AN EXAMINATION OF GROUND COVER
FOR
THE ASSESSMENT OF ECOSYSTEM CONDITION
IN
THE SILVER-LEAVED IRONBARK WOODLANDS OF
THE DESERT UPLANDS

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Abstract

The thesis examines the ecosystem condition of silver-leaved ironbark woodlands within a spatial and temporal context. This relatively homogenous ecosystem, which covers over one million hectares within the Desert Uplands bioregion in Central Queensland, provides the ideal environment in which to examine relationships of condition variables across scales from site to landscape. The results establish that links exist between grazing pressures on ground cover and a number of ecosystem condition variables. The remotely sensed Ground Cover Index (GCI) was found to have significant relationships with field measurements. The current ecosystem condition assessment method using the Ground Cover Disturbance Index (derived from the GCI) is assessed and improvements are suggested that allow climatic effects to be accounted for to ensure the index better reflects ecosystem functionality. The GCI, together with field ground cover measurements are explored to assess ecosystem condition with respect to field measurements of avian and plant diversity. The GCI was found to have a significant relationship with perennial grass cover. Perennial grass abundance and richness were found to have significant relationships with ground cover and are recommended as indicators of ecosystem condition. Avian groups as indicators were less effective with both strong positive and negative relationships with ground cover existing within the groups. The bird habitat assemblage group shows promise for further investigation for the assessment of ecosystem condition. Finally, linear mixed effect modelling established the relationship between ground cover and climate and grazing explanatory variables; the effects of each variable were qualified and the predictive power of the model was validated against ground cover measurements.

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Dedicated to my good friend

I dedicate this work to my good friend Andrea Lingard who loved this landscape as much as I. Let’s hope and pray that mining of the Galilee Basin does not destroy these wonderful woodlands.

Certificate of Authorship and Originality of Thesis

The work contained in this thesis has not been previously submitted either in whole or in part for a degree at Central Queensland University or any other tertiary institution. To the best of my knowledge and belief, the material presented in this thesis is original except where due reference is made in the text.

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This article is my own work – I collected and analysed the data. My supervisors assisted with the interpretation of the results and conclusions drawn.

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Copyright statement

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Chapter One

1.1 Introduction

In the past decade there has been substantial effort to detail and recommend comprehensive monitoring frameworks to assess and monitor the trend of biodiversity condition in the rangelands of Australia (Smyth and James 2004; Fisher and Kutt 2007; Bastin and ACRIS management committee 2008; Bastin *et al.* 2009; Kutt *et al*. 2009; Smyth *et al.* 2009; Eyre *et al.* 2011). This study, while not innovative in biological assessment techniques, is a unique attempt to integrate biological knowledge from different spatial scales across a temporal period of two decades. This study is motivated by the question of how to quantify the ecosystem condition of rangeland ecosystems in a holistic manner, taking into consideration attributes at the site level in order to incorporate them into landscape quantification. The study endeavours to establish an integrated methodology whereby the spatial and temporal dimensions of rangeland woodlands are incorporated. A methodology is developed with which the effects of pastoral activities can be measured and monitored from the scale of site to that of paddock, to property and, ultimately, to landscape scale thus capturing the condition of the ecosystem in its entirety.

The silver-leaved ironbark woodlands of the Desert Uplands bioregion in Queensland cover over a million hectares and as such form the most widespread ecosystem with grazing production values for the region (see Figure 1.1). Human management of this ecosystem is seen as an integral component of these woodlands as they have been grazed for over 150 years. Prior to pastoral settlement there was an aboriginal presence for over 40,000 years that had long-term ecological consequences of their landscape burning and hunting practises (Flannery 1994). While the woodlands may not contain all of their original ecological components post-European settlement, this study found that the ecosystem supports over 190 plant species and 110 bird species (see Appendices A and C) and have limited disturbance.

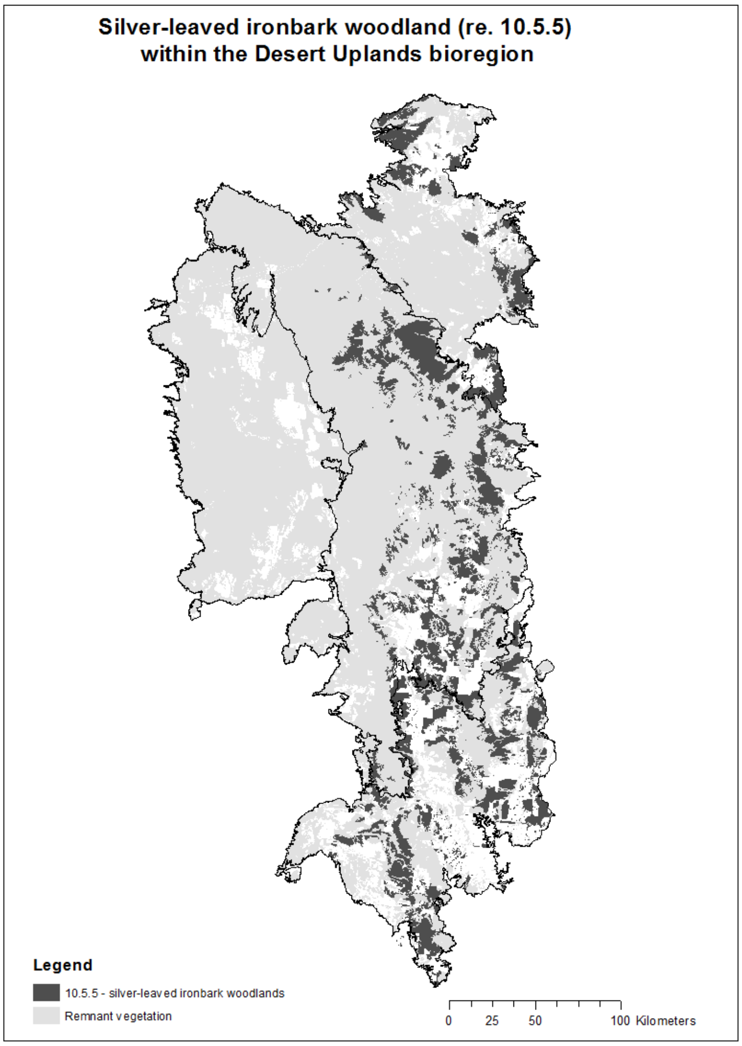


Figure 1.1 The distribution of the silver-leaved ironbark woodlands set within the context of remnant vegetation in the Desert Uplands bioregion.

This large and relatively homogeneous ecosystem provides the ideal environment in which to explore the relationships that inter-connect this landscape from the site to the whole region. Ground cover measurements are used as the context within which to investigate relationships between scales, and the potential to use remote sensing to assess landscape condition and trend is tested. The study was conducted within the remnant area of the ecosystem where the projective foliage cover was less than 20%, as this is the cover at which accurate measurement of ground cover from Landsat imagery is possible (see Figure 1.1).

This extensive woodlands ecosystem in the Desert Uplands provides essential wildlife habitat while, at the same time, it is a fundamental component of grazing enterprises in the bioregion. Overgrazing which results in trampling and defoliation can lead to land degradation which is expressed in a loss of soil moisture storage capacity and nutrient availability (Pickup *et al.* 1994), and is likely to have negative consequences for both production and conservation outcomes.

Rangelands are a valuable resource for both biodiversity maintenance and primary production. As human pressures increase, it is increasingly important for government and research agencies to monitor and assess the condition of these landscapes. A multi-spectral remotely sensed ground cover index allows exploration of the variations of ground cover across the ecosystem's distribution in one time frame and also at the one location across 21 years.

In order for landholders and resource managers to be able to ascertain the impacts of their grazing management on the ecological components of their properties they need to know what the ecosystem condition is: - at the site, paddock, property and landscape scale. Landholders need to know what the impacts of their grazing management are and the wider property and landscape ramifications of their actions. An understanding of both the drivers of the decline in condition or improvement, and the indications of either change in condition are necessary.

There is extensive scientific work on the description and quantification of key ecological condition attributes of eucalypt woodlands such as regeneration of canopy species, litter, tree dieback, presence of weeds, presence of exotic species, native plant species diversity and presence of tree hollows (Parkes 2004; Oliver 2003; Eyre *et al.* 2005; and McIntyre 2002). Current site assessment methodologies do not allow for paddock, property or indeed landscape attributes; however they consider the impacts of the surrounding landscape matrix on the site by assessing landscape attributes such as the size of the remnant area the site falls within, fragmentation, distance from water and isolation. A reliable ecosystem condition index that could be ascribed by pixel using remote sensing methods would provide a consistent, repeatable and time efficient assessment tool for widespread ecosystems.

Challenges exist in separating the relatively local changes in grazing management from the broad spatial and temporal patterns, such as the landscape mosaic and long-term weather patterns (Fensham and Fairfax 2008; Price *et al*. 2010; Reside *et al.* 2010; Scarth 2010). It is difficult to separate the grazing impacts on ecosystem condition from those caused by inter-annual variability and trends in rainfall (Prince *et al*. 1998; Wessels *et al*. 2007). Indeed, it is hard to distinguish the cumulative effects of grazing from long-term effects of climate (Pringle and Landsberg 2004). However, by examining the recovery of ecosystems from disturbance, it can be deduced that ecosystems with high integrity should be relatively resilient to environmental change or stresses and should be able to recover to their original condition after a perturbation (Holling 1973; Lavorel and Garnier 2002). A resilient system returns to the reference state following disturbance (De Angelis *et al.* 1989).

Following the State and Transition concept, environmental drivers and land use can cause semi-arid woodlands to switch between several vegetation states (Westoby *et al.* 1989). These states can broadly be categorised into a productive and desirable perennial grass dominated state and several degraded states dominated by annual vegetation, woody vegetation or bare ground (Westoby *et al*. 1989; Reynolds *et al*. 2007). In response to grazing, semi-arid woodlands can exhibit pronounced thresholds as the degradation process is non-linear and difficult to anticipate (Jeltsch *et al.* 2000; Vetter 2005; Gillson and Hoffman 2007).

One key aim of this study was to determine whether ground cover and a remotely sensed index of ground cover can be an effective indicator of ecosystem condition at a landscape scale. A second aim is to determine if modelling of a two decadal dataset of ground cover indices and grazing variables can distinguish the effects of grazing from rainfall effects on ground cover and provide a tool with which to predict the outcomes of changes in the combinations of both rainfall and grazing variables.

1.2 Methodology

This study examines the use of ground cover as surrogate for ecosystem condition at the site and landscape scale. A ground cover index derived from remotely sensed data (Scarth *et al.* 2006) is used as a measurement of ground cover (or inversely bare ground as in Chapter Three) across the ecosystem's spatial extent and over the years 1988 to 2008. The ground cover measurements from the index are correlated to the actual ground cover measurements taken at 91 sites in May and October 2007, which is close to actual date of capture of the Landsat TM satellite imagery for that year. Through categorisation of the ground cover measurements of the 91 sites and a further 300 randomly located sites, we can extrapolate this data to explain the overall ground cover as a surrogate of ecosystem condition of the ecosystem at a landscape scale. Measurements of paddock size, distance to water, areas of more and less palatable land types in paddock, and fire scars were taken to assess whether these variables explained deviations of the ground cover's expected response to rainfall. The relationships of ground cover with plant and bird diversity at a site level are examined and the extrapolation of these results to the landscape level is discussed.

Chapter Three has been published in the Rangeland Journal in 2009 and the methodology for field collection of ground cover attributes and environmental variables is presented in this chapter. The methodology follows that set out by Hassett *et al*. (2000) and has been used to calibrate and validate ground cover index estimations. Chapter Three is the work of the author, particularly the data collection and analysis. My supervisors who are co-authors on the article assisted with interpreting the results, especially in respect to the concluding remark about the threat of buffel grass.

1.3 Thesis outline

The structure of the thesis is outlined in this section, and displayed in Figure 1.2.

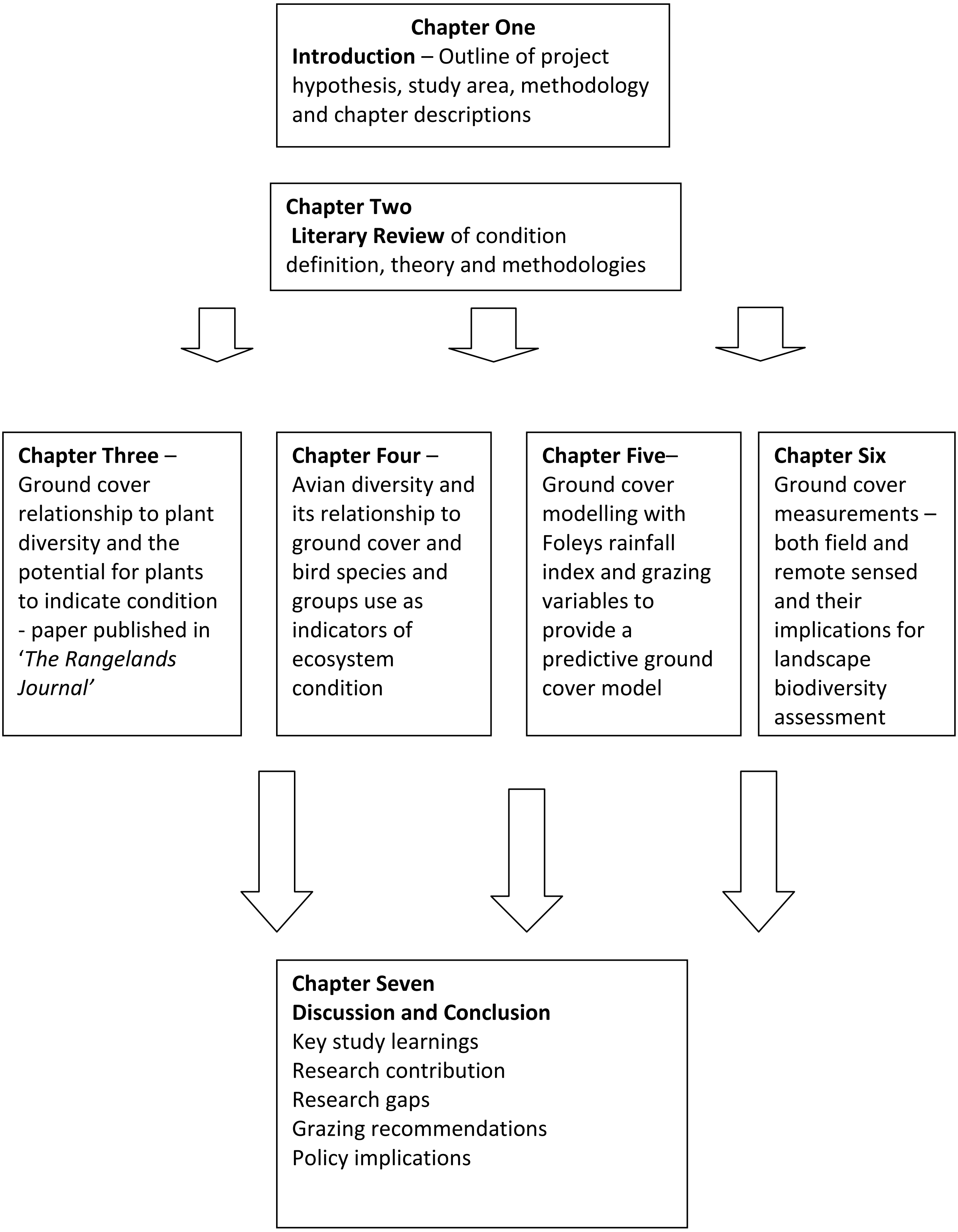


Figure 1.2 Thesis structure

Chapter Two

Ecosystem condition and ecosystem function are defined and their relationship to biodiversity explored. Furthermore, the development of ecosystem condition models with drivers of ecosystem condition investigated. The usage of remote sensing data for ecosystem condition assessment and for the measurement of the drivers of ecosystem condition, are reviewed with particular emphasis on grazing as a key driver of ecosystem condition. Various remote sensing methods to detect and distinguish grazing induced changes in ground cover from those of rainfall are examined.

Chapter Three

In this chapter, the use of a Bare Ground Index (the inverse of the now termed 'Ground Cover Index' (Scarth *et al.* 2006)) is assessed as a rapid methodology for the assessment of biodiversity condition of an ecosystem. The Multiple Regression Bare Ground Index (MRBGI) or Ground Cover Index (GCI) has been developed for semi-arid cover estimation in Queensland and is suitable to estimate cover when vegetation is sparse rather than continuous (Karfs *et al.* 2009; Scarth *et al.* 2006). It is used by the State Department of Natural Resources and Mines to monitor condition and trend in ground cover across Queensland (Scarth *et al.* 2006). The effectiveness of ground cover as a satisfactory correlate to plants in the ground layer (a key biodiversity attribute) is examined. The relationship of plant species richness and abundance to ground cover is examined within the silver-leaved ironbark woodlands in the Desert Uplands bioregion.

In this study, the hypothesis is that some plants, especially palatable perennial grasses, could serve as (i) indicators of biodiversity condition, and (ii) biodiversity indicators of the diversity of other taxa occurring in the ecosystem.

Chapter Four

The use of birds and bird groups as environmental indicators were reviewed from the literature, exploring the advantages and disadvantages of their use for this purpose. Avian composition in terms of species, groups and a habitat assemblage grouping were assessed in terms of their relationships with ground cover and the associated environmental variables using regression curve analysis. These relationships were further explored through constrained canonical ordinations and pattern analysis. The use of dietary, foraging and 'habitat assemblage' bird groups as indicators of biodiversity condition was reviewed from the literature to compare with this study's results. Averaging the group responses by ground cover segments improved the confidence in the responses.

In this study, the hypotheses are tested that (i) some bird species or groups could have significant relationships to ground cover and other environmental variables, (ii) some birds could be ecological indicators for the condition of the ecosystem, (iii) whether dietary and foraging bird functional groups could be effective indicators of ground cover and key woodland environmental variables, (iv) that habitat assemblage groups are useful indicators of ground cover and consequently ecosystem condition, and (v) that the ground cover index is an effective indicator of bird diversity in an open woodland ecosystem.

Chapter Five

This chapter is focused on using rainfall and grazing variables to develop predictive relationships with ground cover. In recent years, remote sensing studies have progressed from simply detecting land cover change to understanding the driving forces of land cover changes and to be able to model cover changes in order to better predict cover change outcomes. Some examples from across the world are detailed to demonstrate the applications of remote sensing cover analysis.

Cover indices that can be used with Landsat TM imagery are reviewed, including NDVI and PD54 methods. This review suggests that the usage of the GCI should be the most reliable and available index for usage in Queensland.

Linear mixed effect modelling is undertaken to quantify and distinguish the effects of climate and grazing variables on ground cover. A dataset of 300 randomly generated sites across the distribution of the silver-leaved ironbark woodland was used in the statistical analysis. The dataset includes twenty-one years of measurements of cover, rainfall index, paddock size, distance to water and area of more palatable and less palatable land types within the paddock for each of these sites measured from Landsat imagery. The predictive power of the three models is validated using ground cover data collected from 50 field sites in October 2007.

Chapter Six

In this chapter, the hypotheses that (i) ground cover measurements could be indicative of biodiversity condition and (ii) ground cover measurements could be indicative of other habitat attributes were tested. Ground cover field measurements and remotely-sensed measurements via the GCI were used to develop and test predictive models that incorporate grazing variables and rainfall.

The predictive power of the ground cover index is explored. Relationships are examined with data from field sites, alongside a remotely sensed ground cover index, and its application to various scales from paddock, property and the landscape. The importance of ground cover for ecosystem function is established from the literature and the ABCD land condition framework is examined in terms of how it relates to ground cover (McIvor *et al*. 1995). The relationship between biodiversity and ecosystem function is explored from spatial and temporal perspectives, the Richards/Green Functionality Index is presented and the limitations of the Ground Cover Disturbance Index are discussed.

Actual ground cover field measurements and the remotely sensed Ground Cover Index are tested for their relationships to actual ground cover measurements, environmental variables and grazing variables. Improvements of ground cover trend assessment are explored and improvements to the Queensland ground cover disturbance index are recommended.

Chapter Seven

The hypothesis and aims of the study are revisited with the study key findings presented. The contribution to knowledge of this study is outlined and how these improve ecosystem condition assessment and management of savanna woodlands in the rangelands are detailed briefly. Areas of further research are recommended.

Chapter Two

Ecosystem condition and its assessment

2.1 Introduction

In this chapter, ecosystem condition and ecosystem function are defined and their relationship to biodiversity explored. Furthermore, the development of ecosystem condition models are reviewed and discussed in relation to the silver-leaved ironbark woodlands of the Desert Uplands bioregion in central Queensland, and then the drivers of ecosystem condition are investigated. The usage of remote sensing data for ecosystem condition assessment and for the measurement of the drivers of ecosystem condition, are reviewed with particular emphasis on grazing as a key driver of ecosystem condition. The various methods to detect and distinguish grazing induced changes in ground cover from those of rainfall are examined.

2.2 Ecosystem condition and ecosystem function

An ecosystem is a functional and dynamic complex that results from the interactions of abiotic, biotic and anthropogenic components (Tansley 1935). The term 'ecosystem condition' refers to the state of the biological, physical and chemical characteristics of the ecosystem (or ecosystem properties), and the processes and interactions that connect them (Daily 1997). 'Biodiversity' describes the complexity of biological systems and the term refers to the variety in the composition and structural aspects of living organisms at different levels of organisation (Mooney 2002). Functional traits of species are important drivers of ecosystem properties and the total suite of functional traits in an ecosystem are a major component of ecosystem properties (Hooper *et al*. 2005). Species composition, richness, evenness and interactions all both respond to and influence ecosystem properties. Ecological processes and biodiversity maintain the function of ecosystems in both the immediate ecological and long-term evolutionary timeframes. There is an implicit acceptance that biodiversity also refers to the interrelatedness of biological components and its importance in maintaining the diversity of life (Wallace *et al.* 2004). There has been a re-emergence of the bi-directional relationship between the conservation of biodiversity and ecosystem processes (Hooper *et al.* 2005).

Ecological integrity is the maintenance of the functional attributes characteristic of a locale, including normal variability (Pellant *et al*. 2000). Disturbance regimes refer to a temporal and spatial pattern of a particular event. To prevent the deterioration of ecosystems it is necessary to derive criteria that can be used to prohibit human activities that deteriorate ecosystem condition, and as a counterpart, criteria that stimulate activities that ameliorate the consequences of previous harmful decisions (Haila 1997).

The effects of biodiversity on ecosystem functioning needs to be contextualised with respect to which components of biodiversity are affecting which components of functioning. Variation in ecosystem properties can result from fluctuations in the environment from year to year, directional changes in conditions and abiotic or biotic disturbance. Because many ecosystem properties fluctuate naturally over time, the difficult task is to determine the bounds of natural fluctuations in order to better understand whether anthropogenic induced fluctuations are outside the natural ranges of variability and therefore present a new threat to the sustainability of the ecosystem function (Chapin *et al.* 1996). Abiotic factors interact with functional traits of organisms to control ecosystem properties (Lavorel and Garnier 2002). In some instances, changes in biota can have greater effects on ecosystem properties than changes in abiotic conditions, such as impacts of invasions where a single species can strongly influence ecosystem properties (Levine *et al*. 2003). On the other hand, abiotic conditions, disturbance regimes and functional traits of dominant plant species can have a greater effect on many ecosystem properties than plant species richness (e.g. Enquist and Niklas 2001). At the landscape scale, a diversity of species with different sensitivities to a suite of environmental conditions should lead to greater stability of ecosystem properties and therefore the number of species necessary to maintain ecosystem function increases with increasing spatial and temporal scales. In other words, if an ecosystem is subject to a variety of natural and human-caused environmental stresses or disturbances, then having a diversity of species that encompass a variety of functional response types ought to reduce the likelihood of loss of all species capable of performing particular ecological processes, as long as response traits are not the same as or closely linked to effect traits (Lavorel and Garnier 2002). Studies of ecosystem recovery after disturbance have often found that ecosystems that were capable of a rapid recovery (exhibiting greater resilience) were those with a higher diversity of response types (e.g. a mixture of seeders and sprouters in the case of fire; Lavorel 1999).

Ecosystem functioning can be defined as all the ecosystem processes that determine the rate of matter and energy fluxes (Hooper *et al*. 2005). Indeed, global survey measurements carried out by Maestre *et al.* (2012) found there to be a significant relationship between perennial plant species richness and ecosystem multi-functionality, suggesting that species richness may be particularly important for maintaining ecosystem functions linked to C and N cycling, which sustain C sequestration and soil fertility.

The following section details the development of the theoretical models of how rangeland ecosystems function. Ecosystem models help guide what data to collect and how that data is analysed to determine on-ground management decisions (Westoby *et al.* 1989).

2.3 Ecosystem Condition Models

2.3.1 The Clementsian – Range Model

In the rangelands, a vegetation composition measure has long been endorsed as the index whereby vegetation condition and productivity can be assessed to account for grazing pressures (Stafford Smith in Hodgson and Illius 1996; Landsberg and Crowley 2004). Originally this was based on the equilibrium dynamics of succession towards a climax vegetation state (Clements 1928). The Clementsian model supposes that drought affects vegetation in a similar manner to grazing and, conversely, above seasonal rainfall accelerates the successional tendency. According to this model, land management should lower grazing pressure in response to drought in order to stabilize the condition and trend of vegetation along the continuum (Westoby *et al.* 1989; Friedel 1991; Laycock 1991).

The concept of Clementsian succession was found wanting in the rangelands because it supposes a given rangeland has a single persistent state (the climax) in the absence of grazing (Westoby *et al.* 1989). Perhaps if the seasonality of rainfall were constant then vegetation in semi-arid rangelands would advance to a more stable state in the absence of disturbance. This range succession model assumes that grazing pressure produces changes which are also progressive to a poorer condition state and in the opposite direction to the successional tendency. The model supposes that all possible states of vegetation exist along a single continuum from heavily grazed, early successional, poor condition to un-grazed, climax, excellent condition. Condition is therefore the vegetation's position along the continuum and trend is the direction along the continuum. The management objective is to choose a stocking rate that establishes a long-term balance between the pressure of grazing and the successional tendency (Westoby *et al.* 1989).

This model, on the whole, encapsulates the essential features of the traditional rangelands management approach. However, it does not relate very well to grazing systems that involve deferred and rotational grazing. In these grazing systems, grazed areas are subdivided into paddocks and stock is moved amongst the paddocks so that each paddock has both rested and grazed periods. These grazing systems can be implemented to reduce selective grazing and/or to provide a rest from grazing at plant seeding times of the year (Westoby *et al.* 1989).

The main limitation to the Clementsian range succession model is that vegetation changes in response to grazing have been found to be not continuous, or reversible and/or consistent. For example, perennial grasslands in areas where there is strongly seasonal rainfall have been converted to annual grasslands by grazing, but when the grazing has stopped they have often not reverted back to perennial grass dominance (Naveh 1967).

Current ecological theory allows for alternative stable states, discontinuous and irreversible transitions, non-equilibrium communities and stochastic effects in succession (Friedel 1997; Bestelmeyer *et al.* 2003; Stringham *et al.* 2003). There are mechanisms which are known to produce complex ecosystem dynamics such as: demographic inertia where some plant populations may require a rare event for establishment to occur and once established the resulting cohort may persist for a long time (for example, silver-leaved ironbark sapling establishment after an above average wet season (Fensham *et al.* 2003)); a grazing catastrophe where plant abundance may vary discontinuously and irreversibly due to stocking rate changes (for example, intensive and constant grazing pressure around watering points (Andrew 1988; James *et al.* 1999; Landsberg *et al*. 2003)); priority in competition where alternative stable states can result depending on the initial abundance of competitors (where adult species may have an advantage over seedlings (Augspurger 1984)); fire positive feedback where some species promote fire and are also themselves promoted by fire; and a vegetation change that triggers a persistent change in soil conditions (e.g. soil erosion) (Westoby *et al.* 1989).

In the rangelands, any one of the above mechanisms can produce alternative persistent vegetation states. Single climatic events, fire and/or grazing may change vegetation condition and trend in a manner that is not reversible or consistent with the Clementsian succession model (Laycock 1991). The limitations of this model are most apparent in the arid and semiarid rangelands where episodic events are important and the impacts of grazing and intrinsic vegetation change can act intermittently.

2.3.2 State and Transition Model

An alternative model is the State and Transition model where change is described by both a discrete set of vegetation 'states' and a discrete set of 'transitions' between states. 'States' are the potential alternative vegetation states at a site. 'Transitions' between states are triggered by management actions (e.g. change in stocking rates, burning, or the introduction of exotic species) and/or natural events (weather, fire). Transitions can occur quickly (via a fire) or over a long period of time (vegetation thickening – when the woody vegetation thickens as a result of wet events or from decreased competition from pasture) (Westoby *et al.* 1989).

State and Transition models help represent different states where an important change in the land occurs from a management perspective. Plant variation due to seasonal phenology of plants would not be broken up into different states, whereas important changes in botanical composition would be encompassed (Westoby *et al.*1989). While species composition may vary substantially in response to disturbances, ecosystem variables, including species richness, productivity and energy use, may remain relatively constant (Chesson and Case 1986; Brown *et al.* 2001). It could even be said that species fluctuations may represent a compensatory mechanism that contributes to ecosystem stability (Morgan Ernest and Brown 2001). The concept of stability defined by the resistance and resilience of plant communities underlies the use of State and Transition models. Resistance is defined as the ability of the system to remain the same while external conditions change, whereas resilience is the ability of the system to recover after it has been disturbed (Holling C.S. 1986; Holling *et al*. 2001; Walker and Abel in Gunderson 2002). Fully functional ecosystems are both resistant to change and resilient (when different disturbances occur) or able to recover without external energy inputs, thereby maintaining stability while allowing for fluctuating combinations of plant species over time (Ludwig and Tongway 1994). Therefore, within a state there exists the potential for a large variation in plant composition, which is merely a reflection of plant dynamics and scale. A change in state is the result of a shift across a threshold: a change in the ecological integrity of the site's primary ecological processes resulting in a different potential plant composition (Stringham *et al*. 2003). For example, in silver-leaved ironbark woodlands severe long-term overgrazing can lead to the dominance of unpalatable wiregrasses (*Aristida spp*.) which is difficult to reverse (Ash and Corfield 1998).

The State and Transition model helps characterize different possible states and conditions that induce transitions. Friedel (1997) defined the concept of thresholds as a boundary in space and time between two states (environmental change between domains of relative stability), which is not reversible on a practical time scale without substantial inputs of energy. This model also allows for categorization of management opportunities under different circumstances, such as when failure to burn or heavy grazing could result in an unfavourable transition that is difficult to reverse (Westoby *et al.* 1989).

Stafford Smith and Pickup (1993) identified two limitations to the State and Transition approach. First, the model does not readily account for spatial links across the landscape; grass cover loss and erosion in one area may lead to deposition and shrub increase in another (therefore it is important to recognize that some transitions may have extrinsic causes that depend upon landscape context rather than local management; Bestelmeyer *et al.* 2003). Secondly, the emphasis on the event of a transition takes attention away from chronic precursors of change and different rates of transitions. In a recent development of the model, Bestelmeyer *et al.* (2011) identified three spatial scales at which spatial patterns and processes manifest: patches, sites and landscapes, and identified three classes of spatial processes that govern heterogeneity in state transitions at each scale and that can be considered in empirical studies, State and Transition Management narratives and management interpretations. First, spatial variations in land-use driver history (e.g. grazing use) can explain differences in the occurrence of state transitions within land areas that are otherwise uniform. Secondly, spatial dependence in response to drivers imposed by variations in soils, landforms and climate can explain how the likelihood of state transition varies along relatively static environmental gradients. Thirdly, state transition processes can be contagious, under control of vegetation-environment feedbacks, such that the spatio-temporal evolution of state transitions is predictable.

The State and Transition model is a valuable management and categorization paradigm, and forms the basis to develop better scientific paradigms. A scientifically robust model requires a more generalized concept, recognizing the complexity of change at different temporal scales, from which it could then be simplified down to management rules for different systems.

A State and Transition model with threshold levels is more effective. The model can improve the capacity of managers to evaluate the costs and consequences of management decisions (Bestelmeyer *et al.* 2003). From a land manager's perspective, the transitions of condition states that are affected by management actions are of interest. Managers have many tasks that naturally revolve around an annual cycle. They include animal husbandry considerations e.g. cattle mustering for weaning, seasonal patterns (wet season versus dry season, summer versus winter) and financial cycles. Decisions about how many stock and what type of stock to return to each paddock are made yearly or more frequently (Watson *et al.* 1996). Land managers need to assess whether their stocking rate will change the state of their paddocks in a detrimental way.

The distinction between the range model and State and Transition models ultimately is derived from their origins; from equilibrium and non-equilibrium paradigms respectively. The Range model uses a univariate approach which emphasises grazing as the primary driver of vegetation dynamics. In comparison, State and Transition models accommodate additional complexity using a multivariate approach that explicitly incorporates multi-dimensions (e.g. fire and climate variability), in addition to grazing and relaxes assumptions of system predictability, stability and allows for potential non-equilibrium states (Briske *et al.* 2003).

The effectiveness of a model is given by its ability to predict the consequences of natural disturbances and/or management activities with acceptable precision over timescales relevant to management (Stringham *et al.* 2003). Rumpff *et al.* (2011) used a State and Transition model within an adaptive management framework to focus ecological assessment of the management influences on the restoration of woodlands composition and structure in southeast Australia in more than a static temporal way. The use of state and transition models within an adaptive management framework allows the use of multiple variable states rather than a univariate condition state.

Increasingly, conceptual models of how the ecosystem responds to key drivers are being incorporated and used as the basis for establishing biodiversity monitoring programs. Eyre *et al.* (2011) used a conceptual state and transition model to outline the predicted response of vertebrate species assemblages to three key rangeland drivers – rainfall, management pressure (grazing or altered fire regime) and predation. The combination of these drivers prompts transitions between states that are characterised by other species assemblages. A Bayesian belief network allows the effective and workable framework of an adaptive use of a process model (McCann *et al.* 2006; Rumpff *et al.* 2011).

2.3.2.1 State and Transition Model Relevant to the Eucalypt woodlands in the Desert Uplands

McIvor *et al.* (1995) sought to establish a quantitative index of land condition for the tropical tallgrass pastures in North Queensland based on vegetation and soil observations and measurements, thereby relating pasture production and species responses to this index. This land condition index has relevance to the silver-leaved ironbark woodlands as the pasture composition is very similar, the only difference being the presence of spinifex (*Triodia pungens*) in this study's ecosystem. In the tropical tall grass pastures of the Australian rangelands, a decline in land condition shows as a reduction in the palatable, perennial grasses with an increase in the unpalatable perennial grasses, annual grasses and forbs. There is also an increase in herbaceous and woody weeds, and a decrease in ground cover with possible erosion and other deleterious soil changes (McIvor *et al*. 1995). At the seven sites they established near Charters Towers, McIvor *et al.* (1995), divided plants into three groups. Group one consisted of the six perennial grasses that were common on good condition plots but whose presence decreased markedly as condition declined (State 1 or good condition as per Figure 2.1). Group two were the eight annual grasses which were almost absent from good condition plots but were recorded to increase as condition declined. The third group included three perennial grass species that were most common in intermediate condition sites (State 2 as per Figure 2.1). The first group of decreaser perennial grasses was equivalent to 'A' condition plots as defined by Ash *et al.* (1993), while the 'annual grasses and forbs state' was equivalent to 'D' condition (State 3 as per figure 2.1). Ash *et al*. (1993) included a state (2) of 'increaser perennial and annual grasses', which is probably an unstable, transient state accounting for their B and C condition plots. State 4 is an alternate state where substitution by exotic pastures has occurred.

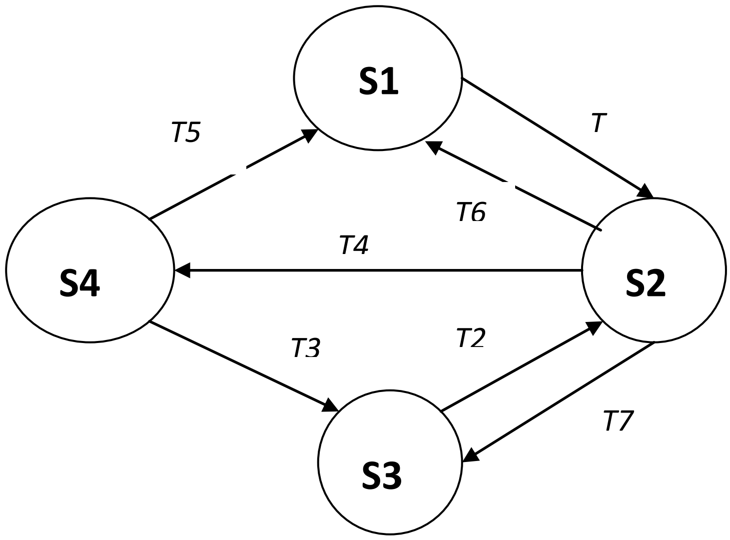


Figure 2.1 System diagram of State and Transition model for tropical savanna woodlands in northern Queensland. State 1 = Native perennial grass, State 2 = Unstable perennial and annual grass; State 3 = Annual grass; State 4 = Naturalised perennial grass. The *T* values are transition rates for changing from one state to another (figure modified from Scanlan 1994)

The following table illustrates a general land condition classification which is the basis of the QPI Stocktake framework and is based on the work of Ash *et al.* (1993) for assessing all land types at a paddock/property scale or landscape level. For ease of interpretation condition is classified into four main groups. However, it should be noted that exact threshold limitations for each condition rank for specific land types have yet to be established and is different to the State and Transition model above as land condition D could be State 3 and State 4 is undifferentiated from State 1 in A land condition (native versus non-native perennial grass presences). This current study was conducted in the intact areas of an ecosystem that covers over 500,000ha, but about 30% of the ecosystem has been cleared, predominantly in the southern end of its distribution. There has also been preferential clearing of more palatable ecosystems in the vicinity from which exotic pastures and legumes are spreading into the remnant areas of these woodlands.

Table 2.1 'ABCD' framework and broad classifications

Condition Features	Condition A	Condition B	Condition C	Condition D
Palatable, perennial grasses	> 70%	>50% and <70%	>30 and < 50%	< 30%
Ground cover	> 70%	> 40% and < 70%	< 40% and > 30%	< 30%
Weed presence	Less than 10%	10-30%	30 – 50%	> 50%
Soil condition	Good surface condition – no erosion	Some decline – some signs of previous erosion and/or susceptibility to erosion	Obvious signs of past erosion and high current susceptibility	Severe erosion or scalding
Woodland thickening or dieback	None	Some	General	Abundant
Productive pasture growth capacity	High	Down 25%-55%	Down 55%-80%	Down 80-100%

McIvor *et al*. (1995) included soil components in forming a condition index, comprising the following: surface crust development, soil surface micro-relief, litter and cryptogam cover and erosion features.

2.3.3 Non-equilibrium versus Equilibrium Theory

Ecosystems at equilibrium are characterised by competition, resource limitations, density dependence, few stochastic effects and tight biotic coupling (De Angelis and

Waterhouse 1987). Grazing represents a biotic process that internally regulates ecosystem behaviour by imposing negative feedbacks on vegetation processes (Briske *et al*. 2003). Non-equilibrium theory suggests that highly variable climate is the major driver of plant system dynamics with only a weak coupling of the herbivores to the plant community. Non-equilibrium theory is based on the assumption that ecosystems possess a limited capacity for internal regulation (Ellis and Swift 1988; Wu and Loucks 1995). Non-equilibrium ecosystem behaviour is more dynamic and less predictable than equilibrium system behaviour, and this implies that there is greater potential for vegetation change associated with periodic and often stochastic climatic events (Westoby *et al.* 1989).

Briske *et al.* (2003) suggest that ecosystems can reflect both equilibrium and non-equilibrium dynamics. They explored data that demonstrated the effects of both grazing and climate on vegetation dynamics. Their data indicated that the effect of herbivores on plant biomass increases as inter-annual rainfall decreases, but the ability of the herbivores to modify species composition increases with increasing rainfall (Chase *et al.* 2000). The disproportionate effect of herbivory on productivity and composition probably results from an increasing expression of selective herbivory with increasing primary productivity, even though the intensity of the plant-herbivore interaction may vary. Therefore, both herbivory and rainfall interact to structure plant communities (Chase *et al*. 2000).

Three types of studies have been carried out to distinguish equilibrium and non-equilibrium vegetation dynamics and in doing so, assess the reversibility of the vegetation dynamics, by examining the strength of plant competition and plant-herbivore interactions compared to climate induced vegetation dynamics. Firstly, changes over time involved examination of species replacement through time following a reduction in grazing or exclusion in grazing (Fuhlendorf *et al.* 2001). Secondly, the study of spatial community variation along a grazing utilisation gradient established by distance to water was examined (Fernandez-Gimenez and Allen-Diaz 1999; Ryerson and Parmenter 2001). Thirdly, there has been examination of the strength of correlation of vegetation change to grazing intensity compared to inter-annual rainfall patterns over time (Fuhlendorf *et al.* 2001). Additionally, non-equilibrium systems are at play in the existence of event-driven vegetation dynamics such as drought induced plant death or episodic plant recruitment (Friedel 1991). Chesson and Case (1986) suggest that while species composition may vary substantially in response to disturbances, ecosystem variables, including species richness, productivity and energy use, may remain relatively constant. Species fluctuations may represent a compensatory mechanism that contributes to ecosystem stability.

Ecological patterns and processes are often scale dependent, suggesting that as the spatial and temporal dimensions change, the pattern, rate and direction of change will also fluctuate. Ryerson and Parmenter (2001) showed that species-specific and site-specific vegetation changes after the removal of herbivores were not followed by a change in total perennial basal cover at the landscape scale.

Fuhlendorf *et al.(*2001) recorded vegetation dynamics following the removal of livestock in a juniper-oak savanna in Texas, U.S.A, showing that while the mid and short grass composition was affected primarily by the removal of grazing, total plant basal area was affected primarily by inter-annual rainfall and grass density was significantly affected by both grazing removal and rainfall. These results demonstrate that total basal area is not an effective indicator of grazing intensity in this system and its extensive use may be the reason that vegetation dynamics appear to be more responsive to climate than grazing. Response group composition and mean basal area per plant were more responsive to grazing than to inter-annual rainfall over the long-term. This indicates that structural attributes would be more effective indicators for monitoring vegetation dynamics in response to grazing than measures of plant basal area. Therefore a broad set of vegetation attributes, including individual species or specific functional groups, are likely to demonstrate that grazing and climate variability together drive vegetation dynamics, rather than just climate.

The work of Fuhlendorf *et al.* (2001) illustrates that grazing intensity established the long-term direction of compositional and structural change. However, a severe episodic drought influenced the short-term rate and course of vegetation change in the juniper-oak savanna in Texas, USA. Drought reduced plant density, but plant density eventually recovered and became proportional with grazing intensity. This less persistent response of plant community composition to rainfall variability rather than to grazing intensity is partially a function of the non-selective, intermittent effects of drought compared with the more chronic, selective influence of grazing on individual species or functional groups (Illius and O’Connor 1999). Intensive selective grazing often establishes the long-term course of vegetation change, while episodic climatic events often exert short-term effects on this rate and course (Fuhlendorf *et al.* 2001). The effects of grazing and rainfall variation together support the inference that stochastic rainfall variation does not maintain a system in perpetual non-equilibrium state (Weins 1984), but, rather, superimposes fluctuations on an otherwise directional response of community composition to grazing intensity. Therefore, the question to ask is ‘when do equilibrium and non-equilibrium dynamics apply’ rather than: which model is correct? In the long term, biotic factors may change the response of the ecosystem to short-term abiotic drivers such as rainfall (Caughley 1987).

2.3.4 Other conceptual models of ecosystem dynamics

The Resilience and regenerative capacity model (Holling 1973) is based on resilience as a measure of the persistence of ecosystems and the ability to absorb change and disturbance and still maintain the same relationships between attributes of the system. In other words, the capacity of an ecosystem to recover following the removal of disturbances is used in the grazing gradient method (Pickup *et al.* 1994; Chilcott *et al.* 2003).

The trigger-transfer-reserve-pulse model (Ludwig and Tongway 1997) is based on the concept that rainfall triggers biophysical activity, whereby activity products are transferred across the landscape by water and wind and deposited in different parts of the landscape to produce an ecosystem ‘pulse’. Sites are then assessed in terms of their capacity to retain resource or ‘leakiness’ (Tongway and Hindley 2005). Reference condition refers to the comparison of habitat attributes between a desired state – usually defined as pristine or undisturbed (Habitat Hectares (Parkes *et al.* 2003; Biometric (Gibbons *et al.* 2005), Biocondition (Eyre *et al.* 2005).

In this study, the State and Transition model (with elements of both equilibrium and non-equilibrium at play) best describes the possible condition states that occur in the silver-leaved ironbark woodlands based on the premise that both rainfall and grazing acting together are the main drivers of ecosystem condition change in this environment.

2.4 Drivers of ecosystem condition change

Management driven disturbances can affect ecosystem condition in the rangelands by way of influencing the persistence of ecosystems, species and critical ecosystem processes (Eyre *et al.* 2011). These disturbances include clearing of vegetation, fragmentation, grazing pressure, changed fire regimes, feral animals, exotic plants, mining activities and climate change (Woinarski and Fisher 2003; Beeton *et al.* 2006). Vegetation clearing is a major threat to biodiversity in Australia (Saunders *et al.* 1991; Kirkpatrick 1994). Habitat loss is recognised as the primary effect of clearing on biota, with fragmentation of the remaining habitat into an increasing number of smaller, more isolated patches a significant secondary effect (McAlpine *et al.* 2002). Ecosystem processes are greatly impaired in small remnants due to the increased exposure to changes in fluxes of radiation, water, wind and nutrients (Hobbs 1993). Fairfax and Fensham (2000) found that remnants adjacent to developed areas with exotic pastures were particularly susceptible to exotic pasture invasion such as buffel grass (*Cenchrus ciliaris*) and that the establishment of buffel grass caused a reduction in native pasture diversity. Gill and Williams (1996) reported that once exotic pastures are established in an agricultural landscape, there is general avoidance of burning, which leads to a build-up of fuel loads and an increased susceptibility of the remnant areas and the biota they support to intense fires. Increased pasture biomass tends to fuel intense fires, leading to incremental tree death and eventual loss of these remnants. Or conversely, the remnant areas double up as cattle camps due to their proximity to artificial waters and become denuded of ground cover diversity and regenerative capacity (Fensham and Fairfax 2002).

Climate change modelling for Queensland predicts increased temperatures of 1 - 3^{o}C, with decreases in rainfall of up to 25%, increased climate variability, and enhanced drying associated with El Nino events (CSIRO 2012). Species propensity for extinction under these extreme climate changes will be dependent on the ability of species to disperse along climatic gradients. The maintenance of large tracts of vegetation straddling gradients may be important if climate change is rapid (McAlpine *et al.* 2002). Therefore understanding the drivers of ecosystem change in relatively intact landscapes, such as the silver-leaved ironbark woodlands, is important to understanding the vegetation and fauna dynamics in an environment with high climatic variability.

The consequences of fire on ecosystem condition vary, depending on factors such as the frequency, timing, intensity, spatial extent and patchiness of fire. Fire can have a substantial effect on vegetation composition and structure (Kutt *et al.* 2009).
The frequency and extent of fire is readily and accurately assessed by remote sensing. Reasonable data is available for most regions of the rangelands. The North Australia Fire Information website provides data on fire frequency, time since last burnt, and fire scars (updated every eight days), mostly available as shapefiles, for northern Australia (ACRIS 2008). In the silver-leaved ironbark woodland most wildfires are started through lighting strikes and because these occur during the period of summer rainfall, they are least likely to induce long-term permanent effects on ecosystem condition. Therefore sites with fire scar evidence were removed from the analysis dataset of this study. Indigenous people probably lit fires outside of these wet season lightning fire events and therefore the current absence of fires in the dry season may have increased the extent and intensity of lightning fires as witnessed by severe widespread fires in the Desert Uplands in 2011. This is not to say that fire is not an important driver of ecosystem condition, but it is more likely to have long lasting effects in combination with grazing.

Climate via rainfall provides a context for interpreting changes in biodiversity in relation to short (seasonal) and long-term (decadal) climatic change. ACRIS (2008) provide examples of how rainfall data is used as a measure of seasonal quality for interpreting change in landscape function derived from pastoral monitoring data. Livestock grazing is likely to have detrimental impacts on conservation values, especially in relatively intact, uninvaded ecosystems on unproductive soils (Lunt *et* *al.* 2007). Livestock grazing can cause a range of ecosystem changes that may be difficult to reverse. Prior to European settlement, Australian landscapes had not experienced heavy grazing by ungulate herbivores. The introduction of cattle and sheep had moderate to catastrophic impacts on soils, ecosystem processes, vegetation and fauna (Noble and Tongway 1983; James *et al.* 1999; Gale and Haworth 2005). Grazing is still a major degradation driver in many Australian ecosystems (James *et al.* 1999; McKeon *et al.* 2004). Indeed, it can be hard to separate the historical impacts of grazing from the current grazing regime. Therefore, current grazing regimes may potentially have positive, neutral or negative effects on biodiversity, in the same places where livestock grazing originally caused substantial damage to ecosystem condition (Lunt *et al.* 2007).

Early disturbance-diversity models suggested that disturbances such as grazing would have varying effects on diversity, depending on the intensity (Grime 1973). The intermediate-disturbance hypothesis predicted that diversity would be low in undisturbed conditions (owing to the competitive exclusion of dominant perennial species), high at moderate grazing levels (due to the reduction of biomass of dominant perennial species and enhanced recruitment of less competitive species) and low under heavy grazing (due to the intolerance of many species to regular defoliation). In silver-leaved ironbark woodlands, there are areas where all three types of grazing intensity produce the predicted plant diversity effects. Known effects of heavy grazing on silver-leaved ironbark pastures are an increase in the unpalatable wire grasses (*Aristida spp*.), coupled with losses of palatable perennial grasses such as *Bothriocloa ewartiana*, *Themeda triandra* and *Chrysopogon fallax*

(Ash and Corfield 1998). Milchunas *et al.* (1988) argued in their review of grazing impacts on plant diversity, that these impacts are mainly driven by the interactions of evolutionary history of grazing by large herbivores and site productivity in terms of above ground net primary production. They found that the site productivity and history of grazing were more important than the actual intensity of grazing. Intermediate responses are likely if there is little evolutionary exposure to heavy grazing and medium productivity as in the case of the silver-leaved ironbark woodlands. Implicit in the intermediate disturbance hypothesis is that if high intensity grazing leads to a reduction in diversity, then diversity can be increased again by reducing grazing intensity. However, as described earlier in the State and Transition models, grazing can cause a range of irreversible ecosystem changes. Livestock grazing affects ecosystems in other ways than just the direct effects on vegetation. Grazing induces land degradation by affecting soil moisture storage capacity and nutrient availability through trampling and defoliation (Pickup *et al.* 1994; Roth 2004). Livestock can affect ecosystems by soil compaction, increases in erosion, and nutrient deposition. Effects of grazing on soils, plants and animals are most pronounced close to watering points where declines in species diversity and palatability are commonplace (James *et al.* 1999; Landsberg *et al.* 2003).

Grazing- induced floristic changes appear to have been less substantial in many tropical woodlands receiving summer rainfall (as is the case in the Desert Uplands) than in temperate woodlands, where pastures remain dominated by diverse native species, despite changes in their life-form composition (McIntyre and Martin 2001; McIntyre and Lavorel 2001).

2.5 Remote sensing for assessment of ecosystem condition

Multi-sensor approaches may be particularly useful to assess changes in ecosystems, especially when combined with ancillary data such as field observations (Rose *et al.* 2014). Satellite technology plays an increasingly important role in detecting, mapping, understanding and predicting changes in the environment. Such approaches can increase the understanding of the ecological ramifications of disturbances and help identify thresholds of disturbance above which there are substantial effects on species and ecosystems and could determine how disturbances affect processes such as carbon sequestration and nutrient cycling. The integration of data on intrinsic biological factors, extrinsic environmental drivers and historical and current species distribution and abundance can help understand ecosystem condition dynamics (Elith and Leathwick 2009).

Remote sensing provides data on extrinsic environmental drivers such as land cover and primary productivity (via Normalised Difference Vegetation Index (NDVI)). With remote sensing data at finer spatial and temporal resolution one can understand distribution and abundance of certain species; - such as fractional land cover, density of human structures, habitat quality for a given species, land and sea surface temperature, fire dynamics, phenology, topography and vertical vegetation structure. NDVI seasonal measurements, at the ecosystem level and regional scale, have emerged as a suitable tool to quantify overall productivity and biomass (Running *et al.* 2004; Turner *et al.* 2006).

Many plant traits are sensitive indicators of ecosystem condition. Changes in pigments may indicate a variety of stresses, including disease, pollution or adverse weather conditions (Ustin *et al.* 2009). The specific wavelength location of the 'red edge', the steep increase in vegetation reflectance from red to near-infrared wavelengths, can also indicate vegetation stress (Li *et al*. 2005), as can leaf water content, temperature and changes in productivity. A discrepancy between observed and potential productivity could suggest degradation (Kienast *et al*. 2009). Ecosystem condition can be empirically modelled by spatial environmental data layers (Zerger *et al*. 2009).

Remote sensing can help understand, monitor and predict ecosystem responses and resilience to multiple stressors. Ecosystems and their processes are constantly changing in response to natural and anthropogenic disturbances. Remote sensing offers cost-effective information on the ecosystem extent, status, trends and responses to stressors over large areas. High spatial resolution and frequent revisits document long-term effects of extreme events.

Disturbances can be mapped indirectly (as areas susceptible to disturbance) using spatial soils, vegetation, and climate data or plant traits (Yebra *et al*. 2013) or directly from remote sensed observations (Frolking *et al*. 2009). Disturbances can be detectable from single image data if distinct legacies are made (e.g. fire scars). Multi-date imagery can detect disturbances and recovery through changes in reflectance. High temporal resolution sensors such as MODIS and the Landsat imagery support remote sensing of disturbance with detailed temporal trajectories and time series analyses (Kennedy *et al*. 2007), including detection of both abrupt and subtle changes in ecosystem condition (Verbesselt *et al*. 2010).

To understand the effects of agriculture on species and ecosystem functions it is necessary to systematically assess the rates and locations of expansion and intensification of agriculture. The spatial and temporal resolution from satellite observations allow mapping of these small to large changes. The combination of images with high temporal and low spatial resolution (that capture the timing of vegetation changes such as changes in phenology) such as MODIS with images with high spatial and low temporal resolution, such as Landsat allow the assessment of agricultural impacts (Rose *et al.* 2014).

While remote sensing can detect disturbance associated with changes in land cover, ecosystems can also be disturbed without land cover change, which makes such disturbances more challenging to detect. Detectable land cover conversion may not accompany changes in composition, structure and function, including changes in vegetation and soils caused by varying levels of livestock grazing, changes in species composition and vegetation structure caused by non-native invasive species, increased tree mortality caused by insect outbreaks and air pollution, and the myriad effects of climate change.

A major challenge for conservation managers is to devise management systems that integrate grazing production and sustainability of ecosystem values and functions. A spatially-explicit quantification of aboveground net primary production (ANPP) and its intra and inter annual variability would allow for the delineation of sustainable stock densities and grazing areas and timing, based on the relationship of ANPP and stock consumption (Oesterheld *et al*. 1992). A pilot system has been implemented to support grazing management based on monitoring ANPP on extensive ranches in the Argentine pampas (Grigera *et al*. 2007). Tracking herbivore stock densities and ANPP could be used to assess the role of overgrazing on desertification processes (Paruelo *et al.* 2008).

Integration of paleo-ecological and paleo-climatological data, contemporary observations of ecosystem status and trend and environmental models can help estimate ecological and economic effects of climate change and therefore allow for the development and assessment of adaptation and mitigation plans (Rose *et al.* 2014).

2.6 Remote sensing methods to detect and measure the effect of grazing on ecosystem condition

Spatial patterns associated with grazing can be separated from other types of spatial variability in the landscape. A conceptual model that describes how rangeland ecosystems behave in response to drivers such as rainfall and grazing, and how past grazing has affected the present state of the ecosystem and its future sustainability is necessary (Pickup, Bastin and Chewings 1994). A model needs to be developed into a set of sampling, measurement and analytical methods that can separate the effects of grazing and other management practises on the indicator variables from other sources of spatial and temporal change. Methods need to be able to detect change at the spatial and temporal scales at which management is carried out – paddock to properties, and months to years.

Grazing gradients occur in arid and semi-arid Australia because domestic animals graze out from artificial waters to which they must return at regular intervals to drink. There is a general decline in animal activity and impact on plant behaviour as distance from water increases. Some grazing gradients may be only temporary and disappear with the next major rainfall event, while others remain for longer periods, indicating that grazing is having a more permanent impact (Pickup *et al.* 1994). These permanent gradients provide a means by which key biotic effects can be separated from other factors. Because of the natural scarcity of water in most regions of the rangelands, grazing by livestock (and also by some native species) tends to be centred on water points, with grazing intensity decreasing with distance from water. Native biota show three types of responses in relation to grazing around water points: 1) the abundance of a species decreases with proximity to water sources (grazing sensitive species or decreasers); 2) the abundance of a species increases with proximity to water (species not favoured by grazing or increasers); and 3) species show no response with distance to water (neutral response) (Biograze 2000).

Through the use of grazing gradients, it is possible to separate out the grazing-induced loss of ecosystem condition from changes in total forage production from short-lived rainfall variability (Pickup *et al*. 1994).

Ground cover (that is the cover types of either green leaf, dead leaf, bare ground, shrub, grass litter or tree litter as per Hassett *et. al*. 2000), if used independently, is not an adequate indicator of rangeland condition. However, utilising frequent time series and the high spatial resolution provided by remotely sensed data, allows for patterns of change in plant cover over both time and space to be identified (Pickup *et al.* 1994). These patterns contain both grazing-induced and abiotic elements which, if separated out, can be used to develop ecosystem condition indicators that are explicitly based on spatial and temporal changes in remotely sensed ground cover indices. Ground cover is correlated with total grass production, which strongly affects animal production in non-equilibrium systems, rather than the actual plant species composition (Ellis and Swift 1988; Ash *et al.* 1992). This situation arises either because all forage has a similar nutritional value due to stocking levels being so high that most forage is consumed between plant growth events, or because the growing season is so short that forage quality decline is independent of grazing selection and off-take. Plant species composition may be a more important consideration in determining animal production under light stocking, when only a limited proportion of the total forage is consumed, or where grazing is so heavy that only the unpalatable species survive. On the other hand, the use of total grass production, as an explanatory variable, is suitable for situations in which the loss of palatable species occurs fairly uniformly across the whole grazed area. The difference in cover with respect to distance from water is then sufficient to describe differences in forage availability along the grazing gradient. Thus, loss of productivity can be inferred from ground cover dynamics as well as ground cover itself (Pickup *et* *al.* 1994). Therefore, the use of ground cover is dependent on the context of grazing evenness at the paddock scale and variability of rainfall.

Pickup *et al.* (1994) describe four vegetation parameters that can be used individually or in combination to assess changes in ecosystem condition. The first three methods use the peak of major vegetation pulses to assess ecosystem condition and, as such, they reflect the amount of vegetation produced rather than its value for grazing. These methods assume that the amount of forage present is proportional to the total vegetation cover:

a) <u>Wet period average cover</u> – A model of the grazing-induced reduction of ground cover after the best plant growth condition experienced within a reasonable time.

b) <u>Wet period cover variance</u> – This is used where there is soil and water redistribution as a result of grazing. It is not effective on sand plains, such as the ironbark woodlands in the Desert Uplands, as redistribution is limited.

c) <u>Above or below expected response to rainfall – (Resilience method)</u> whereby each location should have an expected response to rainfall, which represents plant behaviour in an un-grazed or sustainably grazed condition. The difference between observed and expected responses provides a measure of the resilience of the ecosystem. Below average responses indicate a loss of resilience and above average responses indicate a high level of resilience. The model should incorporate the increase in cover after a rainfall event as a function of rainfall magnitude and the initial state of the vegetation as indicated by cover before the event (end of dry season). The Resilience

Method should be taken from data that is a sufficient distance from water where there is no detectable Wet Period Average Cover gradient. If this is not possible because grazing effects extend to the paddock boundaries, then areas close to water should be avoided.

d) Rate of cover depletion over time after a growth pulse – This method quantifies the location and intensity of grazing from the spatial variable component of the pattern of cover depletion over time. The amount of forage present is not always proportional to the total vegetation cover because past grazing may have resulted in an increase in the proportion of unpalatable species present, as well as a reduction in the amount of cover produced from rainfall. This method is capable of recognizing grazing-induced changes in the proportion of forage present. By looking at the spatial pattern of cover change during the decay phase, vegetation cover decreases because of grazing which should vary systematically with distance from water, whereas other processes of cover loss should on average occur at a constant rate across each landscape type. Paddocks where the proportion of forage is high are likely to be grazed out more rapidly close to water than at a distance, since initially stock do not have to graze far to get their nutritional requirements. Conversely, in paddocks where the preferred forage is limited, stock will have to search more widely and the rate of cover depletion will not vary so much with distance to water.

2.7 Ground cover as a landscape indicator of ecosystem condition

In conclusion, ground cover can be assessed over large areas by remote sensing which makes it potentially attractive as a measure of ecosystem condition. However, the utility of ground cover as an indicator for ecosystem condition has yet to be established, in part because of the limitations in the technology and also because the response of species to changes in different types of cover is poorly understood. Methods to distinguish between annual and perennial herbaceous cover, ground cover from over-storey canopy cover, and decreaser (i.e. under grazing) and increaser perennial grasses can be problematic. Nevertheless, distinguishing perennial grasses from annuals is possible by using imagery from extended rainless periods at the end of the dry season (when most annuals would have disappeared). A tree mask to separate ground cover from over-storey canopy cover has been developed for Queensland as part of SLATS (State-wide Land Cover and Trees Study; DSITIA 2012).

In Queensland, estimates of ground cover are made annually for the whole state using Landsat TM imagery. The Ground Cover Index (GCI) is calculated from a multiple regression model of the reflectance of bare ground in bands 3, 5 and 7 and is calibrated to field-based measurements. It has been developed for semi-arid cover estimation in Queensland and is able to estimate cover when vegetation is sparse rather than continuous (Scarth *et al.* 2006; Karfs *et al.* 2009). GCI integrates total organic soil surface cover, including green and senescent grasses and forbs, grass, litter and cryptogams. However, separating the cover of increaser and decreaser species, and native and exotic species remains a challenge. These distinctions may be important in terms of land condition and the suitability of habitat for native biota. Eyre *et al.* (2011) suggest assessing sites of optimal condition, likely to be far from artificial water and hence grazing pressure (> 6km from permanent water (Fensham and Fairfax 2008)) and assessing sites with average condition that are broadly representative of moderate to significant pressure by the management issue under investigation (e.g. grazing pressure) in order to track the effects on biodiversity under the most prevailing disturbance regime.

Despite these contemporary limitations, it is likely that remotely-sensed ground cover data has potential as a surrogate for habitat condition (Kutt *et al.* 2009). Temporal trends in projective foliage cover or woody cover may indicate broad changes in the suitability of an area for a range of species dependent on trees, forests or woody shrubs, for example. Further work is required to validate ground cover as an indicator for biodiversity and how to interpret trends in ground cover in relation to particular taxa. Issues to be considered include not only the response to changes in absolute cover but also changes in type of cover. The measurement of ecosystem/ landscape function or the ability of the landscape to capture and retain water and nutrients can be measured through the leakiness index (Ludwig *et al.* 2007), which is based on the spatial configuration of patches of vegetation cover obtained from Landsat imagery. A prerequisite for the leakiness index is a ground cover index. However, linkages between ecosystem function and biodiversity have yet to be established. It is possible that adverse changes to biodiversity may occur more rapidly than the ability to detect changes in ecosystem function using remote sensing. Understanding and measuring ecosystems at multiple scales is important (Pringle *et al.* 2006).

Bastin *et al.* (2012) used a dynamic reference cover method to remotely detect management related trends from climate. They used a minimum ground cover image derived from Landsat TM imagery across all years to identify areas of most persistent ground cover in years of lowest rainfall. They calculated the difference between the window's central pixel and its surrounding reference pixels to provide a seasonal interpreted measure of management effects. Areas of persistent ground cover during drier years provide benchmarks defining resilient landscapes, with high biodiversity integrity and high landscape functionality (Ludwig *et al.* 1997).

While the ground cover index has been demonstrated as a reliable remotely sensed measurement of ground cover, ground cover changes and trends over time, it is important that through the State and Transition model we understand the relationship of ground cover to other ecological components of the silver-leaved ironbark ecosystem, namely – key plant functional traits such as native perennial grasses and bird species that are likely to be correlated such as granivores and bird species that nest on the ground.

In this study the relationship of ground cover to understory plant species, bird species, grazing variables and rainfall are explored in order to better quantify ecosystem condition and to determine whether thresholds of ground cover show significant correlations with bird and plant composition. The dynamics of ground cover across the distribution of the silver-leaved ironbark woodland were examined with respect the relationship with the key drivers – rainfall and grazing.

Chapter Three:

Can bare ground cover serve as a surrogate for plant biodiversity in grazed tropical woodlands?

3.1 Abstract

A bare ground index derived by remote sensing would provide a rapid methodology for assessing the biodiversity condition of an ecosystem, provided ground cover is a satisfactory correlate with key biodiversity attributes. The relationship between plant species richness and the abundance of individual species was examined in relation to ground cover within silver-leaved ironbark (*Eucalyptus melanophloia* F. Muell.) woodlands in the Desert Uplands bioregion of north-eastern Australia. There was significant correlation between the bare ground index and ground cover and biomass measurements. Twenty-four species, including the perennial grasses *Sehima nervosum* (Rottler) Stapf, *Themeda triandra* Forssk. and *Bothriocloa ewartiana* (Domin) C.E. Hubb., were significantly and negatively correlated with bare ground. *Scleroleana birchii* (F.Muell.) Domin and *Sida fibulifera* Lindl. displayed significant positive relationships with increasing bare ground, and where abundant indicate heavy grazing in this land type. The study suggests that satellite-derived data does provide a meaningful methodology for assessing vegetation condition although it is strongly associated with seasonal conditions, but is only useful in relation to the regional average for a land type. The findings suggest that plant diversity is

maintained at a relatively high level throughout most of these woodlands in the Desert Uplands.

Additional keywords: bare ground index, biodiversity condition, grazing, plant composition, silver-leaved ironbark.[1]

3.2 Introduction

Ground cover is functionally related to several key indicators of land condition, including runoff and soil loss (Scanlan and McIvor 1993; Dube *et al*. 1999). Losses of ground cover can be caused by climate, fire or grazing and can lead to changes in 'land condition' or the functional integrity of the landscape.

Ground cover can be interpreted from satellite imagery, which provides a spatially explicit and attractive monitoring option for estimating ground cover and trends over large areas. Satellite imagery needs to be correlated with ground cover measured in the field, and validated via multi-temporal image analysis, and multi-temporal datasets so the assessment of condition may be quantified. A remotely-sensed cover index should not be used in isolation from site assessment because it helps to place sites within a landscape context (Gibbons and Freudenberg 2006). There is increasing evidence that remotely sensed trends in cover, combined with field surveys can be used to assess landscape condition (Wallace *et al*. 2006, Ludwig

[1] This chapter was published in 'The Rangeland journal ' in 2009.

et al. 2007), but the procedure needs further testing for individual land types, particularly where there is tree cover (Ward and Kutt 2008).

McKeon *et al.* (2004) concluded from the results of five grazing trials in Queensland that for normal (average rainfall) and wet (above-average rainfall) years, the basal cover of grass under heavy grazing was 88% of that under light grazing. The ‘grazing gradient’ method shows that where there is a higher intensity of grazing close to water there is less ground cover than away from water (Pickup *et al.* 1994). This has been verified in other field based assessment employing ‘distance to water-point’ gradients (Landsberg *et al*. 2003). Grazing in heavily utilised areas leads to losses in palatable perennial grasses and an increase in bare soil (Mott 1987; Ash 2004 in McCullough and Musso 2004). In this study, we assume that spatial variation in ground cover within a single land type represented by Landsat data at one time is primarily caused by grazing, however, climate, fire and soil variations also play a role. Landholders in the rangelands need to ensure that they maintain the ecological integrity of the ecosystems which support their grazing operations in these changing times, where increased accountability for sound management practises is demanded by the broader public. For example, in Queensland, under the Delbessie Agreement (State Rural Leasehold Land Strategy), the lease renewal process will require landholders to maintain or improve the land condition of their property (Department of Natural Resources and Water 2007). However, information about the condition of ecosystems is typically complex to collect, assess and interpret. A rapid method of ground cover assessment would greatly aid landholders’ responses in terms of tactical land management changes directed to minimising negative grazing impacts.

The herbaceous layer in eucalypt woodlands provides the fodder for domestic livestock and other herbivores, and contains most of the plant diversity (Lindenmayer *et al.* 2005). Other studies have shown that the plant composition of the ground layer in semi-arid woodlands is sensitive to grazing (Fensham and Skull 1999), and plant measurements have considerable potential to be effective indicators of the response of rangeland biodiversity to land use (Landsberg and Crowley 2004).

The extensive semi-arid open woodlands of the Desert Uplands bioregion in north-eastern Australia (Fig. 3.1) are a uniform land type and extensively grazed. They are, therefore, suitable to assess whether ground cover is an effective surrogate for plant diversity. In the Western Arid Region Land Use Study part four (WARLUS) (Turner *et al*. 1978) the Desert Uplands bioregion is described as 'the present condition of the majority of these lands is fair to good' because 'the land types most susceptible to land deterioration have not as yet been subjected to intensive use'. Production from cattle grazing in these woodlands has increased over time through a more even spread of waters, the use of nutrient supplements, introduction of exotic plant species and some clearing of vegetation (Pringle and Landsberg 2004; MacLeod and McIvor 2006). By 2008, the Australian Collaborative Rangelands Information System (ACRIS) reports that in the Alice Tableland subregion of the Desert Uplands bioregion there is significant loss of landscape function and that this is combined with an increased level of utilisation reflecting increases in stock numbers in periods of low pasture production between the periods of 1976-90 and 1991-95 (Bastin and the ACRIS Committee 2008). It is still unclear what the long-term grazing capacity is and what the trade-offs in terms of ecosystem integrity and composition involved in the various possible grazing regimes.

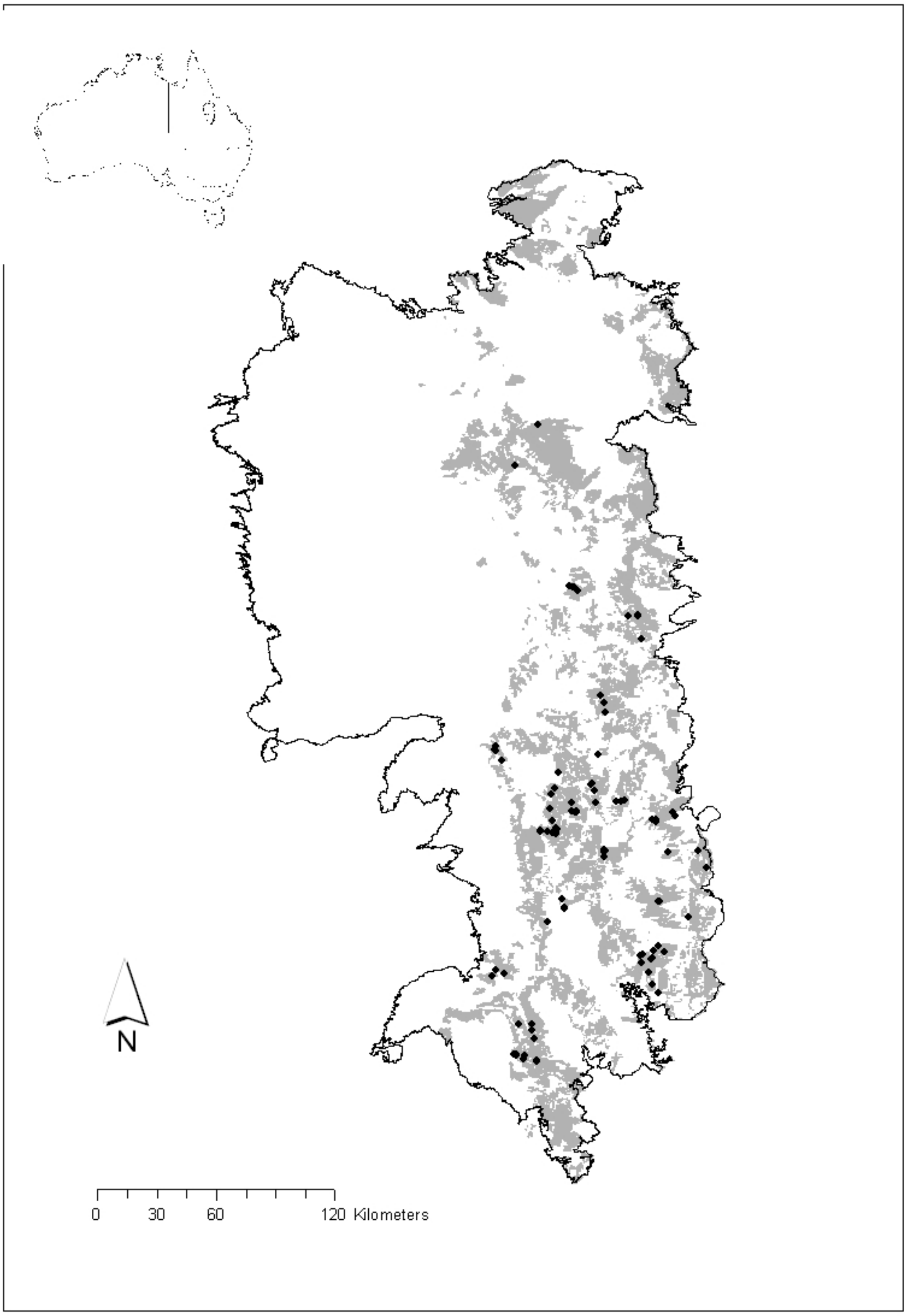


Figure 3.1 The Desert Uplands bioregion in Queensland, Australia, with the target regional ecosystem (non-remnant and <20% projective foliage cover excluded) according to the ground cover index. This represents a total area of 465 084 ha. The sites are indicated (symbol).

We used satellite imagery and ground cover survey methods to develop a relationship between ground cover and biodiversity, the latter focussing on the composition of the herbaceous layer of the eucalypt woodland. After exploring the relationship between plant species richness, abundance and composition with bare ground, we conclude with a preliminary assessment of plant biodiversity for the target ecosystem at the regional scale.

3.3 Materials and methods

3.3.1 Study area and target ecosystem

Silver-leaved ironbark (*Eucalyptus melanophloia* F. Muell.) woodlands cover over a million hectares in the Desert Uplands bioregion (Fig. 3.1) and are the most widespread ecosystem with grazing production values in the bioregion. The herbaceous layer of this ecosystem is a grassland which has a fairly even spread of tussocks. The targeted ecosystem for this study is regional ecosystem 10.5.5; - silver-leaved ironbark woodland with an open grassland understorey of *Aristida spp*. and/or *Triodia spp.* on loamy red and yellow earths and undulating sand plains (Sattler and Williams 1999). Mean annual rainfall varies from 490mm in the north of the study area to 560mm in the south, and is summer dominant.

Site selection and sampling

Twenty-five grazing properties with the highest proportion of the targeted regional ecosystem were selected. Within these properties, 91 sites that could be readily accessed and which represented a wide range of ground cover were selected for study.

Sites were selected to equally represent the four categories developed from the multiple regression bare ground index (MRBGI) for September generated by the Queensland Department of Natural Resources and Water (Scarth *et al.* 2006). These categories were: (a)≤ 25% bare ground, (b) 26-50% bare ground cover, (c) ≥ 50 – 75% bare ground cover and (d) ≥ 75% bare ground cover. The MRBGI is used by the Queensland Government to monitor ground cover (the converse of bare ground) across a range of land/vegetation types in Queensland's grazing lands and has been extensively validated with ground data. Cleared areas have often been developed with exotic pasture.

The MRBGI has been shown to be unreliable when tree cover is greater than 20%, so areas mapped as non-remnant (24.6%) and areas with greater than 20% projective foliage cover (39%) were excluded as study sites. Care was taken to ensure that the sites were independent, although some sites were in the same paddock but always > 1km apart.

Ground cover data was collected to verify the index and the relationship between bare ground and plant species composition was examined.

The centre-points of each site were located on the predetermined coordinate of the MRBGI pixel. A 100m tape was run north-south and another east-west centred on the point. A metal pin was used to score point intercepts for 200 recordings of cover (every 1m along the tape) in three canopy categories – understorey, mid-storey and upper-storey. The understorey was scored as bare ground, litter, cryptogam, dead attached plant material or live attached plant material. Both the mid and upper-stories, where present, were scored into three categories: branch, live leaf or dead leaf. Five 0.25 x 0.25m quadrats were cut at 20m intervals along the north-south transect, stored in paper bags, dried for 48h at 280^{o}C and weighed to establish the pasture biomass. The distance from water was determined for each site.

Eight vegetation subplots (10 x 2m) were established 2m apart along the north-south transect line. All plant species in each subplot were recorded with nomenclature following Henderson (2002) and voucher specimens lodged at the Queensland Herbarium. The frequency of occurrence from the eight subplots provided a measure of abundance (0-8) for each species at each site. The field sites were assessed in May 2007, at the end of an average wet season, a good time for identifying herbaceous flowering plants. The timing of the field sample was eight months after the satellite capture because of the time required for processing.

The MRBGI for each site was compared to the field measures of bare ground (excluding, litter, cryptogam, and dead leaf matter) using linear regression analysis. Plant species were grouped by life form (annual forb, annual grass, perennial forb, perennial grass, total annual, total perennial and total species) and richness and abundance (the total abundance scores for each group) compared with bare ground using linear regression. Distance to water was compared to ground cover and to species abundance and species richness. Abundance is scored as the frequency of each species in the eight subplots, whereas species richness is the sum of the species found at each site.

The data for many of the species were skewed by high numbers of absences, a problem for statistical modelling. This was overcome statistically by averaging the abundance scores into nine segments of 10 sites (11 for the last segment) after ordering sites by their bare ground index. These averaged scores were compared with the mean bare ground score for each segment using linear regression.

The proportion of MRBGI segments for the target regional ecosystem – silver-leaved ironbark woodlands (RE 10.5.5) is also presented and discussed in relation to plant biodiversity condition of the regional ecosystem at the regional scale.

3.4 Results

The bare ground field measurements of May 2007 were strongly related (R^2=0.569, P<0.001, Fig 3.2*a*) to the bare ground index of September 2006, and the biomass measurements taken in May 2007 (R^2=0.264, P<0.001, Fig. 3.2*b*), whereas the bare ground measurements were not significantly related to distance to water (Fig. 3.2*c*). Outlier sites may be due to different levels of wet season growth or grazing pressure between satellite capture and field sampling, or may actually represent limitations in discriminating between bare soil and plant cover in some situations.

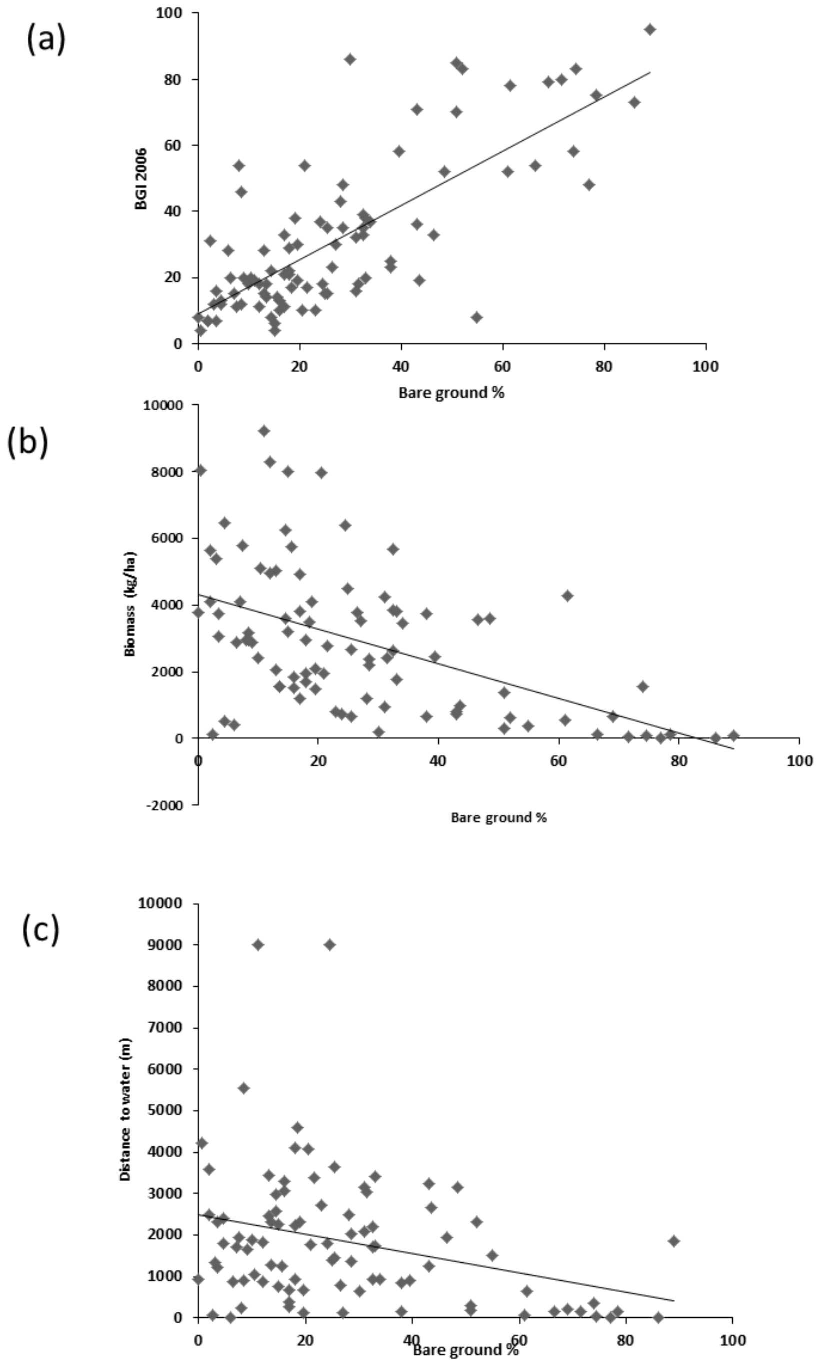


Figure 3.2 Relationship between bare ground measured in the field and (*a*) the bare ground index, (*b*) the biomass, and (*c*) distance to water.

A total of 171 plant species were recorded across the 91 sites, of which 92 species occurred more than six times. Three exotic species were recorded, namely *Cenchrus ciliaris*, *Stylosanthes scabra* and *Opuntia stricta.*

In terms of species richness, the perennial forbs (R^2=0.092, P<0.001), perennial grasses (R^2=0.242, P<0.001) and total species richness (R^2=0.228, P<0.001) were significantly negatively related to the percentage of bare ground.

In terms of species abundance, the perennial forbs (R^2=0.192, P<0.001), perennial grasses (R^2=0.149, P<0.001) and total species abundance (R^2=0.223, P<0.001) were significantly negatively related to the percentage of bare ground.

Twenty-four plants showed significant relationships with increasing bare ground (see Appendix 1). Two species had significant positive relationships with increasing bare ground and 22 had significant negative relationships with increasing bare ground, including seven perennial grasses.

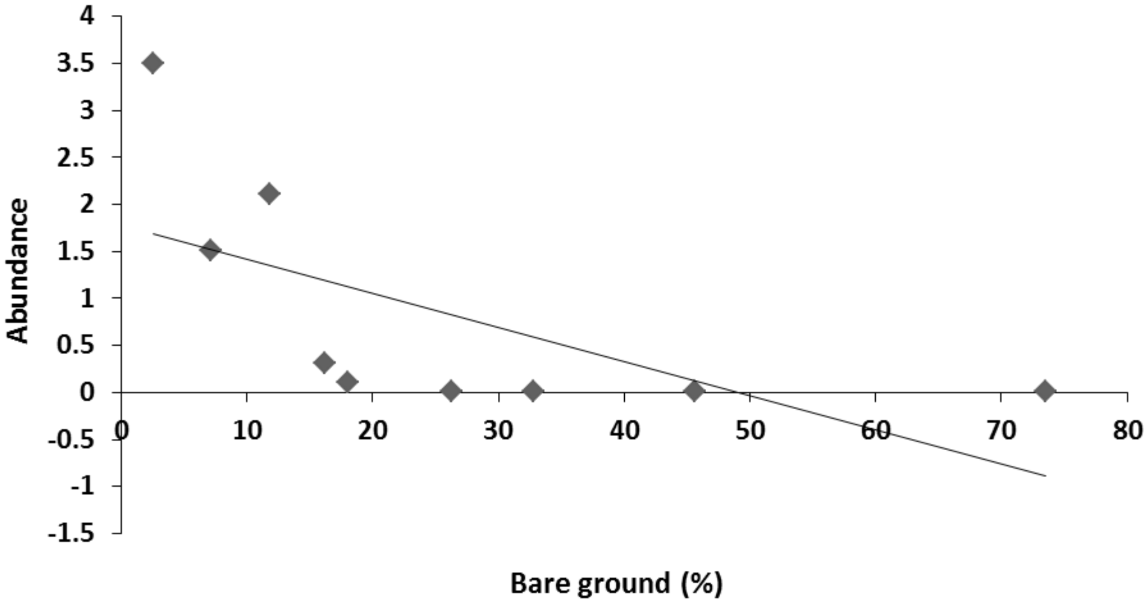
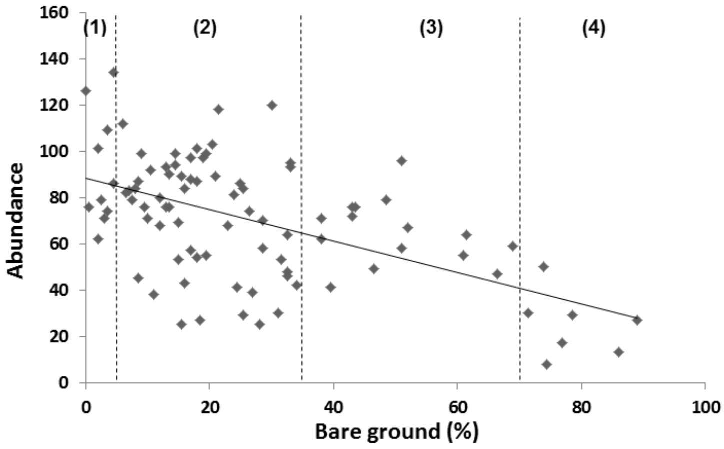


Figure 3.3 Relationship between bare ground and abundance of *Sehima nervosum*.

An example of the relationship between bare ground and occurrence abundance of *Sehima nervosum*, a perennial grass, is shown in Fig. 3.3. A summary of the relationships between plant abundance and richness with bare ground for all plants is shown in Fig. 3.4 *a, b*.

(a)



(b)

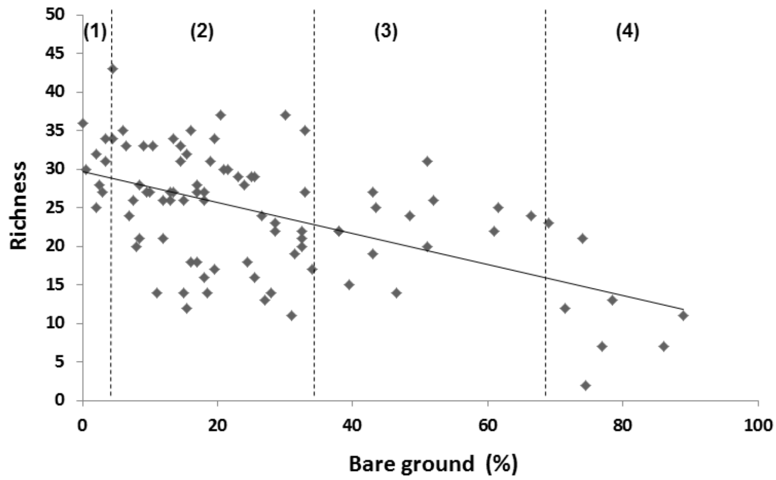


Figure 3.4 Relationship between bare ground and (*a*) abundance, (*b*) richness for all plant species

The MRBGI for Regional Ecosystem 10.5.5, where the projective foliage cover is <20%, (which comprises 61% of the regional ecosystem in the total distribution of the silver-leaved ironbark across the Desert Uplands bioregion) is shown in Figure 3.5. The curve is skewed towards the left indicating that the ecosystem has predominantly high ground cover. The area heavily impacted by grazing - those areas with <25% ground cover – represent only 4% of the entire ecosystem, and are mainly associated with stock watering points and fire scars. These low cover areas have low plant diversity.

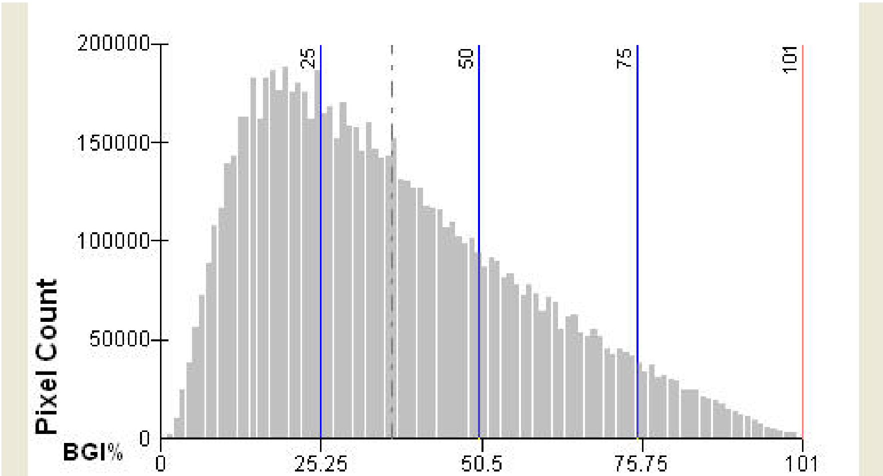


Figure 3.5 Histogram of MRBGI index for the silver-leaved ironbark ecosystem, with 4% of pixels with >75% bare ground, 19% of pixels with >50% and <75% bare ground, 40% of pixels with >50% and <25% bare ground and 37% of pixels with >25% bare ground.

3.5 Discussion

There is a significant relationship between bare ground and the MRBGI (Fig. 3.2). The MRBGI for the ecosystem across its regional distribution suggests that <4% of the area has <25% ground cover and the average ground cover for the ecosystem is 61% (see Fig. 3.5), which indicates that the ecosystem is in reasonable condition. It is likely that the annual forbs recorded in our study at the end of the wet season would not have been present at the time of the satellite capture during the dry season, which would account for the ground cover being higher than the index. The accuracy of the relationship between bare ground and the bare ground index also can be affected by fire scars, the hiatus between capture and sampling, variations in leaf litter, tree and shrub cover and soil cover.

The results confirm that there is a negative relationship between bare ground and the abundance of native plant species, including eight perennial grasses and palatable forbs such as the legume *Rhynchosia minima*. The native perennial grasses are relatively long-lived and, in an undisturbed environment, would dominate the ground layer, usually accounting for 70-90% of the available forage (Ash *et al*. 2001). These perennial grasses typically include *Bothriocloa ewartiana, Themeda triandra* and *Heteropogon contortus* which in other studies were all found to have a significant negative relationship with increasing bare ground (Patridge 2000; Ash *et al.* 2001).

This study also re-enforces findings of other studies which found that some wire grasses (i.e. the common *Aristida jerichoensis)* and wanderrie grasses (*Eriachne mucronata*) which are unpalatable to cattle have a positive relationship with bare ground (e.g. Ash *et al.* 2001). However, *Aristida calycina* had a very significant negative relationship with bare ground, which may be indicative of its adaptations to drought conditions, so our findings support the conclusion by McIntyre and Filet (1997): that the different species of *Aristida* have a range of responses to grazing. Native legumes, such as *Indigofera linnaei, Rhynchosia minima* and *Desmodium varians* decrease under heavy grazing pressure (Patridge 2000). Flannel weeds (*Sida altherophora* and *Sida fibulifera*) and pigweed (*Portulaca olearacea*) have a positive relationship with bare ground (Patridge 2000). Other species with positive relationships with bare ground such as *Gomphrena celosioides, Cynodon dactylon, Scleroleana birchii* and *Dactyloctenium radulans* are often identified as species that invade overgrazed pastures (Anderson 1993).

The relationship between plant abundance and richness with bare ground for the different species (Fig. 3.4 a, b) allows for some categorisation of response types. There appear to be four cover patterns; (1) ≤ 5% bare ground in which the number of plant species is between 30 and 60 species; (2) ≥5% and ≤35% bare ground where there is more variability in the number of plant species from 15 to under 60; (3) ≥35% and ≤70% bare ground where the range of plant species is between 25 and 50; and (4)≥ 70% bare ground where there is <20 plant species. The results show that high levels of ground cover are associated with a consistent richness and abundance of plant species. The areas with <5% bare ground are those areas most likely to have a light grazing history and will be important to maintain high plant biodiversity. Whether they are maintained within the grazed landscape or will need to be protected with incentives for permanent protection needs to be considered, as they will be lost, should grazing intensify in the Desert Uplands.

Some sites with high ground cover with a strong occurrence of *Sehima nervosum* were only 1.4km from water. Distance to water is not the only factor affecting plant diversity and cover as Pringle and Landsberg (2004) describe, and our finding that distance to water did not show a significant relationship with bare ground demonstrates, that many other factors influence grazing effects on ground cover such as paddock heterogeneity, stocking history and configuration of the paddock.

The relationship between grazing and plant species is not a simple linear one, and clearly one snapshot in time does not capture all the cumulative grazing history of a site. Areas with very high ground cover have a different plant composition from those with a moderate ground cover, where grazing may have reduced the dominant palatable perennial grasses, including *Themeda triandra* and *Bothriocloa ewartiana*. The decline of these species with grazing could be explained by their low or intermediate colonising activity and survival as these characteristics make them vulnerable under heavy grazing (McIvor 2007), allowing space for other species and thereby promoting species diversity. As grazing pressure increases further and/or bare ground increases the plant species diversity is reduced which produces a small suite of grazing tolerant species (Lunt *et al.* 2007).

Plant species such as *Dianella longifolia*, *Sehima nervosum* and *Themeda triandra*, which have significant negative responses to increasing bare ground, are worthy indicators of silver-leaved ironbark woodlands in good ecological condition. However, these results cannot be extrapolated outside this ecosystem because different outcomes may be caused by the variable responses of species along soil and climate variables, and the competitive effects of species along those gradients and the magnitude of grazing effects (Vesk and Westoby 2001). *Sehima nervosum* was not a dominant grass species at any sites in this study, but where it was a more important species in basalt woodlands of northern woodlands with more fertile soils than the Desert Uplands (Fensham and Skull 1999), it was absent from cattle –grazed sites even if they were moderately grazed. This indicates that *Sehima nervosum* is a very palatable species and is highly sensitive to cattle grazing.

The results of this study suggest that a bare ground index has the potential to provide some indication of the ecological integrity and plant diversity in this ecosystem. This is reinforced by the findings by Ward and Kutt (2009) that the ground cover temporal mean and variance are potentially useful indicators of disturbance to flora and fauna diversity, especially in homogenous ecosystems. Continued development of better information on the key indicator species, ground cover thresholds and relationships between ground cover and species associations will help provide better information about ecological condition and should help landholders implement management strategies that maintain ecological integrity of this rangeland ecosystem. Remotely-sensed mapping of bare ground could be developed into a powerful property-wide tool for landholders to annually fine-tune their grazing management and enable them to better maintain the ecological basis that underpins their grazing enterprises.

This study suggests that plant biodiversity in the Desert Uplands can persist under current grazing management. This conclusion is tempered by one major caveat. The land type has been extensively clearly particularly in the southern part of the bioregion. With clearing comes the spread of exotic grasses where clearing provides the opportunity for their establishment. The African buffel grass (*Cenchrus ciliaris*) is the main species used and it is well established that where it becomes dominant plant diversity is reduced (Fairfax and Fensham 2000; Jackson 2005). Unpublished quantified data from two sites at Moorinya National Park and another conservation reserve (Glen Innes near the town of Alpha) suggest that buffel grass is spreading without deliberate seeding or fire and grazing disturbance in the target land type (P.

Williams and R. Fensham, unpublished data). Although buffel grass was only found in 14% of our sites, we predict that the further spread of this exotic grass will have a greater impact on plant diversity than cattle grazing *per se*., as long as grazing intensity is not increased.

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3.6 Appendix

Significant relationships with bare ground are listed. For 25 Species with no significant relationships are listed with their frequency in parenthesis: Perennial Grasses**:** *Aristida jerichoensis* (282), *Sporobolus carolii* (14), *Eragrostis lacunaria* (169), *Eriachne mucronata* (217), *Chloris ventricosa* (17), *Eragrostis spartinoides* (10), *Heteropogon contortus (*60), *Digitaria brownii* (171), *Eulalia aurea* (14),), *Enneapogon lindeyanus* (85), *Digitaria ammophila* (38), *Chloris divaricata* (13),*Panicum effusum* (268), *Cymbopogon bombycinus*(14), *Cymbopogon refractus* (7), *Schizachyrium fragile* (26), *Enneapogon virens* (329), *Triodia pungens* (393), *Tragus australianus*(10), *Paraneurachne meulleri* (16),*Cenchrus ciliaris* (106), Perennial Forbs: *Scleroleana muricata* (70), *Scleroleana convexula* (28), *Sida rohlenae* (139), *Cynodon dactylon* (10), *Hibiscus burtonii* (74), *Sida atherophora* (115), *Senna artemisioides subsp. filifolia* (14), *Abutilon sp.* (38), *Phyllanthus sp.* (13), *Indigofera linifolia (*84), *Desmodium varians* (8), *Hybanthus enneaspermus* (21), *Spermacoce brachystemma* (9), *Tephrosia leptoclada* (7), *Wultheria indica* (16), *Chaemasyce drummondii* (33 *Calotis xanthosioidea* (33), *Boerhavia pubescens* (88),*Alternanthera nana* (30), *Cheilanthes sieberi subsp.sieberi* (13), *Peripleura hispidula* (17), *Rostellularia adscendens (14), Gossypium australe* (6), *Corchorus aestuans* (82),*Solanum ellipticum* (46),*Tricoryne elatoir* (13), *Zornia muriculata* (63), *Evolvulus alsinoides subsp.decumdens* (209),Annual grasses: *Aristida holathera* (147), *Perotis rara* (7), *Dactyloctenium radulans* (23), Annual Forbs: *Gomphrena celosioides (*6), *Portulaca oleracea (*12), *Portulaca pilosa* (41), *Oldenlandia mitrasacmoides* (45), *Heliotropium tanythrix* (21), *Alternanthera denticulata (20), Oldenlandia corymbosa* (26), Trees and Shrubs*: Canthium oleifolium* (33), *Acacia melleodora* (10), *Acacia tenuissima* (34), *Opuntia stricta* (16), *Carissa ovata (52), Acacia coriacea subsp. sericophylla* (107), *Stylosanthes scabra* (66), *Petalostigma pubescens* (15),

Species	**Constant**	**Slope**	**R2**	**Significance**	**Direction**
Perennial Grasses					
Aristida calycina var.calycina	7.46	-1.37	0.86	0	-
Sehima nervosum	4.03	-1.11	0.78	0.002	-
Themeda triandra	6.32	-1.12	0.71	0.004	-
Bothriochloa ewartiana	6.70	-1.26	0.68	0.007	-
Chrysopogon fallax	4.79	-0.83	0.65	0.008	-
Themeda avenacea	0.63	-0.17	0.64	0.009	-
Tripogon loliiformis	4.19	-0.84	0.57	0.019	-
Annual Grasses					
Eragrostis sororia	1.40	-0.37	0.84	0.001	-
Fimbristylis dichotoma	4.82	-0.91	0.62	0.012	-
Perennial Forbs					
Sida fibulifera	-0.87	0.58	0.83	0.001	+
Scleroleana birchii	-1.02	0.62	0.48	0.04	+
Dianella longifolia	0.71	-0.18	0.9	0	-
Phyllanthus fuernrohrii	1.08	-0.27	0.83	0.001	-
Brunoniella australis	2.84	-0.64	0.67	0.007	-
Galactia tenuiflora	0.45	-0.12	0.64	0.009	-
Rhynchosia minima	2.50	-0.62	0.58	0.017	-
Phyllanthus similis	0.61	-0.16	0.57	0.018	-
Tephrosia sp.	0.65	-0.19	0.55	0.026	-
Indigofera linnaei	0.68	-0.16	0.48	0.038	-
Stylidium eroirhizum	0.30	-0.07	0.45	0.049	-
Trees and Shrubs					
Maytenus cunninghamii	0.85	-0.20	0.71	0.004	-
Jasminum didymum subsp.didymum	0.43	-0.11	0.72	0.004	-
Grewia retusifolia	1.04	-0.29	0.61	0.013	-
Acacia excelsa	0.69	-0.17	0.47	0.042	-

Chapter Four

Birds as surrogates for ecosystem condition

4.1 Introduction

In this chapter the relationship between bird populations to ground cover and other environmental variables is explored within the context of the silver-leaved ironbark woodlands in the Desert Uplands bioregion. Over 26 insectivorous bird woodland species have been found to decline in many regions of south-eastern Australia (Watson 2011). These woodland dependent species have undergone widespread reductions in abundance and occurrence in southern Australia, is reflected in diminishing populations sizes and reduced distributional ranges (Watson 2011), and with declines even in intact tracts of woodlands (Stevens and Watson 2013). While these declines are attributed to habitat loss, fragmentation, habitat degradation and simplification, Watson's productivity-based hypothesis on the decline of woodland birds is that selective clearing and degradation of remaining woodlands has driven reductions in the biomass of decomposer communities in the soil and litter, thereby reducing food resources available to ground-feeding insectivores. This focuses on ecosystem condition attributes that would lead to the loss of food resources of which ground cover is a key indicator. Fisher and Kutt (2007) identified that on average 19% of bird species across five land types in Queensland and the Northern Territory increased with improved land condition and 17% decreased with deterioration in land condition. The evidence of the loss of insectivorous woodland birds in the southern part of their distribution with one of the likely causes being habitat simplification and degradation, justifies this study's attempt to demonstrate linkages of birds species abundance with ground cover.

4.1.1 Advantages and disadvantages of using birds as indicators

The main advantage in the usage of birds as indicator species is that they are easy to detect and observe using simple survey techniques which are capable of capturing information on several species during the same event (MacNally *et al.* 2004). The taxonomy of birds is well resolved and species are readily identifiable in the field (Furness *et al.* 1993). Most Australian birds have well known distribution, biology, ecology and life histories (Furness *et al.* 1993). Birds are near or at the top of the food chain, making birds sensitive to changes at lower levels (Furness *et al.* 1993; MacNally *et al.*2004; Gregory *et al.* 2005). Many birds pollinate or disperse the seeds of plants, playing a critical role in ecosystem maintenance (Chambers 2008).

There are some characteristics of birds which limit their usefulness as indicators. The high mobility of birds may make it difficult to link responses of birds to specific conditions or stressors on the ground (MacNally *et al.* 2004; Gregory *et al.* 2005). The life span of different species of birds ranges from a few years to a few decades in length (Garnett and Crowley 2000) which can mean that birds are capable of signalling impacts over a long-term exposure, but are less suitable for indicating short-term disturbances (Gregory *et al.* 2005). Birds often respond to secondary effects of stressors as mentioned above in reference to grazing when they respond to changes in vegetation structure as a result of grazing. Additionally, birds have behavioural and physiological traits which make them less sensitive to ecosystem changes than other taxa. For example, birds can regulate fat stores, reducing sensitivity to adverse seasonal conditions (Furness *et al.* 1993).

These factors mean that the use of birds as indicators of bird diversity, ecosystem condition and the impacts of grazing on the environment needs to be adopted with caution. Carignan and Villard (2002) recommend that where species representing various taxa and life histories are used, their selection should be based on sound quantitative data from the region of interest and that caution should be exercised when interpreting trends to separate actual signals from fluctuations that may be unrelated to ecosystem deterioration.

In this study, the main aims were to examine the use of birds as indicators of ecosystem condition; bird diversity within the silver-leaved ironbark woodland ecosystem and grazing impacts.

4.1.2 Guilds as indicators

The Guild (collections of species that exploit the same class of environmental resources in a similar way) indicator approach assumes that all species in a guild respond similarly to environmental change (Root 1967). However, the occurrence of individual species may give little information about the overall guild abundance or diversity because of the different environmental requirements of each species. For example, species that make up an insectivorous foraging group may differ markedly between or even within habitat types, whereas the overall guild abundance and diversity may not differ because of species substitutions.

To a lesser extent, these difficulties also exist when the overall abundance of a guild, rather than an individual species, is used as an indicator (Verner 1984). Large increases or decreases in one or two species can mask the decline or loss of others in the guild. Consistency of population responses among species in a guild is important for guilds to be used as an indicator.

Nevertheless, guilds are useful for evaluating the collective response of multiple species to changes in ecological conditions. In this study, species are categorised into functional groups based on criteria that traditionally define a guild in the form of foraging, migratory and dietary groups (Woinarski and Tidemann 1991; Kutt 2004). Additionally, an attempt to define functional groups based on habitat – in terms of their use of shelter and feeding vegetation stratum (habitat assemblages) has been examined in this study. These functional groups are not guilds as defined by Root (1967) because they do not reflect partitioning of specific ecological resources. Habitat as defined by Canterbury *et al.* (2000) is used, which is, in the narrowest sense, solely vegetation structure rather than the full array of physical, chemical and biological factors in the environment.

Birds have often been used as indicators of biodiversity and species richness (Block *et al.* 1987; Chase *et al.* 2000; Mikusinki *et al.* 2001; Kati *et al.* 2004; Fleishman *et al*. 2005); and birds have been used as surrogate taxa for biodiversity to prioritise areas for conservation management (Moore *et al*. 2003). In a study in Greece Kati *et al.* (2004) found that birds' diversity had a good correlation with woody plant diversity. Fleishman *et al.* (2005) concluded that a small set of bird species with either presence or absence patterns could be indicators of biodiversity if they were found to be correlated functionally with species richness of a large group of organization within the ecosystem. They found that species richness of given taxonomic groups would be more accurately predicted on the basis of species drawn from that taxonomic group. Chase *et al.* (2000) found that a diverse suite of species representing the range of variation in avian communities found in coast sage shrub habitats were useful indicators of the ecosystem diversity.

Birds have previously been considered as indicators of the condition of ecosystems and, in particular, of the condition of rangelands (Bradford *et al.* 1998; Whitford *et al.* 1998). Also birds have been considered as indicators to disturbances such as grazing (Bock and Webb 1984: Knopt *et al.* 1987). In both these cases, birds were seen to respond to a change in the vertical and horizontal vegetation structure as a result of grazing impacts rather than to grazing directly.

4.1.3 Desert Uplands avifauna

Kutt's (2004) fauna survey and studies in the Desert Uplands bioregion found there to be a representation of 229 bird species, 34% of Queensland's avifauna, with 24 birds of conservation significance. The Desert Uplands as a semi-arid bioregion has intermediate species richness, reflecting a transitional location. As the major direction of speciation is via Bassian and Torresian routes towards arid areas (Schodde 1982), these tropical savanna woodlands are more closely related to the source than the arid destination. The Desert Uplands is central to some vicariant speciation associated with the Great Dividing Range. Kutt (2004) suggests that the bird community of the Desert Uplands bioregion may not be at equilibrium and may vary in response to the periodic changes in the continuum of available resources.

The hypotheses tested are the following:

I. There are significant relationships between some bird species and ground cover.
II. There are significant relationships between bird species and the environmental variables associated with ground cover.
III. The dietary and foraging bird functional groups are effective indicators of ground cover and key woodland environmental variables.
IV. 'Habitat assemblage' [2]bird groups are useful as indicators of ground cover and consequently biodiversity condition.
V. The ground cover index is an effective indicator of bird diversity in an open woodland ecosystem.

[2] Habitat assemblage is a construct devised by the author to encapsulate both nesting and feeding requirements into distinct bird groups

4.2 Methodology

4.2.1 Field Measurements

The field sites and methodology described in Chapter Three was used, a) to collect environmental variable information and b) to complete bird surveys at the same sites. While these sites were not selected by their distance from water, this distance was measured for each site. The sites are located over 25 distinct properties and reflect a variety of management practices. The details of paddocks in which the sites were located were recorded, such as the size of the paddock, the vegetation type configurations in the paddocks, the number of waters in the paddock and the number and type of stock and duration of stocking in the paddock. This information was collected to identify whether other infrastructure and management actions (apart from the effects of grazing pressure on ground cover) were significant on bird populations.

At each site, physical surveys were conducted to record a range of bird population and environmental characteristics. Surveys recorded both sightings and calls. The surveys were undertaken in May 2007 at the end of the wet season. Each hectare site was surveyed for 45 minutes in either the morning (0600-1100h) or the evening (1500-1800 h) within which the vegetation plot was placed. The bird data was collected by one experienced observer. Nomenclature followed Christidis and Boles (2008), with both bird species and frequency data recorded. Species were assigned to bird functional groups after Woinarski and Tidemann (1991).

4.2.2 Statistical Analysis

For each site the bird species, richness, evenness and abundance was calculated. Richness is the number of species per site. Evenness is a measure of the relative abundance of the different species making up the richness of a site. As species richness and evenness increase, so diversity increases. Simpson's Index is a measure of diversity which takes into account both richness and evenness (Simpson 1949). Simpson's Index measures the probability that two individuals randomly selected from a site will belong to the same species. With this index, O represents infinite diversity and 1, no diversity. The following formula was used to calculate the diversity of each site:

$$D = \sum \frac{n(n-1)}{N(N-1)}$$

Where n = the total number of organisms of a particular species and N = the total number of organisms of all species.

An index of abundance of a species at a site was calculated as the frequency count of a species multiplied and divided by the highest frequency count of the species found at any site to give a 0-1 abundance value across all the sites.

The species richness, abundance and evenness relationship was first assessed by bi-variate regression with ground cover and then with the key environmental variables. Following this, the relationship between ground cover and the guild groups and habitat assemblage abundance data was assessed by bi-variate tests with ground cover. Then the species, guilds and habitat assemblages across the sites were assessed via constrained Canoco ordinations and PATN assemblage ordination with ground cover and key environmental variables.

4.2.3 Pattern Analysis

A site by species matrix was created to test for bird assemblage patterns using the presence-absence data from the 45 minute site sampling periods. Presence-absence data is considered to be more reliable data than abundance data for determining environmental relationships as noted by previous authors (e.g. MacNally *et al.* 2004; Matern *et al.* 2007; Pavey and Nano 2009). Rare species (those occurring at less than five sites) and sites with less than two species were excluded from the data matrix.

The site species assemblages were assessed using PATN V3.12 (L. Belbin, Griffith University and CSIRO, Australia) for their associations, classification and ordination. This analysis was conducted to assess whether ground cover and the associated environmental variables can explain the species variation between sites. The association of the cube-root transformation of the species data (to normalise and smooth the distribution) between sites was done using a Bray and Curtis association measure. The sites were classified into four groups using an Agglomerative hierarchical classification which weights objects equally. A semi-hybrid multi-dimensional scaling ordination (SHDS) was carried out (multi-dimensional scaling – (MDS) - compares the Bray and Curtis association values using regression with the corresponding Euclidean distance values). The MDS algorithm positions the sites within the data matrix to improve the fit with Euclidean distances. The points are then moved and the Euclidean distance is re-measured, re-iterating the process until any site movement decreases the stress/ improves the ordination fit. In the ordination plot the 23 environmental variables are squeezed into three dimensions. The measure of the fit is the stress. The site groupings and bird groupings created by this association, classification and ordination were then interpreted by a Bray and Curtis histogram and attendant statistics of this association. Dendrograms display the classification. Box and Whisker plots displayed how the environmental variables affect the groupings and how the species or guild groups affect the groupings. These plots were displayed by decreasing Kruskall-Wallis values (one-way analysis of variance) – thereby the most discriminating variables are the highest. Anosim, being the analysis of the similarity within and between group associations, was also carried out. Principal Component Correlation through multiple linear regression ensured that the ordination that best reflects the environmental variables – those with the highest R^2 values. The environmental variables used in the PCC are then randomly re-allocated between sites by Monte Carlo attribution in the ordination (MCAO) and the multiple linear regression (PCC) was re-run.

The direct relationship of bare ground and other environmental variables with bird groups was explored. The groups included taxonomic, seasonal, foraging, dietary, and nesting groups. As well, habitat assemblage groups were identified which combined the nesting and foraging characteristics of each bird to form four groupings – 1) birds that feed and nest in trees, 2) generalists, raptors, pounce feeders – ground foragers that nest in trees or shrubs, 3) ground foragers that nest on the ground, and 4) birds that feed in trees and nest on the ground and in shrubs.

This allowed four habitat group assemblages to be tested against the range of environmental factors of interest. The sheltering and feeding requirements of a bird species were combined to group birds into their habitat requirements. These 'habitat assemblages' groups can be useful to understand the relationships between birds and ground cover and the associated environmental variables.

Relationships between species composition and environmental variables were investigated by constrained ordination using canonical correspondence analysis (CCA) run in CANOCO 4.5 (ter Braak and Šmilauer, 2002). Twenty-three quantitative environmental variables that were predicted a priori to have the potential to impact on bird species composition were used in the CCA (Table 1).

Table 4.1 Environmental variables used in the canonical correspondence analysis (CCA) and the PATN assemblage assessment

Environmental Variable	Type of Data
Live Basal Area	m2/ha
Total Basal Area	m2/ha
Bare ground	percentage cover
Green leaf	percentage cover
Litter	percentage cover
Cryptogam	percentage cover
Midstorey green leaf	percentage cover
Midstorey branch	percentage cover
Overstorey green leaf	percentage cover
Overstorey branch	percentage cover
Total Green leaf	percentage cover
Perennial green leaf	percentage cover
Ground cover	percentage cover
Distance to water	metres
Easting	
Northing	
Stock density	Adult Equivalent/ha
Water density	Ha/water

Bothriocloa ewartiana	percentage cover
Cenchrus ciliaris	percentage cover
Triodia pungens	percentage cover
Grass height	cm

4.3 Results

4.3.1 Species richness, evenness and abundance

A total of 111 bird species were recorded from the 91 sites of which 53 species occurred at more than 6 sites (see Appendix A). These bird species provided representation from 35 taxonomic groups out of a possible 44 groups, and included seven foraging groups, six dietary groups, and five migratory groups (see Appendix B). Those species recorded from the greatest number of sites were: Weebill (61), Striated pardalote (51), Yellow-throated miner (50), Rufous whistler (42), Brown treecreeper (40), Crested bellbird (39) and Jacky winter (36). One endangered bird species, the Black-throated finch, was sighted at three sites on one property and one species, the Diamond firetail, was sighted at the northern extent of its known distribution.

The bird richness, abundance and evenness were examined relative to those species that occurred at more than five sites.

Table 4.2 Summary data for diversity measures of birds in the silver-leaved ironbark woodlands calculated for the 91 sites with 5 or more species recorded

Variable	Minimum	Maximum	Mean	SD
Species Richness	2	27	10.71	± .4.88
Species Abundance	3	140	29.52	± 14.46
Species Evenness	0	0.857	0.147	± 0.139

4.3.2 Hypothesis I

The first hypothesis is tested with the initial data analysis.

I There are significant relationships between some bird species and ground cover.

The abundance of four bird species significantly increased with increasing ground cover, namely the rufous whistler, variegated fairy-wren, hooded robin and grey shrike thrush. The abundance of thirteen bird species significantly decreased with increasing ground cover, namely the pale headed rosella, peaceful dove, crested pigeon, galah, rainbow lorikeet, Torresian crow, zebra finch, red-winged parrot, Australian raven, white-plumed honeyeater, magpie lark, spiny-cheeked honeyeater and blue-faced honeyeater. These regression results are shown in Table 4.3.

Table 4.3 Bi-variate regression results for bird species abundance against ground cover percentage

Species	**Constant**	**Slope**	**R2**	**significance**	**direction**
Pale headed rosella	0.301	-0.003	0.158	0	negative
Peaceful Dove	0.294	-0.003	0.229	0	negative
Crested Pigeon	0.191	-0.002	0.108	0.001	negative
Rufous Whistler	-0.101	0.004	0.126	0.001	positive
Galah	0.159	-0.002	0.107	0.002	negative
Rainbow lorikeet	0.151	-0.002	0.088	0.004	negative
Torresian crow	0.225	-0.002	0.091	0.004	negative
Zebra finch	0.233	-0.002	0.091	0.004	negative
Red-winged parrot	0.171	-0.002	0.077	0.008	negative
Australian raven	0.376	-0.003	0.073	0.01	negative
White-plumed honeyeater	0.206	-0.002	0.073	0.01	negative
Magpie lark	0.19	-0.002	0.071	0.011	negative
Spiny-cheeked honeyeater	0.202	-0.002	0.058	0.021	negative
Blue-faced honeyeater	0.132	-0.001	0.047	0.04	negative
Variegated fairy wren	-0.056	0.001	0.046	0.041	positive
Hooded Robin	-0.053	0.002	0.04	0.058	positive
Grey shrike thrush	-0.054	0.002	0.035	0.077	positive

When the results of 10 averaged bird counts were regressed against the average of 10 groupings of ground cover percentages (see Table 4.4), the data shows that there are species that have significant positive and negative relationships with ground cover in terms of species abundance. When comparing the results of species abundance in both of the above manners, some consistent relationships can be ascertained. The five species with the most significant positive relationships with ground cover are the Rufous whistler, Grey shrike-thrush, crested bellbird, Grey fantail and the Red-browed pardalote and the five species with the most significant negative relationship with ground cover are the Australian raven, galah, crested pigeon, peaceful dove and red-winged parrot as per Table 4.4.

Table 4.4 Regression results of average abundance of bird species against ground cover %

Species	Constant	Slope	R2	Significance	Direction
Rufous Whistler	-0.41	0.017	0.763	0.002	positive
Australian raven	1.508	-0.014	0.089	0.006	negative
Galah	2.171	-0.025	0.66	0.008	negative
Crested pigeon	2.061	-0.021	0.615	0.012	negative
Peaceful dove	2.987	-0.035	0.605	0.014	negative
Red-winged parrot	0.941	-0.01	0.598	0.014	negative
Pale-headed rosella	2.837	-0.029	0.567	0.019	negative
Rainbow lorikeet	1.073	-0.011	0.523	0.028	negative
Torresian crow	2.908	-0.026	0.51	0.031	negative
Grey shrike thrush	-0.155	0.004	0.506	0.032	positive
White-plumed honeyeater	3.184	-0.034	0.503	0.032	negative
Blue faced honeyeater	0.605	-0.007	0.478	0.038	negative
Magpie lark	1.888	-0.02	0.453	0.047	negative
Spiny-cheeked honeyeater	0.883	-0.009	0.423	0.058	negative
Red-browed pardalote	-0.022	0.007	0.42	0.059	positive
Crested Bellbird	0.022	0.01	0.411	0.063	positive
Grey fantail	-0.059	0.003	0.342	0.098	positive
Zebra finch	4.248	-0.039	0.341	0.099	negative

4.3.3 Hypothesis II

II There are significant relationships between bird species and the environmental variables associated with ground cover.

The constrained ordination by canonical correspondence analysis (Figure 4.1) displays the relationship of the species with the most significant relationships with ground cover and their relationship to other environmental variables. The Crested bellbird and Rufous whistler display a strong relationship with environmental variables associated with a good condition ecosystem: mid-storey over-storey, live basal area and litter, whereas the Yellow-throated miner is strongly associated with bare ground and northing. The latitudinal association could also relate to a warmer and drier climate preference for yellow-throated miners.

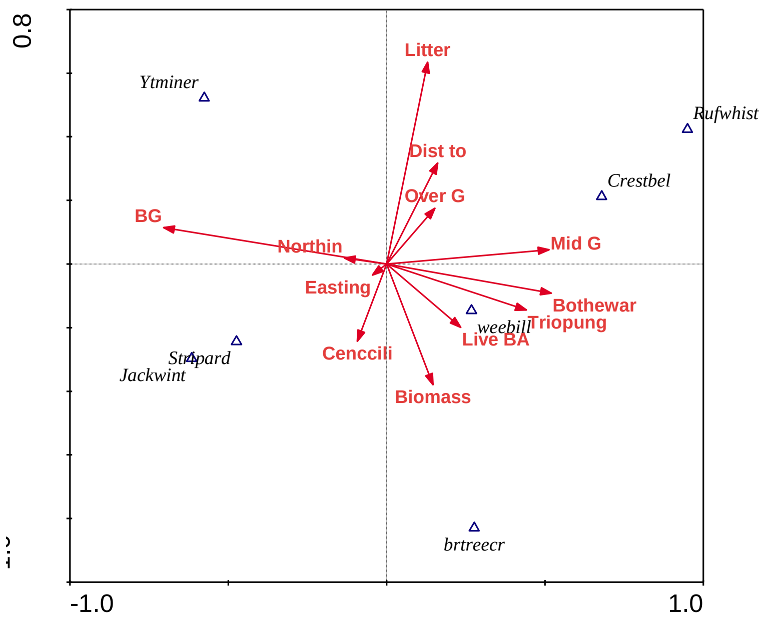


Figure 4.1 CCA ordination diagram showing the birds species, which were most sighted in the study in relation to environmental variables. Arrows represent continuous variables. Increasing arrow length indicates a stronger correlation with environmental gradients. More important variables are further from the origin. Abbreviations: Ytminer = yellow-throated miner, Stripard = striated pardalote, Jackwint = jacky winter, Crestbel = crested bellbird, Rufwhist = rufous whistler, brtreecr = brown treecreeper, BG = % bare ground, Dist to = Distance to water, Over G = Overstorey greenleaf, Mid G = Midstorey greenleaf, Live BA = Live Basal Area, Bothewar = *Bothriocloa ewartiana;* Tripung = *Triodia pungens*, and Cenccili = *Cenchrus ciliaris*

As shown in Figure 4.1, the most influential variables in the ordination are the abundance at the site of the native perennial grass *Bothriocloa ewartiana,* litter, amount of leaf cover in the over-storey and mid-storey and distance to water which are all on the opposite side to the environmental variables associated with grazing – ha/water and AE/ha.

The following two PATN outputs show the results of the Bray and Curtis association, the classification and ordination in Figures 4.2 and 4.3.

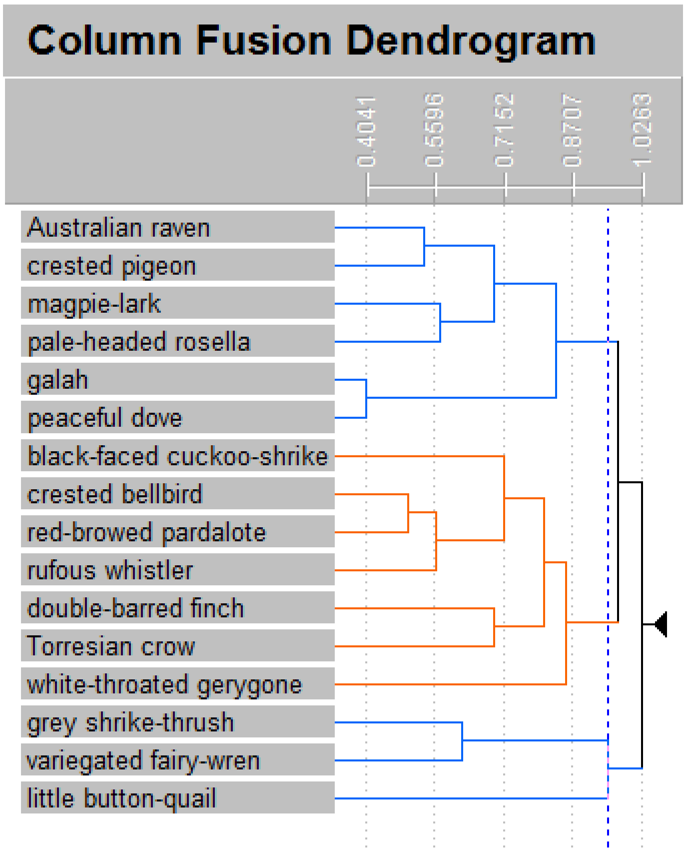
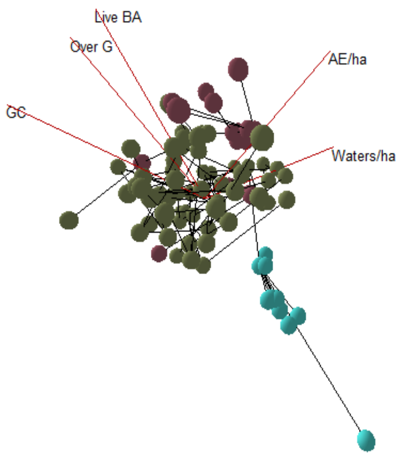
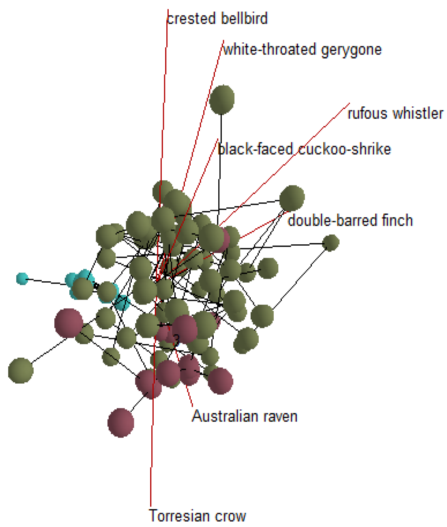


Figure 4.2 Dendrogram of three species groups as associated by environmental variables.

The dendrogram in Figure 4.2 clearly shows three groupings of species by the key environmental variables. The top group shows species mainly associated with more disturbed habitat and low ground cover. The second group are associated with undisturbed habitat and high ground cover. The third group are related to highly undisturbed habitat and high ground cover. In Figure 4.3b the Australian Raven and Torresian crow are in the same area as these variables associated with disturbance and low ground cover and all the other species fall in the area aligned with environmental variables associated with good condition i.e, over-storey and live basal area.



(a)



(b)

Figure 4.3 Ordination plot shows the relationship in terms of (a) the significant environmental variables and (b) the birds with significant relationships with ground cover. The colours relate to the dendrogram shown in figure 2 – here burgundy is the top group, grey green the middle group and light blue the bottom group of the dendrogram.

The ordination plots in Figure 4.3 (b) shows the spatial placement of five species from the middle group in the dendrogram and two species from the top group of the dendrogram in relation to environmental variables with the waters and stocking rate (in Figure 4.3 (a)) determining the position of the Torresian crow and Australian raven (in Figure 4.3 (b)). Therefore, hypothesis two is accepted.

4.3.4 Hypothesis III

III Groups of birds in the dietary and foraging guilds are effective indicators of ground cover and woodland ecosystem condition.

The initial tests compared the number of birds in each guild at the sites against the ground cover and key environmental variables. The following tables and figures present the results of bi-variate regression analysis of dietary and foraging groups with ground cover.

4.3.4.1 Dietary group relationship to ground cover

The insectivores are positively related to ground cover while the granivores, generalists, nectar feeder and frugivores are negatively related to ground cover (see Tables 4.5 a) and b). With respect to abundance of dietary groups, in Table 4.5 b) the granivores, nectar feeders and generalists have a significant negative relationship with ground cover and the insectivores have a significant positive relationship with ground cover.

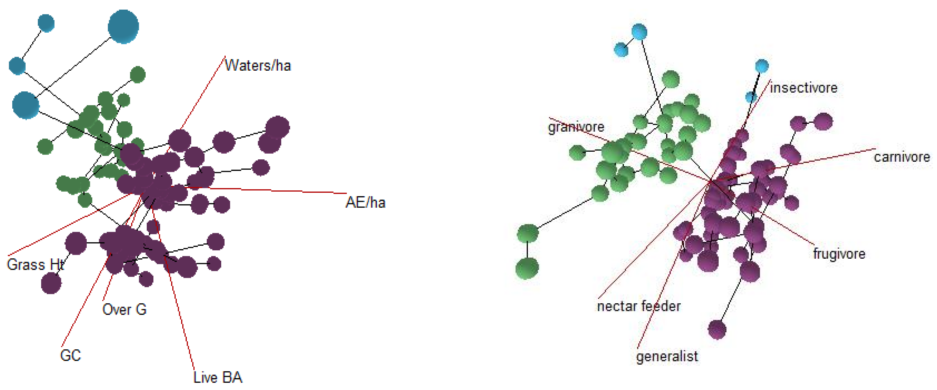
Table 4.5 a) Regression results for dietary groups' richness against ground cover percentage and b) Regression results for abundance of dietary groups against ground cover

a) Dietary Richness

Group	Constant	Slope	R^2	Significance	Direction
Granivore	12.325	-0.127	0.148	0	negative
Generalist	32.092	-0.253	0.119	0.001	negative
Nectar feeder	3.202	-0.038	0.119	0.001	negative
Frugivore	0.725	-0.008	0.076	0.008	negative
Insectivore	6.134	0.08	0.5	0.034	positive

b) Dietary Abundance

Guild	Constant	Slope	R^2	Significance	Direction
Granivore	0.685	-0.007	0.255	0	negative
Nectar feeder	0.151	-0.002	0.088	0.004	negative
Insectivore	1.151	0.014	0.044	0.045	positive
Generalist	3.107	-0.016	0.04	0.057	negative



(a) (b)

Figure 4.4 Ordination plot shows the relationship of Dietary groups with a) the environmental variables associations and b) the dietary groups association by sites

In the ordination of the dietary groups in Figure 4.4, the granivores are associated with low ground cover and are nearer the stocking variables such as AE/ha and waters/ha than the other groups.

4.3.4.2 Foraging group's relationship to ground cover and significant environmental variables

Table 4.6 a) Regression results for foraging groups' richness against ground cover percentage and b) Regression results for abundance of foraging groups against ground cover

a) Foraging Richness

Guild	Constant	Slope	R^2	Significance	Direction
General	17.146	-0.157	0.134	0	negative
Ground and low shrub	22.179	-0.188	0.103	0.002	negative
Pounce feeder	3.584	-0.026	0.04	0.056	negative

b)Foraging Abundance

Guild	Constant	Slope	R^2	Significance	Direction
General	1.386	-0.011	0.095	0.003	negative
Ground	1.464	-0.009	0.049	0.036	negative

With respect to the richness and abundance of foraging groups, the ground and low shrub feeders and pounce feeders have significant negative relationships with ground cover as in Table 4.6.

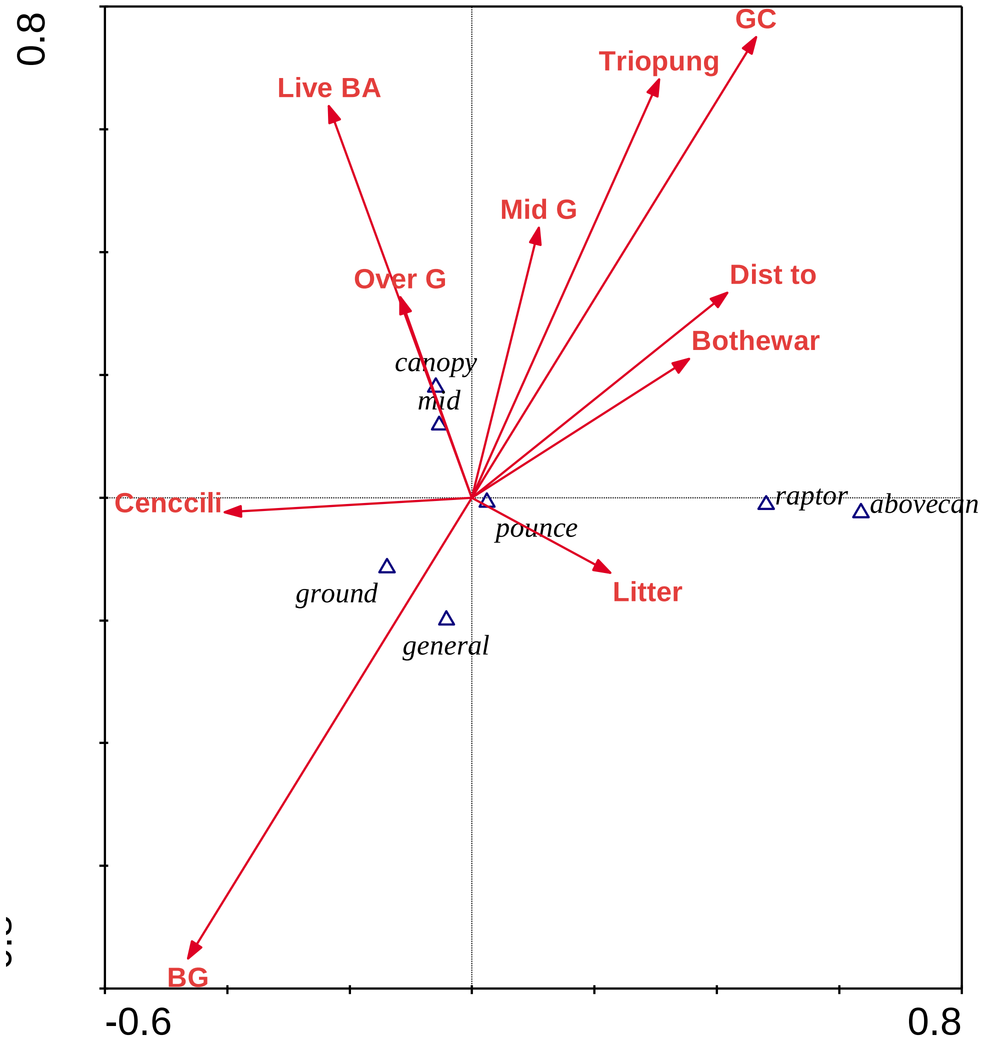
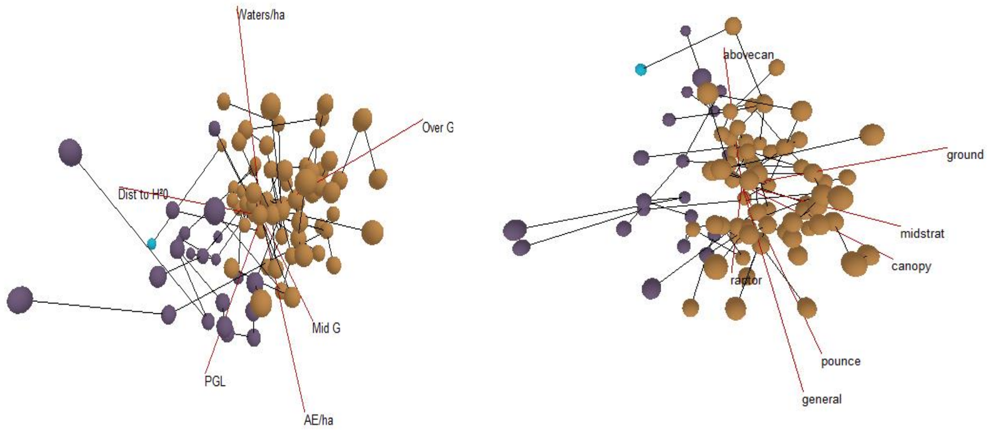


Figure 4.5 The CCA ordination diagram shows foraging group distribution in relation to environmental variables. Arrows represent continuous environmental variables. Increasing arrow length indicates a stronger correlation with environmental gradients. More important variables are further from the origin. Abbreviations: Live BA = live basal area, Over G = Overstorey green leaf, Mid G = Midstorey greenleaf, Triopung = *Triodia pungens*, GC = % ground cover, Dist to = Distance to water, Cenccili = *Cenchrus ciliaris*, Bothewar = *Bothriocloa ewartiana*, BG = % Bare Ground, mid = mid stratum, pounce = pounce feeder, abovecan = above canopy

The constrained ordination in Figure 4.5 shows that ground feeders and generalists have a strong positive relationship to bare ground while other groups are associated with other variables in keeping with good ecological condition.



(a) **(b)**

Figure 4.6 Ordination of foraging groups

The above canopy feeders are closely associated with the over-storey and the mid-stratum with the mid-storey variables as would be assumed as shown in Figure 4.6. No groups within both the dietary and foraging group show significant positive relationships with ground cover, therefore hypothesis three is rejected.

4.3.5 Hypothesis IV

IV 'Habitat assemblage' [3]bird groups are useful as indicators of ground cover and consequently biodiversity condition.

[3] Habitat assemblage is a construct devised by the author to encapsulate both nesting and feeding requirements into distinct bird groups

The group of birds that nest on the ground and feed in trees are very significantly positively related to ground cover and, conversely, the pouncers, raptors and generalists are significantly negatively related to ground cover.

However, in the constrained ordination two groups have a strong relation to the environmental variables associated with good ecosystem condition, namely high basal area, over-storey cover and mid-storey cover. These groups are those that nest on the ground and feed in trees as well as the group that feeds and nests in trees. Birds that fall into this group include the rufous whistler and black-faced cuckoo-shrike.

Table 4.7 Regression results for abundance of habitat groups against ground cover percentage

Habitat group	Constant	Slope	R^2	Significance	Direction
Pouncers and raptors that nest in trees	0.207	0.012	0.339	0.001	-
Feed on the ground and nest in trees	0.261	-0.003	0.095	0.003	+
Generalists that nest in trees	0.424	0.008	0.057	0.022	-

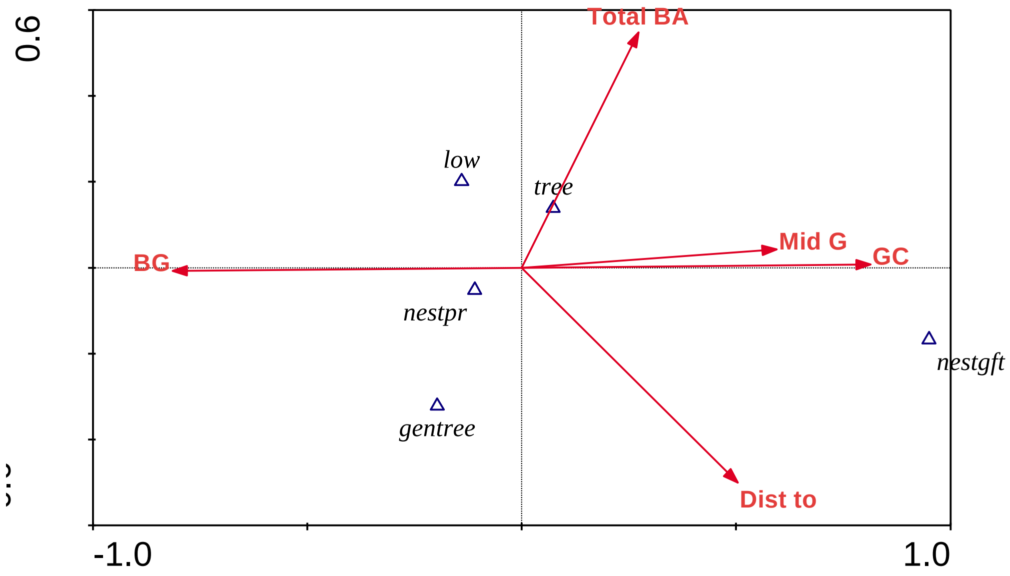


Figure 4.7 CCA ordination diagram shows bird habitat assemblage distribution in relation to environmental variables. Arrows represent continuous environmental variables. More important variables are further from the origin. Abbreviations: tree= nest and feed in trees, low = nest on ground or low shrubs and feed on ground or low shrubs, nestpr = nest in trees and pounce feeders or raptors, gentree = generalist feeder that nest in trees, nestgft = nest on the ground and feed in trees.

The constrained ordination in Figure 4.7 shows the habitat group that nest on the ground and feed in trees as having the positive relationship to ground cover. Additionally, no group is strongly related to bare ground. Additionally, within the habitat assemblage groups of birds, those that nest on the ground and feed in trees are absent from sites with low ground cover as shown in the two way table in Figure 4.8. Therefore hypothesis four is accepted.

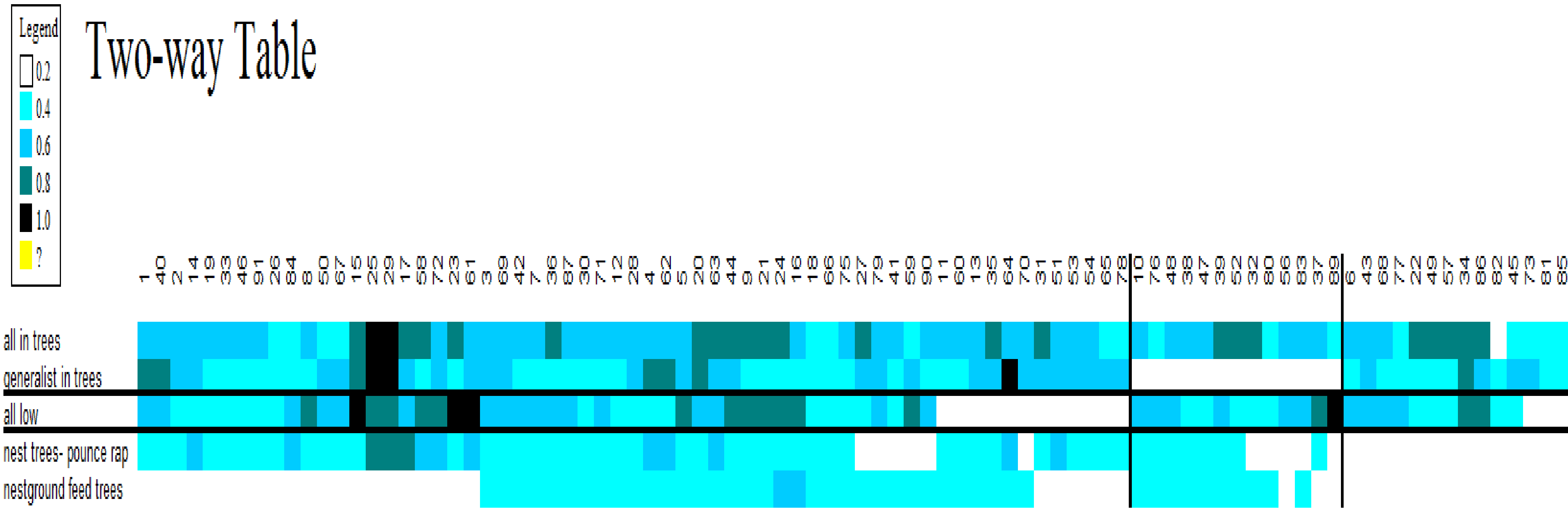


Figure 4.8 Two-way matrix of habitat assemblage groupings by site groupings of ground cover and environmental variables**;** with the lowest ground cover on the right side and the highest ground cover (also the largest group of sites). The darker colours signify increased indexed numbers of species present.

4.3.6 Hypothesis V

VI The ground cover index is an effective indicator of bird diversity in an intact open woodland ecosystem.

The following three graphs in Figure 4.9 show that ground cover does not have a significant relationship with bird species richness or abundance, but does have a significant but weak relationship with Simpson's Diversity index (R^2=.056, sig 0.024).

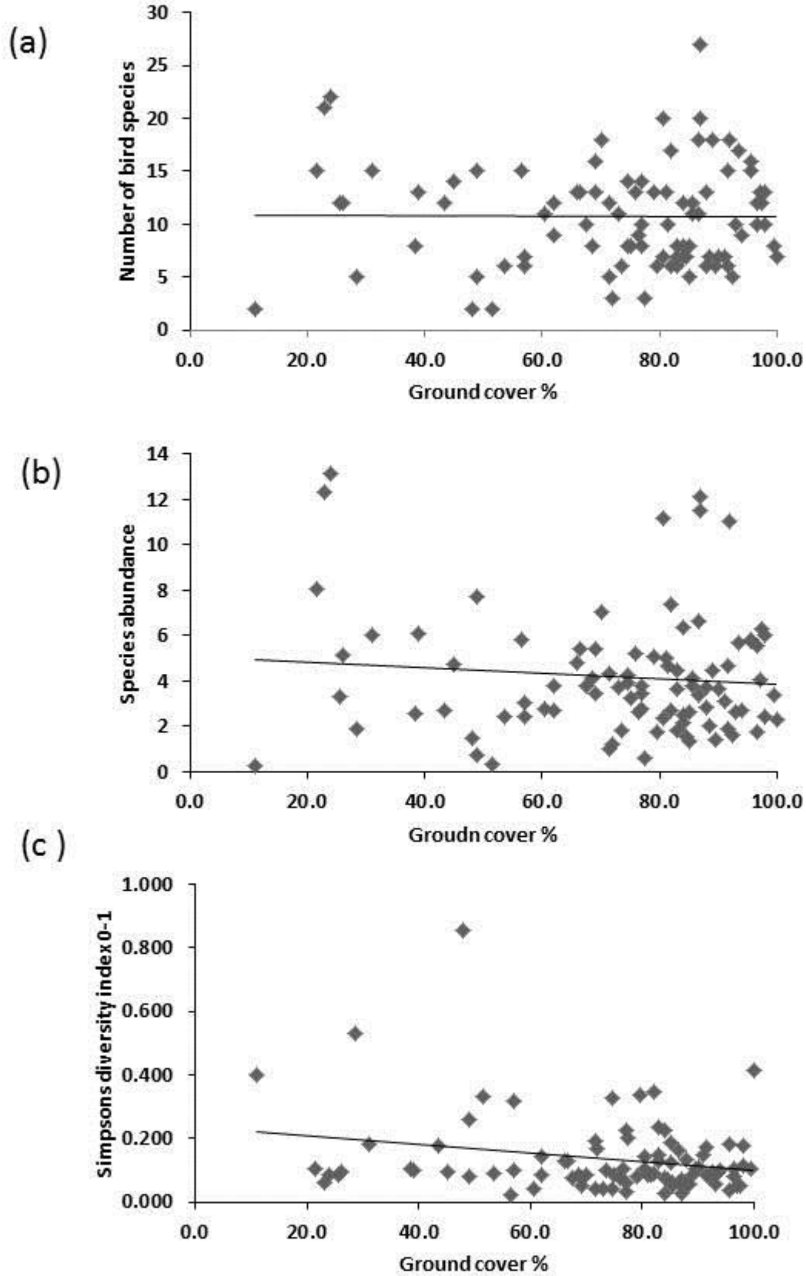


Figure 4.9 Graphs show the bird species (a) richness, (b) abundance and (c) diversity by Simpson's index abundance against ground cover %

Ground cover is indicative of the species diversity but not abundance or richness.

While hypothesis five is accepted, this hypothesis should be tested more robustly.

4.4 Discussion and conclusions

The results of the statistical tests allow the different hypotheses to be examined in turn. The first hypothesis was that there are significant relationships between some bird species and ground cover. The results of the regression analysis between different measures of bird species and ground cover (Tables 4.3 and 4.4) show that this hypothesis can only be accepted consistently for some species; five species such as the rufous whistler had significant positive relationships with increasing ground cover. However, there are thirteen species where a significant negative relationship with increasing ground cover was identified. The abundance of these five species could be used with other biodiversity measures to ascertain the ecological condition of the silver-leaved ironbark woodlands. Low ground cover is associated with a simplification and homogenisation of the ecosystem and reduced ecosystem function (Tongway *et al.* 2003). However, high ground cover will always need to be field verified as to its content of native perennial grasses versus introduced species such as buffel grass that would negatively impact on plant diversity and by consequence of ecosystem function loss; there would be an associated loss of key invertebrate foraging sources (Watson 2012). These results are confounded by the location of low ground cover site near to artificial watering points.

The dietary groups of granivores, frugivores and carnivores, in terms of richness and abundance, are indicative of low ground cover and poor condition while the insectivores are indicative of high cover and good condition. The canopy feeders foraging group is more abundant in high cover, while birds that forage on the ground or in low shrubs are more abundant in low cover. With respect to the habitat assemblages, the pouncers and raptors that nest in trees are more abundant with low cover, while the birds that feed on the ground and nest in trees are more abundant with high cover.

These results demonstrate that insectivorous resident birds that feed in trees and nest on the ground are probably an appropriate group to use as an indicator bird species for the condition of the silver-leaved ironbark woodland. This group can also be an effective indicator of a loss of ground cover resulting from stock grazing pressure because as the habitat decreases in quality, consequently their dietary and shelter requirements disappear. Furthermore, this habitat group, which includes the spotted pardalote, rufous song lark, rufous whistler, Inland thornbill and speckled warbler, are a specialist group with respect to their feeding and nesting requirements, and are sensitive to changes in the environment.

Some strong caveats to these conclusions should be noted. As this study is based on a one-off survey, albeit of ninety-one sites with varying environmental values, these results need further investigation. Indeed, Perry *et al.* (2012) found that repeated sampling over multiple days and at different times of the day provides the best estimate of species richness at a site and improved detectability. To have complete confidence in this habitat group, one would have to survey these sites again under varying grazing regimes and climatic conditions over several years. Another complication demonstrated by this analysis is that there was unequal proportion of sites with low ground cover and consequently these sites are under-represented as demonstrated in the two-way table analysis between species and sites. Nevertheless, these two-way tables demonstrate the strong relationships some species have with high and low ground cover.

Rufous whistlers and grey shrike thrushes display consistent positive regression correlations with increasing ground cover, in both richness and abundance which makes them good indicator species to monitor ground cover condition. This response is consistent with several other studies of bird response to grazing impacts in rangeland woodlands (Woinarski and Ash 2002; James 2003; and Hannah *et al.* 2007).

From the CCA analysis, it is established that other environmental values are correlated with increasing bare ground – such as: loss of litter, reduction in the prevalence of native perennial grasses such as *Bothriocloa ewartiana* and *Triodia pungens*, less mid-storey cover and less tree basal area. With increasing bare ground comes a simplification of the ecosystem. These results demonstrate that birds are affected by grazing pressure and the loss of ground cover, which is reaffirmed by other studies in the region (Kutt 2004; Hannah *et al.* 2007) and in woodlands across Australia (Martin and Possingham 2004; and Maron *et al.* 2005).

Work by Pavey and Nano (2009) suggests that arid Australia bird assemblages are affected more by vegetation patterns than by disturbance and resource pulses. However, within vegetation type variance in habitat quality may prove to affect bird assemblages in a subtle, but measurable manner. Indeed, it is possible that over several years the bird species composition of these woodlands could change purely in response to climatic variation; that there therefore would be changes in the habitat resources available to bird species, as was found in a study of sites in bulloak woodland in Victoria by Maron *et al.* (2005). Additionally, in fragmented landscapes, the isolation of patches, habitat modification, grazing impacts and interspecific interaction with meliphagid miners do influence bird assemblages (Maron *et al.* 2011) and it is likely that these influences will increase in this bioregion with proposed mining activities.

The positive and negative response categorisations of bird species need to be used in a tentative manner as either variation may be likewise a function of fire scars, tree dieback and /or grazing intensity/practices in both the past and the present. In spite of the limitations of the use of indicator bird species, they can be a useful tool to evaluate the condition of an ecosystem. As Carignan and Villard (2002) suggest, cautionary use of indicators should include a) incorporating species representing the range of taxa and life histories, and b) the selection of indicators based on sound quantitative data from the area of interest. They also caution that care should be taken to separate actual meaning from fluctuations that may be unrelated to ecosystem deterioration when interpreting indicator trends.

It would appear that the insectivorous birds that feed around the leaves in the upper canopy are the most affected by a loss of ground cover. The impact is unlikely to be a direct response to the decrease in ground cover, but to the reduction of habitat quality that is likely to be associated with this reduction in ground cover; that is a loss of live trees, foliage cover and general ecological integrity. The carnivorous, granivorous and generalist species are increasers where low ground cover gives them an advantage to their prey selection. In this study, several species reflect the same response to grazing as other studies have identified. In Martin and Possingham (2004) ten species were identified as being more prevalent under low grazing, namely the variegated fairy-wren, weebill, noisy friarbird, little friarbird, yellow-faced honeyeater, white-throated honeyeater, varied sitella, grey fantail and mistletoe bird. In this study, only the variegated fairy-wren and grey fantail had significant positive relationships with ground cover, while birds like the weebill, varied sitella, noisy friarbird and mistletoebird also had positive, but not significant relationships.

A comparison to the Barnards' (1925) notes on birds species at Coomooboolaroo (situated about three hundred kilometres to the east of the Desert Uplands woodlands) show three species remained common in these woodlands: the weebill, the brown treecreeper and the rufous whistler. Interestingly, the crested bellbird had disappeared during the 50 years that the Barnard brothers observed the birds, which they reasoned was for due to its being easy prey for cats. By contrast, in the Desert Uplands' ironbark woodlands they are present. Perhaps crested bellbird numbers should be closely monitored as this suggests they are vulnerable to predation. However, their decline at Coomooboolaroo could have been as a result of grazing impacts and a consequent increase in density of the shrub layer and, therefore, the creation of a homogenous habitat unattractive to the Crested Bellbird at the edge of their distribution (G. Porter *pers. comm*).

Grazing is assumed to threaten ground-foraging birds, however this was not found to be the case in this study. In spite of this, the results indicate that loss of ground cover which could be caused by grazing degradation that contributes to the decline of insectivores by causing a reduction in food availability. Impacts result in habitat degradation such as a decrease in the structure and composition of the vegetation. Habitat structure strongly influences foraging behaviour of birds in woodlands in the mid and upper storeys (Robinson and Holmens 1982), whereas livestock influences the abundance and composition of terrestrial invertebrate fauna (Bromham 1999; Seymour and Dean 1999). If birds have to resort to using more energetically expensive prey-attack manoeuvres or selectively use substrates and microhabitats less available in degraded habitats, such degradation may impact on species compositions by reduction in food availability. This is demonstrated by the habitat group that nest on the ground and feed in trees response in the two way table in Figure 4.8, as this habitat assemblage was not present at sites with low ground cover.

A study of the use of bird species as indicators for biological integrity in the Great Basin Rangelands of the United States concluded that bird abundance and richness metrics had greatest sensitivity at the high grazing impact end, but had limited utility in distinguishing between light and moderate impacts (Bradford *et al.* 1998). This finding is confirmed by the current work, although this study had fewer sites with low ground cover and the results from these low ground cover areas were often confounded by the fact that they were located near an artificial watering point.

Work by Ward and Kutt (2008) showed that ground cover predictor variables explained 83% of bird diversity variation in basalt homogenous eucalypt woodlands. This work provides further evidence that the relationship of bird species with ground cover is related to several environmental variables such as over-storey cover, mid-storey cover and perennial grass cover.

The use of habitat assemblage groupings is useful for examining ecological condition, in particular the group that nest on the ground and feed in trees as it is has an obvious relationship to ground cover. Davies *et al.* (2010) found that the use of feeding and nesting traits helped unravel the impact of artificial waters in the arid zone of Australia and concluded that those bird species that nested on the ground were most susceptible to the impacts of cattle grazing.

It would appear that the species composition of the woodlands avifauna in the Desert Uplands has been resilient to past habitat impacts, with light to moderate grazing impacts having only slight effects, reflecting a relatively stable species composition of avifauna. However, this may be due to non-linear relationships between vegetation characteristics and grazing practices, for example a low impact on canopy structure. The work of Martin and Possingham (2004) indicates that livestock grazing of woodlands alters the vegetation structure by modifying and often removing the understorey vegetation, thereby changing the resources available for birds.

The response to increasing bare ground can be confounded by some birds responding to an increase in the availability of water (James 2003). For example, granivorous birds that need to drink (Galahs, Crested pigeons) are advantaged by water availability which may be associated with low ground cover. In this study, the evidence of Rufous whistlers, Grey shrike-thrushes, crested bellbirds and Red-browed pardalotes being more abundant in sites with more ground cover is probably more conclusive of the effects of grazing than similar trends in raptors and wood swallows (a nomadic species) would be.

Mazaris *et al.* (2010) suggest that 30 to 60 of the most common species (widely distributed over the species distribution) present in an area can be used to predict overall species richness. Based on this recommendation, the four species that have significant positive relationships with ground cover could be used as indicators to the condition of silver-leaved ironbark woodland avian diversity in the Desert Uplands, together with other species with positive but not significant relationships to ground cover. As a suite of resident species with a range of different nesting and feeding requirements, these birds could be indicators of changes in ground cover and environmental variables over time (Price *et al.* 2009). Due to the contextual landscape, in which these silver-leaved ironbark woodlands are found with only 4% under 25% ground cover, the effects of low ground cover areas around artificial watering points are likely to have little impact on bird diversity due to their mobility, but over time low cover areas could increase in area, allow the introduction of exotic species and increased disturbance.

While there are five species, mainly resident insectivorous birds, from the habitat assemblage group of birds that nest on the ground and feed in the mid-storey that are significantly sensitive to decreases in ground cover, these results need further substantiation before birds or bird groups could be recommended as indicators of biodiversity condition in this woodland. However, studies in south-eastern Australia, demonstrated that species that are resident, small-bodied, ground foragers and insectivorous are most likely to decline with the loss of habitat complexity (Reid 1999). Of the five indicator species that decreased in abundance with reduced ground cover in this study, four have been found to also be reduced in the southern part of their distribution. Rufous whistler, crested bellbird, grey shrike thrush and grey fantail are declining woodland birds in south-eastern Australia (Watson 2011; Razeng and Watson 2012; Stevens and Watson 2013). Re-survey of sites in Central Queensland found declines in rufous whistler, grey shrike thrush and grey fantail over 25 years (Woinarski *et al.* 2006). In a survey of different configurations of remnants over two years, rufous whistlers and grey fantail abundances were found significantly more in intact woodland reference sites than pasture sites as were yellow-throated miners (Hannah *et al.* 2007). The presence of aggressive yellow-throated miners could be compounded by habitat fragmentation and simplification with negative effects on smaller insectivorous birds (Maron *et al.* 2011). The red-browed pardalote has been found to have suffered no changes in abundance across its southern distribution and therefore may not be such a reliable indicator (Olsen *et al*. 2003).

In the areas of silver-leaved ironbark with low ground cover, the low abundance of the resident insectivorous birds positively supports Watson's (2011) productivity-based hypothesis for declining woodland birds as evident even in these intact woodlands. Watson's hypothesis indicated that degradation of woodlands results in the reduction of the biomass of decomposer communities in the soil and litter, which in turn reduces the food resources available for ground-feeding insectivores. Increased grazing pressure causes changes in the understorey; declines in palatable species with deep-rooted perennials becoming successively replaced with annuals (McIntyre and Lavoral 1994; Landsberg and Crowley 2004). These ground cover changes decrease the amount of organic matter held in the soil, simplifying the soil structure and decreasing the soil water-holding capacity (Tongway et al. 2003).

Overgrazing and associated trampling by stock also reduces ground cover, disturbs cryptogamic and micro-biotic crusts leading to soil compaction and a decrease in water infiltration as well as topsoil loss from erosion (Ludwig *et al.* 1997). Removal of coarse woody debris results in reduced heterogeneity. Habitat heterogeneity ensures infiltration, litter accumulation and microclimate amelioration (McIvor 2002). Changes from deep-rooted perennial grasses to a domination of shallow-rooted annuals can lead to a shift from low level nitrate availability year round to peaks in otherwise low availability, associated with growth cycles of exotic annuals (McIntyre and Lavorel 1994; Clarke *et al.* 2003). These change soil properties cause decreases in organic matter, soil water content and increases in the availability and homogeneity of inorganic nutrients. These soil changes lead to changes in the composition and function of decomposer communities. Bacteria and fungi are determined by the soil properties and nutrient inputs (Bardgett 2005). Microbial communities determine the rate of decomposition but also affect the structure and composition of other soil biota that feed on them, thereby driving changes throughout the entire food web (Cole and Bardgett 2002). Arthropods and other invertebrate prey become available over an increasingly shorter period and with a reduction in structural complexity of soil render biota more vulnerable to short term climatic variation (Taylor 2008). The composition and accessibility especially of larger taxa, high in nutritional quality found in litter and the topsoil (beetles, their larvae, spiders and moth larvae), are considered the most important drivers of insectivore occurrence (Watson 2011). Invertebrates have a negative relationship with grazing pressure (King and Hutchinson 1983) and a positive relationship with soil moisture and litter depth (Taylor 2008).

The abundance of resident insectivorous birds therefore has a strong relationship to habitat simplification through the loss of ground cover and the associated loss of soil structure and function, and invertebrate composition which is a critical food source for these bird species. These results are very promising, however, further bird monitoring in drought conditions when low ground cover is most evident and widespread would further substantiate these results.

Chapter Five

Can ground cover be predicted using a linear mixed effect model?[4]

5.1 Introduction

Savannas cover one eighth of the globe (Scholes and Archer 1997) and contribute approximately 30% of the terrestrial ecosystem gross primary production (House and Hall 2001). Savannas are extensively utilised for livestock grazing, but heavy grazing can alter species composition and diminish production (Milchunas *et al*. 1988; Scheiter *et al*. 2012), as well as negatively impacting on biodiversity (Frost *et al*. 1986; Hanan and Lehmann 2001; Scholes and Archer 1997). Yet the relationship between pressures and impacts are difficult to predict. A key research need is to couple landscape attributes using remote sensing techniques and ecological modelling to better understand savanna ecological function (Hill *et al.* 2011).

Changes in land cover can significantly affect key aspects of ecological functioning, by impacting on biodiversity, water quality or increasing soil erosion (Houghton *et al.* 1999). Over past decades, studies have moved from detecting and identifying land cover changes (Lambin *et al.* 2001) to understanding the driving forces of landscape change (Antrop 2005) and now towards modelling current land systems in order to predict future cover changes (Veldkemp and Lambin 2001). The remaining challenges are to better isolate and understand the impacts of landholder management actions so that more proactive measures can be taken to protect and improve savanna condition.

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[4] This chapter is structured as a separate journal article.

The separation of the effects of climate from human activity on savanna condition is a major challenge (Evans and Geerkun 2004; Wellens 1997; Wessels *et al.* 2007). This is particularly the case in the savannas of northern Australia where the climate has very stochastic impacts on ecosystem functioning that result in a wide variability of effects.

In northern Australia, the dynamics of semi-arid eucalypt savanna systems are susceptible to the effects of highly variable rainfall. The seasonality of rainfall and droughts accounts for first order components in spatial and temporal ground cover variation. Modelling can help determine whether an anthropogenic signal from grazing pressures can be measured using the Ground Cover Index (GCI) and establish the relative role of factors such as differences in stock management that can affect in the amount of ground cover at a site level (Archer 2004). If ground cover can be predicted based on its relationship to climatic and grazing management variables then ground cover forecasting and sustainable management can be more confidently implemented. Readily available archived remote sensed ground cover data and historic weather data allows for these dynamic relationships to be explored more thoroughly.

This study uses an innovative approach involving linear mixed effect modelling to test whether climate, landscape and grazing management variables are significant explanations of gradual change in ground cover in semi-arid eucalypt savannas. This applied model aims to describe and predict how the ground cover functions in relation to the chosen grazing variables.

In this study, modelling of both field and remote-sensed data was undertaken to ascertain the drivers of ground cover change in the silver-leaved ironbark woodlands of Desert Uplands region of central western Queensland, Australia. This savanna ecosystem covers over 1,000,000ha of the bioregion and as a significant production and biodiversity resource, is worthy of the focus to ensure that both resources are maintained. The linear mixed effect modelling used in this study was derived from the ground cover directly measured for 91 sites in October 2007, as well as remotely-sensed GCI data for these 91 sites over 21 years and remotely-sensed GCI data for 300 randomly generated sites over 21 years. This allowed several modelled parameters that are expected to drive ground cover to be tested for relevance and therefore ecosystem condition to be assessed (see Figure 1).

The research allows three important questions to be addressed. The first is to identify the suitability of remote sensed data to assess ground cover in semi-arid savannas (this exercise is about confirming the robustness of the GCI in an actual case study), the second is to separate out the influence of climate and grazing management variables on ground cover, whilst the third is to identify how these relationships function over time. The resulting models are then validated against actual ground cover measurements to determine their accuracy. The paper is structured in the following way. The research issue is contextualised and research hypotheses are developed in the next section, methods are explained in section three and results in section four. Discussion and conclusions follow in the last two sections.

5.2 Contextualising the research questions

Understanding impacts on savanna condition has been a key focus of previous research. Several studies have proposed techniques using Rain Use Efficiency to identify anthropogenic land degradation (Diout and Lambin 2001; Wessels *et al.* 2007) or removing climatic signals from NDVI time series by means of the residual trend methods (Archer 2004; Evans and Geerkun 2004: Hermann *et al*. 2005; Wessels *et al.* 2007). Most of these satellite time-series analysis uses coarse spatial resolution imagery from NOAA/AVHRR satellites which limits detailed heterogeneity at the fine scale necessary for land management application in terms of paddocks and properties.

Due to the variation in grazing variable influences across a paddock or property, Illius and O'Connor (1999) recommend that research should identify the grazing characteristics that predispose some areas of ecosystems towards degradation, where other areas appear to be resistant. Focus on the spatial heterogeneity in susceptibility to grazing impacts and the preservation of core areas to maintain ecological integrity needs to happen (Illius and O'Connor 1999).

Modelling can link levels of ground cover with explanatory variables, and enables the separation of grazing variables effects on ground cover from the major climatic variable; rainfall in this ecosystem of focus. In the rangelands, skewed annual rainfalls result in long periods of aridity, interrupted by occasional heavy rains with bias towards longer periods of dry years and greater lag correlation between years that leads to longer wet and dry periods than elsewhere in world (McMahon *et al.* 2008).

As information about grazing numbers and grazing management are rarely available in a precise nature, researchers typically use other variables that are indicative of grazing intensity. Distance to water is well documented as a major variable that influences spatial expression of grazing impacts. The physiological dependence of livestock on water results in activity being concentrated in the vicinity of a watering point, and dissipating rapidly with increasing distance from it (Valentine 1947; Lange 1969; Andrew and Lange 1986). Therefore, water placement plays an important role in how livestock behave and spatially impact on an ecosystem. Ground cover and grass cover, as well as several decreaser grass species of ground cover plant species have previously been found to be significantly affected by distance to water (Landsberg *et al*. 2003). Therefore the distance to water is expected to be a reliable measure of grazing impact.

The size and shape of paddock has strong influence on grazing behaviour especially in relation to the location of the watering point. Larger paddocks allow for greater dispersion of impact to other parts of the paddock. Landscape heterogeneity in a paddock can also influence grazing behaviour. Preferential grazing is common in rangelands with contrasting landscapes, and their associated pasture types. Grazing activity is often focused on the more productive, often lower-lying land types (Coughenour 1991; Cridland and Stafford-Smith 1993; Landsberg and Stol 1996). Intensification of infrastructure – both paddock size and number of waters has occurred over the years in order to make use of previously under-utilised country (Fisher *et al.* 2004).

Westoby *et al.* (1989) identified several ecological processes which result in non-linear vegetation dynamics such as episodic periods of drought or favourable rainfall, altered grazing or fire regimes or severe soil erosion. These events can result in thresholds being crossed both spatially and temporally. Functional thresholds are thought to lag behind the structural or compositional thresholds (Briske *et al*. 2005). When a state transition has occurred over a sufficient spatial area, a distinct set of cross-scale inter-actions can be initiated at the landscape scale that link even distant sites together. Changes in land surface condition can have a pronounced effect on weather, climate and local meteorology (Bryant *et al.* 1990; Pielke *et al.* 1998). The landscape scale cover of highly vegetated versus poorly vegetated states can influence meso-scale climate via the influence of vegetation on dust aerosols and soil surface temperature that intensify local drought and vegetation loss (Cook, Miller and Seager 2009). There is little work to indicate the areal extent, continuity and nature of state change needed to initiate feedbacks at the landscape or larger scales (Bestelmeyer *et al.* 2011). The ability to predict and manage transitions in many ecosystems would be improved by knowledge of spatial processes. This would help identify, where, when and under what circumstances undesirable transitions or opportunities to promote desirable transitions are likely to occur.

From a study in northern Queensland, Fisher and Kutt (2007) found in their comparison of grazing variables between land condition classes that distance to water was significantly different between condition classes for Queensland basalt sites. The differences in condition were most pronounced across fence-lines (between paddocks and/or properties), presumably arising from differences in stocking rates and/or other grazing management systems at a paddock/property scale over moderate timeframes.

In this study paddocks and properties were considered to be random effects. Using GIS satellite imagery, it was possible to measure the area of the paddock in which the site was located as well as the distance the site was from water and the amount of more palatable and less palatable ecosystems in comparison with the silver-leaved ironbark woodland within the paddock. As the paddock area changed for several sites over 21 years so too did the area of less and more palatable land types and the distance to water. Modelling of GCI data in this study was carried out to better understand drivers of ground cover change. From previous study results, ground cover has been established as a suitable surrogate for ecosystem condition, especially in terms of plant ground cover diversity and that the GCI is a reliable measure of ground cover.

In this study, the hypothesis tests whether the following climatic parameters; - the 24 month Foley's index (Foley 24) (Fensham 1999) and grazing variables:-paddock size (PS), distance to water (DTW), the area of more palatable land type (Sweet) in the paddock and the area of less palatable land types (Hard) in the paddock, have a measurable effect on the amount of ground cover over a two decade period. All four paddock measurements are surrogate measures for grazing pressure and have been demonstrated to correlate with grazing in the field assessments carried out on the 91 sites.

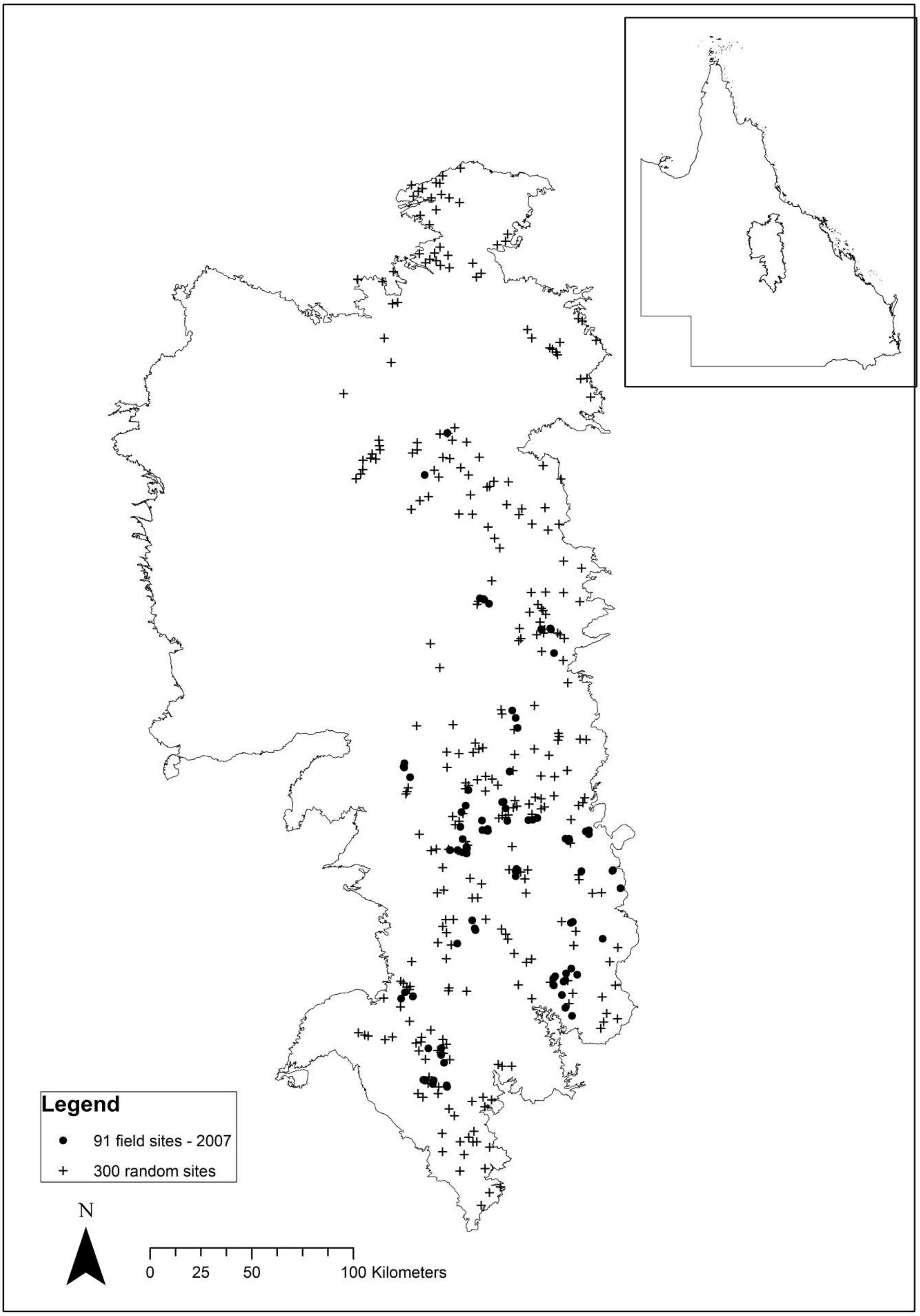


Figure 5.1 Location of the 2007 91 field sites (•) and 300 random digital sites (+) within the Desert Uplands bioregion with the TM Landsat imagery

5.3 Method

The Ground Cover Index (GCI) is calculated from a multiple regression model of the reflectance of bare ground in landsat bands 3, 5 and 7 and is calibrated to field-based measurements. It has been developed for semi-arid cover estimation in Queensland and to be able to estimate cover when vegetation is sparse rather than continuous (Scarth *et al.* 2006; Karfs *et al.* 2009). The GCI is suitable over a range of soil types, variety of woody cover amounts and for both green and dry vegetation. The GCI is the index used as the basis of the modelling for this study[5] .

In this study, ground cover datasets were collected in the field in both May and October 2007 for 91 field sites across 25 properties. This actual ground cover data was collected at 100m by 100m sites as per the methodology outlined by Hassett et al. (2000).

Additionally, remotely sensed GCI index data was collected for these 91 field sites and an additional series of 300 randomly generated sites across the silver-leaved ironbark ecosystem for the 21 years between 1988 and 2008. All sites are located in the ecosystem where the tree projective foliage cover is less than 20%, as that is where the GCI is most accurate (Scarth *et al.* 2006).

The remote sensed GCI for each site (n=300 x t =21years) and (n=91 x t=21 years) was recorded for the central 25m X 25m pixel of the GCI regenerated by the

[5] The Queensland Government now produces a seasonal fractional cover data which is a per pixel estimate of green cover, non-green cover and bare ground. The total green cover and non-green cover of the fractional cover data is equivalent to the GCI (D.Tindall pers.com.)

Queensland Department of Environment and Resource Management (Scarth *et al*. 2006), derived from Landsat TM satellite imagery for the dry season from July to October each year from 1988 to 2008. Observations on the presence of a fire-scar were recorded as a dummy variable equal to one when the fire-scar was present. The area of the paddock (PS), distance to water (DTW), the area of more palatable ecosystems in the paddock than silver-leaved ironbark where the site is located (Sweet) and the area of less palatable ecosystems in the paddock than silver-leaved ironbark (Hard) were measured from Landsat TM imagery for each site for each year. Sites with fire scar data and the data from the following two years were deleted from the dataset as these abrupt cover changes were few in number (only 152 records over 21 years) and would confound the explanation derived from the other variables for those sites.

Fensham *et al*. (2005) affirmed the importance of multiyear variations in rainfall for both overstorey and understorey cover dynamics. Foley's Rainfall Deficit Index (Dm,y) (Foley 1957; Maher 1973; Fensham 1999) is a rainfall deficit standardized by the mean annual rainfall (A) of a site. The Foley rainfall deficit index (Fensham 1999) was calculated for each site using monthly rainfall data and three index periods: 12 months, 24 months and 36 months. The Foley index is simply calculated as the sum of the deficit of observed rainfall from the expected rainfall over monthly intervals for a specified period of time. The deficit index is calculated for each *m* month of each year *y* as actual annual rainfall for 1, 2 or 3 years before every month less expected (long-term average) rainfall for that period, divided by the mean annual rainfall:

$$D_{m,y} = \sum_{i=y-X+1}^{y} \frac{a_{m,y} - Ax}{A}$$

where $a_{\_m,y}$ is the summed monthly rainfall over the previous 12, 24 or 36 months of year *y,* that is, from month *m to m-11,* and *A* is the long-term annual average rainfall; *x* is the chosen deficit period which is in this case is 12, 24 or 36 months.

The minimum Dm,y for each year one degree grid cell combination was then determined for all 91 sites and 300 random sites with records from at least 1898 used to estimate the long-term average expected rainfall. This simplified a large data set consisting of values for every month within a given year and every site within a given grid cell. Thus, for each site three Foley series were developed for each of the 21 years of the dataset representing three alternative lengths of 12, 24 and 36 months for the Foley rainfall deficit indices.

Measurements of GIS attributes were also collected for each of the 91 field sites in order to compare the remotely sensed ground cover index measurements with field measurements. The field ground cover measurements are important for two reasons; a) to ensure that the Ground Cover Index is accurately measuring ground cover and b) to test the predictability of the models with actual field collected ground cover data. Therefore, for each of the 300 randomly generated sites and each of the 91 field sites (these two data sets are independent) the following variables were recorded: the ground cover index, the distance to water (DTW), presence of fire scar, area of the paddock (PS), the area of more palatable land types within the paddock (Sweet), the area of less palatable land type within the paddock (Hard) and the Foley index for three periods – 12 months (Foley_12), 24 months (Foley_24) and 36 (Foley_36) months for each year from 1988 to 2008. The measurements were taken to assess whether these variables explained deviations of ground cover from the expected response to rainfall.

The initial steps in the data analysis were to test for relationships between ground cover and other variables in turn. Graphing and bivariate tests will be used to critically explore relationships between pairs of key variables. Once these exploratory analyses are completed, multivariate analysis will be employed to predict ground cover as a function of a number of variables.

As the data is spatially hierarchical (sites are likely to be more related to sites within the same property than to other sites within other properties because of management factors) and has repeated measures, fixed effects are combined with nested levels of random effects in a linear mixed effect model for the multivariate analysis. This allows the analysis of repeated measures and spatially nested data without succumbing to the problems of non-independence and pseudo-replication. The nested hierarchy of spatial effects and repeated measures inherent in a 21 year dataset necessitated the use of this modelling technique. The model structure allows the investigation of the relative importance of variation between years and sites. Knowledge of where the stochasticity in the system occurs allows for better treatment of the fixed effects and ultimately provides more informed predictions with respect to the fixed effects.

To analyse differences in the cover of the 300 random sites, Linear Mixed Effects Models (hereafter LMEM) using R 2.11.1 (R Foundation for Statistical Computing; R package nlme) were applied. The relative cover of the 300 sites was modelled as a function of the Foley index (12, 24, 36 months) with all factorial combinations of paddock size, distance to water and area of more palatable land type and area of less palatable land type within the nested random effects structure of site/property. The dependent variable was arc sine transformed to ensure it has a more normally smooth distribution. Testing identified auto-correlation of errors (repeated measures through time) so an autoregressive model of order one (AR (1) correlation) was used. Diagnostic plots to check model assumptions were applied (Pinheiro and Bates 2000); there was no evidence of correlation of observations within sites and it was assumed that within group errors were normally distributed.

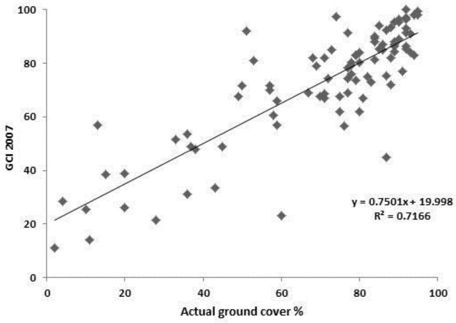
The best-fit models were found using the step Akaike's Information Criterion (AIC) (Venables and Ripley 2002). The step AIC function removes fixed effects one at a time from the full model and compares the AIC value to the full model. If the AIC value of the new model is smaller than the AIC from the full model then the variable is removed. Best-fit models can be found using forward or backward techniques, so in this application both methods were tested. To test the significance of different fixed effects, the remaining explanatory variables were removed one at a time from the best-fit model using likelihood ratio statistics to test for model improvements. Maximum likelihood was used when comparing nested models to simplify the model for fixed effects (Pinheiro and Bates 2000; Ives and Zhu 2006).

5.4 Results

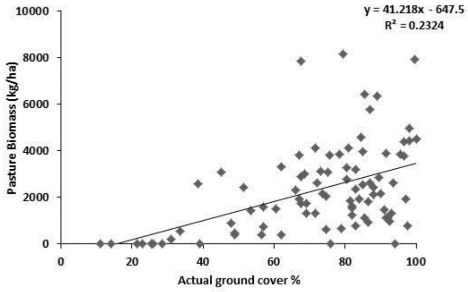
These applied models aim to describe and predict the relationship between the ground cover and the chosen grazing variables. In this study, the effects of rainfall and other grazing variables are explored using bivariate analysis and via a linear mixed effect model of multivariate analysis. Bivariate analysis enables an understanding of the key variable relationships with ground cover while multivariate analysis via the models measures the combined effects of each variable. The results of three models are reported and then these are tested for their predictive power with respect to field ground cover measurements collected in October 2007 from 50 sites.

5.4.1 Test one

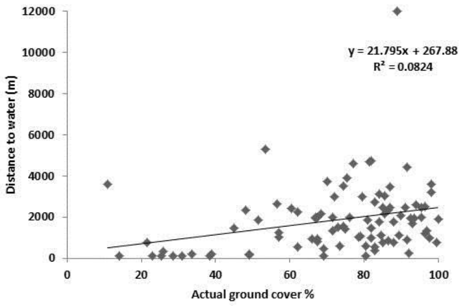
The focus of the initial tests was to assess the relationship of the ground cover measurements in the field sites against both the remote sensing data and other variables. Data from the 91 sites with field measurements was used for these tests. The correlation between the Actual ground cover in October 2007, the Ground cover index and the predicted ground cover from model 1 were examined and T tests assessed the variance between the GCI measurements and model predicted ground cover. The relationship between ground cover measured in the field and the GCI proves to be well correlated across the 91 sites. When ground cover was examined with respect to pasture biomass and distance to water there are positive relationships, which are expected. As ground cover increases so too does pasture biomass. The further from water, the higher the ground cover (see Figure 5.2).



(a)



(b)



(c)

Figure 5.2 Relationship between actual ground cover measured in the field in October 2007 and (*a*) the Ground Cover Index 2007, and (*b*) the pasture biomass and (c) distance to water

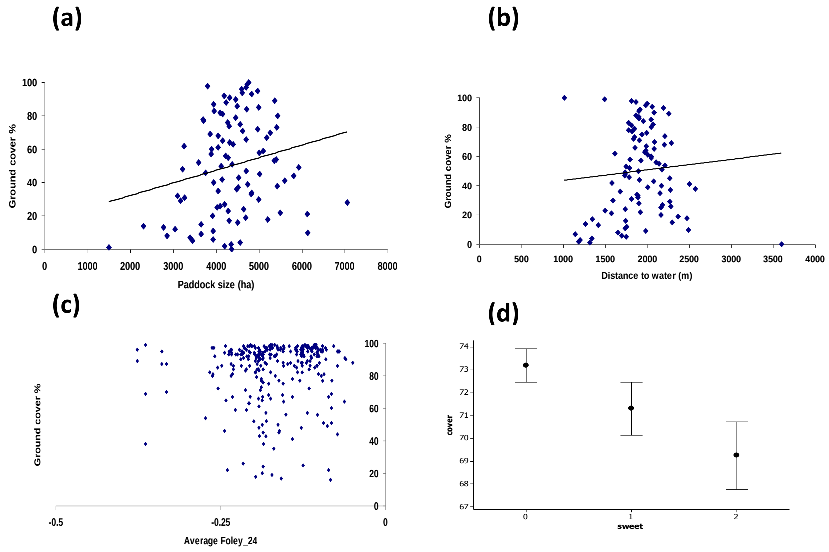


Figure 5.3 Mean ground cover versus (a) paddock size (ha), (b) distance to water (m), (c) foley index for 24 months and (d) grouped areas of more palatable land type within paddock (where 0=none, 1 = less than 50% and 2= more than 50%)

5.4.2 Test two

The second test was to examine the relationship of ground cover to other grazing variables. This is examined repeatedly with the use of field data across the 91 sites and a larger data set of 300 randomly generated sites across the entire silver-leaved ironbark distribution in the Desert Uplands, and spanning over 21 years of archived remotely-sensed data.

Looking at the larger data set of the 300 random sites (see Table 5.1), ground cover as determined by the GCI has a positive relationships with increasing paddock size, distance to water and the rainfall index (see Figure 5.3). On the other hand, ground cover has a negative relationship with the area of more palatable land types (Sweet).

Table 5.1 Summary statistics for the five variables from the 300 random site dataset used in the models

Variable	Mean	Min	Max	St.Deviation
Foley 24 (Foley_24)	-0.169 -	1.341	1.255	0.492
Distance to water (DTW)	1.948m	58m	9,825m	1,448.86
Area of more palatable (SWEET)	1007ha	0	10,918ha	1348.022
Area of less palatable (HARD)	269ha	0	1037ha	317.876
Paddock Size (PS)	5369ha	51ha	56,045ha	7761.032

Turning to the longitudinal data, the pattern of ground cover in Figure 5.4 shows the annual changes in average ground cover over the 21 years. There is significant inter-annual variation in cover, but the variation within the years is very small, suggesting that weather is a major driver of ground cover.

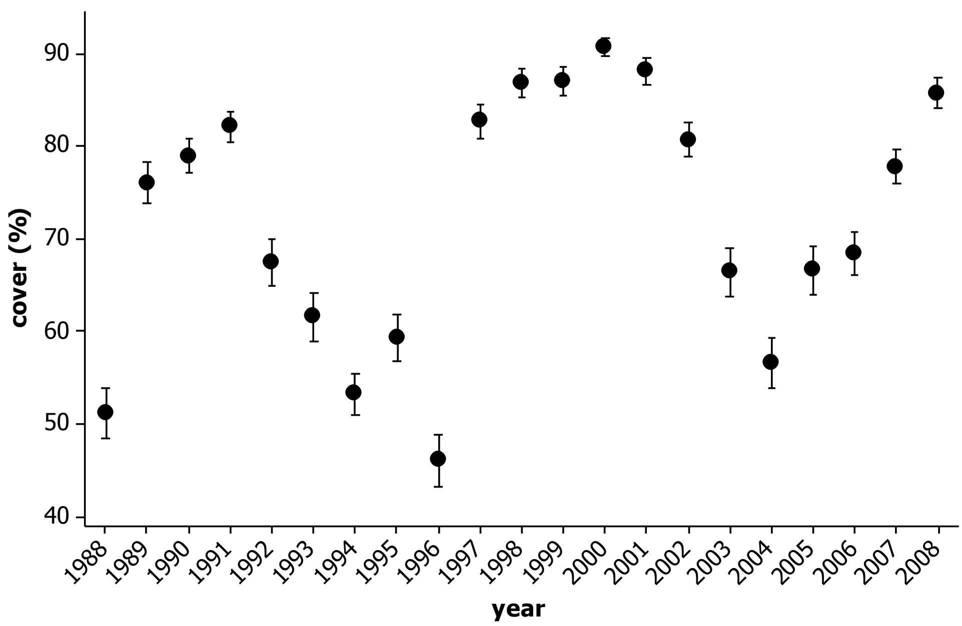


Figure 5.4 The mean ground cover per year for the 300 sites with standard deviation bars

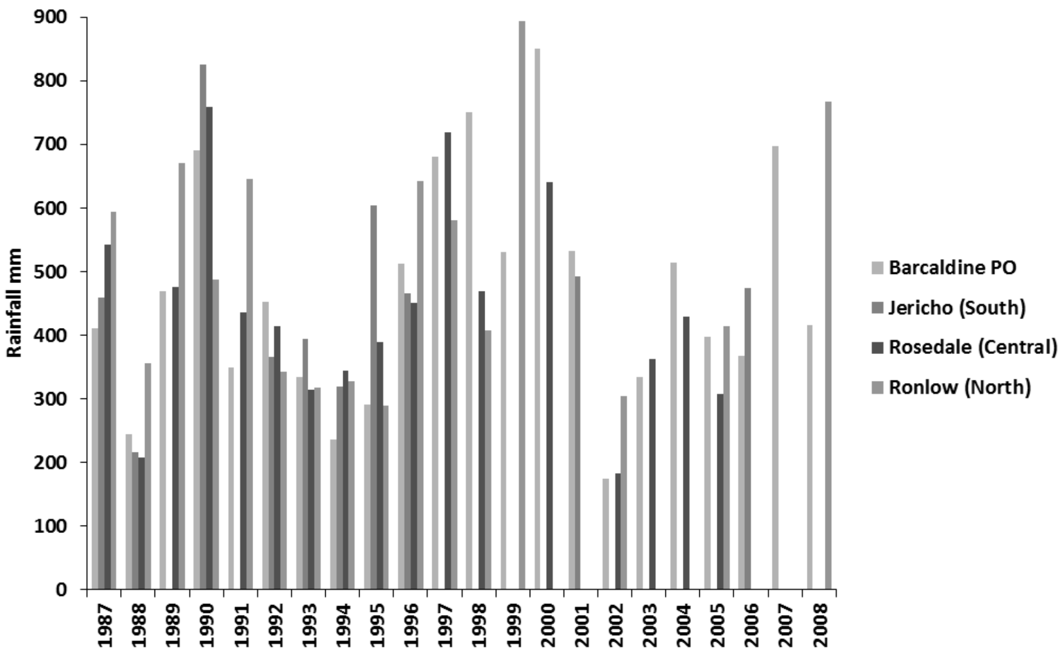


Figure 5.5 The annual rainfall from 1987 to 2008 from four rainfall collection stations across the distribution of silver-leaved ironbark in the Desert Uplands – Ronlow in the north, Rosedale centrally, Jericho in the south and Barcaldine south-west of the study area

When the inter-annual variation in ground cover is then examined in relation to the rainfall deficit index using Foley 24, the patterns become apparent. In the wet years, such as 2008, 2001, 2000, 1999, 1998, 1997 and 1991, the Foley 24 deficit index is above 0 and the ground cover is concentrated at or above 80%. In the dry years, such as 2002, 1994, 1996 and 1988, the Foley deficit index is generally below 0 and the variation in ground cover is very high, with a number of observations at low levels (see Figure 5.6). The actual annual rainfall is presented in Figure 5.5 to show the wet and dry year's actual rainfall at three locations across the Desert Uplands.

Exploratory examination of the relationships between ground cover and each of the 12, 24, and 36 month Foley Indices found that the 24 month index had the most significant relationship with ground cover. For brevity, therefore, all models reported from the analysis were only estimated with this index.

The modelling approach is substantiated by significant bi-variate relationships that exist between the index and actual ground cover measurements and the correlation of the actual ground cover with biomass and distance to water, consistent with the field data results as summarised in Figure 5.2.

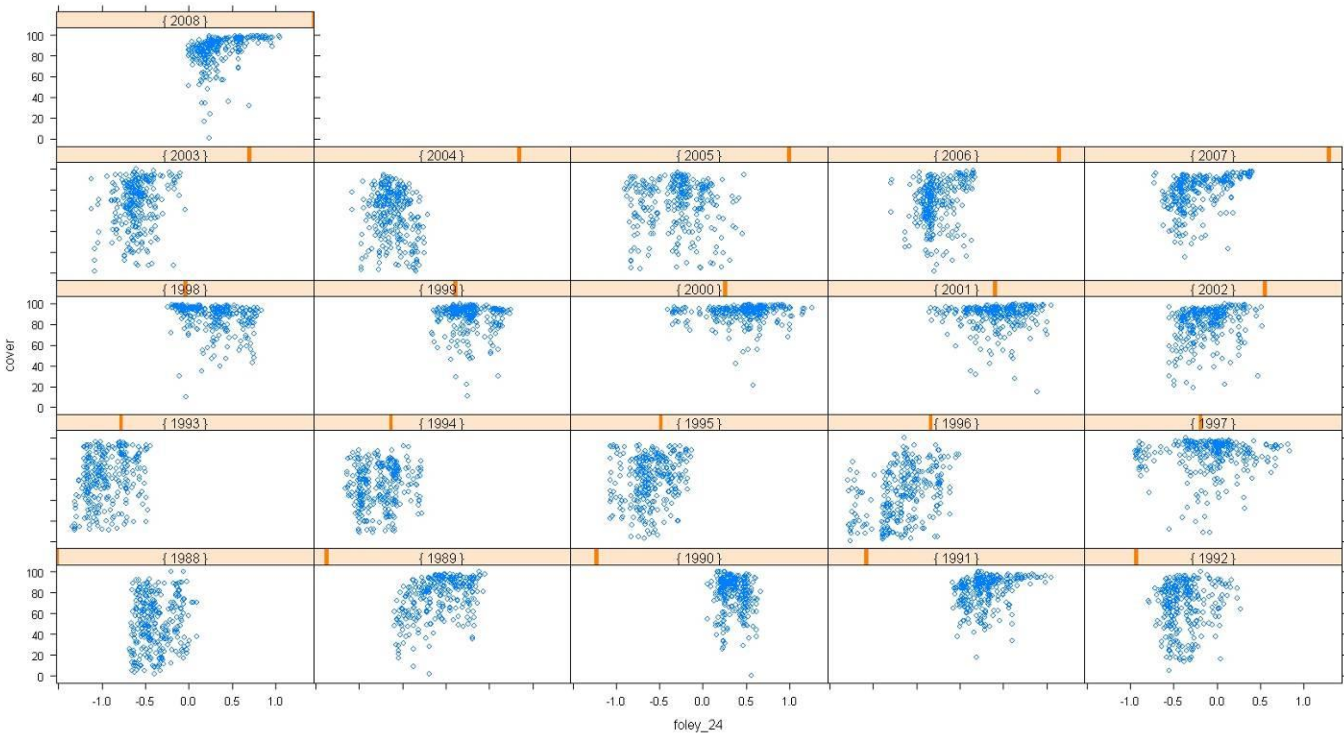


Figure 5.6 21 panels showing ground cover by Foley 24 index by year for the 300 randomly located sites

5.4.3 Test three

This third test involves examining the effects of grazing variables on ground cover over 21 years, therefore incorporating findings of both of the previous tests. A linear mixed effect model was used to examine the effects of the five explanatory variables on ground cover and to take into account the location of sites within properties with annual measurements of the explanatory variables over the 21 years.

The ANOVA (F and p values) results of three linear mixed effect models are reported to order to understand specific relationships and to test if dry and wet periods have different effects on ground cover.

5.4.4 Multi-variate analysis

The following models were used to examine the effects of the five explanatory variables on ground cover and to take into account the location of sites within properties with annual measurements of the explanatory variables over the 21 years:

Model 1 is for the GCI generated cover over 21 years for the 300 randomly generated sites in order to better represent the entire spatial distribution of the ecosystem (Table 5.2).

Model 2 (wet) is for the GCI generated cover over 21 years for the 300 randomly generated sites for the years preceded by two years of above average rainfall – 1991,1997, 1998, 1999, 2000, 2001 and 2008 (Table 5.3).

Model 3 (dry) is for the GCI generated cover over 21 years for the 300 randomly generated sites for the years preceded by less than average rainfall – 1988, 1994, 1996 and 2004 (Table 5.4).

In all three Models; the 24 month foley index has a significant positive effect on ground cover, which is what is expected. Interestingly, in the model for years preceded by less than average rainfall; - the rainfall index has the largest coefficient, therefore has the greatest effect on ground cover as a result of those preceding drier years.

In all three models the amount of more palatable land types had a consistently strong significant negative effect on ground cover. The rainfall index has significant positive effect. The distance to water variable and the amount of more palatable land type in the paddock both have significant effects on ground cover.

Table 5. 2 Anova results from Model 1

ANOVA Results **Variable**	**F value**	**P value**
Foley_24	**2485.249**	**<.0001**
SWEET	**7.891**	**0.005**
HARD	0.387	0.534
DTW	**8.825**	**0.003**
Foley_24:PS	0.052	0.819
SWEET:PS	2.241	0.135
Foley_24:HARD	1.405	0.236
Foley_24:SWEET:PS	1.484	0.223
Foley_24:SWEET:HARD	**8.87**	**0.003**
AIC = -3650.346	BIC = -3555.968	Loglik= 1834.173

Model 2 examines the sites in years that are preceded by two years of above average rainfall to ascertain whether the explanatory variables are acting in a manner specific to these above average rainfall years. The model is derived from a subset of the 300 model dataset in which the records were preceded by two years of above average rainfall.

Table 5.3 Anova results from Model 2

ANOVA results Variable	F value	P value
Foley 24	**138.213**	**<.0001**
SWEET	2.988	0.084
PS	0.001	0.982
HARD	0.022	0.883
Foley 24:DTW	**24.266**	**<.0001**
SWEET:DTW	1.132	0.288
PS:DTW	**6.037**	**0.014**
PS:HARD	**7.072**	**0.008**
HARD:DTW	2.576	0.109
Foley_24:SWEET:PS	0.015	0.109
Foley_24:SWEET:DTW	**11.018**	**0.001**
Foley_24:PS:DTW	2.302	0.129
Foley_24:PS:HARD	**7.257**	**0.007**
AIC= -2148.963	BIC=- 2047.406	LogLik=1092.482

Model 3 examines whether explanatory variables are acting in a manner specific when preceded by two years of below average rainfall. This model is examines sites preceded by two years of below average rainfall to ascertain whether the explanatory variables are acting in a manner specific to these below average rainfall years.

Table 5.4 Anova results from Model 3

ANOVA results Variable	F value	P value
Foley_24	**40.423**	**<.0001**
SWEET	2.786	0.096
DTW	0.522	0.470
PS	0.153	0.696
HARD	0.152	0.697
Foley_24:DTW	0.052	0.819
SWEET:DTW	**0.651**	**0.010**

Foley_24:PS	0.320	0.572
SWEET:PS	0.227	0.634
DTW:PS	0.043	0.836
SWEET:HARD	0.054	0.152
DTW:HARD	0.057	0.811
PS:HARD	0.554	0.457
Foley_24:SWEET:PS	1.090	0.298
Foley_24:DTW:PS	**4.658**	**0.031**
SWEET:DTW:PS	0.009	0.923
Foley_24:SWEET:HARD	0.156	0.693
Foley_24:DTW:HARD	0.002	0.968
SWEET:DTW:HARD	0.089	0.765
SWEET:PS:HARD	0.501	0.479
DTW:PS:HARD	1.745	0.187
Foley_24:SWEET:DTW:PS	**5.516**	**0.019**
Foley_24:SWEET:PS:HARD	**9.398**	**0.002**
Foley_24:DTW:PS:HARD	0.469	0.494
AIC=-150.7579	BIC=-3.3397	LogLik=104.3790

The wet and dry models yield contrasting results. In the wet model (2) the interaction between the rainfall index and distance to water is the most significant variable, followed by a three way interaction between the rainfall index, more palatable land type and distance to water. In the dry model (3) there was a wider range of significant interactions all involving the rainfall index and paddock size.

5.4.5 Model validation

The purpose of developing models is to help with prediction of ground cover when data about different factors is available. While the models reported in the section above appear to be strong, it is unclear how useful they are for prediction purposes. In this section a series of simulation exercises are reported that test the predictive validity. These use the models reported above to predict the ground cover of a particular site, which are then compared to the actual measured ground cover. An assessment of the prediction power of the three models from the previous section was carried out using 50 observations (see Table 5.5 for an example). These 50 observations come from field data of ground cover measurements collected in the field in October 2007. Of the three models; the first model is the strongest and probably the most readily applicable model. (See Appendix 5.1 for structural model outputs).

Table 5.5 Example of the predicted model output for site 245 and Model 1

Example of the prediction model output:

Variable	**Coefficient**	**Variable for site 245**	**Coefficient X variable**
Intercept	1.1242421		1.124242
Foley_24	0.2678285	0.203565	0.05452
SWEET	-0.0324855	0.2	-0.0065
DTW	-0.0121933	1.48	-0.01805
HARD	-0.0040291	0.042	-0.00017
Foley_24 X PS	0.0066002	0.197662	0.001305
SWEET X PS	0.0007436	0.1942	0.000144
Foley_24 X HARD	-0.0117123	0.00855	-0.0001
Foley_24 X SWEET X PS	-0.0020936	0.060255	-0.00013
Foley_24 X SWEET X HARD	0.0034499	0.00171	5.9E-06
		Sum	1.155279
		sin	0.914908
		Power ^{2}	0.837056
Predicted Ground cover		%	83.70558
			GCI=83

The relationship of the actual ground cover against the GCI was graphed and the variance tested to establish the measurement error. Then the relationship between the actual ground cover and the model predicted ground cover and the relationship of the GCI measurements and the predicted ground cover was used to establish the predictive error (see Figure 5.7 and Table 5.6). A sample size of 50 sites was used in each case.

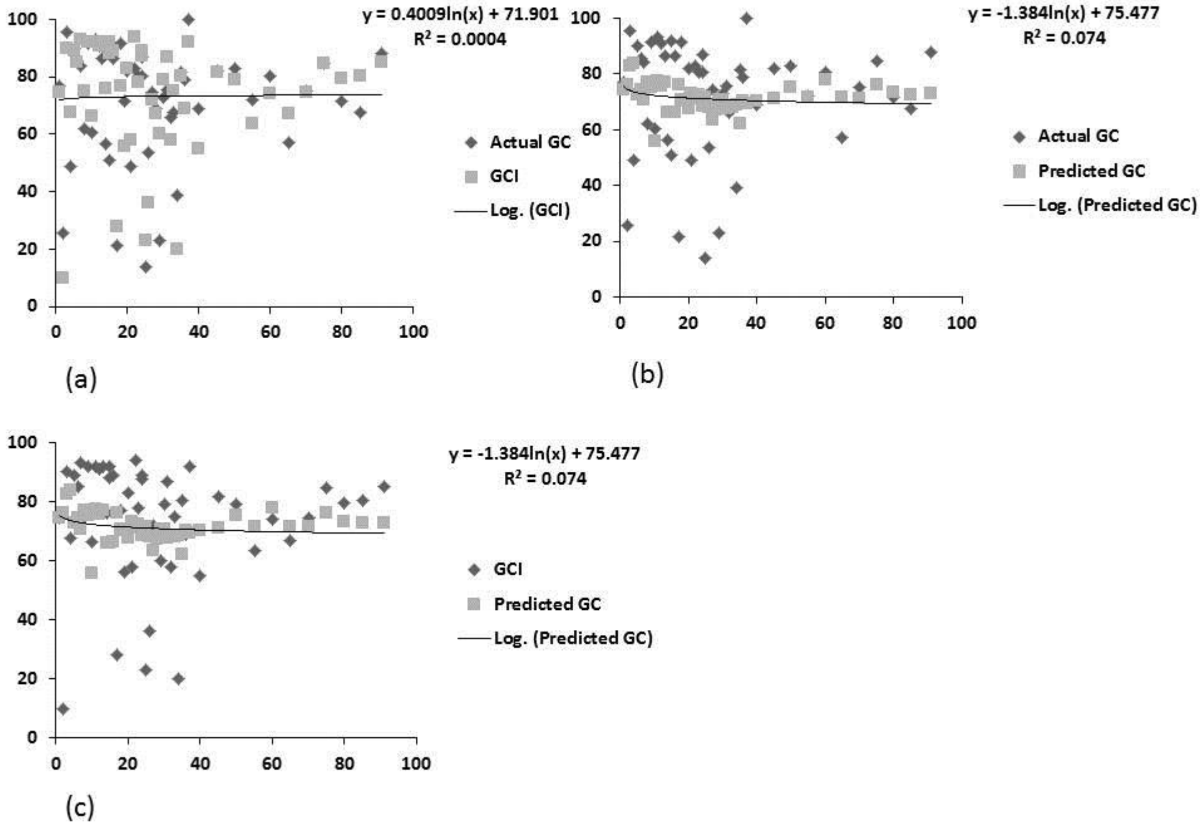


Figure 5.7 Graphs for a) Actual ground cover measurements taken in October 2007 versus GCI measurements in 2007, b) Actual ground cover measurements taken in October 2007 versus the predicted ground cover from the 300 Model, and c) The 2007 GCI versus the predicted ground cover, for 50 sites.

Table 5.6 The correlation statistics of actual ground cover versus GCI measurements, actual ground cover versus predicted ground cover and GCI measurements versus Model 1 predicted ground cover. The sample size for each test was 50.

Predicted values derived from Model 1	**Actual GC versus GCI for October 2007**	**Actual GC for October 2007 versus Predicted GC**	**GCI versus Prediced GC**
Correlation statistic	0.827***	0.031	0.016
Difference in means (standard deviation)	-1.73 (11.846)	0.02 (20.666)	1.75 (20.571)
T-statistic (D.of F.)	-1.033	0.007	0.602

Significance of difference (2 tail test)	0.307	0.995	0.55

***,**,* =significance at 1%, 5% and 10% respectively

Model robustness was examined using data with contrasting rainfall 2 year histories; 1996 was a dry year preceded by dry years, 1991 was wet year preceded by wet years and 2004 was an average rainfall year. By assessing the correlation of the GCI and predicted ground cover through graphs (see Figure 5.8) and T Test the predictive error in the models is established. The T Test results show that the best models are the Dry Model for 1996, the Wet Model for 1991 and 300 Model for 2007 with respect to the expected and predicted variance (see Table 5.7). All models give predictive ground cover close to the actual GCI measurements but without the same level of variance.

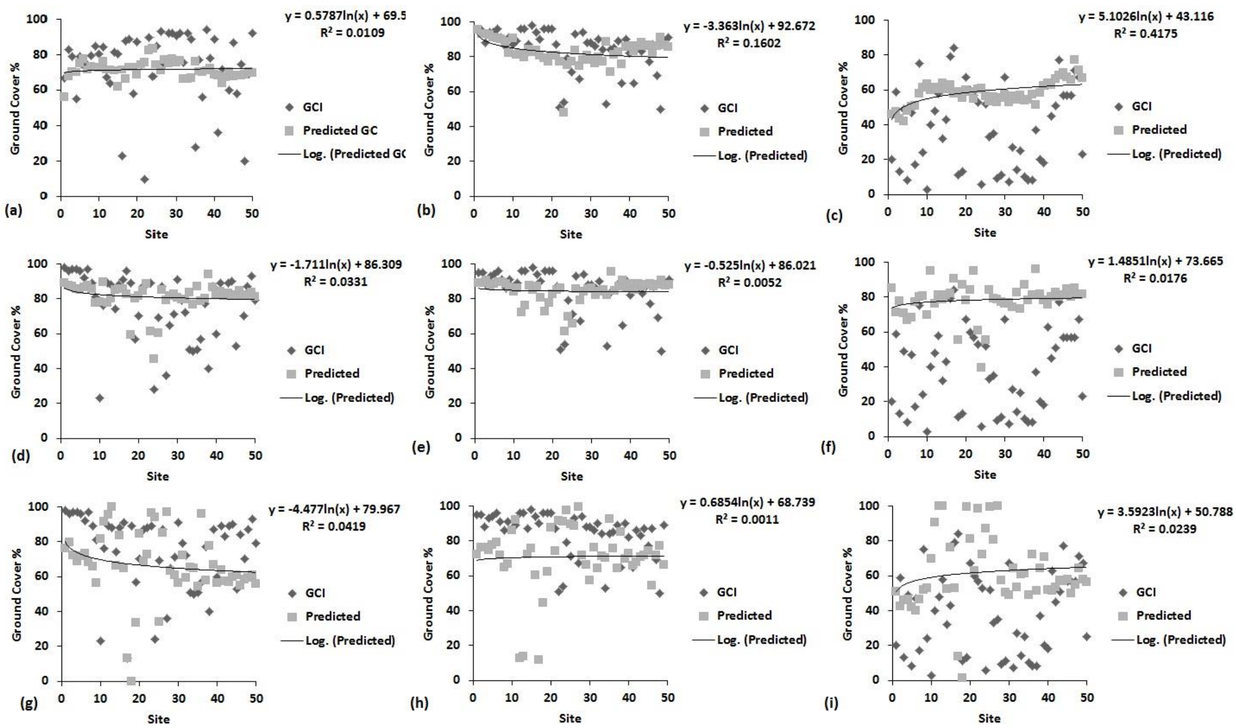


Figure 5.8 The graphs for a) GCI 2007 versus predicted ground cover from model 1, b) GCI 1991 versus predicted ground cover from model 1, c) GCI 1996 versus predicted ground cover from model 1 d) GCI 2007 versus predicted ground cover from model 2 (wet), e) GCI

1991 versus predicted ground cover from model 2 (wet), f) GCI 1996 versus predicted ground cover from model 2 (wet), g) GCI 2007 versus predicted ground cover from model 3 (dry), h) GCI 1991 versus predicted ground cover from model 3 (dry) and i) GCI 1996 versus predicted ground cover from model 3 (dry)

Table 5.7 Statistics of Models 1, 2 (Wet) and 3 (Dry) showing GCI measurements for 2007, 1991 and 1996 and predictive ground cover results from the respective models

		2007	**1991 (wet)**	**1996 (dry)**
Model 1	**Correlation statistic**	0.016	.304**	.411**
	Difference in means (standard deviation)	1.75 (20.571)	1.78 (12.559)	-19.96 (21.869)
	T-statistic (D. F.)	0.602 (49)	1.002 (49)	-6.454 (49)
	Significance of difference (2 tail test)	0.55	0.321	0
Model 2 (wet)	**Correlation statistic**	-0.167	0.149	0.232
	Difference in means (standard deviation)	-8.13 (22.998)	0 (13.272)	-39.7 (23.578)
	T-statistic (D. F.)	-2.499 (49)	0 (49)	-11.906 (49)
	Significance of difference (2 tail test)	0.016	1	0
Model 3 (dry)	**Correlation statistic**	0.005	.369**	0.007
	Difference in means (standard deviation)	6.45 (27.913)	13.66 (25.891)	-23.18 (31.417)
	T-statistic (D. F.)	1.639 (49)	3.731(49)	-5.217 (49)
	Significance of difference (2 tail test)	0.108	0	0

***,**,* =significance at 1%,5% and 10% respectively

5.5 Discussion

In this study, three aims have been achieved. It has been shown that the application of a model of GIS derived data has the ability to predict actual ground cover as demonstrated by the relationships of the 300 model outcomes to actual field measurements of ground cover collected in October 2007 (see Table 5.6 and Figure 5.7). This confirms that the use of remote sensing data to assess ground cover is appropriate and consistent with the creators of the index (Scarth *et al.* 2006).

From the comparison of models derived from 91 sites versus 300 sites, there is value in analysing a larger data set spanning 20 years of changes at a site rather than what is happening at one point in time. Increasing the sample size of data from 91 to 300 sites gives the model more rigour. Additionally, the predicted ground cover from the 300 site model has the same relationship to the actual ground cover measurements as it does to the ground cover index (see Figure 5.7). This model can confidently be used to predict actual ground cover from GIS measured grazing variables and a 24 month rainfall index.

Secondly, the study found that the dominant effect on ground cover was the 24 month rainfall index, where the latter could be quantified and distinguished from the grazing variable effects. This is critical for policy development, as it helps to separate management influences from natural processes. It is important to note that the actual ground cover at a site is not highly correlated to the actual annual rainfall of the ground cover measurement but has a significant correlation to the 24 month foley rainfall index; a measurement of the accumulated rainfall over the preceding 24 months or two years. Therefore when a land manager is setting a stocking rate for a paddock, they should consider the two year rainfall history of the paddock/property. The 24 month Foley rainfall index is an essential parameter for predicting ground cover in any ground cover modelling exercise.

The dominant effect of the 24 month rainfall index as demonstrated in the models is consistent with other studies around the world. Li *et al*. (2013) working in the Saskatchewan, Canada found that rainfall received in the previous year greatly contributes to grass productivity in the current year. They found that the trend of accumulated rainfall was a better predictor than annual rainfall at both temporal and spatial scales. Li *et al.* (2013) also found that stocking intensity was a dominant factor in causing spatial variation in ground cover, but that there was no evidence that past grazing intensity caused a significant negative effect on grasslands. Contrastingly, in a study of rangeland degradation in semi-arid north-eastern South Africa, Wessels *et al.* (2007) found that degradation has a significant influence on the long-term vegetation production, thereby demonstrating that rangelands did not behave in the manner predicted by non-equilibrium theory which predicts grazing to have minimal long-term impacts.

Thirdly, the study found that other grazing management factors are important. Of these distance to water has a significant positive effect and area of more palatable land type in the paddock has a significant negative effect on ground cover. These grazing factor effects only become apparent in a model derived from a large data set. Models could be improved further with characterisation of spatial relationships of water location within the paddock, number of waters within the paddock, location of water in relation to more palatable and less palatable land types, and whether the water was located centrally or in a paddock corner. From the models examined, clearly the amount of more palatable land type in the paddock affects the grazing pressure across the whole paddock. The watering points are usually located within the lower parts of a paddock, especially if they are dams and this would serve to concentrate the grazing pressure. In this ecosystem, the mean paddock size (5,369ha) and mean distance to water (1,948m) make a piosphere grazing effect negligible. The low mean distance to water also explains why the distance to water effect is reduced compared to other studies (Pickup *et al.* 1993; James *et al.* 1999; Pringle and Landsberg 2004).

While the models in this study show promise in their predictive powers, care should be taken in their application, especially in terms of policy reform. Predictive results are close to the actual GCI measurements but do not have the same variability as the actual GCI measurements. The models have predictive power but do not characterise all the variables that may affect ground cover such as soil type, fire events, and stocking history. The models work but are limited by the lack of inclusion of a fundamental grazing variable that would explain the variability in ground cover. In the dry years the models are predicting ground cover at least 20 to 30% higher than it actually is and therefore the models have not incorporated a key explanatory variable that is at play. It is in the dry years that a model would be most useful. In part the inter-actions between variables are counteracting the direct effects of individual variables in dry years as demonstrated in the anova results presented in tables 5.2, 5.3 and 5.4.

The method developed by Bastin *et al*. (2012) to locate areas of most persistent ground cover in years of lowest rainfall from a minimum ground cover image across all years is useful because it does not require measurements of any reference sites or GIS management layers. The Bastin *et al.* (2012) method could be further developed to indicate the recommended levels of ground cover that should be expected in dry years. However, the method does not account for rainfall in the preceding two years, namely the accumulative rainfall effect which may indeed have a greater effect than actual annual rainfall of the actual year being measured. Additionally, looking at the dry years where persistent ground cover exists does not shed light on what is causing lower ground cover levels elsewhere which are the areas of most concern.

In this study, the difference between the wet and dry model is significant; in that whether a site has two years of above rainfall or two years of below average rainfall can result in very contrasting effects on the ground cover at any given time. If a site observation is preceded by above average rainfall, the pasture growth that results from this rainfall makes the ecosystem more resilient to cope with low rainfall with ground cover remaining unchanged and with grazing impacts readily discernible (Model 2), but if a site observation is preceded by below average rainfall, the future resilience of the ground layer can be compromised with ground cover loss evident as a result of various inter-active grazing variables (Model 3).

Omuto *et al*. (2010) used a mixed-effect model of time-series NDVI-rainfall relationship to detect human-induced loss of vegetation cover in the drylands in Somalia. Their time-series mixed effect model of NDVI $_{max}$ and rainfall produced residuals that were correlated with human-induced loss of vegetation accounted for different responses of different vegetation types to rainfall. Modelling that combines both the variation in grazing impact and rainfall variables of this current study together with the responses of different types of vegetation responses to these variables as per Omuto *et al*.'s (2010) work would be useful for the management and ecological maintenance of a temporal and spatially dynamic ecosystem such as these silver-leaved ironbark woodlands.

The use of a model can help to establish threshold ground cover levels – at both a temporal and spatial level with respect to accumulative rainfall effects and grazing variables. With further fine tuning a workable and more accurate model could readily be established in which the rainfall and grazing variables could be better characterised to predict their combined effects. A landholder could readily examine several possible scenarios early in the season to forecast the grazing limitations of the system.

5.6 Appendix: Structural model outputs

Model 1

Predictor Variables	Coefficient	Standard Error	T value	P value
Intercept	1.124	0.016	70.137	0
Foley_24	**0.268**	**0.009**	**30.988**	**0**
Sweet	**-0.032**	**0.011**	**-2.872**	**0.004**
DTW	**-0.012**	**0.004**	**-2.895**	**0.004**
HARD	-0.004	0.003	-1.604	0.109
Foley_24 X PS	0.007	0.003	1.977	0.048
SWEET X PS	0.001	0.000	1.610	0.107
Foley_24 X HARD	-0.012	0.005	-2.393	0.017
Foley_24 X SWEET X PS	-0.002	0.001	-2.528	0.012
Foley_24 X SWEET X HARD	**0.003**	**0.001**	**2.978**	**0.003**
AIC=-3650.346	**BIC=-3555.968**	**LogLik=1839.173**		

Note: Dependent variable is GCI measurement for 300 random sites over 21 years

Model 2

Predictor Variable	Coefficient	Standard Error	t value	p value
Intercept	1.186	0.013	93.101	0
Foley_24	0.167	0.016	10.654	0
SWEET	**-0.062**	**0.016**	**-3.811**	**0.000**
PS	0.000	0.000	2.333	0.020
HARD	-0.027	0.011	-2.478	0.013
Foley_24 X DTW	**-0.023**	**0.009**	**-2.038**	**0.008**
SWEET X DTW	**0.027**	**0.006**	**-4.222**	**0.000**
PS X DTW	**0.000**	**0.000**	**-3.525**	**0.000**
PS X HARD	0.000	0.000	2.578	0.010
HARD X DTW	0.003	0.002	1.718	0.086
Foley_24 X SWEET X PS	**0.000**	**0.000**	**2.994**	**0.003**
Foley_24 X SWEET X DTW	**-0.029**	**0.007**	**-4.285**	**0.000**
Foley_24 X PS X DTW	**0.000**	**0.000**	**2.640**	**0.008**
Foley_24 X PS X HARD	**0.000**	**0.000**	**-2.694**	**0.007**
AIC=-2148.963	**BIC=-2047.406**	**LogLik=1092.482**		

Note: Dependent variable is GCI measurement for 300 random sites over 21 years

Model 3

Predictor variable	**Coefficient**	**Standard Error**	**T value**	**P value**
Intercept	1.043	0.513	20.312	0
Foley_24	**0.357**	**0.069**	**5.164**	**0**
SWEET	**-0.104**	**0.339**	**-3.064**	**0.002**
DTW	-0.050	0.021	-2.382	0.017
PS	-0.030	0.015	-2.083	0.038
HARD	-0.022	0.035	-0.648	0.517
Foley_24 X DTW	**-0.087**	**0.031**	**-2.773**	**0.006**
SWEET X DTW	0.035	0.013	-2.583	0.010
Foley_24 X PS	**-0.070**	**0.020**	**-3.564**	**0.000**
SWEET X PS	0.035	0.004	0.749	0.454
DTW X PS	0.008	0.004	1.882	0.060
SWEET X HARD	**0.064**	**0.024**	**2.053**	**0.008**
DTW X HARD	-0.002	0.010	-0.223	0.824
PS X HARD	-0.001	0.001	-1.476	0.140
Foley_24 X SWEET X PS	0.009	0.004	1.965	0.050
Foley_24 X DTW X PS	**0.029**	**0.007**	**4.079**	**0.000**
SWEET X DTW X PS	-0.001	0.002	-0.968	0.333
Foley_24 X SWEET XHARD	**0.050**	**0.019**	**2.669**	**0.008**
Foley_24 X DTW X HARD	-0.017	0.013	-1.260	0.208
SWEET X DTW X HARD	-0.011	0.006	-1.753	0.080
SWEET X PS X HARD	**-0.001**	**0.000**	**-3.044**	**0.002**
DTW X PS X HARD	0.001	0.001	1.750	0.079
Foley_24 X SWEET X DTW X PS	**-0.006**	**0.002**	**-3.179**	**0.002**
Foley_24 X SWEET X PS X HARD	**-0.001**	**0.000**	**-3.003**	**0.003**
Foley_24 X DTW X PS X HARD	0.000	0.000	0.684	0.494
AIC=-150.7579	**BIC=-3.3397**	**LogLik=104.3790**		

Note: Dependent variable is GCI measurement for 300 random sites over 21 years

Chapter Six

Using the Ground Cover Index to assess ecosystem condition from the site to landscape scale

6.1 Introduction

Vegetation and total surface cover reduce runoff and soil loss by maintaining porosity for greater infiltration, reducing slaking and surface sealing, and directly intercepting rainfall drops (Eldridge and Koen 1993). Previous research has shown that ground cover is key indicator of ecosystem function, especially in terms of preventing runoff and soil loss (Scanlan and McIvor 1993; Dube *et al*. 1999). Therefore a measurement of ground cover and types of ground cover should give a good correlation to ecosystem condition.

Australian rangelands soils are old, fragile and inherently low in carbon, nutrients and organic matter (Ahern *et al*. 1994) and depend on the limited plant growth to help maintain condition (Williams *et al*. 1993). Their soils originate from weathered, deflated, leached and reworked older deposits which means these red and yellow earth kandosols of the Desert Uplands have a low content of phosphorous and nitrogen (Orian and Milewski 2007). Germination and establishment of pasture is only possible during periods of high soil moisture. Low nutrient soils favour longer leaf life spans, with foliage being less digestible to herbivores in an infertile environment (Morton *et al.* 2011). Doubling plant life span halves the yearly uptake of N and P needed to sustain a given area of leaf (Westoby *et al.* 2002). Increased leaf longevity leads to a dominance of perennial species with evergreen foliage. Perennial plants have the capacity to photosynthesise rapidly when soil moisture is available to achieve high levels of standing biomass (Orians and Milewski 2007).

Vegetation promotes ecosystem stability by enhancing soil biological activity and cycling of nutrients (Ludwig *et al.* 1994). Tongway and Ludwig (1994) reported localised enrichment of nitrogen, phosphorus, and soil carbon in close proximity to tussocks in semi-arid rangelands, compared to areas without plants. Grass tussocks act as small barriers, forcing the runoff to a follow a sinuous path compared to movement across bare spaces (Ludwig *et al*. 1996). The physical barriers made by tussocks allow some water and nutrients to infiltrate the soil, and deposition of litter and sediment on the surface. Grazing pressure can cause reduction in root systems and live basal area of tussocks which in turn makes tussocks less resistant to grazing. Heavy grazing can cause reductions in soil-borne plant material and lower nutrient concentrations, which make grass tussocks less resilient from a nutrient cycling perspective (Ludwig *et al.* 1994).

In this chapter, the relationship of the ground cover and environmental measurements are assessed. The strength of the ground cover index (GCI) and a derivative the ground cover disturbance index (GDCI) as a predictor of actual ground cover are evaluated. The relationship of ground cover (both actual field measurements and ground cover index) to other measurements of ecosystem structure, plant and bird diversity is also explored in terms of their application in a landscape biodiversity condition index.

The results from the correlation of field ground cover measurements and plant diversity and abundance is examined in light of the Ground Cover Disturbance Index developed by the Department of Environment and Resource Management for the assessment of ecosystem condition as an input into Biodiversity Planning Assessments at a bioregional scale (Ward 2006). Improvements to this index are outlined.

6.2 Background

6.2.1 Ecosystem Structure and Function over time

Ecosystems are involved in the accumulation, circulation and transformation of energy and matter through biological processes, such as photosynthesis, herbivory and decomposition. Within any ecosystem the non-living part is involved in evaporation, precipitation, erosion and deposition, reacting to the living part, and co-actions between organisms (Dyksterhuis 1958). Ecosystems are dynamic by nature and affected by the climate. Biological diversity occurs at several hierarchical levels, from genes to individuals, from populations, species, communities, and ecosystems to landscapes. At each level there are important relationships between biodiversity and ecosystem functions and between biodiversity and the ways in which ecosystems respond to disturbance. Relationships between biological diversity and ecosystem function are inherently complex and operate at many spatial and temporal scales. At an intermediate scale, contagious disturbance processes such as fire, insect outbreak, plant disease, grazing and water flow dominate the formation of patterns over spatial scales of hundreds of meters to hundreds of kilometres and on time scales of years to decades (Risser 1994). Risser (1994) argues that it is at this intermediate scale where both direct and indirect effects have their greatest consequences for biodiversity and ecosystem behaviour. For example, the relationship of biodiversity with drought recovery fits this model (Tilman and Downing 1994). Grazing, drought and fire occur at intermediate temporal and spatial scales and are a few of the dominating processes that structure grasslands in the central plains of North America (Risser *et al*. 1981).

Similar issues apply to rangelands in Australia. In a report of change of landscape function for the period of 1992 to 2005, Carter *et al.* (2007) stated that there was a large decrease in landscape function based on rapid mobile data assessment of the Alice Tableland subregion of the Desert Uplands, but that the GCI readings[6] indicated no change in landscape function. This was coupled with an increased utilisation in the period of 1991-2005 (24.49%) compared with the period of 1976-1990 (15.03%) based on Aussie-GRASS simulated measurements. These increases in pasture utilisation are attributed to tree clearing and the establishment of exotic pasture species allowing increased grazing intensity. They also suggest that utilisation may have increased due to relatively low rainfall between 2002 and 2005.

[6] GCI readings are a remotely sensed estimate of ground cover derived from Landsat imagery

The Australian Collaborative Rangeland Information system (ACRIS) has proposed a reporting concept based on the Richard/Green Functionality index in which a subregion is reported on by way of three functionality groupings from highly functional to poor functioning against trends of improving, stable and declining condition (see Table 6.1). It is an index of landscape function that has been derived from the frequency and cover of weeds, perennial plant species, cryptogam and soil stability. Functional landscapes are likely to recover quickly from disturbance, and to maintain a consistent vegetation cover through variable seasonal conditions. Dysfunctional landscapes may not recover, take longer to recover or change to a less desirable condition state (Grant *et al.* 2007)

Table 6. 1 Richards/Green Functionality Index

Functionality		Trend	
1	**Highly Functional:** low number of invasive species. Ideal species list, relevant cryptogam cover. Low soil erosion, High perenniality. Landscape patches undisturbed. Bare soil areas restricted.	1	**Improving:** Increasing size/frequency of patches, number of ideal species, perenniality. Decreasing: soil erosion, bare soil areas. Stable or increasing: number of obstructions.
2	**Functional:** Some invasive species, average no. of ideal species. Relevant cryptogam cover not to full potential. Some: soil erosion, perennials, undisturbed landscape patches, bare soil areas.	2	**Stable:** Maintenance of stability or near stability of the above
3	**Poorly Functional:** Many invasive species present. Much soil erosion. Few undisturbed patches, few perennials, large areas of bare soil, few obstructions.	3	**Declining:** Decreasing: size/frequency of patches, perennials, ideal species, relevant cryptogam cover, obstructions. Increasing soil erosion: bare areas and number of invasive species.

6.2.2 Assessing Ground Cover

There is potential for remote sensing data to provide good information about ground cover. Landsat TM has low cost, extensive archival data extending back to 1987.

Landsat TM data cannot provide direct information on plant species composition, but ground cover indices are closely correlated to the amount of vegetation cover present and as such are good surrogate measures of cover (see review of the remote sensed methods in Chapter Two and plant analysis results from Chapter Three).

In northern Greece, Roder *et al.* (2008) carried out linear trend analysis to characterise spatio-temporal patterns of vegetation cover change of a remote sensing data time series for retrospective assessment of rangeland process and interpreted these cover changes in light of grazing practices and management interventions. They used linear spectral mixture analysis to infer quantitative estimates of green vegetation cover on a per-pixel basis, which may be interpreted in the context of land degradation processes. A degradation index was derived which combined direction and magnitude of the trend and the average level of cover with three classes of average cover – low, medium and high, and overall gain rates as strongly decreasing, decreasing, increasing and strongly increasing. Each of these groups was related to stocking rate development to confirm a direct negative relation. The results confirmed that decreasing cover estimates marry with increasing animal stocking rates.

6.2.3 Ground Cover Index

In Queensland, estimates of ground cover are derived on an annual basis from Landsat TM satellite imagery through a ground cover index. The ground cover index (GCI) is calculated from a multiple regression model between Landsat bands 3, 5 and

7 and ground cover measured at sites covering much of the variation in climate, soils and vegetation across Queensland (Scarth *et al.* 2006; Karfs *et al.* 2009). The GCI is derived from transformed Landsat data originating from research by Taube (2000). GCI integrates grasses and forbs, grass and tree litter and cryptogams to provide an estimate of cover which has root measure regression error of +- 13%.

Repeated ground measurement of land condition variables including cover and species composition can provide data which can be aggregated to quantify regional trends (Watson *et al.* 2007) and benchmark condition (Friedel *et al.* 2000). Remote sensing provides quantitative means to consistently identify temporal change in rangeland vegetation at multiple scales from the paddock to property to landscape (Karfs *et al.* 2009). A large body of research has been undertaken in the Australian rangelands over the last 25 years, studying the relationship between reflected surface electromagnetic radiation recorded by Landsat satellites and vegetation cover measures at field sites (e.g. Pickup *et al.* 1993; Bastin *et al.* 1996; Taube 2000; Karfs 2002; Ludwig *et al.* 2007). The capacity to detect and monitor relative change in vegetation cover has been developed. Ecosystem condition statements rely on the analysis or interpretation of patterns of ground cover change, and which can then be extrapolated over a landscape (Karfs 2002). Landsat is suitable to monitor the vast areas of Australian rangelands where the underlying trend in condition is affected by climatic variability operating over decades (Pickup *et al.* 1998).

6.2.4 Ground Cover Disturbance Index

The GCI has then been used to derive the Ground Cover Disturbance Index (GCDI) established by the Queensland Department of Environment and Resource Management (2011). In this index, the mean, variance and trend of the Ground Cover Index has been quantified for the 21 years between 1988 and 2009 based on the following formula: GDCI = GC Mean + GC Trend + GC Standard Variation, with different combinations of mean and trend indicating different levels of disturbance (from very low to very high). The GCDI is inaccurate in areas where the foliage projective cover (FPC) is greater than 20 percent, and also identifies low change areas which are likely to be naturally bare (e.g., bare rock, scalded areas). The GCDI results in the classification of the landscape to sixteen levels of disturbance at the resolution of the 25m pixel of Landsat satellite imagery (see Figure 6.1). This is represented spatially for the silver-leaved ironbark woodlands in the Desert Uplands bioregion in Figure 6.2. However, the index fails to consider the effects of fire scars on trend. Areas with high ground cover have high biomass and therefore are susceptible to wildfire. Additionally, this GCDI index fails to consider increases in trend that are likely to be attributable to increases in rainfall.

Disturbance Level	High Ground Cover	Above Mean Ground Cover	Below Mean Ground Cover	Low Ground Cover
Increasing trend	1 – Very Low (Benchmark)	5 - Low	9 - Medium	13 - High
Slight increase in trend	2 – Very Low (Benchmark)	6 - Low	10 - Medium	14 - High
Slight decrease in trend	3 - Low	7 - Medium	11 - High	15 – Very High
Decreasing trend	4 - Low	8 - Medium	12 - High	16 – Very High

Figure 6.1 Matrix of Ground Cover Disturbance Index shows the sixteen levels of disturbance

In this chapter, the relationship of ground cover with other environmental variables such as total basal area, over-storey cover, mid-storey cover, perennial grass cover, cryptogam cover and litter cover are analysed. The relationship of ground cover to grazing variables are analysed; these include: - distance to water, stocking rate and artificial water density. The spatial-temporal relationship of ground cover at the site, paddock, property and landscape level was explored with respect to the Foley_24 rainfall index to allow for the influence of climate to be controlled. Of most interest is the use of ground cover as a measure of ecosystem condition at the landscape level.

The climate is a dominant driver of ecosystem condition as discussed in Chapter Two with respect to the differences between non-equilibrium and equilibrium theory. Therefore the trend of ground cover over time is examined with a rainfall index, the Foley_24 month index, as a result of modelling carried out in Chapter Five.

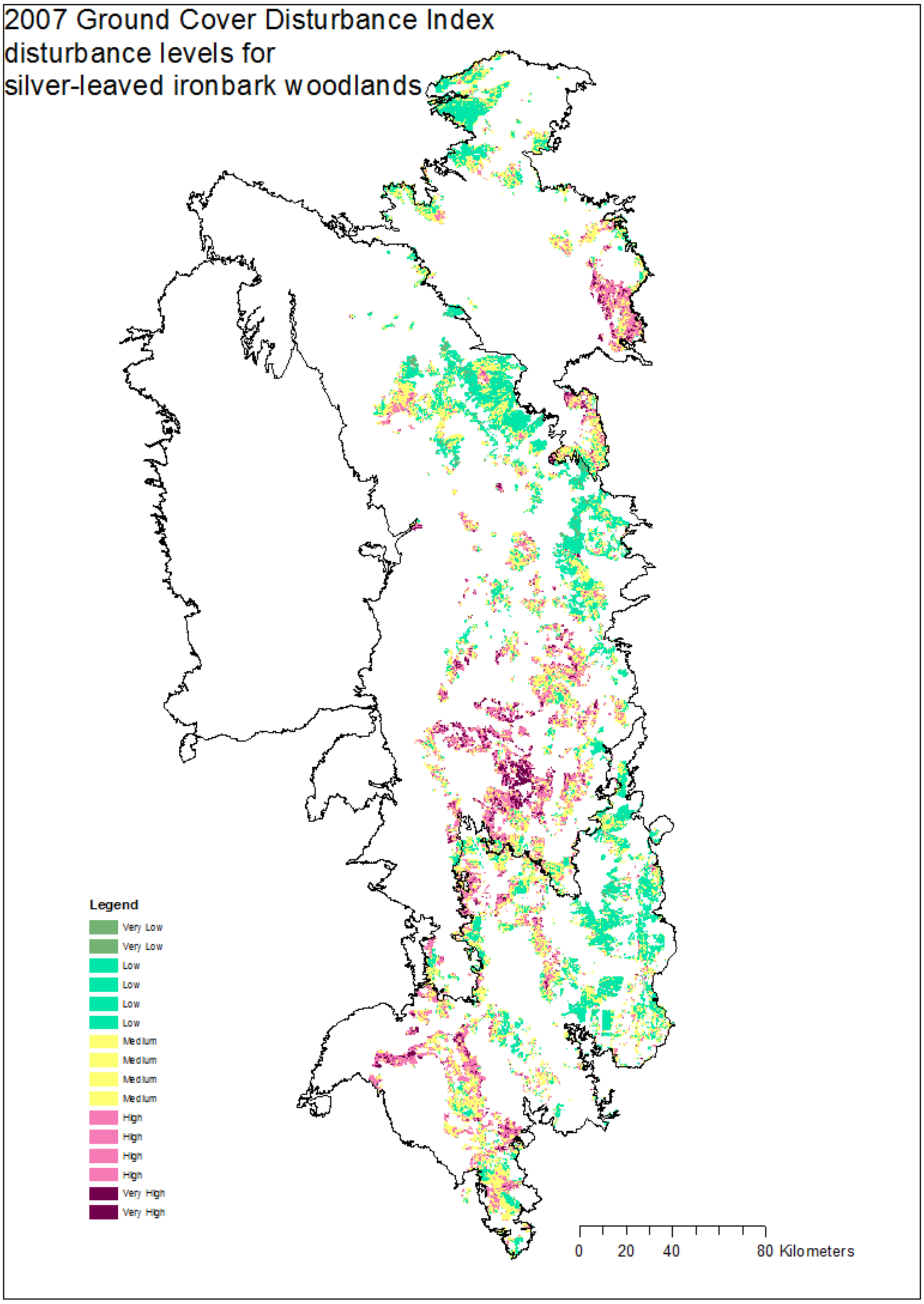


Figure 6.2 The silver-leaved ironbark woodlands classified by the GCDI as at 2007, showing substantial areas of below average ground cover (which have a decreasing trend and low ground cover).

Hypotheses tested:

I Actual ground cover measurements have a significant relationship to other environmental variables such as pasture biomass, perennial green leaf cover, total basal area, mid storey cover and grass height. Additionally, the actual ground cover is significantly correlated to the ground cover index as presented in chapter three and chapter five.

II Actual ground cover measurements are correlated to grazing variables.

III Foley rainfall index is an important explanatory variable of temporal ground cover trends.

IV The ground cover index as categorised by the Ground Cover Disturbance Index to indicate ecosystem condition can be improved with simplification using the Richards/Green functionality index and the incorporation of the Foley's index to account for rainfall variability.

6.3 Methodology

6.3.1 Study area and target ecosystem

Silver-leaved ironbark (*Eucalyptus melanophloia*) woodlands cover over a million hectares in the Desert Uplands bioregion (Figure 6.3) and are the most widespread ecosystem with grazing production values in the bioregion. The herbaceous layer of this ecosystem is a grassland which has a fairly even spread of tussocks. The targeted ecosystem for this study is Regional Ecosystem 10.5.5: - *Eucalyptus melanophloia* woodland with an open grassland understorey of *Aristida* spp. and/or *Triodia* spp. on loamy red and yellow earths and undulating sand plains (Sattler and Williams *eds*. 1999). Mean annual rainfall varies from 490mm in the north of the study area to 560mm in the south, and is summer dominant.

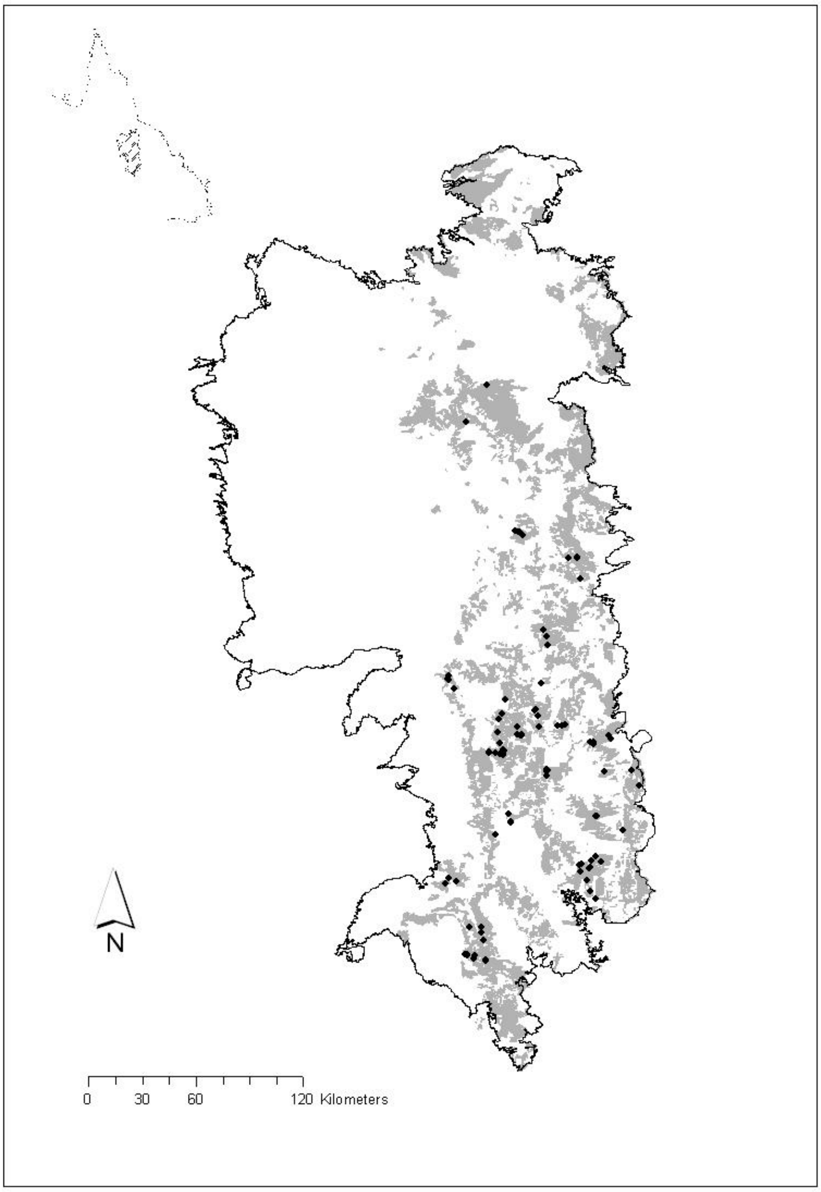


Figure 6.3 The Desert Uplands bioregion in Queensland, Australia, with the target regional ecosystem indicated by the grey shading (non-remnant and ≥20% projective foliage cover excluded). This represents a total area of 465 084 ha. The location of the study sites are indicated (black symbol).

Site selection and sampling was carried out as per the methodology outlined in Chapter Three. Twenty five grazing properties with the highest proportion of the targeted regional ecosystem were selected. Within these properties 91 sites that could be readily accessed and which represented a wide range of ground cover were selected for study (see Figure 6.3). Ground cover data was collected as per the methodology outlined in Chapter Three. This chapter reports on the results of the methodology as it was repeated at the same sites in October 2007.

The actual ground cover for each site was compared to the field measures of ground cover using linear regression analysis. Ground cover measurements were also compared to grazing variables.

6.3.2 Geographic Information System (GIS) assessment

The ground cover index reading for each pixel of each of the 91 sites was recorded for the years 1987 until 2007 – therefore there are 21 records of ground cover for each site. These have been grouped into the 57 paddocks within which these sites fall and the paddock mean, median and standard deviation of ground cover for each of the 21 years was measured from the Landsat derived GCI. Additionally the mean, median and standard deviation of ground cover for each year for the 25 properties on which the paddocks and sites was also determined from the GCI. To assess trends of ground cover across the whole distribution of the silver-leaved ironbark woodlands, the mean, median and standard deviation and trend via slope of ground cover was collated for each year. Note: the site, paddock, property and ground cover assessments only occurred in remnant areas of the ecosystem where the projective foliage cover is less than 20%.

In an examination of the patterns of the GCI measurements of ground cover over time, the following observations were made. In the following Figure 6.3 the location of the 91 sites within paddocks and properties are shown and in Figure 6.4 the boxplots of the variance of ground cover of the pixels (sites), paddock and properties are presented.

The three boxplots of GCI follow similar yearly patterns; however, the range of ground cover within years decreased with an increasing scale from site to paddock to property. There are noticeable differences, especially at the property scale whereby the ground cover mean and variance does not parallel the pattern of the sites and paddocks because within the property area there are other land types (all areas with a projective foliage cover of greater than 20% are masked out). For example, in 1988 the property scale average ground cover is above 80%, while at the paddock and site scale the average is around 40%. This variation at different scales is likely to be caused by a range of factors such as climate variation, wildfires, and different grazing management regimes. The boxplots also reveal the resilience of the ecosystem in its ability to recover in terms of cover after dry years and its capability to capture energy resources to produce organic cover.

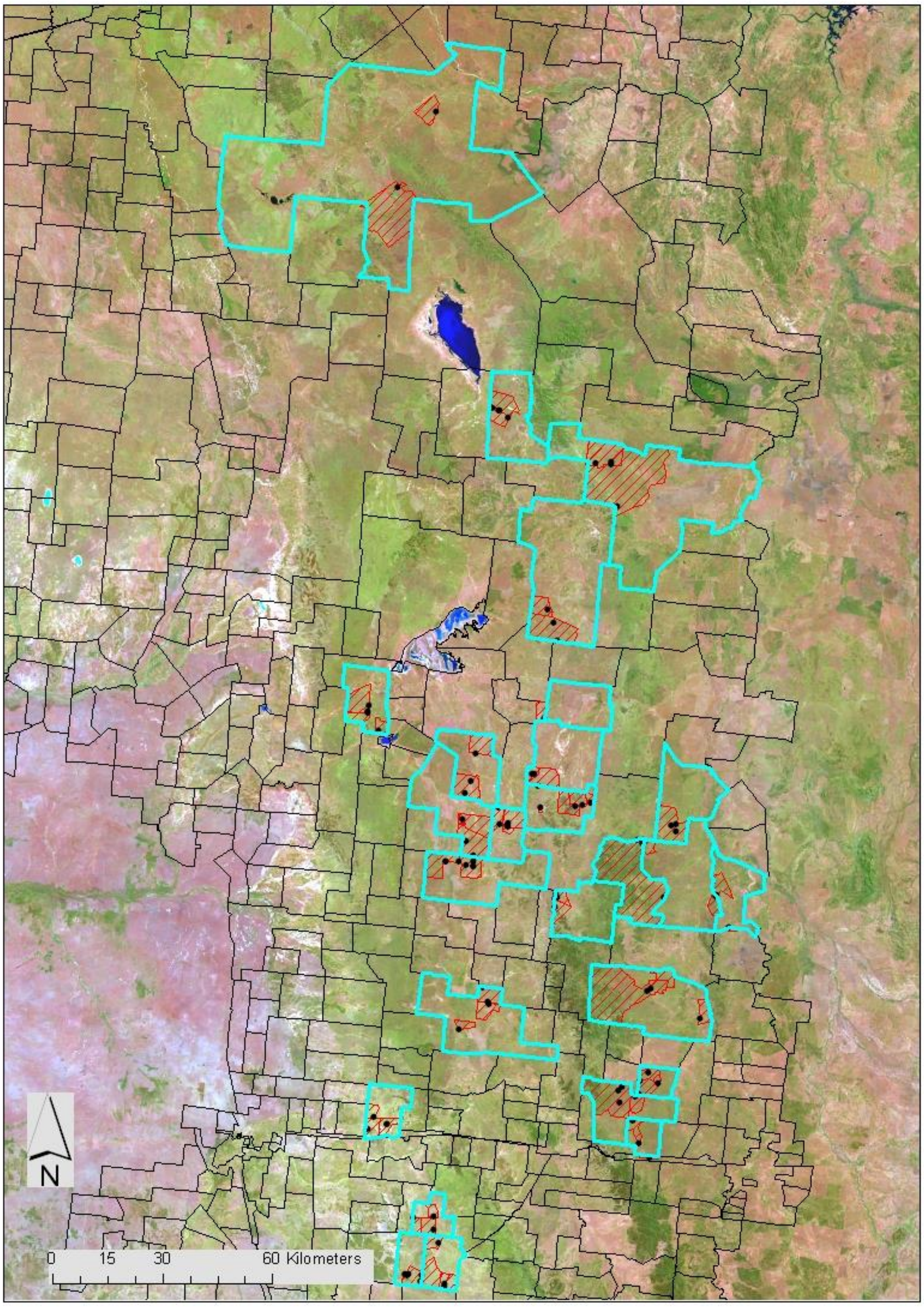
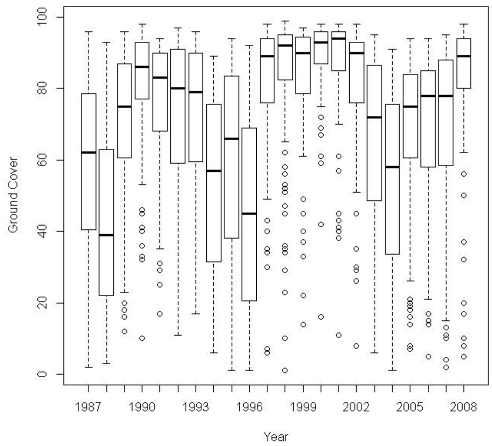
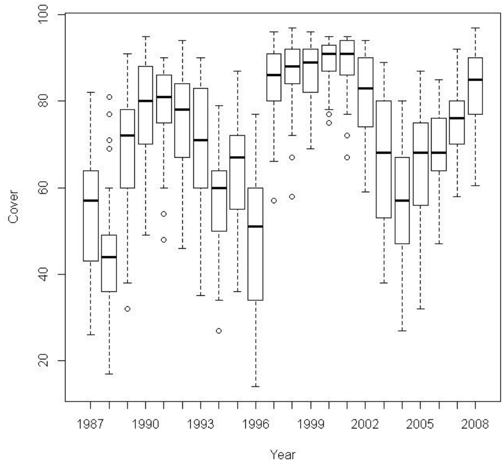


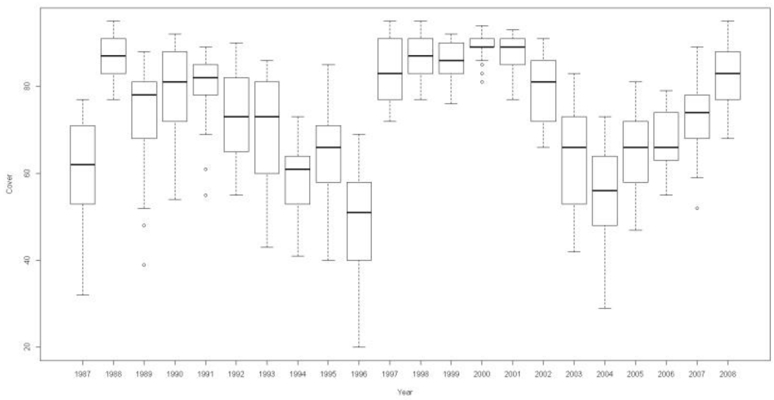
Figure 6.4 Landsat TM imagery showing the location of the 91 sites within the 57 paddocks (red hatching) within the 25 properties highlighted in light blue.



(a)



(b)



(c)

Figure 6.5 Boxplots of the ground cover variation (a) across the 91 sites, (b) across the 57 paddocks and (c) the 25 properties within which the sites are located

In Figure 6.2 the map of the ground cover index for silver-leaved ironbark as at 2007 is coloured by the classifications of the ground cover disturbance index matrix outlined in figure 6.1, showing that most of the ecosystem is considered to be in poor condition. The mean ground cover for the silver-leaved ironbark was 80.05% in 2007 (as shown in Figure 6.6), so the high ground cover would be 89% to 100%, above mean ground cover – 80.5% to 88%, below mean cover 39% to 80.5% and low ground cover would be 38% and below. This shows a lot of the ecosystem as being highly disturbed, however, in reality most of the ecosystem has above the long-term average ground cover.

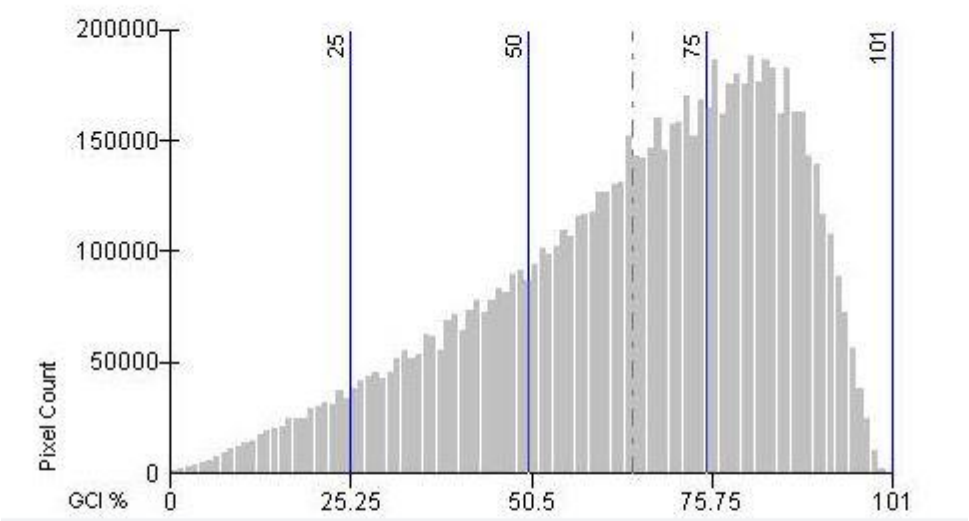


Figure 6.6 Histogram of the 2007 GCI for the silver-leaved ironbark ecosystem with 3% of pixels with <25% ground cover, 17% of pixels with >25% and <50% ground cover, 40% of pixels with between >50% and >75% ground cover and 40% of pixels with <75% ground cover

6.4 Results

Figure 6.7 Photos showing the two extremes of cover – low ground cover and high ground cover in silver-leaved ironbark woodlands (May 2007)

6.4.1 Hypothesis I

The ground cover was significantly correlated with other environmental variables:- specifically total basal area, over-storey cover, mid-storey cover, perennial grass cover, cryptogam cover and pasture biomass (see Figure 6.8 and Table 6.2). This relationship is at the end of dry season for the biomass and shows maintenance of perennial grass cover.

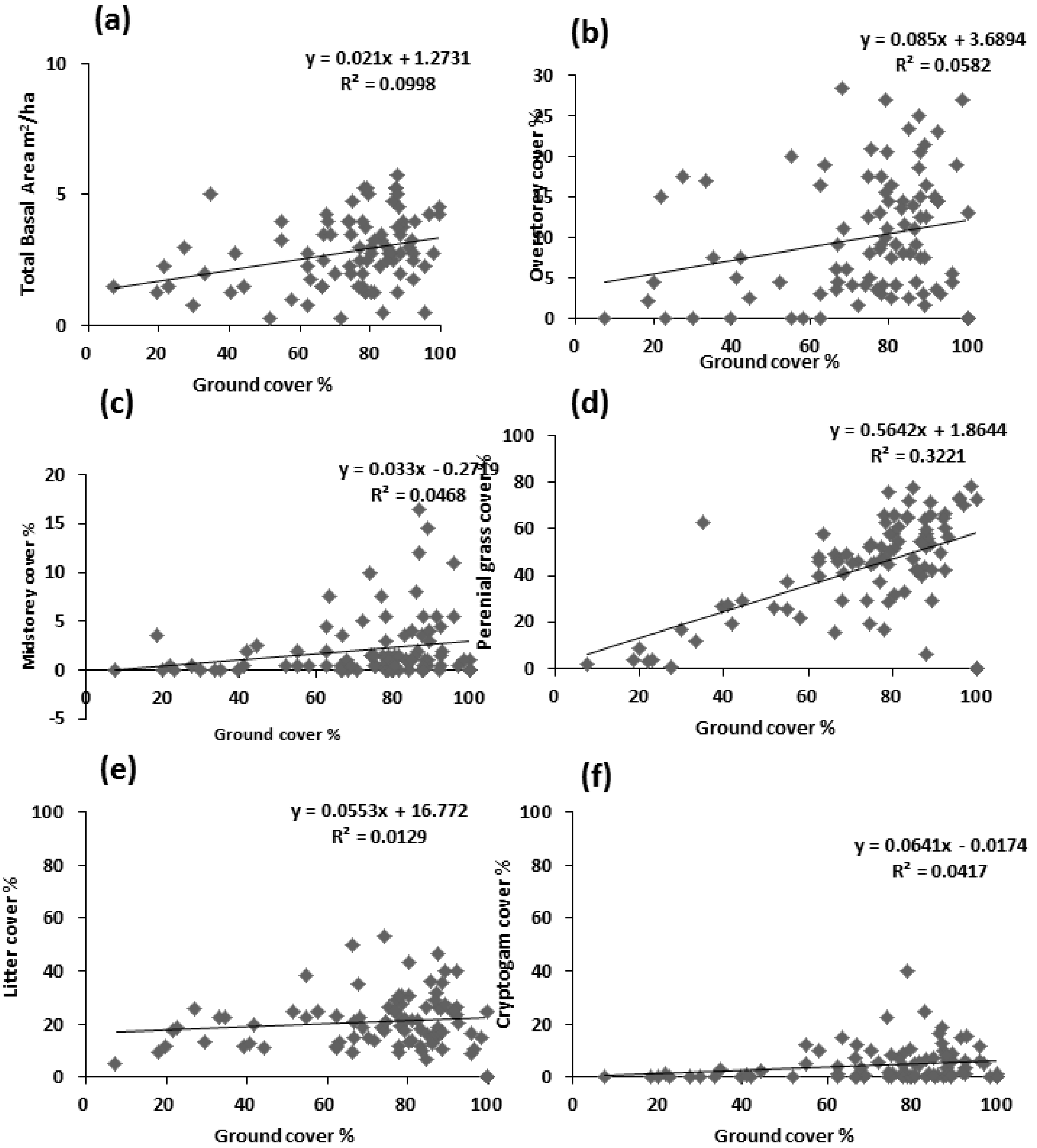


Figure 6.8 Relationship between actual ground cover measured in October 2007 and other environmental variables measured at the same time (a) total basal area (m2/ha), (b) Over-storey cover, (c) Mid-storey cover, (d) Perennial grass cover, (e) Litter cover and (f) cryptogam cover**.**

Table 6.2 Regression of Ground cover October 2007 with environmental variables

Cover Measurement	Constant	Slope	R^2	Significance (linear)
Total Basal Area m^2/ha	1.273	0.021	0.089	.003
Overstorey cover %	3.689	.085	0.048	0.021
Midstorey cover %	.033	-0.272	0.047	0.040
Perennial Grass Cover %	1.864	.564	.314	0
Litter cover %	16.772	.055	0.002	0.284
Cryptogam cover %	-.017	.064	0.031	0.052

The significance of the regression function is used as the test of the relationship between actual ground cover and environmental variables. Field measured ground cover in October 2007 was significantly correlated ($R2$ = 0.717, P < 0.001, Fig. 6.9*a*) to the Ground Cover Index of September 2007, and to the biomass measurements taken in October 2007 ($R2$ = 0.232, P < 0.001, Fig. 6.9*b*). Outlier sites may be due to different levels of wet season growth or grazing pressure between the satellite capture and field sampling, or may actually represent limitations in discriminating between bare soil and plant cover in some situations.

The regression curve fit of actual ground cover measurements has significant relationships with total basal area, over-storey, mid-storey, perennial grass cover, cryptogam cover, pasture biomass and with ground cover measurements taken in October 2007 (see Figure 6.8 and Table 6.2).

From the measurement of the 91 sites there was found to be a significant relationship between the actual ground cover in October and the ground cover index measurements for the same year (see Figure 6.9).

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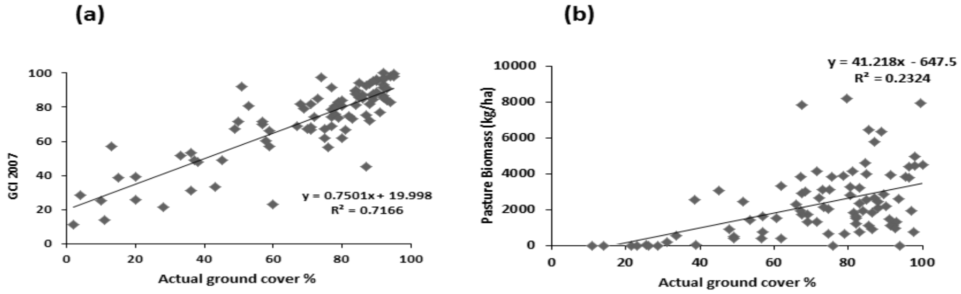

Figure 6.9 Relationship between ground cover measured in the field and (a) the 2007 Ground cover index, and (b) the biomass

Ground cover measured in October 2007 has significant correlation to the GCI and pasture biomass. The most significant relationships are presented in Figure 6.9 and Table 6.3.

Table 6.3 Regression relationship of Ground Cover % October 2007 and GCI

Cover Measurement	Constant	Slope	R^2	Significance (linear)
Biomass kg/ha Oct 07	-313.313	43.292	0.167	0
GCI October 2007 %	-.794	0.962	0.694	0

6.4.2 Hypothesis II

Tests for the third hypothesis reveal that ground cover is significantly related to grazing variables. Data on water density and distance to water were measured from remotely sensed Landsat imagery but the stocking rates were obtained from landholders. The negative relationship of the grazing surrogates ha/water and positive relationship of distance to water to ground cover are significant, but the stocking rate is not (AE/ha)(see Table 6.4 and Figure 6.10).

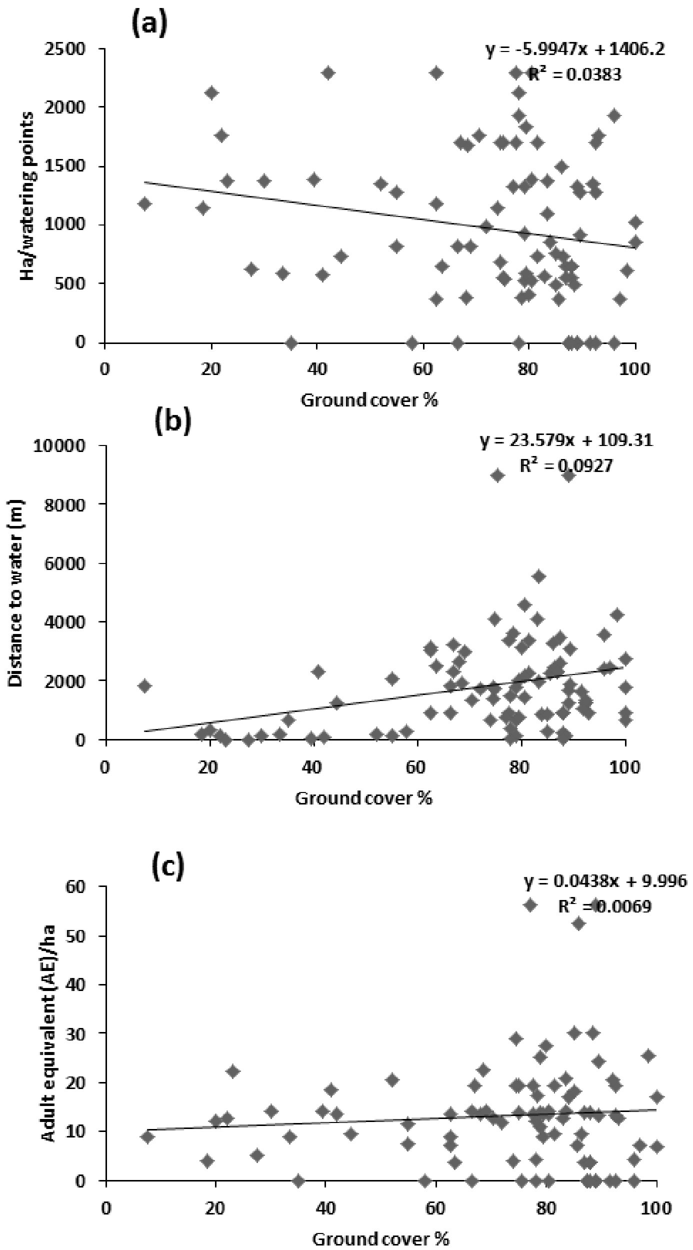

Figure 6.10 Relationship of grazing variables to ground cover (a) ha/water, (b) distance to water (m) and (c) Adult Equivalent stock/ha

Table 6.4 Regression relationship of ground cover to grazing variables

Grazing indicator	Constant	Slope	R^2	Significance
Distance to water (m)	109.314	23.579	0.083	0.003
Ha/watering points	1406.195	-5.995	0.027	0.068
AE/ha	9.996	.044	-.005	0.441

6.4.3 Hypothesis III

Rainfall in the preceding two years is a significant driver of ground cover; while this was established in the previous chapter it is demonstrated here to also identify temporal trends in ground cover change.

When the ground cover index for the whole distribution of silver-leaved ironbark is examined in relation to rainfall, there are definite wet and dry year variations of cover. In the dry years there is a wider range of ground cover across all pixels than in the wet years (see Figure 6.11).

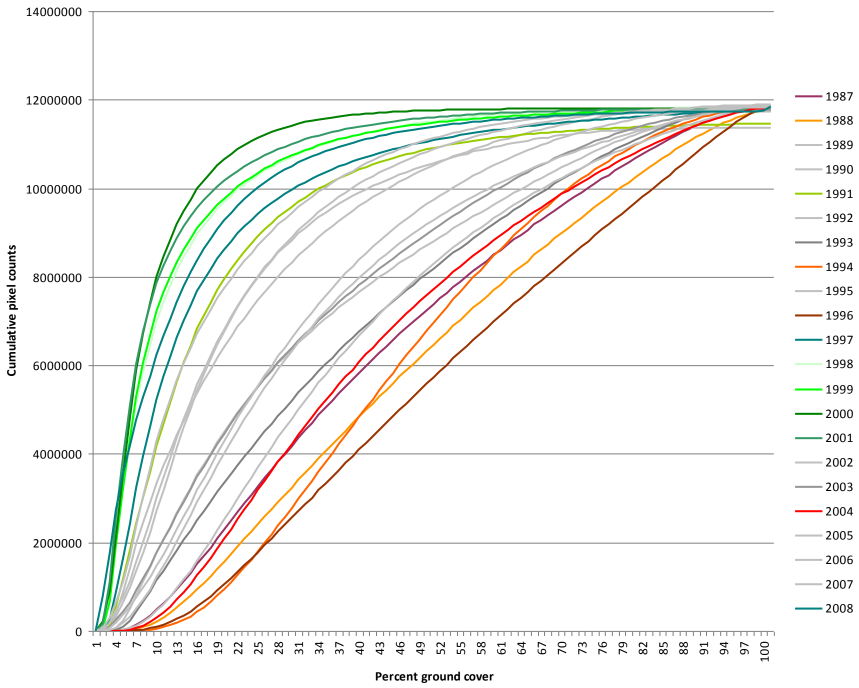

Figure 6.11 Cumulative pixel count by percent ground cover for each year of the silver-leaved ironbark woodlands (the wet years are coloured shades of green and the dry years are shades of red to orange)

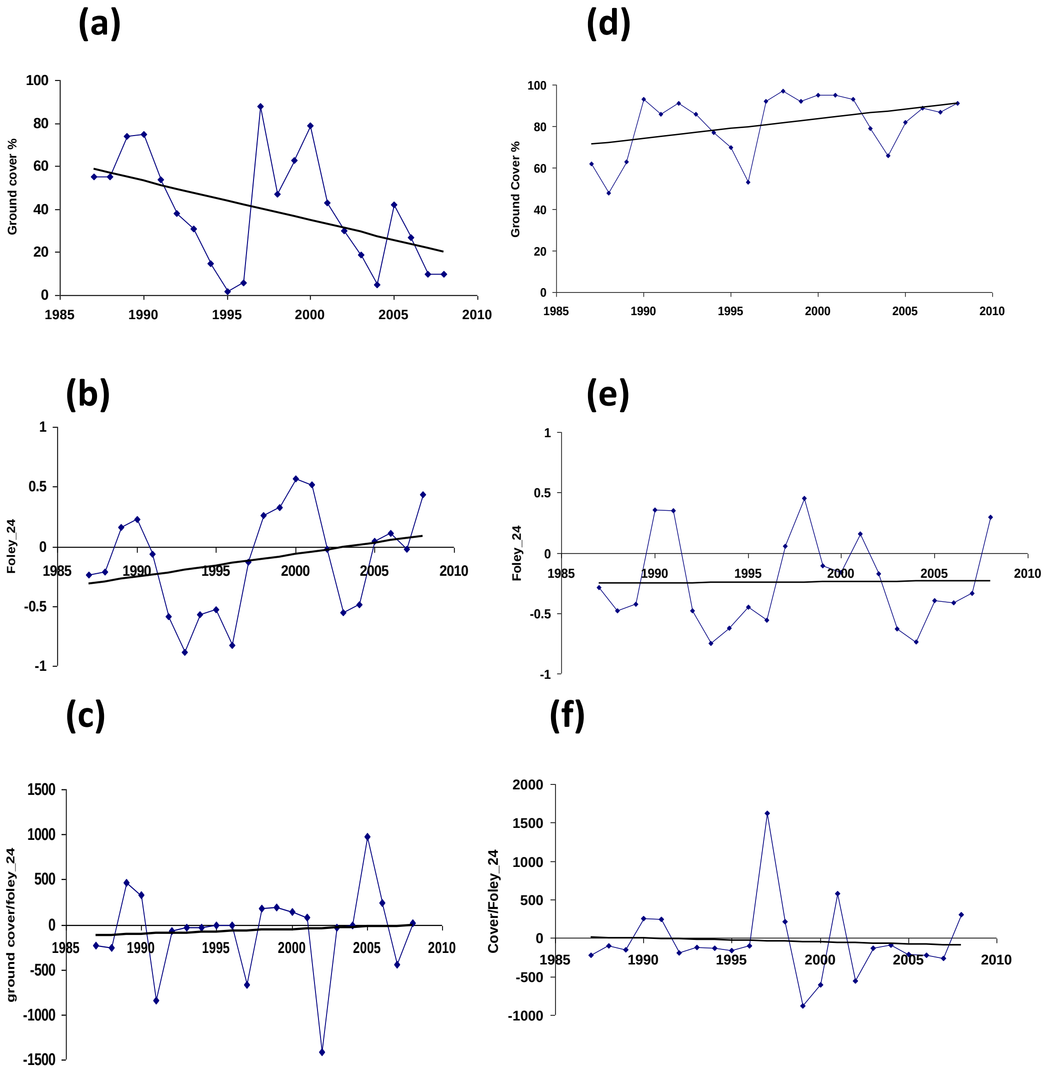

Figure 6.12 Graphs showing two of the 91 sites (a), (b) and (c) pertain to a site with a decreasing ground cover trend and graphs (d), (e) and (f) pertain to a site with increasing ground cover trend. Graphs (a) and (d) show the ground cover trend for both sites over time. Graphs (b) and (d) show the Foley_24 rainfall index trend for each site over time. Graphs (c) and (f) show the cover divided by Foley_24 rainfall index trend over time.

The trends are variable for both ground cover and the Foley_24 rainfall deficit index for the same sites, and, therefore, if the trend of ground cover was corrected to consider the effects of rainfall on ground cover, this would provide a more accurate measurement of ground cover trend alone. Without this consideration, the ground cover of the silver-leaved ironbark woodlands would appear to improving over time, while in reality this is mainly due to increases in annual rainfall from 2007 onwards. The graphs in Figure 6.12 show that the simple but effective application of dividing the cover measurement by the Foley_24 rainfall deficit index correction for rainfall gives a more accurate trend of cover over time.

6.4.4 Hypothesis IV

An assessment of the Ground Cover Disturbance Index for this ecosystem was found to be of limited ecological interpretation due to the lack of consideration of climatic effects. The disturbance matrix is primarily based on mean, the variation in mean cover (standard deviation) and trend and therefore the mean ground cover is over-emphasised in the GCDI. For example, in 2007 the mean cover was 80.05%, and the overall mean ground cover for the ecosystem over the 21 years was 71% (with a minimum mean cover of 51.2% in 1996 and a maximum mean cover of 91% in 1990). When this is related to the research reported in chapter 3; it suggests that at above 65% ground cover the ecosystem retains most of its plant diversity. This long term high mean cover has good ecosystem functionality, as shown by the analysis of plant richness and abundance with ground cover from the field trial data. This revealed four ground cover patterns: the first with 95% ground cover having maximum plant species and therefore near pristine ecosystem condition; the second of between

95% and 65% ground cover had more variability with a range of plant species from 15 to 60; a third with between 65% and 30% ground cover had a range of plant species from 25 to 50; and the last pattern with less than 30% ground cover had less than 20 plant species (see results in Chapter 3).

Incorporating the rainfall index trend and plant diversity relationships with ground cover a simplified ground cover disturbance index matric can be justified as in Table 6.5. This would lead to a revised classification as presented in Figure 6.13.

Table 6.5 Simplified Ground Cover Disturbance Index based on Richards/Green Functionality index and incorporating Foley index

Disturbance Matrix	**Over 70% Ground Cover**	**Between 70% and 30% Ground Cover**	**Under 30% Ground Cover**
Increase Trend(cover/rainfall index)	Very Low	Medium	High
Stable Trend (cover/rainfall index)	Very Low	Medium	Very High
Decrease Trend Cover/rainfall index	Low	High	Very High

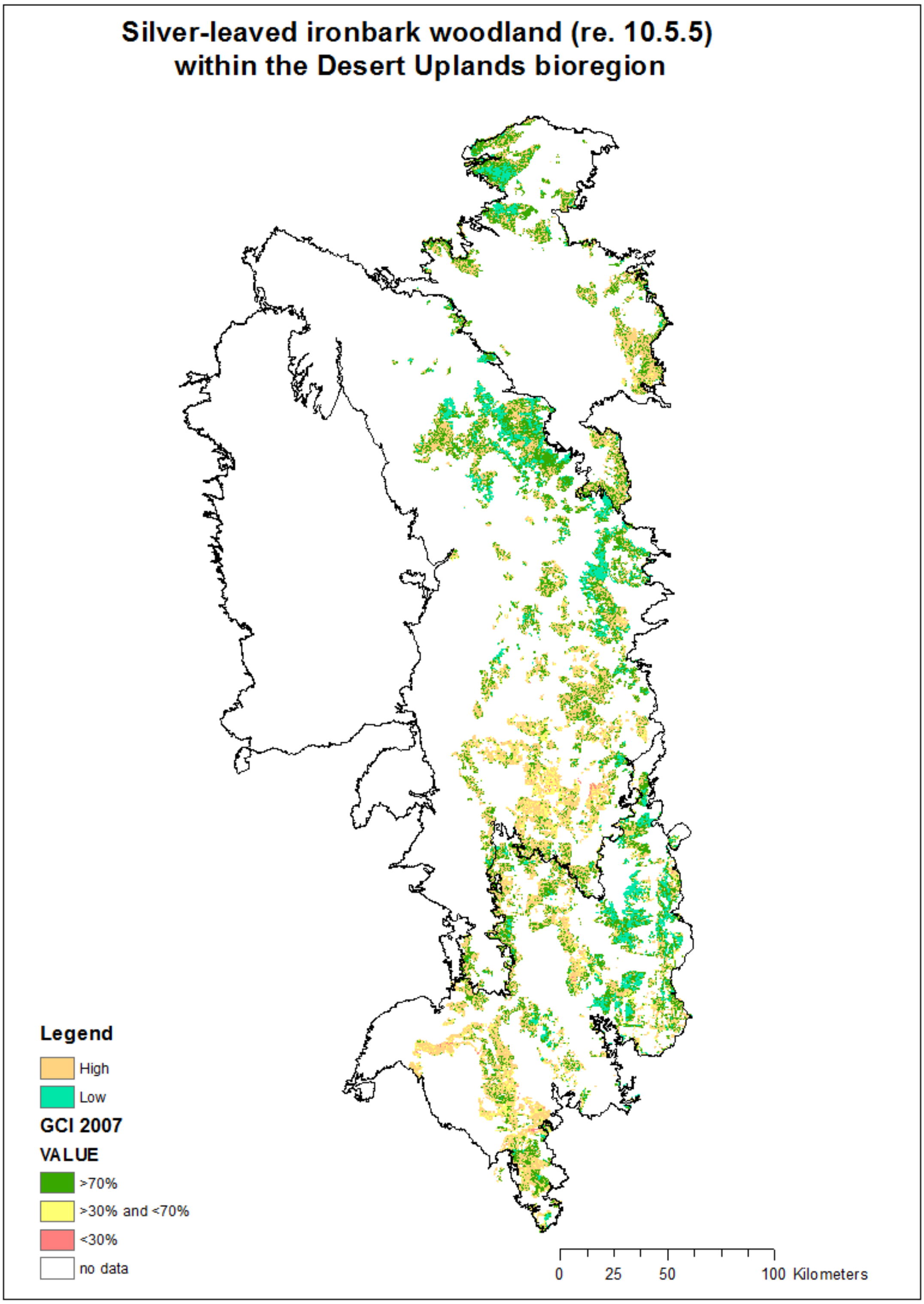

Figure 6.13 Modified Ground Cover Disturbance Index for silver-leaved ironbark woodlands for 2007 taking rainfall index into account

6.5 Discussion and Conclusion

The first key finding is that there is a significant relationship between ground cover measured in the field study and the Ground Cover Index (see Figure 6.9 and Table 6.3). The 2007 Ground Cover Index for the ecosystem across its regional distribution suggests that <4% of the area has <25% ground cover. The average ground cover for the ecosystem in 2007 is 80.05% (see Figure 6.6), which indicates that the ecosystem is in reasonable condition. It is likely that the annual forbs recorded in our study at the end of the wet season would not have been present at the time of the satellite capture during the dry season, which would account for the actual ground cover being higher in May than at the time of the GCI index capture in October. The accuracy of the relationship between ground cover and the Ground Cover Index also can be affected by fire scars, the hiatus between capture and sampling, and variations in leaf litter, tree and shrub cover and soil cover.

Landscape functionality is determined by the spatial arrangement of persistent ground cover in the form of perennial grass species (Ludwig *et al.* 2007). GCI pixel readings can be very dynamic, varying according to the many direct and interacting effects of rainfall, fire, grazing, paddock size, distance to water, proportion of more palatable land types and geographic positions (as evidenced by the upward and downward spikes in the trend in Figure 6.12 (a) and (d)).

The second key finding is that the relationship of ground cover and consequently the GCI is strongly significant with various habitat condition variables such as perennial grass cover, total basal area, over-storey cover, mid-storey cover and cryptogam cover. These are some of the key habitat attributes identified in condition site assessment processes (McIntyre and Hobbs 1999; Parkes *et al*. 2003; Eyre *et al*. 2005; Gibbons and Freudenberger 2006; Kutt *et al.* 2009). Sheffield (2009) found that canopy per cent foliage cover was strongly correlated with both crown cover and canopy health as measured from remote sensed data. She also found that mid-storey cover and pasture cover were strongly correlated with other measures of under-storey cover in a study to determine the correlation between vegetation attributes in the assessment of vegetation condition.

This lends more confidence in the use of the GCI as an indicator of ecosystem condition. Indeed, Bastin *et al.* (2010) reported that there was a distinct spatial patterning of the GCI at Wambiana grazing trials that appears to be related to the grazing management. They also report that there are significant correlations between estimated ground cover and the values of the interim index of landscape function, therefore validating the reliability of the GCI in indicating cover; more so than the performance of the index in indicating landscape function. Intuitively the density of perennial palatable native pastures is a more reliable indicator of landscape function than ground cover, whether estimated directly or derived from remote sensing.

Key results of this study are supported by the conclusions of work by Ward and Kutt (2009) that ground cover temporal mean and variance are potentially useful indicators of disturbance to species diversity and abundance, provided the spatial variability in the climate signal is accounted for.

This study's results support the use of remote sensing data and in particular the GCI as an indicator of biodiversity condition. The strength of remotely sensed data is its ability to provide a spatially continuous sample of the landscape, compared with stand-based assessments of vegetation condition which provide a localised sample within a landscape (Brogaard and Ólafsdóttir 1997; Fassnacht *et al*. 2006; Reinke and Jones 2006). However, while remote sensing provides a synoptic view of extensive areas, the data is limited in terms of spatial and spectral resolution, leading to a coarse measurement of ecosystem condition (Weiers *et al.* 2003).

The third key finding is that using the ABCD land condition classification whereby less than 30% ground cover is 'D' condition, between 30% and 40% is 'C 'condition, between 40% and 70% 'B' condition and over 70% 'A 'condition may be a better way to break up the disturbance levels. The discrepancies between the plant patterns and the ABCD classification can probably be merged to signify a more meaningful classification where over 70% ground cover has high functionality, between 70% and 30% ground cover is functional and under 30% ground cover is less functional. This would be in line with the Richards/Green Functionality Index as proposed by the ACRIS team (Grant *et al.* 2007).

The Ground Cover Disturbance index as presented in Figures 6.1 and 6.2, does not account for climate variability although analysis shows that climate is a key driver of biodiversity condition. A simple but effective method is proposed to account for rainfall. The use of the ground cover index divided by the rainfall deficit index serves to correct for rainfall influences and ensures that a more accurate trend is determined. For simplicity and accuracy only three trend categories are recommended; increase, stable and decrease.

This would result in a simpler, but more meaningful index with which to assess ecosystem condition at the landscape level as shown in Table 6.5 and displayed in Figure 6.13 for 2007.

Chapter Seven

Discussion and Conclusion

7.1 Key Research Aims

One key aim of this study was to determine whether ground cover and a remotely sensed index of ground cover can be an effective indicator of ecosystem condition at a landscape scale. A second aim is to determine if modelling of a two decadal dataset of ground cover indices and grazing variables can distinguish the effects of grazing from rainfall effects on ground cover and provide a tool with which to predict the outcomes of changes in the combinations of both rainfall and grazing variables.

7.2 Key Research Findings – knowledge contribution

7.2.1 Overall finding

A remotely-sensed ground cover measurement does have a significant relationship to actual ground cover, plant diversity and some bird species measurements and therefore has potential as an indicator of ecosystem condition for the silver-leaved ironbark woodlands in the Desert Uplands bioregion of Queensland.

7.2.2 Plants

Twenty-two plants were found to have significant positive relationships with ground cover and two plants were found to have significant negative relationships with ground cover. Both plant richness and abundance were found to be significantly positively related to ground cover. Perennial native grasses are suitable indicators of ecosystem condition across both wet and dry seasons. From the assessment of plant species patterns across four ranges of ground cover, it was found that there was a significant reduction in the richness and abundance of plant species in areas with less than 30% ground cover, an indication that a threshold of functionality had been crossed.

7.2.3 Birds

The relationships of bird species and bird groups (dietary, foraging and habitat assemblage) with ground cover were explored in depth. Significant positive and negative species' relationships to ground cover were established. Birds with significant positive relationships to ground cover include: Rufous whistler, red-browed pardalote, grey shrike thrush, crested bellbird and grey fantail. Bird groups with significant positive relationships with ground cover include: the insectivorous dietary group, the above canopy and canopy feeders from the foraging group and the habitat assemblage of birds that nest on the ground and feed in trees.

Through exploration using constrained ordination and pattern analysis, the above bird groups were found to have strong association with environmental variables of ecosystems in good condition such as live basal area, litter cover, over-storey cover, mid-storey cover, and perennial grass cover. The Simpsons index of diversity was significantly higher with high ground cover.

7.2.4 Modelling

The application of linear mixed effect model to data derived from 21 years of remote sensed data was demonstrated to have the ability to predict actual ground cover. Results of the models demonstrate a measurable grazing effect can be detected at the landscape level. The model provides a method to distinguish the effects of grazing management variables from the key climate variable; the 24 month rainfall index (Foley_24).   The most significant grazing variables that affect ground cover are the area of more palatable land type (with a negative effect) within the paddock and distance to water (with a positive effect). A strong model fit was achieved and model validation quantified the predictive power of three models. This model provides a tool with which to predict the outcomes of changes in the combinations of those grazing variables and rainfall and to adequately assess the impacts of grazing enterprises on woodland's biodiversity condition.

7.2 5 Ground cover

Remotely sensed ground cover measurements have significant relationships to actual ground cover measurements, biomass, perennial grass cover and other environmental variables such live basal area, over-storey cover, mid-storey cover and cryptogam cover. Ground cover was also found to have significant relationships to grazing indicators such as the log of distance to water and density of waters.

Significant variation in ground cover was found between wet and dry years in GCI measurements from the whole ecosystem.

Based on the modelling results of this study, dividing ground cover by the Foley_24 rainfall index over time allows for the ground cover trend to be corrected for rainfall. The high coefficient values in the linear mixed effect models for the Foley rainfall deficit index means that any assessment of ground cover using a remote sensing index needs to take rainfall into account. This measurement is suggested as a more reliable measure of cover trend over time.

A simplification of the Ground Cover Disturbance index, an index that relies on GCI mean, trend and variation to determine biodiversity condition at the landscape scale criteria in Bioregional Planning Assessments (EPA 2002), is recommended so it accounts for rainfall effects.

7.3 Contributions to research into biodiversity condition assessment in the rangelands

In this study, a threshold of thirty per cent ground cover correlates with significant loss of plant species diversity and, therefore ecosystem functionality to perform water infiltration and nutrient cycling is impaired by this loss of cover. Therefore, it is recommended that this threshold is incorporated into a more meaningful biodiversity condition index based on the Richards/Green Functionality Index that incorporates the plant species patterns established in this study into the matrix for disturbance.

Perennial grass cover was found to have a significantly strong relationship with ground cover and, as such, the use of the richness and abundance of these native perennial grasses' relationship to ground cover are re-enforced as reliable indicators of ecosystem condition.

The investigation of bird species abundance, especially resident insectivorous species relationships with ground cover correlated significantly with the native perennial grass species relationships with ground cover.

The use of linear mixed effects models to distinguish the effects of grazing from those of rainfall on ground cover is a novel approach and was successful. The fact that a grazing effect can be detected and quantified at the landscape level, through a modelling approach, is new knowledge. The models can now be applied to case studies to examine the effects of changes in actual dimensions on ground cover under different climate scenarios. The models give land holders, resource managers and policy makers an effective tool with which to understand and control the variables of grazing management.

The ground cover index measurements used in this study are correlated to pasture basal area and ground cover and was found to be significantly affected by grazing variables such as area of more palatable land type and distance to water. This result shows that the equilibrium versus non-equilibrium argument is complicated by cumulative effects of grazing affecting an ecosystem's ability to respond to rainfall events.

7.4 Management Recommendations

If the grazing management in the silver-leaved ironbark were to intensify with more waters, shorter distances to water and more cattle, the ecosystem condition of the these woodlands would deteriorate. The configuration of a paddock and the placement of artificial water within it should be carefully considered because the area of more palatable land type within the paddock plays such a significant direct and inter-active effect with rainfall. Careful placement of infrastructure could lead to more even grazing pressure across land types throughout the paddock. Paddocks separating land types would have a beneficial effect. This becomes increasingly important in wet periods, when preferential grazing of more palatable land types increases; resulting in long term negative effects on ground cover perennial grass composition.

7.5 Further Research Opportunities

As the field assessment of this study was limited in nature, one year and two seasons, and only examined the relationship of the plant and bird biodiversity elements, these methods and results need to be tested in other homogenous ecosystems with low projective foliage cover, to generate more confidence in these findings. The relationship of the ground cover index to biodiversity condition should be examined in other large homogenous ecosystems, such as exist in the Mitchell Grass Downs bioregion. Additionally, the examination of the relationship of invertebrates, reptiles and mammals to ground cover could support the use of ground cover as a biodiversity condition surrogate. This would establish the usefulness of the ground cover index for wider application.

Rangeland research has established what the threshold parameters are when an ecosystem has changed (in terms of its structural and functional traits) to an irreversible, dysfunctional state. The focus now needs to be directed to examination of the spatial and temporal parameters of these threshold changes at the landscape scale. For example, for how long and how extensive does grazing pressure need to occur on a vast ecosystem to change its state to a new system?

In terms of bird diversity, two year, or bi-annual, bird surveys in both the wet and dry seasons once every five years, with a larger data set would strengthen the relationships found in this study. Ideally, the location of low ground cover sites associated with supplement feeding away from water would make the results clearer. The use of projective foliage cover (remote-sensed data on tree cover) should be investigated for its relationship with bird species and therefore its potential as surrogate for bird diversity.

7.6 Ecosystem condition improvement incentive mechanisms

The relationships identified in this thesis provide an assessment mechanism that could be used to underpin incentive mechanisms for landholders. Stewardship payments to landholders as incentive payments to improve and maintain ecosystem condition by way of retaining ground cover through the dry season could be considered through modelling the different grazing parameters in light of rainfall from the preceding two years. The payment for the cost of provision is likely to be higher on the back of dry seasons than on the back of wet seasons, while the opportunity to deliver will be reduced. However, the best opportunity for investment is for incentive schemes to commence on the back of wet seasons, when participants can put mechanisms in place to deliver improvements in ecosystem condition.

7.7 Policy Implications

The availability of an effective remote-sensing tool for the measurement and assessment of ecosystem condition means that a consistent, repeatable and time efficient method is available to project, assess, measure and monitor the effectiveness of different policies and strategies of intervention from extension or regulation through to incentive programs. A model based on remotely-sensed ground cover could be an effective tool for government to ensure that the ecosystem functionality of these woodlands is maintained so that they continue to support a range of bird species (that have declined across their southern distribution) and to maintain ecologically sustainable grazing enterprises.

The model could assist landholders to predict the effects of their grazing management options on ecosystem condition by factoring in the rainfall received over the preceding two years. The management options that the model would be most effective at evaluating are the placement of artificial waters with respect to paddock size and paddock configuration with respect to land types. The model could give landholders an effective method to determine what changes to grazing carrying capacity would be necessary to maintain a certain ground cover threshold. A safe target for landholders to ensure the long-term maintenance of the woodlands ecological condition would be for a minimum of 40% ground cover across at least 40% of their ecosystem distribution in a connected manner across the landscape.

The government could use the model to test and monitor the effectiveness of policy interventions that could be used to ensure the maintenance of ecological condition in the rangelands. Policy makers could use the model to predict the positive or negative impacts of policy intervention on ecosystem condition. By using the model to separate out the effects of grazing from rainfall, policy makers could determine an accurate actual costing of maintaining a threshold of a minimum of 40% ground cover across the spatial distribution of the woodlands for any given period for the forthcoming year from the end of the wet season until the beginning of the next wet season.

In this study, the woodlands have been found to support biodiversity values that have been lost from woodlands in southern Australia. A ground cover index-based model that incorporates rainfall and grazing variables would be a valuable tool for all stakeholders to ensure that grazing pressure on this ecosystem does not undermine the ecosystem condition of these woodlands.

References

Ahern, C.R., Shields, P.G. Enderlin, N.G. & Baker, D.E. (1994). Soil fertility of central and northeast Queensland grazing lands. Information series publication No. Q194065, Queensland Department of Primary Industries, Brisbane.

Anderson, E. (1993). 'Plants of Central Queensland – their Identification and Uses.'(Department of Primary Industries: Brisbane.)

Andrews, M.H. (1988). Grazing impact in relation to livestock water points. *Trends in Ecology and Evolution* **3**: 336-339

Andrew, M.H. & Lange, R.T. (1986) Development of a new piosphere in arid chenopod shrubland grazed by sheep. 1. Changes to the vegetation. *Australian Journal of Ecology* **11** (4): 411-424

Angermeier, P.L. (2000). The natural imperative for biological conservation. *Conservation Biology* **14** (2): 373-381

Antrop, M. (2005). Why landscapes of the past are important for the future? *Landscape Urban Planning* **70**: 21–34

Archer, E. R. M. (2004). Beyond the "climate versus grazing" impasse: Using remote sensing to investigate the effects of grazing system choice on vegetation cover in the eastern Karoo. *Journal of Arid Environments* **57:** 381−408.

Ash, A. & Corfield, J.P. (1998). Influence of pasture condition on plant selection patterns by cattle: its implications for vegetation change in a monsoon tallgrass rangeland. *Tropical Grasslands* **32**: 178-187

Ash, A., Corfield, J., & Ksiksi, T. (2001). The Ecograze Project – developing guidelines to better manage grazing country. CSIRO, Townsville.

Ash, A. & McIvor, J.G. (1998). How season of grazing and herbivore selectivity influence monsoon tall-grass communities of northern Australia*. Journal of Vegetation Science* **9**: 123-132

Ash, A.J., McIvor, J.G. & Brown, J.R. (1993). Land condition and overgrazing: a management paradox for the savannas of northern Australia. Proceedings of the 17th International Grassland Congress, Palmerston North, Hamilton, Lincoln and Rockhampton

Ash, A. J., McIvor, J. G., Corfield, J. P., & Winter, W. H. (1995). How land condition alters plant-animal relationships in Australia's tropical rangelands. *Agriculture, Ecosystems & Environment* **56**: 77–92

Ash, A.J. & Schlink, A.C. (1992). Comparative digestive efficiency of cattle and sheep consuming tropical forages and its significance in predicting digestibility from in vitro techniques. *Proceedings from Australian Society of Animal Production* **19:** 331 -334.

Ash, A.J. & Stafford Smith, D.M. (1996). Evaluating stocking rate impacts: stock don't practice what we preach. *Rangelands Journal* **18** (2): 216-234

Augsperger, C.K. (1984). Seedling survival of tropical tree species: interaction of dispersal, distance, light-gap and pathogens. *Ecology* **65**: 1705-1712

Bardgett, R.D. (2005). 'The Biology of Soil: a community and ecosystem approach'. Oxford University Press: Oxford, U.K.

Barnard, C.A., & Barnard, H.G. (1925) A review of the bird life on Coomooboolaroo Station, Duaringa district, Queensland, during the past fifty years. *Emu* **24:** 252-65.

Bastin,G., & the ACRIS Committee(2008). 'Rangelands 2008 – Taking the Pulse.' (National Land and Water Resources Audit: Canberra).

Bastin, G. N., Pickup, G., Stanes, J., & Stanes, A. (1996). Estimating landscape resilience from satellite data and its application to pastoral land management. *The Rangeland Journal* **18:** 118–135.

Bastin, G., Scarth, P., Chewings, V., Sparrow, A., Denham, R., Schmidt, M., O'Reagain, P., Shepherd, R. & Abbott, B. (2012). Separating grazing and rainfall effects at a regional scale using remote sensing imagery: a dynamic reference-cover method. *Remote Sensing of Environment* **121**: 443-457

Bastin, G., Schmidt, M., O'Reagain, P. & Karfs, R. (2010). Reporting change in landscape function using the Queensland ground cover index. In: *Proceedings of the 16th Biennial Conference of the Australian Rangeland Society,* Bourke (Eds D.J. Eldridge & C. Waters). Australian Rangeland Society: Perth.

Bastin, G. N., Stafford Smith, D. M., Watson, I. C., and Fisher, A. (2009). The Australian Collaborative Rangelands Information System: preparing for a climate of change. *The Rangeland Journal* **31:** 111–125.

Bastin, G.N., Tynan, R.W., & Chewings, V.H. (1998). Ecological consequences of altered hydrological regimes in fragmented ecosystems in southern Australia: impacts and possible management responses. *Rangelands Journal* **20:** 61-76.

Beeton R.J.S. , Buckley K. I., Jones G. J., Morgan D., Reichelt R. E., & Trewin D. (Australian State of the Environment Committee), (2006). Australia State of the Environment 2001. Independent report to the Australian Government Minister for the Environment and Heritage.

Begon, M., Harper, J.L. & Townsend, C.R. (1986). Ecology: Individuals, Populations and Communities. Blackwell, Oxford.

Bestelmeyer, B.T., Brown, J.R., Havstad, K.M., Alexander, R., Chavez, G., & Herrick, J.E. (2003). Development and use of state-and-transition models for rangelands. *Journal of Rangeland Management* **56:**, 114–126.

Bestelmeyer, B.T., Goolsby, D.P. & Archer, S.R. (2011). Spatial perspectives in state and transition models: a missing link to land management? *Journal of Applied Ecology* **48**: 746-757

Biograze (2000). Biograze Technical Fact Sheet No. 5: Environmental Management Systems and biodiversity. CSIRO, Alice Springs.

Block, W.M., Brennan, L.A. & Gutierrez, R.J. (1987). Evaluation of guild-indicator species for the use of resource management. *Environmental Management* **11**: 265-9.

Bock, C.E. & Webb, B. (1984) Birds as grazing indicator species in southeastern Arizona. *Journal of Wildlife Management* **48**: 1045-9

Bradford, D.F., Franson, S.E., Neale, A.C., Heggem, D.T., Miller, G.R. & Canterbury, G.E. (1998). Bird species assemblages as indicators of biological integrity in Great Basin rangeland. *Environmental Monitoring and Assessment* **49**: 1-22

Briske, D.D., Fuhlendorf, S.D. & Smeins, F.E. (2005). State and Transition Models, Thresholds, and Rangeland Health: A synthesis of ecological concepts and perspectives. *Rangeland Ecological Management* **58**: 1-10.

Briske, D.D., Fuhlendorf, S.D. & Smeins, F.E. (2003). Vegetation dynamics on rangelands: a critique of the current paradigms. *Journal of Applied Ecology* **40** (4): 601-614

Bromhan, L., Cardillo, M., Bennett, A.F. & Elgar, M.A. (1999). Effects of stock grazing on the ground invertebrate fauna of woodland remnants. *Australian Journal of Ecology* **24** (3): 199-207

Briggs, S.V. & Freudenberger, D. (2006). Assessment and monitoring of vegetation condition: Moving forward. *Ecological Management and Restoration* **7** (1): S74-S75.

Briske, D.D., Fuhlendorf, S.D. & Smeins, F.E. (2003). Vegetation dynamics on rangelands: a critique of the current paradigms. *Journal of Applied Ecology* **40**: 601-614

Brogaard, S., & Ólafsdóttir, R. (1997). Ground-truths or Ground-lies? Environmental sampling for remote sensing application exemplified by vegetation cover data. Department of Physical Geography, Lund University, Sweden. *Available at* http://www.nateko.lu.se/Elibrary/LeRPG/1/LeRPG1Article.pdf

Bromhan, L., Cardillo, M. Bennett, A.F. & Elgar, M.A. (1999) Effects of stock grazing on the ground invertebrate fauna of woodland remnants. *Australian Journal of Ecology* **24** (3): 199-207

Brown, J.R. & Archer, S. (1989). Woody plant invasion of grasslands: establishment of honey mesquite (*Prosopis glandulosa var glandulosa*) on sites differing in herbaceous biomass and grazing history. *Oecologia* **80** (1): 19-26

Brown, J.H., Whitham, T.G., Morgan Ernest, S.K. & Gehring, C.A. (2001). Complex species interactions and the dynamics of ecological systems: long-term experiments. *Science* **293**: 643-650

Bryant, N.A., Johnson, L.F., Brazel, A.J., Balling, R.C., Hutchinson, C.F. & Beck, L.R. (1990). Measuring the effect of overgrazing in the Sonoran Desert. *Climatic Change* **17:** 243–264

Bullock, J.M. (1996). Plant competition and population dynamics. In: Hodgson, J. & Illius, A.W. (eds.) The Ecology and Management of Grazing systems. CAB International New York, p. 69-100

Canterbury, G.E., Martin, T.E., Petit, L.J. & Bradford, D.F. (2000). Bird communities and habitat as ecological indicators of forest condition in regional monitoring. *Conservation Biology*.**14**: 1179-96.

Cardoso, P.G., Bankovic, M., Raffaelli, D. & Pardal, M.A. (2007). Polychaete assemblages as indicators of habitat recovery in a temperate estuary under eutrophication. *Estuarine, Coastal and Shelf Science* **71**: 301-8

Carignan, V. & Villard, M.A. (2002). Selecting indicator species to monitor ecological integrity: a review. *Environmental Monitoring and Assessment* **78**: 45-81

Carter, J., Silcock, R., Bastin, G. & Schliebs, M. (2007). *Australian Collaborative Rangeland Information System, Reporting Change in the Rangelands – 2007: Queensland Information for the National Report*. Queensland Department of Natural Resources and Water, Brisbane, Australia.

Caughley, G. (1987). Kangaroos, their ecology and management in the sheep rangelands of Australia. Cambridge University Press, England.

Chambers, S.A. (2008). Birds as Environmental Indicators: Review of literature. Parks Victoria Technical Series. No.55. Parks Victoria, Melbourne.

Chapin, F.S., III., Torn, M. S.& Tateno, M. (1996). Principles of ecosystem sustainability. *American Naturalist* **148**: 1016-1037

Chase, J.M. (2003). Community assembly: when should history matter? *Oecologia* **136**: 489-498

Chase, M.K., Kristan, W.B., Lynam, A.J., Price, M.V. & Rotenberry, J.T. (2000). Single species as indicators of species richness and composition in California coastal sage scrub birds and small mammals. *Conservation Biology* **14**: 474-87

Chesson, P.L. & Case, T.J. (1986). Overview: Non-equilibrium community theories: chance, variability, history, and coexistence. Community Ecology (eds. J. Diamond & T.J. Case) pp. 229-239. Harper & Row Publisher, N.Y.

Chilcott, C.R., McCallum, B.S., Quirk, M.F., & Paton, C.J. (2003). Grazing Land Management Education Package Workshop Notes – Burdekin. Meat & Livestock Australia Limited, Sydney.

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

Christensen, P.E. & Kimber, P.C. (1975). Effect of prescribed burning on the flora and fauna of south-west Australian forests. *Proceedings of the Ecological Society of Australia* **9**: 85-106.

Christidis, L., & Boles, W.E., (2008). Systematics and Taxonomy of Australian Birds. CSIRO, Collingwood, Victoria.

Clarke, K.R. & Warwick, R.M. (2001) Change in marine communities: an approach to statistical analysis and interpretation. Second ed. PRIMER – E, Plymouth, UK.

Clarke, P.J. (2003). Composition of grazed and cleared temperate grassy woodlands in eastern Australia: patterns in space and inferences in time. *Journal of Vegetation Science*, **14,** 5-14

Clements, F.E. (1928). Plant succession and indicators: a definitive edition of plant succession and plant indicators. Hafner Press, New York.

Cole, L. & Bardgett, R.D. (2002). Soil animals, microbial activity and nutrient cycling. In '*Encyclopedia of Soil Science'*. Ed. R.Lal pp. 72-75. Marcel Dekker Inc: New York).

Cook, B.I., Miller, R.L. & Seager, R. (2009). Amplification of the North American ''Dust Bowl'' drought through human-induced land degradation. Proceedings of the National Academy of Sciences, USA, **106,** 4997–5001.

Coughenour, M. B. (1991). Invited synthesis paper: Spatial components of plant-herbivore interactions in pastoral, ranching and native ungulate ecosystems. *Journal of Range Management* **44:** 530-542.

Cridland, S. & Stafford Smith, M.D. (1993). *Development and dissemination of design methods for rangeland paddocks which maximise animal production and minimise land degradation.* Department of Agriculture Western Australia, Perth.

CSIRO (2010). Climate Change in Australia – Technical report.

Daily, G.C. (Ed.) (1997). Nature's services: societal dependence on actual ecosystem. Washington D.C. Island Press.

Davies, K.F., Melbourne, B.A., James, C.D., & Cunningham, R.B. (2010). Using traits of species to understand responses to land use change: Using birds and livestock grazing in the Australian arid zone. *Biological Conservation* **143**: 78-85

DeAngelis, D. L. & Waterhouse, J. C. (1987). Equilibrium and nonequilibrium concepts in ecological models. *Ecological Monographs.* **57:** 1–21.

de Bello, F., Lavorel, S., Gerhold, P., Reier, Ü., & Pärtel, M. (2010). A biodiversity monitoring framework for practical conservation of grasslands and shrublands. *Biological Conservation* **143:** 9–17.

Department of Environment and Resource Management. (2011). *Biodiversity Planning Assessment, Desert Uplands Bioregion Landscape Expert Panel Report,* Central West Region: Department of Environment and Resource Management, Queensland Government.

Department of Natural Resources and Water (2007). Delbessie Agreement. State Rural Leasehold Land Strategy. Available at: www.nrw.qld.gov.au (accessed 9 March 2009).

Diouf, A. & Lambin, E. F. (2001). Monitoring land-cover changes in semi-arid regions: Remote sensing data and field observations in the Ferlo, Senegal. *Journal of Arid Environments.* **48:** 129−148.

Dube, S., Kalua, F. & Mkungurutse (1999) The importance of ground-cover in reducing erosion and run-off in a semi-arid rangeland. Sixth International Rangeland Congress Proceedings. Townsville. VI International Rangeland Congress, 706-707.

Dudley, D.R. & Karr, J.R. (1981). Ecological perspective on water quality goals. *Environmental Management* **5** (1): 15-68

Dufrene, M. & Legendre, P. (1997). Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs* **67**: 345-366

Dyksterhuis, E.J. (1958). Ecological principles in range evaluation*. Botanical Review* **24**: 253-272

Eldridge, D.J. & Koen, T.B. (1993). Run-off and sediment yield from a semi-arid woodland in eastern Australia. II. Variation in some soil hydrological properties along a gradient in soil surface condition. *Rangeland Journal* **15** (2): 234-246

Elith, J. & Leathwick, J.R. (2009). Species distribution models: ecological explanation and prediction across space and time. *Annual Review of Ecology Evolution and Systematics* **40**: 677-697

Ellis, J.E. & Swift, D.M. (1988). Stability of African pastoral ecosystems: alternative paradigms and implications for development. *Journal of Range Management* **41**: 450-459

Enquist, B.J. & Niklaus, K.J. (2001). Invariant scaling relations across tree-dominated communities. *Nature* **410**: 655-660

Environmental Protection Agency (2002). Biodiversity Assessment and Mapping Methodology. Biodiversity Planning Unit. Brisbane.

Evans, J. & Geerken, R. (2004). Discrimination between climate and human-induced dryland degradation. *Journal of Arid Environments* **57:** 535−554.

Eyre, T.J., Fisher, A., Hunt, L.P. & Kutt, A.S. (2011). Measure it to better manage it: a biodiversity monitoring framework for the Australian rangelands. *Rangeland Journal* **33:** 239-253.

Eyre, T.J., Kelly, A.L. & Neldner, V.J. (2006). BioCondition: A terrestrial vegetation condition assessment tool for Biodiversity in Queensland. Field Assessment Manual. Version 1.5. Environmental Protection Agency, Biodiversity Sciences Unit, Brisbane.

Eyre, T.J., Kelly, A., Neldner, V.J., McCosker, J. & Kutt, A. (2005*) BioCondition: A Terrestrial Vegetation Condition Assessment Tool for Biodiversity in Queensland,* Version 1.3. Environmental Protection Agency, Biodiversity Sciences Unit, Brisbane.

Fairfax, R.J. & Fensham, R.J. (2000). The effect of exotic pasture development on floristic diversity in central Queensland, Australia. *Biological Conservation* **94**: 11-21.

Fassnacht, K. S., Cohen, W. B. & Spies, T. A. (2006). Key issues in making and using satellite-based maps in ecology: A primer. *Forest Ecology and Management* **222**: 167-181.

Fensham, R.J & Fairfax, R.J. (2002). Aerial photography for assessing vegetation change: a review of applications and the relevance of findings for Australian vegetation history. *Australian Journal of Botany* **50**(4): 415-429

Fensham, R.J. & Fairfax, R.J. (2008). Water-remoteness for grazing relief in Australian arid-lands. *Conservation Biology* **141:** 1447-1460

Fensham, R. J., Fairfax, R.J., Butler, D.W. & Bowman, D.M.J.S. (2003). Effects of fire and drought on a tropical eucalypt savanna colonised by rainforest. *Journal of Biogeography* **30**: 1405-1414

Fensham, R.J., Fairfax, R.J. & Ward, D.P. (2009). Drought-induced tree death in savanna. *Global Change Biology* **15**: 380-387

Fensham, R. J. & Holman, J.E. (1999). Temporal and spatial patterns in drought-related tree dieback in Australian savanna. *Journal of Applied Ecology* **36**: 1035-1050

Fensham, R. J. & Skull, S. D. (1999). Before cattle: a comparative floristic study of Eucalyptus savanna grazed by macropods and cattle in North Queensland, Australia. *Biotropica* **31**: 37–47

Fernandez-Gimenez, M.E. & Allen-Diaz, B. (1999). Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. *Journal of Applied Ecology* **36**, 871–885.

Ferraro, P.J. (2011). The future of payments for environmental services*. Conservation Biology* **25** (6): 1134-1138

Fisher, A. & Kutt, A. (2006). *Biodiversity and land condition in tropical savannah rangelands: summary report*. Tropical Savannas CRC, Darwin.

Fisher, A. & Kutt, A. (2007).Biodiversity and land condition in tropical savannah rangelands: technical report. Tropical Savannas CRC, Darwin.

Fisher, A., Hunt, L., James, C., Landsberg, J., Phelps, D., Smyth, A. & Watson, I. (2004) Review of total grazing pressure management issues and priorities for biodiversity conservation in rangelands: A resource to aid NRM planning. Desert Knowledge CRC Project Report No. 3; Desert Knowledge CRC and Tropical Savannas Management CRC, Alice Springs.

Flannery, T.F. (1994). The Future Eaters. An ecological history of the Australasian lands and people. Reed, Chatswood.

Fleishman, E., Thomson, J.R., MacNally, R., Murphy, D.D. & Fay, J.P. (2005). Using indicator species to predict species richness of multiple taxonomic groups. *Conservation Biology* **19**: 1125-37

Foley, J.C. (1957). *Droughts in Australia. Review of records from earliest years of settlement to 1955.* Bulletin no. 47. Bureau of Meteorology, Commonwealth of Australia, Melbourne, Australia.

Foran, B.D., Bastin, G. & Shaw, K.A. (1986). Range assessment and monitoring in arid lands: the use of classification and ordination in range survey. *Journal of Environmental Management* **22**: 67-84

Ford, H.A. (1985). A synthesis of the foraging ecology and behaviour of birds in eucalypt forests and woodlands. In *Birds of Eucalypt Forests and Woodlands: Ecology, Conservation, Management* (Eds A. Keast, H.F. Recher, H. Ford and D. Saunders) pp. 249-254 (Surrey Beatty: Chipping Norton).

Ford, H.A. (1985). The bird community in eucalypt woodland and eucalypt dieback in the Northern Tablelands of New South Wales. In *Birds of Eucalypt Forests and Woodlands: Ecology, Conservation, Management* (Eds A. Keast, H.F. Recher, H. Ford and D. Saunders) pp. 333-340. (Surrey Beatty: Chipping Norton).

Friedel, M.H. (1997). Discontinuous change in arid woodland and grassland vegetation along a gradient of cattle grazing in Central Australia. *Journal of Arid Environments* **37**: 145-164

Friedel, M.H. (1990). Some key concepts for monitoring Australia's arid and semi-arid rangelands. *Australian Rangeland Journal* **12**: 21-24

Friedel M.H. (1991). Range condition assessment and the concept of thresholds: a viewpoint. *Journal of Range Management* **44**: 422-426.

Friedel, M.H. (1994). How spatial and temporal scale affect the perception of change in rangelands. *Rangeland Journal* **16:** 16-25

Friedel, M.H., Laycock, W.A., & Bastin, G. N. (2000). Assessing rangeland condition and trend. *In*: 'Field and laboratory methods for grassland and animal production research'. (Eds L. T. Mannetje and R. M. Jones.) pp. 227–262. (Commonwealth Agricultural Bureaux Publication: Wallingford, UK.)

Friedel, M.H., Pickup, G. & Nelson, D.J. (1993). The interpretation of vegetation change in a spatially and temporally diverse arid Australian landscape. *Journal of Arid Environments* **24**: 241-260

Friedel, M.H. & Shaw, K. (1987). Evaluation of methods for monitoring sparse patterned vegetation in arid rangelands. I. Herbage. *Journal of Environmental Management* **25**: 297-308

Frolking, S., Palace, M.W. Clark, D.B., Chamber, J.Q., Shugart, H.H. & Hurtt, G.C. (2009). Forest disturbance and recovery: a general review in the context of space-borne remote sensing of impacts on aboveground biomass and canopy structure. *Journal of Geophysical Research – Biogeosciences* **114**: GOOE02

Frost, P., Medina, E., Menaut, J.C., Solbing, D., Swift, M. & Walker, B. (1986). Responses to stress and disturbance: A proposal for collaborative programme of research. *Biology International* **10:** 1-81

Fuhlendorf, S.D., Briske, D.D. & Smeins, F.E. (2001). Herbaceous vegetation change in variable rangeland environments: the relative contribution of grazing and climatic variability. *Applied Vegetation Science* **4**: 177-188

Furness, R.W., Greenwood, J.J.D. & Jarvis, P.J. (1993). Can birds be used to monitor the environment? In: Birds as Monitors of Environmental Change (Eds R.W. Furness & J.J.D. Greenwood) pp. 1-41. Chapman & Hall, London.

Gale, S.J. & Haworth, R.J. (2005). Catchment-wide soil loss from pre-agricultural times to the present: transport and supply limitation of erosion. *Geomorphology* **68:** 314-333

Garnett, S.T. & Crowley, G.M. (2000). The Action Plan for Australian Birds 2000. Environment Australia, Canberra.

Gibbons, P., Ayers, D., Seddon, J., Doyle, S., & Briggs, S. (2005). BioMetric Version 1.8: A Terrestrial Biodiversity Assessment Tool for the NSW Property Vegetation Plan Developer Operational Manual. Department of Environment and Conservation (NSW), Canberra.

Gibbons, P. & Freudenberger, D. (2006). An overview of methods to assess vegetation condition at the scale of the site. *Ecological Management and Restoration* **7**(1): S10- S17.

Gill, A.M. & Williams, J.E. (1996). Fire regimes and biodiversity: the effects of fragmentation of south eastern Australian eucalypt forests by urbanisation, agriculture, and pine plantation. *Forestry Ecological Management* **85**: 261-278.

Gillson, L. & Hoffman, M.T. (2007) Rangeland ecology in a changing world. *Science*, **315**, 53–54.

Grant, R., Schliebs, M., & Bastin, G. (2007). Australian Collaborative Rangeland Information System, reporting change in the rangelands – 2007. New South Wales Information for the National report.

Gregory, R.D., Van Strien, A., Vorisek, P., Meyling, A.W.G., Noble, D.G., Foppen, R.P.B. & Gibbons, D.W. (2005). Developing indicators for European birds. *Philosophical Transactions of the Royal Society B*. **360**: 269-88

Grigera, G., Oesterheld, M., & Pacin, F. (2007) Monitoring forage production for farmers' decision making, *Agricultural Systems*. **94**: 637-648

Grime, J.P. (1973). Competitive exclusion in herbaceous vegetation. *Nature* **242**: 344-347

Hacker, R. B., & Tunbridge, S. B. (1991). Grazing management strategies for reseeded rangelands in the east Kimberley region of Western Australia. *The Rangeland Journal* **13**: 14–35

Haila, Y. (1997). A "Natural" benchmark for ecosystem function. *Conservation Biology* **11**(2):300-307

Hanan, N.P. & Lehmann, C.E.R. (2011). Tree-grass interactions in savannas: Paradigm, contradiction and conceptual models. In M. J. Hill & N.P. Hanan (eds.) *Ecosystem function in savannas. Measurement and modelling at landscape to global scales* (pp. 39-53) Boca Raton, Florida CRC Press.

Hannah, D., Woinarski, J.C.Z., Catterall, C.P., McCosker, J.C., Thurgate, N.Y. & R.J. Fensham (2007). Impacts of clearing, fragmentation and disturbance on the bird fauna of Eucalypt savanna woodland in Central Queensland, Australia. *Austral Ecology* **32**: 261-276

Hassett, R.C., Wood, H.L., Carter, J.O., & Danaher, T.J. (2000). A field method for state-wide ground-truthing of a spatial pasture growth model. *Australian Journal of Experimental Agriculture* **40** (8): 1069-1079

Hein, L., de Ridder, N., Hiernaux, P., Leemans, R., de Wit, A., & Schaepman, M. (2011). Desertification in the Sahel: Towards better accounting for ecosystem dynamics in the interpretation of remote sensing images. *Journal of Arid Environments* **75**, 1164-1172

Henderson, R. J. F. (Ed.) (2002). 'Names and distribution of Queensland plants, algae and lichens.' (Queensland Herbarium, Queensland Environmental Protection Agency: Brisbane.)

Herrmann, S. M., Anyambab, A., & Tucker, C. T. (2005). Recent trends in vegetation dynamics in the African Sahel and their relationship to climate. *Global Environmental Change* **15**, 394–404.

Hill, M.J., Donald, G.E., Vickery, P.J., Moore, A.D. & Donnelly, J.R. (1999). Combining satellite data with a simulation model to describe spatial variability in pasture growth at a farm scale. *Australian Journal of Experimental Agriculture* **39**: 285-300

Hill, M. J., & Hanan, N. P. (2011). Current approaches to measurement, remote sensing and modelling in savannas: A synthesis. In M. J. Hill, & N. P. Hanan (Eds.), *Ecosystem function in savannas: Measurement and modelling at landscape to global scales* (pp. 515 - 545). Boca Raton, Florida: CRC Press.

Hobbs, R.J. (1993). Effects of landscape fragmentation of ecosystem processes in the Western Australian Wheatbelt. *Biological Conservation* **64**: 193 -201.

Hodgson, J. Illius, A.W. (Eds) (1996) The ecology and management of grazing systems. CAB International.

Holling, C.S. (1973). Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* **4**: 1–23.

Holling, C.S. (1986). Resilience of ecosystems; local surprise and global change. In: Clark, W.C., Munn, R.E., (Eds.) Sustainable development of the biosphere. Cambridge (UK): Cambridge University Press 292-317.

Holling, C.S. & Gunderson, L.H.(2001). Reslience and adaptive cycles. In Gunderson , L. Holling, C.S. (Eds.) Panarchy: understanding transformations in human and natural systems. Washington (DC): Island Press.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A.J., Vandermeer, J. & Wardle, D.A. (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecological Monographs* **75** (1): 3-35

Houghton, R.A., Hackler, J.L., & Lawrence, K.T. (1999). The US carbon budget: contributions from land-use change. *Science* **285**: 574–578

Houet, T., Loveland, T.R., Hubert-Moy, L., Gaucherel, C., Napton, D., Barnes, C.A. & Sayler, K.L. (2009). Exploring subtle land use and land cover changes: a framework for future landscape studies, *Landscape Ecology* **25** (2): 249-266

House, J.I. & Hall, D.O. (2001). Production of tropical savannas and grasslands. *Terrestrial Global Productivity* **33**: 363-400

Hughes, P. (1993). Practical experience with grazing systems. Proceedings of a grazing systems seminar. Soil and Water Conservation Association of Australia, Rockhampton. pp. 20-25

Hunt, L.P., Petty, S., Cowley, R., Fisher, A., Ash, A.J. & MacDonald, N. (2007). Factors affecting the management of cattle grazing distribution in northern Australia:

preliminary observations on the effect of paddock size and water points. *Rangeland Journal* **29**: 169-179.

Hunt, L., Fisher, A., Kutt, A. & Mazzer, T. (2006). Biodiversity Monitoring in the Rangelands: A way forward. Vol.2. Case Studies CSIRO NT.

Illius, A.W. & O'Connor, T.G. (1999). On the relevance of non-equilibrium concepts to arid and semi-arid grazing systems. *Ecological Applications* **9** (3): 798-813

Ives, A. R. & Zhu, J. (2006). Statistics for correlated data: phylogenies, space, and time. *Ecological Applications* **16**: 20-32

Jackson, J. (2005). Is there a relationship between species richness and buffel grass (*Cenchrus ciliaris*). *Austral Ecology* **30**: 505–517

James, C. (2003). Response of vertebrates to fence line contrasts in grazing intensity in semi-arid woodlands of eastern Australia. *Austral Ecology* **28**: 137-151

James, C.D., Landsberg, J. & Morton, S.R. (1999). Provision of watering points in the Australian arid zone: a review of effects on biota. *Journal of Arid Environments* **41**: 87-121

Jeffrey, S.J., Carter, J.O., Moodie, K.M. & Beswisk, A.R. (2001). Using spatial interpolation to construct a comprehensive archive of Australian climate data. *Environmental Modelling and Software* **16**(4): 309-330

Jeltsch, F., Weber, G.E. & Grimm, V. (2000) Ecological buffering mechanisms in savannas: a unifying theory of long-term tree-grass coexistence. *Plant Ecology*, **161**, 171.

Karfs, R. A., Abbott, B. N., Scarth, P. F. & Wallace, J. F. (2009). Land condition monitoring information for reef catchments: A new era. *Rangeland Journal* **31**: 69–86.

Karfs, R. & Fisher, A. (2002). Linking landscape function, land condition, grazing and wildlife. In: Proceedings, Fire and Heterogeneity in Savanna Landscapes, 8-12 July 2002, pp. 64. Northern Territory University, Darwin, NT, Australia.

Karr, J.R. & Dudley, D.R., (1981). Ecological processes on water quality goals. *Environmental Management* **5** (1): 55-68.

Kati, V., Devillers, P., Dufrene, M., Legakis, A., Vokou, D. & Lebrun, P. (2004). Testing the value of six taxonomic groups as biodiversity indicators at a local scale. *Conservation Biology* **18**: 667-75

Kelly, J.R. & Harwell, M.A. (1990) Indicators of Ecosystem recovery. *Environmental Management* **14** (5); 527-545

Kennedy, R.E., Cohen, W.B., Schroeder, T.A. (2007). Trajectory-based change detection for automated characterization of forest disturbance dynamics. *Remote Sensing of Environment*. **110**: 370-386

Kienast, F., Bolliger, J., Potschin, M., de Groot, R.S., Verburg, P.H., Heller, I., Wascher, D & Haines- Young, R. (2009). Assessing landscape functions with broad-scale environmental data: Insights gained from a prototype development for Europe. *Environmental Management* **44**: 1099-1120

Keith, D. & Gorrod, E. (2006) The meanings of vegetation condition. *Ecological Management and Restoration* **7**(1): S7-S9

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

Kirkpatrick, J.B. (1994). 'A continent transformed: human impact on the natural vegetation of Australia'. Oxford University Press, Australia.

Knopt, F.L., Sedgwick, J.A. & Cannon, R.W. (1987). Guild structure of a riparian avifauna relative to seasonal cattle grazing. *Journal of Wildlife Management* **52** (2): 280-290

Kutt, A.S. (2004) Patterns in the composition and distribution of the vertebrate fauna, Desert Uplands bioregion, Queensland. PhD Thesis. James Cook University.

Kutt, A., Eyre, T., Fisher, A. & Hunt, L. (2009). A biodiversity monitoring program for Australian rangelands. ACRIS – Biodiversity Monitoring Program, Australian Government.

Lambin, E.F., Turner, B.L., Geist, H.J, Agbola, S.B., Angelsen, A., Bruce, J.W., Coomes, O.T., Dirzo, R., Fischer, G., Folke, C., George, P.S., Homewood, K., Imbernon, J., Leemans, R., Li, X.B., Moran, E.F., Mortimore, M., Ramakrishnan, P.S., Richards, J.F., Skanes, H., Steffen, W., Stone, G.D., Svedin, U., Veldkamp, T., Vogel, C., & Xu, J.C. (2001). The causes of land use and land-cover change: moving beyond the myths. *Global Environmental Change Human Policy Dimensions* **11**: 261–269

Landres, P.B., Verner, J. & Thomas, J.W. (1988). Ecological uses of vertebrate indicator species: a critique. *Conservation Biology* **2**: 316-28

Landsberg, J. & Crowley, G. (2004). Monitoring rangeland biodiversity: Plants as indicators. *Austral Ecology* **29:** 59-77.

Landsberg, J., James, Morton, S.R., Muller, W.J. & Stol, J. (2003). Abundance and composition of plant species along grazing gradients in Australian rangelands. *Journal of Applied Ecology* **40**: 1008-1024

Landsberg, J., Morton, S. & James, C. (1999). A comparison of the diversity and indicator potential of arthropods, vertebrates and plants in arid rangelands across Australia. In: *The Other 99%. The Conservation and Biodiversity of Invertebrates* (Eds. W. Ponder & D. Lunney). pp. 111-20. Surrey Beatty & Sons, Chipping North, NSW.

Landsberg, J. & Stol, J. (1996). Spatial distribution of sheep, feral goats and kangaroos in woody rangeland paddocks, *Australian Journal of Rangelands* **18:** 270-291

Lange, R. T. (1969). The piosphere: Sheep track and dung patterns. *Journal of Range Management* **22:** 396–400.

Lavorel, S. (1999). Ecological diversity and resilience of Mediterranean vegetation to disturbance. *Diversity and Distributions* **5**:1-2.

Lavorel, S. & Garnier, E. (2002). Predicting the effects of environmental changes on plant community composition and ecosystem functioning: revisiting the Holy Grail. *Functional Ecology* **16**: 545-556

Laylock, W.A. (1991). Stable states and thresholds of range condition on North American rangelands: A viewpoint. *Journal of Range Management* **44**: 427-433

Leigh, J.H., Wood, D.H., Holgate, M.D., Slee, A. & Stanger, M.G. (1989). Effects of rabbit and kangaroo grazing on two semi-arid grassland communities in central-western New South Wales. *Australian Journal of Botany* **37**(5): 375-396

Leslie, R.G., Mackey, B.G. & Preece, K. (1988). A computer-based method of wilderness evaluation. *Environmental Conservation* **15**: 225-232

Levine, J.M, Vila, M., D'Antonio, C.M., Dukes, J.S., Grigules, K. & Lavorel, S. (2003) Mechanisms underlying the impacts of exotic plant invasions. *Proceedings of the Royal Society of London, Series B* **270**: 775-781

Li, Z., Huffman, T., McConky, B. & Townley-Smith, L. (2013). Monitoring and modelling spatial and temporal patterns of grassland dynamics using time-series MODIS NDVI with climate and stocking data. *Remote Sensing of Environment* **138**: 232-244

Li, L., Ustin, S.L. & Lay, M. (2005). Application of multiple endmember spectral mixture analysis (MESMA) to AVIRIS imagery for coastal salt marsh mapping: A case study in China Camp, CA, USA*. International Journal of Remote Sensing* **26**: 5193-5207.

Lindenmayer, D.B. & Burgman, M. (2005). Practical Conservation Biology. CSIRO Publishing, Collingwood.

Lindenmayer, D., Crane, M. & Michael, D. (2005). 'Woodlands: a disappearing landscape.' (CSIRO Publishing: Collingwood, Vic.)

Lovett, G. M., Canham, C. D., Arther, M. A., Weathers, K. C. & Fitzhugh, R. D. (2006). Forest ecosystem responses to exotic pests and pathogens in eastern North America. *BioScience* **56**: 395–405

Ludwig, J.A. & Bastin, G.N. (2008). Rangeland condition: its meaning and use. A discussion paper prepared for the Australian Collaborative Rangeland Information System (ACRIS) management committee.

Ludwig, J. A., Bastin, G. N., Wallace, J. F., & McVicar, T. R. (2007). Assessing landscape health by scaling with remote sensing: when is it not enough? *Landscape Ecology* **22**: 163–169

Ludwig, J.A. & Tongway, D.J. (1997). A landscape approach to rangeland ecology. In: Landscape Ecology Function and Management . *Principles from Australia's Rangelands.* Ludwig, J., Tongway, D., Freudenberger, D., Noble, J., and Hodgkinson, K. (Eds). pp. 1 – 12. CSIRO Publishing, Collingwood.

Ludwig, J.A. & Tongway, D.J. (1992). Monitoring the condition of Australian arid lands: linked plant-soil indicators. In *Ecological Indicators* (Eds D.H. McKenzie, D.E. Hyatt and V.J. McDonald) pp 765-772. Elsevier, Essex.

Ludwig, J.A. & Tongway D.J. (1996). Rehabilitation of semiarid landscapes in Australia. II Restoring vegetation patches. Restoration Ecology 4: 398-406

Ludwig, J.A., Tongway, D.J., Bastin, G.N. & James, C.D. (2004). Monitoring ecological indicators of rangeland functional integrity and their relation to biodiversity at local and regional scales. *Austral Ecology* **29** (1): 108-123

Ludwig, J.A., Tongway, D.J. & Marsden, S.G. (1994). A flow-filter model for simulating the conservation of limited resources in spatially heterogeneous, semi-arid landscapes. *Pacific Conservation Biology* **1:** 209-213

Lunt, I. D., Eldridge, D. J., Morgan, J. W. & Witt, B. (2007). A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia. *Australian Journal of Botany* **55**: 401–415

McAlpine, C.A., Suttcliffe, T., Eyre, T.J. & Taylor, K. (2002). One hundred and fifty years of landscape change for two sub-regions of the Southern Brigalow:

Patterns and management implications. In: Landscape health in Queensland (Eds. A. Franks and J. Playford). Royal Society of Queensland, Brisbane.

McCann, R. K., Marcot, B. G. & Ellis, R. (2006). Bayesian belief networks: applications in ecology and natural resource management. *Canadian Journal of Forest Research* **36**: 3053–3062.

McCosker, J., Rolfe, J. and Fensham, R. (2009). Can bare ground cover serve as a surrogate for plant biodiversity in grazed tropical woodlands? *The Rangeland Journal* **31**, 103–109.

McCullough, M. & Musso, B. (Eds) (2004). 'Healthy Rangelands – Principles for Sustainable Systems.' (Tropical Savannas CRC: Darwin, NT.)

McGeogh, M.A. (1998). The selection, testing and application of terrestrial insects as bioindicators. *Biological Reviews* **73**: 181-201

McElhinny, C., Gibbons, P., Brack, C. & Bauhus, J. (2006). Fauna-habitat relationships: a basis for identifying key stand structural attributes in temperate Australian eucalypt forests and woodlands. *Pacific Conservation Biology* **12:** 89-110

McIntyre, S. & Filet, P. G. (1997). Choosing appropriate taxonomic units for ecological survey and experimentation: the response of *Aristida* to management and landscape factors as an example. *The Rangeland Journal* **19**: 26–39

McIntyre, S. & Hobbs, R. A (1999). Framework for conceptualising human effects on landscapes and its relevance to management and research models. *Conservation Biology* **13** (6): 1282-1292

McIntyre, S. & Lavorel, S. (2001). Livestock grazing in subtropical pastures: steps in the analysis of attribute response and plant functional types. *Journal of Ecology* **89**: 209-226.

McIntyre, S. & Martin, T.G. (2001). Biophysical and human influences on plant species richness in grasslands: comparing variegated landscapes in subtropical and temperate regions. *Austral Ecology* **26**: 233 -245.

McIntyre, S., McIvor, J.G. & Heard, K.M. (Ed.) (2002). Managing and Conserving Grassy woodlands. CSIRO Publishing.

McIvor, J. G. (2007). Pasture management in semi-arid tropical woodlands: dynamics of perennial grasses. *The Rangeland Journal* **29**: 87–100.

McIvor, J.G., Ash, A.J. & Cook, G.D., (1995). Land condition in the tropical tallgrass pasture lands 1. Effects on herbage production. *Rangeland Journal* **17** (1): 69-85.

McKeon, G. M., Hall, W. B., Henry, B. K., Stone, G. S. & Watson, I. W. (2004). 'Pasture Degradation and Recovery in Australia's Rangelands: Learning from History.' (Queensland Department of Natural Resources, Mines and Energy: Brisbane.)

MacLeod, N. D. & McIvor, J. G. (2006). Reconciling economic and ecological conflicts in the use of grazing lands. *Ecological Economics* **56**: 386–401

MacNally, R., Ellis, M. & Barrett, G. (2004). Avian biodiversity monitoring in Australian rangelands. *Austral Ecology* **29** (1): 93-99

McVicar T.R. & Jupp D.L.B. (1998). The current and potential operational uses of

remote sensing to aid decisions on drought exceptional circumstances in Australia: a review. *Agricultural Systems* **57**:399–468.
Maestre, F.T., Quero, J.L, Gotelli, N,J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M.A., Soliveres, S., Escolar, C., García-Palacios, P., Berdugo, M., Valencia, E., Gozalo, B., Gallardo, A., Aguilera, L., Arredondo, T., Blones, J., Boeken, B., Donaldo, B., Abel, A., Conceição, A. A., Cabrera, O., Chaieb, M., Derak, M., Eldridge, D.J., Espinosa, C.I., Florentino, A., Gaitán, J., Gabriel Gatica, M., Ghiloufi, W., Gómez-González, S., Gutiérrez, J.R., Hernández, R.M., Huang, X., Huber-Sannwald, E., Jankju, M., Miriti, M., Monerris, J., Mau, J.L., Morici, E., Naseri, K., Ospina, A., Polo, V., Prina, A., Pucheta, E., Ramírez-Collantes, D.A., Romão, R., Tighe, M., Torres-Díaz, C., Val, J., Veiga, J.P., Wang, D. & Zaady, E. (2012). Plant species richness and ecosystem multifunctionalilty in global dry lands. *Science* **335**: 214-217
Maher, J.V. (1973). Meteorological aspects of drought. *The Environmental, Economic and Social significance of Drought* (ed. J.V. Lovett), pp. 41-54. Angus and Robertson, Sydney, Australia
Maron, M. & Lill, A. (2005). The influence of livestock grazing and weed invasion on habitat use by birds in grassy woodland remnants. *Biological Conservation* **124**: 439-50
Maron, M., Main, A., Bowen, M., Howes, A., Kath, J., Pilette, C. & McAlpine, C.A. (2011). Relative influence of habitat modification and interspecific competition on woodland bird assemblages in eastern Australia. *Emu* 111: 40-51
Martin, T.E. (1995). Avian life history evolution in relation to nest sites, nest predation and food. *Ecological Monographs* **65**: 101-27
Martin, T.G. & Possingham, H.P. (2004). Predicting the impact of livestock grazing on birds using foraging height data. *Journal of Applied Ecology* **42**: 400-408
Matern, A., Drees, C., Kleinwachter, M. & Assmann, T. (2007). Habitat modelling for the conservation of the rare ground beetle species Carabus variolosus (Coleoptera, Carabidae) in the riparian zones of headwaters. *Biological Conservation* **136**: 618-627
Mazaris, A.D., Kallimanis, A.S., Tzanopoulous, J., Sgardela, S.G. & J.P. Pantis. (2010). Can we predict the number of plant species from the richness of a few common species, families or orders? *Journal of Applied Ecology* **47:** 662-670
Mikusinski, G., Gromadzki, M. & Chylarecki, P. (2001). Woodpeckers as indicators of forest bird diversity. *Conservation Biology* **15**: 208-17
Milchunas, D.G & Lauenroth, W.K. (1993). Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs* **63** (4): 327-366
Milchunas, D.G., Sala, O.E. & Lauenroth, W.K. (1988). A generalised model of the effects of grazing by large herbivores on grassland community structure. *American Naturalist* **132**: 87-106
Mooney, H.A. (2002). The debate on the role of biodiversity in ecosystem functioning. In *Biodiversity and ecosystem functioning* by M.Loreau, S. Naeem and P. Inchausti, (Eds). Pp 12-17. Oxford University Press, Oxford, UK.
Moore, J.L., Balmford, A., Brooks, T., Burgess, N.D., Hansen, L.A., Rahlek, C. & Williams, P.H. (2003). Performance of sub-Saharan vertebrates as indicator

groups for identifying priority areas for conservation. *Conservation Biology* **17**: 207-18

Morgan Ernest, S.K. & Brown, J.H. (2001). Homeostasis and compensation: the role of species and resources in ecosystem stability. *Ecology* **82**: 2118-2132

Morton, S.R., Stafford Smith, D.M., Dickman, C.R., Dunkerley, D.L., Friedel, M.H., McAllister, R.R.J., Reid, J.R.W., Roshier, D.A., Smith, M.A., Walsh, F.J., Wardle, G.M., Watson, W.W. & Westoby, M. (2011). *A fresh framework for the ecology of arid Australia, Journal of Arid Environments* **75**: 373-329

Mott, J. J. (1987). Patch grazing and degradation in native pasture of the tropical savannas in Northern Australia. *In*: 'Grazing Land Research at the Plant–Animal Interface'. (Eds P. P. Horan, J. Hodgson, J. J. Mott and R. W. Brougham.) pp. 153–166. (Winrock International: Little Rock, AR.)

Naveh, Z. (1967). Mediterranean ecosystems and vegetation types in California and Israel. *Ecology* **48**: 445-449

Ngakane, S., Slade, G. & Stuart-Hill, G (1999). Development of a seasonal vegetation monitoring system for Botswana. *Sixth International Rangeland Congress Proceedings*. Townsville, Australia. VI International Rangeland Congress Inc, pp. 810-811.

Noble, J.C. & Tongway, D.J. (1983). Herbivores in arid and semi-arid rangelands. In 'Australian soils: the human impacts'. (Eds. J.S. Russell, R.F. Isbel) pp. 243-270. University of Queensland Press.

Northup, B.K., Brown, J.R. & Ash, A.J. (2005). Grazing impacts on spatial distribution of soil and herbaceous characteristics in an Australian tropical woodland. *Agroforestry System* **65**: 137-150

Noy-Meir, I. (1973). Desert ecosystems: environment and producers. *Annual Review of Ecological Systems* **4**: 23-51

Noss, R.F. (1990). Indicators for monitoring Biodiversity: a hierarchical approach. *Conservation Biology* **4** (4): 355-364

Numata, I., Roberts, Dar A., Chadwick, O.A., Schimel, J., Sampaio, F.R., Leonidas, F.C. & Soares, J.V. (2007). Characterisation of pasture biophysical properties and the impact of grazing intensity using remotely sensed data. *Remote Sensing of Environment* **109**: 314-327

Oba, G., Stenseth, N.C. & Lusigi, W.J. (2000). New perspectives on sustainable grazing in management in arid zones of sub-Saharan Africa. *Bioscience* **50**: 35-51

O'Connor, T.G. & Roux, P.W. (1995). Vegetation changes (1947-71) in a semi-arid, grassy dwarf shrub land in the Karoo, South Africa: influence of rainfall variability and grazing by sheep. *Journal of Applied Ecology* **32**:612-626

Oesterheld, M., Sala, O.E., & McNaughton S.J. (1992). Effect of animal husbandry on herbivore-carrying capacity at a regional scale. *Nature* **356**: 234-236

Oliver, I. (2003). An expert panel approach to the assessment of vegetation condition within the context of biodiversity conservation. Stage 1: The identification of condition indicators. *Ecological Indicators* **2:** 223-237.

Oliver, I. (2004). A framework and toolkit for scoring the biodiversity value of habitat, and the biodiversity benefits of land use change. *Ecological Management and Restoration* **5**: 75-77

Oliver, I., Jones, H., & Schmoldt, D. L. (2007). Expert panel assessment of attributes for natural variability benchmarks for biodiversity. *Austral Ecology* **32**: 453–475.

Olsen, P., Weston, M., & Cunningham, R. & Silcocks, A. (2003). State of Australia's Birds. *Supplement to Wingspan 13 (4)*

Omuto, C.T., Vargas, R.R., Alim, M.S. & Paron, P. (2010). Mixed-effects modelling of time series NDVI-rainfall relationship for detecting human-induced loss of vegetation cover in drylands. *Journal of Arid Environments* **74**: 1552-1563

Orians, G.H. & Milewski, A.V. (2007). Ecology of Australia: the effects of nutrient-poor soils and intense fires. *Biological Reviews* **82**: 393-423.

Orr, D.M. & O'Reagain, P.J. (2011). Managing for rainfall variability: impacts of grazing strategies on perennial grass dynamics in a dry tropical savanna. *Rangeland Journal* **33**: 209-220

Parkes, D., Newell, G & Cheal, D. (2003). Assessing the quality of native vegetation: The habitat hectares approach. *Ecological Management and Restoration* **4**: S29-S38.

Parsons, S.D. (1995). Putting profit into grazing. Resource Consulting Services, Yeppoon, QLD.

Paruelo, J.M., Putz, S, Weber, G., Bertiller, M., Golluscio, R.A., Aguiar, M.R., & Wiegand, T. (2008). Long-term dynamics of a semiarid grass stepped under stochastic climate and different grazing regimes: a simulation analysis. *Journal of Arid Environment* **72**: 2211-2231

Patridge, I. (2000). 'Managing Grazing in the Semi-arid Woodlands: a Graziers Guide.' (Department of Primary Industries: Brisbane.)

Pavey, C.R. & Nano, C.E.M. (2009). Bird assemblages of arid Australia: Vegetation patterns have a greater effect than disturbance and resource pulse. *Journal of Arid Environments* **1**: 634-642

Pearman, P.B. & Weber, D. (2007). Common species determine richness patterns in biodiversity indicator taxa. *Biological Conservation* **138**: 109-19

Perry, J.C., Kutt, A.S., Perkin, G.C., Vanderduys, E.P. & Colman, N.J. (2012). A bird survey method for Australian tropical savannas. *Emu* **112**: 261-266

Pellant, M., Shaver, P., Pyke, D.A. & Herrick, J.E. (2000). Interpreting indicators of rangeland health, version 3, Technical Reference 1734-6. USDI, BLM, Natural Sciences and Technical Centre, Denver, Colorado

Pennington, D.D. & Collins, S.L. (2007). Response of an aridland ecosystem to interannual climate variability and prolonged drought. *Landscape Ecology* **22:** 897-910

Perry, G. L. W. & Millington, J. D. A. (2007). Spatial modelling of succession disturbance dynamics in forest ecosystems: Concepts and examples. *Perspectives in Plant Ecology, Evolution and Systematics* **9:** 191–210

Pickup, G., Bastin, G.N. & Chewings, V.H. (1994). Remote-sensing assessment for non-equilibrium rangelands under large-scale commercial grazing. *Ecological Applications* **4** (3): 497-517

Pickup, G., Bastin, G. N., & Chewings, V. H. (1998). Identifying trends in land degradation in non-equilibrium rangelands. *Journal of Applied Ecology* **35**, 365–377.

Pickup, G. & Chewings, V.H. (1988). Estimating the distribution and patterns of cattle movement in a large arid zone paddock – an approach using animal distribution patterns and Landsat imagery. *International Journal of Remote Sensing* **9**: 1469-1490

Pickup, G., Chewings, V.H. & Nelson, D.J. (1993). Estimating changes in vegetation cover over time in arid rangelands using Landsat MSS data. *Remote Sensing of the Environment* **43**: 243-263.

Pielke, R.A., Avissar, R., Raupach, M., Dolman, A.J., Zeng, X. & Denning, A.S. (1998). Interactions between the atmosphere and terrestrial ecosystems: influence on weather and climate. *Global Change Biology* **4:** 461–475.

Pinheiro, J. C. & Bates, D.M.. (2000). Mixed-Effects Models in S and S-Plus. Springer Verlag, New York.

Price, B., McAlpine, C.A., Kutt, A.S., Phinn, S.P., Pullar, D.V. & Ludwig, D.V. (2009). Continuum or discrete patch landscape model for savanna birds? Towards a pluralistic approach. *Ecography* **32**:745-756

Prince, S.D., Brown de Colstoun, E., & Kravitz, L. (1998). Evidence from rain use efficiencies does not support extensive Sahelian desertification. *Global Change Biology,* **4**: 359-374

Pringle, H.J.R. & Tinley, K.L. (2003). Are we overlooking critical determinants of landscape change in Australian rangelands? *Ecological Management and Restoration* **4**: 180-186

Pringle, H.J.R. & Landsberg, J. (2004). Predicting the distribution of livestock grazing pressures in rangelands. *Austral Ecology* **29**: 31-39.

Pringle, H.J.R., Watson, I.W. &Tinley, K.L. (2006). Landscape improvement, or ongoing degradation – reconciling apparent contradictions from the arid rangelands of Western Australia. *Landscape Ecology* **21**: 1267-1279.

Queensland Department of Primary Industries and Fisheries. (2005). Stocktake - balancing supply and demand. QPIF publications, Queensland.

Queensland Department of Science, Information Technology, Innovation and the Arts. (2012). Land cover change in Queensland from 2009-2010. A state-wide land cover and tree study report. DSITIA, Brisbane.

Razeng, E. & Watson, D.M. (2012). What do declining woodland birds eat? A synthesis of dietary records. *Emu* **112**, 149-156

Recher, H.F. & Christensen, P.E. (1981). Fire and the evolution of the Australian biota. In *Ecological Biogeography of Australia* (ed. A. Keast) pp. 135-162. (Dr. W. Junk: The Hague)

Reid, J.R.W. (1999). Threatened and declining birds in the New South Wales Sheep-Wheat Belt: 1. Diagnosis, characteristics and management. In Consultancy report to NSW National Parks and Wildlife Service. CSIRO Wildlife and Ecology, Canberra.

Reside A.E., Van Der Wal J.J., Kutt A.S., & Perkins G.C. (2010) Weather, Not Climate, Defines Distributions of Vagile Bird Species. *PLoS ONE* **5**(10): e13569. doi:10.1371/journal.pone.0013569

Reinke, K. & Jones, S. (2006). Integrating vegetation field surveys with remotely sensed data. *Ecological Management & Restoration* **7**:S18-S23

Reynolds, J.F., Stafford Smith, D.M., Lambin, E.F., Turner, B.L., Mortimore, M., Batterbury, S.P.J., Downing, T.E., Dowlatabadi, H., Fernandez, R.J., Herrick, J.E., Huber-Sannwald, E., Jiang, H., Leemans, R., Lynam, T., Maestre, F.T., Ayarza, M. & Walker, B. (2007) Global desertification: building a science for dryland development. *Science*, **316**, 847–851.

Richardson, A.J. & Weigland, C.L. (1977). Distinguishing vegetation from soil background information. *Programme Engineering for Remote Sensing* **43:** 1541-1552.

Risser, P.G., Birney, E.C., Blocker, H.D., May, W.S, Parton, W.J. & Weins, J.A. (1981). The true prairie ecosystem. Hutchinson and Ross, Stroudesburg, Pennsylvania.

Risser, P.G (1995). Biodiversity and ecosystem function. *Conservation Biology* **9** (4): 1-5

Robinson, S.K. & Holmes, R.T. (1982). Foraging behaviour of forest birds: the relationships among search tactics, diet and habitat structure. *Ecology* **63** (6): 1918-1931

Roder, A., Udelhoven, Th., Hill, J., del Barrio, G. & Tsiourlis, G. (2008). Trend analysis of Landsat – TM and –ETM+ imagery to monitor grazing impact in a rangeland ecosystem in Northern Greece. *Remote Sensing of Environment* **112**: 2863-2875

Rose, R.A., Byler, D, Eastman, J.R., Fleishman, E., Geller, G., Goetz, S., Guild, L, Hamilton, H., Hansen, M., Headley, R., Hewson, J., Horning, N., Kaplin, B.A., Laporte, N., Leidner, A., Leimgruber, P., Morisette, J., Musinky, J., Pintea, L., Prados, A., Radeloff, V.C., Rowen, M., Saatchi, S., Schill, S., Tabor, K., Turner, W., Vodcek, A., Vogelmann, J., Wegmann, M., Wilkie, D. & Wilson, C. (2014) Ten ways remote sensing can contribute to conservation. *Conservation Biology* **0**:1-10

Roth, C.A. (2004). A framework relating soil surface condition to infiltration and sediment and nutrient mobilisatoin in grazed rangelands of northeastern Queensland, Australia *Earth Surface Processes and Landforms* **29**: 1093–1104

Root, R.B. (1967). The niche exploitation pattern of the blue-gray gnatcatcher. *Ecological monographs* **37**(4): 317-350

Rumpff, L., Duncan, D. H., Vesk, P. A., Keith, D. A. & Wintle, B. A. (2011). State-and-transition modelling for adaptive management of native woodlands. *Biological Conservation* **144**: 1224–1236.

Running, S.W., Nemani, R., Heinsch, F.A., Zhao, M., Reeves, M. & Hashimoto, H. (2004). A continuous satellite-derived measure of global terrestrial primary production. *Bioscience*. **54** (6):547-560

Ryerson, D.E. & Parmenter, R.R. (2001). Vegetation change following removal of keystone herbivores from desert grasslands in New Mexico. *Journal of Vegetation Science* **12**: 167-180

Sattler, P. S. and Williams, R. D. (Eds) (1999). 'The Conservation Status of Queensland's Bioregional Ecosystems.' (Environmental Protection Agency: Brisbane).

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

Scanlan, J.C. (1994). State and transition models for rangelands. 5. The use of state and transition models for predicting vegetation change in rangelands. *Tropical Grasslands* **28**: 229-240

Scanlan, J.C. & Burrows, W.H. (1990). Woody over-storey impact on herbaceous understorey in *Eucalyptus spp*. communities in central Queensland. *Australian Journal of Ecology* **15**: 191-197

Scanlan, J.C. & McIvor, I.G. (1993). Pasture composition influences on soil erosion in Eucalyptus woodlands of northern Queensland. *Proceedings of the XVII International Grassland Congress*: 65-66

Scanlon, T.M., Caylor, K.K., Manfreda, S., Levin, S.A. & Rodriguez-Iturbe, I. (2005). Dynamic response of grass cover to rainfall variability: implications for the funcition and persistence of savanna ecosystems. *Advances in Water Resources* **28**: 291-302

Scarth, P., Byrne, M., Danaher, T., Henry, B., Hassett, R., Carter, J., & Timmers, P. (2006). State of the paddock: monitoring condition and trend in ground cover across Queensland. *In*: 'Proceedings of 13th Australasian Remote Sensing and Photogrammetry Conference'. (The Remote Sensing and Photogrammetry Commission of Spatial Sciences Institute: Canberra, ACT).

Scarth, P., Roder, A., Schmidt, M., & Denham, R. (2010) Tracking grazing pressure and climate interaction – the role of Landsat fractional cover in time series analysis. *Proceedings of the 15th Australasian Remote Sensing and Photogrammetry Conference*.

Schodde, R. (1982). Origin, adaptation and evolution of birds in arid Australia, pp 191-224. In 'Evolution of the flora and fauna of arid Australia (Eds. W.R. Baker and P.J.M. Greenslade). Peacock Publication, South Australia.

Scholes, R.J. & Archer, S.R. (1997). Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics* **28**: 517-544

Seymour, C.L. & Dean, W.R.J. (1999). Effects of heavy grazing on invertebrate assemblages in the succulent Karoo, South Africa. *Journal of Arid Environments* **43(3):** 267-286

Sheffield, K. (2009). *Multi-spectral remote sensing of native vegetation condition.* PhD thesis. School of Mathematical and Geospatial Sciences, RMIT University.

Simberloff, D. (1998). Flagships, umbrellas and keystones: is single-species management passé in the landscape era? *Biological Conservation* **83**: 247-257

Simpson, E.H. (1949). Measurement of diversity. *Nature* **163**: 688

Smyth, A. K., Brandle, R., Chewings, V., Read, J., Brook, A., & Fleming, M. (2009). A framework for assessing regional biodiversity condition under changing environments of the arid Australian rangelands. *The Rangeland Journal* **31:** 87–101.

Smyth, A.K. & James, C.D. (2004). Characteristics of Australia's rangelands and key design issues for monitoring biodiversity. *Austral Ecology* **29**: 3-15.

Smyth, A., James, C. & Whiteman, G. (2003). Biodiversity Monitoring in the Rangelands: A way forward, report to Environment Australia, vol. 1, Centre for Arid Zone Research, CSIRO Sustainable Ecosystems, Alice Springs.

Spooner, P.G. & Allcock, K.G. (2006). Using a State-and-Transition approach to manage endangered *Eucalyptus albens* (White Box) woodlands. *Environmental Management* **38**: 771–783.
Stafford Smith, D.M. (1984). Behavioural ecology of sheep in the Australian arid zone. Dissertation. Australian National University, Canberra, Australia
Stafford Smith, D.M. (1996). Management of rangelands: paradigms at their limits. In: J. Hodgson and A.W. Illius (Eds.) The ecology and management of grazing systems. Wallingford, U.K. CAB International, pp. 325-357.
Stafford Smith, D.M. & Pickup, G. (1990). Pattern and production in arid lands. *Proceedings of the Ecological Society of Australia* **16**: 195-200
Stafford Smith, D.M. & Pickup, G. (1993). Out of Africa, looking in: understanding vegetation change and its implications for management in Australian rangelands. In: Behnke, R.H., Scones, I. & Kerven, C. (Eds) Range Ecology at Disequilibrium. Overseas Development Institute/International Institute for Environment and Development/Commonwealth Secretariat, London, pp.196-226
State Land Asset Management (2011). *Delbessie Agreement (State Rural Leasehold Land Strategy): Guidelines for determining lease condition Version 2.* Department of Environment and Resource Management, Brisbane
Stevens, H.C. & Watson, D.M. (2013). Reduced rainfall explains avian declines in an unfragmented landscape: incremental steps toward an empty forest? *Emu* **113,** 112-121
Stringham, T.K., Krueger, W.C. & Shaver, P.L. (2003). State and transition modelling: an ecological process approach. *Journal of Range Management* **56**:106-113.
Tainton, N.M. & Walker, B.H. (1993). Grasslands of southern Africa. In: Ecosystems of the world 8B: Natural Grasslands (Ed. R.T. Copeland) pp.265-290.Elsevier
Tansley, A.J. (1935). The use and abuse of vegetational concepts and terms. *Ecology* **16**: 248-307
ter Braak, C.J.K. & Smilauer, P. (2002) CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (Version 4.5) Microcomputer Power, Ithaca, New York.
Taylor, S.G. (2008). Leaf litter invertebrates in box-ironbark forest: composition, size and seasonal variation in biomass. *Victorian Naturalist* **125**, 19-27.
Taube, C.A. (2000). Estimating ground cover in semi-arid rangelands using Landsat Thematic Mapper Imagery. *Master's Thesis*. Geographical Sciences and Planning Department, University of Queensland.
Thrash, I. & Derry, J.F. (1999). The nature and modelling of piospheres: a review. *Koedoe* **42(**2): 73-94
Tilman, D. & Downing, J.A. (1994). Biodiversity and stability in grasslands. *Nature* **367**: 363-365
Tomkins, N.W., O'Reagain, P.J., Swain, D., Bishop-Hurley, G. & Charmley, E. (2009). Determining the effect of stocking rate on the spatial distribution of cattle for subtropical savannas. *Rangeland Journal* **31**:267-276.
Ludwig, J.A., Tongway, D.J. & Marsden, S.G. (1994). A flow-filter model for simulating the conservation of limited resources in spatially heterogeneous, semi-arid landscapes. *Pacific Conservation Biology* **1:** 209-213

Tongway, D.J., Sparrow, A.D. & Friedel, M.H. (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

Tongway, D. J., & Hindley, N.L. (2004) Landscape function analysis manual: procedures for monitoring and assessing landscapes with special reference to minesites and rangelands Version 3.1 CSIRO Sustainable Ecosystems, Canberra, Australia.

Tongway, D.J. & Ludwig, J.A. (1997). The nature of landscape dysfunction in rangelands, Chapter 5. In: J.A. Ludwig, D.T. Tongway, D. Freudenberger, J. Noble and K. Hodgkinson (Eds), Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia, pp. 49-61.

Tothill, J.C. (1971). A review of fire in the management of native pasture with particular reference to north-eastern Australia. Tropical Grasslands **5**(1): 1-10

Turner, E. J., Beeston, G. R., Lee, A. N., Ahern, C. R. & Hughes, K. K. (1978). Western Arid Use Study, Part IV. Technical Bulletin 23, Division of Land Utilisation, Department of Primary Industries, Brisbane.

Turner, M. G. (2005). Landscape ecology: What is the state of the science? *Annual Review of Ecology, Evolution, and Systematics* **36**: 319–344

Turner , W. et al, (2003) Remote sensing for biodiversity science and conservation. *Trends in Ecology and Evolution* **18**: 306-314

Ustin, S.L. et al. (2004) Using imaging spectroscopy to study ecosystem processes and properties. *Bioscience* **54**: 523-534

Veldkamp, A. & Lambin, E.F. (2001). Predicting land-use change. *Agricultural Ecosystem Environments* **85**:1–6

Venables, W. N. & Ripley, B.D.. (2002). Modern Applied Statistics with S. Fourth Edition edition. Springer, New York.

Verbesselt, J., Hyndman, R., Newnham, G. & Culvenor, D. (2010). Detecting trend and seasonal changes in satellite image time series. *Remote Sensing of Environment* **114**: 106-115.

Verner, J. (1984). The guild concept applied to management of bird populations. *Environmental Management* **8**:1-14

Vesk, P. A. & Westoby, M. (2001). Predicting plant species responses to grazing. *Journal of Applied Ecology* **38**: 897–909

Vetter, S. (2005) Rangelands at equilibrium and non-equilibrium: recent developments in the debate. *Journal of Arid Environments*, **62**, 321–341.

Vogelmann, J.E., Xian, G., Homer, C. & Tolk, B. (2012). Monitoring gradual ecosystem change using Landsat time series analysis: case studies in selected forest and rangeland ecosystems. *Remote Sensing of Environment in press, available online)*

Walker, B. & Abel, N. (2002). Resilient rangelands: adaptations in complex systems. In (Eds. L.H. Gunderson & C. Holling) Panarchy: understanding transformations in human and natural systems. Island Press.

Walker, H.H., Ludwig, D., Holling, C.S. & Peterman, R.M. (1981). Stability of semi-arid savanna grazing systems. *Journal of Ecology* **69**: 473-498

Wallace, J., Behn, G. & Furby, S. (2006). Vegetation condition assessment and monitoring from sequences of satellite imagery. *Ecological Management & Restoration* **7**: S31–S36

Wallace, J.F., Caccetta, P.A. & Kiiveri, H.T. (2004). Recent developments in analysis of spatial and temporal data for landscape qualities and monitoring. *Austral Ecology* **29** (1): 100-107

Ward, D. (2006). *Rangeland ground cover disturbance data package. Technical report version 1.1.* Queensland Government. Environmental Protection Agency and Queensland Parks and Wildlife Service.

Ward, D.P. & Kutt, A.S. (2009). Rangeland biodiversity assessment using fine scale on-ground survey, time series of remotely sensed ground cover and climate data: an Australian savanna case study *Landscape Ecology* **24**: 495-507

Watson, D.M. (2011). A productivity-based explanation for woodland bird declines: poorer soils yield less food. *Emu* **11**1, 10-18

Watson, I.W., Burnside, D.G. & Holme, A. M. (1996). Event –driven or continuous: which is the better model for managers? *Australian Rangeland Journal* **18**: 351-369

Watson, I. W., Thomas, P. W. E. & Fletcher, W. J. (2007). The first assessment, using a rangeland monitoring system, of change in shrub and tree populations across the arid shrublands of Western Australia. *The Rangeland Journal* **29**, 25–37.

Weiers, S., Bock, M., Wissen, M., & Rossner, G. (2003). Mapping and indicator approaches for the assessment of habitats at different scales using remote sensing and GIS methods. *Landscape and Urban Planning* **1007**: 1-23.

Weins, J.A. (1984). On understanding a non-equilibrium world: myth and reality in community patterns and processes. In D.R. Strong, D. Simberloff, L. Abele and A.B. Thistle (Eds.). Ecological communities: Conceptual issues and the evidence. Princeton, NJ: Princeton University Press, pp. 439-458

Wellens, J. (1997). Rangeland vegetation dynamics and moisture availability in Tunisia: An investigation using satellite and meteorological data. *Journal of Biogeography* **24:** 845−855.

Wessels, K. J., Prince, S. D., Malherbe, J., Small, J., Frost, P. E. & Van Zyl, D. (2007). Can human-induced land degradation be distinguished from the effects of rainfall variability? A case study in South Africa. *Journal of Arid Environments* **68:** 271−297.

West, N.E. (1999). Accounting for rangeland resources over entire landscapes. *Sixth International Rangeland Congress Proceedings*. Townsville, Australia. VI International Rangeland Congress Inc., pp. 726-735

Westoby, M. (1980). Elements of a theory of vegetation dynamics in arid rangelands. *Israel Journal of Botany* **28**: 169-194

Westoby, M., Falster, D.S., Moles, A.T., Vesk, P.A. & Wright, I.J., (2002). Plant ecological strategies: some leading dimensions of variation between species. *Annual Review of Ecology and Systematics* **33**: 125-159.

Westoby, M., Walker B.H. & Noy-Meir, I. (1989). Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* **42**: 266-274

Whitford, W.G., De Soyza, A.G., Van Zee, J.W, Herrick, J.E. & Havstad, K.M. (1998). Vegetation, soil and animal indicators of rangeland health. *Environmental Monitoring and Assessment* **51:** 179-200.

Williams, J., Helyar, K.R., Greene, R.S.B. & Hook, R.A. (1993). Soil characteristics and processes critical to the sustainable use of grasslands in arid, semi-arid and seasonally dry environments. Proceedings*, Seventeenth International Grassland Congress,* pp. 1335-1350.

Woinarski, J.Z.C., Andersen, A.N., Churchill, T.B. & Ash, A.J. (2002). Response of ant and terrestrial spider assemblages to pastoral and military land use, and to landscape position, in tropical savanna woodland in northern Australia. *Austral Ecology* **27**: 324-333

Woinarski, J.C.Z. & Ash, A. J. (2002). Responses of vertebrates to pastoralism, military land use and landscape position in an Australian tropical savanna. *Austral Ecology* **27**: 311-323

Woinarski, J.C.Z. & Fisher, A. (2003) Conservation and the maintenance of biodiversity in the rangelands. *The Rangeland Journal* **25** (2):157-172

Woinarski, J.C.Z., McCosker, J.C., Gordon, G., Lawrie, B., James, C., Augusteyn, J., Slater, L., & Danvers, T. (2006). Monitoring change in the vertebrate fauna of central Queensland, Australian, over a period of broad-scale vegetation clearance, 1973-2002. *Wildlife Research* **33**, 263-274

Woinarski, J.C.Z. & Tidemann, S.C. (1991) The bird fauna of a deciduous woodland in wet-dry tropics of Northern Australia. *Wildlife Research* **18**: 479-500

Wu, J. & Loucks, O.L. (1995). From the balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. *Quarterly Review of Biology* **70**: 439-466.

Wulder, M.A., White, J.C., Goward, S.N., Masek, J.G., Irons, J.R., & Herold, M. (2008) Landsat continuity: Issues and opportunities for land cover monitoring. *Remote Sensing of Environment* **112**: 955-969

Yebra, M., Dennison, P.E. Chuvieco, E., Riano, D., Zylstra, P., Hunt, E.R., Danson, F.M., Qi, Y., & Jurdoa, S. (2013). A global review of remote sensing of live fuel moisture content for fire danger assessment: Moving towards operational products. *Remote Sensing of Environment* **136**: 455-468.

Zerger, A., Gibbons, P., Seddon, J., Briggs, S. & Freudenberger, D. (2009). A method for predicting native vegetation condition at regional scales. *Landscape Urban Planning* **91**: 65-77

Zha, Y., Gao, J., Ni, S., Lui, N., Jiang, J., & Wei, Y. (2003) A spectral reflectance-based approach to quantification of grassland cover from Landsat TM imagery. *Remote Sensing of Environment* **87**: 371-375

Appendix A

Bird species list with taxonomic, seasonality, dietary, foraging and habitat group numbers.

Common Name	Taxon Name	Taxonomic group	*Seasonality group*	Dietary group	Foraging group	Habitat group
Horsfield's bronze-cuckoo	*Chrysococcyx basalis*	11	*2*	5	1	1
black-faced cuckoo-shrike	*Coracina novaehollandiae*	29	*1*	1	1	1
white-bellied cuckoo-shrike	*Coracina papuensis*	29	*5*	1	1	1
brown treecreeper	*Climacteris picumnus*	20	*1*	5	1	1
mistletoebird	*Dicaeum hirundinaceum*	38	*1*	4	1	1
restless flycatcher	*Myiagra inquieta*	28	*1*	5	1	1
black-chinned honeyeater	*Melithreptus gularis*	23	*5*	1	1	1
blue-faced honeyeater	*Entomyzon cyanotis*	23	*5*	1	1	1
little friarbird	*Philemon citreogularis*	23	*5*	1	1	1
noisy friarbird	*Philemon corniculatus*	23	*5*	1	1	1
rufous-throated honeyeater	*Conopophila rufogularis*	23	*5*	1	1	1
spiny-cheeked honeyeater	*Acanthagenys rufogularis*	23	*5*	1	1	1
varied sittella	*Daphoenositta chrysoptera*	26	*1*	5	1	1
grey shrike-thrush	*Colluricincla harmonica*	27	*1*	1	1	1
red-browed pardalote	*Pardalotus rubricatus*	22	*5*	5	1	1
spotted pardalote	*Pardalotus punctatus*	22	*1*	5	1	4
striated pardalote	*Pardalotus striatus*	22	*1*	5	1	1
weebill	*Smicrornis brevirostris*	22	*1*	5	1	1
western gerygone	*Gerygone fusca*	22	*5*	5	1	1
white-throated gerygone	*Gerygone olivacea*	22	*1*	5	1	1
yellow thornbill	*Acanthiza nana*	22	*1*	5	1	1
rufous songlark	*Cincloramphus mathewsi*	40	*3*	5	1	4
rainbow lorikeet	*Trichoglossus haematodus*	10	*5*	3	1	1
grey fantail	*Rhipidura fuliginosa*	28	*3*	5	2	1
brown honeyeater	*Lichmera indistincta*	23	*5*	1	2	1
grey-fronted honeyeater	*Lichenostomus plumulus*	23	*5*	1	2	1
singing honeyeater	*Lichenostomus virescens*	23	*5*	1	2	1
striped honeyeater	*Plectorhyncha lanceolata*	23	*5*	1	2	1
white-plumed honeyeater	*Lichenostomus penicillatus*	23	*5*	1	2	1
crested bellbird	*Oreoica gutturalis*	27	*1*	5	2	2
rufous whistler	*Pachycephala rufiventris*	27	*1*	5	2	2
inland thornbill	*Acanthiza apicalis*	22	*1*	5	2	2
speckled warbler	*Chthonicola sagittata*	22	*1*	5	2	3
jacky winter	*Microeca fascinans*	24	*1*	5	2	2
ground cuckoo-shrike	*Coracina maxima*	29	*1*	1	3	1
spotted nightjar	*Eurostopodus argus*	15	*1*	5	3	3
bush stone-curlew	*Burhinus grallarius*	5	*1*	6	3	3
masked lapwing	*Vanellus miles*	42	*5*	1	3	3
common bronzewing	*Phaps chalcoptera*	8	*5*	2	3	2
diamond dove	*Geopelia cuneata*	8	*5*	2	3	2
peaceful dove	*Geopelia striata*	8	*5*	2	3	2
squatter pigeon (southern subspecies)	*Geophaps scripta scripta*	8	*1*	2	3	3
pallid cuckoo	*Cuculus pallidus*	11	*2*	5	3	2
shining bronze-cuckoo	*Chrysococcyx lucidus*	11	*2*	5	3	2
white-winged triller	*Lalage sueurii*	29	*5*	5	3	2
apostlebird	*Struthidea cinerea*	33	*1*	1	3	2

Australian raven	*Corvus coronoides*	32	*1*	1	3	2
little crow	*Corvus bennetti*	32	*1*	1	3	2
red-backed fairy-wren	*Malurus melanocephalus*	21	*1*	5	3	3
variegated fairy-wren	*Malurus lamberti*	21	*1*	5	3	3
crimson chat	*Epthianura tricolor*	44	*5*	5	3	3
Richard's pipit	*Anthus novaeseelandiae*	35	*1*	5	3	3
chestnut-rumped thornbill	*Acanthiza uropygialis*	22	*1*	5	3	2
yellow-rumped thornbill	*Acanthiza chrysorrhoa*	22	*1*	5	3	2
black-throated finch	*Poephila cincta*	36	*5*	2	3	2
diamond firetail	*Stagonopleura guttata*	37	*5*	2	3	2
double-barred finch	*Taeniopygia bichenovii*	36	*5*	1	3	2
plum-headed finch	*Neochmia modesta*	36	*5*	2	3	2
zebra finch	*Taeniopygia guttata*	36	*5*	2	3	2
hooded robin	*Melanodryas cucullata*	24	*1*	5	3	2
red-capped robin	*Petroica goodenovii*	24	*5*	5	3	2
grey-crowned babbler	*Pomatostomus temporalis*	25	*1*	1	3	2
spotted bowerbird	*Chlamydera maculata*	34	*1*	4	3	3
brown songlark	*Cincloramphus cruralis*	40	*3*	5	3	3
galah	*Cacatua roseicapilla*	9	*1*	2	3	2
budgerigar	*Melopsittacus undulatus*	10	*5*	2	3	2
emu	*Dromaius novaehollandiae*	1	*1*	1	3	3
little button-quail	*Turnix velox*	7	*1*	2	3	3
black kite	*Milvus migrans*	3	*5*	6	4	2
black-breasted buzzard	*Hamirostra melanosternon*	3	*1*	6	4	2
brown goshawk	*Accipiter fasciatus*	3	*1*	6	4	2
spotted harrier	*Circus assimilis*	3	*1*	6	4	2
square-tailed kite	*Lophoictinia isura*	3	*5*	6	4	2
wedge-tailed eagle	*Aquila audax*	3	*5*	6	4	2
whistling kite	*Haliastur sphenurus*	3	*5*	6	4	2
Australian hobby	*Falco longipennis*	4	*1*	6	4	2
brown falcon	*Falco berigora*	4	*5*	6	4	2
nankeen kestrel	*Falco cenchroides*	4	*1*	6	4	2
Australian owlet-nightjar	*Aegotheles cristatus*	15	*1*	1	5	2
crested pigeon	*Ocyphaps lophotes*	8	*5*	1	5	2
channel-billed cuckoo	*Scythrops novaehollandiae*	11	*2*	1	5	2
Australian magpie	*Gymnorhina tibicen*	31	*1*	1	5	2
magpie-lark	*Grallina cyanoleuca*	28	*1*	5	5	2
willie wagtail	*Rhipidura leucophrys*	28	*1*	5	5	2
yellow-throated miner	*Manorina flavigula*	23	*1*	1	5	2
cockatiel	*Nymphicus hollandicus*	10	*5*	1	5	2
sulphur-crested cockatoo	*Cacatua galerita*	9	*1*	1	5	2
pale-headed rosella	*Platycercus adscitus*	10	*1*	1	5	2
red-winged parrot	*Aprosmictus erythropterus*	10	*5*	1	5	2
southern boobook	*Ninox novaeseelandiae*	12	*1*	1	5	2
barn owl	*Tyto alba*	13	*5*	1	5	2
rainbow bee-eater	*Merops ornatus*	18	*2*	5	6	4
black-faced woodswallow	*Artamus cinereus*	31	*1*	5	6	1
little woodswallow	*Artamus minor*	31	*1*	5	6	1
masked woodswallow	*Artamus personatus*	31	*5*	5	6	1
white-breasted woodswallow	*Artamus leucorynchus*	31	*1*	5	6	1
white-browed woodswallow	*Artamus superciliosus*	31	*5*	5	6	1
fairy martin	*Hirundo ariel*	39	*1*	5	6	4
tree martin	*Hirundo nigricans*	39	*1*	5	6	1
white-necked heron	*Ardea pacifica*	43	*1*	6	7	2

blue-winged kookaburra	*Dacelo leachii*	17	*1*	1	7	2
forest kingfisher	*Todiramphus macleayii*	17	*1*	1	7	2
laughing kookaburra	*Dacelo novaeguineae*	17	*1*	1	7	2
red-backed kingfisher	*Todiramphus pyrrhopygia*	17	*5*	1	7	2
sacred kingfisher	*Todiramphus sanctus*	17	*2*	1	7	2
grey butcherbird	*Cracticus torquatus*	31	*1*	1	7	2
pied butcherbird	*Cracticus nigrogularis*	31	*1*	1	7	2
Torresian crow	*Corvus orru*	32	*1*	1	7	2

Appendix B

Tables of a) Taxonomic groups, b) seasonality groups, c) dietary groups, d) foraging groups and e) habitat groups

a) Taxonomic groups

Code	Taxonomic groups	Common name
1	*Dromaridae*	emus
2	*Phasianidae*	true quails
3	*Accipitridae*	eagles
4	*Falconidae*	falcons
5	*Burhinidae*	stone-curlews
6	*Otididae*	bustard
7	*Turnicidae*	button-quails
8	*Columbidae*	pigeons & doves
9	*Cacatuidae*	cockatoos
10	*Psittacidae*	parrots
11	*Cuculidae*	cuckoo's
12	*Strigidae*	boobook's
13	*Tytonidae*	barn owl's
14	*Podargidae*	frogmouth's
15	*Aegothelidae*	owlet nightjar's
16	*Apodidae*	swifts
17	*Halyconidae*	tree kingfishers
18	*Meropidae*	bee-eaters
19	*Coraciidae*	dollarbirds
20	*Climacteridae*	treecreepers
21	*Maluridae*	fairy wrens
22	*Pardalotidae*	pardalotes, thornbills, gerygones etc.
23	*Meliphagidae*	honey eaters
24	*Petroicidae*	robins
25	*Pomatostomidae*	babblers
26	*Neosittidae*	sitella's
27	*Pachycephalidae*	whistlers, shrike's etc.
28	*Dicruridae*	flycatchers
29	*Campephagidae*	cuckoo-shrike's, trillers
30	*Oriolidae*	orioles & figbirds
31	*Aratamidae*	woodswallows, magpies
32	*Corvidae*	ravens & crows
33	*Corcoracidae*	choughs & apostlebirds
34	*Ptilonorhynchidae*	bowerbirds
35	*Alaudidae*	larks
36	*Fringillidae*	finches
37	*passeridae*	firetails, weavers
38	*Dicaeidae*	mistletoebird
39	*Hirundinidae*	swallows & martins
40	*Sylviidae*	warblers
41	*Zosteropidae*	white-eyes
42	*Charadriidae*	lapwings

b) Seasonal groups

Code	Seasonal group
1	resident
2	summer immigrant
3	winter immigrant
4	uncertain
5	nomadic

c) Dietary groups

Code	Dietary group
1	generalist
2	granivore
3	nectar feeder
4	frugivore
5	insectivore
6	carnivore

d) Foraging groups

Code	Foraging group
1	canopy
2	mid stratum
3	ground & low shrub
4	raptor
5	general
6	above canopy
7	pounce feeder

e) Habitat groups

Code	Habitat group
1	Feed and nest in trees
2	Feed on ground and nest in trees
3	Feed and nest on ground
4	Feed in trees and nest on ground

Appendix C

List of Plant species and their lifeforms

NAME	Lifeform
44*Abutilon sp.*	PF
142*Acacia coriacea*	S
115*Acacia excelsa*	S
130*Acacia leiocalyx*	S
210*Acacia melleodora*	S
124*Acacia salicina*	S
226*Acacia tennuissima*	S
154*Achyranthes aspera*	AF
7*Alternanthera denticulata*	AF
236*Alternanthera nana*	PF
158*Archidendropsis basaltica*	S
5*Aristida calycina*	PG
163*Aristida calycina var. calycina*	PG
215*Aristida calycina var. praealta*	PG
15*Aristida holathera*	AG
29*Aristida jerichoensis*	PG
224*Austrachloris dichanthioides*	PG
189*Bidens pilosa*	AF
118*Blumea diffusa*	AF
125*Boerhavia pubescens*	PF
40*Bothriochloa ewartiana*	PG
34*Brachiaria piligera*	AG
178*Brachychiton populneus*	S
72*Brunoniella australis*	PF
53*Bulbostylis barbata*	AF
26*Camptacra barbata*	PF
111*Canthium oleifolium*	S
192*Capparis mitchellii*	S
90*Carissa ovata*	S
136*Cassytha filiformis*	S
91*Cenchrus ciliaris*	PG
67*Centipeda minima*	AF
3*Chaemasyce drummondii*	AF
123*Chamaecrista absus*	AF
80*Cheilanthes sieberi*	PF
162*Chloris divaricata*	PG
144*Chrysocephalum appiculata*	PF
4*Chrysopogon fallax*	PG
180*Clerodendrum floribundum*	S
110*Corchorus aestuens*	PF
120*Corymbia dallachiana*	S
102*Crotolaria montana*	AF
131*Cymbopogon bombycinus*	PG
51*Cymbopogon refractus*	PG
112*Cyperus fulvus*	PF

126	*Dactylotenium radulans*	AG
93	*Desmodium macrocarpum*	PF
229	*Desmodium varians*	PF
89	*Dianella longifolia*	PF
16	*Digitaria ammophila*	PG
128	*Digitaria bicornis*	PG
41	*Digitaria brownii*	PG
86	*Digitaria ciliaris*	PG
97	*Dipteracanthus australasicus*	AF
241	*Echinochloa colona*	AG
193	*Einadia nutans*	PF
50	*Enneapogon lindleyanus*	AG
6	*Enneapogon virens*	AG
105	*Enteropogon acicularis*	PG
107	*Eragrostis 1*	PG
194	*Eragrostis elongata*	PG
52	*Eragrostis lacunaria*	PG
173	*Eragrostis leptocarpa*	AG
46	*Eragrostis sororia*	AG
129	*Eragrostis spartinoides*	PG
82	*Eragrostis speciosa*	PG
139	*Eremophila longifolia*	S
22	*Eriachne mucronata*	PG
165	*Eriochloa procera*	AG
85	*Erythroxylum australe*	S
27	*Eucalyptus melanophloia*	S
49	*Eulalia aurea*	PG
140	*Euphorbia tannensis*	PF
13	*Evolvulus alsinoides*	PF
14	*Fimbristylis dichotoma*	PF
2	*Galactia tenuifolia*	PF
155	*Geijera parviflora*	S
66	*Glycine clandestina*	PF
211	*Glycine falcata*	PF
10	*Glycine tomentella*	PF
227	*Gomphrena celosioides*	AF
77	*Goodenia glabra*	PF
204	*Goodenia hederacea*	PF
57	*Goodenia hirsuta*	PF
121	*Grewia retusifolia*	S
213	*Heliotropium brachygyne*	AF
232	*Heliotropium cunninghamii*	AF
38	*Heliotropium tanythrix*	AF
87	*Heteropogon contortus*	PG
216	*Hibiscus burtonii*	PF
237	*Hibiscus sturtii*	PF
32	*Hybanthus enneaspermus*	PF
31	*Indigofera colutea*	AF
113	*Indigofera hirsuta*	AF
37	*Indigofera linifolia*	AF
117	*Indigofera linnaei*	PF
138	*Indigofera pratensis*	PF
198	*Ipomoea coptica*	AF
168	*Ipomoea gracilis*	PF

132	*Jasminum didymum*	S
166	*Lasiantha nipans*	S
134	*Marsdenia rostrata*	PF
240	*Marsdenia viridiflora*	PF
104	*Maytenus cunninghamii*	S
70	*Melhania oblongifolia*	S
33	*Melinus repens*	PG
108	*Murdannia graminea*	PF
179	*Myoporum desertii*	S
234	*Oldenlandia corymbosa*	AF
21	*Oldenlandia mitrasacmoides*	AF
160	*Opuntia stricta*	S
11	*Panicum effusum*	PG
135	*Parsonsia lanceolata*	S
64	*Paspalidium gracile*	PG
221	*Paspalidium rarum*	AG
79	*Peripleura hispidula*	AF
48	*Perotis rara*	AG
143	*Petalostigma pubescens*	S
145	*Phyllanthus fuernrohrii*	PF
74	*Phyllanthus simplex*	PF
152	*Polycarpaea corymbosa*	AF
24	*Polygala linariifolia*	PF
116	*Polymeria pusilla*	PF
150	*Portulaca bicolor*	AF
83	*Portulaca oleracea*	AF
30	*Portulaca pilosa*	AF
122	*Pterocaulon serrulatum*	PF
61	*Pterocaulon sphacelatum*	PF
230	*Ptilotus sp*	PF
54	*Rhynchosia minima*	PF
106	*Rostellularia adscendens*	PF
222	*Salsola cali*	AF
9	*Sauropus trachyspermus*	PG
20	*Schizachyrium fragile*	AG
114	*Sclerolaena muricata*	PF
157	*Sclerolaena viuricata*	PF
119	*Sehima nervosum*	PG
223	*Senna artemisoides subsp filiformis*	S
8	*Sida atherophora*	PF
62	*Sida cunninghamii*	PF
203	*Sida fibulifera*	PF
127	*Sida sp 1*	PF
181	*Sida sp 2*	PF
238	*Sida sp 3 (Silvery)*	PF
68	*Solanum ellipticum*	PF
25	*Spermacoce brachystema*	AF
206	*Sporobolus australasicus*	AG
196	*Sporobolus caroli*	PG
100	*Stylidium eglandulosum*	PF
101	*Stylidium eriorrhizum*	PF
156	*Stylosanthes scabra*	S
88	*Tephrosia brachyodon*	AF
133	*Tephrosia flagellaris*	AF

47	*Tephrosia leptoclada*	AF
141	*Tephrosia sp*	PF
188	*Themeda avenacea*	PG
12	*Themeda triandra*	PG
19	*Tragus australianus*	AG
36	*Tricoryne elatoir*	PF
1	*Triodia pungens*	PG
55	*Tripogon loliiformis*	PG
153	*Urochloa*	AG
98	*Ventilago viminalis*	S
169	*Vernonia averea*	PF
78	*Vernonia cinerea*	PF
76	*Wedelia spilanthoides*	PF
243	*Xenostegia tridentata*	PF
43	*Zornia muriculata*	PF