 1963; Pringle and Riches 1996). Mulga landscapes may be a case where range condition assessment overestimates 'range degradation', due to an inappropriate reliance on palatable perennial plant species which do not provide the bulk of the forage on which livestock production depends (Wilson and MacLeod 1991; Behnke and Abel 1996a; b).

Thus there appears to be a contrast in the capacity of range condition to serve its primary purpose among my study landscapes. That this has not been widely recognised is symptomatic of an Australia-wide paucity of research aimed at testing underlying assumptions made in range condition assessment. In particular, little if any calibration has been undertaken between assessments and inferences made from them in different types of landscape (Holm 1994; but see Ash *et al.* 1995; Atkins 1998).

It is because of shortcomings like these that range condition assessments provide little support for pastoral managers. The concept was developed in North America as an adaptation of Clements' Plant Succession Theory (Clements 1916; Sampson 1917; 1919). An underlying assumption was that predictable change in vegetation composition could be achieved solely by manipulating stocking rate. This has been shown not to be the case and the adaptation of Plant Succession Theory to arid rangelands is now widely acknowledged as inappropriate (Connell and Slatyer 1977; Noble 1979; Smith 1989; Westoby, Walker and Noy-Meir 1989a; b).

Because change in arid rangelands does not occur in a predictable, linear fashion, there is no simple means of inducing a change from 'poor' to 'good' range condition. The opportunities for major change may be infrequent (Noble 1986) and management requirements at such times may differ considerably between types of landscapes. A reliance on range condition to assess the capacity of rangelands to provide forage for livestock may have precluded decades of action learning (management as experiments) (Holling 1978; Walters1986). The challenge now is to develop more realistic models of grazing-forage interactions (Westoby, Walker and Noy-Meir 1989a) , including systems of assessment that are more descriptive and can be used to better support pastoral decision-making. How this might be done is discussed in section 9.4.

9.4.2 Nature conservation

Range condition has considerable potential as a good metaphor for 'biodiversity condition'. Palatable perennial plants (the focus of range condition assessment) may be amongst the most threatened groups of organisms in arid rangelands because to survive they must persist during dry periods when grazing stresses are extreme (Noble and Slatyer 1980; Westoby 1980). Extreme levels of grazing under such circumstances are generally beyond evolutionary experience in Australian arid lands , where a general lack of naturally persistent surface water would have limited large herbivore distribution and density before pastoral development (Landsberg and Gillieson 1996; Landsberg, O'Connor and Freudenberger 1999). Range condition therefore assesses 'focal life forms' (Lambeck 1997). Perennial plant species are also 'ecosystem drivers' (Walker 1992) in arid rangelands (Slatyer 1975; Ludwig and Wiens in press) in that they have a strong influence on ecosystem patterns and processes. Thus they are also probably important in maintaining the ecosystem stability necessary for many other life forms to persist (Ehrlich and Walker 1998). In focussing on 'focal life forms' and 'ecosystem drivers' range condition may be a more general predictor of 'biodiversity condition', at least in terms of flora, than it is of 'pastoral condition'.

However, range condition may over-estimate impacts of grazing on ants, particularly in mulga landscapes where few species appear to be disadvantaged by increased grazing. This is probably because the 'ecosystem drivers' (mulga trees) appear largely unaffected by grazing, and the 'focal life form' -palatable perennial plants- is a minor component of these ecosystems. This illustrates that the 'focal species' approach is likely to be more conservative/sensitive than the 'ecosystem driver' approach for monitoring biodiversity. Using the latter approach, numerous understorey plant species could be lost from some landscapes (e.g. those structured around mulga) without any discernible change in the species that confer stability on the system.

9.4.3 Natural resource management

Patterns and processes in arid landscapes are strongly influenced by perennial vegetation (Tongway and Ludwig 1994; Ludwig and Tongway 1995b), though the strength of this influence varies among types of landscapes and the influence of grazing. Where sheep 'eat the architecture', range condition is probably a good surrogate of ecological integrity. This conclusion applies both to chenopod landscapes and sandy banks in sandy mulga landscapes. However, where unpalatable woody species are typical secondary successors (Archer 1994; Whitford 1995; Reynolds *et al.* 1999), this relationship may break down.

Range condition assessments will probably overestimate adverse impacts on ecological integrity in landscapes where physical landscape patterns and processes are dominant influences on ecosystem structure, and/or where 'ecosystem engineers' are either unattractive to stock or beyond the reach of grazing. Mulga landscapes are good general examples of this, though the sustainability of these systems in the very-long term (many decades/centuries) will depend on the capacity of 'ecosystem engineers' to sustain themselves by recruitment under grazing (Lange and Purdie 1976; Lange and Willcocks 1980).

In summary, range condition assessment has some advantages, but also considerable limitations. It is probably best at indicating forage availability during droughts and it may also be useful as an indicator of 'biodiversity condition'. It is a reasonable indicator of ecological integrity in chenopod landscapes, but may overestimate change from an undisturbed state in mulga landscapes. From the various viewpoints of my study, range condition assessments were most useful in chenopod landscapes, reflecting the relevance of what was assessed (perennial understorey plants) to the objectives and critical processes involved.

The limitations of range condition assessment are due largely to the limited ecological information that can be ascertained from visual assessments, even from experienced practitioners. These assessments are essentially interpretations couched in value-laden terms (e.g. 'good', 'fair' or 'poor' for pastoralism). While the underlying ecological model allows for multiple semi-stable states, this information is lost once the interpretation is recorded. For instance, 'very poor' condition may be recorded both for relatively stable sites invaded by woody weeds and for actively eroding sites virtually devoid of perennial vegetation.

Thus within any single range condition class, different ecological states might require different interpretation, depending on the land management perspective. That is, there may be considerable variation within a single range condition class in the capacity of the land to support different types of stock (Stuart-Hill and Hobson 1991), to reflect 'biodiversity condition' (Stuart-Hill and Hobson 1991; Pringle and Burnside 1996; Smith, Holechek and Cardenas 1996; Nelson 1999) or to reflect ecological integrity (Ludwig and Tongway 2000). Nevertheless, my results show that the range condition assessments practised in Western Australia can be interpreted usefully in some situations. In particular, some of the information provided in range condition assessments has considerable potential for supporting conservation planning in my study region (Blood 1995).

9.4.4 Beyond traditional range condition assessment

The challenge is to develop an assessment system that focuses on ecological observations that can be interpreted later from different management perspectives. There are numerous practical problems. For instance, what criteria should be used for distinguishing states? In Chapter 8 I mentioned a case where a vegetation scientist might say woody weed infestation of mulga drainage lanes represents a 'flip', while a geomorphologist might see the change as trivial. One has to make judgements in the process of classifying the real world.

In proposing some avenues for progress I am guided by five underlying principles:

(1) Any system of assessment must be influenced by perception of contemporary objectives (Holmes 1994; Mentis 1984).

(2) Assessment should be based on ecological observations (perhaps based on interpretations of impacts), not inferences according to a specific (limited) suite of objectives (Stuart-Hill and Hobson 1991; National Research Council 1994; West *et al.* 1994).

(3) Hierarchical stratification is required in order to optimise the efficiency of selected surrogates (McArthur 1964; Noss 1990; Abensperg-Traun *et al.* 1997).

(4) Assessment must be multi-disciplinary and inclusive.

(5) Ecological theory and methods of assessment need to be fully integrated (Joyce 1993).

There is a trade-off, between consistency over large time and spatial scales, and efficiency of each monitoring programme within its socio-economic context (West *et al.* 1994; Friedel and Laycock 1996). I agree with many others (Society for Range Management 1990; National Research Council 1994; West *et al.* 1994) that the primary classification of ecological sites should be based on physical environmental properties which evolve at much longer time scales (usually) than biological attributes. These primary classifying properties should define recurring patterns of landforms and major soil types. In Australia, this level of classification is termed a land system (Christian 1958; Christian and Stewart 1968). Landscape processes are also critical in differentiating similar land systems. For instance, chenopod shrublands on plains with texture contrasting soils - such as the landscapes I have studied here - are far more resistant to grazing and severe soil erosion than similar soil-vegetation associations flanking major creek or river systems. These latter land systems occupy similar positions as far as catchment toposequences are concerned, but experience considerably stronger fluvial energy and are susceptible to soil erosion (Wilcox and McKinnon 1972; Pringle 1994b).

Nested within land systems are land units: the building blocks of land systems. They are distributed in a distinctive pattern and recognisable individually as landform/soil/vegetation associations. This level of landscape classification is analogous to the 'range site' and its successor, the 'ecological site' in North America (Society for Range Management 1983; 1990; 1995).

Once sites have been classified according to their bio-physical environment, the next task is to distinguish semi-stable domains of attraction, the 'states' in the State and Transition model (Westoby, Walker and Noy-Meir 1989a; Borman and Pyke 1994). This approach assumes that a State and Transition model applies universally, which may not be so (Stafford-Smith 1992). However, should a model of more continuous variation apply, this variation could be treated as a large domain of attraction.

Elaborate multivariate methods have been used to distinguish 'states' objectively (Stuart-Hill and Hobson 1991). This seems unnecessarily complicated and impractical if underlying ecological models are to be developed for all major or critical ecological sites within a region. It may be more efficient to use a more traditional, 'expert' system of classification based on vegetation structure and dominant taxa. For example, the chenopod landscapes I have studied can be described in the following states:

- diverse low shrublands dominated by *Atriplex vesicari* ;
- low shrublands dominated by *Maireana pyramidata, Lawrencia squamata or Frankenia* spp.;
- tall shrubland dominated by *Hakea preissii*;
- a largely stable herbfield; and
- an unstable, partially stripped plain with patches of vegetation.

I derived these states simply by looking at my site notes. This could be done for many ecological sites already covered by rangeland surveys in arid Western Australia (Pringle 1991), based on site sheets. It could also be done using many different forms of information, including that from biological surveys by Government agencies or mining companies.

Classification at this level should aim to encompass both the 'ecosystem drivers' and 'ecosystem engineers' and hence have some extrapolative value for other aspects of biodiversity (Walker 1992) and ecological integrity (Archer 1990; Reynolds *et al.* 1999). However, a classification of site notes alone may be too general to encompass species that

are less common and influential on ecosystem processes, but may be more easily removed by disturbances such as grazing (e.g. sparse palatable shrub species in mulga landscapes).

Thus another trade-off must be made, between trying to include all species and being able to make many assessments over large tracts of land. The use of species response groups such as I have derived in this study and which have been used for decades in rangeland survey in Western Australia (Wilcox and McKinnon 1972; Pringle 1998b) might provide a solution. Assessments based on the relative abundance of species belonging to different response groups within states might provide enough information to indicate trends, without making the task impractical. Thus there are two levels of assessing grazing impacts: the ecological state, and the prevalence of different indicator species within that state.

The approach is not specific to assessing the impacts of grazing. For instance, it could be used to assess the impacts of sulphur dioxide emissions from mineral processing plants. The situation would become far more complicated when pressures overlapped, as interactive effects would then need to be considered.

A future system of assessment might then have the following hierarchical structure:

- region/catchment
- land system
- land unit or ecological site
- state
- status of indicator species or response groups

The system could be developed through a number of complementary means including research such as this, reviewing previous research and survey information, and by adopting an adaptive approach to implementation. The adaptive approach could involve more detailed measurements at a subset of assessed sites, to check assumptions or estimations based on first principles where information was lacking.

For instance, there may be insufficient time to conduct detailed assessment of landscape patterns and processes, but one might predict that vegetation structure and cover might be reasonable surrogates. This could be tested using Landscape Function Analysis (Tongway 1995; Tongway and Hindley 2000) at a subset of sites. The ability of the system to reflect changes in other attributes such as aspects of faunal diversity or carbon accounting could also be assessed at subsets of sites. This adaptive approach may build understanding of the

ecosystems being assessed and their management requirements, as well as identifying areas for future research where the system appears to fail (i.e. where applied first principles do not conform to subsequent observations).

The system I outline might be applied generally, with local augmentation where specific issues arise. For instance, it may be necessary to record emergent properties such as vegetation cover at a subset of sites in order to link ground-based assessments to remotely sensed assessments (Pickup, Tynan and Chewings 1998; Pickup, Bastin and Chewings 1998). Alternatively, there may be areas of particular conservation value where additional attributes may be pertinent. Recording the presence of feral cats or foxes may be extremely useful in areas in which native fauna are being re-introduced to their former range. Similarly, recording the presence of exotic plants may be useful for prioritising potential areas to be included in an expanded reserve system.

Such a hierarchical system of assessment might also provide a framework for future management and research in a manner similar to the 'management map' approach developed in southern Africa (Stuart-Hill and Hobson 1991). For instance, different stakeholders might express preference ratings for various states within an ecological site. Research questions might include: are a particular set of attributes (e.g. ground nesting birds or palatable grasses) actually present in a particular ecological state or, what management is required to induce a transition from one ecological state to a more desired one?

If range condition assessments are not nested within an explicit hierarchy such as the one I have outlined, they will continue to be a partially effective system for assessing the capacity of the land to support livestock and a useful indication of some general level of grazing impact on landscapes. They are unlikely, however, to help pastoralists make decisions about directing change in a desirable direction. They could also be misleading to conservation planners in regions where exotic plant species are prominent and palatable ('good' from a pastoral perspective) (Pringle and Burnside 1996; Young and Allen 1997). They may also fail to convey substantial change in landscape pattern or process because of the large variation within condition classes.

9.4.5 Some issues regarding implementation of a new system

There are substantial administrative challenges confronting implementation of a new, more expansive and costly system of range assessment. The challenges are amplified by the persistent poor terms of trade within the wool industry: Governments are unlikely to increase

funding of environmental programmes for a land use that has been in decline for some time (Holm 1993; Anon. 1993b). Furthermore, the 'user-pays' principle is impractical: few pastoralists are in a position to fund assessment of their use of resources.

There are economic and technical reasons why an expanded system of assessment requires an effort between Government agencies. Firstly, additional funds will be needed to conduct an expanded system of assessment. Under the concept of ecosystem management, where legal boundaries are not treated as boundaries for broader environmental management, it is appropriate that assessments dealing with issues such as nature conservation and carbon accounting receive funding additional to that supplied for traditional range assessment. The participation of the private sector in funding new ecologically sustainable resource systems could also be nurtured (Daily and Walker 2000). Secondly, multi-disciplinary input will be required to implement an expanded system of assessment. The technical expertise needed to incorporate wider perspectives than pastoralism resides across a number of environmental agencies, academic institutions and the private sector.

In practice, I foresee the system of assessment I have outlined as part of a broader and more cohesive approach to environmental management. This broader system could include aspects of land use planning (e.g. expanding reserve systems and designating areas for various levels of protection from grazing), setting of objectives for different categories of land (e.g. the Excised Management Units of Morton and colleagues (Morton *et al.* 1995), and systems of assessment and review.

If the system of assessment is to be nested within regional environmental systems, it cannot be fully implemented until land use categories have been delineated and objectives and associated indicators assigned to them. In other words, there needs to be tailoring of a general system of assessment to investigate specific objectives that may (should) vary across regions (Mentis 1984; Friedel and Laycock 1996). The development of different land use categories will implement a shift from a mentality of reserve systems versus stations, each with their single systems of assessment (Shea *et al.* 1997). Rather, land use categories will have blends of objectives, reflecting variation of emphasis on resource and conservation goals (Morton *et al.* 1995; Pringle 1995a).

The adoption of blends of objectives will signify a shift from traditional 'environmental apartheid' to functional multiple land use (Cohen 1992). Without the integration of conservation and resource planning nested within scales of management (i.e. region down to individual paddocks and watering points), pursuit of ESD will remain piecemeal and inefficient.

Finally, the involvement of land managers (e.g. pastoralists, miners and indigenous groups) is essential. The areas involved are too large for Government officers to do all works and monitoring. The challenge is to develop mechanisms to make participation attractive to the people who most directly use or value the resources being assessed (Pringle 1995a; Binning 1997; Binning and Young 1997; Shea *et al.* 1997; Briggs 1998).

9.5 A framework for assessing the spatial distribution of grazing impacts

In this section I attempt to generate plausible models of the effects of grazing history in a spatial context. This allows for an assessment of conflicts and complementarity among key perspectives in a real rangeland context (i.e. relative to the locations of watering points, fences, paddocks, stations and so forth). In the process, I develop a conceptual framework for assessing current patterns of rangeland use that might provide useful insights with which to reconcile future use patterns with evolving social imperatives (e.g. a greening of urban attitudes to rangelands).

9.5.1 Conceptual models of grazing impacts

Most research into grazing impacts has been conducted to improve the efficiency of pastoral management and has involved comparisons among stocking rates or land condition classes (Ash and Stafford-Smith 1996; Holecheck *et al.* 1999). Yet there is an abundance of evidence showing that grazing intensity varies considerably across large, arid zone paddocks (see Chapter 3 and references therein). Thus much previous research on grazing impacts is difficult to translate into practical paddock management and monitoring/assessment (Pickup 1989; Stafford Smith and Pickup 1990; Ash and Stafford-Smith 1996).

Where paddock-scale (or similarly broad scale) approaches have been adopted, the underlying focus has generally been on pastoralism and land condition in that context (Senft *et al.* 1987; Hellden 1988; 1991; Coughenour 1991; Bastin *et al.* 1993; Bailey *et al.* 1996; Pickup, Bastin and Chewings 1998). Examples in the traditional rangeland literature of spatial approaches beyond pastoral value systems are rare (e.g. Lange and Purdie 1976).

More recently, research has accelerated into the spatial implications of grazing on aspects of biodiversity (Thrash, Theron and Bothma 1995; Smith, Holechek and Cardenas 1996; Landsberg, James and Morton 1997; Thrash 1998; James, Landsberg and Morton 1999; Nelson 1999). However, these approaches are either conceptual or based on distance from water *per se*. For instance, Nelson et al (1999) recommend proportions of rangelands that should be maintained in different seral states to meet a range of conservation and use

objectives. In recent research by CSIRO (Australia), it was recommended -amongst other things- that a substantial proportion of the arid Australian biota requires the degree of protection from grazing usually provided only in areas remote from watering points (Landsberg, James and Morton 1997; Landsberg *et al.* 1999).

However, there has been little spatially explicit modelling of profiles of impacts (such as I have presented previously) from which to consider ecosystem management of pastoral paddocks and stations. I return to this theme after presenting a novel approach to spatial modelling of grazing impacts. I adopt the Pressure-State-Response approach used in state of environment reporting in Australia (State Of Environment Advisory Council 1997). 'Pressure' refers to the activity of concern (in this case grazing); 'State' refers to the state of the environment (e.g. species richness or soil salinity), and 'Response' refers to the societal action (e.g. introduction of land clearing bans) that is taken as a result of assessing Pressures and States.

9.5.2 PRESSURE: Grazing history in paddocks

The context of my study is rangeland pastoralism in a region where grazing is the overriding pressure on the state of the environment (Pringle, Van Vreeswyk and Gilligan 1994). I used two quite contrasting methods to model the distribution of grazing history within paddocks, and I will focus on the implications of the differences between them.

Model 1: Track-density based model of Stocking History (SH)

The Stocking History (SH) model predicts a consistent pattern of rapid decline in grazing with increasing distance from watering points (Figure 9.3) and a negative exponential function described this pattern well (Table 9.1, Figure 9.4). I used REML to do the modelling because I had an unbalanced design, but when I ran the same model using GLIM as an exploratory tool, the model accounted for 87.2 per cent of variation in (natural log transformed) Stocking History (SH). While GLIM is likely to be an anti-conservative approximation of variance accounted for (Patterson and Thompson 1971; Robinson 1987), it indicates a very good fit nonetheless. The statistical model using raw values of Stocking History (SH) violated underlying statistical assumptions (and accounted for only 44.2 per cent of variation).

Model 2: RANGEPACK/Paddock model of Stocking Rate (SR)

There was little difference in the goodness of fit using the raw Stocking Rate (SR) estimates (Figure 9.5) and their natural log transformations, but the latter was slightly superior and indicated a weak exponential function best accounted for its variation with distance from water (Table 9.2; Figure 9.6). Using GLIM as an exploratory tool to compare the raw and log

transformed models, the variance accounted for was respectively 78.8 per cent and 82.9 per cent. In both models, the diagnostics conformed to underlying statistical assumptions.

Both log transformed models are highly correlated with distance from water (which reflects its influence in their formulation, see Chapter 3). However, for almost all response variables modelled, they far out-performed distance from water alone in accounting for systematic variation. This suggests that the modelling process was critical to this study and indicates that studies based on distance from water alone may not have accounted for substantial systematic variation in response variables.

That the two models of grazing history show a strong difference in the pattern of decline with increasing distance from water is the basis of further interpretation in this section.

If one or other model estimates a response variable better, one can cautiously predict the pattern of grazing impact on that variable. For instance, if Stocking Rate (SR) provides the superior fit for a particular variable, then it is likely that the impact of grazing on that variable dissipates gradually away from watering points and so extends over considerable areas of a typical 8km x 8km paddock. Conversely, if the Stocking History (SH) model describes better the pattern of variation in another variable, the impact of grazing on that variable is likely to be more intense and largely restricted to areas in close proximity to watering points.

Thus there are two quite clearly contrasting models of spatial patterning of grazing impacts, depending on the variable being assessed. They beg different conceptual models for future planning of rangeland use.

For those variables whose variation is described better by the Stocking Rate (SR) model, a conceptual model of extensive fragmentation of landscapes may be appropriate. Watering points scattered throughout arid rangelands (Valentine 1947; Landsberg and Gillieson 1996) represent foci around which landscapes are widely altered with benchmark areas (those impacted negligibly by grazing) probably restricted to places far from watering points (say > 8 km in the case of sheep).

For those variables whose grazing impact is predicted better by the Stocking History (SH) model, a conceptual model of intensive, but localised impacts might be more appropriate. Watering points then represent many scattered, small areas of high impact in a largely intact landscape.

Not only does the pattern of decline in grazing impacts away from watering points vary considerably among response variables, it may also differ among types of landscapes or their strata. This is indicated by differences in the slopes of fitted relationships for different

landscapes or strata. I address this nested effect firstly by classifying a response as either strongly exponential (i.e. Stocking History response) or almost linear (Stocking Rate response) if the effects are strongly in favour of one model or the other and then incorporating landscape-grazing interactions into fitted responses.

Mathematically:

Var (Y) = a + b*G

Ln(G) = c+k*D ; (see figures 9.4 and 9.6)

$G = e^{c+k*D}$; and

Var (Y) = a+b(e^{c+k*D}).

Where;

'Var (Y)' is a response variable from Chapters 4-8,

'G' is grazing estimate; either the Stocking History (SH: sheep years/ha) or Stocking Rate (SR: sheep/ha), and

'D' is distance from water (km),

and **a**, **b**, **c** and **k**.

Where **Var (Y)** was transformed prior to statistical modelling in earlier chapters (e.g. electrical conductivity of topsoil was transformed to its inverse), this is accommodated by appropriate modification of the equations. In such cases, the transformations also strongly influence the rate at which impacts of grazing decline away from watering points.

9.5.3 STATE: some selected case variables

For most response variables, comparison of Wald Statistics indicated that either the Stocking History (SH) or Stocking Rate (SR) model was clearly better in estimating variance. Sometimes this involved just a (main) grazing effect, in other cases there were landscape effects and interactions between grazing and landscape effects (see tables in Chapters 4-8). There was some degree of consistency among attributes representing a perspective (pastoralism, natural resource management or nature conservation). Response variables showing either a strong negative exponential or almost linear pattern with increasing distance from water are presented in Table 9.2.

Response variables representing pastoral values changed in a nearly linear fashion away from watering points, indicating that impacts on pastoral resources extend for many

kilometres and probably affect all or most of normal-sized paddock of 8 km x 8 km (Figure 9.7). Aspects of floristic diversity show similar nearly linear patterns of impact, which is as expected considering that the focus of range condition is the assessment of vegetation (Figure 9.8).

In contrast, aspects of ant diversity, and ecological integrity (landscape organisation and properties of topsoil) are more tightly focussed on watering points, with a more exponential decline in impact away from water (Figures 9.9 and 9.10). That impact on aspects of ant diversity share strongly exponential increases with proximity to watering points with attributes such as size and proportional area of bare patches and topsoil salinity is consistent with ants being primarily sensitive to variation in physical environmental parameters, rather than changes in floristic composition.

Thus there is some general consistency within perspectives, but quite large differences in the magnitude of impacts among landscape types. This provides a complex, but reasonably well structured basis for considering future patterns of pastoral, station and regional management of arid rangelands.

9.5.4 RESPONSE: Implications of spatial models from different perspectives

To illustrate the spatial implications of different patterns of grazing impact I consider paddocks that are 8km square, with fences aligned north-south and east-west, and with watering points in all four corners. This is typical for arid rangelands in Western Australia (Pringle 1994a). Furthermore, the persistence of some level of pastoral use will be assumed (i.e. a ‘close it all down’ option will not be considered). My aim is to focus on patterns of use that optimise performance as judged from different perspectives for realistic future scenarios.

My scenarios are based on spatial patterns of grazing impact from different perspectives. They are certainly not meant to be blueprints for future rangeland management. Rather, they are intended to highlight conflicts and complementarity that can be evaluated with this approach.

Pastoralist

I interpret the pastoralist's objectives to include optimal use of forage in paddocks and maintenance of palatable perennial plants in order to ‘drought proof’ the production system.

The major issue from this perspective is the decline in Sustainable Carrying Capacity that extends far from watering points. This is a particular problem in chenopod landscapes, so it

may be worth maintaining watering points as they are (Figure 9.11), rather than spreading them to further spread grazing activity. Widely spaced watering points will help maintain areas of intact chenopod shrublands far from watering points and so preserve the 'drought durability' of chenopod paddocks.

In mulga landscapes this is less of a worry as it is essentially 'seasonal country' and these landscapes are relatively resistant to grazing. Therefore, it might be worth piping water away from bores or wells in paddock corners and perhaps installing a central watering point, to allow a greater distribution of grazing from each watering point (Figure 9.12) (Stafford-Smith 1987). This will enhance the capacity of stock to graze all parts of the paddock and so simplify management (by reducing spatial heterogeneity of grazing and forage availability). Stock will be able to make much better use of ephemeral forage before it desiccates with more evenly spread grazing.

Natural Resource Manager (e.g. agricultural adviser)

I interpret the primary objectives of the natural resource manager to be the prevention of degradation of the pastoral resource in terms of maintaining or improving range condition. I also interpret a desire on behalf of natural resource managers to see resources being used optimally (Burnside *et al.* 1990; Cridland and Stafford Smith 1993).

From this perspective, declines in Sustainable Carrying Capacities extending far from watering points indicate that lighter stocking is needed overall. Furthermore, considerable degradation around watering points in chenopod landscapes in particular indicates that there is a case for more watering points and fewer sheep per watering point (Burnside *et al.* 1990; Cridland and Stafford Smith 1993). Thus the emphasis is on spreading grazing more evenly to prevent resource degradation and optimise forage utilisation across paddocks at 'moderate to light' stocking rates (Figure 9.12).

Poor correlation between range condition and forage availability in mulga landscapes might beg review of the resource manager's assessment of these landscapes. Given their inherent stability, it may be worth spreading watering points (Figure 9.12) and stocking these landscapes heavily in good seasons to make best use of ephemeral forage (Wilcox 1963). This might allow for reduction of stock numbers in chenopod landscapes and so provide an opportunity for shrub recruitment and for adult shrubs to regain vigour.

The main differences in perspectives between the pastoralist and the natural resource manager is that the pastoralist wants maximum sustainable yield from paddocks, whereas the natural resource manager is firstly concerned with degradation and then with the

efficiency of resource use. Both the pastoralist and the natural resource manager want to spread grazing more evenly in mulga landscapes, but the pastoralist may accept degradation around watering points in chenopod landscapes in order to maintain a 'haystack' of persistent forage for drought durability.

Nature conservationist (e.g. regional conservation biologist)

I interpret the primary objectives of the nature conservationist to be the maintenance of regional populations of grazing-sensitive species within paddocks. This might seem an unrealistic objective and an unfair demand on pastoralists. However, nature reserve systems in arid rangelands of Australia are poorly representative of their regional biodiversity (Specht, Roe and Boughton 1974; Thackway and Cresswell 1995; Pringle 1995a). Off-reserve conservation is critical under such conditions.

The major issue from a nature conservation perspective is the proportion of plant species that are poorly adapted to grazing and decline in abundance for considerable distances from watering points. The rate of decline with increased grazing is strongest in mulga landscapes, though there are more (and a greater proportion) of decreaser species in chenopod landscapes. One way of increasing the area of paddocks considerable distances from water would be to close all but one (corner) watering point in each paddock. This could be done such that the resulting pattern consists of a watering point every second paddock corner in either a north-south or east-west direction, with remaining watering points are 16km apart (Figure 9.13). An acceptable trade off from a point of view of species conservation may be to increase stock numbers at the remaining watering points. Sacrifice zones (Valentine 1947) could be allowed to expand in return for considerably expanded areas remote from watering points. The closing down of many watering points would also ensure that animals not restricted by fences (e.g. kangaroos, feral goats) would also have to focus their activity around sparsely distributed watering points.

9.5.5 Comparing implications of different perspectives

Under the forgoing scenarios there is potential for strong conflict in mulga landscapes between pastoralists and natural resource managers and nature conservationists. From the perspective of the pastoralists and natural resource managers, there is benefit in increasing the density of watering points, while from the perspective of the nature conservationist there is benefit in substantially reducing the density of watering points.

In chenopod landscapes, natural resource managers could see benefit in greater densities of watering points, while nature conservationists could see this as detrimental. From a pastoralist perspective, the optimum density may be somewhere in between:

(1) they may not see much benefit to balance the cost of increasing the density of watering points (and then maintaining them), but

(2) they may be disadvantaged by a reduction in the area available for efficient grazing under a nature conservation scenario.

There may be other benefits for the pastoralist in the nature conservation scenario, such as decreasing maintenance costs considerably and perhaps reducing reliance on fences for stock management.

In summary, from a nature conservationist's perspective, fragmentation of areas readily grazed by stock may be desirable in arid rangelands. From the perspective of a natural resource manager (and to a lesser extent pastoralist), however, the desirable scenario is likely to further fragment the distributions of particularly grazing-sensitive plants (Noy-Meir 1996; Landsberg *et al.* 1999), thus increasing the chances of local and regional extinctions.

Are there opportunities for trade-offs? If nature reserve systems are comprehensive, adequate and representative, then the status quo of watering points in corners of paddocks may be acceptable, with a few minor provisos. One proviso would be that traditional range condition assessment remains part of any formal system of assessing land condition, provided it continues to be based on response groups of perennial plant species (‘a focal life form’). Secondly, areas within paddocks of particular nature conservation value (e.g. a highly palatable rare plant) would probably require complete protection from grazing with the added security of formal and legally binding (for Government and pastoralist) management agreements (Binning 1997; Binning and Young 1997; Briggs 1998).

However, while nature reserve systems in the arid zone of Australia remain inadequate, conflict exists, particularly in mulga landscapes. In these landscapes, desirable distributions of watering points are diametrically opposite. A solution might be to upgrade nature reserve systems, such that reliance on off-reserve nature conservation is reduced. This is consistent with international, national and State Government endorsed treaties and policies on maintaining biodiversity (Anon. 1980; 1984; 1987; 1999).

The status of regional nature conservation may depend on the existing densities of watering points and historical pastoral management (range condition). This will vary from one region to the next and according to types of landscapes within regions. I elaborate below.

In Western Australia for instance, more recently developed pastoral districts in the arid zone are generally ‘in better condition’ (less degraded) than those where pastoralism developed late last Century (Pringle 1991). (In eastern Australia the industry is considerably older). In

my modelling of grazing history (Chapter 3), the age of watering points appeared to affect the degree to which grazing was concentrated around them. With increasing age, it appears that stock tend to graze further from water, perhaps in response to expansion away from watering points of forage-deficient, degraded areas (Andrew and Lange 1986a; 1986b). This is likely to reduce the degree to which impacts are confined near watering points: patterns of impact may well become more linearly related to distance from water with increasing age of watering points. Populations of grazing-sensitive species may then become increasingly fragmented with accumulated exposure to grazing.

Another complicating factor is the effect of regional hydrogeology on the availability of good quality groundwater. Riverine catchments (e.g. the Murchison and Gascoyne River catchments) generally have better supplies and quality of groundwater than the internally draining salinaland (salt lake catchments) and the Nullarbor Plain (karst landscapes). Thus the density of watering points is greatest in riverine catchments, which are also the oldest pastorally and have large areas of severe degradation (Wilcox and McKinnon 1972; Curry *et al.* 1994). In old, well-watered pastoral regions, substantially unaltered (benchmark) areas are likely to be scarce (Blood 1995; Landsberg and Gillieson 1996).

Finally, the availability of suitable groundwater may vary within a district. For instance in this study, finding areas of chenopod landscape extending beyond 7km in paddocks was relatively easy; because it has been historically difficult to find good quality groundwater. Conversely, in mulga landscapes I could not find suitable paddocks conforming to my originally intended design because suitable groundwater is easily accessed (Allen 1994) and watering points are regularly located in most paddock corners.

Thus conflicts between the goals of nature conservation and pastoralism are probably most acute in the older pastoral districts based on river systems and least on the Nullarbor Plain, where the industry is youngest, watering points are often widely-spaced (Cridland and Stafford Smith 1993) and chenopod landscapes predominate.

9.5.6 Implications for future management

The spatial modelling indicates that for regional maintenance of floristic diversity at least, substantial areas on stations will need to be protected from grazing. This is at least partly because of the inadequacy of reserve systems in arid Australian rangelands. The question then becomes; how can this protection from grazing be achieved? I suggest three alternatives:

(1) fence off areas,

(2) shut down many watering points,

(3) substantially reduce the number of stock per watering point.

None of these alternatives is likely to be practical under prevailing systems of pastoralism and rangeland administration. The first option is likely to lead to least disruption to pastoral management, but is costly and would need external sources of funds. The second option will be extremely disruptive to pastoral management and may further jeopardise the viability of pastoral enterprises. The third option is unlikely to be economically feasible because pastoralists need to maximise the number of sheep using infrastructure that is costly to maintain, in order to survive economically. Further expansion of watering points is within the rights of pastoral lessees, but is contrary to ESD, at least in principle.

The most pragmatic solution may be to expand the reserve system and maintain the status quo on stations with the two provisos mentioned above, regarding range condition and specific conservation values. However, the way the reserve system is expanded could be done creatively, in a manner that optimises conservation benefit, benefits pastoralists and is cost efficient. Instead of whole stations being bought up and converted to reserves (as is usual practice), smaller blocks of land chosen for their conservation value could be reserved within a matrix of pastoral management. This would spread the areas protected from grazing across the region, increasing replication of protected biodiversity. Furthermore, pastoralists could be contracted to help manage these scattered reserves (e.g. maintaining fences, controlling feral goats) in return for a stewardship payment (Pringle 1995a; Beilin and Fleming 1997; Shea *et al.* 1997). The use of local, experienced land managers may be far more cost effective for conservation agencies than managing scattered reserves from regional centres.

9.6 Concluding comments

Grazing appears to have a deleterious effect on attributes representing all three key land management perspectives. This may reflect the failure of biota and landscapes to adjust to the introduction of the evolutionarily novel disturbance associated with pastoralism.

Range condition (as currently assessed) was correlated with impacts of grazing on other attributes, but did not provide information on how these impacts varied among and within types of landscapes. A new system of range assessment is therefore required, one that provides more ecological information and allows assessments to be interpreted from different perspectives.

If ecologically sustainable development is a genuine goal, many more areas than currently exist need protection from grazing in my study region. The best way to achieve this may be to develop a network of reserves much smaller than individual stations, within the pastoral matrix.

Finally, the processes of land use planning and assessment of land condition are inseparable. Land use planning provides the local objectives to be assessed, while assessments reveal the regional effectiveness of patterns of land use and local land management objectives. They are inter-dependent processes and require integration across Government agencies and the participation of other regional stakeholders.

Table 9.1 Modelling Stocking History (SH) and Stocking Rate (SR)

Model of grazing history (used as the response variable)	Model term	Wald statistic	d.f.	p-value
Stocking History (SH): based on track densities); using natural logarithms	landscape sub-type	6.4	2	0.04
	distance from water (DW)	4110.6	1	<0.001
	landscape sub-type.DW	12.5	2	0.002
Stocking Rate (SR): based on RANGEPACK/ Paddock model); using natural logarithms	landscape sub-type	15.9	2	<0.001
	distance from water (DW)	1552.8	1	<0.001
	landscape sub-type.DW	55.2	2	<0.001

Table 9.2 Classification of spatial impact on variables with increasing distance from water
E = Strongly exponential; L = linear (L)

Perspective	Response variable	Increase (I)/ decrease (D)	Shape
Pastoralism	Range condition score (high = worse)	D	L
	Sustainable carrying capacity	I	L
	Pastoral integrity	I	L
Biodiversity	Decreaser plant species richness	I	L
	Increaser plant species richness	D	L
	Ant species richness	D	E
	Decreaser ant species richness	D	E
Ecological integrity	Chenopod shrub cover	I	E
	Size of open areas, chenopod	D	E
	Potentially mineralisable nitrogen, sandy mulga	D	L
	Available phosphorus, mulga	D	L
	Organic carbon	I	E
	Electrical conductivity, chenopod	D	E

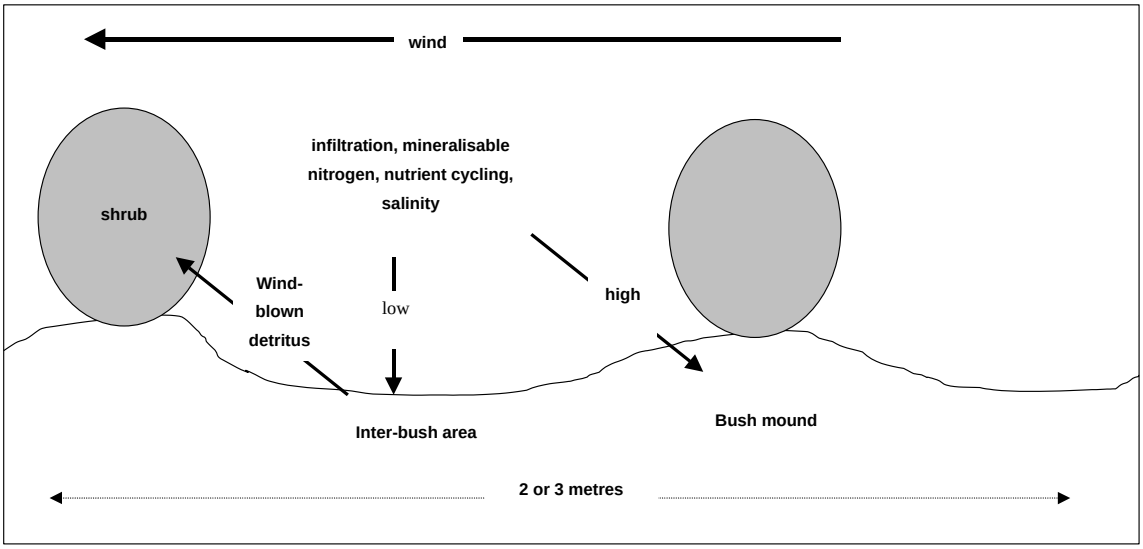


Figure 9.1 Intact patterns and processes in chenopad landscapes.

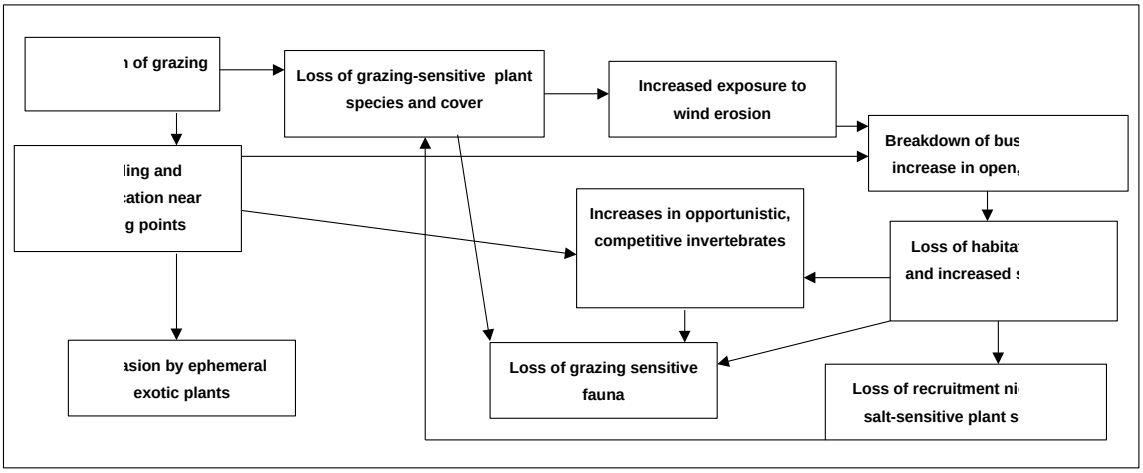


Figure 9.2 A model of degradation in chenopad landscapes.

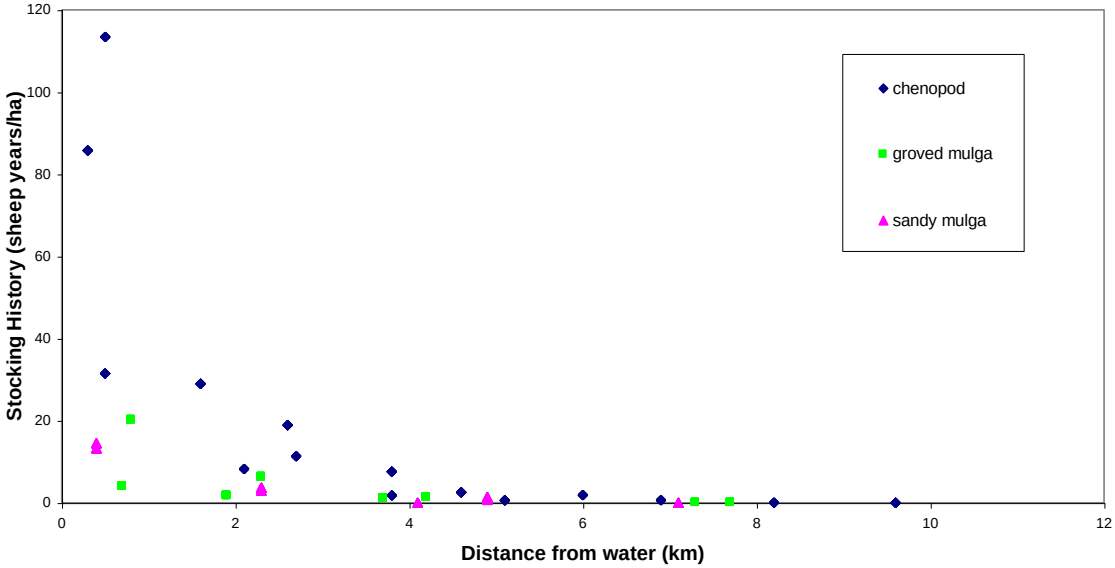


Figure 9.3 Modelled stocking history (SH) with distance from water.

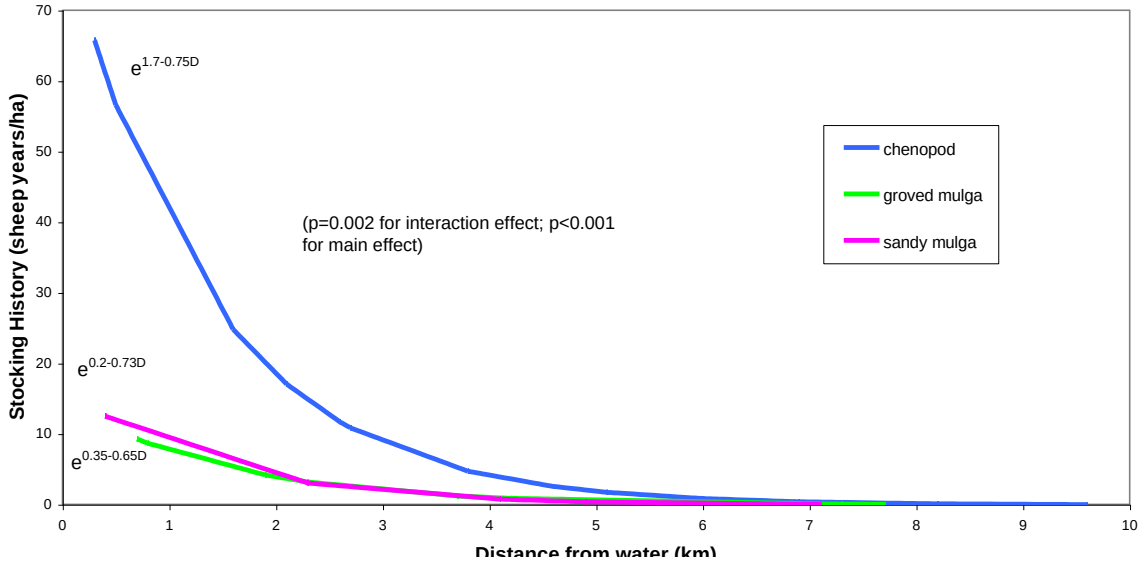


Figure 9.4 Fitted stocking history (SH) with distance from water.

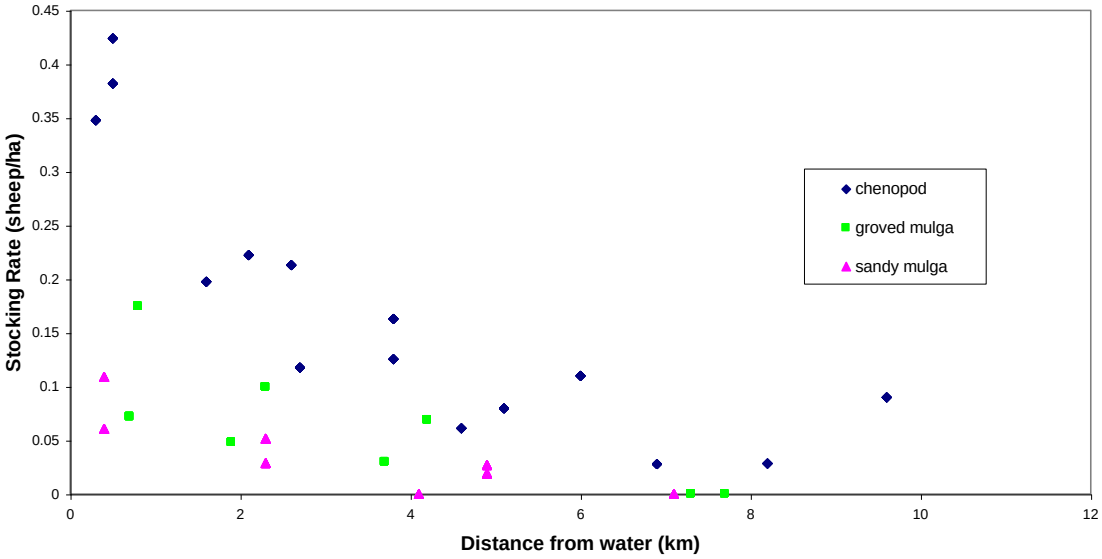


Figure 9.5 Modelled stocking rate (SR) with distance from water.

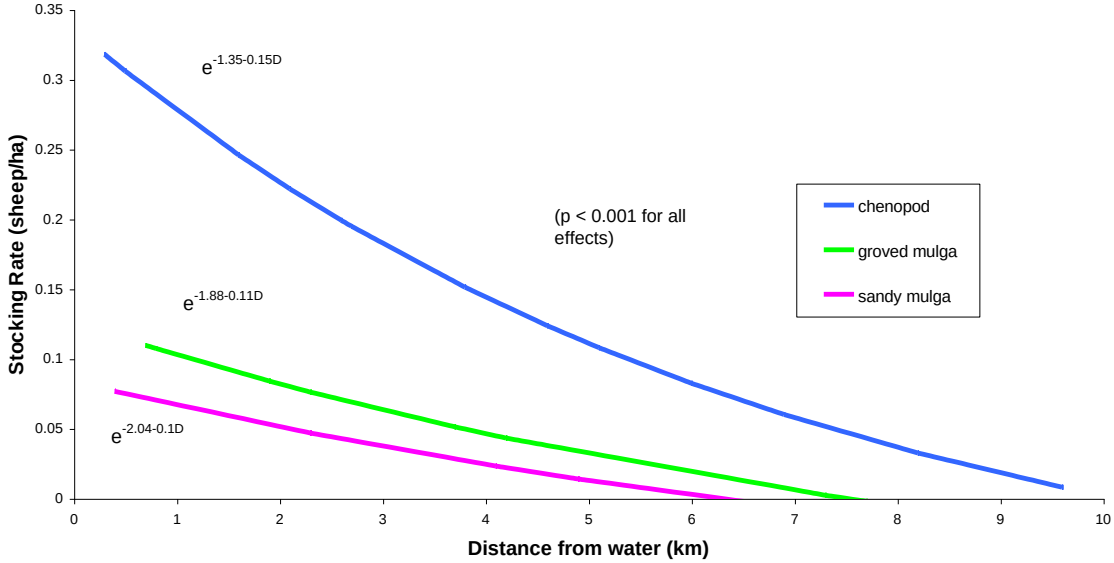


Figure 9.6 Fitted stocking rate (SR) with distance from water.

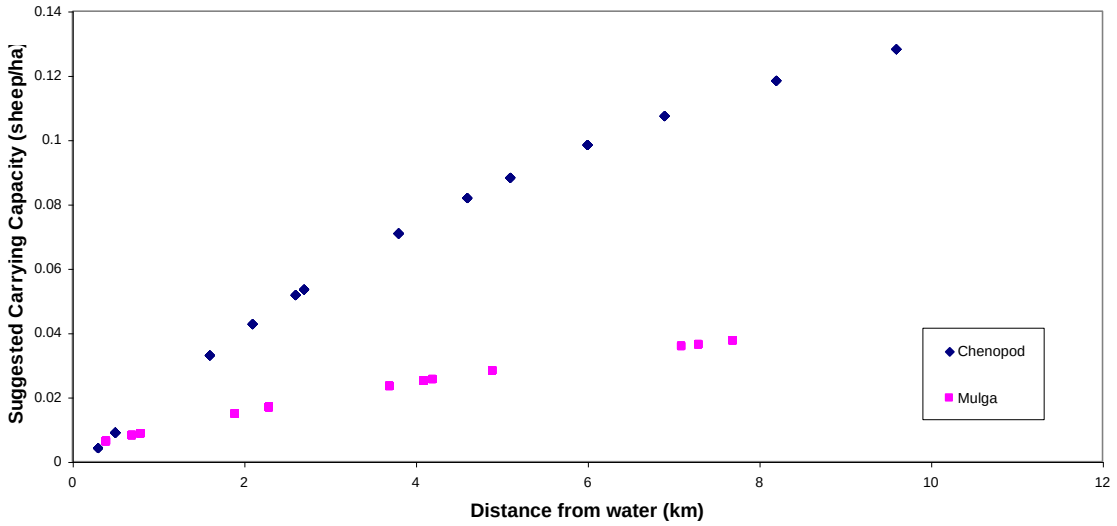


Figure 9.7 Interpreted influence of distance from water on suggested carrying capacity (SCC).

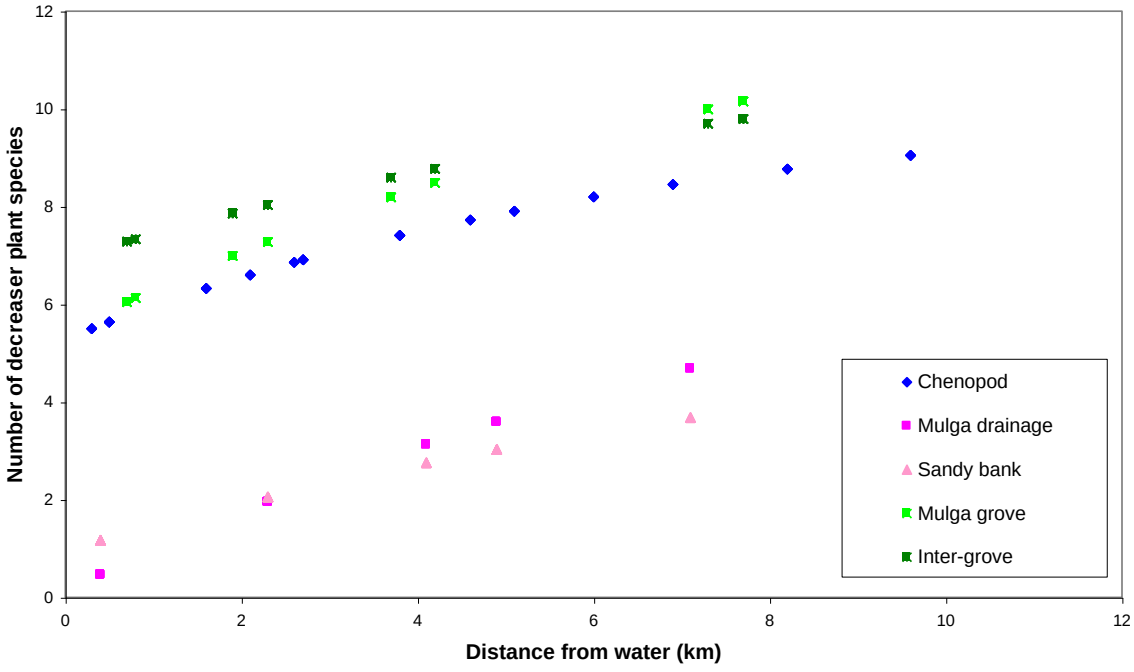


Figure 9.8 Spatial impacts on decreaser plant species richness.

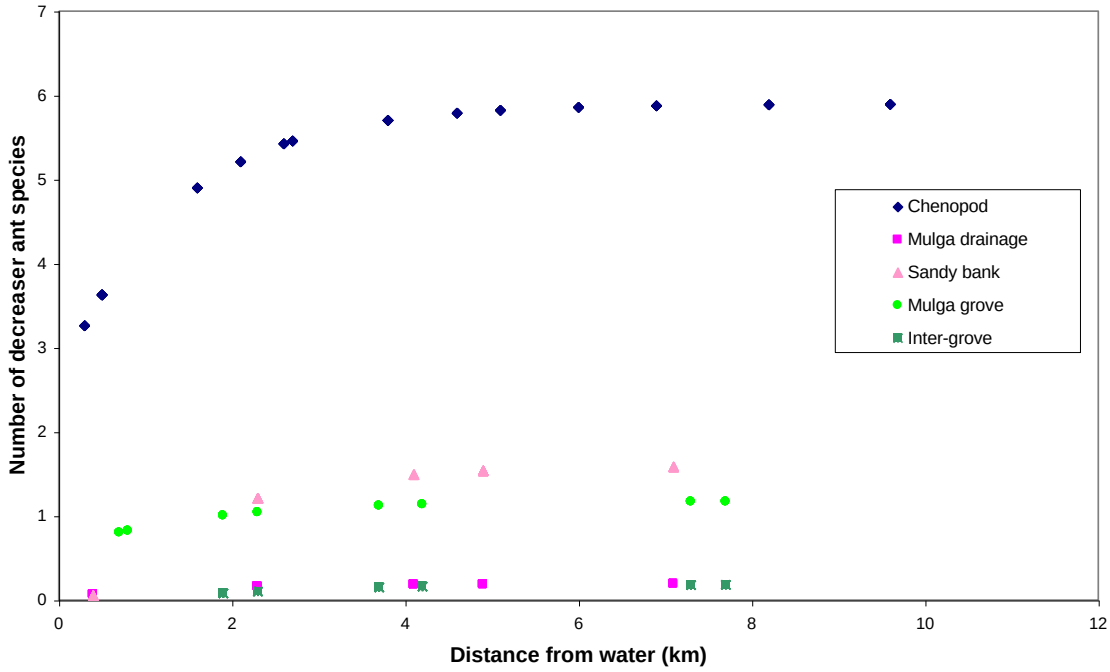


Figure 9.9 Spatial impacts on decreaser ant species richness.

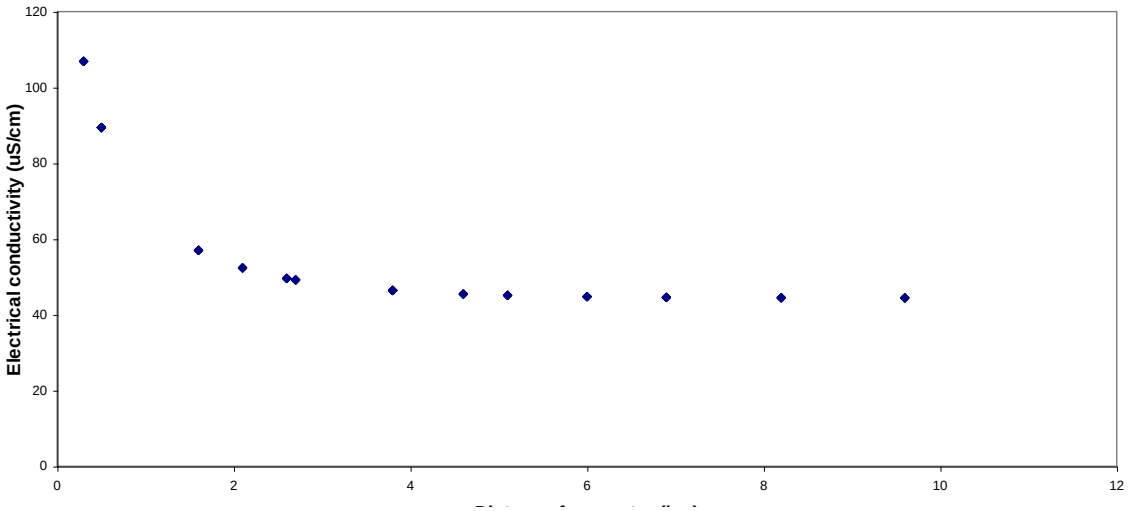


Figure 9.10 Interpreted influence of distance from water on topsoil salinity of inter-bush areas in chenopad landscapes.

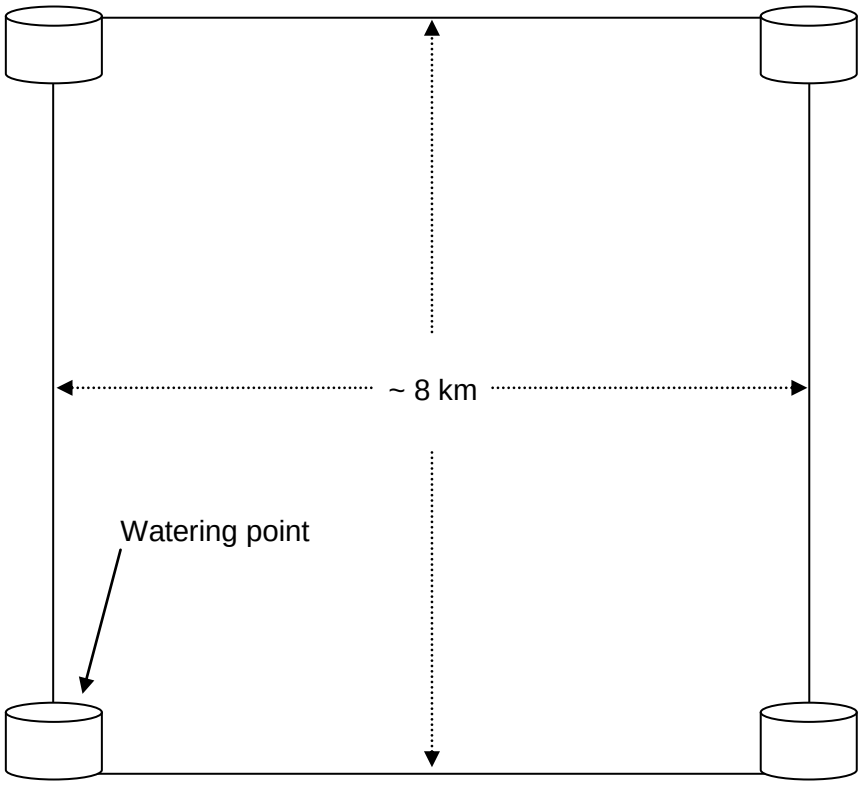


Figure 9.11 A typical paddock in Western Australia's arid zone.

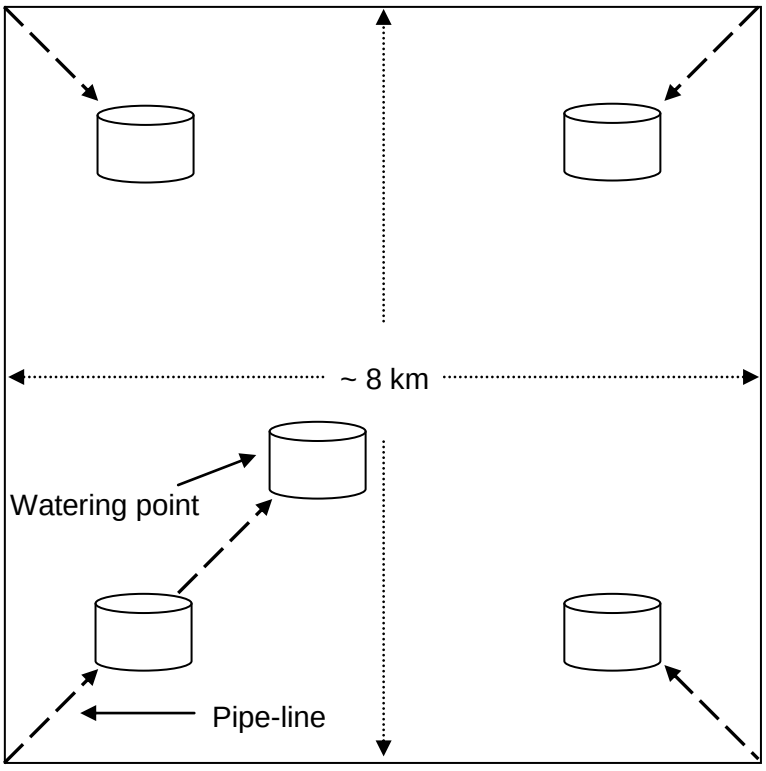


Figure 9.12 New look paddock layout to spread grazing more evenly.

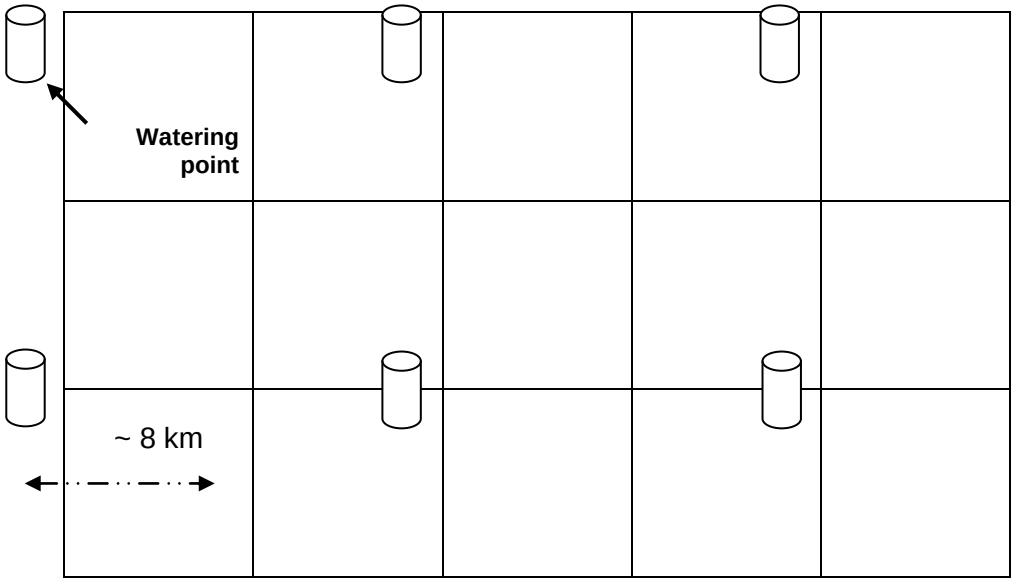


Figure 9.13 New layout to provide water-remote area Appendix 1.

APPENDIX 1. GLOSSARY

CONTENTS:

KEY LAND MANAGEMENT PERSPECTIVES

LANDSCAPE DESCRIPTORS

SAMPLING DESCRIPTORS

GRAZING DESCRIPTORS

TERMS USED IN GRAZING MODELS

KEY LAND MANAGEMENT PERSPECTIVES

(1) *Biodiversity:* The variety of all life forms - the different plants, animals and micro-organisms, the genes they contain and the ecosystems of which they form part.

(2) *Biological integrity:* The full compliment of naturally evolved organisms and the ecosystem processes that sustain them

(3) *Ecological integrity:* The capacity of landscapes to redistribute and conserve resources.

(4) *Natural capital:* The structure, stability and ability of soil to absorb and store water, and the conservation and distribution of nutrients, and the assimilation and conservation of energy.

(5) *Range condition:* The degree to which grazing has affected the long-term capacity of land to support the production of meat and fibre, relative to the potential for that type of land to do so.

LANDSCAPE DESCRIPTORS

(1) *Range type:* An area of land with characteristic landforms, soils and vegetation. This is analogous to the concepts of 'range site' (Society for Range Management 1983), 'ecological site' (Society for Range Management 1995) or 'land unit' (Pringle, Van Vreeswyk and Gilligan 1994) concepts. Pastoralists might call it 'pasture type': an area that has distinctive pasture and management characteristics.

(2) *Landscape:* A large area (at least several km^2) with a distinctive physiography and a characteristic pattern within.

(3) *Landscape type:* I studied three types of landscapes: groved mulga landscapes patterned with mulga groves and inter-groves, sandy mulga landscapes patterned with sandy banks and mulga drainage lanes, and chenopod landscapes patterned with chenopod flats/plains (+/- sandy banks) adjacent to salt lakes.

(3) *Land unit:* An area of several hectares to thousands of hectares with the same landform, major soil type and vegetation. Mulga groves and inter-groves occur on a groved plain unit, sandy banks and mulga drainage lanes are separate land units within sandy mulga landscapes, and chenopod flats/plains and sandy banks are land units in chenopod landscapes.

(4) *Landscape stratum: ('stratum')* One of the characteristic components of a landscape, such as a mulga grove, inter-grove, sandy bank, mulga drainage lane, bush mound or inter-bush area.

(5) *Patch:* A general term for a component of a higher level area, such as a mulga tree or clump in a mulga drainage lane, or 'woody patches'; which include bush mounds and mulga groves.

SAMPLING DESCRIPTORS

(1) *Paddock:* A large fenced area (often about 6 000 ha) used for retaining stock.

(2) *Watering point:* An artificial source of water from a bore or well accessing groundwater.

(3) *Grazing gradient: ('gradient')* a succession of sampling areas (sites) trending away from a watering point, all aligned within a particular quadrant (e.g. north-west) and a particular landscape type.

(4) *Site:* An area of intense sampling at a particular distance from a watering point.

(5) *Sampling locus:* A location within a site (i.e. a replicate) around which sampling is organised.

GRAZING DESCRIPTORS

(1) *Grazing activity:* all activities of stock, including herbivory, trampling, camping (resting), defecating, urinating and decomposing.

(2) *Grazing distribution:* pattern of grazing activity in space.

(3) *Grazing history:* pattern of grazing activity accumulated over time in a general and qualitative sense.

(4) *Grazing pressure:* the difference between forage availability and forage demand.

(5) *Grazing model:* A mathematical combination of variables intended to predict the distribution of grazing activity in space and/or time. Model outputs are in stocking units (e.g. sheep years per hectare).

(6) *Grazing surrogate:* A simple variable that has intuitive or empirical basis as a direct co-variate of grazing activity, but is not necessarily a precise predictor.

(7) *Grazing estimate:* A quantitative estimate of grazing history used. Three grazing estimates are used in subsequent chapters to investigate how variation in grazing and landscapes affect response variables (e.g. ant species richness). They are: Stocking History (SH), Stocking Rate (SR) and distance from water (DW).

TERMS USED IN GRAZING MODELS

(1) *Relative grazing activity (RA):* the modelled proportion of a whole paddock's grazing activity occurring in a particular place or pixel.

(2) *Stocking density (SD):* the number of stock occupying an area at an instant (e.g. sheep per hectare).

(3) *Stocking history (SH):* the accumulated exposure of an area to grazing activity over time, modelled as number of sheep years per hectare, using track density data.

(4) *Stocking rate (SR):* the long-term average number of stock occupying an area over a specific management time unit; modelled as the (long-term average) number of sheep per hectare per year. SR was developed using the RANGEPACK/Paddock model.

APPENDIX 2.
LIST OF IDENTIFIED PLANT SPECIES SHOWING RESPONSE GROUPS FOR LANDSCAPE STRATA

Nomenclature is from the Census of vascular plants of Western Australia (Green; 1985, and further supplements, which all contain the formal authorities). Some species designated 'sp.' have been included on the basis that they were different species from other, fully identified species.

Landscape strata

Chenopod landscape: **C** = chenopod plain

Sandy mulga landscape: **D** = mulga drainage lane; **W** = sandy bank

Groved mulga landscape: **G** = mulga grove; **IG** = inter-grove

Response group codes

D = decreaser; I = increaser; N = nil; S = singleton

(see Chapter 5 for definitions)

***** denotes exotic species

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Adiantaceae	*Cheilanthes sieberi*		S		N	
Aizoaceae	*Disphyma crassifolium*	N				
	Gunniopsis quadrifida	N				
	Gunniopsis septifraga	S				
	Tetragonia eremaea		S			
Amaranthaceae	*Ptilotus* sp.	S	I	I		
	Ptilotus aervoides				D	D
	Ptilotus divaricatus				S	
	Ptilotus fusiformis		S	S		
	Ptilotus gaudichaudii		N		D	D
	Ptilotus gomphrenoides	N				
	Ptilotus macrocephalus				S	
	Ptilotus obovatus	S	N	N	D	D
	Ptilotus polystachyus	S	N	N	S	S
	Ptilotus spC-33	S				
Anthericaceae	*Dichopogon capillipes*	S				
	Thysanotus speckii				N	
Apiaceae	*Daucus glochidiatus*	N			S	
	Trachymene ornata				S	
Asclepidaceae	*Marsdenia australis*	S	N	S	N	
	Rhyncharrhena linearis		S		I	

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Asphodelaceae	*Bulbine semibarbata*	N				
Asteraceae	*Actinoble oldfieldiana*		S	N	I	I
	Actinoble uliginosum		N	N	D	N
	Angianthus lanigerus	N				
	Angianthus pusillus	S				
	Angianthus pygmaeus		N		S	S
	Angianthus tomentosus	S				
	Brachyscome cheilocarpa	N			D	D
	Brachyscome ciliaris	S				
	Brachyscome ciliocarpa	N	N	N	N	N
	Brachyscome iberidifolia	N		S	N	N
	Calocephalus multiflorus	D				
	Calotis hispidula	N	N	N	D	I
	Calotis multicaulis		N	S	D	D
	Calotis spM-48		S			
	Centipeda cunninghamii		S			
	Cephalipterum drummondii		N		N	N
	Chthonocephalus pseudevax	S	N	N	N	I
	Chthonocephalus pygmaeus	N	N	N	S	N
	Cratystylis subspinescens	N				
	Erymophyllum ramosum	N			S	N
	Feldtonia nitens		S			
	Gilruthia osbornei		S	S		
	Gnephosis arachnoidea		N		S	S
	Gnephosis eriocephala	N	N	S	D	D
	Gnephosis foliata	S			D	N
	Gnephosis multiflora				S	S
	Gnephosis spJ-36	S				
	Gnephosis tenuissima	S	N	N	N	N
	Gnephosis trifida	N				
	Helipterum craspedioides		N	N	N	I
	Hyalosperma demissum		N	S	S	S
	Hyalosperma glutinosum ssp. venustrum	S				
	Isoetes graminifolia		N	S	D	D
	Lawrencia squamata	N				
	Lemooria burkittii	N	N	S	D	D
	Minuria cunninghamii	N				
	Myriocephalus appendiculatus				N	N
	Myriocephalus guerinae	S				
	Podolepis canescens		S			
	Podolepis capillaris	N	N	N	S	N

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Asteraceae	*Podolepis lessonii*		D	N		
	Podolepis spM-8		D	N	D	N
	Pogonolepis stricta	N	S		D	D
	Rhodanthe battii				S	
	Rhodanthe charleslayae		N	S	N	N
	Rhodanthe chlorocephala	N	N	N	N	N
	Rhodanthe laevis		S			
	Rhodanthe maryonii	N	N	N	N	N
	Rhodanthe propinqua	S				
	Rhodanthe spM-57		S			
	Rhodanthe stricta	N				
	Schoenia cassiniana	S	N	N	N	N
	Senecio glossanthus	N				
	Senecio lautus			S		
	Senecio spC-11	D				
	*Sonchus oleraceus**					
	*Sonchus tenerrimus**					
	Tietkensia corrickiae		N	N	N	N
	Vittadenia eremaea	N			S	
	Waitzia acuminata		S	N	S	
Boraginacaeae	*Omphalolappula concava*				S	
Brassocaceae	*Lepidium oxytrichum*		S		I	S
	Lepidium rotundum	S				
	Menkea australis	N	N		S	S
	Stenopetalum anfractum	S	N	N	S	
	Stenopetalum pedicellare	N				
Caesalpiniaceae	*Senna artemisioides*		S		S	
	Senna filifolia		S			
	Senna glutinosa		S		S	
Campanulaceae	*Wahlenbergia gracilenta*	N	N	N	N	N
Centrolepdiaceae	*Centrolepis polygala*	N				
Chenopodiaceae	*Atriplex bunburyana*	S				
	Atriplex codonocarpa	N				
	Atriplex semilunaris	S				
	Atriplex vesicaria	N				
	Chenopodium melanocarpum			S		
	Dysphania rhadinostachya					I
	Halosarcia halocnemoides	I				
	Maireana amoena	N				
	Maireana atkinsiana	D				

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Chenopodiaceae	*Maireana carnosa*	N	N	N	N	N
	Maireana convexa	S	S	N	S	S
	Maireana eriosphaera	S				
	Maireana georgei	S				
	Maireana glomerifolia	S				
	Maireana lobiflora	N				
	Maireana planifolia		N	N	I	
	Maireana platycarpa	N				
	Maireana pyramidata	N				
	Maireana spW-35			S		
	Maireana triptera		S	S		
	Maireana villosa	N	S	S	I	S
	Rhagodia eremaea		N	S	N	
	Salsola kali	S				
	Sclerolaena burbidgei	S				
	Sclerolaena cornishiana				S	S
	Sclerolaena cuneata	N				
	Sclerolaena densiflora	N	N	S	N	D
	Sclerolaena diacantha	N	N	N	D	D
	Sclerolaena eurotioides	N				
	Sclerolaena microphylla		S	S		S
	Sclerolaena obliquicuspis	S				
	Sclerolaena parviflora	S				
	Threlkeldia diffusa	N				
Cholanthaceae	*Spartothamnella teucriiflora*		N		D	
Colchicaceae	*Wurmea dioica*	S				
Convolvulaceae	*Convolvulus erubescens*	S				
Crassulaceae	*Crassula sieberiana*	N	N	N	N	N
Cuscutaceae	*Cuscuta epithymum**					
Euphorbiaceae	*Euphorbia drummondii*	N	S		D	D
	Euphorbia spD-11				N	S
Frankeniaceae	*Frankenia interioris*	N				
	Frankenia irregularis	S				
	Frankenia pauciflorus	N				
Geraniaceae	*Erodium cygnorum*	N	N	N	N	N
Goodeniaceae	*Brunonia australis*			S		
	Goodenia mimuloides	S			S	
	Goodenia occidentalis	N	N	N	N	N
	Goodenia sp.				D	S
	Goodenia spM-64		S			

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Goodeniaceae	*Scaevola spinescens*	S				
	Velleia rosea	N	N	S	N	N
Juncaginaceae	*Triglochin calcitrapa*	N				S
Malvaceae	*Abutilon cryptopetalum*		N		N	
	Abutilon otocarpum				I	I
	Hibiscus burtonii				S	S
	Sida calyxhymenia	S	S		D	S
	Sida fibulifera		S	N	I	
	Sida spC-101	S				
	Sida virgata		S	N	I	
Marsileaceae	*Marsilea drummondii*		S	S		
Mimosaceae	*Acacia aneura*		N	N	N	S
	Acacia craspedocarpa		S		S	S
	Acacia tetragonophylla		N		N	
Myoporaceae	*Eremophila battii*	S				
	Eremophila delisseri	I				
	Eremophila forrestii		N	N	N	S
	Eremophila gilesii		N	N	I	N
	Eremophila glabra	S				
	Eremophila hydrophana				D	
	Eremophila latrobei		S		D	
	Eremophila metallicorum		S			
	Eremophila scoparia	N				
	Eremophila spD-22				S	
	Eremophila spectabilis		I	I	I	
	Eremophila spI-99	S				
	Eremophila spuria				I	
Papilionaceae	*Swainsonia gracilis*	N				
	Swainsonia incei				S	
	Swainsonia kingii	N				
	Swainsonia spW-16			S		
Phormiaceae	*Dianella revoluta*		S		S	
Plantaginaceae	*Plantago debilis*	N	N		N	S
Poaceae	*Amphipogon caricinus*			S		
	Aristida contorta	N	N	N	I	I
	Aristida holathera	S				
	Austrostipa campylachne		S			
	Austrostipa elegantissima	N	S			
	Austrostipa platychaeta		S		S	
	Austrostipa scabra	N			S	
	Austrostipa spM-36		S	S		

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Poaceae	*Bromus arenarius*	N				
	Dactyloctenium radulans	S				
	Digitaria brownii		N		N	
	Enneapogon polyphyllus	N			S	S
	Enneapogon sp	D				
	Enteropogon acicularis	N				
	Eragrostis dielsii	N	N	S		N
	Eragrostis eriopoda		N	N	N	S
	Eragrostis falcata	D				
	Eragrostis lanipes		S	N		
	Eragrostis pergracilis	N	S		N	N
	Eriachne helmsii			S	S	
	Eriachne pulchella	N	S	N	N	I
	*Hordeum leporinum**	S				
	Iseilema eremaea	S				
	Monachather paradoxa		N	N	I	S
	Nothodanthonia caespitosa	S				
	Paspalidium gracile	N			N	N
	*Pentaschistis airoides**					
	*Rostraria cristata**					
	Sporobolus australasicus				N	
	Thyridolepis multiculmis		S	N		
	Tragus australianus	N				
	Tripogon loliiformis	N	N		N	S
	*Vulpia myuros**	S				
Polygalaceae	*Polygala isingii*				N	N
Polygonaceae	*Emex australis**	S				
Portulacaceae	*Calandrinia brevipedata*	N				
	Calandrinia eremaea		S	N		N
	Calandrinia polyandra	N	S	N	N	N
	Calandrinia primuliflora	S	N			
	Calandrinia spJ-18	S				
	Calandrinia uniflora	N				
	Portulaca oleracea		S		N	D
Primulaceae	*Anagallis arvensis**					
Proteaceae	*Grevillea berryana*				N	
	Hakea preissii	N				
	Hakea recurva			S		
Rubiaceae	*Canthium attenuatum*				S	
	Canthium lineare				S	
	Synaptantha tillaeacea	N	S			

Family	Scientific name	Landscape strata				
		C	D	W	G	IG
Santalaceae	*Santalum lanceolatum*				S	
Solanaceae	*Nicotiana occidentalis*	S			I	
	Nicotiana rosulata				S	
	Solanum ferocissimum				S	
	Solanum lasiophyllum	S		S	I	N
	Solanum orbiculatum	S				
	Solanum terraneum				S	
Zygophyllaceae	*Tribulus astrocarpus*				N	D

APPENDIX 3.

ANT SPECIES LIST SHOWING RESPONSE GROUPS FOR LANDSCAPE STRATA

Nomenclature is from Shattuck (1999), species codes are related to specimens held by Alan Andersen, CSIRO Darwin. Commas within species codes indicate species composites undifferentiable without resorting specimens (seven species). Unidentified specimens have been removed from the list (four specimens).

Landscape strata

Chenopod landscape: **C** = chenopod plain

Sandy mulga landscape: **D** = mulga drainage lane; **W** = sandy bank

Groved mulga landscape: **G** = mulga grove; **IG** = inter-grove

Response group codes

D = decreaser; I = increaser; N = nil; S = singleton

(see Chapter 6 for definitions)

Family	Scientific name	Landscape strate				
		C	D	W	G	IG
Aenictinae	Aenictus spA	S				
Dolichoderinae	Dolermyrma spA	N				
	Iridomyrmex ?bicknelli	N				
	Iridomyrmex agilis	N	S	N	N	S
	Iridomyrmex discors	S				
	Iridomyrmex greensladei	D	S	S	S	S
	Iridomyrmex mayri		S			
	Iridomyrmex nr. greensladei		N	D	N	N
	Iridomyrmex spA		S			S
	Iridomyrmex spAA	N				
	Iridomyrmex spB			S		
	Iridomyrmex spBB	N				
	Iridomyrmex spC		S	N	N	N
	Iridomyrmex spD		S	N	I	N
	Iridomyrmex spE		S	N	N	I
	Iridomyrmex spF	N	N	N	N	N
	Iridomyrmex spG				S	S
	Iridomyrmex spH255	S				
	Iridomyrmex spH255		I		S	N
	Iridomyrmex spI		N	N	N	I
	Iridomyrmex spJ		S			S

Family	Scientific name	Landscape strate				
		C	D	W	G	IG
	Iridomyrmex spK	N		N		
	Iridomyrmex spL		S		S	S
	Iridomyrmex spM		S	S	S	S
	Iridomyrmex spN					S
	Iridomyrmex spO		N	S		
	Iridomyrmex spP		I	I	N	N
	Iridomyrmex spQ	I		S	S	S
	Iridomyrmex spR		S		S	
	Iridomyrmex spS		I		N	N
	Iridomyrmex spT		S			
	Iridomyrmex spU	N				
	Iridomyrmex spV	S				
	Iridomyrmex spW	D				
	Iridomyrmex spX	N				
	Iridomyrmex spY	D				
	Iridomyrmex spZ	I				
	Tapinoma spA		N	N	N	N
	Tapinoma spB			S	D	S
	Tapinoma spC				S	N
	Tapinoma spD	N				
	Tapinoma spE	D				
	Ochetellus sp	S				
Formicinae	Calomyrmex nr. cyanea	S				
	Camponotus ?leae			D		
	Camponotus dryandrae		S	N		N
	Camponotus nr. cyanea		N		N	S
	Camponotus nr. Whitei		S			
	Camponotus spA			S		
	Camponotus spB				S	
	Camponotus spC		S			
	Camponotus spD				N	
	Camponotus spE		N	D	N	N
	Camponotus spF			S		S
	Camponotus spG	N				
	Camponotus spH	D				
	Camponotus spI	S				
	Camponotus spJ	S				
	Melophorus spA	N	N	I	N	N
	Melophorus spAA	S				
	Melophorus spB	N	S	N		S
	Melophorus spBB	D				
	Melophorus spCC	I				
	Melophorus spCC,DD	D				

Family	Scientific name	Landscape strate				
		C	D	W	G	IG
	Melophorus spCC,SY	D				
	Melophorus spD			S		
	Melophorus spDD	D				
	Melophorus spDD,S,Y	D				
	Melophorus spE	I	N	N	N	N
	Melophorus spEE	S				
	Melophorus spEE,CC	N				
	Melophorus spF		S	S		
	Melophorus spG		S			
	Melophorus spH			S	S	S
	Melophorus spH167	S				
	Melophorus spH181	S				
	Melophorus spI					S
	Melophorus spJ		N	N	N	N
	Melophorus spK		S			
	Melophorus spL				N	
	Melophorus spM		S	I	N	I
	Melophorus spN		D	N	I	N
	Melophorus spN(brunea gp)	D				
	Melophorus spN(mjombergi gp)	D				
	Melophorus spO		N	N	N	N
	Melophorus spP		S			
	Melophorus spQ	S				
	Melophorus spR	D				
	Melophorus spS	S				
	Melophorus spT	D				
	Melophorus spU	S				
	Melophorus spV	D				
	Melophorus spW	N				
	Melophorus spX	I				
	Melophorus spY	N				
	Melophorus spZ	N				
	Opisthopsis spA		S			
	Polyrachis spA		S	S	S	
	Polyrachis spB		S			
	Polyrachis spC			S	S	S
	Polyrachis spD			S		
	Polyrachis spE					I
	Polyrachis spF				N	
	Polyrachis spG					S
	Polyrachis spH				N	
	Polyrachis spI				S	N
	Polyrachis spJ				S	

Family	Scientific name	Landscape strate				
		C	D	W	G	IG
	Stigmacros spA		N	N	I	
	Stigmacros spB		S	S	S	
	Stigmacros spC		S			
	Stigmacros spD				N	
	Stigmacros spE	S			S	
Myrmicinae	Colobestroma spA	S				
	Epopostruma spA				S	
	Monomorium ?fieldi	D				
	Monomorium ?whitei	S				
	Monomorium sp75		S			
	Monomorium spA	S	N	S	D	N
	Monomorium spAA	N				
	Monomorium spB	N	N	N	N	N
	Monomorium spBB	S				
	Monomorium spC		N	D	N	N
	Monomorium spCC	N				
	Monomorium spD	S			S	
	Monomorium spDD	I				
	Monomorium spE	N	N	N	N	I
	Monomorium spEE	N				
	Monomorium spF			D	N	N
	Monomorium spFF	S				
	Monomorium spGG	N				
	Monomorium spGG			N		N
	Monomorium spH		S	S	S	N
	Monomorium spHH	S				
	Monomorium spHS28	N				
	Monomorium spI	N	S	S	I	N
	Monomorium spJ		N	N	N	I
	Monomorium spK	N		S	S	S
	Monomorium spL		S			
	Monomorium spM		N		S	S
	Monomorium spN		S			
	Monomorium spO		S			
	Monomorium spP		I			
	Monomorium spQ					S
	Monomorium spR	N		S		
	Monomorium spS		S	S		
	Monomorium spT		S			
	Monomorium spU		S			
	Monomorium spV	N		N		S
	Monomorium spW	D				
	Monomorium spX	N				S

Family	Scientific name	Landscape strate				
		C	D	W	G	IG
	Monomorium spY	D				
	Monomorium spZ	N				
	Pheidole sp1		S			
	Pheidole spA		N	N	N	S
	Pheidole spB		N	S	S	N
	Pheidole spC		S		S	
	Pheidole spD		S		S	S
	Pheidole spE	N		N	S	S
	Pheidole spF		S			N
	Pheidole spGG		N		S	D
	Pheidole spH		N		S	
	Pheidole spI			S		S
	Pheidole spJ		S			
	Pheidole spK	I				
	Pheidole spL	N				
	Pheidole spM	N				
	Pheidole spN	N				
	Pheidole spO	D				
	Pheidole spP	N				
	Pheidole spQ	D				
	Pheidole spR	S				
	Pheidole spS	D				
	Solenopsis spA		S		S	
	Tetramorium spA		S	S	N	
	Tetramorium spAP		S	N	N	
	Tetramorium spAT					S
	Tetramorium spAU					S
	Tetramorium spB		S			
	Tetramorium spC		I			S
	Tetramorium spD			N	S	
	Tetramorium spE			I	N	N
	Tetramorium spF			S	S	S
	Tetramorium spG		S			S
	Tetramorium spH			N	D	N
	Tetramorium spH1			S	S	
	Tetramorium spI			S	S	
	Tetramorium spJ				S	S
	Tetramorium spK		N	N	S	N
	Tetramorium spK1			S		
	Tetramorium spL,N,O,P	N				
	Tetramorium spL,S,T,U	D				
	Tetramorium spL,U,P,Q	D				
	Tetramorium spM				S	

Family	Scientific name	Landscape strate				
		C	D	W	G	IG
	Tetramorium spN		S			
	Tetramorium spN1		S			
	Tetramorium spO			S		
	Tetramorium spT				S	
	Tetramorium spW			S	N	
	Tetramorium spX				N	
	Tetramorium spY		S	S	N	S
Ponerinae	Anochetus spA	S	S		S	S
	Bothroponera spA	S				
	Brachyponera lutea	S	S			
	Cerapachys ?princeps			S		
	Cerapachys nigraventris	S				
	Cerapachys spA		S		S	
	Cerapachys spB				S	
	Cerapachys spC	S	S	S	N	
	Cerapachys spD	D				
	Cerapachys spE	N				
	Leptogenys nr. aderzi		S	S	N	S
	Odontomachus spA		N		S	S
	Rhytidoponera ?metallica	D				
	Rhytidoponera nr. maniae	I				
	Rhytidoponera spA		S	S	I	I
	Rhytidoponera spB	N	N	N	I	I
	Rhytidoponera spC	S			I	
	Rhytidoponera spD		N	S	N	
	Rhytidoponera spE		N	N	N	
	Rhytidoponera spF	S				S
	Rhytidoponera spG				S	S
Pseudomyrmecinae	Crematogaster spA				N	
	Crematogaster spA				S	
	Meranoplus spA		S		S	S
	Meranoplus spB		N	N	N	
	Meranoplus spC		I			
	Meranoplus spD			S		
	Meranoplus spE		S	S	S	
	Meranoplus spF				S	
	Meranoplus spH		N		N	S
	Meranoplus spI	D	S	N		
	Meranoplus spK	N				
	Meranoplus spL	S				
	Meranoplus spT				S	
	Meranoplus spW		S			
	Meranoplus spZ				N	S

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