

THE UNIVERSITY OF ADELAIDE

---

Landscape scale measurement and monitoring of  
biodiversity in the Australian rangelands

---

Thesis presented for the degree of

Doctorate of Philosophy

Kenneth Clarke

B. Env. Mgt. (Hons), University of Adelaide

November 2008

Faculty of Sciences, Discipline of Soil and Land Systems

## Chapter 1: Introduction

### 1.1 Motivation for the research

As we become more environmentally aware, and as economic values are placed on natural ecosystems (Costanza *et al.* 1997), managers have begun to appreciate the potential cost of allowing further degradation of our natural systems. In the Australian rangelands this has resulted in an increased desire to monitor and manage biodiversity (Smyth *et al.* 2004). Currently however, there is no suitable method for monitoring biodiversity in the extensive Australian rangelands. Hence, there is a clear need for a tool or tools to fill this gap, to allow monitoring of temporal and spatial change in biodiversity and therefore inform the prioritisation of conservation goals and assist in sustainable pastoral management. But how to best measure biodiversity?

The term biodiversity is so all-encompassing that direct measurement is not possible, and it is necessary to measure other features which vary with biodiversity: surrogates. In the language of Sarkar (2002), a true-surrogate represents biodiversity directly, while an estimator-surrogate represents a true-surrogate, which in turn represents biodiversity.

Sarkar (2002) argued that species richness was one of the few suitable true-surrogates for biodiversity because firstly, species are a well defined and understood biological category, and secondly, species richness is measurable. However, measuring total species richness over the entire rangelands is impractical and we therefore desire a surrogate of total species richness, or an estimator-surrogate for biodiversity.

Thus the estimator-surrogate we seek must co-vary with total species-richness, and while the evidence for cross-taxon surrogates is equivocal, there is significant supporting evidence. At broad scales the species-richness of many phylogenetic groups is determined by climatic variables: trees (Currie and Paquin 1987; O'Brien 1993; O'Brien 1998; O'Brien *et al.* 2000); vascular plants (Venevsky and Venevskaia 2005); mammals (Badgley and Fox 2000); butterflies (Hawkins and Porter 2003; Hawkins and Porter 2003); and bird species (Hawkins *et al.* 2003). The species-richness of each of these groups varies in response to similar environmental variables, and we hypothesise that the species-richness

of one of these groups, woody plants, is an estimator-surrogate for biodiversity. This use of cross-taxon biodiversity surrogates is supported by the meta-analysis of 27 biodiversity studies by Rodrigues and Brooks (2007).

While currently there are no methods of measuring or monitoring biodiversity in extensive areas such as the Australian rangelands, historically traditional field-based methods such as quadrat surveys have collected flora and fauna species data. However, it would be prohibitively expensive and time consuming to use field-based surveys to map the majority of variation in species across the Australian rangelands once, let alone regularly as required by a monitoring program.

Therefore, field surveys are unsuitable for measuring and/or monitoring biodiversity in the Australian rangelands for several reasons. Field surveys are not capable of collecting data at a similar scale to the broad extent of the rangelands, the consistency of their data may vary over time, they are relatively costly, and consequently are repeated irregularly. An alternative to these ground based measures is the use of satellite remote sensing, which collects data that is spatially comprehensive, calibrated and therefore consistent, relatively inexpensive, has a high temporal frequency, and is biologically relevant.

Remotely sensed data which covers the Australian rangelands is collected on a regular basis by sensors onboard several satellites, and can be obtained for no or minimal cost. Some individual sensors have been collecting data for years, while some series of sensors have been collecting data for over three decades. Consequently there are now substantial archives of remotely sensed data covering the Australian rangelands.

There is a need for a method of measuring and monitoring biodiversity in the Australian rangelands which is not addressed by current field-based methods. Remotely sensed data are biologically relevant, spatially-extensive, calibrated and therefore temporally consistent. Furthermore, extensive archives of low-cost remotely sensed data exist over the Australian rangelands. Therefore, there is a clear need to examine the potential for remote sensing to improve biodiversity measurement and monitoring in the Australian rangelands.

## 1.2 Thesis topic and structure

This research has the overarching goal of developing better tools for the monitoring of biodiversity in the rangelands of Australia. Existing vegetation quadrat survey data and remotely sensed imagery were recognised as rich sources of biologically relevant data. The first specific aim of the thesis was to review the potential and limitations of the vegetation quadrat survey data. This review informed the methods developed to address the second specific aim: to derive an ecologically and mathematically sound biodiversity index from the vegetation quadrat survey data. The final aim of the thesis was to derive indices of biodiversity stress from the remotely sensed data. The remotely sensed indices of biodiversity stress were evaluated against the vegetation quadrat survey data index of biodiversity.

This thesis is structured with six chapters, some of which were written for publication as peer-reviewed journal articles. The chapters written as articles are included as submitted, which necessitates some repetition of material presented in the introduction and review chapters (Chapters 1 and 2 respectively). Additionally, these articles necessarily use the plural 'we', due to the contribution of co-authors. To ensure consistency this convention has been followed in the remainder of the thesis.

The thesis begins with a general introduction and brief overview of the need for and motivation behind this research, an outline of the structure of the thesis, and finally an introduction to the study area (Chapter 1). Next, Chapter 2 begins with a brief explanation of key terms and concepts which will be used throughout the rest of the thesis. This is followed by a review of the causes of and pressures on biodiversity at broad scales, and of current remote sensing methods of measuring and monitoring biodiversity. Finally, Chapter 2 ends with an outline of two potential surrogates of biodiversity stress which could conceivably be generated at little cost from a combination of satellite and climate data.

Prior to attempting to develop a surrogate of biodiversity from the vegetation quadrat survey data, the assumption that these data could record species richness was tested. Chapter 3 presents the results of this analysis, submitted to *Applied Vegetation Science* as Clarke, K., Lewis, M., and Ostendorf, B., 'False negative errors in vegetation surveys', and

identifies an intrinsic limitation of the vegetation quadrat survey data: false-negative errors render it impossible to estimate species richness at the quadrat scale.

With the limitations identified, Chapter 4 develops a method for extracting an index of biodiversity from the vegetation quadrat survey data. This article, submitted to *Ecological Indicators*, as Clarke, K., Lewis, M., and Ostendorf, B, ‘Additive partitioning of rarefaction curves: removing the influence of sampling on species-diversity in vegetation surveys’, combines rarefaction and additive partitioning methods to allow the extraction of an index of vegetation species diversity from the vegetation quadrat surveys, free from the influence of sampling effort.

In Chapter 5, two theoretical surrogates of biodiversity stress are developed from a combination of remotely sensed and climate data, and these surrogates are validated against the index of vegetation species diversity developed in Chapter 4. Surrogate 1 is based on the hypothesis that the difference between net primary production (NPP) and expected primary productivity (EPP) is indicative of biodiversity stress; Surrogate 2 is based on the hypothesis that overgrazing decreases average NPP and rainfall use efficiency (RUE), and increases variation in NPP and RUE.

Chapter 6 reviews the findings of the research and the extent to which the aims have been met. Key contributions to knowledge are identified, the limitations to generalisation are clarified, and the wider implications are discussed. The thesis ends with a summary of important areas for future research.

## **1.3 Study area**

### **1.3.1 Location and infrastructure**

The study was conducted in central Australia in a region stretching from the top of Spencer Gulf in South Australia to the Northern Territory border (Figure 1). The region contains several small towns, ranging in population from ~80 to 4000 (Table 1), and is otherwise sparsely populated by pastoralists. The smallest towns, Lyndhurst, Marla, Marree and Oodnadatta are resource points on some of the more travelled tracks of the region. The town of Woomera services a defence rocket range, Coober Pedy is a centre for opal mining and arid tourism, and Roxby Downs services the Olympic Dam copper, uranium, gold and

silver mine. Apart from these towns and associated mines the region contains very little additional built infrastructure. A major sealed road runs along the western margin of the study area, and a sparse network of minor roads spreads throughout the rest of the study area, some sealed and some unsealed. Finally, the Dingo Fence, a pest-exclusion fence, bisects the study area, stretching east-west just north of Coober Pedy.

**Table 1. Population centres in study area.**

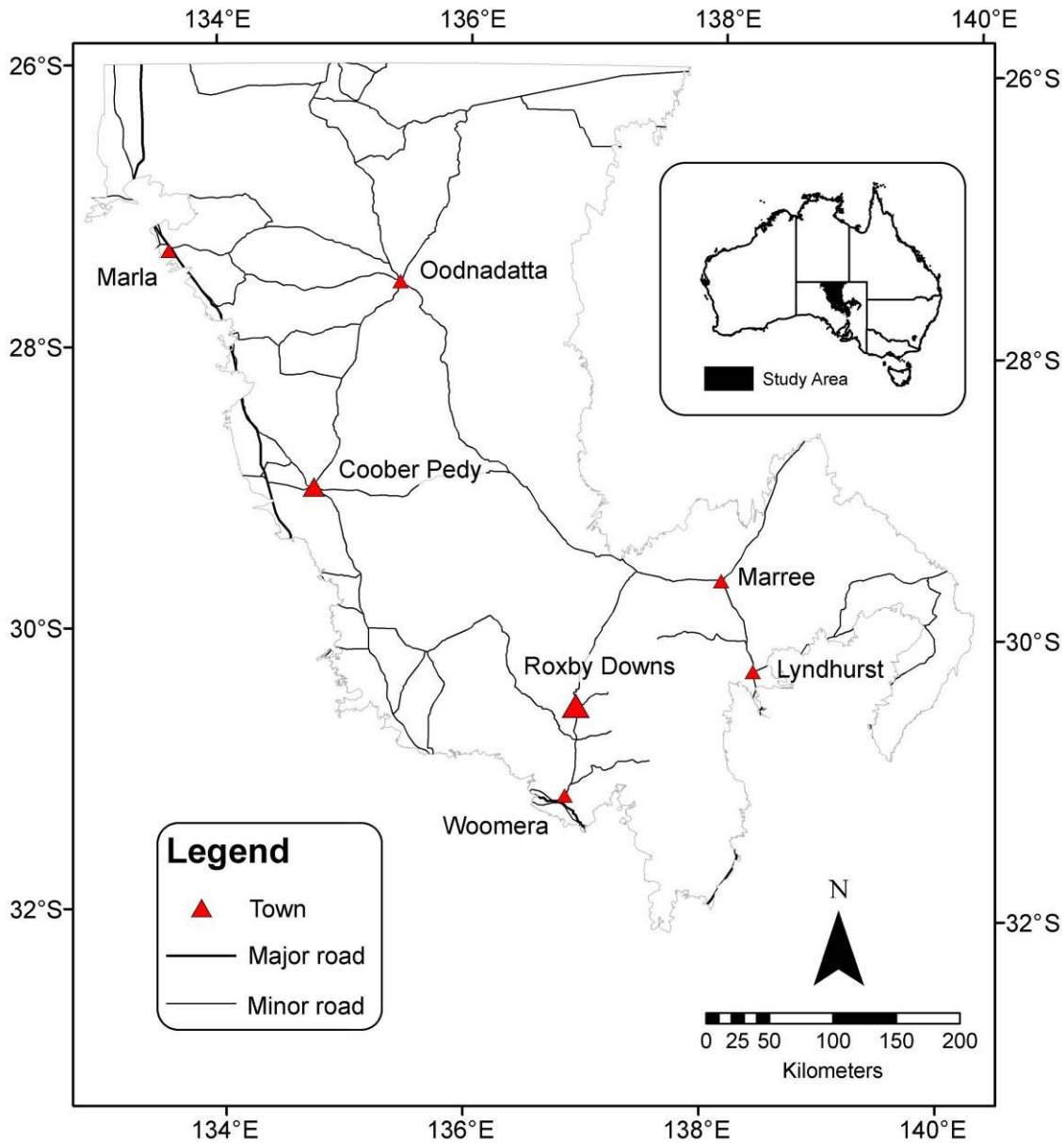
<b>Town</b>	<b>Population (approx)</b>
Coober Pedy	1916
Lyndhurst	<100
Marla	243
Marree	80
Oodnadatta	277
Roxby Downs	4000
Woomera	300

### 1.3.2 Physical geography and climate

The study area is large, approximately 210,000 km<sup>2</sup>, but contains little geographic variation. The majority of the area comprises flat or gently sloping plains, with some notable exceptions: a line of breakaways, or mesas, stretches along the western margin of the study area; the Simpson Desert, an area of extensive dune-fields covers the north east of the study; and the crescent shape of Lake Torrens, a large ephemeral salt lake dominates the south of the study area (Figure 2).

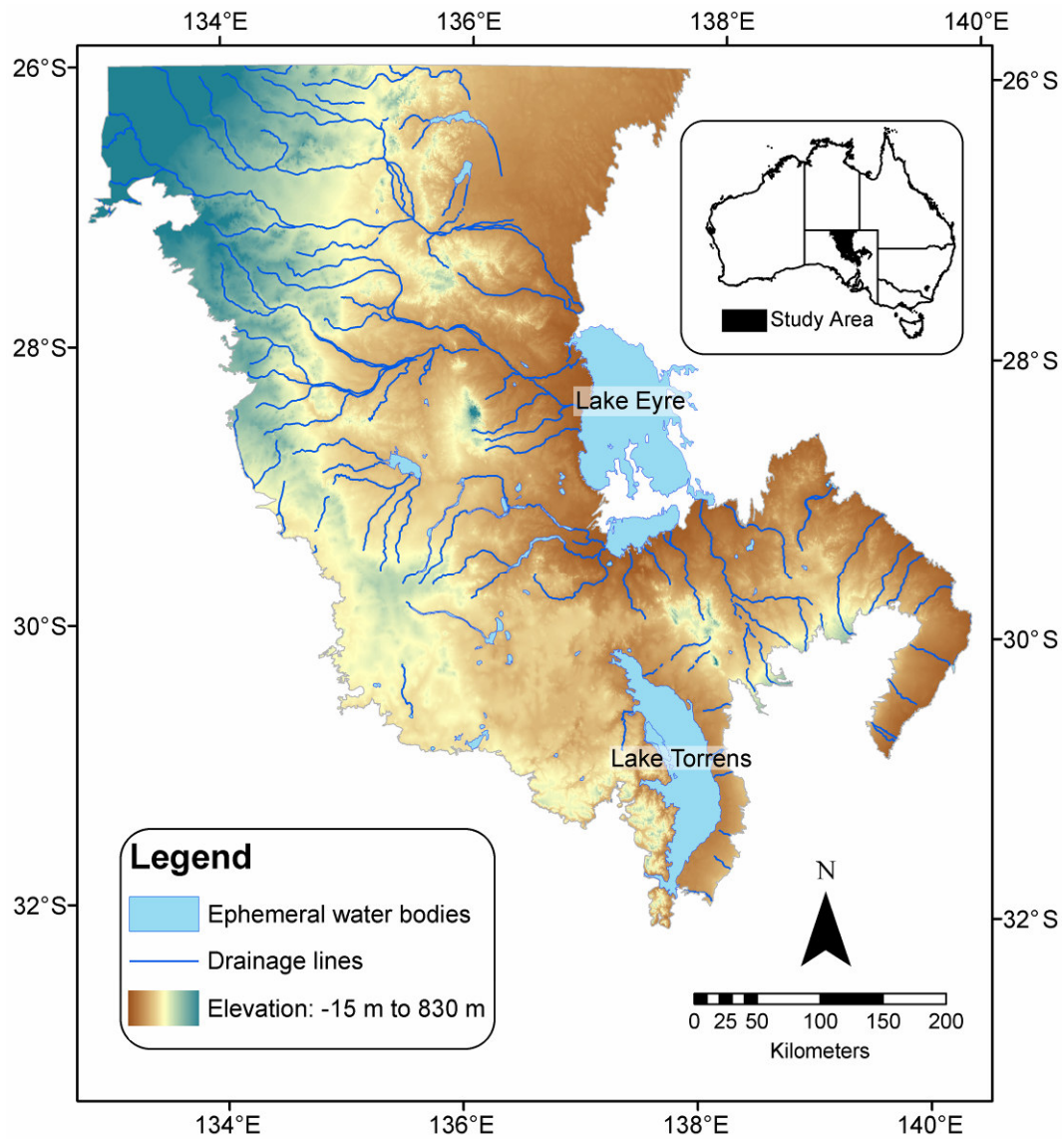
Several small wetlands and mound-springs occur around the middle north of the study area. The mound springs are fed by an artesian aquifer, and are probably the only permanent natural water sources in the study area. Due to their permanency these springs support a diverse range of vegetation, invertebrates, and small mammals.

The climate of the study area is uniformly dry and hot. Average annual rainfall across the area ranges from approximately 300 mm per annum in the south to 100 mm per annum in the north. However, this rainfall is highly variable, and no rain or many times the average may be received in a given month or year. The average and variation in monthly rainfall for Coober Pedy, near the middle of the study area, is presented in Figure 3. This temporal pattern of rainfall is typical of the entire study area.



**Figure 1. Study area location and built infrastructure.**

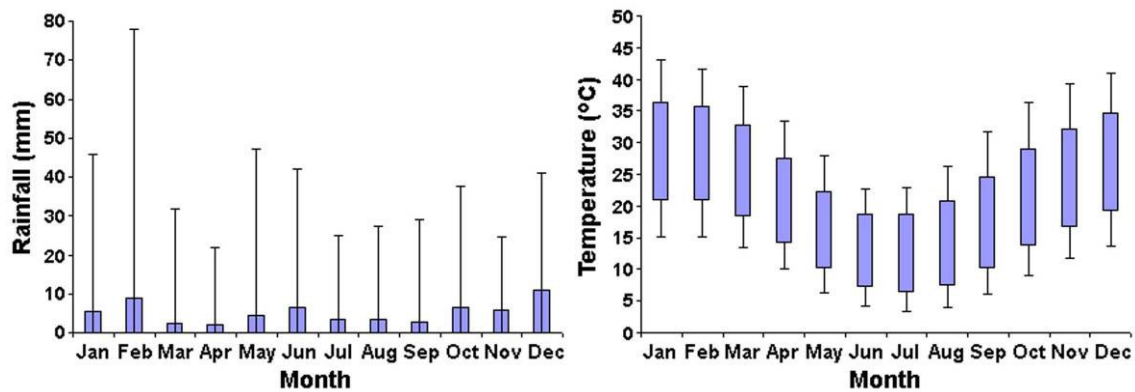
The average daily temperature for Coober Pedy is presented in Figure 3. The graph depicts the mean daily minimum and maximum temperatures (box) and the 1<sup>st</sup> decile minimum and 9<sup>th</sup> decile maximum temperatures (°C). The average maximum temperature in January and February is near 36 °C (96.8 °F), and daily maxima as high as 43 °C (109.4 °F) are reasonably common.



**Figure 2. Physical geography of the study area.**

When the study area experiences one of the large infrequent rainfall events, runoff flows into various salt lakes and wetlands. The majority of the region drains into Lake Eyre, just east of the middle of the study area (Figure 2). Those drainage lines which do not flow into Lake Eyre feed instead into other salt lakes and wetlands. The majority of the region's salt lakes and wetlands are ephemeral, due to the combination of low rainfall and high temperatures.



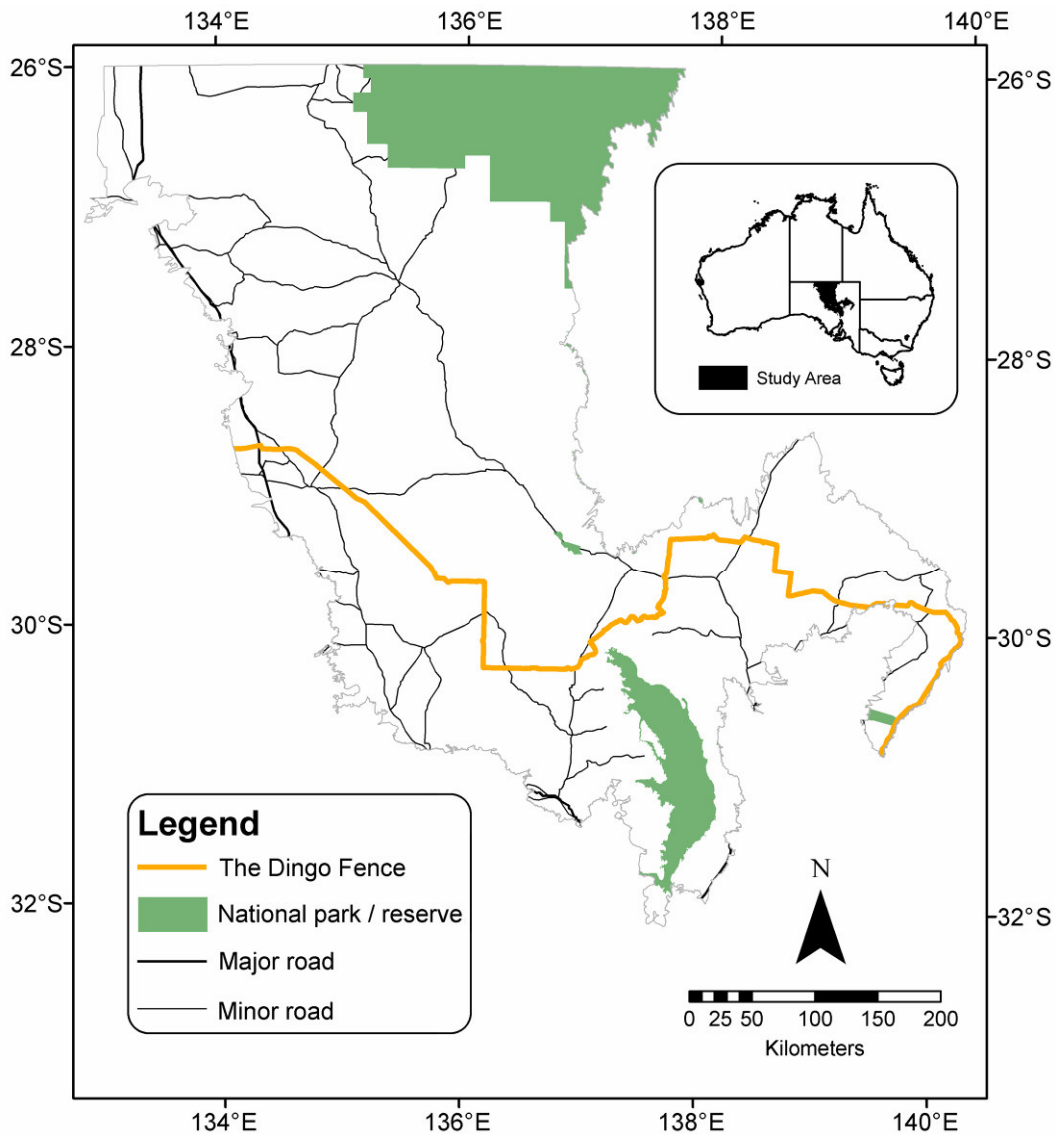


**Figure 3. Average and variability in rainfall (mm) and temperature (°C) for Coober Pedy, calculated from 70 years of climate records. In rainfall chart, the box represents the median rainfall, whiskers represent the 1<sup>st</sup> decile and 9<sup>th</sup> decile rainfall. In temperature chart, the box represents mean daily minimum and maximum temperatures, whiskers represent 1<sup>st</sup> decile daily minimum, and 9<sup>th</sup> decile daily maximum temperatures. Data courtesy of the Australian Bureau of Meteorology.**

### 1.3.3 Ecology and land use

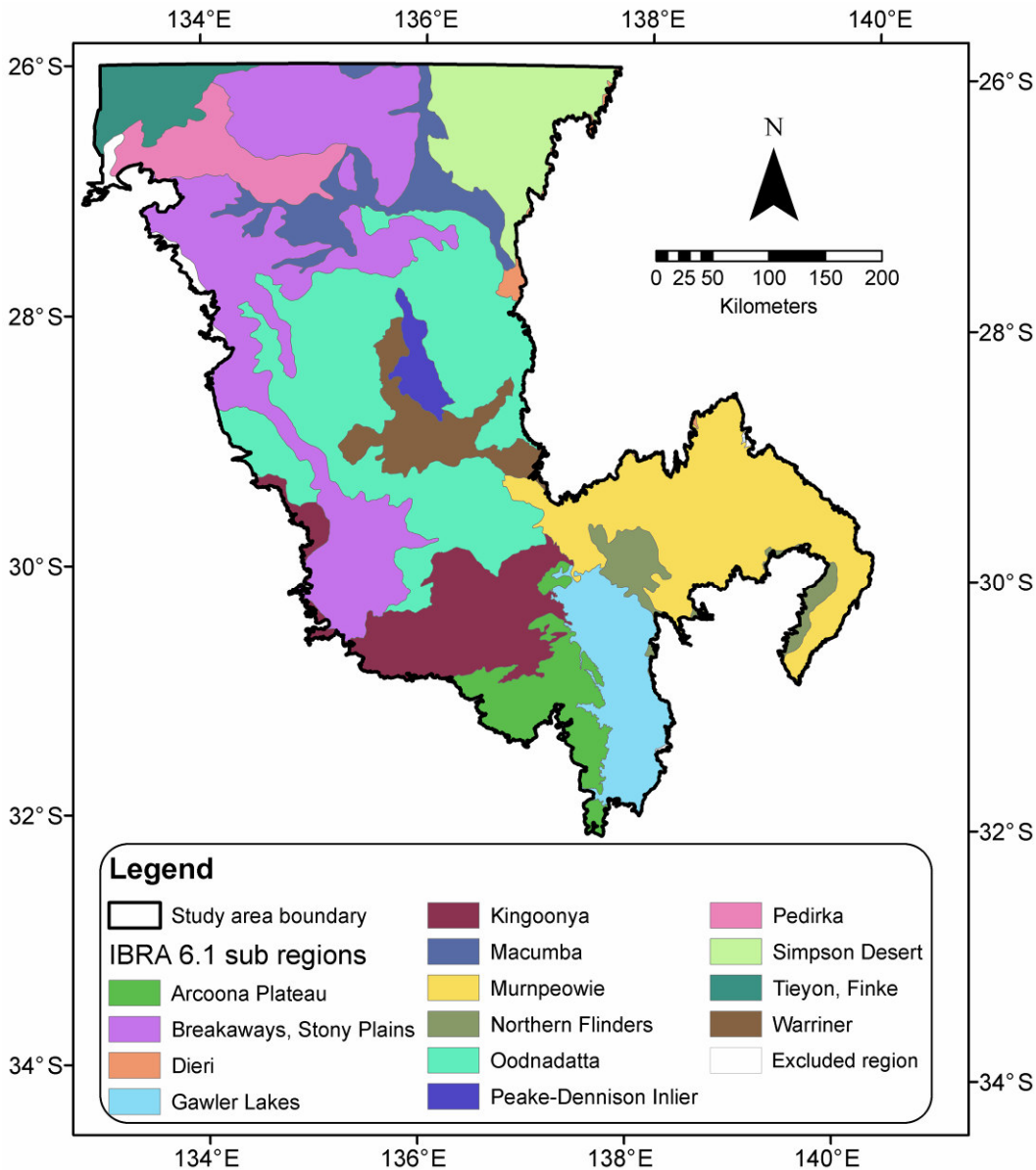
The dominant land use in the study area is grazing on large properties held under long-term pastoral lease, primarily by cattle north of the Dingo Fence, and sheep to the south. The north east of the study area is covered by the neighbouring Witjira National Park and Simpson Desert Regional Reserve, Lake Torrens is incorporated in the Lake Torrens National Park, and a small amount of the remaining area is covered by mining leases (Figure 4). Due to the lack of natural water sources, the proportion of the landscape accessible to domestic stock is largely determined by the location and frequency of artificial stock watering points.

The study area has been classified into areas of similar biological communities, as influenced by climatic and geographical conditions by the Interim Biogeographic Regionalisation of Australia 6.1 (IBRA). The study area IBRA 6.1 sub-regions are presented in Figure 5, and descriptions of the characteristics used to define each sub-region are presented in Appendix 1.



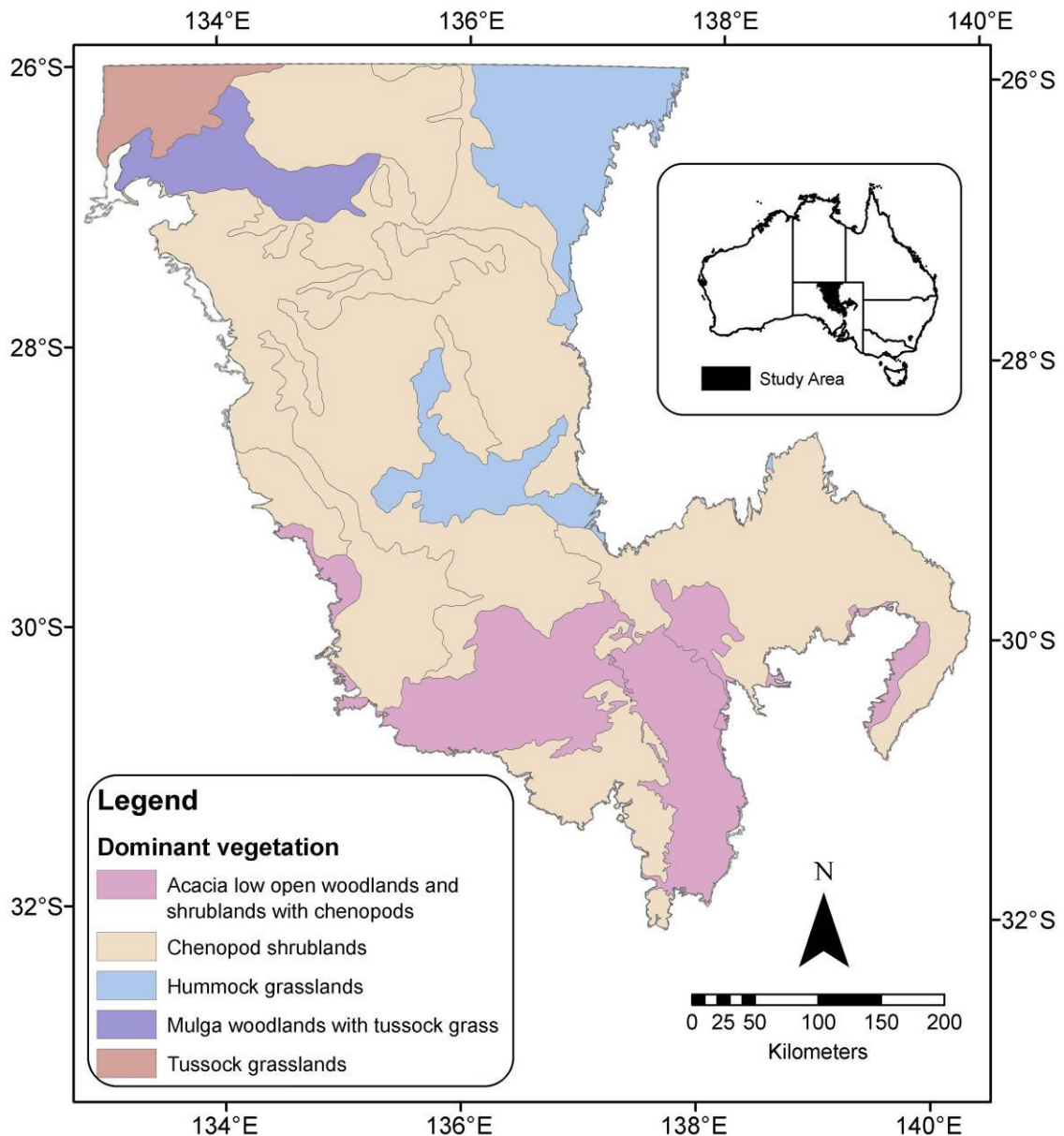
**Figure 4. Land use in study area.**

More specifically, the ecology of the region is strongly influenced by the infrequency and scarcity of water and high temperatures. The IBRA sub-regions in the study area are characterised by one of five dominant vegetation communities (Figure 6). In descending order of proportion of the study area covered these are chenopod shrublands (65.32%, see example in Figure 7), arid and semi-arid acacia low open woodland and shrublands with chenopods (15.71%, see example in Figure 8), hummock grasslands (11.26%, see example in Figure 9), Mulga (*Acacia aneura*) woodlands and tall shrublands with tussock grass (4.02%), and tussock grasslands (2.87%).



**Figure 5. IBRA sub-regions**

Additionally, there is a strong association between vegetation and land form. The chenopod shrublands dominate the regions extensive areas of plains, low hills and plateaus, and on some depositional plains (Figure 7, Figure 10, Figure 11); the arid and semi-arid acacia low woodlands are associated with the regions depositional plains (Figure 8); and the hummock grasslands, tussock grasslands and Mulga woodlands with tussock grass are all found on the regions dunefields and sand plains (Figure 9, Figure 12).



**Figure 6. Dominant vegetation communities.** Grey lines show IBRA sub-region borders.

In addition to the dominant vegetation type, the study region contains many small ephemeral plant species which emerge after major rainfall events and quickly complete a life cycle. Due to their short life cycles, detection of these species is largely dependent on recent rainfall and consequently their ranges are not well understood.

While the thesis focuses mainly on vegetation species, the study area contains many native fauna species, including at least 156 bird species, 81 reptile species, 30 mammal species and seven frog species. The majority of the regions' native fauna species are small, the only large fauna are the Emu (*Dromaius novahollandiae*), Wedge-tailed Eagle (*Aquila*

*audax audax*), Parentie (*Varanus giganteus*), Red Kangaroo (*Macropus rufus*), and the Dingo (*Canis familiaris dingo*) (Brandle 1998).

Finally, the presence of other introduced plants and animals, in addition to domestic stock, is worth noting. Invasive introduced plant species compete with native species, and account for 6% of all recorded plant species in the region (Brandle 1998). This competition is strongest in wet or disturbed environments, such as along drainage lines and close to stock watering points. Introduced camel (*Camelus dromedarius*) and rabbit (*Oryctolagus cuniculus*) populations compete with native herbivores and domestic stock, and put additional pressure on native vegetation. Finally, introduced fox (*Vulpes vulpes*) and cat (*Felis catus*) populations put undue pressure on small native marsupials and reptiles through predation.

#### 1.3.4 Conservation objectives

The conservation objectives of the Stony Plains region are influenced by two somewhat aligned goals: the outright desire to conserve the natural environment, and the desire to maintain the capacity of natural systems to support livestock production. The South Australian state government Strategic Plan set a ‘no species loss’ target (Government of South Australia 2007), which acknowledges the importance of conservation of natural systems, and particularly species. In the detailed strategy document (Department for Environment and Heritage 2007) it is acknowledged that there is currently inadequate understanding of the distribution and status of many South Australian species, and a need for inventory and monitoring of native species for conservation. Concurrently, the Pastoral Land Management and Conservation Act (1989) requires that the pastoral leases which make up the majority of northern South Australia are managed sustainably, and provides a mandate to monitor the pastoral leases to ensure this requirement is met.

The monitoring required to meet these two targets is currently performed by two vegetation quadrat surveys, which collect vegetation species information in the study area: the Biological Survey of South Australia (BSSA), and the South Australian Pastoral Lease Assessment (SAPLA). These surveys provide the best quality field data for the study area, and the analyses in this thesis examine data collected by these surveys over a 14 year

period, from 1990 to 2003. Due to the differing goals of these two surveys, there are significant differences in their collection methodologies.

The Biological Survey of South Australia (BSSA) is a biological inventory survey, which aims to complete state-wide coverage by 2015. The objective of the BSSA is to gather enough information to allow adequate and appropriate management to conserve South Australia's biodiversity. To this end the BSSA aims to determine the distribution and condition of terrestrial plant and vertebrate species, and to establish a base line for future monitoring.

Because the BSSA is an inventory survey, sites are chosen to be representative of the majority of vegetation communities in an area, and within each vegetation community are biased towards areas less disturbed by grazing. A botanical expert is involved in all surveys, and voucher specimens are collected for species not identified on site. The plant inventory is conducted in square quadrats of one hectare, or an equivalent rectangular area if placed in elongated vegetation communities (Heard and Channon 1997). Vegetation surveys are usually only conducted once per site, although several sites were resurveyed twice yearly for approximately eight years. The location and distribution of the 892 BSSA sites within the study area are presented in Figure 13. Finally, the results of the BSSA Stony Plains survey can be found in Brandle (1998).

The South Australian Pastoral Lease Assessment (SAPLA) is designed to monitor the effect of livestock grazing on land condition of pastoral leases. The information gathered by the survey provides the objective information necessary for government to assess stocking levels.



**Figure 7. Chenopod shrubland.**



**Figure 8. Acacia low open woodland.**



**Figure 9. Simpson Desert. Photo courtesy of Patricia Mc.**



**Figure 10. Stony gibber, typical of Arcoona Plateau IBRA 6.1 sub-region and some parts of other sub-regions. Photo courtesy of Patricia Mc.**



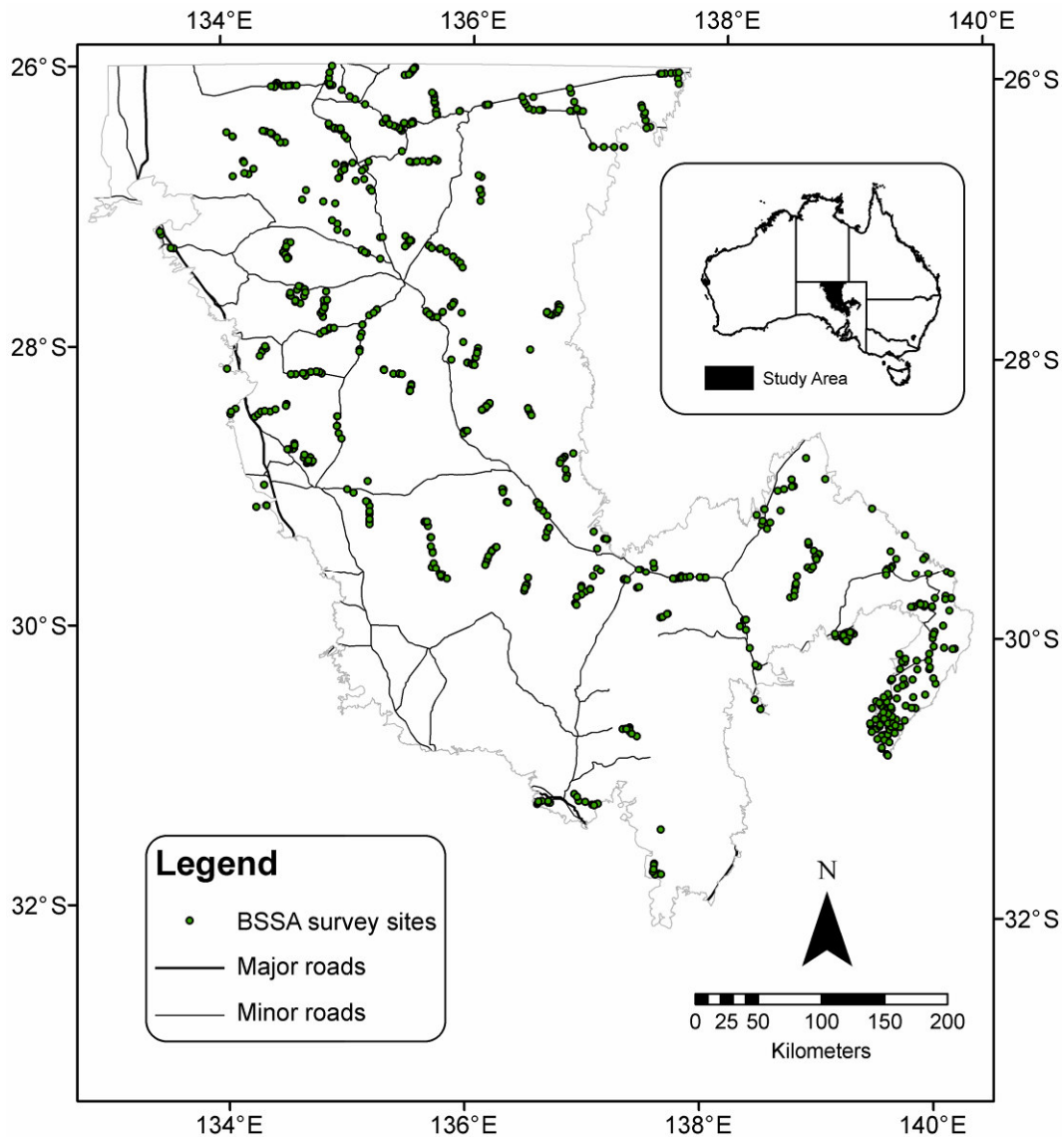
**Figure 11. Stony plains.**



**Figure 12. Open woodland and tussock grass along an ephemeral creek.**

The SAPLA aims to monitor land condition in all paddocks under pastoral lease, and the survey includes several measures including restricted random sampling of land condition along station tracks, photopoint records, quadrat surveys, and some transects. The thesis

considers only the data collected by the quadrat survey, which are conducted around photopoint locations.



**Figure 13. Biological Survey of South Australia (BSSA) site locations.**

Quadrat sites are placed within the grazed area around water points, known as the piosphere, but not in the immediate vicinity of the water point (Lange 1969; Department of Water, Land and Biodiversity Conservation, 2002). In the sheep grazing properties of the southern study area, SAPLA monitoring points are located approximately 1.5 km from watering points, while in the cattle grazing properties in the north they are located approximately 3 km from watering points. Because SAPLA sites are located within stock piospheres they are more likely to be degraded than BSSA sites. Unlike the BSSA, no



botanical expert is involved with SAPLA surveys in the field. SAPLA staff conduct the surveys and attempt to identify all vegetation species, while voucher specimens of any unknown species are collected for later identification. An area of 100 to 200 metres radius is surveyed at each site. Because the SAPLA is designed to monitor change in range condition, sites are revisited at regular intervals. The location and distribution of the 1185 SAPLA sites within the study area are presented in Figure 14. The higher density of sites in the south is noteworthy, and corresponds to the smaller paddocks associated with sheep grazing.

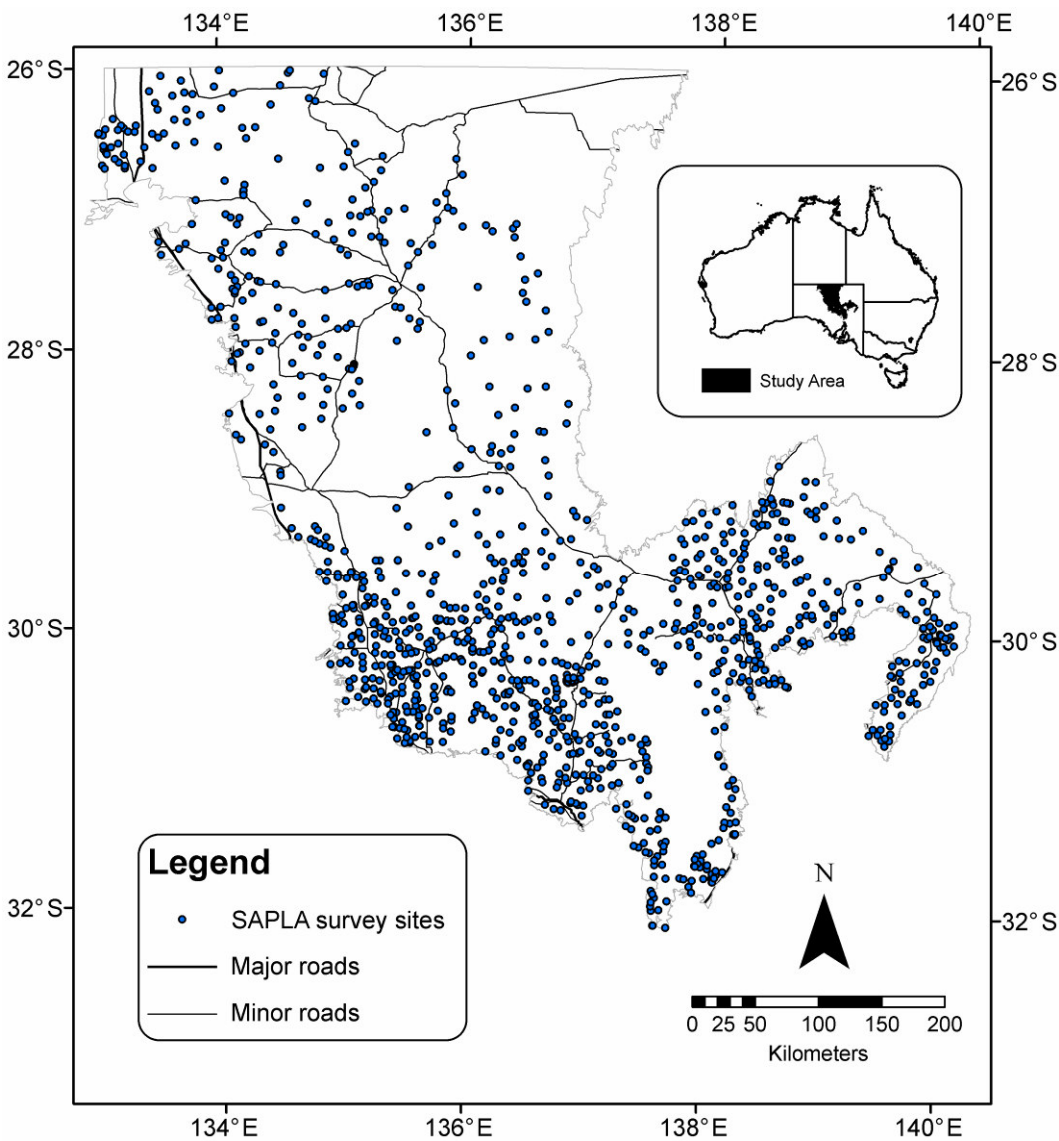


Figure 14. South Australian Pastoral Lease Assessment (SAPLA) site locations.

## 1.4 References

- Badgley, C. and D. L. Fox (2000) Ecological biogeography of North America mammals: species density and ecological structure in relation to environmental gradients. *Journal of Biogeography* **27**: 1437-1467.
- Brandle, R., Ed. (1998) A biological survey of the Stony Deserts, South Australia, 1994 - 1997, Biological Survey and Research Section, Department of Environment, Heritage and Aboriginal Affairs & National Parks Foundation of South Australia Inc.
- Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. O'Neill, J. Paruelo, R. Raskin, P. C. Sutton and M. van den Belt (1997) The value of the world's ecosystem services and natural capital. *Nature* **387**: 253-260.
- Currie, D. J. and V. Paquin (1987) Large-scale biogeographical patterns of species richness of trees. *Nature* **329**: 326-331.
- Department for Environment and Heritage (2007) No species loss: a nature conservation strategy for South Australia 2007 - 2017. Adelaide, Australia.
- DWLBC (2002) Pastoral lease assessment, technical manual for assessing land condition on pastoral leases in South Australia, 1990-2000. Adelaide, Department of Water, Land and Biodiversity Conservation, Pastoral Program, Sustainable Resources.
- Government of South Australia (2007) South Australia's strategic plan 2007. Adelaide.
- Hawkins, B. A. and E. E. Porter (2003) Does herbivore diversity depend on plant diversity? The case of California butterflies. *American Naturalist* **161**(1): 40-49.
- Hawkins, B. A. and E. E. Porter (2003) Water-energy balance and the geographic pattern of species richness of western Palearctic butterflies. *Ecological Entomology* **28**: 678-686.
- Hawkins, B. A., E. E. Porter and J. A. F. Diniz-Filho (2003) Productivity and history as predictors of the latitudinal diversity gradient of terrestrial birds. *Ecology* **84**(6): 1608-1623.
- Heard, L. and B. Channon (1997) Guide to a native vegetation survey: Using the Biological Survey of South Australia. Adelaide, SA, Department of Environment and Natural Resources.
- Lange, R. T. (1969) The piosphere: sheep track and dung patterns. *Journal of Range Management* **22**: 396-400.
- O'Brien, E. M. (1993) Climatic gradients in woody plant species richness: towards an explanation based on an analysis of Southern Africa's woody flora. *Journal of Biogeography* **20**: 181-198.
- O'Brien, E. M. (1998) Water-energy dynamics, climate, and prediction of woody plants species richness: an interim general model. *Journal of Biogeography* **25**: 379-398.
- O'Brien, E. M., R. Field and R. J. Whittaker (2000) Climatic gradients in woody plant (tree and shrub) diversity: water-energy dynamics, residual variation, and topography. *Oikos* **89**(3): 588-600.
- Rodrigues, A. S. L. and T. M. Brooks (2007) Shortcuts for biodiversity conservation planning: the effectiveness of surrogates. *Annual Review of Ecology, Evolution, and Systematics* **38**(1): 713-737.
- Sarkar, S. (2002) Defining "Biodiversity"; Assessing Biodiversity. *The Monist* **85**(1): 131-155.
- Smyth, A. K., V. H. Chewings, G. N. Bastin, S. Ferrier, G. Manion and B. Clifford (2004) Integrating historical datasets to prioritise areas for biodiversity monitoring? Australian Rangelands Society 13th Biennial Conference: "Living in the outback", Alice Springs, Northern Territory.
- Venevsky, S. and I. Venevskaia (2005) Hierarchical systematic conservation planning at the national level: Identifying national biodiversity hotspots using abiotic factors in Russia. *Biological Conservation* **124**: 235-251.

## Chapter 2: Literature review

### 2.1 Introduction

This review covers several important topics which develop the logic behind the work conducted in this thesis. Firstly the literature on the ecological determinants of and pressures on biodiversity is reviewed with a view to identifying potential surrogates of biodiversity which are relevant to the study environment and measurable with remotely sensed data. Next, current remote sensing methods of measuring or monitoring biodiversity are reviewed. Finally, the potential biodiversity surrogates identified through this review process are outlined.

However, specific biodiversity and scale terminology is used in the review and throughout the thesis, and therefore this terminology will be clarified before proceeding any further.

#### 2.1.1 Biodiversity phenomena: $\alpha$ -, $\beta$ - and $\gamma$ -diversity

Throughout this thesis the terminology of Whittaker (1972) is used to describe different biodiversity, or more correctly species-diversity phenomena. In this terminology  $\alpha$ -diversity is the species richness at a site of standard size;  $\beta$ -diversity is the difference in species composition between these sites; and  $\gamma$ -diversity is the species diversity of a region. Thus  $\alpha$ - and  $\gamma$ -diversity are absolute measures, while  $\beta$ -diversity is a comparative measure.

#### 2.1.2 Scale in biodiversity studies

In studies on determinants of biodiversity in the past there seems to have been some confusion as to whether “scale” refers to the extent of a study or the size of the samples it uses. Because many ecological phenomena are scale dependant (Lyons and Willig 2002), and because  $\alpha$ -,  $\beta$ - and  $\gamma$ -diversity are often discussed in relation to scale, it is important that the use of the term is clarified.

Whittaker *et al.* (2003) argue that the scale of a study is determined by the size of its samples. The reason for this is twofold: it is not possible to examine the spatial fluctuation of variables that change over distances smaller than the sample size; and, the effect of

variables that have a subtle effect over larger distances will be masked by small scale variance if sample size is too small to capture an average of community structure at the appropriate scale. However, many studies into biodiversity pattern have not used sampling scales appropriate to the scale at which the variables of interest change, and this confusion of extent and scale has needlessly confounded our understanding of broad scale patterns of biodiversity (Whittaker *et al.* 2003). For the sake of clarity, this document accepts the definition of scale given by Whittaker *et al.* (2003): the scale of a study is determined by the size of its samples, not the extent of the study. This definition of scale is often referred to as grain.

The general terms, micro, meso and macro scale are used frequently in the biodiversity literature to describe the spatial scale of studies, but are almost never adequately defined. Although there seems to be general agreement in the use of these scales at their extremes, there is room for confusion at their boundaries. In the context of this discussion of biodiversity we believe a rational classification of these scales can be arrived at by relating them to the scale of variation of diversity phenomena ( $\alpha$ -,  $\beta$ - and  $\gamma$ -diversity). Hence, we consider studies of micro-scale variation in biodiversity to correspond to  $\alpha$ -diversity; studies of macro-scale variation in biodiversity correspond to  $\gamma$ -diversity; and the poorly-defined middle ground of meso-scale studies of biodiversity may, depending on the specifics of the given study, correspond to either  $\alpha$ -,  $\beta$ - or  $\gamma$ -diversity.

An approximate guide to the scale, sample grain appropriate to that scale, and the diversity phenomena measurable at that scale is presented in Table 2. This table is only intended as a guide to aid clarity and consistency in the discussion to follow, and is not intended as a statement that the given form of biodiversity only and always varies at the defined scale. It is acknowledged that the scale at which different types of biodiversity vary probably changes to some extent depending on climatic and other variables. Indeed there is some evidence that this is the case, Ohmann and Spies (1998) found that community structure varied at finer scale in the dryer than in the wetter regions of Oregon.

**Table 2. Approximate guide to scale, sample grain and corresponding biodiversity phenomena**

Scale	Approximate sample grain	Diversity phenomena measurable
Micro	< 0.01 km <sup>2</sup> (1 ha)	$\alpha$ -diversity
Meso	0.01 – 100 km <sup>2</sup>	$\alpha$ -, $\beta$ - or $\gamma$ -diversity
Macro	> 100 km <sup>2</sup>	$\gamma$ -diversity

### 2.1.3 Determinants of biodiversity

This section discusses the main determinants of biodiversity and reviews the relevant literature. However, the dissection of these different variables is confounded by the interrelatedness of ecological processes. For instance, it has been hypothesised that greater primary productivity leads to greater biodiversity (Abrams 1995), and that climate is the chief determinant of primary productivity (O'Brien 1993; Hawkins *et al.* 2003). Indeed, there is a strong demonstrated link between primary productivity and biodiversity (Cardinale *et al.* 2006) However it is clear that other factors such as soil type (Miller *et al.* 2002), landscape degradation (Bastin *et al.* 2002), and disturbance also influence primary productivity.

This discussion is structured, as far as possible from information gleaned through this review, with the variable which explains the greatest amount of variation in biodiversity first (climate/productivity), through to those variables which explain lesser or poorly defined amounts. Because of the interrelatedness of ecological processes and the difficulty of teasing out the influence of different variables this structure is intended only as a guide to the relative contributions of variables to biodiversity, and not an absolute assessment.

#### **Climate and productivity**

The species-energy hypothesis proposes that the availability of energy determines biodiversity (Wright 1983). Indeed, there is evidence that the majority of variation in species richness of plants (Wright 1983; Currie and Paquin 1987; Adams and Woodward 1989; O'Brien 1993; O'Brien 1998; O'Brien *et al.* 2000; Venevsky and Venevskaja 2005), mammals (Currie 1991; Badgley and Fox 2000), butterflies (Hawkins and Porter 2003) and bird species (Currie 1991; Hawkins *et al.* 2003) at broad scales is determined by climatic variables associated with energy availability.

One of the best supported explanations for the relationship between climatic variables and biodiversity is the Productivity Theory. This theory reasons that the greater the amount and duration of primary productivity the greater the capacity to generate and support high biodiversity (O'Brien 1993; Whittaker *et al.* 2003). However, some have questioned why greater productivity should not simply lead to larger populations without increasing species richness (Willig *et al.* 2003). Some theoretical explanations were advanced by Abrams (1995):

1. Increased productivity increases the abundance of rare species, reducing their extinction rates;
2. Increased productivity increases the abundance of rare resources or combinations of resources and conditions that are required by specialists;
3. Increased productivity increases intraspecific density dependence, allowing coexistence of species, some of which would be excluded at lower productivity;

and a fourth theoretical explanation was provided by Whittaker *et al.* (2003), that

4. Over large geographical areas, cells of generally high productivity will contain scattered low productivity sites, and their species will contribute to the diversity measured across high productivity regions.

Abrams (1995) cited evidence for each of these possible explanations, while the strong and consistent correlations found in many studies suggest that the relationship between climate and species richness is relatively direct (Turner 2004). Indeed, the Productivity Theory is further supported by a recent meta-analysis of 111 biodiversity experiments which found that, in general, the most diverse systems were also the most productive (Cardinale *et al.* 2006).

Thus, climatic variables determine primary productivity, which in turn determines biodiversity. But which climatic variables are important, and specifically how do they determine primary productivity? Primary productivity is a function of maximised water and optimised energy, or “water-energy balance,” (O'Brien 1993; Hawkins *et al.* 2003). Furthermore, measures of water-energy balance have been demonstrated to explain the

majority of variation in tree species richness in Africa, South America, the United States and China (O'Brien 1998); of vascular plant species richness globally (Venevsky and Venevskaja 2003); of butterfly species richness in western/central Europe, northern Africa and California (Hawkins and Porter 2003; Hawkins and Porter 2003); of mammal species richness in North America (Badgley and Fox 2000); and of bird species richness globally (Hawkins *et al.* 2003).

This idea, that at the macro-scale water-energy dynamics are the primary determining factor for species richness, was formalised by O'Brien (1998) with the Interim General Model (IGM) of water-energy dynamics for the prediction of woody plant richness. O'Brien (1998) found that for Africa, woody plant species richness was best described as a function of maximised water and optimum energy, or:

$$\text{Species richness} = \text{water} + (\text{energy} - \text{energy}^2)$$

Thus for a given energy level species richness increases as available water increases. The relationship of energy to species richness is more complex. At very low and at high energy levels species richness is zero and approaches a maximum for energy levels in between the two extremes. This is because the availability of water for biotic processes is dependent on energy: too little energy and water is solid, too much energy and water becomes a gas.

Finally, recent refinement of the IGM has led to its generalisation to the theory of 'biological relativity to water-energy dynamics' (O'Brien 2006). This theory states that the capacity for life to exist is determined by the capacity for liquid water to exist, and that water-energy dynamics are a fundamental mechanism of evolution, through natural selection. However, O'Brien (2006) make the point that at global scales water-energy dynamics should be the primary determinant of biotic dynamics, but that the relationship will necessarily dissolve into apparent chaos locally.

Thus, at broad scales the relationship between water-energy dynamics and species richness has been demonstrated by significant macro-scale studies, and is relatively consistent across the globe. The extent of the study area for the current research is large enough that water-energy dynamics are expected to play a major role in determining regional species richness ( $\gamma$ -diversity).

## Topography

It is well established that areas of increased topographic relief are associated with increased species richness, as compared to flatter areas (Simpson 1964; Richerson and Lum 1980; Badgley and Fox 2000; O'Brien *et al.* 2000). For instance Richerson and Lum (1980) found that topographic heterogeneity had a strong effect on patterns of flora and fauna species richness. Diversity of flora was highest in the mountainous regions of California and lowest in the flatter regions.

It appears that topography affects species richness in four primary ways:

1. By supporting a wider range of climatic zones and hence habitats than an equivalent “flat” area at the same latitude. For instance a mountainous tropical region could contain habitats ranging from tropical rainforest in the lowest regions, through temperate and alpine forests to tundra above the snow line. Normally these habitats would be separated by many degrees of latitude (or hundreds of kilometres) but the high relief of the region allows communities that favour these habitats to coexist within a relatively small spatial area (Turner 2004). This effect is at a maximum at the equator and decreases towards the poles, i.e. mountains from middle latitudes support temperate to tundra/snow habitats while arctic mountainous regions only sport snow.
2. The difference in solar illumination angle between north and south facing slopes create microclimates that would normally be found several degrees of latitude apart (Turner 2004). This effect increases with distance from the equator.
3. According to Hewitt (1996), during climatic warming populations would move upward in elevation toward peaks, splitting into allopatric populations. In periods of cooling populations would then retreat down the mountains and become sympatric again. While this process may generate species in periods of cooling it would potentially result in extinctions in periods of warming. It is therefore not clear that this process would necessarily lead greater species richness in areas of high relief.



4. A quadrat projected onto an area of high relief will have a greater real area than one placed over a flat area. For instance, an quadrat containing an area with an average slope of  $45^\circ$  will contain approximately 40% more real area than an equivalent flat area (Turner 2004). Thus some of the apparently greater richness in areas of high relief is an artefact of viewing the world from a simplistic mapping perspective which portrays the globe as a sphere (O'Brien *et al.* 2000; Whittaker *et al.* 2003; Turner 2004).

But how significant is the effect of topography on species richness and at what scale is it measurable? Several studies at the macro scale have found that after climate, topography was the next strongest explanatory variable for species richness (Richerson and Lum 1980; Currie and Paquin 1987; O'Brien *et al.* 2000). While some other studies have not separated out the relative significance of the different factors, models based on topography and climate have been demonstrated to be capable of explaining the majority of variation in the spatial distribution of species richness (Simpson 1964; Badgley and Fox 2000; Guégan *et al.* 2001; Venevsky and Venevskaja 2003).

The scale at which topography influences variation in species richness is likely to be the scale at which the topography varies, which will depend on the particular topographic feature. It is fair to say however that topography has been found to influence species richness on a range of scales. Some authors have demonstrated the significant role of topography in determining species richness at continental, or macro-scale (Simpson 1964; O'Brien *et al.* 2000), while others have stated that not only does topography have an influence on species richness at smaller scales, but also that it might be a more important determinant of species richness than climate at these scales (Ohmann and Spies 1998).

Thus, the potential impact of topography should be considered during the development of any biodiversity index. Specifically, it is possible that topographic variation within the study area will play a role in determining the distribution of biodiversity.

### **Topographic redistribution of rainfall**

Much of the precipitation in the study area occurs as infrequent high volume events. In these events precipitation which falls on ridges and slopes, or runoff areas, is redistributed

by overland or sub-surface flow to areas of shallower relief, or run-on areas (HilleRisLambers *et al.* 2001). Run-on areas are observed to have high primary productivity and biodiversity (Ludwig *et al.* 2004), a result predicted by the previously discussed theory of 'biological relativity to water-energy dynamics' (O'Brien 2006) due to the locally increased availability of water. However, the increase in productivity and biodiversity is the result of more than the simple topographic redistribution of precipitation.

A positive feedback between vegetation density and water infiltration in semi-arid areas allows greater water infiltration in more vegetated areas (HilleRisLambers *et al.* 2001). Vegetation reduces raindrop impact and compaction and increases soil organic matter content which improve water infiltration rates (Cross and Schlesinger 1999; Schlesinger *et al.* 1999; Wainwright *et al.* 1999; Sparrow *et al.* 2003; Tongway *et al.* 2003), which in turn allow greater vegetation growth and so on.

The combination of topographic redistribution of water, and the positive feedback between plant density and water infiltration create regions of locally increased water availability (above that predicted by regional precipitation records). In combination with the theory of 'biological relativity to water-energy dynamics' these factors explain the observed increase in productivity and biodiversity in run-on areas.

### **Area and heterogeneity**

The respective influences of area and heterogeneity on biodiversity are somewhat intertwined and difficult to separate. This section provides a brief overview of the literature relating to area and biodiversity, and habitat heterogeneity and biodiversity, and finally a summary of the probable link between area and habitat heterogeneity.

In 1973 Terborgh suggested that the tropics support more species than other latitudinal belts because they cover a larger area, and the two tropical belts are neighbours whereas other latitudinal belts are separated by the tropics. Hence any species with environmental tolerances that allow it to exist in the tropics has the potential to populate any area within the tropics without having to cross through unfavourable latitudes, while species from

other latitudes are unlikely to be able to populate their corresponding latitudinal belt in the opposite hemisphere.

Of course species cannot freely move within the global extent of an entire latitudinal belt due to geographic obstacles. However this theory suggests, as does the Theory of Island Biogeography (MacArthur and Wilson 1967) that diversity of species should increase as the size of a landmass in a latitudinal band increases. This has been demonstrated to be the case for islands (i.e. larger islands support more species than smaller comparable islands) and there is some evidence for this effect on larger landmasses, but this theory is still controversial and far from proven (Turner 2004). For instance Rahbek and Graves (2001) found a trough in avian  $\gamma$ -diversity in central Amazonia, instead of the peak predicted by this theory.

Alternatively, it has been suggested that areas of a given size that contain more diverse habitats will support a greater biodiversity; this theory is sometimes referred to as the habitat heterogeneity hypothesis (Hawkins *et al.* 2003). Research in this area is strongly related to the research into the effect of topography on species richness, since the mechanism proposed by that work is topographic heterogeneity, rather than absolute elevation.

Rahbek and Graves (2001) found a strong relationship between topographic heterogeneity and species richness of birds, and O'Brien *et al.* (2000) found a similar relationship between topographic heterogeneity and woody plant species richness. In the case of riverine fish Guégan *et al.* (2001) demonstrated that at the macro scale the majority of variation in species richness was determined by net primary productivity (NPP) and habitat heterogeneity. Lastly Miller *et al.* (2002) found habitat heterogeneity was important at the micro-scale for determining plant species diversity in old growth hardwood forests in the northern USA.

The link between topographic heterogeneity and increased species richness as found in the studies by Rahbek and Graves (2001) and O'Brien *et al.* (2000) has already been discussed in the preceding section. The findings by Guégan *et al.* (2001) and Miller *et al.* (2002) do suggest that habitat heterogeneity plays a strong role in determining the species richness of

riverine fish and plant species respectively. However, it is unclear how universal this relationship is, and whether it is likely to apply within the study area.

In summary, it seems probable that the habitat heterogeneity hypothesis goes a long way to explaining the association between area and biodiversity: larger areas will often contain more diverse habitat, and therefore more species. Regardless, this discussion is academic since the data available for this thesis does not cover a large enough area to test this theory.

### **Soil type**

Since productivity plays a significant role in determining the spatial variation of biodiversity it seems reasonable to conclude that any factor which influences productivity might also have influence the distribution of biodiversity. Soil type is such a factor, and is examined in this section.

In old-growth hardwood forests of northern USA Miller *et al.* (2002) found that species presence and absence was strongly determined by soil type, to the point where most species only occurred on one soil type. Additionally, many South Australian soils are strongly associated with distinctive vegetation communities (Specht and Specht 1999). Importantly Miller *et al.* (2002) found that some soil types supported communities with greater species richness ( $\alpha$ -diversity) than others, although this finding was not quantified. However a study in an Amazonian rainforest (Tuomisto *et al.* 2003) found no relationship between species richness and variation in soil type.

Thus, it is possible to synthesise the probable influence of soils on biodiversity from the literature, and the influence may be different for  $\alpha$ - and  $\gamma$ -diversity. The relationship of soils and  $\alpha$ -diversity appears relatively clear: particular vegetation communities are strongly associated with certain soils, and some soils happen to support vegetation communities with a higher  $\alpha$ -diversity than other soils. However, there does not appear to be any evidence that particular soil types consistently support greater  $\alpha$ -diversity than others.

The relationship of soils and  $\gamma$ -diversity is less clear. The literature suggests that the strong association of particular vegetation communities with certain soils is common. This association may have ramifications for the influence of soil heterogeneity on  $\gamma$ -diversity,

although this is open to interpretation: if different soils support different vegetation communities with similar  $\alpha$ -diversity, then the  $\gamma$ -diversity of a region would be strongly influenced by the number of different soil types present. However, this is predicated on the assumption that there is little overlap in the vegetation communities supported on different soils.

To summarise, it seems probable that soil heterogeneity would influence vegetation  $\gamma$ -diversity in the study area. However, due to the extent of the study area and the relatively coarse scale of available soil maps, it may not be possible to examine the influence of soil-heterogeneity on biodiversity.

### **The influence of environmental variability on speciation**

Environmental variability has sometimes been postulated to have the potential to generate the global gradient of biodiversity, and is therefore a potential determinant of biodiversity distribution worth examining. Turner (2004) states that the balance between the rates of speciation and extinction must determine to some extent the biodiversity of a given community. If speciation rate is determined by generation time (and is therefore higher in the tropics because of higher temperatures) and extinction rate is determined by environmental fluctuations that occur in absolute time then the change in temperature from the equator to the poles could potentially generate the latitudinal gradient of species richness (Turner 2004).

However, there is significant evidence against the hypothesis that environmental variability decreases biodiversity. Richerson and Lum (1980) found mean values of temperature and precipitation to be more important for explaining biodiversity distribution than seasonality and irregularity. This could suggest that longer-term trends in climatic variables and their effect on the water-energy balance is a more important determinant of biodiversity than climatic variability.

In a study of bird species Bromham and Cardillo (2003) found no evidence to support the idea that rates of molecular evolution increase towards the tropics, one of the key mechanisms of this theory. Willig *et al.* (2003) argued that the environmental variability

hypotheses should be discarded if studies of other taxa fail to find evidence for greater evolutionary rates at the tropics.

Due to the relatively uniform environment of the study area (see Chapter 1, Study area), the environmental variability hypothesis would be expected to play a minor role in determining the distribution of biodiversity. For this reason, and due to the evidence against the environmental variability hypothesis, this hypothesis will not be investigated further in this thesis.

### **Fire**

The effects of fire on biodiversity differ depending on the landscape. Vegetation cover in the study area is so sparse that the region experiences almost no fires. This observation is backed by the satellite fire mapping conducted by the Department of Land Information, Western Australia which recorded few fire hotspots or scars within the study region from 1998 to 2004 (Figure 15).

However a small part of the region is covered by Acacia wooded landscapes which are fire-prone (Hodgkinson 2002). These woodlands are characterised by dense groves of vegetation with sparse interpatch areas; fuel loads in the interpatch areas can reach 800-1800 kg ha<sup>-1</sup> and up to 7000 kg ha<sup>-1</sup> in groves. This patchy distribution of fuel results in a high spatial variability in fire intensity; although the low fuel load in interpatch areas can prevent fires from spreading. Thus while these woodlands can burn, fire frequency is typically low (Hodgkinson 2002).

Fires have different effects on different species within a landscape type, for instance, in one case a summer fire in central Australia greatly reduced grass biomass and increased forb biomass (Griffin and Friedel 1984). While fires cause the death of perennial plants the new spaces within the landscape allow the germination of a new generation of perennials. Indeed the pattern of shrub recruitment within Australian rangelands is strongly influenced by fire (Hodgkinson 2002). Thus the small portion of the study area which is prone to burning has evolved to cope with, and to some extent depend on fire. Fire is likely to have neither a net positive or negative influence on biodiversity within the study area.

#### 2.1.4 Pressures on biodiversity

While there is no doubt that environmental variables play a large role in determining the potential biodiversity of a given location, human disturbance in the form of land-use change has been identified as the most critical driver of biodiversity loss over the next 100 years (Sala *et al.* 2000). Specific causes of this habitat loss include urban expansion, and overgrazing which leads to erosion, although the former is not likely to be a significant pressure in the South Australian rangelands. This section examines two pressures on biodiversity, grazing induced landscape degradation, and invasion by exotic species.

##### **Grazing induced degradation**

The main pressure on biodiversity in the study area is landscape degradation, caused primarily by the grazing of introduced stock. In healthy perennially-vegetated landscapes vegetation occurs in patches with relatively bare inter-patch areas, with nutrients concentrated in vegetated patches (Reynolds *et al.* 1997). Grazing-induced landscape degradation, or overgrazing, causes loss of vegetation from patches, and through wind and water erosion the loss of nutrient-rich soil and plant litter from the landscape (Shaver *et al.* 1991; Northrup and Brown 1999; Meadows and Hoffman 2002). Additionally, this degradation has been demonstrated to reduce soil physical and nutrient cycling, and hence soil fertility, and to reduce water infiltration (Schlesinger and Pilmanis 1998; Cross and Schlesinger 1999; Tongway *et al.* 2003; Lechmere-Oertel *et al.* 2005), reducing potential primary production.

NOTE:  
This figure is included on page 31  
of the print copy of the thesis held in  
the University of Adelaide Library.

**Figure 15. Fire history in the Stony Plains IBRA, courtesy of the Department of Land Information, Western Australia. Fires mapped from National Oceanic and Atmospheric Administration (NOAA) Advanced Very High Resolution Radiometer (AVHRR) data.**

Indeed, there is evidence that overgrazing reduces net primary productivity and rainfall use efficiency (the amount of phytomass produced per unit rainfall) at broad scales. Generally, research has found that in arid and semi-arid rangelands net primary production and rainfall use efficiency are lower in degraded systems, and higher in less- or non-degraded systems (Le Houerou 1984; Snyman and Fouché 1993; Snyman 1997; Snyman 1998; Holm *et al.* 2002; Holm *et al.* 2003). However, the relationship may be somewhat more complex. In a 12 year study in semi-arid Australian rangelands, Holm *et al.* (2003) found



that net primary productivity and rainfall use efficiency were lower on average, and more temporally variable in a degraded landscape, as compared to a non-degraded landscape. In infrequent years of unusually high rainfall, increased primary production from ephemeral growth resulted in net primary production on the degraded landscape similar to, or higher than on the non-degraded landscape. This concurrent increase in temporal variability of net primary production and rainfall use efficiency with increased degradation was also found by Kelly and Walker (1976) in a semi-arid Zimbabwean rangeland.

Thus, there is a strong link between overgrazing and decreases in the measurable landscape function variable, rainfall use efficiency. There is also a corresponding link between grazing induced degradation and reduced woody perennial vegetation  $\gamma$ -diversity. However, the relationship between degradation and species-diversity is not simple, and may vary throughout the world's rangeland systems.

In Australia, James *et al.* (1999) found that grazing selectively removed palatable plants from within the piosphere, the zone of grazing influence around stock water points, locally reducing native plant species diversity. In southern Australia, Landsberg *et al.* (2002) found a grazing induced increase in plant species abundance at the local scale, and a decrease at the regional scale. However, of the species found to increase in abundance with proximity to water, most only exhibited this tendency on one transect in the study. Of the plant species which exhibited trends of abundance in relation to water on two or more transects, only two species increased with proximity to water, while four species decreased. Thus, an alternative interpretation is that the results of Landsberg *et al.* (2002) show a consistent grazing induced decline in species abundance at regional scales, a consistent decrease in species abundance at local scales, and an inconsistent increase in species abundance at local scales. Subsequent work by Landsberg *et al.* (2003) in Australia found that ground layer (< 50 cm height) plant species which decreased in abundance with proximity to water were significantly outnumbered by those which increased. Importantly, Landsberg *et al.* (2003) also found significantly more 'singletons', plants which were recorded only once out of all sites, at the sample points most remote from water. This suggests a grazing induced loss of rare and uncommon species close to stock water points. Landsberg *et al.* (2003) found no relationship between upper layer plant species (> 50 cm height) and proximity to water. In a final Australian study,

McIntyre *et al.* (2003) found that increased grazing pressure lead to an increase in the  $\alpha$ -diversity of introduced or invasive species, and a decrease in native plant  $\alpha$ -diversity.

In a hyper-arid system in northern Africa, Ali *et al.* (2000) demonstrated an increase in plant  $\alpha$ -diversity at moderate grazing pressure, and a decrease in plant  $\alpha$ -diversity at higher grazing pressure. In South Africa, Todd (2006) found that sustained heavy grazing in a semi-arid system negatively impacted the plant  $\alpha$ -diversity and structural diversity of the landscape. Conversely, another study in South Africa found no consistent relationship between plant species-diversity and distance from watering points (Thrash *et al.* 1993).

Finally, in the Mojave Desert of North America, Brooks *et al.* (2006) found that both annual and perennial plant cover, and  $\alpha$ -diversity decreased with proximity to stock watering sites. Conversely, introduced plant abundance increased with proximity to watering sites.

While the effects of grazing on plant species diversity across the globe may be somewhat variable, the results within Australia appear consistent. Grazing leads to an overall decrease in native plant  $\alpha$ - and  $\gamma$ -diversity, and to an increase in invasive plant  $\alpha$ -diversity.

### **Exotic species invasion**

Invasion of exotic species has been identified as a serious potential threat to biodiversity in regions with Mediterranean climates outside Europe, such as South Australia, due to their long isolation and extensive convergent evolution (Sala *et al.* 2000). Exotic plant and animal species threaten native species through competition, predation and herbivory.

Predation by the introduced fox *Vulpes vulpes* and cat *Felis catus* is a serious pressure native mammal populations, and has caused the decline and extinction of many Australian mammals (Smith and Quin 1996; McKenzie *et al.* 2007). Additional pressure has been placed on small Australian mammals by competition from the introduced rabbit *Oryctolagus cuniculus*.

Finally, herbivory by the introduced rabbit and domestic and feral stock cause grazing induced degradation, covered in the previous section. It should also be noted that there

appears to be a synergy between grazing pressure and invasion of exotic vegetation species, which is enhanced by increased grazing (McIntyre *et al.* 2003).

## **2.2 Surrogates for monitoring biodiversity**

This section reviews recent approaches to measuring biodiversity, or factors which were identified as likely to be related to biodiversity by the preceding review of the determinants of biodiversity. Five broad categories of approach are reviewed: generalised dissimilarity modelling; vegetation community classification; indicators of landscape condition; measures of landscape heterogeneity; and gamma-ray spectrometry.

### **2.2.1 Generalised dissimilarity modelling**

Smyth *et al.* (2007) took a risk-based approach to identifying potential biodiversity surrogates in some of the Stony Plains of South Australia, a subset of the same area examined in the thesis. Primary threats to biodiversity in the Stony Plains were identified, and from these a range of relevant potential biodiversity surrogates were identified. These included a range of topographic indices derived from a digital elevation model, as well as the digital elevation model itself; climatic information including temperature, radiation and moisture; Landsat TM bands 2, 3 & 4; a remotely-sensed perennial vegetation index, its variance and contrast; gamma radiometric data; distance to water; grazing gradients, and grazing pressure.

Generalised dissimilarity modelling (GDM) (Ferrier 2002) was employed to examine the  $\beta$ -diversity of persistent native vegetation species (as a surrogate for biodiversity) in relation to the potential biodiversity surrogates. Contrary to expectations, no relationship was found between any of the potential biodiversity surrogates and persistent native vegetation  $\beta$ -diversity.

Smyth *et al.* (2007) listed several potential reasons for the failure of their model, the most significant of which were the low resolution of the climate information, and the temporal difference between collection of the native vegetation data, and the satellite imagery acquisition. The climate information was collected from very few stations and then extrapolated to the whole region, and consequently varied little over the study region. The

vegetation data were collected primarily between 1990 and 2003, while the two satellite images were collected in 2000 and 2002.

### 2.2.2 Vegetation community classification

The classification of remotely sensed imagery to map distinct vegetation communities may be indirectly useful in the mapping and monitoring of biodiversity. For biodiversity mapping or monitoring to be assisted by the classification of remotely sensed imagery several factors must be demonstrated. Firstly, vegetation species must be organised into communities, with little variation in species composition within community, and significant differences in species composition between communities. Secondly, there must be significant spectral differences between vegetation communities. Finally and optionally, natural or anthropogenic disturbances which alter vegetation communities and concurrently alter biodiversity must cause significant spectral changes. While the first point is a widely assumed by community ecologists, the second and third points bear illustration.

The classification of vegetation communities in arid lands is a well researched area. For instance, Lewis (1998) used supervised classification of Landsat TM imagery to map vegetation communities in an arid Australian chenopod shrubland similar to the study area. The remotely sensed classes agreed strongly with community classes generated from field sample sites. In a similar climate in a different country, Tanzania, Tobler *et al.* (2003) identified 15 distinct vegetation communities through field work, and then successfully mapped these communities through supervised classification of Landsat TM imagery.

The mapping of disturbance phenomena relevant to arid areas through remotely sensed imagery classification is a similarly well researched area. Image classification can be used to map fire (Verlinden and Laamanen 2005; Alo and Pontius Jr 2008), anthropogenic vegetation clearance (Cameron and Hart 1998; Cameron *et al.* 2004; Alo and Pontius Jr 2008), and domestic stock grazing impact (Tobler *et al.* 2003). While these disturbance phenomena operate at different scales, all have some impact on vegetation communities and hence biodiversity.

Therefore the theoretical requirements are met, and the use of vegetation community classification to assist in biodiversity mapping and monitoring is valid. However this potential is tempered by the volume of field data required. Classification techniques typically require extensive field data input to training procedures, and/or for validation of classes.

### 2.2.3 Indicators of landscape condition: rainfall use efficiency (RUE) and net primary productivity (NPP)

The approach of measuring landscape function or degradation does not allow us to directly determine the biodiversity of an area, although landscape degradation leads to a loss of resources from the system and a reduction in primary productivity. Given that primary productivity appears to play a pivotal role in determining the biodiversity of a given landscape or region it is logical to conclude that landscape degradation leads to a reduction in biodiversity. Hence measures of landscape degradation allow us to monitor one of the most significant pressures on biodiversity in the arid rangelands.

Rainfall use efficiency (RUE) and net primary productivity (NPP) have been found to decrease as the landscape becomes more degraded (Holm *et al.* 2002; Holm *et al.* 2003), though the relationship may not be a simple one. While one study found that degraded landscapes always produced less phytomass and had poorer RUE (Holm *et al.* 2002) another study found that degraded landscapes had lower average RUE and NPP but were overall more variable and in fact produced higher maximum NPP in response to better than average rainfall (Holm *et al.* 2003). In the case of the second study it was suggested that the greater NPP in degraded sites was due to a freeing of resources formerly tied up in perennial vegetation patches for use by fast growing annual species, and a lack of competition by perennial vegetation. Interestingly Holm *et al.* (2002) found no difference in the response of low-shrubland and low-woodland in terms of RUE.

The knowledge gained from these studies has been applied in the examination of the potential for remote sensing to measure RUE and NPP as surrogates for landscape condition (Holm *et al.* 2003). A model based on rainfall, landscape and NPP data collected on-ground was used to estimate NPP and RUE from 1992 to 1999 for a large area of Western Australia. These data were compared to estimates of NPP and RUE generated

from NDVI indices of the National Oceanic and Atmospheric Administration's (NOAA) Advanced Very High Resolution Radiometer (AVHRR) satellite imagery. There was good agreement between ground-based and remotely sensed estimates of NPP but less agreement between estimates of RUE (Holm *et al.* 2003).

In a similar study (Wessels *et al.* 2004) evaluated a time series of AVHRR NDVI imagery in comparison to a ground based assessment of degradation in South Africa. Wessels *et al.* (2003) found that the remotely sensed and ground-measured productivity were consistently lower in degraded than non-degraded areas, even in times of greater than average rainfall.

The field based research to date suggests that RUE or NPP are good indicators of landscape degradation (Holm *et al.* 2002). It seems probable that a measure of NPP or RUE, or perhaps a measure of change in NPP or RUE over time could be a useful method for measuring changes in landscape degradation and hence pressure on biodiversity in the arid and semi-arid rangelands of Australia.

#### 2.2.4 Measures of landscape heterogeneity

The potential for measures of landscape heterogeneity in measuring or monitoring biodiversity has a sound grounding in ecological theory. Strong relationships have been found between landscape heterogeneity and plant (O'Brien *et al.* 2000), bird (Rahbek and Graves 2001) and riverine fish (Guégan *et al.* 2001) species richness. Indeed Tongway *et al.* (2003) found that grazing in the Australian rangelands caused landscape degradation which lead to homogenous landscapes close to livestock watering points while sites remote from watering points remained more heterogeneous. This section examines two methods of measuring landscape heterogeneity, spectral variation and landscape leakiness.

##### **Spectral variation**

The spectral variation hypothesis (SVH) (Palmer *et al.* 2002) takes the form: greater spatial variation within the environment correlates with greater species richness, which in turn correlates with greater spectral variation.

Studies examining the SVH have met with varying degrees of success ranging from no relationship to quite strong relationships between spectral variation and species richness.

Two studies at the micro-scale, one in Amazonian rainforest (Tuomisto *et al.* 2003) and the other in Oklahoma in the USA (Palmer *et al.* 2002), found no relationship between spectral heterogeneity and species richness. Palmer *et al.* (2002) hypothesised that spectral variation would be better able to predict species richness at meso- to macro-scales. In a later test of this hypothesis Rocchini *et al.* (2004) examined spectral variation at two larger scales 100m<sup>2</sup> (still within our definition of micro-scale but larger than the sample size used by Palmer *et al.* (2002)) and 1 ha (meso-scale). This study found that spectral heterogeneity explained some variation in species richness at the micro-scale (approximately 20%) and significantly more variation at the meso-scale (approximately 50%).

In the arid lands of Australia watering points, which are degraded by heavy stock use, were found to have a significantly higher moving standard deviation index (MSDI) value than non degraded areas (Jafari *et al.* 2008). However, the relation of MSDI to species diversity was not examined.

The most promising study of this type found that variation in NDVI generated from Landsat TM imagery was positively correlated with measured species richness and explained 65% of the variation. When the variation in NDVI was combined with a weighted abundance of vegetation types map, generated from a supervised classification, a multiple regression indicated that these two variables significantly explained 79% of variation in species richness of plants (Gould 2000).

Some authors have had reasonable success in mapping biodiversity with these methods. Since this method relies on variation in measured pixel values it may be suitable for mapping and monitoring of biodiversity. However, none of these studies have been conducted in arid environments, and variation in biodiversity is scale dependent, as illustrated by the findings of Rocchini *et al.* (2004).

However, thorough assessment of this approach to mapping biodiversity would require the acquisition and analysis of several scales of remotely sensed imagery. The financial and temporal cost of this analysis is beyond the scope of this thesis.

## Landscape leakiness

Landscape leakiness is a measure of how likely a landscape is to lose resources through wind and water erosion. While measures of landscape leakiness are similar to other measures of landscape heterogeneity, they differ in a key respect. Unlike other landscape heterogeneity measures, landscape leakiness is concerned with how heterogeneous the landscape is as well as how that heterogeneity is distributed.

The measurement of landscape leakiness as a method of evaluating landscape degradation stems from the concept that non-degraded semi-arid landscapes are characterised by a spatial patterning of vegetation patches interspersed by almost bare soil (Aguiar and Sala 1999). Research has shown that soil attributes are better, and key resources are more available in these vegetation patches as compared to the bare inter-patch areas (Cross and Schlesinger 1999; Sparrow *et al.* 2003; Tongway *et al.* 2003; Ludwig *et al.* 2005), and it has been demonstrated that this is a result of retention and capture of water and windborne resources by these vegetation patches (Sparrow *et al.* 2003). Conversely degraded landscapes have fewer vegetation patches (Friedel *et al.* 2003; Ludwig *et al.* 2005) and lose, or 'leak' resources through wind and water erosion (Sparrow *et al.* 2003; Ludwig *et al.* 2005). Grazing has been demonstrated to cause landscapes to become more homogeneous, degraded and leaky, especially closer to stock watering points (Tongway *et al.* 2003). It is this change from a heterogeneous landscape with many flow-obstructing patches to a more homogenous landscape, which leaks resources, that researchers have sought to measure.

A directional leakiness index (DLI) and a multi-directional leakiness index (MDLI) were developed for use with high resolution airborne videography (Bastin *et al.* 2002; Ludwig *et al.* 2002). These indices predict landscape leakiness by modelling resource flow once an image is classified into vegetation patches (flow obstructing patches) and inter-patch area (which promotes resource flow). However these indices do not incorporate topographic information and are therefore limited to modelling uni-directional flow (in the case of the DLI) or flow in a landscape with distinct vegetation bands (in the case of the MDLI). Despite this limitation both indices were able to correctly rank several sample sites from most to least degraded (Bastin *et al.* 2002; Ludwig *et al.* 2002).



While both of these indices are promising, there are a few key factors that limit their usefulness:

1. Both indices require a user to classify the image into “patch” and “inter-patch” areas,
2. They were developed for very high resolution imagery at the micro-scale,
3. Neither index actually models down slope flow, and
4. Neither index is comparable between images or sites due to the lack of a suitable calibration method.

To address the first two limitations Bastin *et al.* (2004) developed a cover based DLI (CDLI). The CDLI uses a measure of cover within pixels from an appropriate vegetation index. This index is suitable for lower resolution imagery such as Landsat TM but still does not model down slope flow and hence assumes uni-directional flow.

The remaining two limitations are critical, and prevent useful application of leakiness indices in biodiversity monitoring. Firstly, broad scale application of leakiness models throughout the study region would require the incorporation of a sophisticated flow model, predicated on a suitably fine resolution digital elevation model, neither of which currently exists. Secondly, the inability to compare leakiness values spatially and temporally renders this method currently inappropriate for monitoring purposes.

### 2.2.5 Airborne gamma-ray spectrometry

Some soils and parent rocks contain radio-elements that emit characteristic gamma-rays as they decay. Many studies have examined the possibility of recording this information with airborne gamma-ray spectrometry (AGS) as a method of mapping soil type and characteristics, possibly under vegetation cover (Cook *et al.* 1995; Bierwirth *et al.* 1996; Wilford *et al.* 1997; Thwaites 2002)

It appears that AGS can potentially provide significant information about the variation in lithology within an area (Wilford *et al.* 1997; Thwaites 2002). AGS has even been

demonstrated to allow mapping of some soil properties such as pH, composition and texture (Bierwirth *et al.* 1996).

However, interpretation of AGS is site specific and requires detailed *a priori* knowledge of the geology and/or geomorphology of a region (Bierwirth *et al.* 1996), or a pedogeomorphic model (Thwaites 2002). Even with extensive ground data, interpretation can be confounded by several factors:

- different elements have different environmental mobilities (Bierwirth *et al.* 1996);
- the relationship between regolith material and gamma-ray response differs from region to region (Wilford *et al.* 1997);
- in some cases different regoliths have similar gamma-ray responses (Wilford *et al.* 1997);
- some regoliths are free of radio isotopes and hence produce no gamma-ray response at all (Wilford *et al.* 1997);
- soil moisture levels can cause variation in the gamma-ray response indistinguishable from that due to variation in the regolith (Wilford *et al.* 1997); and
- the extent to which vegetation impedes measurement of gamma-ray emissions from soil is as yet unclear.

Lastly the costs of data collection limit the use of AGS to micro and meso-scale studies.

AGS appears to show little promise in mapping or monitoring biodiversity. As previously discussed, the ecological literature is inconclusive as to whether soil plays an important role in determining biodiversity, interpretation of AGS requires extensive *a priori* knowledge of the geomorphology of the region, and is only useful in the mapping of some soils. Finally, Smyth *et al.* (2007) found no relationship between AGS and biodiversity in the study region. Considering this, and the uncertainty surrounding whether vegetation might impede the measurement of gamma-ray emissions, and AGS would seem to confound any prediction of biodiversity more than it helped.

## **2.3 Summary and potential biodiversity surrogates**

There are no current comprehensive methods for measuring and/or monitoring biodiversity at the extensive scales of the Australian rangelands, remote sensing or ground based.

Through this review the determinants of and pressures on biodiversity in the study area and at extensive scales were identified. The primary determinant of biodiversity was identified as total primary productivity, and the balance between water and energy availability for life was identified as the main determinant of primary productivity. The largest pressure on biodiversity in the study area was identified as grazing induced degradation.

From the understanding of primary determinants of and pressures on biodiversity gained through this review, and knowledge of the capabilities of remote sensing two potential surrogates were identified. These surrogates are briefly outlined below, and developed more thoroughly in Chapter 5.

### **2.3.1 Surrogate 1**

The first surrogate is based on the differential effect grazing induced degradation on a measure of expected primary productivity and a measure of actual primary productivity. The measure of expected primary productivity is water-energy balance, and the measure of actual primary productivity is derived from remotely sensed data.

Water-energy balance is a determinant of primary productivity, and therefore of possible biodiversity. Because water-energy balance is a function of climatic variables it is independent of grazing disturbance, and is therefore a measure of expected primary productivity in the absence of disturbance.

Actual primary productivity is measured from satellite imagery through the use of integrated normalised difference vegetation index (NDVI). This is a measure of primary productivity after, or including the effect of grazing pressure.

It is hypothesised that the difference between expected and net primary production would provide a measure of biodiversity-reducing disturbance in the study area.

### 2.3.2 Surrogate 2

The second surrogate is inspired by and based on the convincing link between grazing induced degradation and the temporal variability of net primary productivity and rainfall use efficiency. Grazing induced degradation leads to an overall decrease in native plant  $\alpha$ - and  $\gamma$ -diversity, and to an increase in invasive plant  $\alpha$ -diversity. Additionally, over grazing decreases average decreases mean net primary production and rainfall use efficiency, and increases variation in primary production and rainfall use efficiency.

It is hypothesised that the average and variation in net primary productivity and rainfall use efficiency, as measured from satellite imagery and climatic data can provide a measure of grazing induced landscape degradation, and hence pressure on biodiversity.

## References

- Abrams, P. A. (1995) Monotonic or unimodal diversity-productivity gradients: What does competition theory predict? *Ecology* **76**(7): 2019-2027.
- Adams, J. M. and F. I. Woodward (1989) Patterns in tree species richness as a test of the glacial extinction hypothesis. *Nature* **339**: 699-701.
- Aguiar, M. R. and O. E. Sala (1999) Patch structure, dynamics and implications for the functioning of arid ecosystems. *Trends in Ecology and Evolution* **14**: 273-277.
- Ali, M. M., G. Dickinson and K. J. Murphy (2000) Predictors of plant diversity in a hyperarid desert wadi ecosystem. *Journal of Arid Environments* **45**: 215-230.
- Alo, C. A. and R. G. Pontius Jr (2008) Identifying systematic land-cover transitions using remote sensing and GIS: the fate of forests inside and outside protected areas of Southwestern Ghana. *Environment and Planning B: Planning and Design* **35**: 280-295.
- Badgley, C. and D. L. Fox (2000) Ecological biogeography of North America mammals: species density and ecological structure in relation to environmental gradients. *Journal of Biogeography* **27**: 1437-1467.
- Bastin, G. N., J. A. Ludwig, R. W. Eager, V. H. Chewings and A. C. Liedloff (2002) Indicators of landscape function: comparing patchiness metrics using remotely-sensed data from rangelands. *Ecological Indicators* **1**: 247-260.
- Bastin, G. N., J. A. Ludwig, R. W. Eager, A. C. Liedloff and V. H. Chewings (2004) Indices for assessing landscape leakiness at multiple scales. Alice Springs, CSIRO Sustainable Ecosystems.
- Bierwirth, P., P. Gessler and D. McKane (1996) Empirical investigation of airborne gamma-ray images as an indicator of soil properties - Wagga Wagga, NSW. 8th Australian Remote Sensing Conference.
- Bromham, L. and M. Cardillo (2003) Testing the link between the latitudinal gradient in species richness and rates of molecular evolution. *Journal of Evolutionary Biology* **16**: 200-207.
- Brooks, M. L., J. R. Matchett and K. H. Berry (2006) Effects of livestock watering sites on alien and native plants in the Mojave Desert, USA. *Journal of Arid Environments* Special Issue: Deserts of the World Part I: The changing Mojave Desert Guest Editors: K.H. Berry, R. Murphy **67**(Supplement 1): 125-147.
- Cameron, J., S. Childs and L. Pittard-Robb (2004) "Illegal clearers busted from a great height": the use of remote sensing and GIS to manage South Australia's native vegetation. The 12th Australasian Remote Sensing and Photogrammetry Conference, Fremantle, Australia.
- Cameron, J. and D. Hart (1998) Remote sensing of agricultural land cover change 1990-1995: a South Australian perspective. The 9th Australasian Sydney, Australia.
- Cardinale, B. J., D. S. Srivastava, J. E. Duffy, J. P. Wright, A. L. Downing, M. Sankaran and C. Jouseau (2006) Effects of biodiversity on the functioning of trophic groups and ecosystems. *Nature* **443**: 989-992.
- Cook, S. E., R. J. Corner, P. R. Groves and G. J. Greali (1995) Use of airborne gamma radiometric data for soil mapping. *Australian Journal of Soil Research* **34**(1): 183-194.
- Cross, A. F. and W. H. Schlesinger (1999) Plant regulation of soil nutrient distribution in the northern Chihuahuan Desert. *Plant Ecology* **145**: 11-25.
- Currie, D. J. (1991) Energy and large-scale patterns of animal-species and plant-species richness. *American Naturalist* **137**(1): 27-49.
- Currie, D. J. and V. Paquin (1987) Large-scale biogeographical patterns of species richness of trees. *Nature* **329**: 326-331.
- Ferrier, S. (2002) Mapping spatial pattern in biodiversity for regional conservation planning: where to from here? *Systematic Biology* **51**(2): 331-363.

- Friedel, M. H., A. D. Sparrow, J. E. Kinloch and D. J. Tongway (2003) Degradation and recovery processes in arid grazing lands of central Australia. Part 2: vegetation. *Journal of Arid Environments* **55**: 327-348.
- Gould, W. (2000) Remote sensing of vegetation, plant species richness, and regional biodiversity hotspots. *Ecological Applications* **10**(6): 1861-1870.
- Griffin, G. F. and M. H. Friedel (1984) Effects of fire on central Australian rangelands 1: fire and fuel characteristics and changes in herbage and nutrients. *Australian Journal of Ecology* **9**(4): 381-393.
- Guégan, J. F., S. Lek and T. Oberdorff (2001) Energy availability and habitat heterogeneity predict global riverine fish diversity. *Nature* **391**: 382-384.
- Hawkins, B. A. and E. E. Porter (2003) Does herbivore diversity depend on plant diversity? The case of California butterflies. *American Naturalist* **161**(1): 40-49.
- Hawkins, B. A. and E. E. Porter (2003) Water-energy balance and the geographic pattern of species richness of western Palearctic butterflies. *Ecological Entomology* **28**: 678-686.
- Hawkins, B. A., E. E. Porter and J. A. F. Diniz-Filho (2003) Productivity and history as predictors of the latitudinal diversity gradient of terrestrial birds. *Ecology* **84**(6): 1608-1623.
- HilleRisLambers, R., M. Rietkerk, F. v. d. Bosch, H. H. T. Prins and H. d. Kroon (2001) Vegetation Pattern Formation in Semi-Arid Grazing Systems. *Ecology* **82**(1): 50-61.
- Hodgkinson, K. C. (2002) Fire regimes in *Acacia* wooded landscapes: effects on functional processes and biological diversity. *Flammable Australia: The Fire Regimes and Biodiversity of a Continent*. R. A. Bradstock, J. E. Williams and A. M. Gill. Cambridge, Cambridge University Press.
- Holm, A. M., S. W. Cridland and M. L. Roderick (2003) The use of time-integrated NOAA NDVI data and rainfall to assess landscape degradation in the arid shrubland of Western Australia. *Remote Sensing of Environment* **85**: 145-158.
- Holm, A. M., W. A. Loneragan and M. A. Adams (2002) Do variations on a model of landscape function assist in interpreting the growth response of vegetation to rainfall in arid environments? *Journal of Arid Environments* **50**: 23-52.
- Holm, A. M., I. W. Watson, W. A. Loneragan and M. A. Adams (2003) Loss of patch-scale heterogeneity on primary productivity and rainfall-use efficiency in Western Australia. *Basic and Applied Ecology* **4**: 569-578.
- Jafari, R., M. M. Lewis and B. Ostendorf (2008) An image-based diversity index for assessing land degradation in an arid environment in South Australia. *Journal of Arid Environment* **72**: 1282-1283.
- James, C. D., J. Landsberg and S. R. Morton (1999) Provision of watering points in the Australian arid zone: a review of effects on biota. *Journal of Arid Environments* **41**(1): 87-121.
- Kelly, R. D. and B. H. Walker (1976) The effects of different forms of land use on the ecology of a semi-arid region in south-eastern Rhodesia. *The Journal of Ecology* **64**(2): 553-576.
- Landsberg, J., C. D. James, J. Maconochie, A. O. Nicholls, J. Stol and R. Tynan (2002) Scale-related effects of grazing on native plant communities in an arid rangeland region of South Australia  
doi:10.1046/j.1365-2664.2002.00719.x. *Journal of Applied Ecology* **39**(3): 427-444.
- Landsberg, J., C. D. James, S. R. Morton, W. J. Muller and J. Stol (2003) Abundance and composition of plant species along grazing gradients in Australian rangelands  
doi:10.1111/j.1365-2664.2003.00862.x. *Journal of Applied Ecology* **40**(6): 1008-1024.
- Le Houerou, H. N. (1984) Rain use efficiency: a unifying concept in arid-land ecology. *Journal of Arid Environments* **7**: 213-247.
- Lechmere-Oertel, R., G., R. Cowling, M. and G. I. H. Kerley (2005) Landscape dysfunction and reduced spatial heterogeneity in soil resources and fertility in semi-arid succulent thicket, South Africa  
doi:10.1111/j.1442-9993.2005.01495.x. *Austral Ecology* **30**(6): 615-624.
- Lewis, M. M. (1998) Numeric classification as an aid to spectral mapping of vegetation communities. *Plant Ecology* **136**: 133-149.

- Ludwig, J. A., R. W. Eager, G. N. Bastin, V. H. Chewings and A. C. Liedloff (2002) A leakiness index for assessing landscape function using remote sensing. *Landscape Ecology* **17**: 157-171.
- Ludwig, J. A., D. J. Tongway, G. N. Bastin and C. D. James (2004) Monitoring ecological indicators of rangeland functional integrity and their relation to biodiversity at local to regional scales. *Austral Ecology* **29**: 108-120.
- Ludwig, J. A., B. P. Wilcox, D. D. Breshears, D. J. Tongway and A. C. Imeson (2005) Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology* **86**(2): 288-297.
- Lyons, S. K. and M. R. Willig (2002) Species richness, latitude, and scale-sensitivity. *Ecology* **83**(1): 47-58.
- MacArthur, R. H. and E. O. Wilson (1967) *The Theory of Island Biogeography*. Princeton, NJ, Princeton University Press.
- McIntyre, S., K. M. Heard and T. G. Martin (2003) The relative importance of cattle grazing in subtropical grasslands: does it reduce or enhance plant biodiversity? *Journal of Applied Ecology* **40**: 445-457.
- McKenzie, N. L., A. A. Burbidge, A. Baynes, R. N. Brereton, C. R. Dickman, G. Gordon, L. A. Gibson, P. W. Menkhorst, A. C. Robinson, M. R. Williams and J. Woinarski (2007) Analysis of factors implicated in the recent decline of Australia's mammal fauna. *Journal of Biogeography* **34**: 597-611.
- Meadows, M. E. and M. T. Hoffman (2002) The nature, extent and causes of land degradation in South Africa: legacy of the past, lessons for the future? *Area* **34**(4): 428-437.
- Miller, T. F., D. J. Mladenoff and M. K. Clayton (2002) Old-growth northern hardwood forests: Spatial autocorrelation and patterns of understory vegetation. *Ecological Monographs* **72**(4): 487-503.
- Northrup, B. K. and J. R. Brown (1999) Spatial distribution of soil carbon in grazed woodlands of dry tropical Australia: tussock and inter-tussock scales. *Proceedings of the VI international rangeland congress*, Townsville, Australia.
- O'Brien, E. (2006) Biological relativity to water-energy dynamics. *Journal of Biogeography* **33**: 1868-1888.
- O'Brien, E. M. (1993) Climatic gradients in woody plant species richness: towards an explanation based on an analysis of Southern Africa's woody flora. *Journal of Biogeography* **20**: 181-198.
- O'Brien, E. M. (1998) Water-energy dynamics, climate, and prediction of woody plants species richness: an interim general model. *Journal of Biogeography* **25**: 379-398.
- O'Brien, E. M., R. Field and R. J. Whittaker (2000) Climatic gradients in woody plant (tree and shrub) diversity: water-energy dynamics, residual variation, and topography. *Oikos* **89**(3): 588-600.
- Ohmann, J. L. and T. A. Spies (1998) Regional gradient analysis and spatial pattern of woody plant communities of Oregon forests. *Ecological Monographs* **68**(2): 151-182.
- Palmer, M. W., P. G. Earls, B. W. Hoagland, P. S. White and T. Wohlgemuth (2002) Quantitative tools for perfecting species lists. *Environmetrics* **13**: 121-137.
- Rahbek, C. and G. R. Graves (2001) Multiscale assessment of patterns of avian species richness. *Proceedings of the National Academy of Science, USA* **98**: 4534-4539.
- Reynolds, J. F., R. A. Virginia and W. H. Schlesinger (1997) Defining functional types for models of desertification. *Plant Functional Types: their relevance to ecosystem properties and global change*. T. M. Smith, H. H. Shugart and F. I. Woodward. Cambridge, New York, Cambridge University Press: 195-216.
- Richerson, P. J. and K. L. Lum (1980) Patterns of plant species diversity in California: relation to weather and topography. *American Naturalist* **116**: 504-536.
- Rocchini, D., A. Chiarucci and S. A. Loiselle (2004) Testing the spectral variation hypothesis by using satellite multispectral images. *Acta Oecologica* **26**: 117-120.
- Sala, O. E., F. S. I. Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Sykes, B. H. Walker, M. Walker and D. H. Wall (2000) Global biodiversity scenarios for the year 2100. *Science* **287**: 1770-1774.

- Schlesinger, W. H., A. D. Abrahams, A. J. Parsons and J. Wainwright (1999) Nutrient losses in runoff from grassland and shrubland habitats in Southern New Mexico: I. rainfall simulation experiments. *Biogeochemistry* **45**(1): 21-34.
- Schlesinger, W. H. and A. M. Pilmanis (1998) Plant-soil interactions in deserts. *Biogeochemistry* **42**: 169-187.
- Shaver, G. R., K. J. Nadelhoffer and A. E. Giblin (1991) Biogeochemical diversity and element transport in a heterogeneous landscape, the north slope of Alaska. *Quantitative Methods in Landscape Ecology*. New York, Springer-Verlag.
- Simpson, G. G. (1964) Species densities of North American mammals. *Systematic Zoology* **13**(2): 57-73.
- Smith, A. P. and D. G. Quin (1996) Patterns and causes of extinction and decline in Australian conilurine rodents. *Biological Conservation* **77**(2-3): 246-267.
- Smyth, A. K., R. Brandle, A. Brook, V. H. Chewings, M. Fleming and J. L. Read (2007) Methods for identifying, selecting and interpreting indicators for assessing biodiversity condition in desert Australia, using the Stony Plains bioregion as a case study, Desert Knowledge Cooperative Research Centre.
- Snyman, H. A. (1997) The influence of range condition on the hydrological characteristics in semi-arid rangeland. The XVIII International Grassland Congress.
- Snyman, H. A. (1998) Dynamics and sustainable utilization of rangeland ecosystems in arid and semi-arid climates of southern Africa. *Journal of Arid Environments* **39**(4): 645-666.
- Snyman, H. A. and H. J. Fouché (1993) Estimating seasonal herbage production of a semi-arid grassland based on veld condition, rainfall, and evapotranspiration. *African Journal of Range and Forage Science* **10**(1): 21-24.
- Sparrow, A. D., M. H. Friedel and D. J. Tongway (2003) Degradation and recovery processes in arid grazing lands of central Australia. Part 3: implications at landscape scale. *Journal of Arid Environments* **55**: 349-360.
- Specht, R. L. and A. Specht (1999) Australian plant communities: dynamics of structure, growth and biodiversity, Kyodo, Singapore.
- Terborgh, J. (1973) On the notion of favorableness in plant ecology. *American Naturalist* **107**(956): 481-501.
- Thrash, I., G. K. Theron and J. d. P. Bothma (1993) Impact of water provision on herbaceous community composition in the Kruger National Park, South Africa. *African Journal of Range and Forage Science* **10**(1): 31-35.
- Thwaites, R. N. (2002) Airborne gamma-ray spectrometry in soil-landscape modelling for upland, erosional forestlands. 11th Australasian Remote Sensing and Photogrammetry Conference, Brisbane.
- Tobler, M. W., R. Cochard and P. J. Edwards (2003) The impact of cattle ranching on large-scale vegetation patterns in a coastal savanna in Tanzania. *Journal of Applied Ecology* **40**(3): 430-444.
- Todd, S. W. (2006) Gradients in vegetation cover, structure and species richness of Nama-Karoo shrublands in relation to distance from livestock watering points  
doi:10.1111/j.1365-2664.2006.01154.x. *Journal of Applied Ecology* **43**(2): 293-304.
- Tongway, D. J., A. D. Sparrow and M. H. Friedel (2003) Degradation and recovery processes in arid grazing lands of central Australia. Part 1: soil and land resources. *Journal of Arid Environments* **55**: 301-326.
- Tuomisto, H., A. D. Poulsen, K. Ruokolainen, R. C. Moran, C. Quintana, J. Celi and G. Canas (2003) Linking floristic patterns with soil heterogeneity and satellite imagery in Ecuadorian Amazonia. *Ecological Applications* **13**(2): 352-371.
- Turner, J. R. G. (2004) Explaining the global biodiversity gradient: energy, area, history and natural selection. *Basic and Applied Ecology* **5**: 435-448.
- Venevsky, S. and I. Venevskaia (2003) Large-scale energetic and landscape factors of vegetation diversity. *Ecology Letters* **6**(11): 1004-1016.
- Venevsky, S. and I. Venevskaia (2005) Hierarchical systematic conservation planning at the national level: Identifying national biodiversity hotspots using abiotic factors in Russia. *Biological Conservation* **124**: 235-251.



Verlinden, A. and R. Laamanen (2005) Long Term Fire Scar Monitoring with Remote Sensing in Northern Namibia: Relations Between Fire Frequency, Rainfall, Land Cover, Fire Management and Trees. *Environmental Monitoring and Assessment* **112**(1-3): 231.

Wainwright, J., A. J. Parsons and A. D. Abrahams (1999) Rainfall energy under creosotebush. *Journal of Arid Environments* **43**(2): 111-120.

Wessels, K. J., S. D. Prince, P. E. Frost and D. van Zyl (2003) Assessing the effects of human-induced land degradation in the former homelands of northern South Africa with a 1 km AVHRR NDVI time-series. *Remote Sensing of Environment* **91**: 47-67.

Wessels, K. J., S. D. Prince, P. E. Frost and D. van Zyl (2004) Assessing the effects of human-induced land degradation in the former homelands of northern South Africa with a 1 km AVHRR NDVI time-series. *Remote Sensing of Environment* **91**: 47-67.

Whittaker, R. H. (1972) Evolution and measurement of species diversity. *Taxon* **21**(2/3): 213-251.

Whittaker, R. J., K. J. Willis and R. Field (2003) Climatic-energetic explanations of diversity: a macroscopic perspective. *Macroecology: concepts and consequences*. T. M. Blackburn and K. J. Gaston. Oxford, Blackwell Science: 107-129.

Wilford, J. R., P. Bierwirth and M. A. Craig (1997) Application of airborne gamma-ray spectrometry in soil/regolith mapping and applied geomorphology. *AGSO Journal of Australian Geology & Geophysics* **17**(2): 201-216.

Willig, M. R., D. M. Kaufman and R. D. Stevens (2003) Latitudinal gradients of biodiversity: pattern, process, scale, and synthesis. *Annual Review of Ecology and Systematics* **34**: 273-309.

Wright, D. (1983) Species-energy theory: an extension of species-area theory. *Oikos* **41**: 496-506.

## Chapter 3: False-negative errors in a survey of persistent, highly-detectable vegetation species

Authors: Clarke, K., Lewis, M., Ostendorf, B.

Key words: *false-negative errors; biological surveys; presence-absence data; perennial vegetation*

### 3.1 Introduction

Fauna and flora surveys collect presence-absence data for a variety of reasons, including inventory surveying, ecological monitoring and change detection for environmental management. The data generated by these surveys informs environmental management and policy formation. However, errors within the data collected by these surveys are often overlooked or ignored. One such problem present in many fauna and flora surveys is the presence of false-negative errors; the failure to record a species which was actually present.

Several studies have examined false-negative errors in vegetation surveys and demonstrated that both common (Kéry *et al.* 2006) and rare (Alexander *et al.* 1997; Slade *et al.* 2003; Kéry *et al.* 2006; Regan *et al.* 2006) plant species are imperfectly detected with a single survey. These studies highlight the need for mark-recapture methods in vegetation surveys to minimise non-detection of un-common species.

However, we believe that false-negative error rates of common vegetation species are still frequently underestimated. Only one study (Kéry *et al.* 2006) has examined false-negative errors rates for common vegetation species, and that study examined only six species in the Swiss Alps. It is necessary to examine false-negative rates of common species in other regions to determine the breadth and severity of non-detection errors in conventional vegetation surveys.

Many conventional flora surveys assume that the majority of large persistent vegetation species are highly or perfectly detected. We hypothesise that this assumption results from underestimating the difficulty of detecting even large vegetation species consistently and that this underestimation is a result of sedentary nature of vegetation: it is assumed that if a plant is within the quadrat its detection is largely a function of plant size, surveyor persistence and difficulty of identification (ie. presence of fruiting bodies, distinctive habit, etc.). Methods have been developed for minimising detection error due to these sources, such as expert judgement of necessary search time and taking of voucher specimens. Finally, many vegetation communities are not climax communities, or contain many annual or ephemeral species. Thus, when species lists vary on repeat visits to a site, or from site to site within one land system, the variation is considered natural and expected.

At least two broad scale conventional vegetation surveys are conducted in South Australia; the Biological Survey of South Australia; and the South Australian Pastoral Lease Assessment. False-negative rates have not been quantified for these surveys, even though moderate levels of false-negative errors have been demonstrated to significantly effect ecological measures such as site occupancy and hence range and rarity (Tyre *et al.* 2003). It is important that false-negative rates in these studies are examined so that their effects may be counteracted.

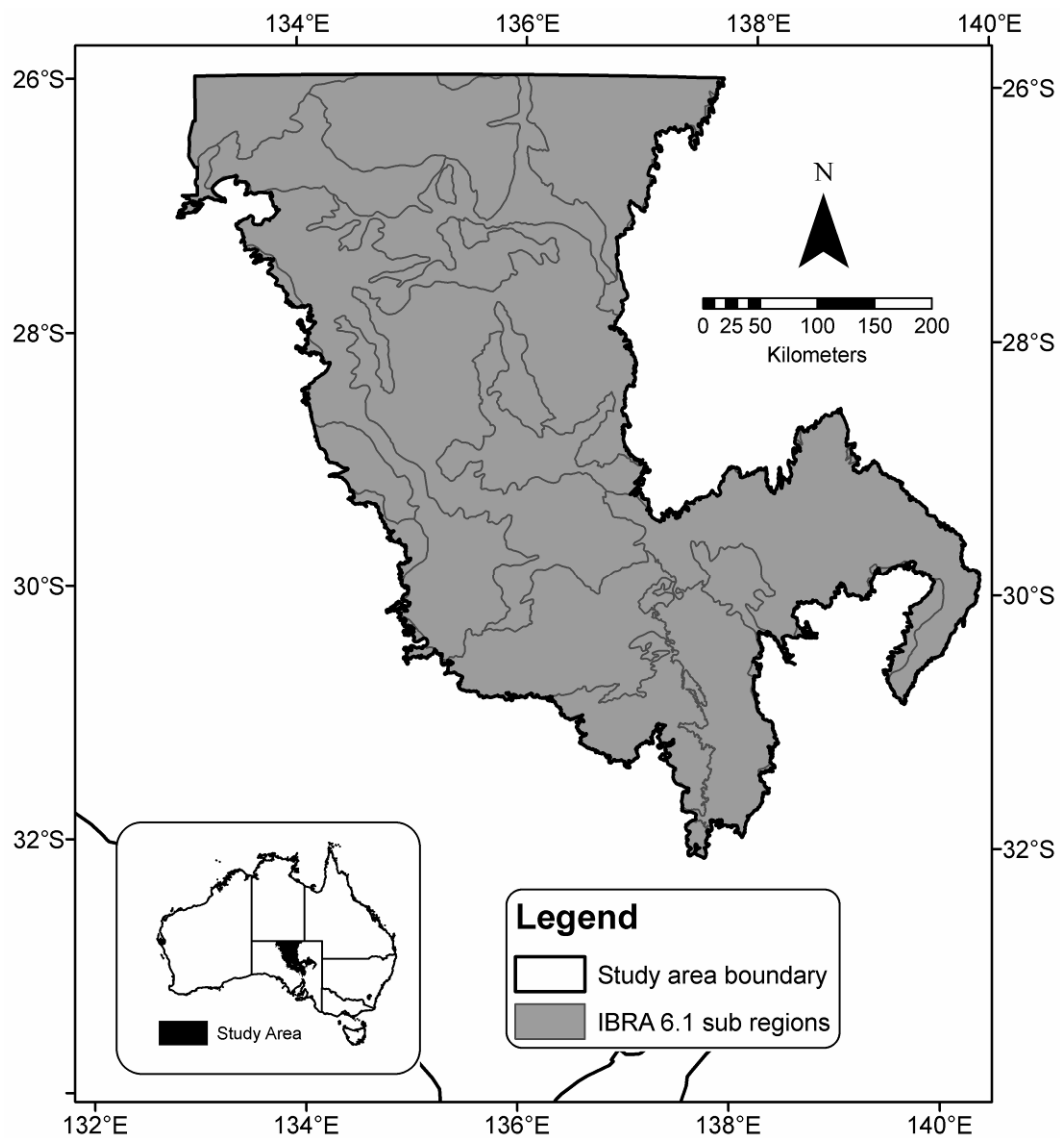
In this paper we aim to evaluate the extent of false-negative errors in the Biological Survey of South Australia. We first examine data from several sites surveyed by the Biological Survey of South Australia, controlling for confounding variables, and produce a conservative assessment of false-negative errors for that survey. We then apply the lessons of that analysis to predict the extent of non-detection errors in the South Australian Pastoral Lease Assessment.

## **3.2 Methodology**

### **3.2.1 Study Area**

The study was conducted in central Australia and stretches from the top of Spencer Gulf in South Australia to the Northern Territory border (Figure 16). Specifically the study contains the entire Stony Plains region, as defined in the Interim Biogeographic

Regionalisation of Australia (IBRA) 6.1 as well as other adjacent IBRA sub-regions. Average annual rainfall across the area ranges from approximately 300 mm per annum in the south to 100 mm per annum in the north. While this area is very large (approximately 210,000 km<sup>2</sup>) it contains little geographic variation, and the majority of the area comprises flat or gently sloping plains with few shallow ephemeral watercourses. The dominant vegetation cover is chenopod shrubland, although there are significant areas of tall shrubland and low open woodland with grass or chenopod understorey (Laut *et al.* 1977).



**Figure 16.** Study area; Interim Biogeographic Regionalisation of Australia (IBRA) sub-regions displayed within study area.

Throughout the area the dominant land use is pastoral grazing of sheep in the south and cattle in the north. However, the low rainfall of the region provides few natural watering points for livestock. To increase the proportion of the landscape usable by livestock many artificial water points have been established.

### 3.2.2 Survey data

We analysed data from two conventional vegetation surveys: the Department for Environment and Heritage's Biological Survey of South Australia (BSSA); and the Department of Water Land and Biodiversity Conservation's South Australian Pastoral Lease Assessment (SAPLA). The majority of the data were collected over fourteen years, from 1990 to 2003. Any data collected outside this period was excluded to ensure both surveys covered the same time span. The two surveys have different goals and therefore different data collection methodologies and site selection biases.

The aim of the BSSA is to create an inventory of native species and therefore sites are generally chosen in areas less disturbed by grazing. A botanical expert participates in all surveys, and voucher specimens are collected for species not identified on site. The plant inventory is conducted in square quadrats of one hectare, or an equivalent rectangular area if placed in elongated vegetation communities. Vegetation surveys are usually only conducted once per site, although four BSSA sites were visited twice yearly for 8 years as part of a small-mammal monitoring program. These four sites were marked with photopoint pegs to, to aid re-location, and both fauna and flora surveys were conducted on each visit to these four sites.

Prior to analysis, all species names in the survey data were checked for currency and consistency and updated where necessary. With the aid of advice from the South Australian Herbarium, species were classified as either perennial or ephemeral, where perennials were defined as plants with an expected lifespan of three or more years in the study area. All ephemeral plant species were excluded from the analysis for two reasons. Firstly, most ephemeral species are physically small making them easily overlooked even when present at a site. Secondly, ephemeral plant populations vary greatly with preceding rain and were likely to confound the generation of species richness estimates.

### 3.2.3 False-negative analysis

Our aim was to test the assumption that vegetation species are highly or perfectly detected by conventional vegetation surveys. To this end we required sites with repeat surveys over a relatively short time (MacKenzie *et al.* 2002; Tyre *et al.* 2003; Gu and Swihart 2004).

While many SAPLA sites were surveyed on multiple occasions, the majority of these repeat surveys were separated by one or more years. In addition, all SAPLA sites are placed within the piosphere, and therefore changes in stocking rate could result in real changes in the species composition of a site and confound our assessment of false-negative errors. Hence, we start our evaluation of false-negative errors in conventional vegetation surveys with the aforementioned frequently re-surveyed BSSA sites.

#### **Biological Survey of South Australia**

Four Biological Survey of South Australia sites were visited twice yearly for eight years as part of a small mammal monitoring program. Although the monitoring program focussed primarily on small mammal monitoring, characterising land condition was an integral part of that monitoring. Thorough vegetation surveys were conducted during each site survey in order to characterise land condition. It is these data that were analysed for false-negative errors. Detailed records of survey history allowed analysis of species observations made by different surveyors.

Two different vegetation survey methodologies were conducted at the four sites, and the surveys were conducted by up to six different principal surveyors. The first three surveys at sites 11031, 10478 and 10294, and the first four surveys at site 9599 were conducted over a 1ha area according to the standard Biological Survey methodology. Additionally, these surveys were all conducted by one principal surveyor, experienced in rangeland vegetation surveys, referred to hereafter as surveyor A. Subsequent visits were conducted by several different principal surveyors (surveyors B, C, D, E and F) using a non-standard vegetation survey methodology over a larger area: 8ha for sites 9599 and 11031; and 4ha for sites 10478 and 10294.

The data collected by surveyors A, B and C were analysed to determine whether any false-negative errors were made, and if so, how many. In the context of this study, a false-

negative error was defined as the failure to record a species at a site during surveys subsequent to the visit on which it was first recorded. The data collected by surveyors D, E and F were not analysed because each surveyor conducted only one survey at each site.

The analysis attempts to control for two potentially confounding factors. Firstly, since our aim was to extract a conservative estimate of false-negative errors. Hence, we only included species described by BSSA staff as highly-detectable, persistent and easily identifiable. This resulted in a list of one perennial grass and seven woody-perennial shrub species. To emphasise the point, these species are perennial and therefore less likely to be influenced by short-term rainfall events.

Secondly, observer skill can play a significant role in determining the number of species recorded. A less skilled observer may fail to notice the minor taxonomic difference between two species in a genus, artificially reducing the number of species recorded. To control for observer skill, the above definition of false-negative error was applied separately to the data collected by each surveyor. Hence, the recording of a species presence by one surveyor had no bearing on the evaluation of false-negative error rate for another surveyor. Additionally, by analysing the data collected by each surveyor separately each analysis covers a shorter period of time which increases the validity of the assumption of demographic closure.

After examining false-negative error rates at each site, we estimate the detection probability of each species across all four BSSA sites. This was calculated as a proportion of the number of times a species was detected out of the number of potential detections. It is reasonable to calculate this statistic for the study as a whole, rather than by surveyor, because observer skill has already been controlled for.

### **3.3 Results**

#### **3.3.1 Biological Survey of South Australia**

The data collected by surveyors A, B and C for sites 9599, 11031, 10478 and 10294 are presented in Tables 1, 2, 3 and 4 respectively. Where an observer failed to detect a plant previously recorded by them at that site they were considered to have made a false-negative error.

### Site 9599

Five highly-detectable, woody-perennials were recorded at least once at site 9599 (Table 3). Surveyors B and C only visited site 9599 once each; hence their records are not displayed in Table 3. Surveyor A made a total of five false-negative errors over four visits to site 9599. Of particular note is the false-negative record on visit two, for *Atriplex nummularia*, a distinctive, medium-to-large, perennial salt bush.

### Site 11031

Only two highly-detectable woody-perennial vegetation species were recorded at site 11031 (Table 4). Surveyor A made only one false-negative error over three surveys, failing to detect *Frankenia serpyllifolia*, while surveyor B made three false-negative errors in four surveys, all failing to detect the one species, *Astrebla pectinata*. Surveyor C did not record either of these species in three surveys.

**Table 3. False-negative errors at BSSA site 9599**

Visit Number	1	2	3	4
<i>Abutilon fraseri</i>	x	fn	fn	fn
<i>Abutilon halophilum</i>				x
<i>Astrebla pectinata</i>	x	x	x	x
<i>Atriplex nummularia</i>	x	fn	x	x
<i>Frankenia serpyllifolia</i>	x	fn	fn	x
Surveyor	A	A	A	A
Area surveyed (ha)	1	1	1	1

† x indicates the species was recorded at the site on that date; **fn** indicates a false-negative error, i.e. the surveyor failed to detect and record a species which was present at the site.

**NB:** Surveyors B and C only visited this site once, hence their records are not displayed.

Site 11031 is situated in a cracking-clay depression, and supported almost no vegetation when completely dry. The site was reasonably wet from visit 1 to 7 inclusive, and very dry from visit 8 onwards. This possibly explains the failure of Surveyor C to record any of the species highly-detectable woody-perennial in their three visits.

Three specific notes on site placement should be made here. Site 11031 was situated in a cracking-clay depression, and supported almost no vegetation when completely dry. This site was wetter from visit one to seven, and significantly dryer from visit eight onwards. The change from wet to dry conditions occurred at the same point as the change from surveyor B to C, and is therefore unlikely to have any influence on the analysis, and



certainly not on the results of surveyor A. Sites 10478 and 10294 are both situated within 2km of a stock watering point, hence within the piosphere. Both sites experienced consistent, mild grazing pressure over the study period.

**Table 4. False-negative errors at BSSA site 11031**

Visit Number	1	2	3	4	5	6	7	*	9	10	11
<i>Astrebla pectinata</i>				x	fn	fn	fn				
<i>Frankenia serpyllifolia</i>		x	fn								
Surveyor	A	A	A	B	B	B	B		C	C	C
Area surveyed (ha)	1	1	1	8	8	8	8		8	8	8

\*The principal surveyor for visit 8 was not recorded; this visit is excluded from analysis.

† x indicates the species was recorded at the site on that date; **fn** indicates a false-negative error, i.e. the surveyor failed to detect and record a species which was present at the site.

### Site 10478

At site 10478 (Table 5), seven highly-detectable woody-perennial vegetation species were recorded, the most for any of the Biological Survey sites. This site experienced mild grazing pressure over the study period.

Surveyor A made three false-negative errors in three surveys, surveyor B made seven false-negative errors in four surveys, and surveyor C made 12 false-negative errors in six visits. Of particular note is the detection, non-detection and re-detection of *Abutilon halophilum* by surveyors A and B.

**Table 5. False-negative errors at BSSA site 10478**

Visit Number	1	2	3	4	5	6	7	8	9	10	11	12	13
<i>Abutilon halophilum</i>	x	fn	x	x	fn	x	fn			x	fn	fn	fn
<i>Astrebla pectinata</i>	x	x	x	x	x	x	fn		x	x	x	x	x
<i>Atriplex nummularia</i>	x	x	x	x	x	x	x	x	x	x	x	x	x
<i>Chenopodium auricomum</i>	x	fn	fn					x	fn	fn	fn	fn	fn
<i>Maireana aphylla</i>					x	x	fn		x	fn	fn	fn	fn
<i>Sclerostegia medullosa</i>				x	fn	fn	fn						
Surveyor	A	A	A	B	B	B	B	C	C	C	C	C	C
Area surveyed (ha)	1	1	1	4	4	4	4	4	4	4	4	4	4

† x indicates the species was recorded at the site on that date; **fn** indicates a false-negative error, i.e. the surveyor failed to detect and record a species which was present at the site.

### Site 10294

Five highly-detectable woody-perennial vegetation species were recorded at Site 10294 (Table 6). As with site 10478, site 10294 experienced consistent, mild grazing pressure over the study period.

Surveyor A made two false-negative errors in three surveys, surveyor B made four false-negative errors in four surveys and surveyor C made seven false-negative errors in six surveys. All three surveyors recorded, then failed to record, and then recorded again at least one species: Surveyor A and *Astrebla pectinata* and *Frankenia serpyllifolia*; Surveyor B and *Abutilon halophilum*; and Surveyor C and *Frankenia serpyllifolia* and *Sclerostegia medulla*.

**Table 6. False-negative errors at BSSA site 10294**

Visit Number	1	2	3	4	5	6	7	8	9	10	11	12	13
<i>Abutilon halophilum</i>	x	x	x	x	fn	x	x						x
<i>Astrebla pectinata</i>	x	fn	x	x	x	x	x			x	x	x	x
<i>Atriplex nummularia</i>	x	x	x	x	x	x	x	x	x	x	x	x	x
<i>Frankenia serpyllifolia</i>	x	fn	x	x	fn	fn	fn	x	fn	fn	fn	fn	x
<i>Sclerostegia medullosa</i>				x	x	x	x	x	fn	fn	fn	x	x
Surveyor	A	A	A	B	B	B	B	C	C	C	C	C	C
Area surveyed (ha)	1	1	1	4	4	4	4	4	4	4	4	4	4

† x indicates the species was recorded at the site on that date; **fn** indicates a false-negative error, i.e. the surveyor failed to detect and record a species which was present at the site.

**NB:** *Sclerostegia tenuis* was only recorded on visits 8 and 16, on which *Sclerostegia medullosa* was recorded as absent. Because these two species are easily confused, and were never recorded together at the site we resolved that records of *Sclerostegia tenuis* were misidentifications of *Sclerostegia medullosa*, and re-classified them as such.

### Detection Probability

Examining species detection probabilities across all sites, several species stand out for their frequent non-detection by surveyors (Table 7). Four species have detection probabilities less than 0.5: *Frankenia serpyllifolia*, *Maireana aphylla*, *Abutilon fraseri*, and *Chenopodium auricomum*. However, the very low detection probability of *Abutilon fraseri* is based on only four possible detections. Of particular note is the relatively-low detection probability of 0.57 for *Sclerostegia medullosa*, a distinctive and relatively un-palatable plant (Kutsche and Lay 2003).

Interestingly, even *Atriplex nummularia* has a detection probability of less than 1. This large distinctive species (Jessop 1978) was present at three of the four sites, and was detected on all but one survey.

**Table 7. Species detection probabilities across all BSSA sites**

Species	Possible Detections	Detections	Detection Probability
<i>Abutilon fraseri</i>	4	1	<b>0.25</b>
<i>Abutilon halophilum</i>	20	13	<b>0.65</b>
<i>Astrebla pectinata</i>	31	26	<b>0.84</b>
<i>Atriplex nummularia</i>	30	29	<b>0.97</b>
<i>Chenopodium auricomum</i>	9	2	<b>0.22</b>
<i>Frankenia serpyllifolia</i>	19	8	<b>0.42</b>
<i>Maireana aphylla</i>	8	3	<b>0.38</b>
<i>Sclerostegia medullosa</i>	14	8	<b>0.57</b>

### 3.4 Discussion

We believe the conditions we set to evaluate false-negative error frequencies in this study were generous: the study was limited to the most visible, persistent, easily-identifiable perennial vegetation species. Many more species than the ones used in this study were considered by an arid-land botanist to be perennial in the study area but were excluded on other grounds. For instance, some are cryptic without fruiting bodies while others can be grazed to an extent which hinders identification.

Furthermore, we have considered the data collected by each surveyor in isolation: detection of a species by one surveyor and the subsequent non-detection of that species by another surveyor was not considered a false-negative error. This approach was necessary when considering the results of surveyor A in relation to the results of the other surveyors, as surveyor A employed a different survey methodology to the other surveyors. However, surveyors B and C used the same methodology and should have surveyed the same area. It could therefore be argued that it would have been reasonable to combine the data collected by surveyors B and C for evaluation. Conducting the analysis under these conditions would have resulted in many more false-negative errors.

Finally, the assumption of demographic closure is critical to this analysis; two steps were taken to minimise the violation of this assumption. Firstly the analysis was limited to persistent perennial vegetation species, and secondly through controlling for differences in

surveyors the dataset was analysed in blocks of no more than six years. Together these minimised the probability of local colonisation or extinction over the time span of the study.

Even after limiting this study to the most easily-detected perennial vegetation species, and controlling for observer skill, we revealed frequent false-negative errors by all surveyors, at all sites, for all species. Thus, we have demonstrated in the study area that even highly detectable vegetation species often have detection probabilities significantly less than one.

This result is interesting in the context of the collecting study: these data were the only measure of land condition at these sites, an important part of the small-mammal monitoring program for which this data was collected. Any measure of land-condition extracted from this vegetation survey data which did not account for the vegetation false-negative detection rates would be seriously flawed.

While our analysis was conducted on data collected by the Biological Survey of South Australia (BSSA) in arid South Australia, we believe our findings have broader implications. Surveyor A conducted the vegetation survey with the standard BSSA methodology over the standard 1ha area. Therefore, it seems reasonable to conclude that all BSSA flora surveys in the region contain frequent false-negative errors. These frequent false-negative errors will limit the ends to which BSSA data can be put, and without correction will invalidate analyses and conclusions drawn from these data.

Surveyors B and C utilised a non-standard survey methodology, so we can only draw general conclusions from their performance. To summarise, these two trained surveyors utilised one survey methodology to conduct repeat vegetation surveys of three sites. On all but one occasion, when false-negative errors were possible, these surveyors recorded false-negative errors.

Examining the results of surveyors A, B and C together, we see that three experienced surveyors employing two different survey methodologies all frequently failed to detect species present.

### 3.4.1 Ramifications for similar vegetation surveys

In addition to BSSA, there are other broad scale vegetation surveys in this region that generate vegetation data which may be used for similar purposes, and which may also suffer from false-negative errors.

One such survey is the South Australian Pastoral Lease Assessment (SAPLA). This survey is designed to monitor the effect of livestock grazing on land condition. Hence sample sites are placed within the piosphere but not in the immediate vicinity of the water point. In the sheep grazing properties of the southern study area, SAPLA monitoring points are located approximately 1.5 km from watering points. In the cattle grazing properties in the north of the study area SAPLA monitoring points are located approximately 3 km from watering points. Because SAPLA sites are located within stock piospheres they are more likely to be degraded than BSSA sites. Unlike the BSSA, no botanical expert is involved with SAPLA surveys in the field. SAPLA staff conduct the surveys and attempt to identify all vegetation species, while voucher specimens of any unknown species are collected for later identification. An area of 100 to 200 metres radius is surveyed at each site. Because the SAPLA is designed to monitor change in range condition, sites are revisited at regular intervals. There were 1185 SAPLA sites within the study area.

It was not possible to analyse the SAPLA data in as much detail as the BSSA sites, due to the lack of detailed site-scale information on surveyor identity and site history. However, we argue that it is possible, and reasonable, to estimate the magnitude of false-negative errors in SAPLA data without such an analysis. There are three key differences between the BSSA and SAPLA surveys: area surveyed; botanical expertise; and expected grazing pressure.

Firstly, the area surveyed by the BSSA is usually one hectare. In the BSSA sites examined in this paper surveyor A conducted a standard survey over one hectare, while surveyors B and C conducted surveys over four or eight hectares, depending on site. By contrast, SAPLA sites vary between one and two hundred meters in radius, or approximately three to 12 hectares. Assuming a fixed search time, rates of false-negative errors are likely to increase with surveyed area, by allowing the surveyor's less time per unit area. However,

since the amount of time allocated to SAPLA site surveys is un-known, it is not possible to estimate the effect of quadrat size differences on false-negative error rates.

Secondly, the two surveys differ in their declared level of surveyor botanical expertise. Standard BSSA surveys include a botanical expert, while SAPLA surveys are conducted by range monitoring professionals with some botanical experience. While the BSSA sites examined in this study were not standard surveys, we did limit this analysis to the most visible, persistent, easily-identifiable perennial vegetation species. Therefore, the difference in botanical expertise is likely to result in similar or higher rates of false-negative errors in SAPLA surveys.

Finally, BSSA and SAPLA sites are subject to different levels of grazing. The BSSA is an inventory survey, and therefore sites are biased away from heavily grazed areas, whereas the SAPLA is a range condition monitoring survey, and sites are placed within the piosphere and therefore frequently grazed. Moderate to heavy grazing is likely to reduce identifiability of plant species by removing fruiting bodies, damaging or reducing visible leaves, and at worst reducing plants to un-identifiable stubs. Therefore, we expect the greater grazing pressure to increase false-negative error rates at SAPLA sites.

Thus, all differences between the BSSA and SAPLA would lead us to expect higher rates of false-negative errors in the SAPLA. False-negative errors in the SAPLA data are a serious concern and may confound measures of range condition. For instance, the apparent loss of one or more species from a site may simply be the result of a false-negative error, or could be a serious result of grazing induced degradation.

### 3.4.2 Wider implications

Beyond the two vegetation surveys which contributed data to this study, our analysis has implications for vegetation surveying in general. The results support the findings of Kéry *et al.* (2006), that a single site-survey may miss some of the most detectable vegetation species, and will probably miss an even greater proportion of the less detectable vegetation species. Hence we recommend that vegetation surveys adopt measures to gauge the detectability of species, and to correct for false-negative errors, as is already done in some fauna surveys.

The problem of false-negative errors in both flora and fauna surveying has received some attention in recent years, and the same counter-measures are recommended. The work of Gu and Swihart (2004) and MacKenzie *et al.* (2002) focused on correcting for imperfect detection of small mammals and amphibians respectively. Both studies highlighted the need for multiple sampling occasions at each site in estimating and correcting for non-detection. Modelling by Tyre *et al.* (2003) determined that three repeat visits to a site was the minimum required to remove bias caused by moderate levels of false-negative error in fauna surveying, and six visits were required to effectively eliminate the effects of false negative errors. Finally, Kéry *et al.* (2006) estimated the number of site visits required to detect high and low detectability species with 95% confidence as two and four visits respectively.

It is clear that caution is necessary when interpreting the results of conventional vegetation surveys. We have demonstrated that the implicit assumption of perfect detection in many vegetation surveys is invalid. Without multiple sampling occasions at each site, or some other method for estimating species detectability, the results of vegetation surveys may contain serious biases and inaccuracies. Without correcting for these biases and inaccuracies ecological measures such as alpha- or beta diversity will be flawed and the necessity of management response to an apparent species loss will be unclear.

### **3.5 Acknowledgements**

This work was supported by funding from the Australian Desert Knowledge Cooperative Research Centre (DKCRC) and the University of Adelaide. In addition, two DKCRC partners provided the vegetation survey data that made this analysis possible, the South Australian Department for Environment and Heritage supplied the Biological Survey of South Australia data; and the South Australian Department of Water, Land and Biodiversity Conservation supplied the South Australian Pastoral Lease Assessment data. Special thanks to the staff of the Biological Survey of South Australia, especially Robert Brandle, for the site and survey specific information which made this detailed analysis possible. Also, special thanks to Helen Vonow of the South Australian State Herbarium for help in classifying and updating vegetation species lists.

### 3.6 References

- Alexander, H. M., N. A. Slade and D. W. Kettle (1997) Application of mark-recapture models to estimation of the population size of plants. *Ecology* 78(4): 1230-1237.
- Gu, W. and R. K. Swihart (2004) Absent or undetected? Effects of non-detection of species occurrence on wildlife-habitat models. *Biological Conservation* 116(2): 195-203.
- Jessop, J. P. (1978) *Flora of South Australia*. Adelaide, Government Printer.
- Kéry, M., J. Spillmann, C. Truong and R. Holderegger (2006) How biased are estimates of extinction probability in revisitation studies? *Journal of Ecology* 94: 980-986.
- Kutsche, F. and B. Lay (2003) *Field guide to the plants of Outback South Australia*, Openbook Print.
- Laut, P., G. Keig, M. Lazarides, E. Loffler, C. Margules, R. M. Scott and M. E. Sullivan (1977) *Environments of South Australia: Province 8, Northern Arid*. Canberra, Division of Land Use Research, CSIRO: 116-168.
- MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle and C. A. Langtimm (2002) Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83(8): 2248-2255.
- Regan, T., M. A. McCarthy, P. W. Baxter, F. D. Panetta and H. P. Possingham (2006) Optimal eradication: when to stop looking for an invasive plant. *Ecology Letters* 9(7): 759-766.
- Slade, N. A., H. M. Alexander and D. W. Kettle (2003) Estimation of population size and probabilities of survival and detection in Mead's milkweed. *Ecology* 84(3): 791-797.
- Tyre, A. J., B. Tenhumberg, S. A. Field, D. Niejalke, K. Parris and H. P. Possingham (2003) Improving precision and reducing bias in biological surveys: estimating false-negative error rates. *Ecological Applications* 13(6): 1790-1801.