The Role of Wilderness in Nature Conservation

B.G. Mackey, R.G. Lesslie, D.B. Lindenmayer,
H.A. Nix and R.D. Incoll

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The School of Resource Management and Environmental Science
The Australian National University
Canberra ACT 0200

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Executive Summary

Overview

This report was commissioned by Environment Australia, Australian and World Heritage Group, World Heritage and Wilderness Branch as a background briefing for internal policy formulation within the portfolio. The contract brief required that we examine the role of wilderness in nature conservation and that we:

1. Review, assess and document information and published discussion on the significance of the wilderness condition of places, for nature conservation, addressing the broad meaning of the term 'nature conservation'.
2. Undertake quantitative analysis of the role of wilderness for nature conservation using available data and applying the methodology proposed in the tender.
3. Describe the role that wilderness areas can, or can not, play in nature conservation and discuss the range and prevailing themes of current theory relevant to the nature conservation role of wilderness, indicating where this is relevant to the Australian environment.
4. Prepare principles for incorporating the wilderness concept into strategic conservation planning and management noting the range and possible conflict of nature conservation objectives.
5. Provide a report documenting the above, including an executive summary of major findings.

Given these terms of reference, we begin by acknowledging that both wilderness and nature conservation are cultural concepts and we review their history and development. Next we assess processes that threaten species and ecological conservation and then evaluate relationships between these threatening processes, 'wilderness' and 'nature conservation' drawing upon current ecological theory within an Australian context. Generally, 'wilderness' and 'nature conservation' values are inversely related to current levels of disturbance by modern technological society as well as the presence of threatening processes. Most importantly, we distinguish between attempts to define 'wilderness' as a binary condition (in/out) based on a defined threshold and 'wilderness quality' as a continuum that can be estimated using explicit, repeatable and quantitative methods.

The National Wilderness Inventory (NWI) for Australia developed a particular set of indicators based on the continuum concept and these were used in an attempt to assess possible correlations between indicators of wilderness quality (inversely related to level disturbance) and numbers of threatened plants and animals. National-level data on the distribution of numbers of threatened vertebrate animal species and plants were intersected with NWI Indices. The results are indicative, but equivocal. Generally, high numbers of threatened species correlate with low wilderness quality, but both the coarse spatial resolution of the species data and their aggregate form, as well as the absence of indicators for key threatening processes (such as feral pests and predators in the NWI) militate against any firm conclusions. More detailed analysis of selected regions that have soundly based digital data on plant and animal distributions will be a necessary condition for resolution of questions posed in this study.

In the interim, we explored the need for environmental context when interpreting wilderness quality. We show how NWI ratings vary in relation to:

a. IBRA regions (Interim Biogeographic Regionalisation of Australia).

b. An environmental domain classification based on terrain attributes.

Both of these regionalisations are indicative of environmental heterogeneity, known to be important for biodiversity and hence 'nature conservation' and for concepts of representativeness in reserve design.

Finally we develop a synthesis which draws upon a very extensive literature review and the foregoing indicative analyses. We conclude by proposing a set of notional principles for incorporating wilderness concepts into strategic conservation planning and management, accepting that these provide a platform only for extensive and continuing development.

Main conclusions

'Wilderness' has been criticised on a number of levels, *viz*. that it:

- is only a cultural concept
- has no objective, empirical and hence scientific basis
- is irrelevant to nature conservation in general, and biodiversity conservation of in particular

Additionally:

- wilderness areas tend to be spectacular icon areas that are protected at the expense of more threatened and smaller habitat areas

- the conservation of biodiversity is ultimately dependent on off-reserve management rather than wilderness areas, there being a fundamental limit to the amount of land that can be reserved
- because humans have inhabited Australia continuously for at least 40,000-50,000 years 'wilderness' has little or no relevance
- biodiversity conservation requires a focus on the habitat of rare, endangered and threatened species, whose distributions may not be represented in 'wilderness areas'.

The validity of these criticisms is very dependent on matters of definition.

1. Definitions of 'wilderness'

A critical distinction must be made between concepts and definitions of (a) wilderness quality and (b) wilderness area:

a. *Wilderness quality* is the extent to which any specified unit area is remote from and undisturbed by the impacts and influence of modern technological society.

b. *Wilderness areas* are places where wilderness quality is recognized and valued by society and are defined using arbitrary thresholds of remoteness, naturalness and total area.

Given this important distinction we argue that:

- variation in wilderness quality across the landscape can be measured using explicit, repeatable and quantitative methods. The National Wilderness Inventory (NWI) uses a particular set of indicators developed by one of the authors (RGL)
- wilderness quality is defined as a function of levels of disturbance associated with modern technological society and, as such, does not deny the reality of aboriginal history
- wilderness areas are indeed cultural constructs to the extent that threshold criteria are intrinsically value-based and their existence is fundamentally controlled by the demand for and supply of remote and natural places.

2. Ecological integrity and species conservation

The conservation of biodiversity in practice has a strong species-based focus, often with a particular emphasis on the conservation of rare or threatened species. From this perspective, the challenge for *in situ* conservation is to identify a set of critical habitat features for a species, and ensure that land management proceeds in such a way that these are maintained in the landscape. On this basis, off-reserve management is an essential component. There is no doubt that this approach to nature conservation is critical and must continue.

However, at a more general and global level there is a focus on the ecological systems and processes which underpin life on Earth. The key characteristics of ecological systems are that (a) they are self-regenerating, (b) the biota have a capacity to self-regulate their environment, and (c) they have resilience, that is, they are able to absorb the impact of external perturbations up to a point - beyond which they flip into a different (usually a lower energy) system state.

It is critical to recognise these two, complementary, dimensions to nature conservation as the relevance of *wilderness quality* and *wilderness areas* will vary depending on where our focus lies. For example, a given species may survive in a landscape where the resilience of the dominant landscape ecosystem (and hence its role in regional and global ecosystem processes) is destroyed. In these cases, the concept of a *wilderness area* may not be relevant to management of the species. Rather the critical question becomes to what extent *wilderness quality indicators* reflect the impact of processes that threaten the persistence of the species in the landscape.

As a general rule, systems must be large enough to incorporate the impact of the largest scaled perturbation. This can be readily appreciated by considering the impact of fire, where a small reserve might be burnt out, whereas a large, environmentally heterogenous areas is more likely to contain unburnt areas. Similarly we can consider the potential effects of global warming, which will involve large scale changes in meso-climatic regimes. In these circumstances *wilderness areas*, by definition large, may contribute significantly to the long term integrity of ecological systems - even though the thresholds were not defined with this purpose in mind.

There is no theoretical conflict between species conservation and ecological conservation, as species habitats are emergent properties of ecosystems, which in turn are comprised of populations of species. But in practice, scientists tend to focus on one rather than the other, and public policy generally regards them as distinct, if not unrelated, areas of concern.

3. Indicators of threatening processes

Wilderness quality indicators are by definition indicators of the extent to which modern industrial society has impacted the pre-European landscape. They may also indicate processes that threaten the healthy functioning of ecosystems, and the persistence in a landscape of populations of native species. We have identified a number of key threatening processes that stem from the impact of modern technological society, namely:

- changing fire regimes
- changing hydrological regimes
- roading
- changing vegetation cover
- introduced species
- accelerated global change.

These processes rarely act in isolation from each other, and their impact varies with the environmental context and the target species. Our literature review examined their impact on nature conservation (in particular species conservation) and the potential correlation with the National Wilderness Inventory (NWI) indicators. We concluded the following:

- there are insufficient data for pre- and post-colonisation fire regimes to permit evaluation of the extent to which any landscape is disturbed
- the intensity, frequency and type of fire is not necessarily located with reliably measured material evidence of modern technological society
- the extent to which NWI indicators capture changed fire regimes is therefore unknown
- many of the impacts on river systems and hydrological regimes are spatially correlated with infrastructure developments like dams and weirs. Consequently, wilderness quality indicators related to the presence, density or distance from such structures may be a good predictor of these threatening processes
- roads have a number of adverse impacts on nature conservation associated from their establishment including the isolation and fragmentation of populations, the enhanced dispersal of weeds and ferals, and are a major source of mortality for populations of animals as a result of collisions with motor vehicles
- these roading impacts should be strongly captured by NWI indicators
- changes in the vegetation includes such affects as vegetation loss and fragmentation, degradation of vegetation structure and alteration to the floristic composition
- NWI indicators record the relative intensity of a given land use - grazing, agriculture, logging, and consequently will generally be correlated with these threatening processes
- the impact of introduced predators and herbivores in certain arid environments appears to have been critical in the decline and extinction of certain critical weight range mammals
- the feral predators and herbivores of concern here have spread away from much of the infrastructure and land use associated with modern technological society, and therefore are not always correlated with NWI indicators.

The NWI indicators therefore should capture the signal of many threatening processes. However there are certainly some key threatening processes for which the NWI indicators are inadequate, and for which additional indicators need to be developed. There are also some threatening processes for which it may be technically impossible to develop suitable disturbance indicators.

4. Relevance of high wilderness quality to nature conservation

A review of conservation theory suggests the following:

- large reserves are usually better that small reserves
- large populations or connected populations in a metapopulation are usually better than small populations
- certain human actions may elevate extinction risk of a species
- fragmentation may reduce the amount of habitat, increase edge-effects, and subdivide and isolate populations
- resilience requires maintenance of the evolved primary productivity in a landscape which is defined by the dominant autotrophs, decomposers, and other taxa, that maintain the landscape’s resource ‘infrastructure’.

Many of the disturbances associated with modern technological society cause fragmentation, degrade the native vegetation, and elevate extinction risk. It follows, that wilderness areas and places with a high wilderness quality, all other things being equal, will provide for larger reserves, support larger or better connected metapopulations, reduce extinction risk, be less fragmented, and posses greater resilience.

5. Data analysis

We examined the hypothesis that areas with low wilderness quality correlate with areas that have higher numbers of rare, threatened, or endangered species. However this work was severely constrained by the lack of relevant data - at required levels of spatial resolution:

- within the available time, we were able to obtain aggregated data relating to the distribution of certain threatened plant and animal groups, and subject these to simple, descriptive, exploratory data analysis techniques

- the report discusses in detail the many limitations of the data and the analyses. More analysis is required using better field data and focussing on suites of taxa assigned to more refined groups (e.g. functional groups or foraging guilds).

Nonetheless, some important conclusions could still be reached:

- there is a broad, overall trend, mainly driven by the vascular plant data, indicating that the number of threatened species decreases as total wilderness quality as measured by the NWI increases, i.e. larger numbers of threatened species are found in areas characterised by the intense impacts of modern technological society
- on this basis, for some species groups and in many environments, the NWI indicators appear potentially useful measures of both impacts and threatening processes.

Such a general trend was not evident for threatened mammals, rather the relationship is far more complex (i.e. positive in some cases, negative in others):

- certain areas in the arid zone have high threatened species numbers and high wilderness quality
- this may be due to the impact of exotic animals on key refuge areas in patchy, low productivity environments
- while these impacts are a consequence of modern technological society they are not captured by the current NWI indicators.

Following from these results, we also tried to demonstrate how the ecological interpretation of wilderness quality data (such as derived from the NWI) varies with the environmental context:

- only by examining wilderness quality within an environmental context can relationships be established between wilderness quality indicators and threatening processes; for example, the ecological impact of roading is different in moist forest ecosystems compared with arid shrubland
- this was illustrated by showing how wilderness quality varies across IBRA regions
- another perspective was given by generating a continental terrain classification, and mapping how wilderness quality varies across selected terrain units.

We recommend further research along these lines as environmental context is the key to interpreting the ecological significance of wilderness quality data.

6. Development and implementation of conservation strategies

Nature conservation can only be achieved through an integration of on- and off-reserve strategies. In both areas, wilderness quality data and wilderness areas can make important contributions. Though the role they play will vary depending on the kind of landscape under consideration. We have identified four broad landscape classes:

a. *High quality reserve landscapes* - these are large, dedicated nature conservation reserves of high ecological integrity.

Dedicated reserves provide maximum legislative protection from threatening processes. Reserves need to be:

- large enough to absorb large-scaled perturbations
- spatially configured to promote flow and migration of genomes
- representative (in terms of taxonomy, community organisation, productivity, and environment)
- possessed of the highest possible ecological integrity.

These qualities are likely to be found in, or promoted by, *wilderness areas*. Hence they should where possible form the core of a dedicated reserve network.

In many cases, low ecological integrity will have to be included in the reserve system as high integrity places are lacking due to land use history. No *wilderness areas* may be left in these landscapes, however *wilderness quality data* can help identify places that possess the highest ecological integrity for a given ecosystem type, that is, it can help identify ‘the best of what is left’.

b. *Restoration reserve networks* - these are large dedicated reserves which may presently have low ecological integrity, but are the best available examples of an ecosystem type.

Landscapes that are being managed in order to restore the ecological integrity will, amongst other things, aim to reduce the impact of threatening processes associated with modern technological society. Hence wilderness quality data can be used to monitor the success of such ecological restoration programs.

c. *Remnant landscapes* - highly disturbed landscapes which have been largely cleared and replaced by exotic vegetation; only remnant patches of native vegetation remain.

d. *Production landscapes* - areas dominated by native vegetation from which natural resources are harvested.

In both these landscape types, small remnant patches may be critical for the continued persistence of viable populations of a given species. *Wilderness areas* may be absent, but *wilderness quality data* can be used to track variation in disturbance over time at these core locations. In the case of production landscapes, wilderness quality data can also be used to monitor the state of the surrounding landscape matrix. The condition of the matrix can be critical to the viability of the populations in the habitat patches.

7. Integrated landscape conservation

All other factors being equal, a landscape of high wilderness quality will better promote nature conservation objectives than one with low wilderness quality:

- this general result holds even in those areas where NWI values are shown to correlate poorly with critical threatening processes
- the SLOSS (single large or several small) argument is only relevant to issues of wilderness quality and reserve design when dealing with small remnant patches in a heavily disturbed environment
- all other factors being equal, a very large area of high wilderness quality will always be the preferred conservation strategy.

A national assessment of the conservation reserve network must include an Optimal Gap Analysis, that is, identification of 'the best of what is left'. This is complementary to the notion of a CAR (comprehensive, adequate and representative) system.

However due to the extensive ecological degradation that has occurred over the last 200 years, nature conservation requires the contribution of locations from all four landscape types noted in #6 above, and hence the contribution of places with relative low wilderness quality. This can only be achieved through strategic planning and partnerships at national, regional and local scales. For example, within a given region, core wilderness areas must be complemented with remnant patches, corridors, regenerating sites, and buffers, in neighbouring more disturbed landscapes.

The wilderness protection regime set in place by the South Australian Government Act (1992) provides a good model of how the concept of wilderness quality can be used as a management tool. Here, the emphasis is on the wilderness continuum and high wilderness quality is established as a planning and management goal. The wilderness concept therefore can be applied as an instrument for ecological restoration.

8. Ecological integrity

Faced with the daunting challenge of ensuring the maintenance of ecological integrity we know that both the concepts and tools are currently inadequate for the task. However we do know that a species-by-species approach while necessary is in itself insufficient to ensure the protection of system-level characteristics such as ecosystem resilience. Some other important conclusions can also be made:

- wilderness areas, and locations with high wilderness quality, make a critical contribution to global biogeochemical cycles, and to the role the biota play in regulating Earth's environment
- because wilderness areas by definition are relatively large, in certain environments they will span a range of climatic gradients and therefore potentially provide refugia for certain species, or will facilitate species migration, in the face of accelerated global climate change
- high wilderness quality landscapes retain evolved vegetation communities which may represent the maximised primary productivity given prevailing environmental conditions and disturbance regimes.

9. Management principles

The following management principles emerged from our review *(management* is considered here from both a strategic and operational perspective):

- management aimed at promoting wilderness quality in an area may provide for uses that are compatible with the objective of minimising the impacts of modern technological society
- land that is allocated for protection of wilderness quality may still be subject to threatening processes, and hence may require active management to ameliorate their impacts
- the use of the wilderness concept in management is underpinned by the notion of the wilderness continuum; all other factors being equal (and assuming that an appropriate set of indicators have been used), higher wilderness quality indicates 'the best of what is left' for nature conservation purposes
- as defined here, low wilderness quality means that a landscape has been heavily modified by exposure to modern technological society; therefore those threatening processes associated with that exposure will also be present
- wilderness areas provide important opportunities for maintaining ecosystem integrity
- a conservation strategy based on wilderness areas only will be inadequate due to the the severe level of disturbance now experienced by

many Australian landscapes; production landscapes and restoration landscapes therefore also have critical contributions to make

- an integrated landscape conservation strategy will have wilderness areas as the core, complemented with 'the best of what is left'.

Development of the concept of wilderness

Introduction

The concept of ‘wilderness’, like most other constructs under the broad umbrella of nature conservation, has changed substantially during the last century. If our objective is to understand its current role in nature conservation, and perhaps gain insight into its future purpose, it is necessary to have some appreciation of the historical shifts in meaning and use over this period. It also needs to be understood that the traditional concept of wilderness is an artefact of New World societies. Wilderness requires a frontier; a benchmark or break-point which separates the structure and function of natural systems from systems dominated by agricultural and industrial enterprises. Within societies where the breakpoints between natural and managed systems are subtle or non-existent, or where economic and social activity is not easily distinguished from the broader environmental setting, the traditional notions of wilderness have little or no meaning. As discussed in more detail below, this is the case with indigenous Australian societies.

Shifts in the concept of wilderness, and its role in nature conservation, have occurred in response to two driving forces. The first is evidence of rapid and sustained transformation of natural systems due to agricultural and industrial development. The second, a product of the first, is public awareness and appreciation of the value of natural systems. This century has seen public concern for both wilderness and nature conservation extend from spiritual and aesthetic concerns, through narrowly defined interests, such as the reservation of environments that provide particular recreation and other nature-based benefits, or the protection of species with special appeal, to concerns which encompass natural systems and processes as a whole. Today, terms once restricted to the scientific literature, such as biological diversity, and ecosystem integrity now appear frequently in the media and more popular writing. Notions of wilderness are now often invoked within this wider context.

It is not the purpose of this discussion to re-articulate the history of the wilderness concept in nature conservation, as this has been done exhaustively elsewhere (in Australia, c.f. Mosley 1978, Robertson *et al.* 1992, Griffiths 1996). It is, however, necessary to briefly review this history to discern trends in the application and use of the concept. It is also important to set a framework against which we have undertaken our analyses and also related broad concepts of conservation science. The development of the wilderness concept in Australia began in the late 1800s, reflecting changing perceptions of the natural environment in Australia, United States and Europe. Early Australian environmentalists were inspired by ideas from the northern hemisphere, particularly the US, which was the focus for development of the wilderness concept.

Development of wilderness concept in U.S.

During the early and mid 1800s, the North American philosophers, Emerson, Thoreau and others greatly influenced western perceptions of the natural environment, writing of its capacity to inspire. The perception of the environment in spiritual, even mystical terms, was a dramatic change from the views of the early settlers and pioneers. For them, the wilderness was to be conquered and civilised; their survival depended upon it. Concurrent with these changing perceptions was a closing frontier and an increasingly urbanised population. City dwellers were beginning to seek recreational opportunities in the wild. National parks were established at Yellowstone in 1872 and subsequently at Yosemite in 1890. The declaration of these parks had little to do with the conservation of wilderness (Nash 1978). The congressional act that declared Yellowstone established a ‘public park or pleasuring ground for the benefit and enjoyment of the people’ (US Statutes at Large, 17, cited in Nash 1978). Tourism in these parks relied on convenient transportation (railways), hotel accommodation, and spectacular scenery.

However, during the 1910s, values began to change. The US Forest Service, which managed public land chiefly for forestry and recreational purposes, began to recognise value in undeveloped areas. From this time, the Forest Service actively advanced the cause of wilderness protection (Leopold 1921) and in 1924 designated the 232,000 ha Gila National Forest in New Mexico for wilderness recreation. Recreation was the sole sanctioned purpose for reserved wilderness (Gilligan 1953). ‘Wilderness areas are first of all a series of sanctuaries for the primitive arts of wilderness travel, especially canoeing and packing’ (Leopold 1949, p. 248). Regulations banning roads, hotels, permanent camps and logging in wilderness areas were introduced in 1939 (Baldwin 1972 cited in Nash 1978).

Post-war concerns in the USA over increasing resource-driven demands for access to, and exploitation of, remaining wild lands, prompted vigorous campaigning for federal legislation to protect wilderness. In 1964, the US Wilderness Act was passed. This act, and subsequent legislation, established the National Wilderness Preservation System (NWPS) and a legislative and administrative framework for the review, evaluation and designation of wilderness on federal lands. Exhaustive roadless area review and evaluation processes conducted by the major federal land-holding agencies have been continuing, amid controversy, since the passage of the act. One feature of the contention that surrounds the wilderness evaluation and assessment from the 1970s to the present is the 'purism' debate; the issue being how 'wild' areas need be for inclusion within the NWPS. The NWPS today contains the majority of the land area set aside within the US for nature conservation. Recreation remains a primary tenet of the system, but increasingly the focus is shifting to biological and ecological values. Wilderness is now strongly promoted as a key element in the maintenance of biological and ecological values (Grumbine 1990). The NWPS is increasingly recognised as the core estate around which efforts to protect ecological diversity and integrity should be centred (Noss 1991, Noss and Cooperrider 1994).

Development of wilderness concept in Australia

During the late 1800s, urban Australians were beginning to develop an appreciation for antipodean environments. Sentiment was aroused by artists such as the Heidelberg impressionists, Roberts, Streeton and Conder and poets like Lawson and Patterson. Photographers such as Lindt and Caire also increased people's enthusiasm for the Australian bush. The period saw a number of organisations established to promote the enjoyment of the Australian environment. For example, the Field Naturalists Club of Victoria was founded in 1880, the Melbourne Amateur Walking and Touring Club in 1894 and Walhalla Mountain Association in 1907. At the turn of the century, writers such as Donald Macdonald and Charles Barrett were encouraging interest in Australian environments (Griffiths 1989); an important step towards an appreciation of Australian wilderness.

The first National Park in Australia and the second in the world was established at Point Hacking in NSW (now the Royal National Park) and, in the tradition of Yellowstone National Park, the imperative for protection was recreation (Mosley 1978, Ramsey 1995). The establishment of national parks followed throughout Australia. Bardwell (1974) concludes that the factors important in the establishment of the first national parks were urbanisation, the rapid growth in popularity of outdoor recreation, expansion of the conservation and national park movements, and trends in relationship between humans and nature. In 1933, Myles Dunphy formed the National Parks and Primitive Areas Council (NPPAC) to promote the establishment of primitive areas (equivalent to wilderness areas) in NSW. The NPPAC believed that national parks could be divided into tourist areas for development, and primitive areas for wilderness recreation. The NPPAC were happy for the primitive areas to be established in rugged, unproductive country and national parks in more developed and accessible locations (Mosley 1978). Dunphy proffered as a primary justification for wilderness protection the 'recreational purposes of man-kind, where he can rid himself of the shackles of ordered existence ... to escape his civilisation' (Dunphy 1934, p.202)

Dunphy produced a plan of primitive areas in NSW, leading to the establishment of Tallowa Primitive Area in 1934, Morton Primitive Area in 1938, Heathcote Primitive Area in 1943, and in 1944 the reservation of Mount Kosciusko State Park which provided for 10% to be nominated as a primitive area. The delineation of the primitive area proved to be a contentious issue, which highlighted the contrasting views on wilderness value. Scientists from organisations such as the Royal Zoological Society of NSW and the Linnean Society of NSW wanted access to be restricted for scientific purposes only (Mosley 1978). Dunphy objected strongly to this (Johnson 1974). In 1962 the Kosciusko Primitive Area was finally set aside for science and recreation. In the following years, various pieces of legislation were introduced concerning wilderness. The NSW National Parks and Wildlife Act of 1967 included clauses for wilderness areas within National Parks and State Parks. The act determined that only simple survival huts would be permitted in wilderness and all areas should be maintained in a wilderness condition (Mosley 1978).

Demand for the recognition and conservation of wilderness gathered impetus during the 1970s and 1980s, the concept being strongly promoted by the nature conservation movement, and becoming established in the public mind as a goal for nature conservation. Several landmark conservation development conflicts, like the flooding of Lake Pedder in 1972, and the successful campaign to prevent damming of the Franklin River in 1983, provided foci for support and action. Legislative provisions specifically for wilderness protection appeared in a number of states in response. Other developments during these decades included the completion of inventory surveys and land evaluation programs around Australia to identify wilderness resources (Robertson *et al.* 1992).

The period saw a steady transition in the emphasis on wilderness conservation from its recreation focus to one which embodied both recreation and biological conservation. This expanded role was recognised in the wilderness criteria proposed by Helman *et al.*

(1976), which were formulated specifically to identify areas capable of supporting wilderness-based recreation needs and the population viability requirements of macropods. As awareness of challenges facing the maintenance of biological diversity increased, wilderness was increasingly seen in eco-centric terms, providing opportunities for the protection of these values (Kirkpatrick 1994b). Indeed wilderness received explicit recognition as a key ecosystem and habitat attribute within Annex 1 of the 1992 Convention on Biological Diversity (United Nations Framework Convention on Biological Diversity 1992). By the 1990s, despite controversy, wilderness had found broad community acceptance both as a nature conservation objective and a land allocation category. Wilderness values, along with biodiversity and old growth values, formed the agreed set of core criteria for the establishment of a national forest reserve system under The National Forest Policy Statement (Commonwealth of Australia 1992) and these reserve criteria underpin the present joint commonwealth and state Regional Forest Agreements process.

Hostility to the wilderness concept in Australia has traditionally been utilitarian, usually arising where there are competing economic resource development opportunities. However, other areas of conflict have emerged, some within the realm of conservation, such as the conflict between wilderness protection and recreational access and use (Kirkpatrick 1994). There is also conflict when wilderness areas are allocated to reserves at the expense of land with lower wilderness quality but containing, for example, rare and endangered species.

Another key area of tension relates to the cultural association with the wilderness concept. Its cultural origins in western frontier societies tends to limit its appeal in those societies that do not share such a tradition. Wilderness has no meaning within indigenous societies (Flannery 1994), a situation which raises complex issues concerning perceptions of nature and the rights of culture and land which are particularly acute in the case of indigenous Australians (c.f. Robertson *et al.* 1992). Some argue, in this context, that wilderness is inherently limited by its origins in the 19th century romantic, chauvinistic ideal of virgin land (Preston and Stannard 1994). Others argue that the concept has evolved and spread from its cultural roots, and that its appeal has broadened sufficiently to be embraced on a more global basis (Robertson *et al.* 1992).

One approach to this problem has been to define wilderness in terms of the impact of modern and technological society. This is not to say that pre-modern societies did not have technology nor impacted on their environments. Rather, it recognises that indigenous societies were often low-energy societies whose activities were far more constrained by prevailing environmental conditions and the prevailing primary productivity of landscapes, compared to societies with access to the capacity for large scale economic developments as exemplified by the agricultural and pastoral industries, urbanisation, and the use of fossil fuel in conjunction with technologies such as the internal combustion engine.

Defining wilderness

Application of the wilderness concept requires an operational definition. Operational definitions of wilderness have changed over time, reflecting stages in the development of the wilderness concept. Although this history, like that of the wilderness concept, has been reviewed thoroughly elsewhere (see Robertson *et al.* 1992), it is worth briefly tracing the historical trends in approach as this provides useful insight into the way in which the wilderness concept has been applied, and to its potential in nature conservation.

The first operational definitions of wilderness were developed in the USA in the early part of the 20th century when wilderness was viewed primarily in recreational and heritage terms. Leopold (1921) defined wilderness as a continuous stretch of country preserved in its natural state big enough to absorb a two week pack trip, and kept devoid of roads, artificial trails, cottages, or other works of man, with a minimum area of 500,000 acres (c. 200,000 ha). Marshall, in the 1930s, defined it as a natural area that could not be traversed in a single day, specifying minimum size requirements of 300,000 acres (c. 140,000 ha) for forested environments and 500,000 acres for arid and semi-arid environments (United States Outdoor Recreation Resources Review Commission (ORRRC), 1962). The ORRRC (1962) itself defined wilderness as an area of land at least 100,000 acres (c.40,000 ha), containing no roads constructed for passenger car traffic in deserts and plains, existing as a single unit with boundaries reasonably free of indentation, and showing no significant disturbance due to technological activity.

The US Wilderness Act of 1964 provided a definition which has underpinned US wilderness policy and management since that time. The Act establishes wilderness as an area of land

> retaining its primeval character and influence, without permanent improvements or human habitation which (1) generally appears to have been affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable; (2) has outstanding opportunities for solitude or a primitive and unconfined type of recreation; (3) has at least five thousand acres of land or is of sufficient size to make practicable its

> preservation and use in an unimpaired condition … (Wilderness Act 1964 United States Congress, sect. 2c).

In Australia, operational definitions of wilderness took their lead from those in the US, but developed to specifically take account of the biocentric, ecological benefits increasingly attributed to wilderness. The series of inventory surveys of wilderness conducted in Australia during the 1970s and early 1980s defined wilderness in a way which recognised and accommodated these benefits. Helman *et al.* (1976) defined wilderness using a recreation criterion requiring a wilderness area to be of sufficient size whereby a substantial part thereof would be more than half a days walk from the nearest road access point, and an ecosystem conservation criterion based on a minimum area requirement for the maintenance of a viable population of the largest macropod. Four wilderness criteria were established (1) a minimum core area of 25,000 ha, (2) a core area free of major indentations, (3) a core area of at least 10 km width, and (4) a buffer zone surrounding the core area of 25,000 ha or more. Adjustments to this definition in this, and subsequent, surveys were made for coastal and arid/semi-arid areas.

A review of this history shows that shifts and changes in operational definitions of wilderness have occurred as a result of shifts and changes in the demand for, and supply of, suitable land. Demand has increased as the number and type of benefits ascribed to wilderness have increased. At the same time, the supply of land capable of providing these benefits has generally declined in line with development pressures. One result of this process is the successive operational re-definition of wilderness, such that increasingly small and more disturbed areas have been considered suitable for wilderness protection (Lesslie 1991).

Recognition of this process has meant that more recent definitions of wilderness tend to be couched in terms which are flexible enough to adapt to changing community valuations. A capacity for flexibility is fully realised in the approach to wilderness definition and identification in the Commonwealth Government’s National Wilderness Inventory (NWI). The NWI is founded on the ‘wilderness continuum’ concept (Lesslie and Taylor, 1985) which holds that the wilderness condition exists at a certain point on a spectrum of remote and natural conditions where the community recognises and places value on the existence of these conditions. Acknowledging that this point is difficult to define with any precision, and that it will, in any case, shift over time, the NWI places emphasis on measuring the extent to which locations are remote from, and undisturbed by, the influence of modern technological society (see also Kirkpatrick and Haney 1980). It does so by quantitatively measuring variation in remoteness and naturalness across the landscape using four indicators: (1) *remoteness from settlement* (remoteness from places of permanent habitation); (2) *remoteness from access* (remoteness from established access routes); (3) *apparent naturalness* (the degree to which the landscape is free from the presence of permanent structures associated with modern technological society; and, (4) *biophysical naturalness* (the degree to which the natural environment is free from biophysical disturbance caused by the influence of modern technological society).

Flexibility is similarly reflected in the definition of a wilderness area proposed by Robertson *et al.* 1992. They define a wilderness area as

> an area that is, or can be restored to be:
>
> > of sufficient size to enable the long-term protection of its natural systems and biological diversity;
> >
> > substantially undisturbed by colonial and modern technological society; and
> >
> > remote at its core from points of mechanised access and other evidence of colonial and modern technological society (p.26)

A recent Commonwealth definition of wilderness additionally makes explicit the distinction between impacts and influences of modern technological society and those of indigenous societies.

> Wilderness areas are large areas in which ecological processes continue with minimal change caused by modern development… Indigenous custodianship and customary practices have been, and in many places continue to be, significant factors in creating what non-indigenous people refer to as wilderness and wild rivers. (Commonwealth of Australia 1997 p.130.)

Also note the use of Wilderness as defined by the National Forest Policy

> “Land that, together with its plant and animal communities, is in a state that has not been substantially modified by, and is remote from, the influence of European settlement or is capable of being restored to such a state; is of sufficient size to make its maintenance in such a state feasible; and is capable of providing opportunities for solitude and self-reliant recreation” (Commonwealth of Australia 1992).

If flexibility is required for the operational definition of wilderness, then this has important implications for mechanisms of wilderness protection. It follows, for instance, that emphasis should be placed on the protection of wilderness quality in areas regarded as suitable for protection, rather than ensuring potential areas reach a particular wilderness threshold or standard. Any area may reasonably be selected for wilderness protection if its wilderness quality, in a given context, is sufficient to support its protection. Areas with high wilderness quality will be a priority for protection regardless of context. Areas of lesser quality will also have potential importance where these qualities have value because of their environmental context.

A capacity for discretion in the evaluation and selection process for wilderness is evident, for instance, in legislation established for the protection of wilderness in NSW. Subsection 6(1) of the NSW Wilderness Act of 1987 requires that wilderness constitute an area that

> is in a state that has not been substantially modified by humans and their works or is capable of being restored to such a state; … is of sufficient size to make its maintenance in such a state feasible; and … is capable of providing opportunities for solitude and self-reliant recreation.

Legislation established for the assessment, identification, protection and management of wilderness in South Australia, likewise does not prescribe a rigid formula for the identification of wilderness areas. The wilderness criteria described in section 3(2) of the SA Wilderness Protection Act of 1992 state that:

> (a) the land and its ecosystems must not have been affected, or must have been affected to only a minor extent, by modern technology; and
>
> (b) the land and its ecosystems must not have been seriously affected by exotic plants or animals or other exotic organisms.

Notably the wilderness quality of land may receive protection under the provisions of the Wilderness Protection Act, if it ‘meets the wilderness criteria to a sufficient extent to justify its protection as wilderness’ or ‘…to enable it to be restored to a condition that justifies its protection as wilderness...’. The express purpose for the constitution of areas under the Act is for the management of land for the protection of wilderness and the restoration of land to its condition before European colonisation.

Emphasis on wilderness management, rather than the meeting of any particular definitional condition, recognises the inherent variability of wilderness quality across the landscape and allows for variation in the degree of significance which may be attached to particular levels of wilderness quality in different ecological settings. Regardless of the existing level of wilderness quality, the objectives remain constant, namely; management for the protection and enhancement of wilderness quality.

Implications for role of wilderness in nature conservation

Important insights into the role which wilderness has played in nature conservation emerge from the history of the development of the concept and its definition. Some key observations on this history and its implications for nature conservation are noted below.

1. The wilderness concept has its origins in western frontier culture. Applications of the concept (in Australia and elsewhere) reflect these values. For societies where the separation of nature and economy is not distinct these concepts of wilderness have limited validity. Within indigenous societies, where such distinctions are non-existent, the concept is meaningless.

2. Wilderness is dependent upon the existence of modern technological activity (more particularly, its absence). Proponents of wilderness typically refer to the quality of remoteness from, and absence of influence by, the impacts and influences of modern technological society.

3. The wilderness condition is both subjective and relative. As Nash (1978 p.1) notes:

> Wilderness has a deceptive correctness at first glance. The difficulty is that while the word is a noun it acts like an adjective. There is no specific material object that is wilderness.

4. The notion of a wilderness area is flexible. Wilderness areas are recognised where wilderness quality has value to society. Wilderness areas are designated when it is accepted that this value exceeds that of alternative competing uses. Criteria for wilderness have changed as social valuations of wilderness quality have changed, and as the supply of land with wilderness potential has diminished. Wilderness quality was once valued primarily for the spiritual, aesthetic and recreational benefits it provided. An eco-centric focus on wilderness quality is predominant today.

5. Modern valuations of wilderness have cross-cultural relevance, to the extent of the ubiquity of modern technological activity, its impact on ecosystem structure and function, and the necessity for the maintenance of ecosystem structure and function. In a modern context, wilderness has connotations akin to that of ecosystem integrity. Dearden (1989) comments that, notwithstanding the power of recreational and transcendental arguments that have worked to establish wilderness systems, future appeal will lie in a much broader approach focussed on sustainable development and the role that natural areas play in regulating essential life processes, wildlife pools, genetic reservoirs, scientific inquiry and education, Deardon (1989) concludes that these concerns are neither as culture-based as the psychological *raison d'etre*, nor as geographically limited.

6. With regard to the definition of wilderness, distinctions must be made between i) wilderness quality, ii) wilderness areas, and iii) wilderness quality estimates as measured for example by the wilderness quality indicators of the NWI.

Wilderness quality can be defined as absence of, and remoteness from, the impacts and influence of modern technological society. These conditions are relative and vary continuously across the landscape, from highly developed landscapes through to those where the impacts and influence of technological activity is minimal.

Wilderness areas can be defined as those places in the landscape where remoteness and naturalness (wilderness quality) is sufficiently valued. The point on the wilderness quality spectrum where wilderness quality is sufficiently valued is intrinsically subjective and difficult to define and is fundamentally controlled by the demand for, and supply of, remote and natural places.

Measures and estimates of wilderness quality, such as the NWI indicators, are a means for measuring and quantifying remoteness and naturalness. Because of the complexity and variability of these attributes, such measures can only provide partial expressions of wilderness quality. Wilderness quality may be measured using indicators other than those used in the NWI.

Development of the concept of nature conservation

Introduction

This section aims to very briefly scope the goals, activities and ideas that are now encompassed by the concept of nature conservation. Firstly, some historical content is provided, which inevitably involves a degree of overlap with the development of the concept of wilderness - the extent of this overlap is itself interesting and important.

Some historical perceptions

The meaning of nature conservation has developed considerably over the past few centuries. This change has been driven by two forces. First, the advent of the industrial revolution and explosion in the human population wrought great change on rural environments in Europe and North America, and was a catalyst for greater levels and rates of resource extraction. Throughout the world, pastoral and agro-forestry development resulted in rapid loss of vegetation and land degradation. Concern with nature conservation therefore increased as economic development led to a scarcity of landscapes where pattern and process were dominated by natural processes, and the impact of people was secondary. The second driving force was advances in the ecological sciences that led to a more profound understanding of the origins, functions and meaning of life on Earth.

There are three fundamental reasons why people are concerned with nature conservation. The first relates to the need to maintain the biosphere so that it can continue to provide resources and conditions essential to human survival and well being. The second concerns recognition that other living organisms have intrinsic value irrespective of their value to humans. This leads to a respect for other forms of life and acceptance of a duty of care towards them. The third reason stems from the joy people derive from experiencing landscapes dominated by natural processes rather than modern technological- society.

Interestingly, the utilitarian values noted above were evident early in the European settlement of Australia. Powell (1976) quoted an 1865 article in '*The Argus*' (a Melbourne newspaper)

> In protecting the forests we do more than increase the growth of timber - we prevent waste of soil, we conserve the natural streams, it is not improbable that we prevent decrease in the rainfall, and it is certain that we largely affect the distribution of storm waters.

The intrinsic values of nature were also recognised at around the same time. Dr Ferdinand von Mueller (again cited in Powell 1976) in a public address in 1866 articulated his belief that

> What is vitality, and what mortal will measure the share of delight enjoyed by any organism? Why should even the life of a plant be expended cruelly and wastefully... that individual life, whatever it may be, which we often so thoughtlessly and so ruthlessly destroy, which we never can restore, should be respected.

Despite these eloquent pleas, economic development continued largely unabated without concerns for nature conservation. Similar concerns were still being expressed into the 20th century. Jones (1925) argued that the forests of Australia had three main uses. First, their unique character and beauty. This was considered to have:-

> "a very important influence on national welfare by virtue of the beautiful and health giving surroundings afforded by the forests for our hours of leisure"

Second, their value in conserving and regulating the run-off of rain water; and third, their value in providing essential primary products, especially timber. The latter two were considered to have direct economic value, with the conclusion that

> "a brief consideration of them will convince us that our continued prosperity...is intimately connected with the conservation of our forest wealth"

Also, by the 1920s, it was obvious that in addition to large scale transformations in the vegetation cover of landscapes, many species were, as a result of human activity, extinct or threatened.

It is evident that the basic values that underpin our concerns for nature conservation have been motivating forces in our culture for at least 100 years. Certainly these values and concerns are now shared more broadly through our community than they were in the 1860s (though they are still far from being universally accepted principles).

We also note that the concepts of nature conservation and wilderness have developed based on shared common concerns for nature, and both reflect similar aspects of the interactions between people, society and nature. The non-utilitarian value of wild nature to humans is an integral part of the motivating force behind concerns to conserve nature. Conversely, land with high wilderness quality is now being increasingly viewed as being fundamentally important to ecological conservation (Noss and Cooperrider 1994).

Advances in the ecological sciences

The underlying motivational forces behind nature conservation changed little over the last 100 years, and they share common roots with the development of the concept of wilderness. However, what has changed dramatically at that same time is our understanding of what constitutes nature, the structure and function of the planet's ecology, and an increased understanding of the impacts of human activity and how these can be ameliorated. These, in turn, have led to a re-evaluation of the role of wilderness in nature conservation (Noss and Cooperrider 1994). The following advances in the ecological sciences were not entirely driven by a concern with nature conservation, they were in part, informed and moulded both scientific and popular conceptions, thereby fuelling public support for nature conservation to be given greater weight in public policy.

Evolution and natural selection

Life forms are not static, but rather change and diversify through time. The recognition and acceptance of biological evolution and the central role of natural selection in this process (see Cockburn 1991) continues to have a profound effect on the way we perceive the biota and our relationship to it. The present set of taxa inhabiting a landscape, region or continent are not fixed in time, but rather represent one of a set of biological forms that have or will occupy this landscape over the course of time. In response to the insights provided by the theory of evolution, nature conservation goals have expanded to encompass the maintenance of the capacity for evolution (e.g. Government of Victoria 1988, Commonwealth of Australia 1992). This is in addition to the goal of preserving nature in its current form. Acceptance of biological evolution as a major scientific paradigm has profoundly altered how we humans view ourselves and our relationship to nature.

Advances in autecology

The study of single taxon-environmental relations requires detailed, empirical studies that are time consuming and require a significant investment in field studies. The last 100 years has witnessed a tremendous increase in knowledge about the distribution, life history, and habitat requirements of organisms, in particular, vertebrates and vascular plants (Morrison *et al.* 1992). The concept of the niche was formalised by Hutchinson (1957) who defined it as a hyper-dimensional space specifying the range of environmental conditions to which a taxa is physiologically and ecologically adapted. The latter being a function of competition and other biotic interactions. Habitat, therefore, can be viewed as those locations that contain the environmental conditions and resources defined by a taxon's niche. Or as Hutchinson puts it, 'habitat is the address and niche the occupation'.

While the notions of a niche and habitat are now well established, it is only recently that computer technology has been available to analyse niche relations and habitat distributions at landscape scales (e.g. Austin *et al.* 1990). We now have some ability to understand the physical environmental determinants of niche and the distribution and availability of species-specific habitat resources on a landscape-wide basis.

Conservation biology

As noted above, ecological science has often been motivated and informed by the goal of nature conservation. However, the last 15 years has seen a deliberate and successful attempt to formalise conservation biology as a mission-orientated discipline (Soulé 1987). That is, a scientific discipline motivated to use all available methods to maintain the integrity of natural ecosystems and stem the loss of biodiversity (Hedrick *et al.* 1996), but with a particular focus on species as the unit of conservation (Caughley and Gunn 1995).

Caughley (1994) noted that conservation biology has focused on two main themes (but see critique by Hendrick *et al.* 1996). The first concerns small populations studies, which examine the effects of loss of genetic variation, and demographic and environmental stochasticity, on the risk of extinction of small populations of often rare, threatened and endangered species. The second focuses on the processes by which a species declines such that only small populations remain.

A major line of investigation in conservation biology therefore has been the prediction of the point a population becomes unable to sustain its existence. This was originally formulated in terms of identifying minimum viable populations but has developed into notions of quantifying the probability of extinction; a more integrated notion that includes consideration of spatially structured population dynamics and larger scaled effects such a habitat loss (Lindenmayer and Possingham 1995). More recent work on meta-population studies (e.g. Hanski and Gilpin 1991) has also helped integrate life history and niche and habitat resource studies with population ecology and genecology (see Simberloff 1988, Lindenmayer 1996). Conservation biology also has helped focus attention on the genetic diversity found within species (e.g. Osborne and Norman 1991), and in so doing, made populations of species rather than a species *per se* the operational unit of conservation evaluation. Therefore, in practice, species management involves the management of populations and their habitat.

Ecosystem theory

The concept of the ecosystem was advanced by Tansley in 1935 (Tansley 1935). In the same year, Troll introduced the concept of landscape ecology (Troll 1971). The idea behind both concepts was to focus attention on the study of the full and complex interrelationships between living organisms (the "biocoenosis") and the environment. Ecosystems therefore by definition involve the exchange of energy, water, and nutrients, between plants, animals, microorganisms and the atmosphere, hydrosphere and lithosphere. Landscape ecology has a particular focus on the spatial configuration of the various biophysical elements of ecosystems, and how these change through space and time (Hansson and Angelstam 1991, Forman 1996). A number of key concepts of relevance to nature conservation stem from ecosystem theory.

a. A functional perspective

Both ecosystem science and landscape ecology examine living organisms from a functional perspective (for example, primary producers, secondary consumers, decomposers) in addition to a taxonomic perspective. Ecosystem processes can be studied at a range of space/time scales - site, catchment, biome, planet - together they constitute essential life support systems for humans, as their healthy functioning ensures a continuing supply of water, air, productive soils, and various renewable natural resources. Species and landscapes therefore have additional nature conservation value that derives from the roles they play in the maintenance of these ecosystem processes. Indeed, on this basis, a number of authors have highlighted the importance of maintaining functionally viable populations that play key roles in ecosystem processes like pollination (e.g. Conner 1988, Carthew and Goldingay 1997).

b. Ecosystem resilience

Holling (1996) defined ecosystem resilience as the ability of a system to absorb change and variation without flipping into a different state where the variables and processes controlling structure and behaviour suddenly change. He further argued that resilience is the property that sustains ecosystems. This concept is important because it implies that while ecological systems are dynamic, these changes are bounded. Thus, systems can be perturbed such that if resilience is destroyed they can be permanently degraded. The resilience of ecosystems can result in relative homeostasis for very long periods of time, despite environmental change and variation. A fundamental finding of ecosystem resilience theory is that systems must be large enough to absorb or incorporate the largest scaled perturbation on the basis that there are limits to the level of disturbance an ecosystem can absorb before resilience is destroyed.

Earth system science

In parallel with the development of ecosystem theory, increasing attention was being paid to the effect of living organisms on the total global environment. These types of interplays were recognised a long time ago by the English geologist Hutton (1788), who based his studies upon hydrological and nutrient cycles.

This notion that the Earth and Life sciences should be viewed in an integrated way under the aegis of geophysiology was also strongly promoted this century by the Russian scientist Verndansky (1945). Lovelock's Gaia world view (see Lovelock 1979) both resurrected the concept, and further advanced the view that, rather than simply adapting to their environment, organisms alter their environment with subsequent implications for the evolution of all biota. As noted by Markos (1995), the Gaia world view states that Earth constitutes a single system within which homeostasis is maintained by active feedback processes operated automatically and unconsciously by the biota - Earth therefore can be viewed as a self evolving entity that uses solar radiation for self maintenance (regeneration) and self-organisation. From this perspective, Earth itself becomes the ultimate unit of nature conservation.

The organisation of ecological systems

Recognition that life (including human life) is an emergent property of Earth, and is sustained through highly interconnected and coupled biophysical systems, has promoted research into how these systems are organised and have generated such complexity. Four billion years ago the planet was a lifeless sphere. Now, the planet is inhabited by at least 30 million species of plants, animals and microorganisms, encompassing many trillions of populations and a larger number of individual organisms. The total number of unique genotypes is doubtless very large, and perhaps beyond estimation even within an order of magnitude. Various theories have been proposed to account for how such a high degree of ecological complexity could have emerged and how it continues to be sustained.

New theories of complexity and complex adaptive systems (Kraufman 1995) and hierarchy theory (Koestler 1967, Allen and Star 1982), together with the concepts of gaia and geophysiology, have lead to abandonment of the conventional mechanistic model of nature. As discussed by Abram (1991), the mechanistic paradigm views nature as akin to a machine, comprising parts that can be tinkered with, examined in isolation, and is inherently linear. Modern ecological concepts force us to view nature differently, as systems driven by nonlinear feedbacks, where the various components co-evolve, adapting in response to changing conditions, yet also collectively interacting to provide the homeostasic and resilience required for organisms to persist in a landscape. One implication of these ideas is that there is no single unit of conservation, with species being but one of several criteria that can be applied to define valid ecological entities (Allen and Hoekstra 1992).

Policy and management responses

Advances in the ecological sciences have profoundly altered our understanding of nature and how units of conservation are defined. A range of policy and management themes has emerged in response to this more complex notion of nature and what is required for its conservation.

Sustainable development

Recognition that humans are dependent upon the biosphere has promoted the notion of sustainable development. That is, development that operates within the regenerative capacities of Earth. Sustainable development aims to alter the patterns of human production, consumption and reproduction to reduce their environmental impact, and provide a social and economic system which sustains rather than degrades nature.

Biodiversity

Advances in autecology, ecosystem studies and conservation biology have forced the consideration of the full extent of biological expression on the planet. Biodiversity is now a well established concept, defined as the diversity of genes, species and ecosystems. Biodiversity mean more than species richness, that is, the number of different species. It also includes the many and varied ecological differences that occur within and between organisms (see Harper & Hawksworth 1994), and between organisms and their environment.

Wildlife preservation was usually defined in terms of a subset of taxa, charismatic species, large vertebrates, economically valuable species or flagship species. Consequently, the majority of other species and life forms were ignored. The concept of biodiversity has brought attention to the full gamut of life and the fact that by far the greatest number of living organisms are very small and cryptic to humans. The scope of nature conservation has consequently expanded to include all forms of life on Earth, even invertebrates and microorganisms, as well as the inter-relationships between then.

Ecosystem management

Advances in species-focused conservation biology and ecosystem studies have directed attention to consider wildlife preservation in the context of those ecosystem processes that produce the ongoing supply of habitat resources needed to sustain wild populations of plants, animals and microorganisms. For example, nature conservation management strategies must consider the wider landscape context that provides the environmental context for a given endangered population or meta-population (e.g. Lamberson *et al.* 1992). For example, Clark and Minta (1994) have recommended that an ecosystem management approach be employed for the integrated resource use and conservation of nature in the greater Yellowstone ecosystem in central U.S.A.

Land use evaluation and planning

During the late 1960s, nature conservation became accepted as a legitimate land use that had to be considered in land use evaluation, planning and allocation, alongside forestry, water catchment management, agriculture etc. Subsequently, considerable effort has been invested in developing land evaluation techniques for nature conservation and examining the tradeoffs between nature conservation and competing land uses (for example, see Costin & Groves 1973, Austin *et al.* 1977). Various criteria have been established for evaluating the nature conservation value of a location, such as representativeness, rarity, and diversity (see Margules & Usher 1981, Mackey *et al.* 1988). More recently, various procedures have been developed for the design of dedicated reserve networks based on these criteria (see Pressey *et al.* 1993, Church *et al.* 1996, Willis *et al.* 1996).

Summary

As noted above, the fundamental reasons for conserving nature have changed little over the last 100 years. Our understanding of what constitutes nature has however been transformed, and our knowledge concerning how nature functions and the impact of humans has similarly increased. This together, with the accelerating loss of biodiversity throughout the world, and the ongoing degradation of ecosystem resilience, has intensified the need for nature conservation.

The ecological sciences have provided critical new insights into the dependence of humans on these ecosystem processes, and the role of living organisms in their maintenance. The survival of humans is now inexorably linked to the conservation of nature.

Nature conservation is now as concerned with processes and functions, as it is with physical entities and structures. This does not weaken the need for species-orientated conservation because living organisms comprise the basic entities and structures of ecosystems, and mediate all processes and functions. Conserving species requires conserving habitat which in turn maintains key ecosystem processes and functions (see Soulé 1994).

The results of ecological sciences indicate that there are real (though often hard to define) biophysical limits within which human activity has to operate. Sustainable development does not mean that every hectare of Earth's land surface must be mined, farmed or otherwise directly manipulated either to extract resources or used as a sink for waste. Rather, a range of strategies is required if nature conservation objectives are to be achieved.

It is not possible for humans to live without impact. All life consumes other life. But ultimately humans must learn how to conduct their affairs such that natural processes (that is, processes that are self-sustaining and that can continue without the need for human intervention) can continue. This will inevitably involve ensuring that there are wild populations of plants, animals and microorganisms interacting with each other and their physical environment. In this way, new forms of life can continue to evolve and adapt in response to changing environmental conditions.

The fact that some species can exist in highly disturbed landscapes with very low levels of wilderness quality should not be used to ignore the fact that the maintenance of the process of evolution requires *in situ* conservation of a large pool of wild biotic populations, operating within a landscape dominated by self-supporting and self-regenerating ecosystem processes, of sufficient size and integrity to ensure long term resilience.

In summary, nature conservation now encompasses no less than (i) the world's biodiversity, (ii) maintenance of essential life support processes, and (iii) the maintenance of Earth's evolutionary potential.

Ecology by definition involves the interactions between the biotic and abiotic components of nature. Many phenomena which are commonly considered to be purely physical or biological are in fact biophysical in nature. For example, there are physical dimensions to an animal's habitat (Morrison *et al.* 1992) and conversely the chemical composition of the atmosphere involves the biota (Gorshkov *et al.* 1994). In this report we focus on phenomena that are the result of ecological interactions rather than the purely biological or physical components of nature.

We therefore define nature conservation as having two main foci:

1. conservation biology (with a primary focus on species conservation) and
2. conservation ecology (with a primary focus on ecosystems conservation).

However we recognise that there are other components of nature that are highly valued, for example, geomorphic features which are considered to have scientific value.

Introduction to threatening processes

A range of processes associated with modern technological society can be identified that either alter wilderness quality and/or threaten the conservation of nature. We have chosen to illustrate the role of threatening processes by examining six broad and often interrelated types of disturbance relevant to Australia:

a. changes to fire regimes

b. changes to hydrological regimes

c. roading as a threatening process

d. changes to vegetation cover

e. introduction of exotic species

f. accelerated global warming.

The six classes do not necessarily occur independently of each other, often the processes act simultaneously or in close association. Our aim in outlining the impacts of these classes of threatening processes is **not** to produce a definitive review of the impacts of each process on the Australian environment, as this would constitute a major study in its own right. Rather, we briefly indicate the historic and potential impact of each of these processes on the conservation of nature. The sketches we create of the threatening processes provide examples of how modern technological society can disturb natural environments. A number of other classes of threatening process which alter wilderness quality could also have been included (for example, recreation, pollution, and hunting).

The impact of these threatening processes is likely to intensify as human population increases (State of the Environment Australia, 1996).

Changing Fire Regimes

Fire is a significant ecological factor in most Australian environments (see Gill *et al.* 1981, Groves 1981, Williams and Gill 1995). Humans have been considered a major agent of fire in the Australian landscape for at least 50,000 years and possibly up to 150,000 years (Hallam 1985, Flannery 1994). Presently, most fires are started by people, and are generally extinguished quickly, whereas prior to European settlement, fires were generally not actively extinguished (Gill 1977). King (1963) reviewed early historical literature that referred to forest fire and concluded that European colonisation of Australia had, (i) bought about a change in the nature of the forest, (ii) reduced the area of forest burned annually, and (iii) produced more severe and damaging bushfires.

Significant changes in human-induced fire regimes occurred with the arrival of Europeans in Australia (e.g. Griffin *et al.* 1986). The spread of European settlement displaced traditional indigenous burning regimes (King 1963). Aborigines lit fires in the landscape for a number of reasons including the regeneration of food plants, assistance in game hunting, the location of food items, the maintenance of mosaic vegetation in different seral stages, ceremony and religious purposes and signalling (Nicholson 1981, Hallam 1985, Burbidge 1985, Mangglamarra *et al.* 1991). Aboriginal use of fire was not consistent Australia-wide; firing varied in location, season, and frequency (King 1963, Gill 1977).

Early Europeans utilised fire for a variety of purposes and these were different in timing, intensity, and frequency from Aboriginal fires. Fire is currently utilised in the pastoral industry for purposes such as the promotion of new pasture, the extension of pasture growing season, the control of woody weeds, gradual clearance of native vegetation, establishing improved pasture species, a hazard reduction measure against wildfires, the release of nutrients, and the destruction of source areas for cattle ticks (Leigh and Noble 1981, Stocker and Mott 1981). There may be some similarities in fire use by pastoralists and Aborigines. However, it is anticipated that different outcomes will arise from their different goals and patterns of use. For example, burning to promote growth for cattle may occur with an intensity, frequency, and seasonality different from burning for purposes such as game hunting.

Evidence for changing fire regimes

Direct evidence for the changes in fire regimes following European colonization is difficult to establish. There is a lack of detail concerning Aboriginal application of fire beyond historical anecdotes and guesswork (Nicholson 1981, Christensen and Abbott 1989). One source of information on fire history comes from analysis of fire injury to living plants. Lamont and Downes (1979) and Burrows *et al.* (1995) investigated fire injury in jarrah (*Eucalyptus marginata*) and grasstrees (*Xanthorrhea preissii* and *Kingii australis*), respectively. Both groups concluded that significant changes in fire regimes have occurred since European settlement.

Effect of fires on biota

Fire has the potential to have significant impacts on the distribution and abundance of biota. For example, Gill and Bradstock (1995) recorded four species of Gymnospermae and fifteen species of Angiospermae that have become locally extinct due to the effects of fire. Changes in fire regimes are considered to be one of the factors contributing to the dramatic loss of small to medium sized mammal species from semi-arid/arid environments of Australia (Burbidge and McKenzie 1989, Morton 1990). Exotic herbivores and predators, agricultural development, and drought have also contributed to these losses (Burbidge and McKenzie 1989, Morton 1991, Tunbridge 1991).

Another example is the contraction of the wet sclerophyll forest ecotone between tropical rainforest and dry sclerophyll forest in northern Queensland. These types of patchy or ecotonal environments are maintained by fire regimes and it appears that the exclusion of such disturbance processes may lead to the local extinction of a range of flora and fauna species. The wet sclerophyll forest ecotone supports endemic bat and ant faunas (Harrington, pers. comms.) and is probably critical for the long-term conservation of a range of species such as the northern sub-species of the yellow-bellied glider (*Petaurus australis reginae*) (Harrington and Sanderson 1994).

A classic example of the effect of changing fire regimes on fauna is the case of the mala (*Lagorchestes hirsutus*) (Bolton and Latz 1978). This species has suffered a dramatic decline over the last fifty years which has been associated with the absence of Aboriginal fire regimes. Areas typically supporting the species were characterized by the presence of shelter and forage, which are represented by different seral stages of spinifex grassland. It appears that this mosaic of seral stages is created by cool winter fires, a fire regime which has been largely lost with the movement of Aborigines to stations and other settlements during the last fifty years. However, the influence of vegetation mosaics on wildlife distributions was further tested on Barrow Island in Western Australia (Short and Turner 1994). They found that the distribution of several species; golden bandicoots (*Isodon auratus*), northern brush-tailed possums (*Trichosurus vulpecula arnhemensis*), and burrowing bettongs (*Bettongia lesueur*), was **not** influenced by the vegetation mosaic. Other factors no doubt were operating in this relatively small island of finite habitats.

Interaction of fire with exotic species, grazing, and forestry

The introduction of exotic species during the last 200 years has also altered fire regimes in Australian environments. Stocker and Mott (1981) outline the effect of two different exotic species on natural fire patterns and subsequent vegetation distributions. The introduced scrambling grass (*Melinus minutiflora*) rapidly colonizes severely burnt sites in areas of tropical Australia, altering vegetation composition and resisting the spread of fire through the community. Similarly, *Lantana camara* (a scrubby climbing species) prevents or slows grass invasion following the disturbance of rainforest or late stage forest regeneration. Lantana can then act as a barrier to low intensity burns early in the dry season, further altering community composition. The South African veldt grass, *Ehrharta calycine*, has been attributed with changing “the whole cycle of regeneration after fires” in King’s Park, Perth (Baird 1977). The veldt grass is thought to outcompete native herbaceous species and cause a decline in shrub seedling survivorship.

Grazing has the potential to influence fire regimes by altering vegetation structure. Cattle grazing in the Kimberley region has resulted in a reduction of understorey structure in rainforest patches, permitting the increased germination of savannah grasses (McKenzie and Belbin 1991). Cattle shelter in these dense, shady patches and trample seedlings and small trees and shrubs, opening up the patch to sunlight and drying winds. This provides access for dry season savannah fires to enter rainforest patches and initiate further structural and spatial changes to the plant community. Aboriginal people value rainforest patches in the Kimberley as sources of food. They often lit low intensity burns around the rainforest patches to prevent intense fires later in the dry season (Mangglamarra *et al.* 1991). The rainforests appear to be exposed to increased fire risk due to cattle grazing and the absence of Aboriginal fire management. Grazing on levee bank forests and woodlands in tropical Australia has resulted in the replacement of grasses with the tall woody herb *Hyptis suaveolens* (Stocker and Mott, 1981). The outcome is a decrease in frequency and intensity of fire which has been linked to an increase in the woody perennial component of these ecosystems.

The advent of colonization and modern technological society has clearly altered fire regimes in many landscapes. This has probably changed elements of vegetation structure and composition, and in certain areas, led to a decline in species as a result of a change in the supply of habitat resources. However, there are insufficient data for pre- and post-colonisation fire regimes to permit evaluation of the extent to which any given landscape is disturbed, This is one reason why an index of fire disturbance is not included in the Lesslie wilderness model employed by the NWI. Another is that changes in the intensity, frequency, and types of fires is not necessarily located with reliably measured material evidence of modern technological society.

Changing Hydrological Regimes

A rapid increase in knowledge and technology over the last 200 years has permitted the control and regulation of vast amounts of water in Australia (Ghassemi *et al.* 1995). The impoundment and regulation of waterflow has lead to dramatic changes in the Australian environment. Few "wild rivers" are now left in Australia and, paradoxically, some of the most pristine catchments are protected as sources of water for urban centres, for example, the Hawkesbury-Nepean. It should also be noted that while there may be few wild river systems there are a large number of stream segments that are wild. The effect of changing hydrological regimes can be considered by examining the volumes of water now controlled by human structures. Currently, 81,000 GL of water are held in storage facilities. Annually, about 15,000 GL p/a are distributed on 2,679,000 ha of irrigated land, sourced from rivers and groundwater (State of the Environment Advisory Council 1996, p. 7-11). The harnessing of vast quantities of water has many influences on normal river and stream dynamics. Even relatively minor impoundments, such as farm dams, can have a significant impact. In the Lal Lal Reservoir catchment in Victoria, farm dams have reduced average annual streamflow by 7% which increased to a 50% reduction in streamflow during drought years (Victorian State of the Environment 1988 cited in State of the Environment Advisory Council 1996, p 7-9). This pattern is undoubtably repeated throughout Australia (there are 300,000 small farm dams in Victoria alone).

Salinity

Salinity is a major problem of increasing seriousness in Australia (see review by Ghassemi *et al.* 1995), and it can occur as a result of a range of factors including (i) large-scale irrigation practices, (ii) clearing of vegetation, (iii) the replacement of trees by shallow-rooted plants such as crops and pastures, and (iv) the discharge of saline agricultural or industrial waters. Salinisation has an enormous cost in Australia not only in financial terms but also with respect to other criteria (Burgman and Lindenmayer, in press). The impacts of salinisation on biodiversity can be substantial (Metzeling *et al.* 1995). Scalding and drainage seeps can lead to loss of plant communities (Williams, 1987), and their associated fauna and result in major changes to soil flora and fauna (George *et al.* 1995). For instance, in the Avon River in south-west Western Australia, a species of freshwater mollusc was replaced by a taxon that was tolerant of brackish conditions. Hart *et al.* (1991) and Metzeling *et al.* (1995) describe a wide range of taxa that are potentially at risk from salinisation ranging from frogs, aquatic plants and macro- and microinvertebrates. In Victoria, preliminary work has resulted in the listing of almost 100 species being listed as susceptible to increasing salinity (Salinity Planning Working Group, 1992; in Ghassemi *et al.* 1995). These include plants, terrestrial vertebrates and fish.

Effect of changing hydrological regimes on river biota

Dams and weirs have been built throughout most river systems in densely settled Australia. Often these dams have been established in areas of high conservation value, such as the Tumut dam system in Kosciusko National Park and the Thomson Dam in Victoria (Melbourne and Metropolitan Board of Works 1975). Dam construction has been shown to alter invertebrate communities at a number of localities including Dartmouth Dam on the Mitta Mitta River (Doeg *et al.* 1984, cited in Lake and Marchant 1990), the Blue Rock Dam on the Tanjil River (Chessman *et al.* 1987), and the Thomson Dam on the Thomson River (Doeg *et al.* 1987). Chessman *et al.* (1987) found a 30-40% reduction in faunal density after construction of the Blue Rock Dam. (Blyth *et al.* 1984, cited in Lake and Marchant 1990) found that such effects penetrated 20 km downstream of the impoundment.

The impacts of dam construction may be short term (< 5 years, Gippel and Stewardson 1995), but harvesting of water once the dam is complete has the potential for long-term effects on the river environment. Damming and irrigation in the Murray-Darling system has resulted in a 44% drop in water volumes in the lower Murray (Walker and Thoms 1993). Engineering works in the Snowy Mountains have redirected 99% of the natural water flow from the Snowy River into inland water systems (Snowy Genoa Catchment Management Committee). Since the advent of weirs and upstream dams, low flows have decreased 5-fold and moderate flows have increased 2-fold in the lower Murray (Walker and Thoms 1993). Concurrent with these changes in the Murray River, there has been a decline in the range and abundance of native fish and invertebrates (Cadwallader 1978). Gastropods have also suffered the effects of changes in hydrological regimes, with 13 of 14 native species becoming locally extinct (State of the Environment Advisory Council 1996). Flora and fauna that occupy wetlands, billabongs (Hillman 1986), and floodplains (e.g. *Eucalyptus* *camalduensis*) (Dexter 1978) are particularly at risk due to changes in flow volume and patterns.

The establishment of physical barriers such as weirs and dams, have direct impact on migratory fish. One study estimated that between one-third and one-half of aquatic habitats of the south-east coastal drainages have been obstructed by physical barriers (Harris 1984). This study located 293 barriers in these river systems which had the potential to influence 26 migratory fish species.

"River improvement" includes engineering works that alter the natural channels of a river for irrigation purposes, for the protection of banks and other human structures or features. This process often involves the removal of debris and the clearing of vegetation around a watercourse, which have an important influence on habitat for stream and river species. Hortle and Lake (1983) found that the absence of such habitat attributes (snags, area of slack water, length of bank fringed with vegetation) accounted for lower fish abundance and species richness in channelised areas of the Bunyip River, Victoria.

Long-term monitoring at Lake Pedder indicates a loss of species and biodiversity since the lake was dammed in the early 1970's. (Lake, 19..). It was further argued that in the event of restoration of the lake it would be unlikely to result in a return of the species that have been lost since it was dammed.

Groundwater

Australia is a dry continent; in arid and semi-arid parts of the country, groundwater is the only permanent and plentiful source of water. Since artesian water sources were discovered in the 1880s, human reliance on groundwater has grown to the point that natural hydrological regimes are being disturbed (Ghassemi *et al.* 1995). Disturbance of hydrological regimes in arid and semi-arid environments is of particular concern because of the important role groundwater performs in maintaining springs, wetlands, and vegetation (State of the Environment Advisory Council 1996 p7-24).

The utilization of groundwater from the Great Artesian Basin (GAB), Australia's largest source of artesian water, highlights the problems confronting groundwater management in Australia. The GAB lies beneath 1.7 million km^2 of inland Australia, supporting 600 natural springs (Habermehl 1983) and sustaining a diversity of vegetation and animal life. Land development has led to the construction of 23,000 artificial water points (3,000 artesian bores and 20,000 sub-artesian bores) (Habermehl 1983) within the Basin. Flows from individual artesian bores can exceed 10,000 L/day with outputs estimated at 1,500 million L/day across the entire basin. Large volumes of water are wasted each year as many bores flow continuously once they have been sunk. An estimated 90% of water obtained from the GAB is lost to evaporation and seepage from bore drains (State of the Environment Advisory Council 1996 p6-50). It is unwise to continue the wastage of such large volumes of water, especially when the long time scales involved in groundwater dynamics are considered. Water entering the Great Artesian Basin on the western edge of the Great Dividing Range, has been measured to take 1.8 million years to reach the southern section of the Basin (Torgensen *et al.* 1991). Historic levels of harvesting have already reduced artesian flows output from bores (Habermehl 1983) and nearly one-fifth of bores in south-west Queensland have ceased to flow (State of the Environment Advisory Council 1996, p6-50).

The widespread establishment of artificial water points has the potential to alter flora and fauna in arid and semi-arid Australia (Landsberg *et al.* 1997). A study of the major arid and semi-arid pastoral rangelands of Australia found few pastorally-productive areas further than 10 km from artificial water points (Landsberg and Gillieson 1997). An examination of biota along transects centred on water points found grazing led to major changes in species composition. At sites adjacent to the water points, between 15-38% of species groups declined in abundance relative to remote sites. Additionally, between 10-33% of species groups increased in abundance at sites adjacent to water points relative to remote sites. A range of taxonomic groups were examined in the study by Landsberg *et al.* (1997) including overstorey plants, understorey plants, plants in the soil seedbank, birds, reptiles, small mammals, and several invertebrate groups.

Many of the impacts on river systems and the hydrological regimes are spatially correlated with infrastructure developments like dams and weirs. Consequently, indices related to the presence, density, or distance from such structures (such as found in the Lesslie wilderness model) may be good predictors of threatening processes.

Roading as a Threatening Process

Roads can be a conduit through which nature conservation values are threatened by human activity (Scott, 1994). Impacts may be intended (e.g. grazing, hunting and mining) or inadvertent (e.g. weed and pathogen dispersal). Where roads exist there is often the need to undertake active ameliorative conservation management.

Frood and Calder (1987) identified road and track construction as a major threatening process influencing the conservation value of Victorian forests. Changes in sediment loads and patterns of water flow in streams may result from the construction of roads and this may have a negative impact on sediment loads in aquatic ecosystems and their associated flora and fauna (Michaelis 1984, Grayson *et al*. 1992, Metzeling *et al*. 1995).

Several authors have speculated that roads may provide conduits for the movement of feral predators (e.g. the Red Fox and Feral Cat) in the forests of south-eastern Australia (e.g. May and Norton 1996). In the case of wood production forests in south-eastern Australia, roading networks established as part of timber harvesting operations may allow feral animals to gain access to areas where they have not previously occurred, although there is presently only limited evidence to support these assertions (cf. Catling and Burt 1995). This may, in turn, lead to increased predation pressure on native animals such as small mammals, bandicoots and macropods (Robertshaw and Harden 1989, May and Norton 1996) and hence jeopardise the long-term persistence of populations of some species.

Roads may pose barriers to the movement of animals and this could serve to isolate and fragment populations thereby increasing their susceptibility to extinction. For example, small forest mammals may be reluctant to cross roads and tracks (Burnett 1992, Richardson 1992, Lindenmayer *et al.* 1994). Barnett *et al*. (1978) found that overgrown logging tracks more than eight metres wide impeded the movement of the Brown Antechinus (*Antechinus stuartii*).

Motor vehicles that use roads may have a number of other important impacts on nature conservation. Mortality resulting from collisions with vehicles may have a major impact on the dynamics of wildlife populations, although this is a topic on which only limited Australian research has been undertaken (Bennett 1991). Ehmann and Cogger (1985) made crude estimates of road-kill rates of amphibians and reptiles in Australia. They based their calculations on information on the extent of the sealed road network in Australia, the amount of roading that passes through suitable habitat in various parts of the continent, the density of animals per unit distance, and numbers of road kills. Using this information as a guide, Ehmann and Cogger (1985) estimated that approximately 5.5 million frogs and reptiles are killed every year on sealed Australian roads. The actual number of deaths is likely to be much higher if the extent of herpetofauna mortality on unsealed roads is taken into account and the substantial extension of the sealed road network in the past decade since Ehmann and Cogger (1985) completed their calculations is considered. Populations of other groups of animals are highly susceptible to the effects of road-kill mortality. For example, in many parts of rural Australia, the Magpie (*Gymnorhina tibicen*) nests along roadsides, and mortality rates among the juveniles of this species can be particularly high (Carrick 1963, Bennett 1991). Other “high profile” Australian species for which collisions with motor vehicles may have an important effect on population dynamics include Carnaby's Cockatoo in southwestern Australia (*Calyptorhyncus funereus latirostris*) (Saunders 1990), Eastern barred Bandicoot (*Parameles gunnii*) (Brown 1989, Backhouse *et al*. 1995) and the Koala (*Phascolarctos cinereus*) (Lee and Martin 1988).

Another impact of roading is the increased incidence of deliberate and accidental fires (Gill, personal communication).

There is considerable potential for vehicles that use roads to act as dispersal vector for weeds (e.g. Wace 1977, Forman 1996). Wace (1977) germinated more than 18 500 seedlings representing almost 260 species from samples collected from mud and sludge washed from vehicles cleaned in a commercial car washing facility in the centre of Canberra. Seedlings from exotic plant taxa figured prominently in the plant samples; more than 50% of the total number of seedlings were assigned to a broad group typically occurring in disturbed environments such as rubbish dumps. Thus, motor vehicles may considerably assist in the dispersal process of an extensive number of weeds. Hence, the control of weeds is likely to be very difficult in areas intersected by roads, especially those with a large number of high density of roads (Burgman and Lindenmayer, 1997).

Finally, roading also indirectly affects nature conservation values by facilitating activities such as the illegal taking of wildlife and plants, waste dumping, grazing, hunting and the removal of material such as sand, gravel, soil, rocks, timber and flower seeds.

In summary, roads may have a number of adverse impacts on nature conservation values associated with their establishment including the isolation and fragmentation of populations, the enhanced dispersal of weeds and feral animals, and a major source of mortality for populations of animals as a result of collisions with motor vehicles (Bennett, 1991). These processes could, in turn, lead to changes in the species composition of landscape ecosystems as well as negative changes in the dynamics of populations of native organisms.

Changing Vegetation Cover

European settlement of Australia has led to significant changes in vegetation cover (State of the Environment Advisory Council 1996, p6-11). At least 710,000km^2 of forest, woodland, open woodland, and shrubland have been converted to grassland or pasture. A comparison of vegetation cover in 1788 and 1988 found that over 1,220,000km^2 of vegetation has changed cover class (forest, woodland, open woodland, shrubland, grassland/pasture) since 1788 (17% of total land mass). Cropping, forestry, mining, grazing, and human settlements are activities which have led to these changes in vegetation cover. For example, about 60% of Australia's land area is occupied by agricultural and pastoral enterprises (State of the Environment Advisory Council 1996, p. 2-24). Vegetation change is a key process in converting natural systems to agricultural systems. Thus, in one regional study, vegetation cover led to major changes in biota (Hobbs *et al.* 1992), soil properties (Nulsen 1992), and the hydrological balance (McFarlane *et al.* 1992). Clearly, changes in vegetation cover can be substantial and influence the function, structure, and composition of ecosystems.

We discuss the influence of vegetation cover changes by outlining the potential impacts of grazing, habitat fragmentation and forestry practices.

Livestock grazing

The major influence of grazing on landscapes is to cause degradation by altering vegetation structure, plant regeneration, fauna distribution, soil properties, and water flow patterns. The effect of grazing on ecosystems is often not as abrupt as other vegetation changes such as forest clearance. However, its impacts are widespread because grazing is the predominant land-use in Australia.

Introduced grazing animals can have a major effect on native ecosystems by altering species distribution and vegetation structure. Impact varies between different ecosystems and different species within those ecosystems. In the Australian Alps, various shrub species increased or decreased their distributions under the influence of grazing (Wimbush & Costin 1979a,b,c). At the community level, grazing has been unequivocally linked to change from grassland to open heath and open heath to closed heath (Williams 1990, Wahren *etal.* 1994). While shrubs begin to dominate in alpine ecosystems, semi-arid areas may show an entirely different trend. In the semi-arid zone, palatable perennial species appear to be particularly threatened by grazing disturbance (Pressland 1984, Cheal 1993). This can lead to a variety of outcomes in community composition. Wilson (1990) found woody species dominated areas previously supporting perennial species, whereas Cheal (1993) found a complete lack of regeneration of any trees and shrubs in a semi-arid area exposed to grazing pressure. This lack of regeneration is of particular concern as the loss of important functional groups, such as perennial woody shrubs, has the potential to lead to soil erosion, disruption of nutrient cycling, reduction in animal habitat, and reduction in food for plant pollinators (Pettie *et al.* 1995).

Changes in vegetation structure can influence fauna distribution. Grazing may impinge on fauna by directly competing for forage or removing cover. Two lizard species (*Diplodactylus pulcher* and *D. granarensis*) in the W.A. wheat belt were absent from all remnants affected by livestock grazing and trampling (Smith *et al.* 1996). Loyn (1987) also found that grazing substantially influenced habitat quality in remnant vegetation in southern Victoria and this, in turn, effected the composition of bird communities. Grazing has also been shown to impact invertebrate species in a number of locations (Briese and Macauley 1977, Andersen and McKaige 1987, Holt 1996). King and Hutchinson (1983) observed a reduction in the abundance of litter and topsoil microarthropods, nematodes, enchytaeids, and litter dwelling macroinvertebrates in grazed areas.

Grazing influences soil properties and water flow patterns. Grazing and trampling by cattle and sheep have been shown to decrease infiltration rates, root growth, steady state flow rates, and hydraulic conductivities (Profitt *et al.* 1993, Holt 1996). Several authors have also found that grazing increases bulk density of the soil (Carr and Turner 1959a,b, Wimbush and Costin 1979a,b,c, Willatt and Pullar 1983). Grazing has been shown to increase soil temperature range and reduce soil moisture (King and Hutchinson 1983). The impact of grazing on these soil properties may be due to the physical effects of compaction, or alternatively can be due to the loss of organisms associated with the soil (Holt 1996). A study by Holt (1996) found a significant drop in mites, termite diversity and termite activity in heavily grazed semi-arid land. A significant reduction in the number of termite galleries in the upper 25mm of soil was thought to be the major factor causing an increase in bulk density and a decrease in hydraulic conductivities. The effects of grazing on soil properties can be long-term. For example, Braunack and Walker (1985) found the effects of sheep grazing were detectable sixteen years after grazing had ceased.

Measurement of the impact of grazing is difficult on a landscape-wide basis. It is not simply a function of the stock density, but of various ecological factors including the inherent biological productivity and resilience of the landscape. In semi-arid areas, the ecological impacts of exotic grazers may be spatially correlated with artificial water points (e.g. dams and bores).

Vegetation loss and fragmentation effects

There have been major changes in the Australian environment since white settlement (State of Environment Advisory Council 1996). The extent of vegetation clearing has led to major changes in regional vegetation cover (e.g. Graetz *et al.* 1995) and resulted in dramatic reduction of losses of particular plant communities like the Brigalow (Nix 1992, Alexandra 1995, Fensham 1996).

Habitat loss is a major process threatening species persistence (Groombridge 1992). For example, Burgman and Lindenmayer (1998) noted that of those species listed by the IUCN as threatened, about 60% of birds and 80% of mammals had declined as a result of habitat loss. Indeed, in some ecosystems such as those dominated by Brigalow (*Acacia harpophylla*) where extensive areas of native vegetation habitat has been cleared (Nix 1994, Fensham 1996), large losses of biodiversity have undoubtedly occurred (Gordon 1984). Andren (1994) demonstrated that in relatively intact landscapes (such as those with more than 30% of the original vegetation cover still extant), the loss of habitat was often the best predictor of species loss. However, when larger areas of original landscape cover are lost and the remaining areas become more extensively fragmented, the loss of species and/or the extent of declines in the sizes of populations of particular taxa, is greater than would be predicted from the process of habitat reduction alone (Andren 1994). Under these highly fragmented habitat conditions, the persistence of small populations in small, isolated patches is jeopardised. Hence, area and isolation factors also begin to influence population dynamics. Thus, factors which influence small populations such as demographic stochasticity and environmental variability, may become important in these circumstances. Moreover, localised extinctions of sub-populations in an array of small patches may, in turn, place a population at a regional scale, at risk.

Fragmentation of landscapes and habitat can occur as a result of natural processes such as geological events (e.g. volcanic eruptions), wildfires (Pickett and Thompson 1978), or windstorms (Foster 1980). However, clearing and landscape modification by humans is by far the most significant factor resulting in habitat fragmentation. Indeed, habitat fragmentation is considered to be a major process threatening the conservation of biodiversity worldwide and contributing significantly to the present extinction crisis (Wilcox and Murphy 1984, Wilcove *et al.* 1986, Saunders and Hobbs 1991).

These processes result not only in an overall reduction in the available habitat (Andrén 1994), but also sub-division of the remaining areas as well as potentially detrimental biotic and abiotic edge effects at the margins of habitat fragments (e.g. Temple and Carey 1988). Changes in biodiversity are concomitant with modifications to landscapes like habitat fragmentation (Saunders *et al.* 1987, Andrén 1994, Forman 1996). The effects of habitat fragmentation have been examined in a wide range of studies in Australia, encompassing investigations of an array of different groups including:- birds, (e.g. Loyn 1985, 1987), arboreal marsupials (e.g. Suckling 1982; Laurance 1990), reptiles (Sarre 1995), and invertebrates (e.g. Davies 1993; Margules *et al.* 1994).

For example, the impacts of habitat fragmentation have been examined for assemblages of mammals inhabiting the Naringal region of south-western Victoria that has been heavily modified by Europeans during past 150 years (Bennett 1990a, 1990b). The region was first settled by Europeans in the late 1830s when the area was originally covered by “thick forest and was densely wooded”. The forests in the area began to be extensively cleared 30-40 years after first settlement. In addition, forestry activities were extensive throughout the region. By 1947 approximately half the native vegetation in the Naringal region had been cleared, although the rate of forest removal accelerated substantially during the following decades and by 1966 only 19% of the original vegetation cover remained (Bennett 1990b). In 1980, a total of 8.5% of the original vegetation cover remained uncleared. In 1947 about 90% of remnant vegetation occurred as patches exceeding 100 ha in size. In 1980, 92% of remnants measured less than 20 ha and none were larger than 100 ha (Bennett 1990b). The area now supports a large number of small dairy farms and the small patches of native vegetation are either surrounded by pasture or occur as remnant riparian strips or roadside reserves.

Bennett (1990b) compared the historical and present status of populations of mammals in the Naringal area using a range of sources of data including:- historical and anecdotal records, collection of road kills, archival records from museums, and fauna surveys in 39 patches of 0.3 to 92 ha in size. The results of this work showed that at the time of white settlement there were 33 species of native mammals in the Naringal region. Of these six are now extinct:- Dingo (*Canis familiaris dingo*), Tiger Quoll (*Dasyurus maculatus*), Eastern Quoll (*Dasyurus viverrinus*), Common Wombat (*Vombatus ursinus*), Koala (*Phascolarctos cinereus*) and Eastern Pygmy Possum (*Cercartetus nanus*). The first four of these species were hunted extensively as part of pest control programs. The loss of species has been countered by invasions of six introduced taxa:- House Mouse (*Mus musculus*), Black Rat (*Rattus rattus*), Red Fox (*Vulpes vulpes*), Feral cat (*Felis catus*), European rabbit (*Oryctolagus cuniculus*), and Brown Hare (*Lepus capensis*) (Bennett 1990b). While the identity of taxa which comprise the mammal assemblage has changed dramatically, the actual mammal species richness in the Naringal region has actually remained unchanged since white settlement. In addition to the extinction of mammal taxa at Naringal, there has also been a decline in the abundance of many other species. The loss and decline of species has not been a random process and a range of processes have been important, including:- (1) the modification of forest habitat (particularly the loss of mature trees and, in turn, hollows they contain), (2) loss and fragmentation of forest habitat, (3) hunting activities to control populations of vertebrate pests, and, (4) the impacts of exotic plants and animals (Bennett 1990a, 1990b). For example, Bennett (1990a) showed that species like the Southern Brown Bandicoot (*Isodon obelulus*), Sugar Glider (*Petaurus breviceps*) and Red-necked Wallaby (*Macropus rufogriseus*) were absent from smaller patches.

Effects of timber harvesting

Extensive areas of forest have been cleared for agriculture and grazing since white settlement (Resource Assessment Commission, 1992; State of the Environment Australia, 1996) and it is likely there have been large losses of biodiversity as a result (Recher 1996). Large areas of forest in Australia have also been harvested to produce timber and pulpwood (Resource Assessment Commission, 1992; State of the Environment Australia, 1996). Indeed, during the past 25 years there has been increasing public concern about the potential effects of silvicultural practices in places managed for wood production, including those areas of native forest that have been cleared to establish exotic softwoods (e.g. Routley and Routley 1975, Dargavel 1995).

Logging operations may have a number of direct and indirect effects on nature conservation although further work needs to be undertaken to determine the magnitude and importance of such impacts. Some of these potential effects are briefly outlined below; a more detailed treatment of these issues is well beyond the scope of this report and would warrant an extensive major study in its own right. Further, more detailed discussions of this topic can be found elsewhere (e.g. Lindenmayer 1994, Recher 1996, Lindenmayer and Franklin 1997).

The results of a number of studies have revealed that some forms of timber harvesting such as clearfelling (and associated fire regimes) result in major changes in plant species composition and stand structure (Mueck and Peacock 1992, Ough and Ross 1992; Kirkpatrick 1994a, Williams and Gill 1995, Mueck *et al*. 1996). Such changes may, in turn, have detrimental effects on an array of plants and animals (Lindenmayer and Franklin 1997). Notably, traditional silvicultural systems such as selective cutting or shelterwood logging applied over several cutting cycles will have the same cumulative effects on stand structure as clearfelling (Gibbons and Lindenmayer 1997a; Lindenmayer and Franklin 1997).

The distribution and abundance of key structural elements of old forests like large old living and dead trees are often changed markedly by logging operations. For example, Lindenmayer *et al*. (1991a) found significantly fewer trees with hollows in logged and regenerated montane ash forests than areas where harvesting had not taken place. Gibbons and Lindenmayer (1997b) and Gibbons (unpublished data) reported similar results from extensive surveys in East Gippsland. These trees can be an important habitat resource for a wide array of vertebrate and invertebrate taxa (Scotts 1991, Gibbons and Lindenmayer 1997a). Lindenmayer *et al*. (1991b) found that arboreal marsupials were rare or absent from areas of forest which supported limited numbers of hollow-bearing stems.

Logging operations may have other important negative effects on forest ecosystems such as changing the size and spatial arrangement of age classes, particularly stands of old growth forest (Franklin and Forman 1987). The cumulative effects on landscape pattern of many harvested areas is an important factor influencing the distribution of some forest-dependent taxa. For example, the distribution of the Yellow-bellied Glider (*Petaurus australis*) in the montane ash forests of the Central Highlands of Victoria now appears to be virtually confined to water catchment forests that have burnt but **not** logged (Milledge *et al*. 1991, Incoll 1995, Lindenmayer *et al*. 1998). This species appears to have been lost from wood production areas that have been both burnt in large-scale wildfires **and** subject to extensive and intensive logging operations (Lindenmayer *et al*., 1998). These findings appear to be related to the limited areas of old growth montane ash forest that now occur in the Central Highlands of Victoria; the largest single remaining patch in timber production areas is about 50 ha in size whereas the water catchments contains stands exceeding 5000 ha in size (Lindenmayer *et al*., 1998).

Introduced Species

Introduction

European settlement has added an extensive range of species to the Australian biota, including 25 mammals species, 3 reptile and amphibian species, 37 bird species, 8 marine fish species, 21 freshwater fish species, 1500-2000 vascular plants and an unknown number of non-vascular plants and invertebrate species (compiled by Fox 1995). The potential impact of these species may be assessed by their contribution to the estimated species richness. Introduced plants contribute 6.9-9% of estimated plant species richness, birds 3.6%, mammals 8.0%, reptiles 0.4%, freshwater fish 12.2%, and marine fish 0.2% (Fox 1995).

There are a myriad of reasons for the introduction of species to Australian ecosystems including, sport (foxes and freshwater fish), amenity (blackberries, house sparrow), pest control (cane toad), soil stabilization and revegetation activities (grass species), agricultural and plantation crops (*Pinus radiata*), agricultural livestock (sheep, cattle), accidental (Pacific seastar), and transport (camel, horses).

Few ecosystems in the world are free from the impact of introduced species. Biological invasions have contributed to more extinctions than any other human activity except for land-use change (D'Antonio and Vitousek 1992). The impact of introduced species is of particular concern because the process is rarely reversible (Coblentz 1990). Some degraded ecosystems can be rehabilitated (although many species may be lost even after restoration). However, once an introduced species becomes established, it may be extremely difficult to eradicate. Indeed, O'Brien (1990) noted that although there have been many unintended extinctions of native animal in Australia, not one targeted program has yet fully eradicated an introduced vertebrate pest.

To summarise the threats to Australian ecosystems, we can group introduced species into those that:

- **a.** alter resource levels (e.g. food, nutrients, shelter);
- **b.** alter community composition and relationships (e.g. introduced predators);
- **c.** alter disturbance regimes (e.g. fire and flooding);
- **d.** alter the physical environment (e.g. microclimate, river structures).

(modified from Fox 1995)

Changes to resource levels

The reduction in range of the bilby, *Macrotis lagotis*, has been associated with competition for resources with the rabbit, *Oryctolagus cuniculus* (Southgate 1990). There is evidence that the rabbit competes with the bilby for food items such as the sedge *Cyperus bulbosus*. The rabbit may have also reduced other important food sources such as seeds, witchetty grubs from *Acacia* spp., and *Solanum* spp. fruit. Livestock grazing, introduced predators, and changing fire regimes have also been associated with the demise of the bilby.

Changes to community composition and relationships

Rabbits are known to effect vegetation patterns by selective browsing and preventing the regeneration of plant species. In the subalpine region of NSW, rabbits were found to reduce the cover, biomass, and diversity of forb species (Leigh *et al.* 1987). Rabbits also selectively foraged on flowers and seeds, which was expected to have serious implications for post-fire regeneration. The effect of rabbits was magnified after fire as resprouting plants were heavily grazed and soil exposure was prolonged. Within the arid zone, rabbit damage significantly reduces *Acacia* regeneration (Lange and Graham 1983, Auld 1990) These effects on vegetation undoubtedly effect other biota within these systems. For example, the reduction of vegetation cover by rabbits has been associated with the decline of small macropods due to increased risk of predation (Jarman and Johnson 1977).

Introduced predators such as foxes and cats are represented in most Australian ecosystems and have played a role in altering community dynamics (May and Norton 1996). The potential impact of cats may be inferred by a review of recorded prey species: 186 native bird species, 64 native mammal species, 87 reptile species, 10 amphibian species and unknown number of invertebrate species (Paton 1993). May and Norton (1996) have associated the decline or local extinction of at least eight mammal species with predation by feral cats.

Predation of native species by foxes has been shown to significantly alter population sizes, particularly during re-introduction programs (O'Brien 1990). Fox control within two remnant rock wallaby (*Petrogale lateralis*) populations in south-west WA resulted in increases in their numbers by 138% and 223% over four years (Kinnear *et al.* 1988). Three populations monitored without fox control measures experienced declines of 14% and 85% and an increase of 29%, respectively. However, introduced predators have not been directly linked with the extinction of any native species (Burbidge and

McKenzie 1989). Rather, introduced predators have their greatest impact in the presence of other threatening processes such as changed fire regimes, habitat clearance and introduced herbivores. For example, an increase in the number of feral cats on Macquarie Island (a sub-antarctic island south of Tasmania) following the introduction of rabbits is thought to have lead to the demise of red-fronted parakeets (*Cyanorhampus novaezelandiae*). Prior to the introduction of rabbits, feral cats and red-fronted parakeets had co-existed for almost a century (Taylor 1979).

Changes to disturbance regimes

The introduced buffel grass, *Cenchrus ciliaris*, displaces native grasses along watercourses in Central Australia (Humphris cited in D'Antonio and Vitousek 1992). The species is 2-3 times more flammable than native grasses (Latz 1991 cited in D'Antonio and Vitouseko 1992). Thus, the presence of *Cenchrus ciliaris* has transformed river landscapes from barriers to corridors that facilitate fire movement.

Changes to the physical environment

An introduced shade tree, *Tamarix aphylla*, has invaded the Finke River in NT with serious implications for the river landscape (Griffin *et al.* 1989). *Tamarix aphylla* may alter flooding regimes, increase sedimentation rates, reduce channel width, stabilise and alter island and bar dimensions, lower water tables and cause the disappearance of wetland areas (Graf 1978, Loope *et al.* 1988). Studies of the Finke River have shown changes to riparian vegetation with the displacement of *Eucalyptus camalduensis* and the dominance of the understorey by salt-tolerant chenopods and grass at the expense of native herbs. Resident bird and reptile diversity is lower within the *Tamarix aphylla* stands (Griffin *et al.* 1989).

The relationship between the introduced species and NWI indicators is outlined in the results and analysis section.

Accelerated global climate change

The consensus of scientists is that global atmospheric temperatures are likely to rise between 1.5 - 4.5°C, given a doubling of CO_2 due to human activity in the next century (Schneider 1993, Houghton *et al.* 1996). Various studies have predicted that this will have profound ecological impacts particularly for species distribution. Peters and Darling (1985) suggested that the greatest impact would be on: geographically localised species; genetically impoverished species; highly specialised species; poor dispersing annuals; montane, alpine, arctic and coastal communities (also Brereton *et al.* 1995, Mackey and Sims 1993, Root and Schneider 1993).

Global climate change occurs naturally, and the paleontological record shows that many species have been able to successfully migrate to suitable habitat (e.g. Davis 1989). A capacity for species to migrate across the landscape to keep up with movements in favourable habit conditions under accelerated global climate change will be essential. However this can only occur in many cases, where intervening landscapes remain intact. Species migration can be impeded or prevented by habitat loss and fragmentation. Wilderness areas are by definition large and contain relatively intact ecosystems. There are many geographic settings (such as S.E. Australia) where wilderness areas will encompass a broad climatic gradients. Given accelerated global warming, such areas provide locations which can function as refugia and potentially provide the best opportunity for promoting species migration and hence the maintenance of ecosystem resilience, and the evolutionary potential of landscapes.

Ecological theory and nature conservation

This section very briefly discusses the relevance of theories of conservation biology and ecology to nature conservation, and the links between these theories and wilderness quality.

Island biogeography and the "SLOSS" debate

The Theory of Island Biogeography was proposed by Macarthur and Wilson (1967) and has been a controversial topic in conservation science for several decades (see reviews by Gilbert 1980; Margules *et al.* 1982, Burgman *et al.* 1988). Some of the key elements of the Theory of Island Biogeography have been proposed as general principles to guide the design of nature reserves (e.g. Terborgh 1974, Diamond 1975, Diamond and May 1976, Noss and Cooperrider 1994) and highlight the importance of wilderness areas and very large reserves (Grumbine 1990, Noss and Cooperrider 1994). Indeed, they were incorporated in the IUCN's 1980 World Conservation Strategy (World Conservation Union 1980). In Australia, these key principles have been recommended for use in guiding the management of wildlife populations in wood production forests (e.g. Davey 1989).

Given identical places from which to select protected areas (a situation which never exists in reality), there are six general principles which have been derived from the Theory of Island Biogeography which are considered to be relevant to the design of nature reserves. These are:- (after Noss and Cooperrider 1994 and many other authors cited therein).

Principle 1. Large reserves are better than small reserves

Principle 2. A single large reserve is better than a group of small ones of equivalent total area (the so-called SLOSS debate [Single Large of Several Small]).

Principle 3. Reserves close together are better than ones a long way apart.

Principle 4. A compact cluster of reserves is better than a line of reserves

Principle 5. Round reserves are better than long thin ones.

Principle 6. Reserves connected by a corridor are better than reserves which remain unconnected.

Principles 1 and 2 are the ones of greatest potential relevance for the conjoint notions of wilderness and nature conservation (Noss and Cooperrider 1994) and they have been discussed more than any others in the conservation biology literature over the past 20 years (see Simberloff 1988). In Australia these principles have been recognised by the Victorian Land Conservation Council in its findings following its 1991 special investigation into wilderness (Victoria Land Conservation Council, 1991).

General Principle 1

In most circumstances, larger reserves will typically support a greater diversity of habitats, contain more species and larger populations of individual taxa than smaller reserves. Notably, this "general principle" does not owe its origin to Island Biogeography, but rather it comes from earlier research and general observations on species-area relationships (i.e. there are more species in larger areas than smaller ones) (e.g. Preston 1962).

Under conditions where natural vegetation cover dominates a landscape, the general outcome of Principle 1 will typically be quite simple - a single large reserve will usually be better than a number of smaller ones of equivalent size. Moreover, in these landscapes, large reservations are important for observing the impact of natural disturbances such as fire. In these cases, large areas need to be bigger than the typical size or extent of disturbance events (Hobbs and Huencke 1992; Morrison *et al.* 1992) ensuring that some places remain unaffected by perturbation and provide refugia for some organisms to persist and/or source areas for other taxa to eventually recolonise locations recovering post disturbance. Hence, natural connectivity (*sensu* Noss 1991,Forman 1996) remains extant within such heterogeneous landscapes.

The reliability of general principle 1 may change in landscapes heavily disturbed and fragmented by humans and thus places which may contain a mosaic of cleared and uncleared areas. Under such situations, habitat loss and fragmentation can disrupt effective and functional connectivity between undisturbed areas, leading to circumstances where larger reserves may not always be appropriate. For example, studies of terrestrial gastropods in patches of remnant native vegetation in New Zealand showed that they were virtually confined to smaller habitat patches. This was because smaller patches did not support populations of feral pigs (*Sus scrofa*) which were a major predator of the snails (Ogle 1987).

General Principle 2

Where a landscape is dominated by natural vegetation cover, a single large reserve is the preferred reserve option. However, complex trade-offs occur in environments where there has been extensive human-induced habitat loss and fragmentation. For example, a single reserve may be more susceptible to being destroyed by a single catastrophic event than a set of smaller, spatially separated ones. This is a type of risk-spreading strategy, whereby the probability of the simultaneous destruction of reserved lands is reduced by setting aside a group of protected areas. Other factors such as the dispersal ability of the target species and the spatial location and arrangement of areas set aside have an important effect on the size and number of reserves. If a species has poor movement capabilities, then recolonisation of patches where local extinctions have occurred will be impaired. In these cases, fewer larger reserves or a number of reserves located close together may be required. However, as outlined above, this may increase the risk of correlated disturbance events destroying reserved areas.

The importance of this array of considerations was demonstrated by studies aimed at designing reserves for the endangered Leadbeater's Possum within the forests of south-eastern Australia which are also broadly designated for wood production (Lindenmayer and Possingham 1995). Computer simulation analyses modelled populations using a system of reserves ranging in size and number. A system of several intermediate-sized patches was found to be optimum for the conservation of Leadbeater's Possum. This was due to trade-offs between the impacts of processes that influence small populations (like demographic stochasticity) and the influence of fire regimes on the persistence of large reserves in the landscape. The best option for protecting Leadbeater's possum therefore is to reserve a very large area. However, in those landscapes which are heavily disturbed on an ongoing basis by forestry operations, the largest undisturbed patch may not be large enough, and a network of medium sized reserved patches may be optimal.

It is apparent that the trade-offs between single large and several small reserves are dependent upon a number of inter-acting factors including (after Burgman and Lindenmayer 1997):- (1) the extent of landscape fragmentation, (2) spatial contagion in disturbance regimes (or the spatial extent of areas typically impacted by the same catastrophic event such as the dispersion of high-intensity wildfires), (3) the dispersal capabilities of those taxa targeted for conservation, and thus their ability to re-colonise reserves from disturbed areas, and, (4) the demographics of populations in reserves (Akçakaya and Ferson 1990, McCarthy *et al.* 1994).

Summary

In landscapes that remain predominantly undisturbed by humans (eg large areas of high wilderness quality), large contiguous areas will typically support more species and larger populations of a given species than smaller areas. However, in extensively perturbed and sub-divided landscapes, there are circumstances where an array of small reserves may support more species (e.g. Fitzpatrick 1994), and have a higher probability of maintaining viable populations of particular key taxa (see Simberloff 1988). Indeed, it has become clear that there are many concepts associated with the Theory of Island Biogeography that may have only limited value in guiding the design of nature reserves (see Gilbert 1980, Burgman *et al.* 1988, Simberloff 1988). For example, much of Island Biogeography Theory focuses on numbers of species. However, attempts to conserve maximum numbers of species may be flawed (e.g. for communities dominated by exotic taxa); species composition and the conservation of suites of taxa that include rare and threatened taxa may be a more appropriate conservation goal (Gilmore 1990). Moreover, one of the fundamental assumptions underpinning the comparison of reserve strategies (that there are identical areas that sum to the same total size) will never be met in land allocation issues (Simberloff 1988). This is because there will always be differences between any set of areas for a wide range of key characteristics of significance for nature conservation such as environmental conditions, habitat quality, spatial location, and proximity to other areas of suitable habitat or reserves. These factors will, in turn, be critical for making assessments of the value of areas for the conservation of biodiversity and how networks of reserves should be designed (Burgman and Lindenmayer 1997).

Given problems in the validity of the assumptions underpinning the theory of island biogeography, more recent developments in conservation biology have focussed on the maintenance of viable populations of taxa (e.g. Lamberson *et al.* 1992), assessments of extinction risk (e.g. Burgman *et al.* 1993; McCarthy *et al.* 1994), metapopulation dynamics (Hanski and Gilpin 1991) and the relevance of these concepts to design of reserves, large protected areas and biodiversity management strategies (e.g. Murphy and Noon 1992, Armbruster and Lande 1993).

Maintenance of viable populations and the importance of metapopulation dynamics

A major focus of conservation biology has been the identification of the size of populations likely to be viable in the long-term (Shaffer 1981; Caughley and Gunn 1995). Studies on the susceptibility of populations to extinction have indicated that those comprised of a larger number of individuals have a significantly higher probability of persistence (e.g. Thomas 1989, Berger 1990, Tscharntke 1992). Small and/or isolated populations such as those created by habitat loss or habitat fragmentation are at risk of extinction as a result of a number of potentially important factors including (after Shaffer 1981; Burgman *et al*. 1993):- **(1)** genetic stochasticity (Lacy 1993b), **(2)** demographic stochastacity (McCarthy *et al*. 1994), **(3)** environmental variation (Burgman and Lamont 1992, Caughley and Gunn 1995), **(4)** catastrophic events (Ewens *et al*. 1987), **(5)** negative density dependence or Allee effects (Allee 1931), and, **(6)** spatial stochasticity or spatial sub-division (Gilpin 1987).

Approaches to identify processes that may threaten populations of species and increase their risk of extinction include simulation and analytical modelling (e.g. Shaffer 1981) as well as field experiments and observational studies (e.g. Miller and Mullette 1985). Such investigations have been used to assess the efficacy of different management actions and determine which, if any, will have greatest value in minimising the risks of extinction or quasi-extinction (i.e. a population falling below some nominated size) (see Possingham *et al*. 1993).

A number of field studies have shown that some species persist as a meta-population or a set of sub-populations inter-connected by dispersal. A range of types of metapopulations are known (e.g. Levins 1970; Hanski and Thomas 1994), but in each case, occasional dispersal events prevent or reverse localised extinction within habitat patches (Hanski and Gilpin 1991). Populations of the Euro (*Macropus robustus*) inhabiting remnant native vegetation within the Western Australian wheatbelt is a good example of spatially-structured meta-population in the Australian environment (Arnold *et al*. 1993).

The long-term persistence of a meta-population is dependent upon the size, shape and spatial location of suitable habitat patches as well as the dispersal capability of the organism in question (reviewed by Hanski 1994). Connectivity between remnant patches or reserves (*sensu* Bennett 1990a, Noss 1991) such as the provision of wildlife corridors, may promote the dispersal of plants and animals (e.g. Bennett *et al*. 1994) and, in turn, reverse localised extinction thereby enhancing the probability of metapopulation persistence (see reviews by Bennett 1990; Wilson and Lindenmayer 1996).

Given the potential impacts on small populations of the array of processes listed above, it is clear that large, well connected populations are often required to maximise chances of persistence (Noss and Cooperrider 1994). Such populations will often, in turn, require large areas within which to persist (Grumbine 1990; Armbruster and Lande 1993), although the precise amount of land needed will be dependent upon a wide array of factors such as the spatial requirements of species (e.g. home range size), spatial and temporal distribution of suitable habitat, the frequency and intensity of catastrophic disturbances, the prevalence of features that link sub-populations in a landscape such as wildlife corridors, and, the ability of organisms to utilise such landscape features (Burgman *et al*. 1993, Lindenmayer and Possingham 1995).

Habitat fragmentation and the viability of wildlife populations

Although small reserves have conservation value, there are a number of processes typically associated with small fragmented reserves that influence the dynamics of populations and thus the nature conservation value of a given area. These include:

(1) With greater levels of fragmentation there will increasing sub-division of remaining habitat - resulting in smaller habitat patches. Factors which influence small populations in such small remaining patches such as demographic stochasticity and environmental variability, may become important in these circumstances. Moreover, localised extinctions of sub-populations in an array of small patches may, in turn, place a population at a regional scale, at risk.

(2) The process of increasing habitat fragmentation often leads to larger distances between remnant patches and therefore greater levels of patch isolation. These factors can destabilise metapopulations and lead to localised extinctions (Hanski 1994, Lindenmayer and Lacy 1995).

(3) Patch perimeter to interior ratios are changed in fragmented environments and these can result in edge effects such as increased nest predation and brood parasitism or changed micro-climatic conditions (e.g. in windspeeds, light fluxes and temperature regimes - see Saunders *et al.* 1991 and Forman 1996 for reviews).

(4) For many species there is an overall reduction in the amount of habitat. However, for some taxa such as generalist species which can use disturbed environments, the amount of suitable habitat may increase with fragmentation as the area between remnant patches expands; a good example is the case of the Galah [*Cactua roseicapilla*] which uses cleared areas to forage

(Saunders and Ingram 1995, Burgman and Lindenmayer 1997).

The importance of these processes, and the extent to which they influence the persistence of populations in fragmented environments, varies with respect to a large number of factors such as the spatial arrangement of patches in such highly fragmented environments as well the life history and movement capabilities of particular species (Wilson and Lindenmayer 1996).

Ecosystem resilience

Nature conservation within a landscape will be affected by the resilience of its ecosystems. Resilience is the capacity of a system to absorb the effects of, and recover from, a disturbance, without "flipping" into a different state where the variables and processes controlling structure and behaviour suddenly change (Holling 1996). Resilience is the property that sustains ecosystems, and provides them with the degree of homeostasis needed to maintain a supply of habitat resources for a given taxa.

The habitat of a species is a function of ecosystem processes concerning radiation interception and capture, primary productivity, biomass storage, nutrient storage and cycling, and regulation of the water cycle (these processes are, in turn, all mediated by species of plants, animals and microorganisms). The persistence of a taxon in a landscape is dependent on the continued supply of those habitat resources to which it is adapted. This requires that the ecosystem maintains some degree of homeostasis so that conditions are kept within certain limits thereby ensuring required habitat elements are present. The maintenance of productivity in the landscape by autotrophs (upon which all heterotrophs depend) hinges upon the ability of plant species to evolve and adapt their growth form, physiognomy or life history. The on-going productivity of landscapes, and hence the maintenance at any one point in time of viable populations of species, is therefore the product of a balance between: (1) the ecosystem homeostasis needed to maintain a supply of required habitat resources; and (2) the capacity of autotrophs to adopt to changing environmental conditions.

Resilience is threatened when a disturbance event:

(1) rapidly reduces the storage of biomass or nutrients in the landscape (especially if these have been built up slowly),

(2) decreases the inherent rate of primary productivity by interrupting the supply or availability of heat, light, water or mineral nutrients, so that these resources become limiting, or more limiting, for primary productivity,

(3) results in the local extinction of the dominant autotrophs, decomposers and other taxa that perform key roles in maintaining the resource infrastructure,

(4) results in degradation of the vegetation structure (for example, when understorey vegetation is removed in open woodland ecosystems),

(5) disrupts the flow of genetic material through the landscape.

To date, *in situ* conservation has focussed on protecting specific habitat patches of selected taxa, usually vertebrate animals and vascular plants (although higher levels of ecological organisation, such as vegetation communities, are also the object of conservation assessment (e.g. Specht and Specht 1995)). Hence, the maintenance of ecosystem resilience is generally not recognised as a critical objective for nature conservation. This is unfortunate given its significance to ecosystem self-regeneration, homeostasis, habitat supply and ultimately the persistence of viable populations in a landscape. While species-based approaches will always be useful and necessary, nature conservation must include the management of ecosystems (Recher and Lim 1990). Ecosystem management should have as a prime objective the maintenance of ecosystem resilience.

The concept of ecosystem resilience is also applicable at the global scale where the biota can play a major role in mediating biogeochemical cycles. For example, the biota are a key component of the global carbon cycle. Gorshkov *et al.* (1994) quantified the amount of carbon assimilated by global undisturbed terrestrial ecosystems (9 Gt C y^{-1} absorbed) compared to those landscapes if totally disturbed by modern technological society (15 Gt C y^{-1} ejected). They argued that in order to maintain atmospheric homeostasis it will be necessary to retain substantial areas of Earth in an undisturbed state free of the impacts of modern technological society. They further calculated that in order to achieve this, it will be necessary to restore 20% of currently disturbed land to an undisturbed state.

Summary

The preceding outline of current ecological theory indicates that nature conservation is maximised by:

1. Considering the relevance of the following ecological concepts to nature conservation goals:

   (a) *island biogeography*, eg large reserves are usually better than small reserves;

   (b) *maintenance of viable populations*, e.g. large populations or connected populations in a metapopulation are usually better than small populations;

(c) *extinction risk assessment*, e.g. certain human actions or management actions may elevate extinction risk of a species.

2. Reducing fragmentation.

Fragmentation may reduce the amount of habitat, increases edge-effects, and subdivides and isolates populations.

3. Maximising ecosystem resilience.

Increased resilience maximises the capacity of a system to absorb the effects of, and recover from, a disturbance.

The wilderness concept is relevant to these points because wilderness indicators measure:

a. many of the disturbances associated with modern technological society and which may threaten ecosystem resilience, cause fragmentation, and elevate extinction risks,

the isolation of environments from human infrastructure, thus providing an estimate of the size and structure of landscape pattern as relevant to island biogeography theory and metapopulation theory.

Data Analysis and Results

The discussion to date has documented many of the processes associated with modern technological society that threaten the conservation of nature. It should follow that nature conservation is best promoted in those landscapes where these threatening processes are absent or at least minimised. Hence we can hypothesise that (i) threatening processes will be minimised in a landscape with high wilderness quality and (ii) amongst other things, such a landscape will have a relatively low degree of species endangerment. Two types of analyses were undertaken to explore these hypotheses. The first examined the spatial covariation between NWI and the number of threatened species. The second provided different environmental contexts for the NWI data.

More specifically, the objectives of the analyses were to:

(a) examine whether wilderness quality data can be used as a surrogate for threatening processes,

(b) consider the correspondence between NWI indicators and species-based indicators,

(c) illustrate the importance of providing appropriate environmental context when interpreting wilderness indicators.

Note that the analyses undertaken here constitute exploratory data analysis (as defined by Austin and McKenzie 1988) where hypotheses are explored using pattern analysis techniques. This can be contrasted with confirmatory data analysis where, following implementation of a formal experimental design, data are generated that enable a statistical test of significance to be applied to a null hypothesis. The quality of data available for this study was inadequate to support confirmatory data analysis. The results of the exploratory data analyses undertaken here can only be used to indicate whether there are trends in the data that support a specified hypothesis. These analyses do not exclude alternative hypotheses.

1. Threatened species analysis.

Continent-wide data representing the number of threatened plant and animal species were used as a coarse species-based indication of nature conservation status. The threatened species data were obtained from Environment Australia as published in the Australia: State of the Environment Report 1996 (State of the Environment Advisory Council 1996). These data comprise counts of the number of threatened species within each one-degree grid cells covering the continent. The threatened species data were classified into seven groups:

a. vascular plants

b. mammals

c. reptiles

d. amphibians

e. birds

f. vertebrate species combined

g. all species combined.

Wilderness quality was measured using an index of Total Wilderness Quality from the National Wilderness Inventory undertaken by the Australian Heritage Commission (Lesslie and Maslen 1995). This index is a composite measure of measure biophysical naturalness, apparent naturalness, remoteness from access, and remoteness from settlement. These data are georeferenced on a grid with a resolution of 1km.

All spatial analyses were undertaken using the ARCINFO GIS (Environmental Systems Research Institute 1996). The wilderness quality data were overlaid on the threatened species data to determine the extent of spatial covariance in the pattern of the distribution. This continental-wide assessment is illustrated in Figures 1-5. Increasing wilderness quality is indicated by increasingly dense shades of grey (with black representing highest wilderness quality). The number of threatened species is indicated by the diameter of the circle, with the greatest number of threatened species indicated by the largest diameter circle. Each circle represents the area of one grid cell. At various extremes: large black circles represent high wilderness quality and a high number of threatened species; small black circles represent high wilderness quality and a low number of threatened species; small white circles represent low wilderness quality and a low number of threatened species; and large white circles represent low wilderness quality and a high number of threatened species.

The hypotheses noted above would be supported if, at extremes, larger white circles and smaller black circles dominate, i.e. decreasing wilderness quality coincides with an increase in threatening processes, and hence an increase in the number of threatened species.

The results show that the maximum number of threatened species varied between species groups. This is illustrated in Figures 6-11 which show mean values of total wilderness quality within those grid cells that have the same number of threatened species. Figure 12 shows the relationship for all threatened species. These numbers should be kept in mind when interpreting Figures 1-5. For example, the number of threatened reptile and amphibian species is inadequate to establish whether definite relationships exist. Also, note that:

(a) no wilderness quality data were available for south-west Western Australia

(b) there is a large discrepancy between the resolution of the wilderness quality data grid cells (at 1 km) and that of the threatened species grid cells (at about 100 km). The total wilderness quality index was simply averaged based on the approximately 10,000 1 km cells found within each 1 degree cell, and

(c) larger darker circles can be an artifact of the absence of wilderness quality data within the larger threatened species cells, for example, in the south west corner of Western Australia.

2. Environmental context analysis

Ecosystems differ in terms of their characteristic species composition, net primary productivity (by photosynthesising plants), plant and animal life histories and growth forms, and the disturbance regime (particularly the intensity, frequency and seasonality of fire). Given this, it is reasonable to expect the impact of threatening processes to differ between ecosystems. Similarly, we should expect the ecological significance of wilderness quality indicators also to vary. There is, therefore, a need for environmental context when considering the ecological significance of wilderness quality.

While various options are available for eco-regionalisation (see Mackey *et al.* 1988), two continental classifications were used here to provide environmental context for the wilderness quality data. The first is an existing eco-regionalisation of Australia called IBRA (the Interim Biogeographic Regionalisation of Australia) which delineates broad regions based on large-scaled patterns of taxonomic affinities and vegetation types (Thackway and Creswell 1995). This regionalisation was constructed to test the adequacy of the reserve system.

The second is a terrain-classification produced from a digital elevation model (DEM) of Australia. This DEM was generated by Hutchinson and Dowling (1991) and has a resolution of about 2.5 km. The terrain classification is based on two terrain attributes calculated from a 1/40° DEM for Australia, namely (1) elevation percentile and (2) relief. Elevation percentile is calculated by ranking all the elevations that fall within a specified radius of each grid cell, and recording the percentile ranking of the focus grid cell. Elevation percentile therefore provides an index of local topographic position. Each grid cell was assigned to one of twelve terrain units representing different combinations of these two terrain attributes.

The IBRA regionalisation presents one perspective on large-scale variation in particular ecosystem characteristics. Similarly, the terrain classification provides mapped units which should represent variation in the distribution of certain key environmental resources which effect primary productivity and the disturbance regime. In both cases, the mean total wilderness quality index was calculated by GIS overlay using the NWI database.

Figure 13 shows the mean total wilderness quality for each IBRA region. Figure 14 shows the terrain classification for Australia, represented in nine terrain units. Figure 15 shows the location of eight IBRA regions that were selected to highlight how wilderness quality can vary within IBRA regions and between terrain units. Table 1 shows the distribution of mean total wilderness quality for each of these selected regions.

Figure 13 indicates that, even at a gross level of generalisation, there is considerable variation in the wilderness quality of the IBRA regions. This reflects the fact that differing environmental and ecological conditions exert a control on human land use and resource procurement between regions. As discussed below, the variation in wilderness quality in arid Australia has particular ecological significance.

The terrain classification shown in Figure 14 provides another perspective on the environmental determinants of primary productivity, vegetation growth forms, and plant and animal life histories, although topographic controls on water and nutrient distribution must be complemented by climatic and lithological data if the primary environmental regimes of the biota are to be adequately defined (Nix 1986, Mackey *et al.* 1988). Nonetheless, Figure 15 and Table 1 illustrate that the mean total wilderness quality varies within each IBRA region across the different terrain units, and that there are correlations between the physical environment and the level of disturbance associated with modern technological society.

Discussion

The overall trend indicated in Figure 12 is that the number of threatened species decreased as mean total wilderness quality increased. This trend is also evident for vascular plants (Figure 1 and Figure 6). The anomalous vascular plant cells are located on the border of south west Western Australia and are an artefact of the lack of wilderness data for that region. There were major differences between the trends for the relationships for all species and plants and wilderness quality and that derived for threatened mammals and wilderness quality. In the latter case

(Figure 2 and Figure 7), more threatened mammals occurred in cells assigned high values for wilderness quality. However, the spatial distribution of the relationship between the number of threatened mammal species and mean total wilderness quality was highly variable. While strong positive trends are evident in south east Australia, the patterns in the arid zone are far more complex. In the arid zone, a diversity of patterns can be discerned, for example, a zone of high wilderness quality and high numbers of threatened species; a zone of lower wilderness and high threatened species; and a zone of medium wilderness and threatened species. These results suggest there are complex interactions between wilderness quality *per se*, wilderness quality as captured by the NWI indicators, and threatening processes for certain mammal species in central Australia.

The total number of threatened vertebrates, amphibians, birds, reptiles, and plants all exhibited a general trend of increasing threat with decreasing wilderness quality. Conversely, the maximum number of threatened mammals was associated with high wilderness quality. Both trends warrant further explanation.

The National Wilderness Inventory is based upon the Lesslie model of wilderness quality (Lesslie and Taylor 1985, Lesslie *et al.* 1988). This focuses on the spatial patterns of modern technological infrastructure and land use activities. In many environments, these prove to be either direct measures or good surrogates of threatening processes for particular groups of species. However, in certain environments, the NWI indicators do **not** appear to capture some key threatening processes.

In arid and semi-arid areas, a number of inter-related threatening processes are associated with the decline and extinction of mammal species (see Appendix 1). These include introduced predators, introduced herbivores (rabbits and domestic stock), changes in vegetation structure and composition, changing fire regimes, and climatic variability. Exotic species have a variety of impacts on semi-arid/arid mammals. Feral predators directly feed on native mammals. Exotic herbivores, especially domestic stock and feral rabbits, compete with native mammals for food (Southgate 1990) and shelter resources. Various feedback mechanisms also exist. For example, the pastoral industry has tapped ground water resources thereby increasing surface water (Landsberg *et al.* 1997), enabling both prey and predator numbers to remain high when prevailing weather conditions would otherwise have resulted in reduced animal numbers. This results in increased competition for fodder, especially in times of climatic drought. Large populations of feral herbivores, especially rabbits also support large numbers of feral predators, increasing predation pressures on native mammals. Rabbits directly compete with ground nesting and burrowing mammals for den and nesting resources.

These general effects are further exacerbated by the fact that only parts of the landscape support something that approaches a more regular supply of water and nutrients. These habitat *oases* can function as refuges during climatic drought from where animals can disperse to repopulate the surrounding landscape during inter-drought periods. However, such resource-rich landscape patches are attractive to exotic animals, who displace or prey upon, native animals, thereby increasing their vulnerability (Morton 1990).

Exotic animals are now widely distributed across large areas of arid and semi-arid Australia where the highest numbers of threatened mammals occur. These animals are present in the landscape as a result of European settlement. However, their distribution and impact is not necessarily spatially correlated with the infrastructure associated with European settlement and subsequent modern technological development. The diffuse nature of these distributions means that their considerable ecological impact is not always reflected in the NWI indicators. The index of Biophysical Naturalness could theoretically be improved by including additional indicators related to the number or density of exotic animals. However, the main impediments to this are (i) the difficulty of obtaining reliable spatial estimates of, for example, fox and rabbit numbers, (ii) evaluating the impacts of these animals, particularly those not dependent on the presence of permanent surface water and, (iii) temporal variations in the abundance of exotic animals, particularly in response to factors like climatic conditions. Also, potential stocking is not a sufficient indicator, as impact is related to the productivity and resilience of the ecosystem in addition to the density of animals. The Lesslie model does not require an evaluation of the *impact* of the infrastructure and land use patterns associated with modern technological society. It does not attempt to capture the abundance and effects of exotic and feral animals.

The NWI model still has considerable value for nature conservation evaluation in arid environments. While exotic animal distributions may be diffuse throughout a landscape, other threatening processes which are associated with the NWI indicators may be present (such as habitat fragmentation). In these circumstances, and all other factors being equal, the landscape with the highest wilderness quality as measured by the NWI indicators, will be correlated with fewer total threatening processes and will be a better protected conservation environment.

In other environments, and for other groups of species (e.g. plants) the NWI model shows good correlation with numbers of threatened species. In these cases, the threatening processes either equate with or are associated with the infrastructure and land use patterns of modern technological society. This is the case, for example, with threatened plants in woodland environments which have experienced land use pressures that are reflected in the NWI indicators.

The threatened species analyses highlight the importance of examining wilderness quality data within an explicit environmental context. High wilderness quality as modelled by the NWI indicates that certain threatening processes associated with modern technological society are absent. This has important ecological implications in all environments. However, in certain environments, the NWI indicators are not well correlated with all threatening processes (although as noted above, within these latter environments, the relative wilderness quality as measured by the NWI is still ecologically important).

Even at the coarse level of geographic identification provided by the IBRA regions (Figure 13), the spatial distribution of mean total wilderness quality can be seen to vary, especially within the arid zone. The terrain classification is intended to illustrate one method for stratifying the environment to delineate areas that differ in terms of the distribution and availability of resources relevant to wildlife habitat. The classification used here (Figure 14) only does this crudely due to the coarse resolution of the digital elevation model. Nevertheless, for the selected IBRA regions; wilderness quality can be seen to vary between terrain classes (Figure 15). This suggests that the inherent biological productivity and resilience of ecosystem varies significantly within a given IBRA region, which in turn exerts a major control on the intensity and type of land use activity.

The IBRA and terrain classification illustrate two approaches to providing the necessary environmental context. As noted, these analyses are only indicative, and that more comprehensive environmental context requires finer resolution elevation and substrate data, coupled to climatic and vegetation information (e.g. Mackey *et al.* 1989). Data and models are now available to provide environmental context at a range of scales. These analyses are no longer restricted to meso-scaled processes such as climate but can now be conducted at scales more appropriate to key processes operating at a landscape-level in arid and semi-arid ecosystems. Environmental domains based on such appropriately scaled climatic, substrate and terrain data can be good indicators of the spatial variation in the inherent productivity of ecosystems (Mackey 1993, 1994). The nature and impact of threatening processes, as demonstrated by the threatened species analysis, varies as a function of these factors. When combined with these data, the NWI wilderness quality indicators provide a powerful diagnostic capability to assess the relative impact of modern technological society on nature conservation.

While the analyses presented here have been useful, their limitations need to be stressed. Further analyses should be undertaken to account for the following issues.

- An examination of spatial covariance in the distribution of wilderness quality and functionally related subgroups of threatened species (e.g. foraging guilds) would provide a better indication of the relation between wilderness quality and species endangerment. Spatial data about extinct species would add an important dimension to this work.
- The quality of the threatened species data is poor. The one degree grid cells provide only a coarse approximation of the spatial distribution of threatened species.
- The threatened species data and the NWI data are geo-referenced to different grid resolutions. Significant information is therefore lost as the NWI data has to be averaged over a larger area.
- The IBRA regions and the terrain classification provide only a crude approximation of the spatial variation in resource availability and biological productivity. More research is needed to determine the optimum method for providing environmental context, particularly in semi-arid and arid environments. These analyses might encompass continental data sets relating climate, geology and vegetation types to existing patterns of wilderness.
- We have used only threatened species data here due to their availability. However there are many problems associated with the use and interpretation of threatened species data (e.g. there may be more threatened species in wilderness areas because those species have suffered local extinctions elsewhere). Therefore, further analyses should examine other response variables such as numbers of extinct species.
- Additional analyses should ideally take into account bias in the sampling intensity between areas of high and low wilderness quality, as remote areas may be poorly surveyed. This has implications for data interpretation.

In addition to the broader scaled geographic analyses present here other studies are required.

- Finer resolution analysis is required at a sub-regional scale. We note that obtaining both biological and disturbance data at finer scales remains a challenge.
- Experimentally designed analyses are required that enable the application of confirmatory data analysis, where the statistical significance of results can be formally tested.

The analysis used an index of total wilderness quality which is a composite of four indicators developed for the NWI. These four indicators are based in turn on primary geographical data. Other estimates of wilderness quality and other ways of combining the primary data and associated indicators may be valid.

In summary, the results suggest correlations exist between biological conservation, wilderness quality, and the threatening processes associated with modern technological society. However, the nature of these relations varies with the characteristic biodiversity and primary productivity of ecosystems. Hence, the potential interpretive value of wilderness quality data for nature conservation requires appropriate environmental context.

Discussion and Synthesis

Conservation in dedicated reserves

Wilderness quality as defined here is a measure of the absence of the impact of modern technological society. Our review of processes that threaten the conservation of nature, highlights the need for a network of dedicated nature reserves.

Dedicated reserves provide maximum legislative protection from threatening processes, and hence are an indispensable component of a strategy for the conservation of both species and ecosystems. However, a reserve network cannot be comprised only of residual land not wanted for other purposes (the "worthless land hypothesis", Hall 1988). Rather, reserve networks must be carefully designed if they are to achieve the objectives of maintaining biodiversity, ecosystem resilience and the inherent biological productivity of a landscape. Some important design principles include:

(1) reserve systems should be big enough in area to be able to absorb large scaled perturbations. Generally in Australia, these includes fire, *normal* climatic variation, and the prospect of accelerated global warming over the next 50-100 years (though note that all three are linked) (see Noss 1991, Mackey and Sims 1993). Reserve systems therefore must be evaluated on a continental and regional basis taking into account, amongst other things, mesoscale climate and its affect on land surface processes

(2) reserve systems must be spatially configured to promote the flow of genetic material through the region, to allow genotypes to migrate to more favourable conditions, and to allow the influx of better adapted taxa

(3) reserve systems need to encompass, on a regional basis, the full range of patterns in the resource infrastructure that underpin plant and animal habitat as defined by (i) autotroph productivity and (ii) the primary environmental regimes at meso- and topo-scales

(4) reserve systems must possess the highest ecological integrity for that type, that is, be as free as possible from processes that threaten ecosystem resilience and biological conservation. In many cases, places of relatively low ecological integrity will have to be included in the reserve system as high integrity places may be lacking for that ecosystem type. In these cases specific management regimes will be needed to promote restoration of resilience and the characteristic species composition.

Reserve systems therefore must be representative of the genetic, species and ecosystem diversity found in a region, and be configured to promote the viability and adequacy of these phenomona through time (see Commonwealth of Australia 1997)

Wilderness quality data are invaluable for the tasks of designing an optimum reserve network based upon these design features. This is because they can help identify places that possess the highest ecological integrity for a given ecosystem type. The present conservation reserve network is not the result of such a landscape evaluation (Pressy 1995). An optimal gap analysis would combine wilderness quality data with information that accounts for the spatial distribution at meso- and topo-scales of (a) the primary environmental resources that drive biotic response, and (b) the autotroph (photosynthesising) plant cover. These two sets of data define the habitat matrix that all plants and animals are dependent upon.

Ecological restoration

Because so much of Australia has been degraded by interference from modern technological society, an optimum gap analysis for a dedicated reserve network will result in locations being identified that require significant ecological restoration. From this perspective, wilderness quality data can be a useful tool for reserve management as it can indicate the degree of degradation. Furthermore, by monitoring the change in wilderness quality through time, the degree of restoration can be quantified as natural vegetation is re-established, roads closed etc.

In certain circumstances, local extinctions of species can be an indicator of ecological degradation. Local extinctions represent stages on the pathway to global extinction (Clark *et al.* 1990). Local extinctions are also an indicator of a fundamental change in the characteristic diversity of a landscape, that is, the biodiversity that can be supported by a landscape in the absence of interference from modern technological society. Both global and local extinctions occur in the absence of human interference, and are part of the natural process of evolution and natural selection (Burgman and Lindenmayer 1997). However, species extinctions due to human interference are generally accepted as an indicator of a loss of biodiversity. Species extinction can mean either that (i) no viable populations of a species occur in the wild or (ii) not a single individual organism exists in the wild or captivity. However, it is important to remember that global extinctions are the end point of what can be a long and complicated history of human interference. Preoccupation with global extinctions can detract attention from the more ubiquitous issue of local extinctions, where human interference results in a species being eliminated from a landscape or region (Lindenmayer and Gibbons 1997).

Information about local extinctions is extremely difficult to obtain as it requires (i) data about the species that were part of the characteristic diversity of a landscape, and (ii) how population numbers are changing over time. Even in the absence of human interference, population numbers naturally fluctuate through time in response to various factors including weather conditions, seasonal changes and resource availability (e.g. MacNally 1996). These considerations highlight the need for long term monitoring of selected taxa in representative landscapes. As a complement to such studies, or even in their absence, wilderness quality data, when associated with threatening processes, may provide a measure of the likelihood a landscape will experience local extinctions.

Conservation in disturbed and production landscapes

Much of the Australian continent has been disturbed by modern technological society, and consequently has a low wilderness quality. Nonetheless, how these landscapes are managed will have a major influence on the conservation of Australia's biodiversity. Our indicative analyses illustrated that the distribution of wilderness quality varies with different environmental conditions. Much of the most biologically productive landscapes have been converted to agriculture and largely cleared of native vegetation. The remaining large areas of high wilderness quality therefore do **not** represent the full range of biodiversity in Australia.

This suggests that two courses of action are necessary. First, additional dedicated reserves will be needed in areas that are presently heavily disturbed, and managed to promote ecological restoration. Second, production areas will need to be carefully managed to maximise the role they can play in the conservation of biodiversity (Lindenmayer and Franklin 1997).

Much of this report has focussed on the conservation of species of plants and animals, that is, conservation biology. We do not question the need for species-orientated work, nor its place as a key nature conservation objective. However, of equal, and many would argue ultimately greater value, is the need for management aimed at ecological conservation. Here, the focus of attention is the need to protect ecological integrity by not only protecting species composition but also by maintaining ecosystem resilience.

These two conservation goals however highlight the different time scales on which management objectives must focus. Conservation biology may require action such as the manipulation of the local environment to ensure particular habitat conditions continue to occur, or that threatening processes are removed or mitigated. In this way, special management arrangements may enable a target species to persist in a heavily disturbed landscape. It is possible, for instance, for Leadbeater's Possum to persist in logged forest landscapes if landscape-wide management prescriptions are put in place for a period of 500-1000 years that provide for the continued supply of key habitat resources and take into account the risk of fire (see Lindenmayer 1996). In contrast, there are also a range of taxa which have been successful in adjusting to disturbance associated with modern technological society.

The scale at which an organism interacts with the environment is also an important consideration. Some animal species, for example, are immobile, have small home ranges, or are relatively small. These characteristics may enable a population to persist in what appears to be a small area surrounded by unsuitable habitat. In these circumstances, small habitat areas can play an important role in species conservation. For example, Kirkpatrick and Gilfedder (1995) discussed the importance of road-side remnants in the conservation of rare plant species in Tasmania.

As we have defined the terms here whereas the focus in conservation biology is taxon-specific, in conservation ecology, the aim is to ensure the long term viability of ecosystems. This requires the maintenance of ecosystem resilience and the potential and capacity of the constituent species for ongoing evolution (though note that many argue both concepts are in fact encapsulated by the term conservation biology).

As with conservation biology, meeting conservation ecology objectives will require careful management of production landscapes as well as the establishment of a network of dedicated reserves. Over-exploitation of production landscapes can lead to a collapse of ecosystem processes and loss of ecosystem resilience, leading to the degradation of biological productivity. Ultimately the self-regenerating capacity of the landscape can be impaired and lost.

Individual land use activities may have marginal additional impacts on ecosystems. However, the accumulated impacts of a series of events may result in a significant loss of ecosystem integrity (Kirkpatrick 1994). The time scale over which these impacts take effect can be longer than a political term of office, an annual budget, a human life span, or any other time interval to which people and organisations are usually sensitive. These factors mitigate against management for the maintenance of ecosystem resilience in production landscapes.

Integrated landscape conservation

The preceding sections have identified four major classes of landscapes where the role that wilderness quality plays in promoting nature conservation varies:

1. remnant landscapes - highly disturbed landscapes which have been largely cleared and replaced by exotic vegetation. Only remnant patches of native vegetation remain
2. production landscapes - areas dominated by native vegetation but from which natural resources are harvested
3. high quality reserve landscapes - large, dedicated nature conservation reserves of high ecological integrity
4. restoration reserve landscapes - large dedicated nature reserves which may presently have low ecological integrity, but are the best available example of that ecosystem type.

All four classes are needed if we are to ensure both the goals of nature conservation are met, that is, conservation of species and conservation of ecosystems. Presently however, these landscapes are not managed in an integrated way on a regional basis. Rather, they are considered separately, and the potential to maximise the conservation of nature by taking advantage of the relationships between them is ignored. The potential benefits of an integrated landscape conservation strategy has been promoted by many authors, (e.g. Noss 1983, Noss and Harris 1986, Norton and Lindenmayer 1991, Noss 1992, Mott and Bridgewater 1992, Noss 1993, Hobbs *et al.* 1993, Recher 1993, Scotts 1994, Morton *et al.* 1995).

A landscape-wide approach to regional nature conservation has evolved in response to the recognition that the current reserve system is neither adequate in size nor sufficiently representative to maintain diversity and ecosystem function (Pressey 1990). Therefore, to secure the conservation of biodiversity we need to first expand the reserve system, and then look beyond dedicated reserve boundaries and consider modifying land-use activities in the surrounding landscape matrix (Lindenmayer and Franklin 1997).

Noss and Harris (1986) and Noss and Cooperrider (1994) have argued the need for a more integrated landscape approach to nature conservation, suggesting that areas of high conservation value (nodes) be considered as elements of multiple-use modules (MUMs), to create a network across space and time. Such a network would protect and buffer important ecological entities and phenomena, while encouraging the movement of individuals, species, nutrients, energy, and even habitat patches through space and time.

This type of land management approximates the biosphere-reserve concept, developed by UNESCO under the Man and the Biosphere program (Franklin 1977, Batisse 1986, Dyer and Holland 1991). A biosphere-reserve is composed of three zones: a central core, strictly protected for nature conservation purposes; a buffer zone, where activities consistent with the protection of the core area may take place (e.g. research, environmental education and training, tourism, and recreation); and a transition area where traditional land use and resource development is permitted with management to ensure the effects of these activities on the remainder of the reserve is minimised (Batisse 1986). Biosphere-reserves are created to represent the major biomes of the world, rather than exist as conservation units within every landscape (Noss and Harris 1986). Thus, some authors believe that the concept requires updating to address recently acknowledged landscape-scale problems (Dyer and Holland 1991).

The approach of Noss and Harris (1986) has been incorporated into the US Wildlands Project (Wild Earth Wildlands Special Issue 1992, Noss 1993). The Wildlands Project has extended these ideas by increasing the scale of management to the regional and inter-regional level (Newman *et al.* 1992, Noss 1993). Connectivity at the continental level ensures linkages for seasonal movement, dispersal, and long distance range shifts (Noss 1992). This permits conservation at the highest geographic scale of ecosystem function, structure, and composition and thus maximises the value of protected areas and probability of population, process, community, and species persistence.

Scotts (1994) proposed a reserve system for the forests of south-eastern Australia addressing four spatial scales, the regional scale (1:2,500,000), the state forest scale (1:125,000), the forest compartment scale (1:25,000), and the logging coupe scale (1:12,500). Using this approach, core conservation units exist as national parks, nature reserves, flora reserves, old-growth forest reserves, and logging prescription reserves at the highest three levels of this scale. Landscape linkages also occur at regional, state forest, and forest compartment scales. Buffers and habitat retention (eg large hollow trees and large logs) exist at the coupe level. Where timber harvesting activities continue, they are modified in intensity and location to promote the maintenance of species, communities and ecosystems.

The role of wilderness

An integrated nature conservation strategy requires the co-ordinated management of the four landscape categories noted above: remnant landscapes; production landscapes; high quality reserve landscapes and restoration reserve landscapes. Areas of high wilderness quality will be a key component of such a strategy. Soulé and Simberloff (1992) argue that the core reserves of a network should be as large as possible, and there should be many of them. Noss and Cooperrider (1994) maintain that large reserves unquestionably offer the best prospects for the long-term maintenance of ecosystem processes and integrity. The merits of large areas of ecosystems as free as possible from threatening processes have also been outlined above in the discussion in previous chapters on metapopulations, viable populations, and biogeographic theory.

Measures of wilderness quality also are useful in identifying the best available example of an ecosystem type for a restoration reserve. Furthermore, ongoing monitoring of wilderness quality through time can help track the success of ecological restoration programs.

Wilderness quality can also contribute to the management of production landscapes. For example, Morton *et al.* (1995) argued for the protection of small, resource-rich areas in the production landscapes of arid and semi-arid Australia. These so-called Excised Management Units (EMUs) would be embedded within a matrix of land units which were managed to support varying intensities of grazing, tourism, mining or settlement. Wilderness quality data could assist in the identification of the EMUs and the optimal spatial configuration of the surrounding landscape management matrix. Hence, in production landscapes, wilderness quality data can help identify core conservation patches, buffer-zones and corridors, i.e. where threatening processes are minimised. Wilderness quality data are therefore potentially useful at all scales in identifying locations important for nature conservation.

Landscape scale

It is useful to note that the term landscape does not refer to a specific geographic extent, that is, it does not correspond to a defined area of the Earth's surface. A common sense interpretation of the term might define it as somewhere between 10,000 to 100,000 hectares in area.

Landscape pattern can be discerned at a range of scales. Integrated landscape conservation therefore is not restricted to a single scale of analysis. For example, in arid and semi-arid Australia, a production landscape defined at one scale is itself nested within a matrix defined at a larger, regional scale. At this larger scale, a mix of the four major landscape categories may be found, including dedicated reserves of large, high quality wilderness areas

The National Wilderness Inventory

Our review of threatening processes identified the array of activities associated with modern technological society causing local and global extinctions and the degradation of ecosystem integrity. These include vegetation clearance and fragmentation, grazing, logging and introduced species. We noted earlier that the indicators used by the National Wilderness Inventory (NWI) capture some, but not all, of these threatening processes. Where they do, the NWI can be used to evaluate the wilderness quality components of an integrated landscape conservation strategy. In those environments where the correlation between threatening processes and the NWI indicators is poor, research is needed to identify additional indicators to ensure that the measurements of wilderness quality have adequate ecological significance.

Concluding Comments

The concept of wilderness in this study is used in three related, but distinct contexts:

1. wilderness quality
2. measures and estimates of wilderness quality, and
3. wilderness areas.

Wilderness quality constitutes a continuum, that is, the condition of a given landscape can be treated along a spectrum of remote and natural conditions. The NWI indicators (remoteness from settlement, remoteness from access, apparent naturalness, and biophysical naturalness) provide one method of measuring wilderness quality. Wilderness areas are places that meet agreed thresholds of remoteness and naturalness.

Nature conservation now encompasses two broad thrusts, namely, biological conservation and ecological conservation. The former is concerned with the *in situ* conservation of viable populations of species, and the latter with the maintenance of ecosystem integrity. While a robust definition is still being developed, ecosystem integrity can be usefully considered in terms of maintaining the characteristic diversity, composition, structure and productivity of an ecosystem, and thereby promoting ecosystem resilience.

Various processes can be identified that threaten the conservation of nature (so-called threatening processes) that are associated with or stem from the impact of modern technological society, including: changes to fire regimes; changes to hydrological regimes; changes to vegetation cover; and the introduction of exotic species. Changes to the vegetation cover include the effects of livestock grazing, loss of native vegetation cover and fragmentation effects, and timber harvesting. These impacts can influence animals that utilise vegetation for shelter and as a source of nutrition. Wilderness quality as defined here is therefore an indirect measure of the likely absence or presence in a landscape of threatening processes; that is, high wilderness quality equates with an absence of threatening processes that are associated with modern technological society.

Of the major categories of threatening processes examined in this study, all are strongly spatially correlated with the intensity of modern land use and infrastructure. Consequently, the NWI indicators generally provide useful measures of the degree to which landscapes possess nature conservation value. The key exception appears to be the lack of correspondence between the NWI indicators threatened mammals and the impact of certain introduced species in certain arid environments. Further research is required to examine how these effects can be captured.

A considerable body of ecological theory and empirical data now exists to support the proposition that large reserves are an integral and indispensable component of nature conservation strategies (McNeely 1994). Large reserves will typically support a greater diversity of habitats, contain more species and larger populations, and are better able to absorb the impact of disturbances. Small and/or isolated populations such as those created by habitat loss or fragmentation are at risk of extinction as a result of a number of factors such as genetic and demographic stochasticity. The long term persistence of a meta-population is dependent, amongst other things, upon the size, shape and spatial location of suitable habitat. Fragmentation by clearing and roading and loss of habitat can increase their risk of extinction by eliminating the landscape conditions needed to support relatively large, well-connected populations.

The maintenance of ecosystem resilience is a critical nature conservation objective that complements species-based approaches. A resilient ecosystem is self-regenerating, and can maintain sufficient homeostasis to ensure a supply of habitat resources needed to maintain viable populations in the landscape of characteristic species. Nature conservation strategies need to pay more attention to ecosystem management with the aim of maintaining ecosystem resilience. Wilderness quality data may provide one measure of the changes in landscape pattern and the degree to which these elevate risk to the viability of meta-populations and ecosystem resilience. However, the relationship between wilderness quality and ecosystem resilience requires further investigation.

A critical question concerns the role of wilderness areas in nature conservation, that is, relatively large areas that have a high measure of remoteness and naturalness exceeding an agreed set of thresholds. In addressing this question, consideration must be given to whether a landscape is heavily disturbed with little natural vegetation cover remaining, or whether it is largely undisturbed by modern technological society and is still dominated by the characteristic vegetation. Heavily disturbed landscapes are unlikely (by definition) to contain any wilderness areas. Nevertheless in these cases relatively small remnants can play a vital role in maintaining viable populations of some species in those landscapes. The wilderness continuum concept is particularly important here as wilderness quality data can help identify the best of what is left.

In those landscapes which retain a high level of integrity, wilderness areas may still exist, and opportunities remain to capture and protect these in very large dedicated reserves. Where this option exists it remains the most desirable strategy for promoting the conservation of nature. Unfortunately, many landscapes are so degraded that no wilderness areas remain of this ecosystem type. In these cases, wilderness quality data can identify the best remaining examples, and high wilderness quality becomes a management target to promote the ecological restoration of the ecosystem. Therefore, the entire spectrum of wilderness quality has a vital role to play in nature conservation, though the nature of the role will vary with landscape context.

An integrated, landscape-wide approach to nature conservation is needed that utilises the full spectrum of wilderness quality across landscape types. This demands evaluation at both a continental and regional scale. At the core of such a nature conservation strategy must be the largest areas of the highest quality available for that type. These core areas must be complemented however in production landscapes by conservation patches, buffer-zones and corridors. Wilderness quality data are therefore useful at all scales and in all landscape types for identifying locations important for nature conservation.

The indicative environmental analyses presented here, based on the IBRA and terrain classification, point to the diversity and variation found across Australia in terms of the composition, structure, and productivity of landscapes. The ecological impact of threatening processes associated with modern technological society varies between these systems. This must be taken into account when interpreting wilderness quality data. Therefore wilderness quality data, such as provided by the NWI, must have an appropriate environmental context in order to be given an ecological enterpretation.

Research recommendations

1. Further work is needed in reviewing and compiling the scientific literature that documents the impact of threatening processes on the conservation of nature.
2. Further analysis with other response variables is required to test wilderness quality and nature conservation relationships. More formal experimentally-designed studies are also needed that allow for the testing of null hypotheses using statistical tests of significance.
3. Operational definitions of ecological integrity and ecosystem resilience are needed, along with methodologies for their evaluation.
4. Additional wilderness quality indicators should be developed that capture the effects of certain exotic species in certain arid environments.
5. Further options need to be explored of how to best provide appropriate environmental context for interpreting wilderness quality.
6. A continental-wide optimal gap analysis is needed as part of an integrated, landscape-wide approach to nature conservation.

Recommended management principles

This section summarises management principles that have emerged in this report that relate to promoting the conservation of nature through wilderness. *Management* is considered here from both a strategic planning and operational perspective.

1. Wilderness management objectives

The principal objective of wilderness management is to maximise remoteness from, and minimise modifications by, the impacts and influences of modern technological society. This includes, in particular, the protection of indigenous species and ecological processes; and the control and, where practicable, eradication of non-indigenous plants and animals. Wilderness management may provide for uses that are compatible with the protection and enhancement of wilderness quality.

2. Active management in wilderness areas

Land that has been allocated for wilderness protection may still be subject to threatening processes. In these circumstances active management may be required to ameliorate their impacts.

3. The wilderness continuum

This concept underpins the use of the wilderness concept in management. It holds that the wilderness condition exists across a spectrum of remote and natural conditions. Within a given landscape, those places that have the maximum wilderness quality will represent, all other factors being equal, the best of what is left for nature conservation purposes.

4. Wilderness quality as an index of threatening processes

As defined here, low wilderness quality means that a landscape has been heavily exposed to the impacts and influences of modern, technological society. Those threatening processes associated with that exposure will therefore also be present. Where the relationship between measured wilderness quality and threatening processes is strong, wilderness quality data can be useful as an index of both ecosystem integrity and population viability.

5. Wilderness areas and ecosystem integrity

Maintaining ecosystem integrity at a landscape scale requires the protection of areas big enough to, amongst other things, absorb the impact of large scale disturbance, and maintain refugia from which protected populations can disperse. Wilderness areas by definition have higher levels of ecological integrity and are defined at a large spatial scale. Wilderness areas provide important opportunities for the maintenance of ecosystem integrity.

6. Integrated landscape conservation

Many landscapes in Australia are severely disturbed and no wilderness areas remain. A conservation strategy based only on wilderness areas will therefore not be representative of Australia's biodiversity. Production landscapes and landscapes that have been disturbed also have critical roles to play in nature conservation. An integrated landscape strategy will have wilderness areas at its core, complemented with the best of the rest from the surrounding regional matrix. Wilderness quality data assists in identifying the most valuable locations for nature conservation irrespective of where they occur on the wilderness continuum.

References

Abram, D., 1991. The mechanical and the organic: on the impact of metaphor in science. In: S.H. Schneider and Penelope J. Boston, (eds.), *Scientists on Gaia*. The MIT Press, Cambridge, 67-74.

Akçakaya, H.R., and Ferson, S., 1990. *RAMAS/Space User Manual: Spatially Structured Population Models for Conservation Biology*. Exeter Software, Setauket, New York.

Allee, W.C., 1931. *Animal Aggregations. A study in general sociology*. University of Chicago Press, Chicago.

Alexandra, J., 1995. Bush bashing. *Habitat Australia*, **23**: 19-26.

Allen, T.F.H., and Hoekstra, T.W., 1992. *Toward a unified ecology*. Columbia University Press, New York.

Allen, T.F.H. and Starr, T.B., 1982. *Hierarchy: perspectives for ecological complexity*. University of Chicago Press, Chicago.

Andersen, A.N., and McKaige, M.E. 1987. Ant communities at Rotamah Island, Victoria, with particular reference to disturbance and *Rhytidoponera Tasmaniensis. Proceedings of the Royal Society of Victoria*, **99**: 141-146.

Andren, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos*, **71**: 355-366.

Armbruster, P., and Lande, R., 1993. A Population Viability Analysis for African Elephant (*Luxodonta africana*): How big should reserves be? *Conservation Biology*, **7**: 602-610.

Auld, T.D., 1990. Regeneration in populations of the arid zone plants *Acacia carnei* and *Acacia oswaldii*. *Proceedings of the Ecological Society of Australia*, **16**: 267-272.

Austin M.P., Basinski J.J., Cocks K.D., and Ive J.R., 1977. *South Coast Project: Pre-print of Chapters Describing Land Use Planning Methodology*. CSIRO Division of Land Use Research. Technical Memorandum 77/19, Canberra.

Austin, M.P. and McKenzie, N.J., 1988. Data Analysis. In: *Australian Soil and Land Survey Handbook: Guidelines for conducting surveys.* R.H. Gunn, J.A. Beattie, R.E. Reid and R.H.M. van de Graaff (eds.) Inkata Press, Melbourne and Sydney, Ch.15, pp.210-232

Austin, M.P., Nicholls, A.O., and Margules, C.R., 1990. Measurement of the realized qualitative niche: environmental niches of five Eucalyptus species. *Ecological Monographs*, **60**: 161-177.

Backhouse, G.N., Clark, T.W., and Reading, R. (1995). Reintroductions for recovery of the Eastern Barred Bandicoot *Parameles gunnii* in Victoria, Australia. In: Reintroduction Biology of Australasian fauna. M. Serena editor. Surrey Beatty and Sons, Chipping Norton, pp. 209-218.

Baird, A.M., 1977. Regeneration afer fire in King's Park, Western Australia. *Journal of the Royal Society of Western Australia*, **60**: 1-22.

Bardwell, S., 1974. *National Parks in Victoria, 1866-1956: For All the People for All Times.* PhD thesis, Monash University, Melbourne.

Barnett, J.L., How, R.A., & Humphreys, W.F. (1978). The use of habitat components by small mammals in eastern Australia. Aust. J. Ecol., 3, 277-285.

Batisse, M., 1986. Developing and focusing the biosphere reserve concept. *Nature and Resources*, **22**: 2-11.

Bennett, A.F., 1990a. Habitat corridors and the conservation of small mammals in a fragmented forest environment. *Landscape Ecology*, **4**: 109-122.

Bennett, A.F., 1990b. Land use, forest fragmentation and the mammalian fauna at Naringal, south-western Victoria. *Australian Wildlife Research*, **17**: 325-347.

Bennett, A.F. 1991. Roads, roadsides and wildlife conservation: a review. In: Nature Conservation 2: The Role of Corridors. D.A. Saunders and R.J. Hobbs, (eds.). Surrey Beatty and Sons, Chipping Norton.

Bennett, A.F., Henein, K. and Merriam, G., 1994. Determinants of corridor quality: chimunks and fencerows in a farmland mosaic. *Biological Conservation*, **68**: 155-165.

Berger, J., 1990. Persistence of different-sized populations: an empirical assessment of rapid extinctions in bighorn sheep. *Conservation Biology*, **4**: 91-98

Bolton, B.L., and Latz, P.K., 1978. The mala, Lagochestes hirsutus, (Gould) (Macropodidae) in the Tanami Desert. *Australian Wildlife Research*, **5**: 285-293.

Braunack, M.V., and Walker, J., 1985. Recovery of some surface soil properties of ecological interest after sheep grazing in a semi-arid woodland. *Australian Journal of Ecology*, **10**: 451-460.

Brereton, R., Bennett, S. and Mansergh, I., 1995. Enhanced greenhouse climate change and its potential effect on selected fauna of south-eastern Australia: a trend analysis. *Biological Conservation* **72**: 339-354.

Briese, D.T., and Macauley, B.M., 1977. Physical structure of an ant community in semi-arid Australia. *Australian Journal of Ecology*, **2**: 107-120.

Brown, P.R., 1989`. Management plan for the conservation of the Eastern Barred Bandicoot *Parameles gunnii* in Victoria. Arthur Rylah Institue for Environmental Research. Technical Report Series No. 63. Department of Conservation, Forests and Lands, Melbourne.

Burbidge, A., 1985. Fire and mammals in hummock grasslands of the arid zone. In: J.R. Ford (ed.), *Fire Ecology and Management in Western Australian Ecosystems.* Western Australia Institute of Technology, Perth, pp. 91-94.

Burbidge, A., and McKenzie, N., 1989. Patterns in the modern decline of Western Australian vertebrate fauna: causes and conservation implications. *Biological Conservation*, **50**: 143-198.

Burgman, M., Ferson, S., and Akçakaya, H.R., 1993. *Risk Assessment in Conservation Biology*. Chapman and Hall, New York.

Burgman, M.A., Akçakaya, H.R., and Loew, S.S., 1988. The use of extinction models for species conservation. *Biological Conservation*, **43**: 9-25.

Burgman, M.A., Lamont, B., 1992. A stochastic model for the viability of *Banksia cuneata* populations: environmental, demographic and genetic effects. *Journal of Applied Ecology*, **29**: 719-727.

Burgman, M.A. and Lindenmayer, D.B., 1998. *Conservation biology for the Australian Environment.* Surrey Beatty and Sons, Chipping Norton (in press).

Burnett, S., 1992. Effects of a rainforest road on movements of small mammals: mechanisms and implications. *Wildlife Research*, **19**: 95-104.

Burrows, N.D., Ward, B., and Robinson, A.D., 1995. Jarrah forest forest from stem analysis and anthropological evidence. *Australian Forestry*, **58**: 7-16.

Cadwallader, P.L., 1978. Some causes of the decline in range and abundance of native fish in the Murray-Darling River system. *Proceedings of the Royal Society of Victoria,* **90**: 211-224.

Carr, S.G.M., and Turner, J.S., 1959. The ecology of the Bogong High Plains. I. The environmental factors and grass communities. *Australian Journal of Botany*, **7**: 12-33.

Carr, S.G.M., and Turner, J.S., 1959. The ecology of the Bogong High Plains. II. Fencing experiments in grassland C. *Australian Journal of Botany*, **7**: 34-63.

Carrick, R., 1963. Ecological significance of territory in the Australian Magpie, *Gymnorhina tibicen.* Proceedings of the International Ornithological Congress, **13**: 740-753.

Carthew, S.M. and Goldingay, R.L., 1997. Non-flying mammals as pollinators. *Trends in Evolution and Ecology*, **12**: 104-108.

Caughley, G. and Gunn, A. 1995. *Conservation biology in theory and practice.* Blackwell Science, Cambridge, Massachusetts.

Caughley, G.C., 1994. Directions in conservation biology. *Journal of Animal Ecology*, **63**: 215-244.

Cheal, D.C., 1993. Effects of stock grazing on plants of semi-arid woodlands and grasslands. *Proceedings of the Royal Society of Victoria,* **105**: 57-65.

Chessman, B.C., Robinson, D.P., and Hortle, K.G., 1987. Changes in the riffle macroinvertebrate fauna of the Tanjil River, south-east Australia, during construction of Blue Rock Dam. *Regulated Rivers: Research and Management*, **1**: 317-329.

Christensen, P., and Abbott, I., 1989. Impact of fire in the eucalypt forest ecosystems of southern Western Australia: a critical review. *Australian Forestry*, **52**: 103-121.

Church, R.L., Stoms, D.M., and Davies, F.W., 1996. Reserve selection as a maximal covering location problem. *Biological Conservation*, **76**: 105-112.

Claridge, A.W., Tanton, M.T., Seebeck, J.H., Cork, S.J., and Cunningham, R.B., 1992. Establishment of ectomycorrhizae on the roots of two species of *Eucalyptus* from funal spores contained in the faeces of the long-nosed potoroo (*Potorous tridactylus*). *Australian Journal of Ecology*, **17**: 207-217.

Clark, T.W. and Minta, S.C., 1994. *Greater Yellowstone's Future. Prospects for ecosystem science, management and policy.* Homestead Publishing, Moose, Wyoming.

Clark, T.W., Warneke, R.M. and George, G.G., 1990. Management and conservation of small populations. In: *Management and conservation of small populations*, eds. T.W. Clark and J.H. Seebeck. Chicago Zoological Society, Chicago. pp. 1-18.

Coblentz, B.E., 1990. Exotic organisms: a dilemma for conservation biology. *Conservation Biology*, **4:** 261-265.

Cockburn, A., 1991. *An Introduction to Evolutionary Ecology*. Blackwell, London.

Commonwealth of Australia., 1992. *National Forest Policy Statement*. Advance Press, Perth, Australia.

Commonwealth of Australia, 1997. *Nationally agreed criteria for the establishment of a comprehensive, adequate and representative reserve system for forests in Australia.* A report by the Joint ANZECC/MCFFA National Forest Policy Statement Implementation Sub-committee, Canberra.

Commonwealth of Australia, 1997. *Investing in our Natural Heritage - The Commonwealth's Environment Expenditure, 1997-98.* Statement by Senator The Hon. Robert Hill, Minister for the Environment. 13 May 1997. AGPS, Canberra.

Conner, R.N., 1988. Wildlife populations: minimally viable or ecologically functional *? Wildlife Society Bulletin*, **16**: 80-84.

Costin, A.B., and Groves, R.H., (eds.), 1973. *Nature Conservation in the Pacific*. Proceedings of a symposium on nature conservation in the Pacific. Australian National University Press, Canberra.

D'Antonio, C.M., and Vitousek, P.M., 1992. Biological invasions by exotic grasses, the grass fire cycle, and global change. *Annual Review of Ecology and Systematics*, **23**: 63-87.

Dargavel, J. 1995. *Fashioning Australia's forests.* Oxford University Press, Melbourne.

Davey, S.M., 1989. Thoughts towards a forest wildlife management strategy. *Australian Forestry*, **52**: 56-67.

Davies, K.F., 1993. *The Effects of Habitat Fragmentation on Carabid Beetles (Coleoptera: Carabidae) at Wog Wog, South-eastern N.S.W.* Honours thesis, The Australian National University, Canberra.

Davis, M.B., 1989. Lags in vegetation response to Greenhouse warming. *Climate Change*, **15**: 75-82.

Dearden, P., 1989. Wilderness and our common future. *Natural Resources Journal*, **29**: 205-221.

Dexter, B.D., 1978. Silviculture of the River Red Gum forests of the Central Murray Flood Plain. *Proceedings of the Royal Society of Victoria*, **90**: 175-192.

Diamond, J.M., 1975. The island dilemma: lessons of modern biogeographic studies for the design of natural preserves. *Biological Conservation*, **7**: 129-146.

Diamond, J.M., and May, R.M., 1976. Island Biogeography and the design of nature reserves. In: R.M. May, (ed.), *Theoretical Ecology: Principles and Applications*. W.B. Saunders, Philadelphia, pp. 163-186.

Dickman, C.R., Pressey, R.L., Lim, L., and Parnaby, H.E., 1993. Mammals of particular conservation concern in the Western Division of New South Wales. *Biological Conservation*, **65**: 219-248.

Doeg, J.T., Davey, G.W., and Blyth, J.D., 1987. Response of the aquatic macroinvertebrate commmunities to dam construction on the Thomson River, south-east Australia. *Regulated Rivers: Research and Management*, **1**: 195-209.

Dunphy, M., 1934. Equilibrated minds and wilderness. In: Thompson, (ed.), 1986. *Myles Dunphy, Selected Writings*, p. 202. Ballagrin.

Dyer, M.I., and Holland, M.M., 1991. The biosphere-reserve concept: needs for a network design. *BioScience*, **41**: 319-325.

Ehmann, H. and Cogger, H. (1985). Australia's endangered herpetofauna: a review of criteria and policies. pp. 435-437 In: Biology of Australasian frogs and reptiles. (Ed. G. Grigg, R.Shine and H. Ehmann). Surrey Beatty and Sons and Royal Society of N.S.W., Sydney.

Environmental Systems Research Institute, 1996. *ARC/INFO Version 7.0.4.* Redlands, California.

Ewens, W.J., Brockwell, P.J., Gani, J.M. and Resnick, S.I., 1987. Minimum viable population size in the presence of catastrophes. In: *Viable Populations.* (Ed. M.E. Soulé), Cambridge University Press, New York. Pp.59-68.

Fensham, R.J., 1996. Land clearance and conservation of inland dry rainforest in north Queensland, Australia. *Biological Conservation*, **75**: 289-298.

Flannery, T.F., 1994. *The Future Eaters*. Reed Books, Sydney.

Forman, R.T.T., 1996. *Land Mosaics. The Ecology of Landscapes and Regions*. Cambridge University Press, New York.

Foster, R.B., 1980. Heterogeneity and disturbance in tropical vegetation. In: M.E. Soulé, and B.A. Wilcox, (eds.), *Conservation Biology. An Evolutionary-ecological Perspective*. Sinaur, Sunderland, pp. 75-92.

Fox, M.D., 1995. Conserving Biodiversity: impact and management of exotic organisms. In: R.A. Bradstock, T.D. Auld, D.A. Keith, R.T. Kingsford, D. Lunney, and D.P. Sivertsen, (eds.), *Conserving Biodiversity: Threats and Solutions.* Surrey Beatty and Sons, Chipping Norton, pp. 177-183.

Franklin, J.F., 1977. The biosphere reserve program on the United States. *Science*, vol? 262-267

Franklin, J.F., and Forman, R.T.T., 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. *Landscape Ecology* **1**, 5-18.

Frood, D. and Calder, M., 1987. Nature conservation in Victoria. Study Report Volumes I & II. A Report to the Victorian National Parks Association.

George, R.J., McFarlane, D.J. and Speed, R.J., 1996. The consequences of a changing hydrologic environment for native vegetation in southwestern Australia. In: D.A. Saunders, R.J. Hobbs and P.R. Ehrlich.. (eds.), *Nature Conservation 4: the role of networks*, Surrey Beatty and Sons, Chipping Norton.

Ghassemi, F., Jakeman, A.J., and Nix, H.A., 1995. *Salinisation of land and water resources*. University of New South Wales Press, Sydney.

Gibbons, P. and Lindenmayer, D.B., 1997a. The performance of prescriptions for the conservation of hollow-dependent fauna. Implications for the Comprehensive Regional Forest Agreement Process. CRES Working paper. 1997/2. ISBN 0 86740 477 9.

Gibbons, P. and Lindenmayer, D.B., 1997b. Conserving hollow-dependent fauna in timber-production forests *New South Wales National Parks and Wildlife Service Environmental Heritage Monograph Series No. 3.* 110 pp.

Gilbert, L.E., 1980. The equilibrium theory of island biogeography: fact or fiction? *Journal of Biogeography*, **7**, 209-235.

Gill, A.M., 1977. Management of fire-prone vegetation for plant species conservation in Australia. *Search*, **8**: 20-26.

Gill, A.M., and Bradstock, R., 1995. Extinction of biota by fires. In: R.A. Bradstock, T.D. Auld, D.A. Keith, R.T. Kingsford, D. Lunney, and D.P. Sivertsen, (eds.), *Conserving Biodiversity: Threats and Solutions*. Surrey Beatty and Sons, Chipping Norton, pp. 309-322.

Gill, A.M., Groves, R.H., and Noble, I.R. (eds.), 1981. *Fire and the Australian Biota*. Australian Academy of Science, Canberra.

Gilligan, J.P., 1953. *The Development of Policy and Administration of Forest Service Primitive and Wilderness Areas in the Western United States*. PhD thesis. University of Michigan, Ann Arbor.

Gilmore, A.M., 1990. Plantation forestry: Conservation impacts on terrestrial vertebrate fauna. In: J. Dargavel and N. Semple, (eds.), *Prospects for Australian Plantations*. Centre for Resource and Environmental Studies, The Australian National University, Canberra, pp. 377-388.

Gilpin, M.E., 1987. Spatial structure and population vulnerability. In *Viable populations for conservation.* (Ed. M.E. Soulé.) Cambridge University Press.

Gippel, C.J., and Stewardson, M.J., 1995. Development of an environmental flow management strategy for the Thomson River, Victoria, Australia. *Regulated Rivers: Research and Management*, **10**: 121-135.

Gordon, G., 1984. Fauna of the Brigalow Belt. In: The Brigalow Belt of Australia. A. Bailey (Ed.). Proceedings of a symposium held at the John Kindler Memorial Theatre, Queensland Institute of Technology. 23 October 1982. Published by The Royal Society of Queensland, Brisbane. pp.61-70.

Gorshkov, V.G., Kondratger, K.Y. and Losev, K.S., 1994. The natural biological regulation of the environment. *J. Ecol. Chem.* **3**(2): 85-90.

Government of Victoria. 1988 *Flora and Fauna Guarantee Strategy: Conservation of Victoria's Biodiversity*. Department of Conservation and Environment. Melbourne, Australia.

Graetz, R.D., Wilson, M.A., and Campbell, S.K., 1995. *Landcover Disturbance over the Australian Continent: a Contempary Assessment*. Biodiversity Series, Paper No. 7, Department of the Environment Sports and Territories, Canberra.

Graf, W.L., 1978. Fluvial adjustment to the spread of tamarisk in the Colorado Plateau region. *Geological Society of America Bulletin*, **89**:1491-1501.

Grayson, R.B., Finlayson, B.L., Jayasuriya, M.D., and O'Shaughnessy, P.J., 1992. *The effects of a clearfelling and harvesting operation on the quality of streamflow from a Mountain Ash forest.* Land and Water Resources Research and Development Corporation Project UME 1. Land and Water Resources Research and Development Corporation, Canberra, A.C.T.

Griffin, G.F., Saxon, E.C., and Allan, G.E., 1986. Return of fire management to Uluru. In: P.J. Joss, P.W. Lynch, and O.B. Williams (eds.), *Rangelands: A Resource Under Siege*, Australian Academy of Science, Canberra, pp. 602-603.

Griffin, G.F., Stafford-Smith, D.M., Morton, S.R., Allan, G.E., Masters, K.A., and Preece, N., 1989. Status and implications of the invasion of Tamarisk (*Tamarix aphylla*) on the Finke River, Northern Territory, Australia. *Journal of Environmental Management*, **29**: 297-315.

Griffiths, T., 1989. The natural history of Melbourne: the culture of nature writing in Victoria, 1880-1945. *Australian Historical Studies*, **23**: 339-365.

Griffiths, T., 1996. *Hunters and Collectors: the Antiquarian Imagination in Australia*. Cambridge University Press, Melbourne.

Groombridge, B. (Editor), 1992. IUCN Red List of threatened animals. Gland, Switzerland.

Groves, R.H. (ed.), 1981. *Australian Vegetation*. Cambridge University Press, Cambridge.

Grumbine, R.E., 1990. Viable populations, reserve design, and Federal lands management: a critique. *Conservation Biology*, **4**: 127-134.

Habermehl, M.A., 1983. Hydrology and hydrochemistry of the Great Artesian Basin, Australia. *Australian Water Resources Council Series No. 8*, Volume 3, pp. 83-98.

Hall, C.M. (1988). The "worthless lands hypothesis" and Australia's national parks and reserves. pp. 441-59 In: K.J. Frawley and N.M. Semple (eds). *Australia's ever changing forests.* Australian Defence Force Academy, Canberra.

Hallam, S.J., 1985. The history of Aboriginal firing. In: J.R. Ford (ed.), *Fire Ecology and Management in Western Australian Ecosystems*. Western Australia Institute of Technology, Perth, pp. 7-20.

Hanski, I., 1994. Patch occupancy dynamics in fragmented landscapes. *Trends in Ecology and Evolution*, **9**: 131-134.

Hanski, I., and Gilpin, M., 1991. Metapopulation dynamics: brief history and conceptual domain. *Biological Journal of the Linnean Society*, **42**: 3-16.

Hanski, I., and Thomas, C.D., 1994. Meta-population dynamics and conservation: a spatially explicit model applied to butterflies. *Biological Conservation*, **68**: 167-180.

Hansson, L., and P. Angelstam, 1991. Landscape ecology as a theoretical basis for nature conservation. *Landscape Ecology* **5**: 191-201.

Harper, J.L., and Hawksworth, D.L., 1994. Biodiversity: measurement and estimation (preface). *Philosophical Transactions of the Royal Society of London*, **345**: 5-12.

Harrington, G.N., and Sanderson, K.D., 1994. Recent contraction of wet sclerophyll forest in the wet tropics of Queensland due to the invasion by rainforest. *Pacific Conservation Biology,* **1**: 319-327.

Harris, J.H., 1984. Impoundment of coastal drainage of south-eastern Australia, and a review of its relevance to fish migrations. *Australian Zoologist*, **21**: 235-250.

Hart, B.T., Bailey, P., Edwards, R., Hortle, K., James, K., McMahon, A., Meredith, C. and Swalding, K., 1991. A review of the salt sensitivity of the Australian freshwater biota. *Hydrobiology*, **210**: 105-144.

Houghton, J.J., Meiro Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A. and Maskell, K. (eds.) 1996 *Climate Change 1995: The Science of Climate Change.* Contribution of Working Group 1 to the Second Assessment Report of the Intergovernmental Panel on Climate Change.

Hedrick, P.W., Lacy, R.C., Allendorf, F.W., and Soulé, M.E., 1996. Directions in conservation biology: comments on Caughley. *Conservation Biology,* **10**: 1312-1320.

Helman, P.M., Jones, A.D., Pigram, J.J., and Smith, J.M., 1976. *Wilderness in Australia: Eastern New South Wales and South-east Queensland*. Department of Geography, University of New England, Armidale.

Hillman, T.J., 1986. Billabongs. In: P. DeDeckker and W.D. Williams (eds.), *Limnology in Australia*. CSIRO and W. Junk, Melbourne and Dordrecht, pp. 457-470.

Hobbs, R., and Huenneke, L.F., 1992. Disturbance, diversity, and invasion: implications for conservation. *Conservation Biology*, **6**: 324-337.

Hobbs, R.J., Saunders, D.A., and Arnold, G.W., 1993. Integrated landscape ecology: a western Australian perspective. *Biological Conservation*, **64**: 231-238.

Hobbs, R.J., Saunders, D.A., Lobry De Brun, L.A., and Main, A.R., 1992. Changes in biota. In: R.J. Hobbs, and D.A. Saunders, (eds.), *Reintegrating Fragmented Landscapes. Towards Sustainable Production and Nature Conservation*. Springer-Verlag, New York, pp. 65-106.

Holling, C.S., 1996. Surprise for science, resilience for ecosystems, and incentives for people. *Ecological Applications*, **6**: 733-735.

Holt, J.A., 1996. The effects of grazing pressure on soil animals, and hydraulic properties of two soils in semi-arid tropical Queensland. *Australian Journal of Soil Research*, **34**: 69-76.

Hortle, K.G., and Lake, P.S., 1983. Fish of channelized and unchannelized sections of the Bunyip River, Victoria. *Australian Journal of Marine and Freshwater Research*, **34**: 441-450.

Hutchinson, G.E. 1957. Concluding Remarks. *Cold Spring Harbor Symposium on Quantitative Biology,* **22**: 415-427.

Hutchinson, M.F., and Dowling, T.I., 1991. A continental hydrological assessment of a new grid-based digital elevation model of Australia. *Hydrological Processes*, **5**: 45-58.

Hutton, 1788. Theory of the Earth: or an investigation of the laws observable in the composition, dissolution, and rerstoration of land upon the globe. *Royal Society of Edinburgh*, Transactions **1**: 209-304.

Incoll, R.D. (1995). *Landscape ecology of glider populations in old-growth forest patches.* B.Sc. Honours thesis, Department of Zoology, University of Melbourne, Melbourne.

International Union for the Conservation of Nature, 1980. *World Conservation Strategy*. Gland, Switzerland.

Jarman, P.J., and Johnson, K.A., 1977. Exotic mammals, indigenous mammals and land use. *Proceedings of the Ecological Society of Australia*, **10**: 146-166.

Johnson, D., 1974. *The Alps at the Crossroads: The Quest for an Alpine National Park in Victoria*. Victorian National Parks Association, Melbourne.

Jones, O., 1925. Our forests. In: J. Barrett (ed.), *Save Australia: a Plea for the Right Use of our Flora and Fauna*. Macmillan and Co. Ltd., Melbourne, pp. 55-68.

King, A.R., 1963. *Report on the Influence of Colonization on the Forests and the Prevalence of Bushfires in Australia*. Division of Physical Chemistry, CSIRO, Melbourne.

King, K.L., and Hutchinson, K.J., 1983. The effects of sheep grazing on invertebrate numbers and biomass in unfertilized natural pastures of the New England Tablelands. *Australian Journal of Ecology*, **8**: 245-255.

Kinnear, J.E., Onus, M.L., and Bromilow, R.N., 1988. Fox control and rock wallaby population dynamics. *Australian Wildlife Research*, **15**: 435-450.

Kirkpatrick, J.B., 1994a. *A continent transformed. Human impact on the natural vegetation of Australia.* Oxford University Press, Melbourne.

Kirkpatrick, J.B., 1994b. Wilderness, biodiversity conservation, and science. In: W. Barton, (ed.), *Wilderness: The Future.* Environbook, Sydney, pp.164-169.

Kirkpatrick, J., and L. Gilfedder, 1995. Maintaining integrity compared with maintaining rare and threatened taxa in remnant bushland in subhumid Tasmania. *Biological Conservation*, **74**: 1-8.

Koestler, A., 1967. *The Ghost in the Machine*. McMillan, New York.

Kraufman, S., 1995. *At home in the universe: the search for laws of complexity.* Oxford University Press Inc. pp.321.

Krefft, G., 1862. On the vertebrated animals of the Lower Murray and Darling: their habits, economy, and geographic distribution. *Transactions of the Philosophical Society of New South Wales*, 1862-1865, pp. 1-33.

Lacy, R.C., 1993b. Impacts of inbreeding in natural and captive populations of vertebrates: implications for conservation. *Perspectives in Biology and Medicine*, **36**: 480-496.

Lake, P.S., and Marchant, R., 1990. Australian upland streams: ecological degradation and possible restoration. *Proceedings of the Ecological Society of Australia*, **16**: 79-91.

Lamberson, R.H., McKelvey, R., Noon, B.R., and Voss, C. 1992. A dynamic analysis of northern spotted owl viability in a fragmented forest landscape. *Conservation. Biology*, **6**: 505-12.

Lamont, B.B., and Downes, S., 1979. The longevity, flowering, and fire history of the grass trees *Xanthorrhea preisii* and *Kingii australis*. *Journal of Applied Ecology*, 16: 893-899.

Landsberg, J., and Gillieson, D., 1997. Continental analysis of the distribution of water points in arid and semi-arid Australia. In: J. Landsberg, C.D. James, S.R. Morton, T.J. Hobbs, J. Stohl, A. Drew, and H. Tongway, *The Effects of Artifical Sources of Water on Rangeland Biodiversity*. Final Report to the Biodiversity Convention and Strategy Section of the Biodiversity Group, Environment Australia. Environment Australia and CSIRO, Canberra, pp. 141-143.

Landsberg, J., James, C.D., Morton, S.R., Hobbs, T.J., Stohl, J., Drew, A., and Tongway, H., 1997. *The Effects of Artifical Sources of Water on Rangeland Biodiversity*. Final Report to the Biodiversity Convention and Strategy Section of the Biodiversity Group, Environment Australia. Environment Australia and CSIRO, Canberra.

Lange, R.T., and Graham, C.R., 1983. rabbits and the failure of regeneration in Australian arid zone *Acacia*. *Australian Journal of Ecology*, **8**: 377-381.

Laurance, W. F., 1990. Comparative responses of five arboreal marsupials to forest fragmentation. *Journal of Mammalogy*, **71**: 641-653.

Lee, A.K. and Martin, R., 1988. *The Koala. Australian Natural History Series.* University of New South Wales Press, Sydney.

Leigh, J.H., and Noble, J.C., 1981. The role of fire in the management of rangelands in Australia. In: A.M. Gill, R.H. Groves, and I.R. Noble, (eds.), *Fire and the Australian Biota*. Australian Academy of Science, Canberra, pp. 471-495.

Leigh, J.H., Wimbush, D.J., Wood, D.H., Holgate, M.D., Slee, A.V., Stanger, M.G., and Forrester, R.I., 1987. Effects of rabbit grazing and fire on subalpine environment. I. Herbaceous and shrubby vegetation. *Australian Journal of Botany*, **35**: 433-464.

Leopold, A., 1921. The wilderness and its place in forest recreation policy. *Journal of Forestry*, **19**: 718-721.

Leopold, A., 1949. *A Sand County Almanac*. Oxford University Press, New York.

Lesslie, R., 1991. Wilderness survey and evaluation in Australia. *Australian Geographer*, **22**: 35-43.

Lesslie, R., and Maslen, M., 1995. *National Wilderness Inventory Handbook.* 2nd ed., Australian Heritage Commission, AGPS, Canberra.

Lesslie, R., and Taylor, S.G., 1985. The wilderness continuum concept and its implications for wilderness preservation policy. *Biological Conservation*, **32**: 309-333.

Lesslie, R.G., Mackey, B.G., Preece, R.M., 1988. A computer based method of wilderness evaluation. *Environmental Conservation*, **15**: 225-232.

Levins, R., 1970. Extinction. In: *Some mathematical questions in biology.* (Ed. M. Gerstenhaber.) American Mathematical Society, Providence, Rhode Island pp.77-107.

Lindenmayer, D., 1996. *Wildlife and Woodchips: Leadbeater's Possum, A Test Case for Sustainable Forestry*. UNSW Press, Sydney.

Lindenmayer, D.B. 1994. The impacts of timber harvesting on arboreal marsupials at different spatial scales and its implications for ecologically sustainable forest use and nature conservation. *Australian Journal of Environmental Management*, **1**: 56-68.

Lindenmayer, D.B. Cunningham, R.B., Donnelly, C.F. Triggs, B.J. and Belvedere, M. (1994). The diversity, abundance and microhabitat requirements of terrestrial mammals in contiguous forests and retained linear strips in the montane ash forests of the central highlands of Victoria. Forest, *Ecology and Management*, **67**: 113-133.

Lindenmayer, D.B., and J.F. Franklin. 1997. Forest structure and sustainable temperate forestry: A case study from Australia. *Conservation Biology*. (in press).

Lindenmayer, D.B., and Gibbons, P., 1997. Forestry effects - non existent until extinction occurs? *Institute of Forester's Australia Newsletter.*

Lindenmayer, D.B., and Lacy, R.C., 1995. Metapopulation viability of arboreal marsupials in fragmented old growth forests: a comparision among species. *Ecological Applications*, **5**: 183-199.

Lindenmayer, D.B., and Possingham, H.P., 1995. *The Risk of Extinction: Ranking Management Options for Leadbeater's Possum*. Centre for Resource and Environmental Studies, The Australian National University and The Australian Nature Conservation Agency, Canberra.

Lindenmayer, D.B., Cunningham, R.B. and Donnelly, C.F. 1994. The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-east Australia. VI. Tests of the performance of models of nest tree and habitat requirements of arboreal marsupials. *Biological Conservation* **70**, 143-7.

Lindenmayer, D.B., Cunningham, R.B., and McCarthy, M. (in press). Landscape analysis of the occurrence of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-eastern Australia. (*Biological Conservation*).

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T, Nix, H.A., and Smith, A.P. (1991b). The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-east Australia. III. The habitat requirements of Leadbeater's possum, *Gymnobelideus leadbeateri* and models of the diversity and abundance of arboreal marsupials. *Biological Conservation* **56**, 295-315.

Lindenmayer, D.B., M.T. Tanton, and T.W. Norton. 1991a. Differences between the effects of wildfire and clearfelling in montane ash forests of Victoria and its implications for fauna dependent on tree hollows. *Australian Forestry*, **53**, 61-68.

Loope, L.L., Sanchez, P.G., Tarr, P.W., Loope, W.L., and Anderson, R.L., 1988. Biological invasions of arid land nature reserves. *Biological Conservation*, **44**: 95-118.

Lovelock, J.E., 1979. *Gaia, a New Look at Life on Earth*. Oxford University Press, Oxford.

Loyn, R.H., 1985. Birds in fragmented forests in Gippsland, Victoria. In: A. Keast, H.F. Recher, H. Ford, and D. Saunders, (eds.), *Birds of Eucalypt Forests and Woodlands: Ecology, Conservation, Management*. Surrey Beatty and Sons, Chipping Norton, pp. 323-331

Loyn, R.H. 1987. Effects of patch area and habitat on bird abundances, species numbers and tree health in fragmented Victorian forests. Pages 65-77 in D. A. Saunders, G. Arnold, A. Burbidge and A. Hopkins, (eds). *Nature Conservation: The Role of Remnants of Native Vegetation*. Surrey Beatty and Sons, Chipping Norton.

Lunney, D., and Leary, T., 1988. The impact on native mammals of land-use changes and exotic species in the Bega district, New South Wales, since settlement. *Australian Journal of Zoology*, **13**: 67-92.

Lunney, D., Curtin, A.L., Fisher, D., Ayers, D., and Dickman, C.R., 1997. Ecological attributes of the threatened fauna of New South Wales. *Pacific Conservation Biology*, **3**: 13-26.

Macarthur, R.H., and Wilson, E.O., 1967. *The Theory of Island Biogeography*. Princeton University Press, Princeton.

McCarthy, M.A., Franklin, D.C., and Burgman, M.A., 1994. The importance of demographic uncertainty: an example from the helmeted honeyeater. *Biological Conservation*, **67**: 135-142.

McFarlane, D.J., George, R.J., and Farrington, P., 1992. Changes in hydrological balance. In: R.J. Hobbs, and D.A. Saunders, (eds.), *Reintegrating Fragmented Landscapes. Towards Sustainable Production and Nature Conservation*. Springer-Verlag, New York, pp. 146-186

McKenzie, N.L., and Belbin, L., 1991. Kimberley rainforest communities: reserve recommendations and management considerations. In: N.L. McKenzie, R.B. Johnston, P.G. Kendrick (eds.), *Kimberley Rainforest*. Surrey Beatty and Sons, Chipping Norton, pp 453-468.

Mackey B.G., 1993. A spatial analysis of the environmental relations of rainforest structure. *Journal of Biogeography*, **20**: 303-356.

Mackey B.G., 1994. Predicting the spatial distribution of rainforest structural characteristics. *Journal of Vegetation Science*, **5**:43-54.

Mackey B.G., and Sims, R.A., 1993. A climatic analysis of selected boreal trees species, and potential responses to global climate change. *World Resources Review*, **5**(4): 469-487.

Mackey, B.G., Nix, H.A., Hutchinson, M.F., McMahon, J.P., and Fleming P.M., 1988. Assessing representativeness of places for conservation reservation and heritage listing. *Environmental Management*, **12**(4): 501-514.

Mackey, B.G., Nix, H.A., Stein, J., Cork, E. and F.T. Bullen, 1989. Assessing the representativeness of the Wet Tropics of Queensland World Heritage Property. *Biological Conservation*, **50**(1-4): 279-303.

MacNally, R., 1996. A winter's tale: among-year variation in bird community in a southeastern Australian forest. *Australian Journal of Ecology*, **21**, 280-291.

McNeely, J.A. 1994. Protected areas for the 21st century: working to provide benefits to society. *Biodiversity and Conservation*, **3**: 390-405.

Mangglamarra, G., Burbidge, A., and Fuller, P., 1991. Wunambal words for rainforest and other Kimberley plants and animals. In: N.L. McKenzie, R.B. Johnston, P.G. Kendrick, (eds.) *Kimberley Rainforest*. Surrey Beatty and Sons, Chipping Norton, pp.413-421.

Margules, C.R. and Usher, M.B. 1981. Criteria used in assessing wildlife conservation potential: a review. *Biological Conservation*, **21**: 79-109.

Margules, C.R., Higgs, A.J., and Rafe, R.W., 1982. Modern biogeographic theory: are there any lessons for nature reserve design? *Biological Conservation*, **24**: 115-128.

Margules, C.R., Milkovits, G.A., and Smith, G.T., 1994. Contrasting effects of habitat fragmentation on the scorpion *Cercphonius squama* and an amphipod. *Ecology*, **75**: 2033-2042.

Markos, A. 1995. The ontogeny of Gaia: the role of microorganisms in planetary information network. *Journal of Theoretical Biology*, **176**: 175-180.

May, S.A., and Norton, T.W., 1996. Influence of fragmentation and disturbance on the potential impact of feral predators on native fauna in Australian forest ecosystems. *Wildlife Research*, **23**: 387-400.

Melbourne and Metropolitan Board of Works, 1975. *Report on Environmental Study into Thomson Dam and Associated Works. Volume 1. Proposed Works and Existing Environment*. Melbourne and Metropolitan Board of Works, Melbourne, July 1975.

Metzeling, L., Doeg, T and O'Conner, W., 1995. The impact of salinisation and sedimentation on aquatic biota. pp. 126-136 In: *Conserving biodiversity: threats and solutions.* Ed by R.A. Bradstock, T.A. Auld, D.A. Keith, R.T. Kingsford, D. Lunney and D.P. Sivertsen. Surrey Beatty and Sons: Chipping Norton.

Michaelis, F.B., 1984. Possible effects of forestry on inland waters of Tasmania: a review. *Environmental Conservation*, **11**: 331-343.

Milledge, D.R., Palmer, C.L. & Nelson, J.L., 1991. “Barometers of change”: the distribution of large owls and gliders in Mountain Ash forests of the Victorian Central Highlands and their potential as management indicators. pp. 55-65 In: *Conservation of Australia's forest fauna*. (Ed. D. Lunney). Royal Zoological Society of N.S.W., Sydney.

Morrison, M.L., Marcot, B.G., and Mannan, R.W., 1992. *Wildlife Habitat_Relationships: Concepts and Applications*. University of Wisconsin Press, Madision.

Morton, S.R., 1990. The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia*, **16**: 201-213.

Morton, S.R., Stafford Smith, D.M., Freidel, M.H., Griffin, G.F., and Pickup, G., 1995. The stewardship of arid Australia: ecology and landscape management. *Journal of Environmental Management*, **43**: 195-217.

Mosley, G., 1978. A history of the wilderness reserve idea in Australia. In: G. Mosley, (ed.), *Australian Wilderness: Conservation Progress and Plans*. Australian Conservation Foundation.

Mott, J.J., and Bridgewater, P.B., 1992. Biodiversity conservation and ecologically sustainable development. *Search*, **23**: 284-287.

Mueck, S.G., and R. Peacock. 1992. Impacts of intensive timber harvesting on the forests of East Gippsland, Victoria. VSP Technical Report No. 15. Department of Conservation and Environment, Melbourne.

Mueck, S.G., Ough, K. and Banks, J.C., 1996 . How old are wet forest understories ? *Australian Journal of Ecology* **21**, 345-8.

Murphy, D.D., and Noon, B.R., 1992. Integrating scientific methods with habitat conservation planning: reserve design for Northern Spotted Owls. *Ecological Applications*, **2**: 3-17.

Nash, R., 1978. Historical roots of wilderness management. In: J.C. Hendee, G.H. Stankey, R.C. Lucas (eds.),*Wilderness Management*, Forest Service. United States Department of Agriculture, Miscellaneous Publication No. 1365, pp. 27-42.

Newman, B., Irwin, H., Lowe, K., Mostwill, A., Smith, S., and Jones, J., 1992. Southern Appalachian wildlands proposal. *Wild Earth*, Special Issue: The Wildlands Project, pp.46-60.

Nicholson, P.H., 1981. Fire and the Australian Aborigine - an enigma. In: A.M. Gill, R.H. Groves, and I.R. Noble, (eds.), *Fire and the Australian Biota*. Australian Academy of Science, Canberra, pp.55-76.

Nix, H.A., 1986. A biogeographic analysis of Australian Elapid Snakes. In: Atlas of Elapid Snakes of Australia. R. Longmore (ed.), Bureau of Flora and Fauna, *Australian Flora and Fauna Series* No. **7**: 4-15. AGPS, Canberra.

Nix, H.A., 1992. The Brigalow. In: S. Dovers, (ed.), *Australian Environmental History: Essays and Cases*. Oxford University Press, Sydney, pp. 198-233.

Norton, T.W. and Lindenmayer, D.B., 1991. Integrated management of forest wildlife: towards a coherent strategy across state borders and land tenures. In: *Conservation of Australia's forest fauna.* (Ed. D. Lunney). Surrey Beatty and Sons, Chipping Norton, pp. 237-244.

Noss, R.F., 1983. A regional landscape approach to maintain diversity. *BioScience*, **33**: 700-706.

Noss, R.F., 1991. Landscape connectivity: different function at different scales. In: W.E. Hudson, (ed.), *Landscape Linkages and Biodiversity*. Island Press, Washington, pp.27-39.

Noss, R.F., 1992, The Wildlands Project land conservation project. *Wild Earth*, Special Issue: The Wildlands Project, pp. 10-25.

Noss, R.F., 1993. A conservation plan for the Oregon Coast Range: some preliminary suggestions. *Natural Areas Journal*, **13**: 276-290.

Noss, R.F., and Cooperrider, A.Y., 1994. *Saving Nature's Legacy: Protecting and Restoring Biodiversity*. Island Press, Washington D.C.

Noss, R.F., and Harris, L.D., 1986. Nodes, networks, and MUMs: preserving diversity at all scales. *Environmental Management*, **10**: 299-309.

Nulsen, R.A., 1992. Changes in soil properties. In: R.J. Hobbs, and D.A. Saunders, (eds.), *Reintegrating Fragmented Landscapes. Towards Sustainable Production and Nature Conservation*. Springer-Verlag, New York, pp. 107-145.

O'Brien, P., 1990. Managing Australian wildlife. *Search*, **21**: 24-27.

Ogle, C.C., 1987. The incidence and conservation of animal and plant species in remnants of native vegetation with New Zealand. In: D.A. Saunders, G.W. Arnold, A.A. Burbidge, and A.J. Hopkins, (eds.), *Nature Conservation: the Role of Remnants of Vegetation*. Surrey Beatty and Sons, Chipping Norton, pp. 79-87.

Osborne, W. and Norman, J.A., 1991. Conservation genetics of Corroboree Frogs, *Pseudophyrne corroboree* Moore (Anura: Myobatrachidae): Population subdivision and genetic divergence. *Australian Journal of Zoology*, **39**, 285-297.

Ough, K., and J. Ross. 1992. Floristics, fire and clearfelling in wet forests of the central highlands of Victoria. Silvicultural Systems Project Technical Report No. **11**. Department of Conservation and Environment, Melbourne.

Paton, D.C., 1993. Impacts of domestic and feral cats on wildlife. In: G. Siepen, and C. Owens, (eds.), *Proceedings of a Cat Management Workshop*. Queensland Department of Environment and Heritage, Brisbane, pp.9-15.

Peters, R.L. and Darling, J.D.S., 1985. The greenhouse effect and nature reserves. *Bioscience* **35**(11): 707-717

Pettie, N.E., Friend, R.H., and Ladd, P.G., 1995. Grazing in remnant woodland vegetation: changes in species composition and life form groups. *Journal of Vegetation Science*, **6**: 121-130.

Pickett, S.T., and Thompson, J.H., 1978. patch dynamics and the design of nature reserves. *Biological Conservation*, **13**: 27-37.

Possingham, H.P., Lindenmayer, D.B. and Norton, T.W., 1993. A framework for improved threatened species management using Population Viability Analysis. *Pacific Conservation Biology*, **1**: 39-45.

Powell, J.M., 1976. *Environmental Management in Australia, 1788-1914: Guardians, Improvers and Profit: An Introductory Survey*. Oxford University Press, Melbourne.

Pressey, R.L., 1990. *Australian Zoologist*, **26**:

Pressey, R.L. 1995. Conservation reserves in N.S.W. Crown jewels or leftovers ? *Search* **26**: 47-51.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vanewright, R.I., and Williams, P.H., 1993. Beyond opportunism: key principles or systematic reserve selection. *Trends in Ecology and Evolution*, **8**:124-128.

Pressland, A.J., 1984. Productivity and management of western Queensland rangelands. *Australian Rangelands Journal*, **6**: 26-45.

Preston, B., and Stannard, C., 1994. The re-creation of wilderness: the case for an Australian Ecological Reserve System. In: W. Barton, (ed.), *Wilderness: The Future*. Envirobook, Sydney, pp.127-147.

Preston, F.W., 1962. The canonical distribution of commonness and rarity. *Ecology*, **43**: 185-215.

Profitt, A.P.B., Bendotti, S., Howell, M.R., and Eastham, J., 1993. The effect of sheep trampling and grazing on soil physical properties and pasture growth for red-brown earth. *Australian Journal of Agricultural Research*, **44**: 317-331.

Ramsey, A., 1995. What is wilderness? NSW perceptions. *Australian Parks and Recreation*, Summer 1995: 26-30.

Recher, H. F. 1996. Conservation and management of eucalypt forest vertebrates. pp. 339-88. in R. DeGraff and I. Miller (eds*.) Conservation of Faunal Diversity in Forested Landscapes*. Chapman and Hall, London.

Recher, H.F., 1993. The loss of biodiversity and landscape restoration: conservation, management, and survival, an Australian perspective. In: D.A. Saunders, R.J. Hobbs, and P.R. Ehrlich, (eds.), *Nature Conservation 3: Reconstruction of Fragmented Ecosystems*, Surrey Beatty and Sons, pp. 141-151.

Recher, H.F., and Lim, L., 1990. A review of current ideas of the extinction, conservation, and management of Australia's terrestrial vertebrate fauna. *Proceedings of the Ecological Society of Australia*, **16**: 287-301.

Resource Assessment Commission. 1992. Forest and Timber Inquiry. Final Report. Volume 1. Australian Government Printing Service, Canberra, Australia.

Richardson, K., 1992. Assessing the impacts of roads on small mammal movement in forest habitats of Victoria. BS.c Hons. thesis, Deakin University, Melbourne.

Robertshaw, J.D., and Harden, R.H., 1989. Predation on Macropodoidea: A review. In: *Kangaroos, Wallabies and Rat Kangaroos.* G.Grigg, P. Jarman and I. Hume (eds.). Surrey Beatty and Sons, Chipping Norton, pp. 735-753.

Robertson, M., Vang, K., and Brown, A.J., 1992. *Wilderness in Australia: Issues and Options, a Discussion Paper*. Report to the Minister for Arts, Sports, Environment, and Territories.

Root, T. and Schreider, S.H., 1993. Can large-scale climatic models be linked with multiscale ecological studies? *Conservation Biology* **7**(2): 256-270.

Routley, R. and Routley, V. 1975. *The fight for the forests: the takeover of Australian forests for pines, woodchips and intensive forestry*. Research School of Social Sciences, The Australian National University, Canberra.

Sarre, S., 1995. Size and structure of populations of *Oedura reticulata* (Reptilia: Gekkonidae) in woodland remnants: Implications for the future regional distribution of a currently common species. *Australian Journal of Ecology*, **20**: 288-298.

Saunders, D. A., Arnold, G. W., Burbidge, A. A., and Hopkins, A. J. M., (eds.), 1987. *Nature Conservation: The Role of Remnants of Native Vegetation*. Surrey Beatty and Sons, Chipping Norton.

Saunders, D.A. (1990). Problems of survival in an intensively cultivated landscape: the case of Carnaby's *Cockatoo Calyptorhyncus funereus latirostris. Biological Conservation*, **54**: 277-290.

Saunders, D.A., and Hobbs, R.J., (ed.), 1991. *Nature Conservation 2: the Role of Corridors*. Surrey Beatty and Sons, Chipping Norton.

Saunders, D.A., Hobbs, R.J., and Margules, C.R., 1991. Biological consequences of ecosystem fragmentation: a review. *Conservation Biology*, **5**: 18-32.

Saunders, D.A. and Ingram, J., 1995. *Birds of southwestern Australia.* Surrey Beatty and Sons, Chipping Norton, 296pp.

Schneider, S.H., 1993. Scenarios of global warming. In: *Biotic interactions and global change.* (Eds.) Kareira, P.M., Kingsolver, J.G., Huey, R.B. Sinaver Associates Inc., Massachusetts.

Scott, R., 1994. Mechanised Access and Wilderness Protection. In W. Barton (ed.), *Wilderness: The Future.* Envirobook, Sydney, pp.237-248.

Scotts, D., 1994. Sustaining sensitive wildlife within temperate forest landscapes: regional systems of retained habitat as a planning framework. In: S.R. Dovers, and T. Norton, (eds.), *Ecology and Sustainability of Southern Temperate Ecosystems*, CSIRO, Melbourne.

Shaffer, M.L. 1981. Minimum population sizes for species conservation. *Bioscience*, **31**: 131-134.

Short, J., and Smith, A., 1994. Mammal decline and recovery in Australia. *Journal of Mammalogy*, **75**: 288-297.

Short, J., and Turner, B., 1994. A test of the vegetation mosaic hypothesis: a hypothesis to explain the decline and extinction of Australian mammals. *Conservation Biology*, **8**: 439-449.

Simberloff, D.A., 1988. The contribution of population and community biology to conservation science. *Annual Review of Ecology and Systematics*, **19**: 473-511.

Smith, G.T., Arnold, Sarre, S., Abensberg-Traun, M., and Steven, D.E., 1996. The effects of habitat fragmentation and livestock grazing on animal communities in remnants of *gimlet Eucalyptus salubris* woodland in the W.A. wheatbelt. II. Lizards. *Journal of Applied Ecology*, **33**: 1302-1310.

Snowy Genoa Catchment Management Committee, 1996. *Expert Panel: Environmental Flow Assessment of the Snowy River Below Jindabyne Dam*. Snowy Genoa Catchment Management Committee.

Soulé, M.E., 1987. *Viable Populations for Conservation*. New York: Cambridge University Press, pp. 126-139.

Soulé, M.E., 1994. Saving species versus saving ecosystems: is there a conflict? In: by D.W. McKenney, R.A. Sims, M. Soulé, B.G. Mackey & K.L. Campbell, (eds.), *Towards a Set of Biodiversity Indicators for Canadian Forests*. Natural Resources Canada, Sault Ste Marie.

Soulé, M.E., and Simberloff, D., 1986. What do genetics and ecology tell us about the design of nature reserves. *Biological Conservation*, **35**: 19-40.

Southgate, R.I., 1990. Distribution and abundance of the Greater Bilby *Macrotis lagotis* Reid (*Marsupialia: Peramelidae*). In: J.H. Seebeck, P.R. Brown, R.L. Wallis, and C.M. Kemper, (eds.), *Bandicoots and Bilbies*. Surrey Beatty and Sons, Chipping Norton, pp. 293-302.

Specht, R.L., Specht, A., Whelan, M.B. and Hegarty, E.E., 1995. *Conservation atlas of plant communities in Australia.* Centre for Coastal Management and Southern Cross University Press, Lismore.

State of the Environment Advisory Council 1996. *Australia State of the Environment*. CSIRO Publishing, Melbourne.

Stocker, G.C., and Mott, J.J., 1981. Fire in the tropical forests and woodlands of northern Australia. In: A.M. Gill, R.H. Groves, and I.R. Noble (eds.), *Fire and the Australian Biota*. Australian Academy of Science, Canberra, pp 427-442.

Strahan, R., 1983. *Complete Book of Australian Mammals.* Collins Angus and Robertson Publishers, Sydney.

Suckling, G.C., 1982. Value of reserved habitat for mammal conservation in plantations. *Australian Forestry*, **45**: 19-27.

Tansley, A.G., 1935. The use and abuse of vegetational concepts and terms. *Ecology*, **16**: 284-307.

Taylor, R., 1979. How the Macquarie Island Parakeet became extinct. *New Zealand Journal of Ecology*, **2**: 42-45.

Temple, S.A., and Cary, J.R., 1988. Modelling dynamics of habitat interior bird populations in fragmented landscapes. *Conservation Biology*, **2**: 340-347.

Terborgh, J., 1974. Preservation of natural diversity. The problem of extinction prone species. *Bioscience*, **24**: 715-722.

Thackway, R. and Creswell, I., 1995. *An Interim Biogeographic Regionalisation for Australia: A Framework for Setting Priorities in the National Reserve System Cooperative Program.* Australian Nature Conservation Agency, Canberra.

Thomas, V.C., 1989. The ecology of Leadbeater's Possum in the Cockatoo Swamp, Yellingbo State Nature Reserve. B.Sc. Honours Thesis, La Trobe University, Melbourne, Australia.

Torgensen, T., Habermehl, M.A., Phillips, F.M., Elmore, D., Kubik, D, Geoffrey-Jones, B., Hemmick, T., and Gove, H.E., 1991. Chlorine 36 dating of very old groundwater. 3. Further studies in the Great Artesian Basin, Australia. *Water Resources Research*, **27**: 3201-3213.

Troll, C., 1971. Landscape ecology (geoecology) and biogeocenology - a terminological study. *Geoforum*, **8**: 43-46.

Tscharntke, T., 1992. Fragmentation of Phragmites habitats, minimum viable population size, habitat suitability, and local extinction of moths, midges, flies, aphids and birds. *Conservation Biology*, **6**: 530-536.

Tunbridge, D., 1991. *The Story of the Flinders Range's Mammals*. Kangaroo Press, Kenthurst.

United Nations Framework Convention on Biological Diversity, 1992. United Nations Conference on Environment and Development, Rio de Janeiro, Brazil. Department of Public Information, United Nations, New York.

United States Outdoor Recreation Resources Review Commission (ORRRC). 1962. *Wilderness and Recreation - A Report on Resources, Values and Problems.* Study Report 3, Wildland Research Center, University of California, California.

Verdansky, V., 1945. The biosphere and the noosphere. *American Scientist*, **33**: 1-12.

Victoria. Land Conservation Council. 1991. *Wilderness Special Investigation Final Recommendations.* Land Conservation Council, Melbourne.

Wace, N., 1977. Assessment of dispersal of plant species - the car borne flora in Canberra. *Proceedings of the Ecological Society of Australia*, **10**, 166-186.

Wahren, C-H.A., Papst, W.A., and Williams, R.J., 1994. Long-term vegetation change in relation to cattle grazing in subalpine grassland and heathland on the Bogong High Plains: an analysis of vegetation records from 1945 to 1994. *Australian Journal of Botany*, 42: 607-639.

Walker, K.F., and Thoms, M.C., 1993. Environmental effects of flow regulation on the lower river Murray, Australia. *Regulated Rivers: Research and Management*, **8**: 103-119.

Wilcove, D.S., McLellen, C.H., and Dobson, A.P., 1986. Habitat fragmentation in the temperate zone. In: M.E. Soulé, (ed.), *Conservation Biology: the Science of Scarcity and Diversity*. Sinauer, Sunderland, pp. 237-256.

Wilcox, B.A., and Murphy, D.D., 1984. Conservation strategy: the effects of fragmentation on extinction. *American Naturalist*, **125**: 879-887.

Williams, J., and Gill, A.M., 1995. The impact of fire regimes on native forests in eastern New South Wales. *Environmental Heritage Monograph Series* **2**: 1-68.

Williams, R.J., 1990. Grazing cattle within subalpine heathland and grassland communities on the Bogong High Plains: disturbance, regeneration, and the shrub-grass balance. *Proceedings of the Ecological Society of Australia*, **16**: 255-265.

Williams, W.D., 1987. Salinisation of rivers and streams: an important environmental hazard. *Ambio*, **16**: 180-185.

Williat, S.T., and Pullar, D.M., 1983. Changes in soil physical properties under grazed pasture. *Australian Journal of Soil Research*, **22**: 343-348.

Willis, C.K., Lombard, A.T., Cowling, R.M., Heydenrych, B.J., and Burgers, C.J., 1996. Reserve systems for limestone endemic flora of the Cape lowland fynbos: iterative versus linear programming techniques. *Biological Conservation*, **76**: ? ?

Wilson, A.D., 1990. The effect of grazing on Australian ecosystems. *Proceedings of the Ecological Society of Australia*, **16**: 235-244.

Wilson, A.M., and Lindenmayer, D.B., 1996. *The Role of Wildlife Corridors in the Conservation of Biodiversity in Multi-use Landscapes*. Centre for Resource and Environmental Studies, Greening Australia, and The Australian Nature Conservation Agency.

Wimbush, D.J., and Costin, A.B., 1979a. Trends in vegetation at Kosciusko. I. Grazing trials in the sub-alpine zone, 1957-1971. *Australian Journal of Botany*, **27**: 741-787.

Wimbush, D.J., and Costin, A.B., 1979b. Trends in vegetation at Kosciusko. II. Sub-alpine range transects, 1959-1978. *Australian Journal of Botany*, **27**: 789-831.

Wimbush, D.J., and Costin, A.B., 1979c. Trends in vegetation at Kosciusko. III. Alpine range transects, 1959-1978. *Australian Journal of Botany*, **27**: 833-871.

Table 1

Table 1: The relationship between eight selected IBRA regions and mean total wilderness quality.

IBRA region	Terrain	Relief	Percentile	No.cells	wilderness quality Min	Max	Mean
D'Entrecasteaux							
15	2	1	1	11	5	11	8.636
15	5	2	1	119	0	13	3.370
15	4	2	2	4	0	0	0.000
15	9	2	3	4	14	15	14.500
15	7	3	1	981	0	18	4.809
15	6	3	2	1589	0	20	6.683
15	8	3	3	1584	0	20	9.299
Central Highlands							
16	3	1	2	92	0	10	1.533
16	5	2	1	196	0	11	4.883
16	4	2	2	564	0	13	3.566
16	9	2	3	256	0	20	11.102
16	7	3	1	1320	0	19	6.908
16	6	3	2	4748	0	20	7.291
16	8	3	3	4588	0	20	9.478
Sydney Basin							
20	2	1	1	292	0	9	0.671
20	3	1	2	47	0	12	4.064
20	1	1	3	4	7	9	8.000
20	5	2	1	2737	0	16	1.695
20	4	2	2	1646	0	17	2.767
20	9	2	3	1004	0	19	5.197
20	7	3	1	7717	0	17	4.269
20	8	3	3	9460	0	20	6.656
Central Mackay Coast							
43	2	1	1	379	0	16	1.533
43	3	1	2	43	0	3	0.140
43	5	2	1	1599	0	20	3.343
43	4	2	2	306	0	17	5.435
43	9	2	3	4	12	14	13.000
43	7	3	1	2256	0	20	3.676
43	6	3	2	4532	0	20	5.392
43	8	3	3	5270	0	20	7.917
Carnarvon							
54	2	1	1	8333	0	20	11.24
54	3	1	2	27185	0	20	10.53
54	1	1	3	2174	0	20	13.692
54	5	2	1	7682	0	20	9.948
54	4	2	2	29852	0	20	11.518
54	9	2	3	9791	0	20	12.750
54	7	3	1	352	0	16	7.722
54	6	3	2	1429	0	19	10.575
54	8	3	3	2155	0	20	11.547

Table 1 (cont.)

IBRA region	Terrain	Relief	Percentile	No.cells	wilderness quality Min	Max	Mean
Daly Basin							
73	2	1	1	992	0	18	10.458
73	3	1	2	80	0	18	8.137
73	1	1	3	16	8	17	13.625
73	5	2	1	5540	0	20	10.391
73	4	2	2	5492	0	20	12.188
73	9	2	3	2484	0	20	13.557
73	7	3	1	760	0	18	9.512
73	6	3	2	2824	0	20	10.887
73	8	3	3	3360	0	20	12.501
South East Queensland							
74	2	1	1	1436	0	15	2.776
74	3	1	2	451	0	11	1.951
74	1	1	3	96	0	8	2.792
74	5	2	1	6960	0	15	1.383
74	4	2	2	5680	0	14	1.914
74	9	2	3	1800	0	13	2.206
74	7	3	1	10603	0	14	1.166
74	6	3	2	20410	0	16	1.966
74	8	3	3	20224	0	19	3.318
Victorian Midlands							
78	2	1	1	476	0	10	0.962
78	3	1	2	348	0	8	1.095
78	1	1	3	96	0	0	0.000
78	5	2	1	4648	0	11	0.599
78	4	2	2	6188	0	9	0.580
78	9	2	3	2524	0	11	0.739
78	7	3	1	3616	0	12	0.647
78	6	3	2	9868	0	12	1.003
78	8	3	3	10256	0	14	1.902

Figures 1-15

Figure 1

Figure 1: The Relationship Between Wilderness Quality and the Number of Threatened Vascular Plant Species

Data Sources

Wilderness Quality: National Wilderness Inventory
Environment Australia

Threatened Plants: State of the Environment Advisory
Council, 1996

Interpretation:

Small black dots correspond with locations that have high wilderness quality and a low number of threatened species. In contrast, large white dots correspond with locations that have low wilderness quality and a high number of threatened species.

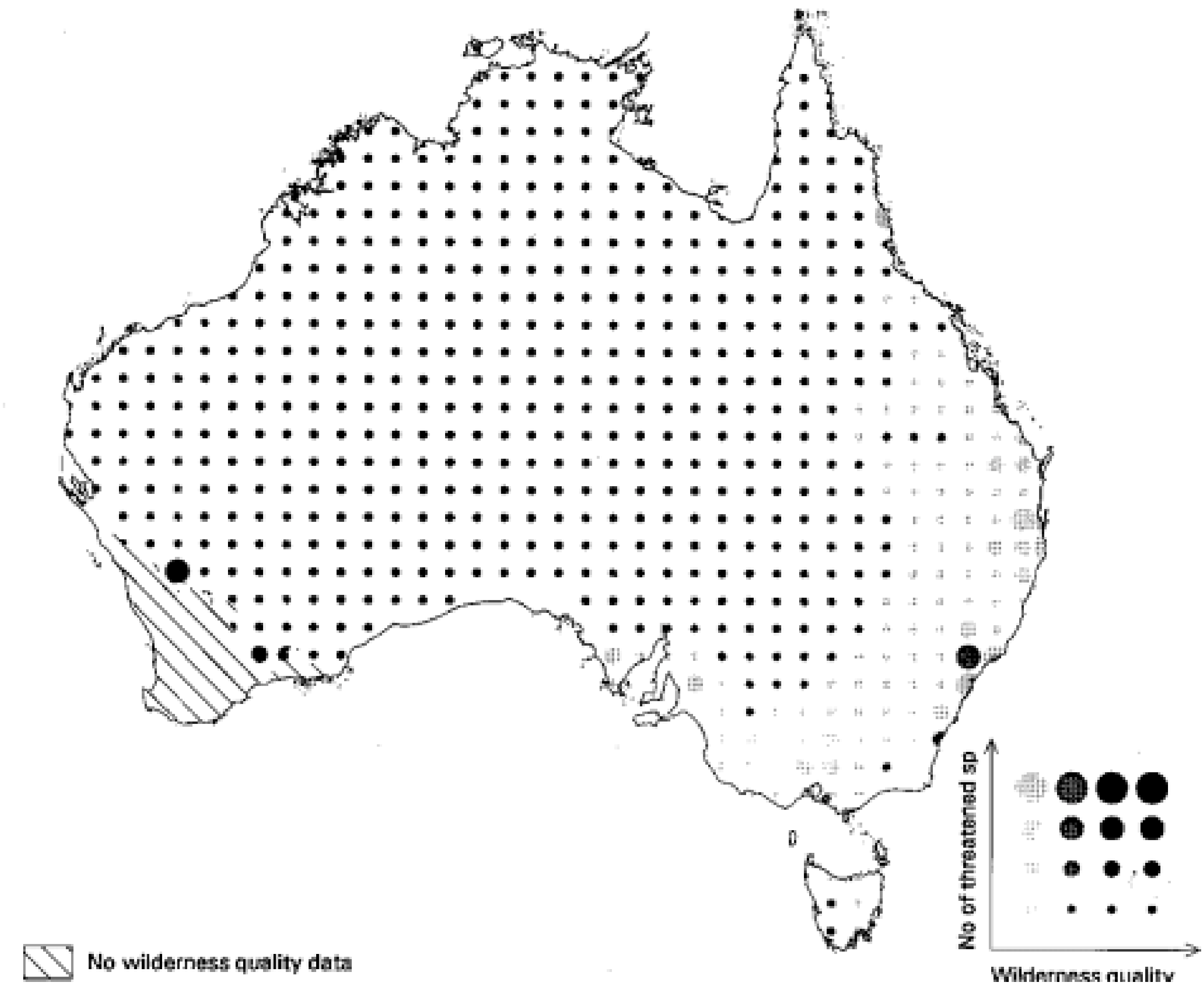


Figure 2

Figure 2: The Relationship Between Wilderness Quality and the Number of Threatened Mammal Species

Data Sources

Wilderness Quality: National Wilderness Inventory, Environment Australia

Threatened Mammals: State of the Environment Advisory Council, 1996

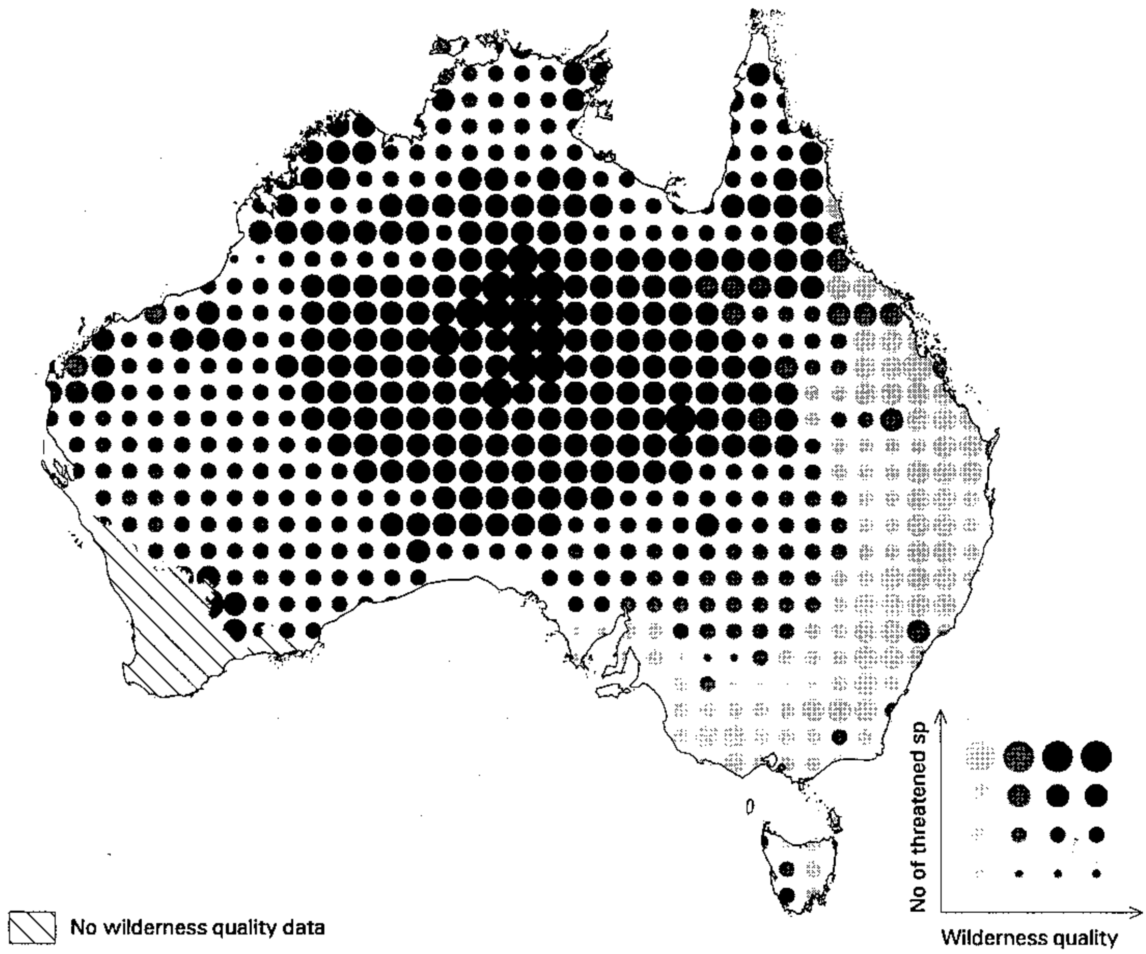


Figure 3

Figure 3: The Relationship Between Wilderness Quality and the Number of Threatened Reptile Species

Data Sources

Wilderness Quality: National Wilderness Inventory, Environment Australia

Threatened Mammals: State of the Environment Advisory Council, 1996

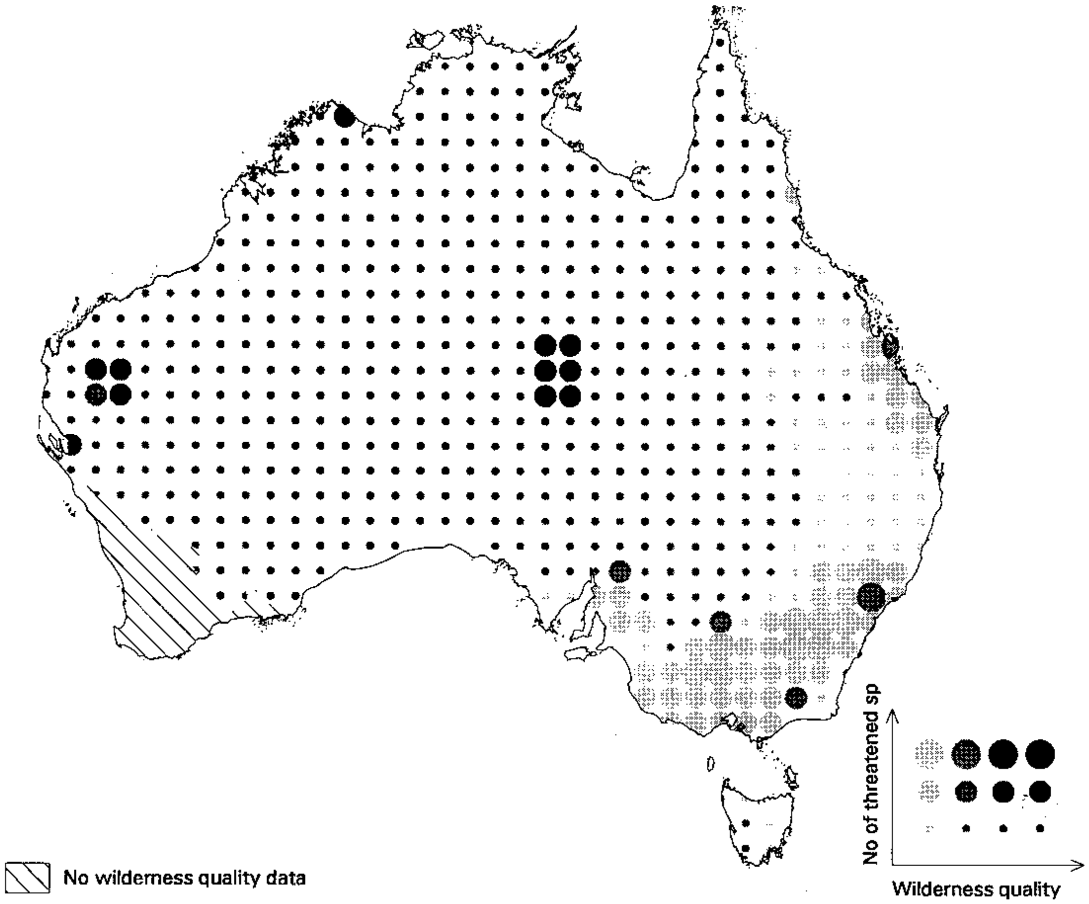


Figure 4

Figure 4: The Relationship Between Wilderness Quality and the Number of Threatened Amphibian Species

Data Sources

Wilderness Quality: National Wilderness Inventory,
Environment Australia

Threatened Mammals: State of the Environment
Advisory Council, 1996

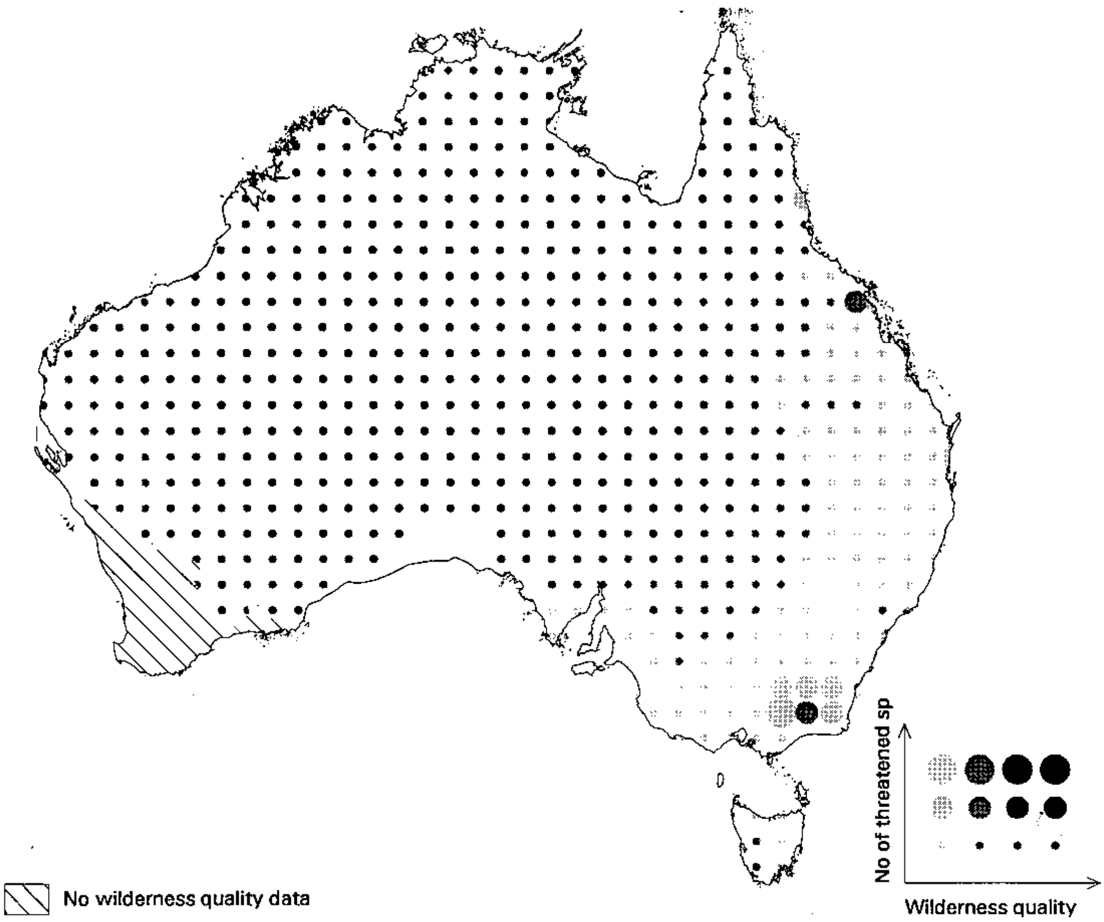


Figure 5

Figure 5: The Relationship Between Wilderness Quality and the Number of Threatened Bird Species

Data Sources

Wilderness Quality: National Wilderness Inventory, Environment Australia

Threatened Mammals: State of the Environment Advisory Council, 1996

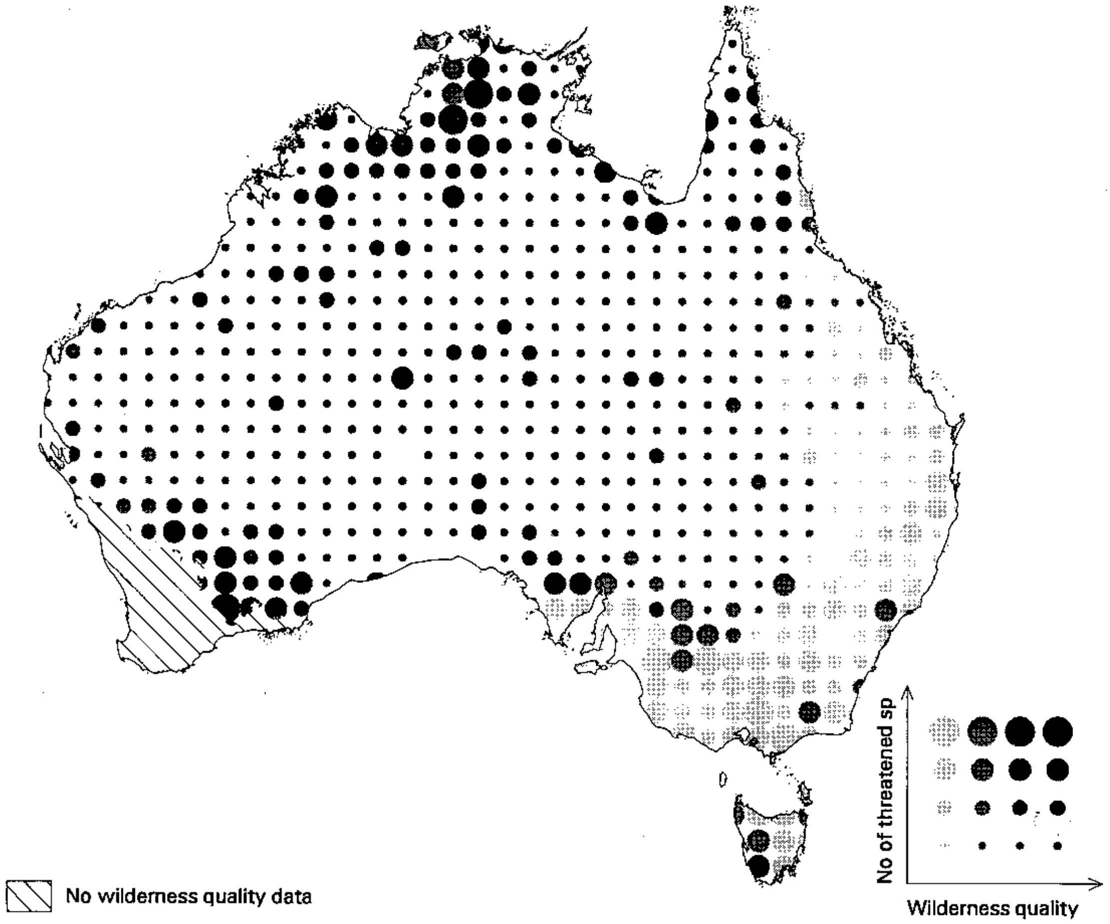


Figure 6

Figure 6: Plots of the number of threatened plant species and wilderness quality in Australia (data from ASOE 1996).

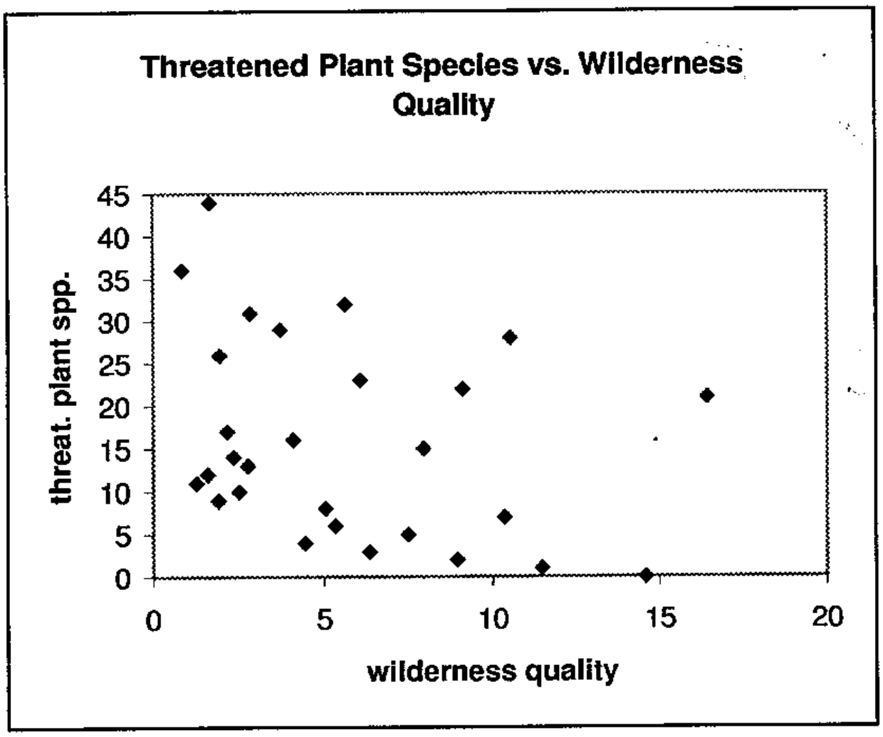


Figure 7

Figure 7: Plots of the number of threatened mammal species and wilderness quality in Australia (data from ASOE 1996).

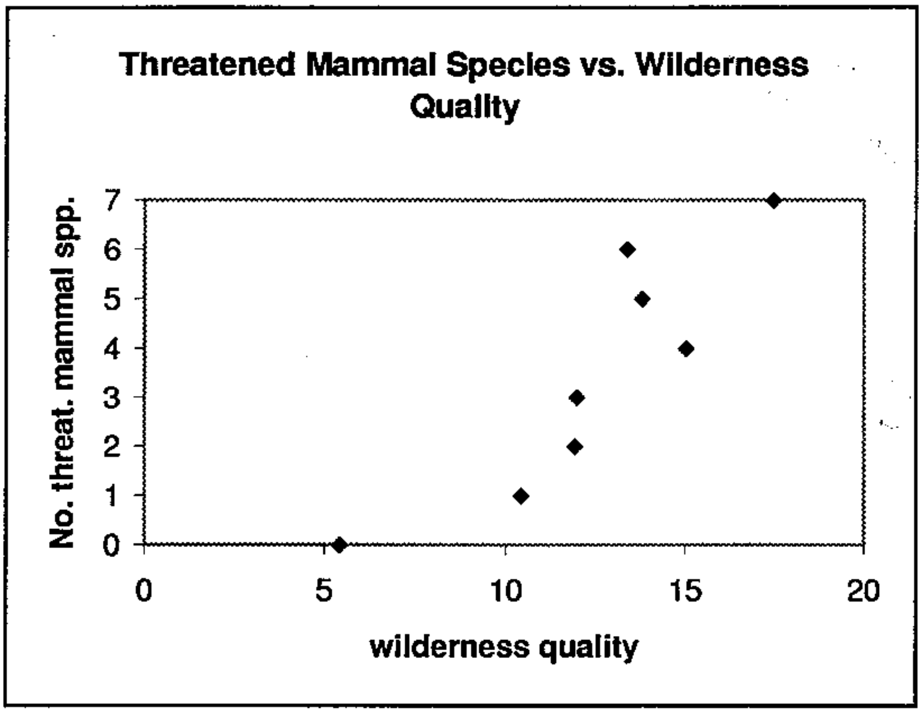


Figure 8

Figure 8: Plots of the number of threatened reptile species and wilderness quality in Australia (data from ASOE 1996).

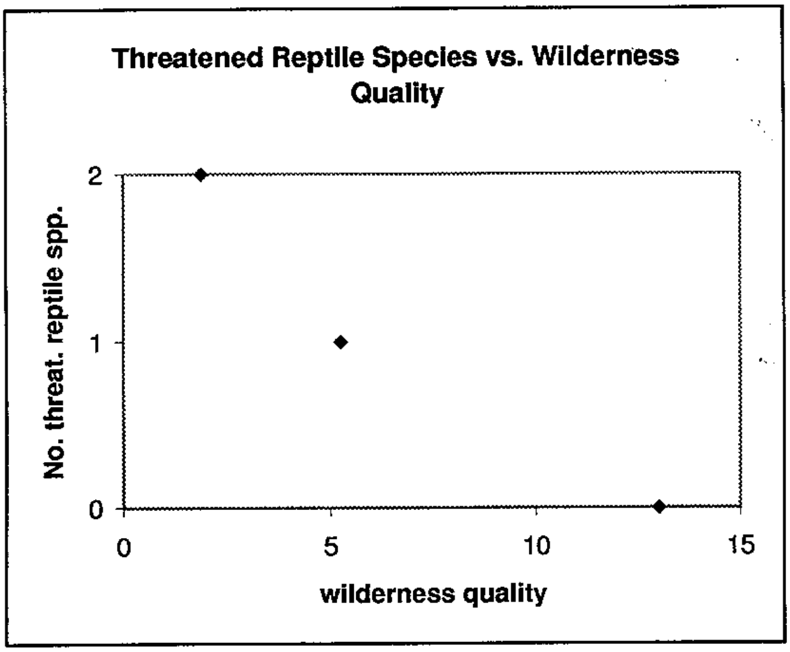


Figure 9

Figure 9: Plots of the number of threatened amphibian species and wilderness quality in Australia (data from ASOE 1996).

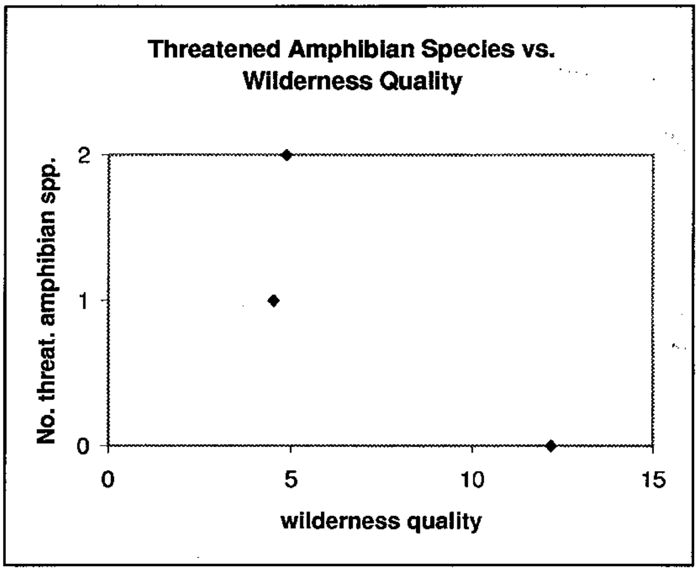


Figure 10

Figure 10: Plots of the number of threatened bird species and wilderness quality in Australia (data from ASOE 1996).

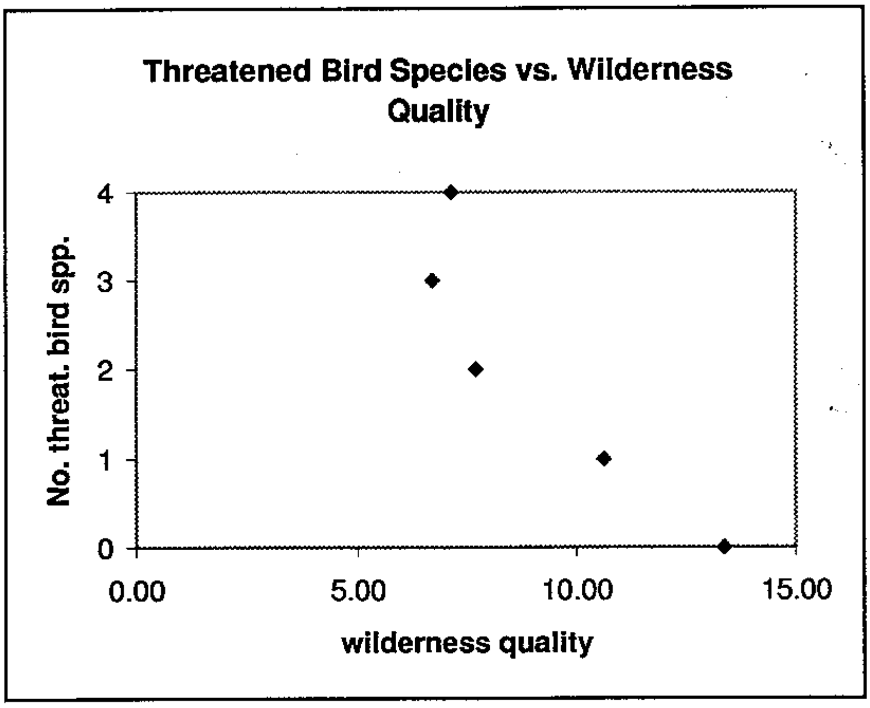


Figure 11

Figure 11: Plots of the number of threatened vertebrate species and wilderness quality in Australia (data from ASOE 1996).

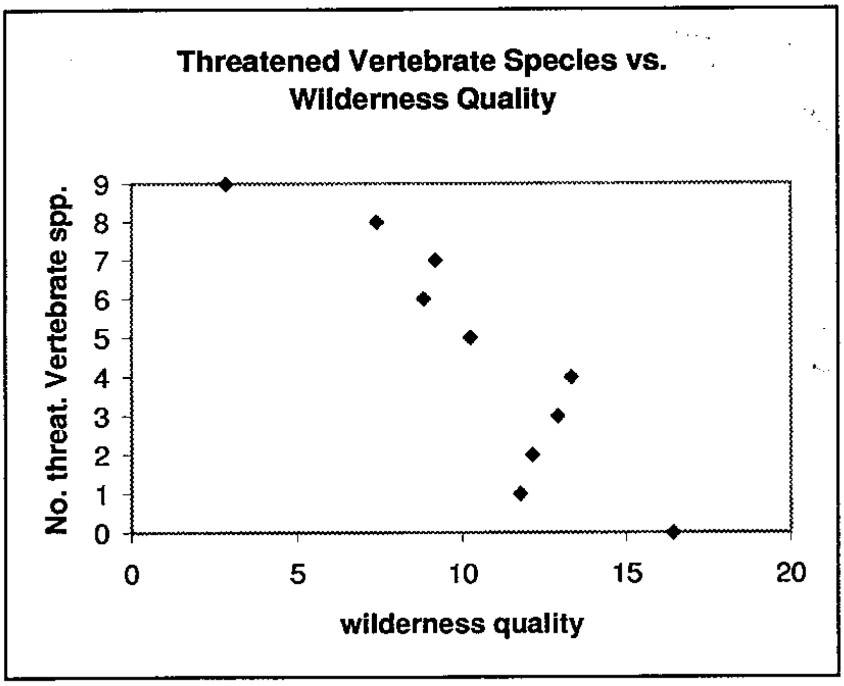


Figure 12

Figure 12: Plots of the number of threatened species and wilderness quality in Australia (data from ASOE 1996).

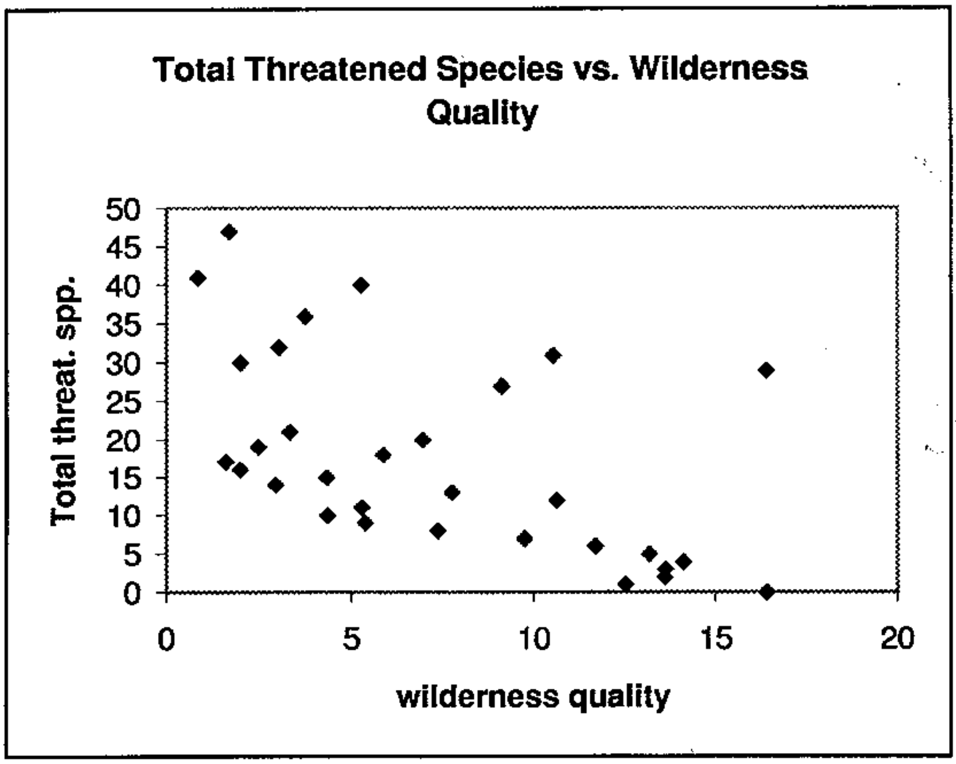


Figure 13

Figure 13: Mean Total Wilderness Quality of IBRA Regions

Data Sources

Wilderness Quality: National Wilderness Inventory Environment Australia

IBRA Regions: Reserve Systems Section Biodiversity Group Environment Australia

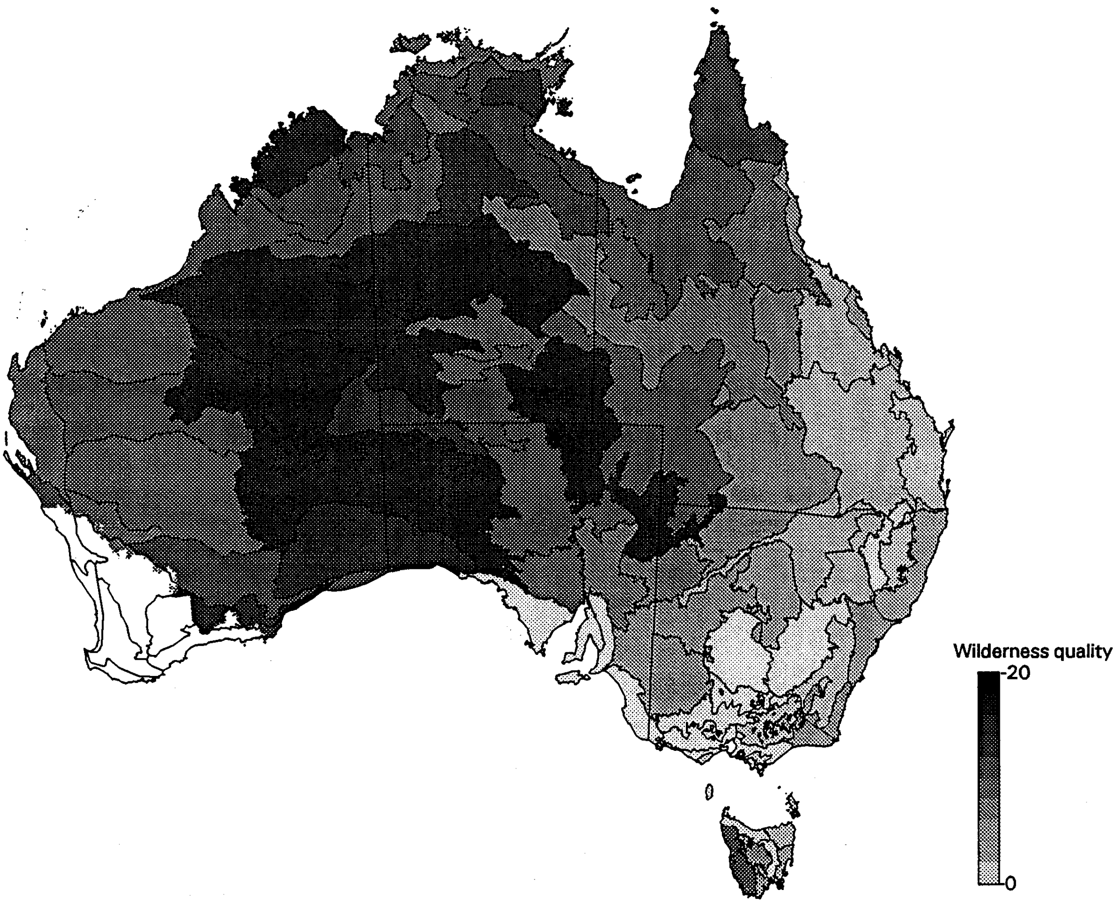


Figure 14

Figure 14: Digital Terrain Classification of Australia using Elevation Percentile and Relief

Data Source

Elevation: Hutchinson and Dowling (1991)

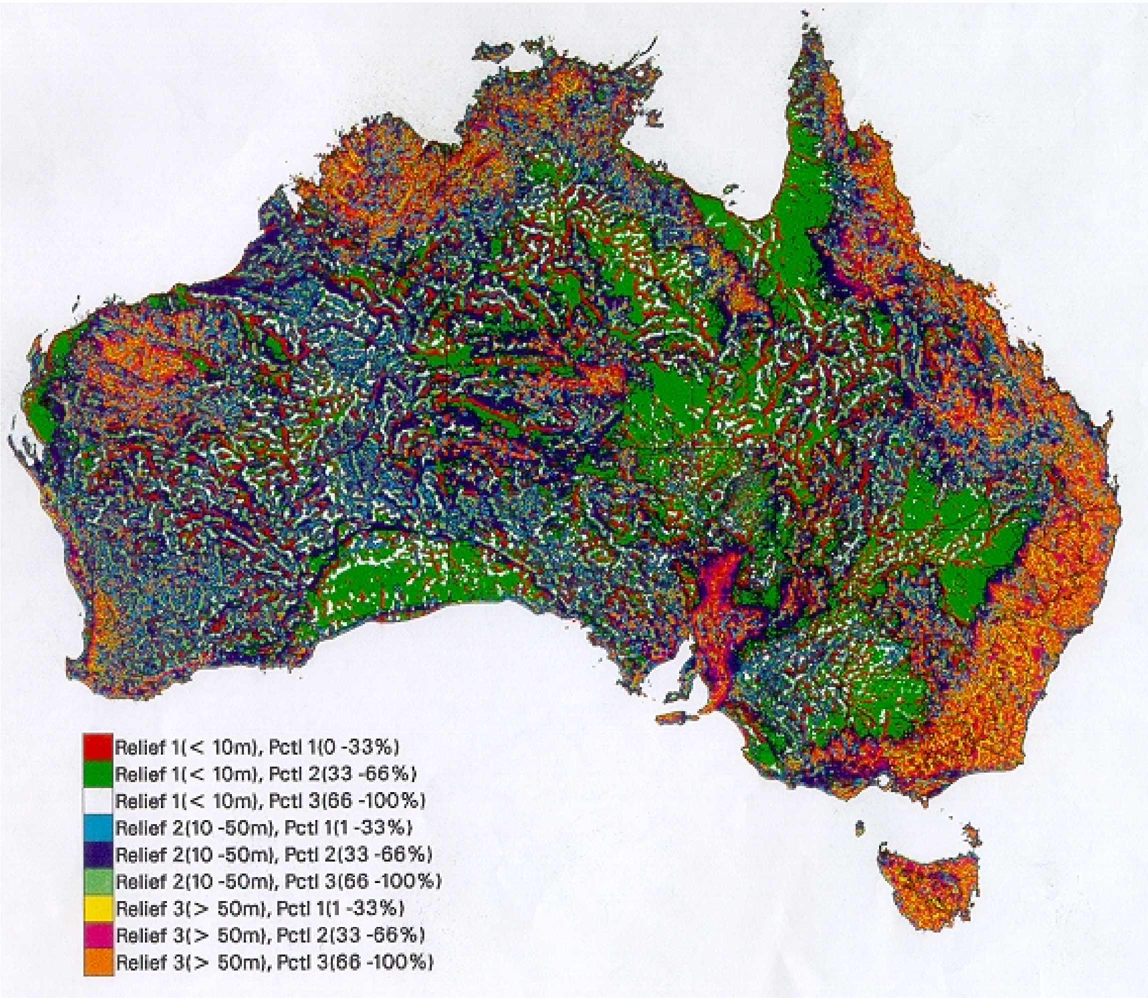


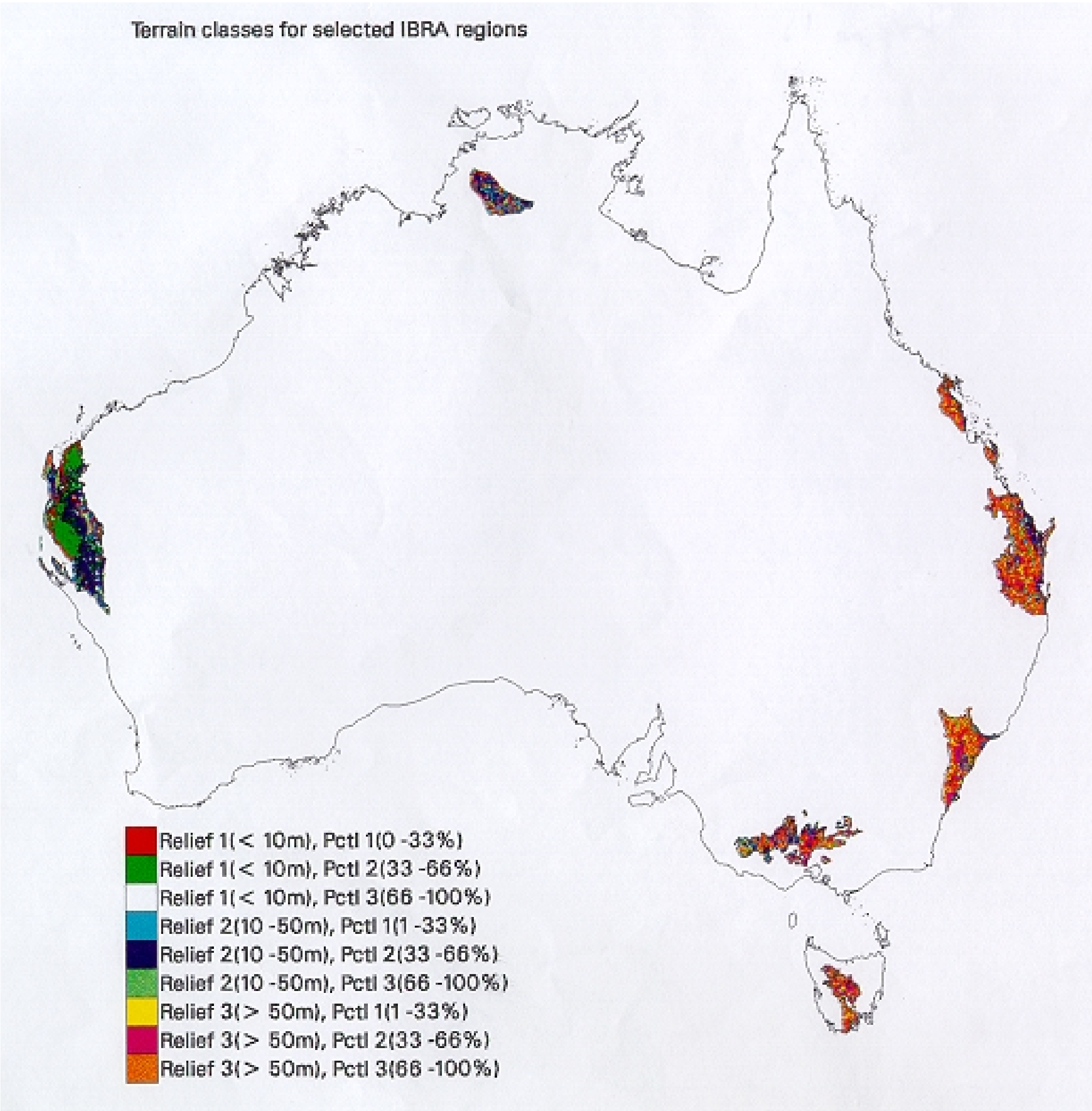
Figure 15

Figure 15: Digital Terrain Classification using Elevation Percentile and Relief for Eight IBRA Regions

Data Source

Elevation: Hutchinson and Dowling (1991)

IBRA Regions: Reserve Systems Section Biodiversity Group Environment Australia



Appendix

The decline and extinction of Australian mammals

The settlement and development of Australian landscapes has coincided with the extinction of 40 terrestrial vertebrate species (20 bird species and 20 mammal species, State of the Environment Advisory Council 1996). Rates of extinction may have slowed in recent years, but there is a strong possibility that further species will become extinct. Currently, of the 1630 extant species of terrestrial and freshwater vertebrates, 16% (280 spp.) are listed by various organisation as endangered, rare or threatened (Recher and Lim 1990). Some species naturally occur as small isolated populations, however the vast majority of declines or extinctions can be attributed to human impacts. The decline and extinction of birds and mammals has been linked with a number of ecological attributes including: ground-dwelling habits, ground or hollow nesting habitats, seed and terrestrial invertebrate-eating, being a large carnivore, occurrence in western grassland and shrubland for birds, herbivory, fungivory, omnivory, resident in arid or semi-arid regions, and ground dwelling habits for mammals (Lunney *et al.* 1997).

Mammal species have experienced the greatest rates of extinction and decline on the Australian continent as documented by various authors (Burbidge and McKenzie 1989, Recher and Lim 1990, Morton 1990, Dickman *et al.* 1993, Lunney and Leary 1988, Short and Smith 1994). The distribution of mammal species decline and extinction has been uneven across the continent (Short and Smith 1994). Intuition would suggest that the greatest rates of extinction would occur in the most altered ecosystems. The most dramatic land cover changes have occurred in south-east and south-west Australia, reflecting the extensive and intensive impact of agricultural and pastoral activities. For example, the Wheatbelt and Darling districts of Western Australia have been cleared for agriculture and pastoralism, removing 65% and 35% of vegetation cover, respectively (Burbidge and McKenzie 1989).

An indication of the effect of European settlement in these areas is well demonstrated by the records of a collecting expedition led by Krefft in the 1850s (Krefft 1862). Krefft established a camp near the Junction of the Murray and Darling River and catalogued 28 species of rodents, marsupials, and monotremes to exist in the region. A comparison of Krefft's data with current distribution maps from Strahan (1983), shows at least 14 species (50%) have ceased to occur in the Lower Murray-Darling region. Many of these species have declined dramatically in range since Krefft's expedition. For example, the numbat (*Myrmecobius fasciatus*) is now only found in south-western Western Australia and the Greater Stick Nest Rat (*Leporillus conditor*) is only found on offshore islands. Three species recorded by Krefft are now presumed extinct (Eastern Hare Wallaby, *Lagorchestes leporoides*; Pig-footed Bandicoot, *Chaeropus occidentalis*; and Lesser Stick Nest Rat, *Leporillus apicalis*). Two species that were considered common by Krefft in the 1850s are now extinct (*Lagorchestes leproides*, Eastern Hare Wallaby), or isolated to offshore islands 3,000 kilometres to the west (*Perameles bougainville*, Western Barred Bandicoot). Krefft (1862) observed that the decline of several species during the 1850s coincided with increasing pastoral activity.

Historical research has revealed similar species loss on the western slopes of the Great Dividing Range (Jarman and Johnson 1977). Their studies found several species to have undergone dramatic declines in range and abundance since European settlement including, Greater Bilby (*Macrotis lagotis*), Rock Wallaby (*Petrogale pencillata*), Bridled Nail-tailed Wallaby (*Onychogalea fraenata*) and Rat Kangaroos, Potoroos, (*Aepyprymnus*, *Bettongia*). Both *Marcotis lagotis* and *Onychogalea fraenata* are now extinct from New South Wales. Jarman and Johnson (1977) associated local extinctions and declines of native marsupials with the removal of ground cover by rabbits and fox predation.

Although, the south-eastern and south-western areas of Australia have undergone the greatest environmental modifications, the highest rates of extinction of mammals have occurred in the arid zone (the semi-arid/arid zone is henceforth referred to as the arid zone) (Burbidge and McKenzie 1989, Lunney *et al.* 1997). Sixty percent (23 spp.) of Australia's endangered or extinct terrestrial vertebrates were/are found in the arid zone (CONCOM 1988 cited in Morton 1990), while the arid zone species constitute only 38% of the terrestrial vertebrate fauna.

Mammals weighing between 35-5500g (the Critical Weight Range) have been postulated to be particularly sensitive to extinction and population decline (Burbidge and McKenzie 1989). Surveys of West Australian fauna have found that all the mammal species that have declined or become extinct on the mainland are non-flying and lie within the Critical Weight Range (CWR). However, not all of the 59 species found within the CWR in WA have declined; 17 species (29%) are stable, while 25 spp. (42%) have declined, and 17 spp. (29%) have become extinct.

Similarly, Dickman *et al.* (1993) reviewed the native mammals of the Western Division of NSW recorded since European settlement in 1788 and found that 34 species lay within the CWR, accounting for 48% of mammals in the region. The extinction rate of CWR species was high with 68% (23 spp.) locally extinct and a further 15% were considered endangered. This compares with only four non-CWR species presumed extinct.

Explanations for mammal decline and extinction in the arid zone

Various authors have explored the reason for the observed declines in mammal diversity in the arid zone (Burbidge and McKenzie 1989, Dickman *et al.* 1993, Recher and Lim 1990, Short and Smith 1994). A number of factors have been suggested as, including the impacts of introduced predators, introduced herbivores (rabbits and stock), disease, hunting, clearing and fragmentation, changes in vegetation structure and composition, changing fire regimes, predation and competition from increased abundance of native fauna, and drought and climate change (Recher and Lim 1990, Morton 1990).

The absence of Aboriginal burning in the arid zone has been attributed to the decline of some mammal species (e.g. *Lagorchestes hirsutus*, mala, Bolton and Latz 1978). Aboriginal burning was thought to create a mosaic of vegetation patterns required by certain mammal species. Short and Turner (1994) investigated the relationships between vegetation mosaics and several species isolated to offshore islands. They concluded that the species persisted on these islands, after becoming extinct or declining on the mainland, due to the absence of introduced predators and not the influence of vegetation mosaics. Unfortunately, assessing the impact of Aboriginal burning on the environment is "conjectural" due to our lack of accurate information on Aboriginal use of fire (Gill 1977, Caughley and Gunn 1966).

Dickman *et al.* 1993 also attributed predation by an introduced species, the feral cat, as a major cause of early mammal extinctions. However, they consider more recent extinctions to be linked to a number of causes including predation by cats and foxes, competition and habitat degradation by rabbits, stock and other introduced herbivores, clearing of trees, changes in fire regimes, and human persecution. It would appear that the majority of extinctions are associated with a number of causes and there is no single explanation. Burbidge and McKenzie (1989) theorise that European settlement has resulted in an "emulated" increase in aridity for many native species in the arid zone. The diversion of environment resources into crop and stock production has fragmented habitats and altered the vegetation, litter fauna, nutrient cycles, evaporation regimes. This reduction in available production has caused CWR mammals to suffer, in particular, because of their limited mobility and high daily metabolic requirements. Predation by introduced species was considered to compound the tenuous position of the CWR mammals.

Morton (1990) expanded on this model by incorporating the dynamics of landscape and climate in the arid zone. Morton (1990) argued that arid and semi-arid fauna depend on high-quality "oases", which are widely dispersed through the landscape, to survive drought periods. Many species relied on these patches of habitat to maintain core populations and permit the expansion into areas of marginal habitat during benign climatic conditions. Thus, CWR species were "inherently prone to the disturbance of their precious patches of habitat" due to their physiological and morphological constraints. Introduced herbivores then arrived in these landscapes, often their initial densities were far in excess of sustainable populations. The rabbits and stock altered the vegetation composition and structure, and most importantly damaged the high quality refuge areas the mammals relied upon. The refuge areas were often the most productive land for grazing animals, thus it was the first areas occupied by pastoralists. The pastoralists particularly relied on the "oases" during drought periods. These were the last areas to support stock once the drought hit, and the first areas to regenerate feed after the drought. This caused further damage to the refuges at a time when mammal populations were particularly vulnerable. The already perilous situation of these mammals was exacerbated by the introduction of exotic predators, who followed the spread of rabbits, and decline in Aboriginal burning patterns which was thought to create vegetation mosaics to the benefit of mammal species.

The multiple-cause models developed by Burbidge and McKenzie (1989) and Morton (1990) were applied to the Western Division of NSW by Dickman *et al.* 1993. They found the models successfully summarized their observations of mammal decline and extinction, although there were several discrepancies. They concluded that several extinctions occurred prior to significant land-use changes (pre-1850) and found that rock-pile mammals were not exempt from declining trends as predicted by Burbidge and McKenzie. They also found 4 non-CWR species presumed to become extinct and at least 5 non-CWR species in decline.

Local extinctions in mesic areas

From the perspective of an individual ecosystem, the distinction between local versus global extinction is meaningless. Once a species has disappeared from an ecosystem at one location, it is irrelevant to the functioning of that ecosystem that the species may continue to exist in other parts of Australia. Similarly, depleted populations may be ineffective in playing their roles within ecosystem processes (Lindenmayer and Gibbons 1997). For example, the functional extinction of mycophagous mammals (e.g. bandicoots and potoroos), species thought to play a role in maintaining ectomycorrhizal relationships between fungi and plants (Claridge *et al.* 1992), may influence key aspects of forest dynamics such as plant growth patterns and recovery of plant communities after fire. Many ecosystems in the arid zone have suffered local extinctions and depleted populations as we have outlined above. However, it is often overlooked that many ecosystems within mesic areas of Australian have also suffered significant changes in species composition and ecosystem function due to local extinctions and depleted populations.

Lunney and Leary (1988) provide an example of this situation in a coastal, mesic area in southern NSW, demonstrating that significant changes in mammal communities and ecosystem function have also occurred in moister areas of Australia of lower extinction rates. They found all native species populations to have declined and there has been local extinction of six species, the eastern quoll (*Dasyurus viverrinus*), a rat kangaroo (probably *Bettongia gaimardi*), two pademelons (*Macropus parma* and *Thylogale thetis*), the wallaroo (*Macropus robustus*), and brush-tailed phascogale (*Phascogale tapoatafa*). A further four species that were once common are now rare and threatened with extinction, koala (*Phascolartes cinerus*), southern brown bandicoot (*Isodon obesulus*), spotted-tailed quoll (*Dasyurus maculatus*), and little red flying fox (*Pteropus scapulatus*). These species are typical of depleted populations and may now be functionally extinct within the ecosystem. Flying fox colonies were reported to have contained 1000s of individuals with colonies “at least half a square mile” in size. Few flying foxes are now recorded and most of the old colony sites have been abandoned.

Lunney and Leary (1988) attribute these changes in mammal distribution to the effects of habitat clearance, and the introduction of hares, rabbits, and foxes. They propose that much of the productive land which is currently utilised for agriculture once supported high density core populations of native species. This mirrors impacts in arid Australia where the key, high quality habitats have been damaged or entirely alienated (see above). The damage to these areas causes subsequent declines throughout the marginal ranges of these arid and mesic species, due to the lack of a healthy “source” population.