UNIVERSITY OF SOUTHERN QUEENSLAND

# Regrowth vegetation in production landscapes: Comparisons between remnant and regrowth Brigalow (*Acacia harpophylla*) communities, Southern Queensland.

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# ABSTRACT

Endangered Brigalow (*Acacia harpophylla* F. Muell. ex Benth.) dominated ecosystems have been extensively cleared and modified for agricultural production throughout the Southern Brigalow Belt Bioregion of southern Queensland, Australia. Intensification of the agriculture in the region has resulted in significant fragmentation and disturbance of natural vegetation remnants. Rehabilitating previously cleared vegetation (regrowth) is an important conservation alternative where there is inadequate cover of intact remnants, that aims to ensure the persistence of Brigalow communities throughout highly modified landscapes. While legislation is now in place to restrict clearing of remnant and Brigalow regrowth in southern Queensland, little is known of the functioning and dynamics of regrowth communities in comparison to remnant vegetation. This research investigates patterns in floristic composition, stand structure and microenvironment of Brigalow remnants and a number of regrowth communities differing in time since clearing.

Thirty-eight sites in the Darling Downs region, southern Queensland, were sampled encompassing Brigalow remnant (two treatments: ‘Remnant’ (sites with an absence of clearing but continue to experience small disturbances such as grazing) and ‘Reserve’ (sites with an absence of disturbance within state and national reserves)) and different aged Brigalow regrowth (<20 years; 20 – 30 years; 30 – 40 years; >40 years). Floristic composition and projected cover of strata were recorded in a 500m$^2$ quadrat at each site. Patch and surrounding landscape context (spatial) environmental data were also determined for each site. Soil chemical data (including soil, pH, organic content, cation exchange capacity and phosphorus and nitrogen concentrations) were determined from bulked soil samples. Landscape spatial data was derived from remotely sensed imagery. Un-weighted paired group arithmetic averaging (UPGMA), non-metric multidimensional scaling (nMDS) ordination, analysis of similarity (ANOSIM) and analysis of variance (ANOVA) were used to identify patterns in composition, functional group richness and stand structure between remnant and regrowth sites. Canonical Correspondence Analysis (CCA), Kruskall Wallis correlations and linear step-wise regression were used to identify potential environmental drivers of vegetation responses (Appendix I).

Multivariate analyses (UPGMA and nMDS) showed general gradients from recent regrowth through older stages of regrowth to remnant and reserve sites for both floristic composition and stand structure data. There were no differences between remnant and old regrowth (>40 years) in terms of stand structure (ANOSIM, $p<0.05$) and total species richness (ANOVA, $p<0.05$), although differences were evident amongst other treatments. Remnant and Reserve sites were compositionally different to older regrowth (ANOSIM, $p<0.05$). No significant invasion of exotic species was observed within any treatment type (ANOVA, $p>0.05$).

These results suggest that stand structure in regrowth vegetation returns to that similar to remnant vegetation within 40 years of initial clearing, predominantly driven by the development of the overstorey trees and woody understorey species. However, Brigalow remnant and old regrowth (>40y) remain different in terms of floristic composition. The significance of this finding is discussed in relation to Alternate Stable State Theory.

The proportion of surrounding vegetation and landuse type, as well as patch factors, such as grazing intensity and soil properties, were highly correlated (CCA) with patterns in composition and stand structure observed across the Brigalow treatments. Regrowth sites exhibited higher proportion of regrowth vegetation as well as cropping and grazing in the surrounding landscape compared to remnant sites. Soil nutrients, such as phosphorus, potassium and cation exchange capacity, and stem density and grazing intensity were also higher within regrowth sites. These findings suggest an environmental influence may be responsible for the differential patterns in composition observed between remnant and regrowth communities in the region.

It is concluded that while structure of Brigalow regrowth returns to that of remnant Brigalow, the distinct floristic species composition of older regrowth may indicate a longer time period is needed for equivalence or a possible alternate stable state in these highly modified agricultural landscapes. These results are significant for the management of highly fragmented communities in production landscapes, where management actions may need to be directed towards a more active approach towards regeneration. To ensure the long-term persistence of Brigalow communities, further investigation of the development trajectory of old Brigalow regrowth is required.

# DECLARATION

I certify that the work reported in this thesis is entirely my own effort, except where otherwise acknowledged. I also certify that the work is original and has not been previously submitted for assessment in any other course of study at this or any other institution.

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Signature of candidate

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Date

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# GLOSSARY

**Agro-ecosystem** – A natural ecosystem or landscape that has been modified to incorporate agricultural production, supporting both cropping and livestock grazing as well as natural vegetation and wildlife.

**Disturbance** – A change in the physical state, composition or processes that operate within a specific ecosystem.

**Ecosystem Functioning** – The collective interactions and processes that operate between biological and physical components within an ecosystem

**Ecosystem Health** – A characteristic of complex natural environments relating to the condition and state of individual species, populations and their interactions.

**Ecosystem Integrity** – see ecosystem health

**Ecosystem Processes** – Physical, chemical and biological reactions which take place such as decomposition and primary production that are essential for the cycling and transfer of energy within a system.

**Ecosystem Properties** – Specific component or components within an ecosystem that are responsible for maintaining and/or creating particular physical characteristics or processes.

**Ecosystem Services** – The direct and indirect human benefits associated with ecosystem attributes and processes.

**Exotic or Introduced Species** – Species not considered endemic or native to an area and have been deliberately of accidently introduced by humans.

**Fragmentation** – The division or break up of a larger intact patch of vegetation increasing isolation.

**High Value Regrowth** – Areas of regrowth vegetation of an *endangered*, *of concern, least concern* Regional Ecosystem or has not been cleared since 31 December 1989 as recognised under the *Vegetation Management and Other Legislation Amendment Act 2009*.

**Regrowth Vegetation** – Areas of natural vegetation that have experienced some form of mechanical disturbance of clearing and have since regenerated to some degree, as recognised by Queensland’s Regrowth Vegetation Mapping 2006.

**Remnant Vegetation** – Natural ecosystems that have no, or very little, evidence of clearing as recognised by Queensland’s Regional Ecosystem Mapping 2003.

**Resilience** – A term relating to the ability of an ecosystem to maintain ecosystem functioning and persist following a physical disturbance.

# Chapter 1: General Introduction and Literature Review

## *1.1 Overview*

Humans have been the dominant drivers of widespread landscape transformation and modification, affecting a significant proportion of Earth's ecosystems at a variety of scales (Vitousek *et al.*, 1997a; Vitousek *et al.*, 1997b; Chapin *et al.*, 2000; Seabrook *et al.*, 2006). The influence of anthropogenic activities on the natural environment plays a critical role in controlling the type and rate of change these systems experience, particularly within production landscapes (Tscharntke *et al.*, 2005; Hobbs *et al.*, 2006). The expansion and intensification of agricultural production has resulted in the modification and fragmentation of many natural ecosystems, affecting ecological processes, ecosystem functioning and ultimately, the ability for an ecosystem to persist in the landscape (Vitousek *et al.*, 1997b; Altieri, 1999).

In terrestrial landscapes, the fragmentation of natural habitat, creating patches of vegetation with reduced connectivity, coupled with additional forms of land degradation, impacts on biodiversity (Saunders *et al.*, 1991; Walker *et al.*, 1999). The management practices within agroecosystems continue to further degrade the condition of remnant vegetation in the landscape (Prober and Smith, 2009). The application of fertilisers and pesticides to increase production yields, impacts both on- and off-site communities, altering soil structure, composition and disrupting soil biota interactions important in nutrient cycling (Vitousek *et al.*, 1997a; Barrios, 2007). Compaction of soil, erosion and salinity associated with livestock grazing also inhibits healthy root growth and water infiltration, affecting the successful recruitment and survival of plant species (Yates *et al.*, 2000). The continued fragmentation of natural ecosystems for agricultural expansion (Prober and Smith, 2009) has resulted in increased levels of isolation between remnant patches in the landscape, affecting natural disturbance regimes, ecological processes and introducing a variety of new perturbations (Saunders *et al.*, 1991; Walker and Salt, 2006).

The amount of physical disturbance an ecosystem can sustain and continue to persist within a landscape, known as ecological resilience, is important within ecological conservation (Gunderson, 2000; Walker and Salt, 2006). Retaining ecosystem resilience assists in maintaining ecological biodiversity and also social and economic components, such as goods and services, within a landscape. However, as resilience declines, it can take progressively smaller disturbances to push an ecosystem into a different state, also known as an alternate state, differing in structure and ecosystem functioning (Scheffer *et al.*, 2001; Walker *et al.*, 2009). The recognition of alternative equilibrium states and what constitutes a “desirable” state enables conservation managers to identify areas requiring restoration in order to maintain some level of heterogeneity and biodiversity across the landscape (Walker and Salt, 2006).

Significant habitat loss worldwide, associated with increased agricultural and urban development and management intensity, has reduced the extent and isolation of natural remnant ecosystems and increased areas of regrowth and degraded vegetation (Brown and Lugo, 1990; Fahrig, 2003). Growing concerns have arisen in relation to the persistence and resilience of natural ecosystems within these fragmented landscapes to withstand continued human disturbances (Walker, 1995; Young, 2000).

The recognition of global environmental threats, such as climate change, has suggested a potential shift in the perception of production management, from one predominantly focussed on optimising production efficiency and output, opting for one of improved sustainable practices and management of resilience in uncertain environments (Fahrig, 2001; Ludwig *et al.*, 2001; Folke *et al.*, 2004; Barrios, 2007). With insufficient remnant vegetation remaining in many landscapes to achieve long term conservation targets, managers have acknowledged that some form of restoration is required (Young, 2000; Bowen *et al.*, 2007).

Australian woodlands have been greatly affected by agricultural practices despite a relatively short history of European settlement, with many communities experiencing issues such as crown dieback and woody thickening (Fensham *et al.*, 2005; Lindenmayer and Fischer, 2007). In order to maintain long-term biodiversity and ensure the persistence of many endangered ecosystems within highly fragmented landscapes, protection and passive restoration of degraded and regrowth vegetation, areas previously cleared of vegetation, has been employed along with the protection of remnant vegetation (McAlpine *et al.*, 2002a). However, the condition and status of regrowth patches, in comparison to remnant vegetation, remains relatively unknown and pose as a vital component in the future persistence of many threatened and endangered communities (Bowen *et al.*, 2007).

This study investigates the floristic composition and stand structure of Brigalow remnant and regrowth communities in South-east Queensland in order to identify patterns in ecosystem function and vegetation response to disturbance regimes and the landscape context. This study will address the following general questions:

- Are there differences in floristic composition and stand structure between remnant and regrowth forests of different aged structure?

- Are environmental habitat factors, surrounding land-use and landscape connectivity factors correlated with the stand structure and composition of Brigalow communities?

## *1.2 Biodiversity and Ecosystem Processes*

Land use practices, resulting in widespread habitat clearing, fragmentation and overexploitation of natural resources, has threatened global biological diversity and disrupted essential functioning and processes within many ecosystems, resulting in the extinction and endangerment of thousands of species (Chapin *et al.*, 1997; Chapin *et al.*, 2000; Hobbs *et al.*, 2006). Ecosystem function refers to the habitat, biological or system properties or the processes that occur within ecosystems (Costanza *et al.*, 1997). Ecosystem processes, such as productivity and plant recruitment, are important for determining the rates of nutrient cycling and energy transformations through trophic interactions, essential for continued persistence and preservation of ecosystem health, also known as ecosystem integrity (Woodley *et al.*, 1993; Schulze and Mooney, 1994). Diversity within ecosystem functioning is fundamental in increasing the efficiency of resource acquisition and accommodating the requirements of species that have strong ecosystem effects (Chapin *et al.*, 1997).

Ecosystem functioning is also invariably linked to the allocation of ecosystem services (Costanza *et al.*, 1997). The regulation of environmental variables, such as climate and disease, and supporting services, comprising soil formation and fertility, are important, particularly within agricultural landscapes (Millennium Ecosystem Assessment, 2005; Costanza *et al.*, 2007).

The degree to which changes in ecosystem functioning affect ecosystem properties, particularly resilience and the magnitude of ecosystem change following disturbance, known as resistance, has become increasingly relevant as anthropogenic activities intensify across landscapes (Tilman *et al.*, 1997; Loreau *et al.*, 2001; Sugden, 2001). The disruption of key processes, such as nutrient cycling, within a system can result in the presence of unwanted ecosystem effects, threatening both individual species and functional groups within a landscape (Walker, 1992; Sinclair and Byrom, 2006). Negative impacts to diversity and ecosystem processes on a local scale can also have flow-on effects, impacting surrounding areas, initiating reduced patch health and widespread degradation (Chapin *et al.*, 1997; Sinclair and Byrom, 2006). The ability of an ecosystem to absorb these effects and sustain ecological processes is vital in the continued functioning of the system in the landscape.

### 1.2.1 Ecological Resilience and Alternative Stable States

Ecological resilience is an emergent property of ecosystems, highly influenced by the self-organisation of an ecosystem over time (Gunderson, 2000). Resilience is defined as a measure of an ecosystems' ability to absorb disturbances in order to persist within the landscape and maintain essentially the same function, structure and composition (Walker *et al.*, 2004a). Resilience is an important concept within land management, ecosystem modelling and conservation (DeAngelis, 1980; Walker *et al.*, 1999). All systems experience some form of gradual change over time, such as climate fluctuations and nutrient loading (Scheffer *et al.*, 2001). However, severe shock events and disturbances, both natural and anthropogenic, can cause notable changes in structural or compositional characteristics, ultimately shifting the equilibrium of the system towards an alternate stable state (Scheffer *et al.*, 2001).

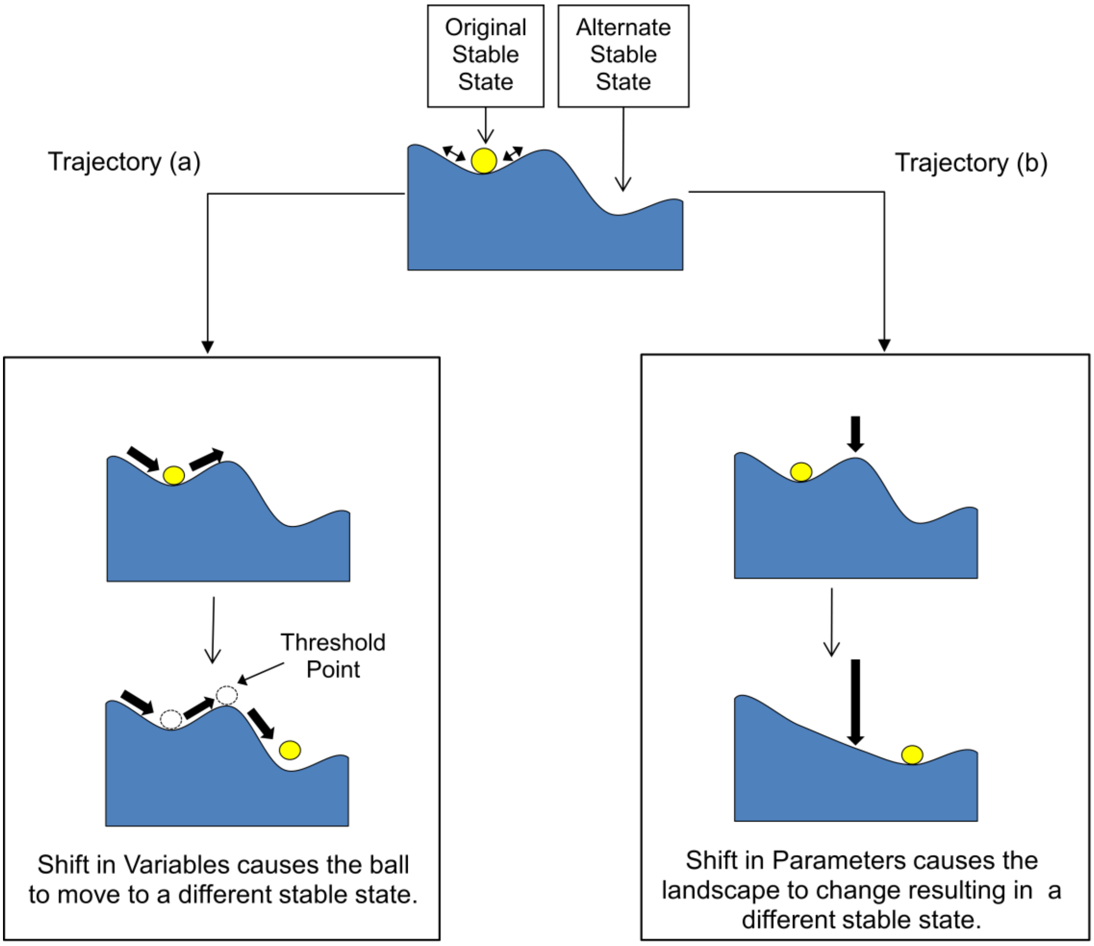
A succinct definition of a "*stable state*" is difficult to determine as each system functions differently from others (Ives and Carpenter, 2007). The stability of an ecosystem can be dependent on numerous factors with no easily definable, universal description (Ives and Carpenter, 2007). Similarly, ecosystems are never "*stable*" in the sense that they do not change and stay at a fixed point in time (Scheffer and Carpenter, 2003). Stability is best described as populations fluctuating around some trend or stable average over time (Scheffer and Carpenter, 2003). The Intermediate Disturbance Hypothesis (Connell, 1978), for example, is based on the notion that at intermediate levels of disturbance, the diversity of species within an ecosystem remains at its highest (Molino and Sabatier, 2001). The continued persistence of these perturbations works as a mechanism leading to relatively stable species coexistence for long periods (Roxburgh *et al.*, 2004). What is categorised as "intermediate" disturbance can be difficult to identify and may be different from system to system, relying on dominant species occurrences and discrete disturbance events (Hobbs and Huenneke, 1992).

Disturbance plays an important role in ecosystem functioning, shaping the physical and behavioural adaptations of individual species (Hobbs & Huenneke, 1992). A natural disturbance can be described as an event or series of events that disrupt existing relationships between organisms and the environment (Gunderson and Holling, 2002). The nature of disturbance regimes produces a mosaic of spatial and temporal variation, contributing to the creation of a heterogenous landscape (Moloney and Levin, 1996).

There are a variety of dimensions to disturbance regimes that influence the response observed within a community. Factors such as magnitude and intensity of disturbances are vital in determining the disruption incurred and the time a system may require in returning to a particular equilibrium (Palmer *et al.*, 1997). Other factors, including frequency and duration in respect to spatial and temporal scales, are also crucial determinants in the regeneration process of a system (Moloney and Levin, 1996; Palmer *et al.*, 1997).

Ecological resilience is centred on the assumption of multiple or alternate stable states of a system (Peterson, 1998; Gunderson, 2000). This term also identifies the level or amount of perturbation a system can withstand before a shift occurs within the system's self-organisation (Gunderson, 2000). Ecosystem thresholds are particular levels of magnitude of ecosystem processes or disturbance which if exceeded can mark sudden changes in feedbacks and responding state variables (Folke *et al.*, 2004). Shocks or disturbances can drive a system beyond its ecological thresholds, causing a divergence in the community trajectory resulting in one of many potential alternate steady states, differing in ecosystem structure and function (Gunderson, 2000; Walker and Salt, 2006).

The recognition of alternate stable states also acknowledges that in some instances, disturbance can shift the trajectory of a successional community, resulting in a different stable state compared with that prior to disturbance (Beisner *et al.*, 2003). There are numerous mechanisms that dictate the stable state of an ecosystem, including the combined densities and interactions of biotic and abiotic components (Ives and Carpenter, 2007). The principles of Alternate Stable State Theory argue that the state of a community is dependent on a group of dynamic state variables, such as population densities and species composition, that are determined by a set of parameters, including migration, emigration birth and death rates (Beisner *et al.*, 2003). The shift between stable states can occur via two pathways, either a change in the parameters, such as birth and death rates, that influence the state variables or a direct change in state variables resulting from a large disturbance (Beisner *et al.*, 2003). This relationship is best described through the threshold models, or "Ball in Cup" diagrams (Figure 1.1). The shift between stable states can occur via two pathways (Figure 1.1), either a change in the parameters (Trajectory b), such as birth and death rates, that influence the state variables, or a direct change in state variables resulting from a large disturbance (Trajectory a) (Beisner *et al.*, 2003).



Adapted from Beisner *et al.* (2003)

**Figure 1.1. Ball in Cup Models identifying the two types of disturbance that can result in a change of stable states.** Small movements away from the original state may occur but be insufficient to result in an alternate state, however when a large disturbance occurs, two trajectories may potentially occur: Trajectory (a) where changes to ecosystem variables, such as species composition, push the state past its threshold point(s) forcing the system into an alternate state; or Trajectory (b) where a change to the parameters causes the system to progress towards a different state.

Changes in community stable states have been observed within a number of studies encompassing both terrestrial and marine biomes (e.g. (Carpenter *et al.*, 1999; Van Langevelde *et al.*, 2003; Ibelings *et al.*, 2007). Cropp & Gabric (2002) identified a tendency of various frequently disturbed aquatic ecosystems to develop towards a state that optimizes resilience. This progression was dependent on numerous imposing factors from the surrounding environment, including nutrient and resource availability, as well as the genetic potential of biological inhabitants (Cropp and

Gabric, 2002). Similarly, Prober *et al.* (2008) identified two potential alternate stable states within temperate grassy woodlands in south-eastern Australia. Disturbance from direct grazing and surrounding agricultural management was found to alter productivity within these communities replacing native grasses, dominated by *Themeda australis*, to a community dominated by exotic annuals (Prober *et al.*, 2008).

Determining the resilience and state of a particular system can be difficult, typically dependent on the landscape context and change over time (Carpenter *et al.*, 2005). Directly assessing resilience involves the recognition of ecosystem thresholds whereby exceeding these boundaries, forces the community into an alternate state (Carpenter *et al.*, 2005). However, specific threshold parameters are rarely identified prior to a crossing of these thresholds, resulting in changes to community components associated with a shift to an alternate stable state (Carpenter *et al.*, 1999; Peterson *et al.*, 2003). Within shallow lake ecosystems, for instance, clear-water regimes are dependent on vast beds of aquatic plants to stabilise sediments and reduce the cycling of phosphorus (Folke *et al.*, 2004). An increase in phosphorus inputs can result in eutrophication and algal blooms, decreasing the abundance of aquatic plants and increasing turbidity (Folke *et al.*, 2004). While the reduction of phosphorus levels can reverse some of the damages imposed, increased turbidity can continue to inhibit the development of benthic vegetation (Folke *et al.*, 2004).

Positive feedback controls between the consumers in an ecosystem and limiting resources are considered to be a principle factor underlying catastrophic ecosystem shifts (Rietkerk *et al.*, 2009). Identifying clear and effective factors, or indicators, within a particular ecosystem is crucial within studies evaluating ecological resilience (Walker & Salt, 2006; Rietkerk *et al.*, 2009).

### 1.2.2 Resilience Surrogates

The recognition of alternate stable states can be determined using resilience surrogates, or indicators (Carpenter *et al.*, 2005). In contrast to threshold identification, resilience surrogates indirectly assess ecosystem components and function, and attempt to infer the relative resilience of a system (Carpenter *et al.*, 2005; Walker *et al.*, 2009). Measuring ecosystem factors, such as functional species diversity and soil properties, provide an indication of the ecosystem functioning and response to change, that can be consequently linked with a system's persistence within a landscape (Walker *et al.*, 1999). Relationships between surrogates and resilience can be highly dynamic and complex, suggesting the use of a suite of surrogates to measure general resilience of a system (Carpenter *et al.*, 2005).

The successful implementation of surrogates is dependent on a clear link with the theoretical concepts of resilience (Carpenter *et al.*, 2005). Walker (1999), for instance, identifies functional group diversity in relation to responses to change and functional contribution, and that they are part of the basis for maintaining ecosystem resilience. The functional group diversity and the response in a system can be vital to managing ecosystems (Carpenter *et al.*, 2005). In general, ecosystems consist of dominant species, in high abundance, and minor species, occupying the tail-end of the distribution (Walker *et al.*, 1999). The tail-end species represent a small proportion of plant biomass and cover and, while they may be similar to dominant species in relation to contributing to ecosystem function, they can differ in tolerance and response to disturbances (Walker *et al.*, 1999). The more variety of species response types, or response diversity, the wider the range of conditions that can sustain functioning and increase the capacity of a community to absorb perturbations (Walker and Salt, 2006). As a result, the abundance of minor species represent a "*reservoir of resilience*" that maintains the persistence of natural function in light of continued or future disturbance and change (Walker *et al.*, 1999).

Other surrogates, such as sampling biomass for ecosystem function and carbon storage and assessing community structure, can also be incorporated to assess the different aspects of a system's resilience and determine the relative state of an ecosystem (Carpenter *et al.*, 2005; Walker *et al.*, 2009). Social-ecological systems is a comprehensive approach developed to assess general resilience, incorporating the social, economic and ecological dimensions of a system, assessing ability to continue to provide goods and services (Walker *et al.*, 2009). However, in the context of this study only the ecological aspects will be assessed.

### 1.2.2 Measuring Resilience and Shifts in Trajectory

The ability to measure general ecological resilience of agroecosystems is important in maintaining both biodiversity and production values across the landscape (Walker *et al.*, 2009). However, the intensification of agriculture in recent years has reduced biodiversity favouring a few select pastoral and cropping species, limiting the variety of species responses to disturbance and ultimately, the ecosystems' resilience in landscape (Altieri, 1999; Fischer *et al.*, 2006). Agricultural practices, including grazing and clearing regimes, have significantly affected the stability of many natural ecosystems (Anderies *et al.*, 2002). Van de Koppel *et al.*, (1997) found at intermediate levels of grazing, multiple state trajectories are possible depending on continued grazing pressures and soil water concentrations.  Herbivory saturation past a certain threshold can result in increased soil water evaporation, leading to a catastrophic shift towards a barren state (Van De Koppel *et al.*, 1997).

Similarly, van Auken (2000), found increased grazing and a reduction in fire frequency to be significant drivers in determining persistent states in semi-arid grasslands of North America.  The grasslands are dominated by a variety of native grass species, such as *Andropogon gerardii* (Bunch Grass) and *Sorghastrum nutans* (Indian Grass); however, following intense grazing and a reduction in fire occurrences, grass cover had significantly declined. As a result of the reduction in grass cover, woody encroachment by *Prosopis* sp. (Mesquite) and *Larrea tridentata* (Creosotebush) had occurred, ultimately transforming the landscape into shrublands (Van Auken, 2000).

The stability domain of a community represents the magnitude of a disturbance a system can endure before a shift in stable state occurs (Holling, 1973 cited in Folke *et al.*, 2004).   The adaptive capacity consists of the ecosystem properties that reflect the alteration in the stability domain of a landscape (Gunderson, 2000).  Prior to human expansion and development, properties that determined a system’s capacity to change, influenced its stability at a relatively slow rate (Gunderson, 2000).  Over time, anthropogenic activities have reduced the environments’ capacity to manage change, causing unpredictable ecosystem shifts and changes in the ability to generate ecosystem services (Folke *et al.*, 2004).  The shifts observed within ecosystems usually correspond to significant catastrophic changes in community structure and function and failure to identify these events can have detrimental consequences, particularly in relation to restoration costs (Carpenter *et al.*, 1999; Beisner *et al.*, 2003; Peterson *et al.*, 2003).

Essential for the management for ecosystem resilience is the detection of drivers influencing regime shifts, identifying where potentials thresholds lie and maintaining particular aspects of a system that safeguard resilience (Walker and Salt, 2006). While species within functional groups are responsible for specific ecosystem functions, individual species responses to disturbances can differ (Walker & Salt, 2006). Therefore, the more variety of species response types, or response diversity, the wider the range of conditions that can sustain functioning and increase the capacity of a group to absorb perturbations (Walker and Salt, 2006). However, the intensification of agriculture in recent years has reduced biodiversity, favouring a few select pastoral and cropping species, ultimately limiting the ecosystems' resilience in landscape.

## *1.3 Anthropogenic Driven Disturbances in Production Landscapes*

The methods of modern agriculture have resulted in the extensive simplification of vast areas of land, substituting native plant and animal species for a small number of cultivated monocultures and domestic livestock (Altieri, 1999; Tilman, 1999). While the expansion of new agricultural land, both in Australia and globally, has decreased in recent years, the management within pre-existing production landscape has intensified in order to meet the demands of the world's growing population (Tilman, 1999). Tilman (1999b) calculated an approximate doubling in cereal production (rice, wheat, corn and barley) based on statistics from the Food and Agriculture Organisation between 1961 and 1996. However, the need to meet food and fibre requirements of the human population has come at the expense of natural ecosystem functioning, altering ecological processes and biodiversity components (Tilman, 1999; Lunt and Spooner, 2005). The clearing and fragmentation of land has been one of the most catastrophic and widespread changes within landscapes and continues to impact ecosystem function and species viability (Vitousek *et al.*, 1997b)

### 1.3.1 Fragmentation in Production Landscapes of Australia

Fragmentation has had a significant impact on natural landscapes and continues to be a key threat to biodiversity (Saunders *et al.*, 1991). Fragmentation is the process of dividing and segmenting the landscape, creating patches smaller in size than that of the contiguous tract of habitat (Bender *et al.*, 1998; Ross *et al.*, 2002). This process creates islands of vegetation in a differing, often inhospitable matrix, affecting the connectivity and ecological functioning throughout the landscape (Saunders *et al.*, 1991; Prober and Thiele, 1995).

Agricultural development within natural landscapes is a chief cause of habitat loss and fragmentation, with current expansion and intensification contributing to further division and hardening of the matrix (Spooner *et al.*, 2004a). Despite continued clearing in Australia, there has been an estimated average decrease of 1/3 (182 600ha) in clearing rates since 1990 (Change, 2009). However, the substantial history of land cover change in Australia has resulted in many landscapes significantly fragmented, with many containing less than 10% of original (pre-European) vegetation remaining (McIntyre and Hobbs, 1999). For example, Fensham (1998) estimates grasslands dominated by Queensland Bluegrass (*Dichantheum sericeum*) on the Darling Downs in Queensland have been reduced to almost 1% of its original area.  Woodland ecosystems throughout Australia have also been heavily cleared in order to accommodate various forms of agriculture and remains one of the greatest threats to biodiversity (Wilson *et al.*, 2002).  Distinguishable from forests by their relative height, wide spacing and low projected foliage cover (10 – 30%), woodland areas have been widely utilised for their fertile soils and grazing potential (Lindenmayer *et al.*, 2005).  In Victoria, for example, Plains Grassy Woodlands and Wimmera Mallee Woodlands were once some of the most common ecosystem types found (Lunt and Bennett, 2000).  However, following European settlement significant land cover change has resulted in less than 3% remaining in state's northern and western plains (Lunt and Bennett, 2000).  Similarly, Poplar Box (*Eucalyptus populnea*) and Mountain Coolibah (*E. orgadophila*) have been reduced in some landscapes to 6% and 28% respectively (Fensham, 1998).  Within these highly fragmented landscapes, factors such as patch geometry and position, become important in maintaining habitat health, quality and long term persistence (Saunders *et al.*, 1991; McIntyre and Hobbs, 1999).

The process of fragmentation affects habitats over a variety of spatial scales and biological organisation, altering species assemblages and ecological processes (Lindenmayer and Fischer, 2007).  Fragmentation not only disrupts natural disturbance patterns within ecosystems, but also introduces novel disturbance regimes associated with activities in the surrounding matrix (Turner, 1989). Saunders *et al*. (1991) recognises two primary effects of fragmentation that can influence remnant patches in the landscape: changes to microclimate and isolation.

Changes to microclimate can occur due to increased infiltration of radiation, wind and altered hydrological cycles, resulting from the removal or modification of vegetation in the surrounding matrix (Saunders *et al.*, 1991). The isolation of remnants also occurs, depending on the degree of modification in the matrix and the distance from surrounding patches (Saunders *et al.*, 1991). Within production landscapes, the extent to which these physical and biogeographical changes occur within a community is influenced by the magnitude and interaction of human disturbances, both within an ecosystem and from the surrounding matrix (Saunders *et al.*, 1991).

Exogenous disturbance, to varying intensities, associated with agricultural practices, such as livestock grazing and distribution of fertilisers and other chemicals, can degrade and modify remaining habitat within fragmented landscapes (Prober and Thiele, 1995). Livestock grazing, for example, has had a significant impact within woodland communities, affecting floral and faunal assemblages and ecosystem functioning ((Yates *et al.*, 2000). Studies have indicated the presence of livestock grazing can lead to changes in community organization and alter understorey structure and composition of remaining vegetation (Yates *et al.*, 2000; Jansen and Robertson, 2001). As grazing pressure increases, changes in vegetation characteristics can occur, usually decreasing the abundance of native plants and supporting the establishment of exotic species (Prober and Thiele, 1995; Yates *et al.*, 2000). In severely exploited landscapes, dryland salinity can also eventuate, reducing the ability of plants to survive and establish (Freudenberger and Brooker, 2004). Lechmere-Oertal *et al*. (2005), for instance, examined succulent Spekboom thickets in South Africa and found reduced soil fertility, water infiltration and retention as a result of unsustainable grazing practices. Overgrazing, resulted in a transition from a predominantly dense perennial shrub layer to an ephemeral grass and forb layer (Lechmere-Oertel *et al.*, 2005). This combination of impacts resulted in a relatively infertile state, considered dysfunctional in comparison to similar ecosystems in the surrounding area (Lechmere-Oertel *et al.*, 2005). The introduction of agricultural systems has also altered fire regimes in many woodland ecosystems affecting species abundance and inhibiting reproductive cycles of numerous native species (Attiwill & Wilson, 2004).

The continued intensification of agricultural lands has introduced a variety of novel perturbations to naturally functioning ecosystems. Technological advances enabling increased yields has been in part due to the increased use of fertilizers, high in phosphorus and nitrogen, restoring fertility and nutrients to degraded and depauperate soils (Tilman, 1999). Soil biota and their associated interactions are critical in nutrient cycling, pest control and the regulation of soil structure, necessary in the establishment and maintenance of entire ecosystem functioning (Barrios, 2007). Increases in soil nitrogen can result in a variety of *in situ* and *ex situ* effects. The conversion of nitrogen into nitrates and nitrites by soil bacteria can result in water pollution via leaching into underground water bodies (Tilman, 1999). Excess nitrogen and phosphorus accumulation in surrounding waterways can cause water acidification, eurotrophication and hypoxia in downstream communities (Vitousek *et al.*, 1997a; Tilman, 1999). De-nitrification also produces nitrous oxide, a greenhouse gas, impacting atmospheric concentrations and ultimately influencing the global carbon cycle (Vitousek *et al.*, 1997a).

Other local impacts of agricultural practices can trigger a cascade of environmental problems, whereby one factor, such as erosion, can result in further degradation, such as nutrient deficiency and recruitment of vegetation, reducing the ability of species to re-establish themselves (Asner *et al.*, 2004). Intensive grazing, for example, can significantly reduce the vegetative cover which can result in erosion of the topsoil by wind and rain (Asner *et al.*, 2004). In the last 50 years, loss of topsoil has doubled, occurring at a rate of 760 million tonnes $yr^{-1}$ worldwide (Walker and Salt, 2006). Continued erosion of topsoil can also result in increased desertification and salinisation (Matson *et al.*, 1997; Asner *et al.*, 2004). Compaction of soil as a result of hooved livestock further decreases water infiltration and changes soil morphology (Yates *et al.*, 2000). Belsky and Blumenthal (1997) analysed a number of studies within Ponderosa Pine communities and found that livestock grazing resulted in the compaction of soil and loss of important soil nutrients. Over time these changes, coupled with changed fire regimes, altered the composition, density and structure of the vegetation (Belsky and Blumenthal, 1997). Consequently, the communities became less resilient to natural disturbance regimes, such as fire and disease, reducing the long-term community viability (Belsky and Blumenthal, 1997).

## *1.4 Altered Ecosystem Functioning within Production Landscapes*

Significant changes to natural ecosystem functioning have occurred within production landscapes since European settlement (Vitousek *et al.*, 1997a). Tree clearing continues to be common practice within agricultural landscapes in an attempt to maximise pasture production and monetary gains (Sangha *et al.*, 2005). The removal of large trees reduces ground litter and competition for light, facilitating grass production in the understorey (McIvor, 2001). Many paddocks are described as containing isolated large spreading trees nearing senescence, with very little or no recruitment due to pressures associated with chronic grazing (Vesk and Mac Nally, 2006). While increases in grass production are beneficial for grazing, overstorey clearing can affect ecosystem processes and natural understorey composition, potentially causing changes in the dominant plant species (McIvor, 2001). Decline and mortality of native trees can limit successful recruitment processes that require a consistent seed supply to enable seedling establishment during favourable years (Close *et al.*, 2008). Saunders *et al.* (2003), for example, found reduced rates of recruitment within *Eucalyptus salmonophloia* woodlands that are continuously grazed.

The modification of natural fire regimes has also impacted the functioning within production landscapes. Fire has played a critical role in the life histories and adaptations exhibited by many of Australia's floral species (Fensham, 1997). Woody thickening throughout many woodland ecosystems has resulted from the alteration of fire regimes, particularly reduced fire frequency (Asner *et al.*, 2004; Fensham, 2008). Woody thickening, also known as woody encroachment, is the process of increased woody plant density, significantly decreasing grazing potential of the land (Asner *et al.*, 2004; Price and Morgan, 2008). Fensham & Fairfax (2005) found thickening of Gidgee (*Acacia cambagei*) to not only impact livestock grazing, but was also seen to be encroaching into nearby Mitchell grasslands in Longreach, Queensland. Other factors believed to be contributing to this phenomena are overgrazing and livestock trampling of herbaceous cover favouring woody seedlings, and increased atmospheric carbon dioxide, benefiting C3 (woody) plant growth, and nitrogen pollution (Asner *et al.*, 2004; Eamus and Palmer, 2007). Thinning required to restore the structure and composition of vegetation within a common range of variability is a contentious issue, with considerations such as imperfect background knowledge, labour shortages and the divergent views of landholders hampering decision making (Fensham, 2008).

From a landscape perspective, disturbance regimes are rarely operating independently from one another, instead they tend to work in synergy, at a variety of spatial and temporal scales (Seabrook *et al.*, 2007). Recognising the ability of an ecosystem to sustain functioning in a variety of situations in relation to surrounding land-uses is essential within conservation initiatives and ultimately, long-term persistence in the landscape (Spooner *et al.*, 2004b).

Within areas of significant historical clearing, a large proportion of existing vegetation consists of highly disturbed vegetation and regrowth patches (Wilson *et al.*, 2002). The regeneration and development of secondary vegetation can be difficult to calculate as these processes can be significantly dependent on land-use history, incorporating intensity, duration and type, as well as other landscape attributes (Bowen *et al.*, 2007).

Protection and preservation of undisturbed habitat patches, or remnant vegetation, has been highly emphasised within conservation initiatives (e.g. Native Vegetation and Biodiversity Program 1994 -2009 (Land and Water Australia, 2009). The distribution of remnant vegetation in the landscape partially reflects the historical events, settlement and development of a region (Wilson *et al.*, 2002). Specifically, factors associated with environmental conditions, rainfall patterns, soil fertility and other underlying economic issues, coupled with social and political trends and institutions, highly influence land management (Wilson *et al.*, 2002). However, recognition of vulnerable communities as a conservation priority tend to be an afterthought following the crossing of thresholds whereby it can be costly or impossible to reverse, particularly in highly developed landscapes (Wilson *et al.*, 2002; Carpenter *et al.*, 2008). This can result in scarce and insufficient representation of remnant vegetation within the landscape (Hansen and Rotella, 2002; Wilson *et al.*, 2002). The condition and health of many remaining woodland habitats has also experienced a gradual decline throughout many agricultural landscapes in Australia (Lindenmayer *et al.*, 2005). During the mid to late 1960s, the premature death and decline of tree health, known as rural dieback, increased significantly throughout a variety of woodland types (Lindenmayer *et al.*, 2005). This reduction in crown cover has been attributed to a variety of causes, including increased insect herbivory, salinity, soil acidification, drought, changes to soil structure and direct mechanical damage from agricultural livestock and machinery (Lindenmayer *et al.*, 2005). The modification of fire regimes has also been accredited to a decline in mature tree health within agricultural landscapes (Lindenmayer *et al.*, 2005). In order to implement effective management, the monitoring and assessment of all vegetation, both remnant and non- remnant communities, remains a primary goal in landscape and biodiversity initiatives (Young, 2000; Wilson *et al.*, 2002).

A shifting focus has been observed in recent years from the preservation of predominantly intact ecosystems to restoring disturbed and degraded communities

(Suding *et al.*, 2004). The management of both remnant and regrowth vegetation is important, particularly within highly disturbed and fragmented landscapes not only for regional and global biodiversity, but also for safeguarding ecosystem function and services (McAlpine *et al.*, 2002a; Bowen *et al.*, 2009b). The creation of ‘Viable Networks’ is one example that attempts to incorporate remnant and regrowth vegetation in a whole landscape approach (McAlpine *et al.*, 2002a). Viable Networks are described as areas of vegetation exhibiting high connectivity, ecological functioning, are capable of self-generation yet can still be maintained within areas subjected to threatening disturbance regimes (McAlpine *et al.*, 2002a).

For example, Gardner *et al*. (2007) conducted a study on amphibian and reptile diversity in secondary forests in the Amazonian forests and found while primary forests contained higher levels of biodiversity, regrowth communities were also of significant value to maintaining diversity on a landscape scale. A similar pattern was observed by Dunn (2004) where older aged regrowth exhibited an increase in bird and ant species richness, contributing to the overall biodiversity at the landscape scale.

Restoration of disturbed or cleared vegetation is commonly undertaken via replanting and seeding of plant species endemic to the target remnant community (Dorrough and Moxham, 2005). However, significant costs associated with labour and ongoing monitoring, and mixed success in numerous restoration projects ultimately reduces its effectiveness over large areas and landscapes (Dorrough and Moxham, 2005). In contrast, natural regeneration relies on the colonisation of woody plants to disturbed or cleared patches from surrounding remnant patches (Vesk and Dorrough, 2006).

Consequently, natural regeneration has been adopted as a cost-effective alternative to replanting and seeding (Dorrough and Moxham, 2005; Vesk and Dorrough, 2006).

While passively regenerated regrowth has occurred within many areas around the world, particularly associated with agricultural abandonment in Central and Southern America, there is limited empirical evidence available in relation to its overall effectiveness in floristic community development (Aide *et al.*, 2000; Young, 2000; Ruiz-Jaén and Aide, 2005; Bowen *et al.*, 2007). Despite extended periods of time, the composition and structure of regrowth is rarely similar to remnant vegetation, with a number of studies indicating the return of only subsets of ecosystem components, such as richness of fauna or some aspects of stand structure (Turner *et al.*, 1997; Lamb *et al.*, 2005). Martin *et al*. (2004) found stand structural characteristics, including stem density, canopy height and tree size distributions, to resemble remnant tropical forest after 40 years in abandoned agricultural land in central Dominican Republic. However, floristic diversity still remained significantly different to undisturbed forests (Martin *et al.*, 2004). Other studies within agricultural landscapes have also found that former agricultural land readily returns to a semi natural state (Wilcox, 1998; Walker *et al.*, 2004b). O'Connor *et al*. (2005) examined abandoned grasslands in South Africa and found while the general structure was comparable to other less disturbed grasslands, the composition and abundance of species still remained significantly different.

In Australia, the value of regrowth vegetation has also been recognised within fragmented landscapes. Bowen *et al*. (2009b), found the proportion of older Brigalow regrowth (>30 years) in the surrounding landscapes in southern

Queensland to have a strong positive relationship on the species richness of woodland dependent birds. Similarly Michael *et al*. (2011) found temperate Eucalypt woodland regrowth in southern New South Wales provided important habitat for numerous reptile species. While numerous studies have investigated the value of regrowth vegetation to faunal movements and potential habitat (e.g. Gascon *et al*., 1999; Neilan *et al*., 2006) few long-term studies have been conducted on the recovery of flora following human disturbance within Australian woodlands.

As Alternate Stable State Theory suggests, the recovery process does not always necessarily suggest a return to identical pre-clearing levels (Beisner *et al.*, 2003). Recovery and succession following a significant disturbance can be influenced by various environmental factors, including individual species biology, competition, stochasticity and the interaction between biotic and abiotic components (Guariguata and Ostertag, 2001).

Further uncertainty in relation to the future impacts of climate change and changes in land use on ecosystem health in Australia, highlights the importance of determining whether regrowth vegetation, under passive regeneration, can attain the same biodiversity value as remnant vegetation in production landscapes. With total landscape restoration within production landscapes unlikely, the success of restoration initiatives for long term ecosystem viability and functioning relies heavily on the complementarity of remaining ecosystem components within the landscape mosaic (Walker *et al.*, 2004b; Lamb *et al.*, 2005). Continued assessment of regrowth communities is therefore required in order to determine the general condition and state if the adoption of natural regeneration as a conservation strategy is to be successful.  However, the status of many regrowth communities remains relative unknown resulting in little indication of the biological value of these communities, compared with relatively undisturbed remnant communities, and biodiversity in the greater landscape  (Nicholson *et al.*, 2009).

## ***1.5 Study Questions and Hypotheses***

Despite evidence of some ecosystem processes and biodiversity values returning to regrowth communities (Martin *et al.*, 2004; Bowen *et al.*, 2009b; Michael *et al.*, 2011), it does not ensure the return of structure and function to that similar of remnant vegetation, or the on-going persistence of these services. Many regrowth communities have been significantly transformed as a result of anthropogenic disturbance, resulting in alternate stable states (Van De Koppel *et al.*, 1997; Carpenter *et al.*, 1999; Van Auken, 2000; Folke *et al.*, 2004).

Current management in Queensland has protected ‘high value’ regrowth, including Brigalow, from further clearing in an attempt to increase the extent of native remnant communities in production landscapes (*Vegetation Management and Other Legislation Amendment Act 2009*). However, little is known of the condition, status and general resilience of regrowth patches of Brigalow communities in southern Queensland, and how they compare to less disturbed remnant vegetation.  If the natural restoration of regrowth is intended to increase the extent of Brigalow communities, investigation is necessary to determine its relative similarity and contribution to its remnant counterparts in light of historical disturbance regimes.  This study will investigate the structural and functional components of remnant and a number of different aged regrowth

Brigalow communities (Regrowth >40 years; Regrowth 30-40 years; Regrowth 20-30 years and Regrowth <20 years) and determine the relative correlation of patch factors and landscape factors to identify major drivers.

The following specific hypotheses will be investigated:

- Floristic composition, stand structure, functional group richness differs between Brigalow remnant, old regrowth (>40 years), intermediate regrowth (30-40y and 20-30y) and recent regrowth (<20y) communities.
- Environmental, land-use and landscape connectivity factors are possible drivers of floristic composition and stand structure of remnant Brigalow vegetation and different aged regrowth.

## *1.6 Thesis Overview*

In order to test the hypotheses, floristic surveys, assessments of stand structure and condition and spatial features were sampled within remnant and a range of regrowth age classes. Chapter 2 provides a general description of the study area as well as a history of agricultural and conservation management in the area. Patterns in floristic composition and stand structure between remnant and regrowth vegetation is investigated in Chapter 3. Total species richness and functional group richness based on life form, perenniality and origin were also assessed across the sites. In Chapter 4 the effects of the landscape context, landscape connectivity (spatial factors) and local patch factors, mainly soil nutrients and general sites condition are also tested in regards to vegetative patterns observed between remnant and regrowth Brigalow. Finally, Chapter 5 analyses the key findings within the studies and discusses the significance in relation to Alternate Stable State Theory and the general resilience of these communities within agricultural landscapes. The management implications and potential future investigations are also discussed.

# Chapter 2: Brigalow Vegetation, Study Area and Design

## *2.1 Study Region*

### 2.1.1 Brigalow Belt Bioregion

The Brigalow Belt Bioregion, spans over 41 million hectares throughout much of inland and eastern Queensland and northern New South Wales (Figure 2.1 map inset), and has experienced one of the most rapid landscape transformations ever documented (Thackway and Cresswell, 1995; Chandler *et al.*, 2007). Due to the highly fertile soils, the region has experienced significant agricultural and regional development over the past 100 years (Johnson, 1984b).

The Southern Brigalow Belt Bioregion occupies over 27 million hectares, 22 million hectares of which falls within Queensland (Dunn and Sahukar, 2003). This region has been significantly modified since European occupation and remains a major sector for agricultural production, occupying the upper catchment of the Murray Darling Basin (Biggs *et al.*, 2005). The region consists predominantly of Quarternary and Tertiary sediments that have experienced significant erosion and aridity, coupled with a history of indigenous firestick practices (Johnson, 1984b). The southern bioregion also occupies a climate transition zone, straddling both the tropical northern climates and temperate south (Johnson, 1984b).

The combination of environmental factors, such as soil and climate, has created a mosaic of vegetations types across the bioregion (Johnson, 1984b). Acacia communities characterise the bioregion, consisting of over 120 species, including *Acacia harpophylla*, *Acacia cambagei* and *A. argyrodendron* (Johnson, 1984b). Eucalypt open forests and woodlands, including those dominated by *E. melanophloia, E. crebra. E. populnea, E. tereticornis* also occupy a vast majority of the south east Queensland region, particularly along the Great Dividing Range, as well as *Callitris* and *Casuarina* open forests, grasslands and semi evergreen vine thickets (Johnson, 1984b). While the acacia-dominated ecosystems are considered relatively biodiverse, much of the diversity in the bioregion is contributed to eucalypt woodlands and semi-evergreen vine thickets (Johnson, 1984a).

### 2.1.2 Moonie and Weir River Catchments

The study was conducted, in western Darling Downs, Southern Queensland, situated in the Southern Brigalow Belt Bioregion. The study area covers approximately 3 600 $km^2$ around the townships of Moonie (27˚ 43’ 01.2”S, 150˚ 22’ 15.6”E) and Westmar (27˚ 55’ 09.4”S, 149˚ 42’ 59.2”E) within the Moonie and Weir River Catchments (Figure 2.1).

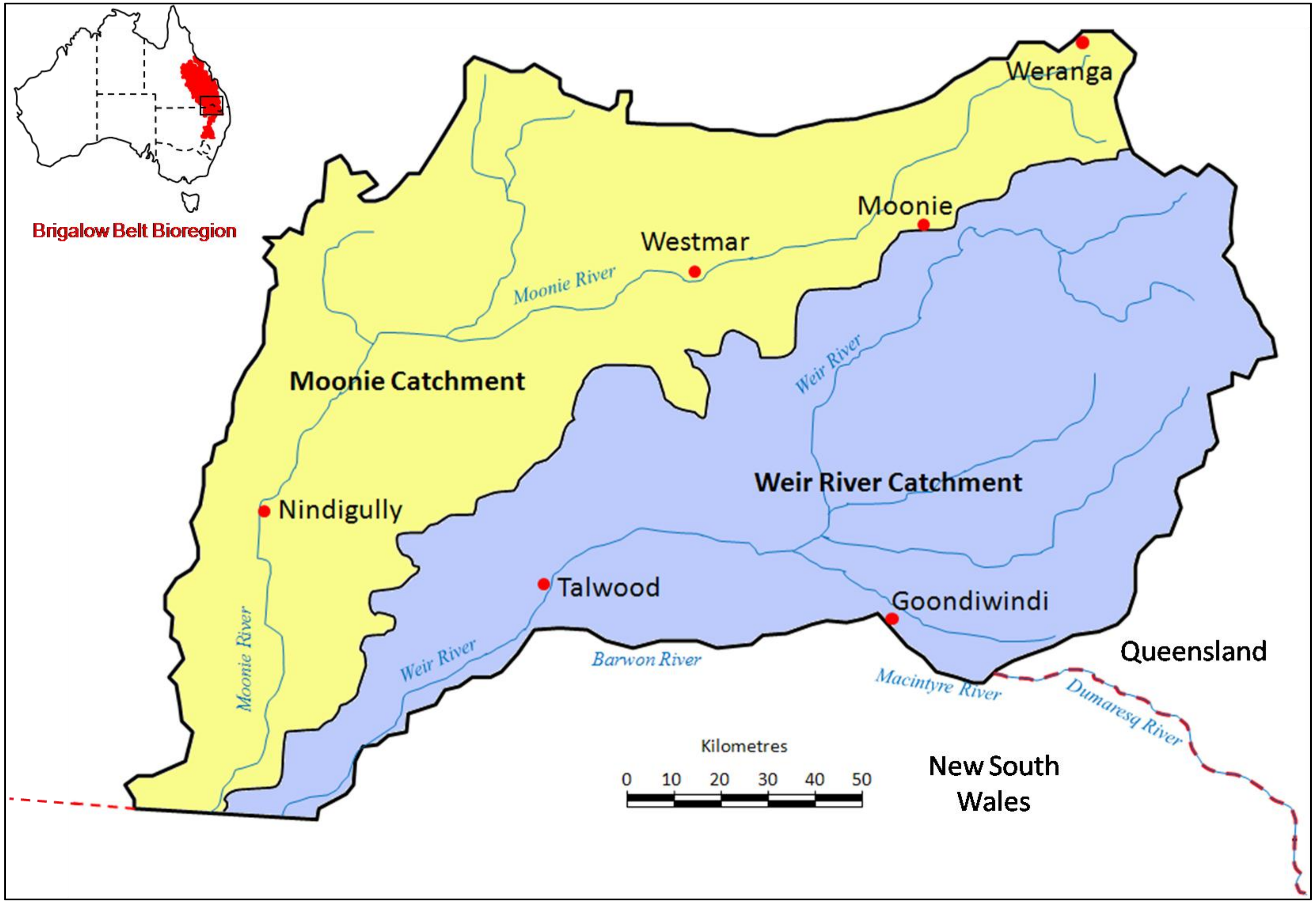


**Figure 2.1.** Map of the Moonie and Weir River Catchments in south eastern Queensland, Australia.

The region is characterised by a predominantly sub-tropical climate, experiencing highly episodic summer dominant rainfall, averaging approximately 618mm annually (Biggs *et al.*, 2005); Bureau of Meteorology, 2010). The average potential evaporation for the region is greater than precipitation, resulting in frequent short droughts (3-4 months) as well as periodic extended droughts (Coaldrake, 1973; Biggs *et al.*, 2005). The study area is approximately 240 m – 260 m above sea level and has an average temperature ranging between 7˚C to 25˚C in the winter months and 17˚C to 35˚C in summer (Bureau of Meteorology, 2010). The area primarily slopes to the west and south (Biggs *et al.*, 2005).

The soils in the study area consist predominantly of grey Vertisols, in particular deep gilgaied clays, along with some areas of brown Sodosols that overlay Tertiary sandstones and other conglomerates (Biggs *et al.*, 2005; DERM, 2009c). Gilgaied clays occur on gently undulating landscapes and are usually very coarsely structured, up to 3 to 4.5m deep, and can experience heavy cracking in relation to soil moisture availability (Coaldrake, 1973; Isbell, 2002). These soils are typically alkaline on the surface with a strongly acidic subsoil ($pH < 4$), depauperate in phosphorus but can contain high salinity in the upper profile (Coaldrake, 1973). Gilgai are small land depressions developed as a result of shrinking and swelling associated with the wetting and drying of the heavy clay soils (DERM, 2009d). This form of microrelief can reach up to 150 cm in depth, usually averaging between 0.6 m and 0.9 m, and can contain different texture, fertility and water moisture on the top, slope and depression (Russell *et al.*, 1967; Coaldrake, 1973).

## *2.2 Vegetation and Management*

### 2.2.1 Brigalow Communities

While Brigalow and other Acacia communities characterize Australia's Brigalow Belt, there are a number of other vegetation types that occupy the area. Ironbark (*Eucalyptus crebra* F. Muell. and *E. melanophloia* F. Muell.) woodlands tend to dominate the hillsides and escarpments, while Poplar Box (*E. populnea* F. Muell.) generally occurs along drainage lines and on sandy alluvial soils and River Red Gums (*Eucalyptus camaldulensis* Dehnh.) along rivers and streams (Biggs *et al.*, 2005). Brigalow (*Acacia harpophylla*) and Belah (*Casuarina cristata* Miq.) communities dominate the extensive clay plains, also referred to as the "Brigalow Claysheet" or "Tara Plains" (Biggs *et al.*, 2005).

There are a number of different types of acacia dominated communities found within the Southern Brigalow Belt Bioregion, usually determined by changes in soils and climatic factors (Johnson, 1984b). *Acacia harpophylla* (Brigalow) communities tend to occupy mid and lower slopes and plains with typical deep cracking tertiary clay soils (Johnson, 1984b). The term 'Brigalow', originated from the Aboriginal people and was used to identify areas of scrub containing, not only *A. harpophylla,* but several other species of Acacia (Bailey, 1984). Nowadays, the term is used to encompass both entire vegetative communities containing *A. harpophylla,* as well as referring to the particularly species (Bailey, 1984).

The Brigalow species is a leguminous tree that can reach up to 24 m in height and can become monospecific in the tree layer (Russell *et al.*, 1967). The species retain its leaves all year round, even during drought, and can grow rapidly when in favourable conditions, particularly following precipitation events (Tunstall and Connor, 1981). Flowering generally occurs from April to October; however, it does not occur every year, and is highly dependent on rainfall conditions (Department of Sustainability, Energy, Water, Population & Communities, 2010). Benson *et al*. (2006, cited in Department of Sustainability, Energy, Water, Population & Communities, 2010) described flowering within the species as highly irregular, noting that some individuals may experience heavy flowering once over a 30 year period. Unlike other acacia species, Brigalow seeds have soft seed coats, and consequently, have a relatively low seed set and rarely stay viable over a year (Coaldrake, 1971; Department of Sustainability, Energy, Water, Population & Communities, 2010). Prolific reproduction via root-suckers can also occur, particularly following mechanical disturbance, where there can be damage to the root or stems, with suckers capable of developing faster than seedling recruits (Coaldrake, 1971; Department of Sustainability, Energy, Water, Population & Communities, 2010). The extent of suckering within the population is dependent on seasonal conditions, the size and age of the individual and the amount of damage caused by mechanical disturbance (Department of Sustainability, energy, Water, Population & Communities, 2010).

While Brigalow can dominate, other species frequently co-dominate, including *Casuarina cristata* and various eucalypt species such as *Eucalyptus cambageana, E, microcarpa* and *E. populnea* (Johnson and Burrows, 1994).

The tree canopy of the community, which can reach up to 20m in height, with an average height of 10 to 15m with some Eucalypt and Belah emergents (Johnson, 1984b; DERM, 2009b). The understorey tends to be dominated by *Geijera parviflora* (Wilga) and *Eremophila mitchellii* (False Sandalwood); however, other species, including *Alectryon diversifolius, Carrissa ovata, Citrus glauca* and *Apophyllum anomalum* can occur (Johnson, 1984b; Biggs *et al.*, 2005; DERM, 2009b). The groundcover is relatively sparse to open and frequently contains large levels of leaf litter biomass, attributed to the aridity inhibited natural organic breakdown (Russell *et al.*, 1967; DERM, 2009b). Grass species found within the ground layer include *Paspalidium caespitosum, P. gracile, Cymbopogon refractus* and *Ancistrachne uncinulata* (Johnson, 1984b; DERM, 2009c). Numerous forb species, such as *Brunoniella australis, Enchylaena tomentosa, Rhagodia nutans* and *Abutilon oxycarpum,* can also be present in the ground layer (Johnson, 1984b; DERM, 2009b).

Remaining Brigalow communities also provide habitat for numerous faunal species, some of which are recognised as threatened or endangered under both state and national legislation (McAlpine *et al.*, 2002a; Department of the Environment, Water, Heritage & Arts, 2010). The region contains one of the highest number of *endangered* and *of concern* regional ecosystems over a large extent (Wilson *et al.*, 2002). There are an estimated 30 threatened plant and animal species that can be found within Brigalow communities, with three known plant species found exclusively within Brigalow remnant and regrowth vegetation (Dwyer *et al.*, 2009).

### 2.2.2 Historic Management of Brigalow

#### *2.2.2.1 History of Vegetation Clearing*

Prior to European settlement, firestick farming was the main anthropogenic disturbance encountered across Australian landscapes, implemented by nomadic indigenous tribes across different seasons (Walsh, 1984). While the use of fire has been shown to influence the species and type of vegetation found across landscapes, significant speculation still exists in relation to the degree to which it was employed within particular regions of Queensland (Fensham, 1997).

According to Isbell (1957, cited in Coaldrake, 1973), prior to initial clearing, around 55% of the area, encompassing the study region and some surrounding landscapes, was made up of Brigalow Belah woodlands, with the remaining comprising Eucalypt and other communities and some small grasslands. Since European settlement in the 1840s, over 61% of remnant woody vegetation in the Brigalow Belt Bioregion has been cleared in order to utilise fertile, nitrogen rich soils for agricultural cropping and grazing (Robertson *et al.*, 1997; Seabrook *et al.*, 2007). A large proportion of clearing in the bioregion has occurred on freehold land, due to the productive nature, substantial coverage and uncontrolled management of the area (Fensham *et al.*, 1998).

Early clearing practices began in the late 1880s, employing ring-barking and axing, in order to make way for pastures (Lloyd, 1984). However, clearing decreased during the 1920s and 1930s with the peak infestation of Prickly Pear (*Opuntia* sp.) throughout much of eastern Australia (Lloyd, 1984; Freeman, 1992). Following the success of the Cactoblastis moth (*Cactoblastis cactorum*) large areas of previously termed "*useless scrub*", densely covered in prickly pear, was once more cleared for production (Lloyd, 1984; Freeman, 1992). According to Skerman (1953, cited in Lloyd, 1984), approximately 2.8 million hectares of Brigalow had been cleared by axe, fire and intense sheep grazing up to 1953; although Skerman (1953) also noted there was significant regrowth in the landscape.

Clearing rates increased throughout the 1950s with the establishment of more efficient, mechanical methods, such as blade ploughing and heavy machinery, allowing developers to readily clear large expanses of Brigalow within a relatively short time period (Anderson, 1984; Seabrook *et al.*, 2006). Clearing rates were also exacerbated following the end of World War II as demand for release of leasehold land for returning soldiers grew (Lloyd, 1984).

The introduction of the Queensland *Brigalow and Other Lands Development Act 1962* institutionalised the clearing of Brigalow communities for the expansion of the pastoral industry (Seabrook *et al.*, 2006). Other development schemes by the Commonwealth and Queensland governments (e.g. Fitzroy Basin Land Development Scheme 1968) also encouraged the clearing of Brigalow land for development and agriculture in return for cheap land and reduced loans (Anderson, 1984; Fensham and Fairfax, 2003). Fensham & Fairfax (2003) estimate the clearing rate in central Queensland at around 1% per annum between 1956 to 1993, reducing the percentage of uncleared vegetation from approximately 93.4% to 55.8%.

However, while clearing rates increased with the use of heavy machinery, the regrowth issue continued to frustrate developers.  The “woody weed” problem is a result of the modifications by land developers on the original Brigalow communities (Anderson, 1984).  Due to the root suckering ability of the species, areas cleared commonly experienced substantial thickening regrowth in the following years (Seabrook *et al.*, 2006).  Johnson (1997) assessed Brigalow regrowth stands in central Queensland and found some dense clonal stands reaching up to 25 000 stems/ha.  Numerous attempts were made to control the regrowth with differing degrees of success (Anderson, 1984; Johnson, 1997).

Continual periodic blade ploughing, successful at removing remaining root structures, was found to be the most effective at controlling regrowth; however, this was only suitable for areas to be used for cropping (Anderson, 1984). Regrowth within grazing pastures proved to be more troublesome, with deliberate burning only providing a temporary solution with minimal effect on stem densities (Anderson, 1984). Intense sheep grazing was found to be useful in impeding thick regrowth; however, following the wool price slump (1969 - 1972) sheep grazing decreased significantly, particularly in the southern Brigalow Belt (Lloyd, 1984; Scanlan *et al.*, 1991).  Aerial spraying of Brigalow regrowth was also trialled during the 1950s and 1960s, although the overall success was highly dependent on the age of the regrowth and the available soil moisture (Anderson, 1984).   More recently, soil applied herbicides, such as Tebuthiuron, have been implemented to assist in the control of woody regrowth thickening (Scanlan *et al.*, 1991).

#### *2.2.2.2 Changes in Land-use*

Since early European settlement in the 1840s, mainly round Tara, the Moonie River Catchment and surrounding areas have undergone significant land-use changes (Lloyd, 1984). Originally utilised for the region's growing beef and wool industries, dairy farming increased during the 1920s and 1930 until its decline in the 1960s following prolonged periods of drought (Biggs *et al.*, 2005). In order to increase stocking rates, Mitchell (*Astrebla* spp.) and Flinders (*Iseilema* spp.) grasses were introduced to many areas, particularly in the Goondiwindi, Taroom and Dalby districts (Lloyd, 1984). Following the collapse of the wool and beef markets in the 1950s and 1970s, respectively, farmers turned to wheat cropping because of its high economic return (Biggs *et al.*, 2005). During the 1970s, summer cropping, particularly sorghum, also increased throughout the region (Biggs *et al.*, 2005).

Prior to the 1950s, grazing and limited cropping (mainly cotton) was prevalent in the Southern Brigalow Belt; however, a significant increase in development in summer cash cropping was observed following 1950 (Lloyd, 1984). While native pastures were found to support livestock, increased stocking rates could be obtained by introducing exotic grass species. *Chloris gayana* (Rhodes Grass) and *Panicum maximum* (Green Panic) were commonly sown in recently cleared Brigalow land and areas previously colonised by prickly pear (Lloyd, 1984). The introduction of *Cenchrus ciliaris* (Buffel Grass) was not observed in the southern region until 1965, following a prolonged period of drought (Lloyd, 1984). Between 1961 and 1976, the area sown to exotic grasses increased from approximately 335, 000 ha to 2, 146, 000 ha before stabilising at around 2, 000, 000 ha in the following years, associated with an increase in cropping (Lloyd, 1984).

Sheep grazing significantly decreased between the early 1960s and 1980s as the cattle industry began to develop following a slump in wool prices (Lloyd, 1984). Despite some policies and schemes in place to entice farmers back to sheep grazing during the 1970's, it was perceived as uneconomic under the labour and financial crises at the time (Lloyd, 1984). Farmers were instead resorting to summer cropping due to the high prices for wheat and sorghum (Lloyd, 1984). The cattle industry and dryland cropping continue to be the primary type of landuse in the Southern Brigalow Belt (Seabrook *et al.*, 2006).

### 2.2.3 Current Land-use

Little change in land-use in the study region has occurred over recent years, with the majority of land dedicated to pastoral and cropping agriculture (Biggs *et al.*, 2005). While grazing is currently the most significant land use in Moonie and Weir River catchments, 79% and 61%, respectively, the combination of summer and winter cropping accounts for up to 65% of the agricultural income of the region (Biggs *et al.*, 2005; Bowen *et al.*, 2009b) (Table 2.1). The year to date (June 2010) estimated gross value of agricultural production in the area is over $409 m (Statistics, 2010). Some areas of forestry also exist within both catchments totally around 19% of land-use (Biggs *et al.*, 2005). While no significant changes in land-use are expected in the future, attributed to limited water availability, woody vegetation cover is expected to increase slightly due to natural regeneration in grazing land by 2020 (Biggs *et al.*, 2005).

Apart from a number of state forests and national parks, the majority of remnant Brigalow vegetation exists as linear strips along roadside and fence-lines (Bowen *et al.*, 2009a). Erringibba National Park (900 ha) and Southwood National Park (7,120 ha) are two of the larger nature reserves in the area that contain Brigalow Communities.

**Table 2.1.** Landuse by catchment based on 1999 Landuse cover data

| | **Moonie Catchment** | | **Weir River Catchment** | |
|---|---|---|---|---|
| | **Hectares** | **%** | **Hectares** | **%** |
| **Grazing** | 1 137 841 | (78.9%) | 940 893 | (61.0%) |
| **Dryland Cropping** | 233 329 | (16.2%) | 256 305 | (16.6%) |
| **Irrigated Cropping** | 4 087 | (0.3%) | 70 621 | (4.6%) |
| **Forestry** | 47 931 | (3.3%) | 238 364 | (15.5%) |
| **Nature Conservation** | 7 769 | (0.5%) | 4 821 | (0.3%) |
| **Other** | 12068 | (1.7%) | 31 050 | (2%) |

Source: (Biggs *et al.*, 2005)

Brigalow communities and other associated/co-dominant ecosystems have been identified as endangered by the Queensland Herbarium (DERM, 2010). However, some regions continued to be cleared, with over one million hectares of woody vegetation removed in the Southern Brigalow Belt since 1995, leaving between 7 – 30% remnant vegetation (Wilson *et al.*, 2002; Seabrook *et al.*, 2006). In Queensland, the implementation of the *Vegetation Management and Other Legislation Amendment Act 2004* aimed to phase out broadscale clearing in Queensland by the end of 2006 (Seabrook *et al.*, 2006). The implementation of this legislation resulted in a considerable decrease, almost 50%, in clearing rates in the following years, falling from 235 000 ha yr$^{-1}$ (129 000ha remnant) in 2006-2007 to 123 000 ha yr$^{-1}$ (56 000ha remnant) in 2007-2008 (DERM, 2009a) (Table 2.2).

**Table 2.2**. Statewide Total Woody and Remnant Vegetation Clearing Rates ('000 ha year$^{-1}$) according Statewide Landcover and Tree Studies (SLATS) 1988 – 2009

| | **Time Period** | **Total Woody Vegetation (000 ha yr$^{-1}$)** | **Remnant Vegetation (000 ha yr$^{-1}$)** |
|---|---|---|---|
| **State-wide Woody and Remnant clearing** | 1995 – 1997 | 340 | 227 |
| | 1997 – 1999 | 425 | 286 |
| | 1999 – 2000 | 758 | 505 |
| | 2000 – 2001 | 378 | 213 |
| | 2001 – 2002 | 498 | 276 |
| | 2002 – 2003 | 554 | 366 |
| | 2003 – 2004 | 482 | 267 |
| | 2004 – 2005 | 351 | 172 |
| | 2005 – 2006 | 375 | 222 |
| | 2006 – 2007 | 235 | 129 |
| | 2007 - 2008 | 123 | 56 |

Furthermore, the introduction of the *Vegetation Management and Other Legislation Amendment Act 2009*, following a clearing moratorium, prohibits the clearing of 'high value' regrowth vegetation. The condition and status of remaining vegetation within the landscape is, hence, central in the persistence and survival of many native species and communities (Fensham, 2008).

The Moonie and Weir River Catchments also experienced a reduction in woody clearing rates in recent years in response to changed Queensland legislation (Table 2.3). Both catchments experienced a relatively high clearing rate during the 1990s and early 2000s. Clearing rates significantly declined from 7410 ha yr$^{-1}$ and 2672 ha yr$^{-1}$ in the period 2006-2007 to 3054 ha yr$^{-1}$ and 2633 ha yr$^{-1}$ respectively in the period 2007-2008 (DERM, 2009a). Regeneration of regrowth cover in the region averages around 4-7% with a high rate of turnover due to multiple re-clearing and regrowth events (Bowen *et al.*, 2009b).

**Table 2.3.** Clearing Rates ('000 ha year$^{-1}$) of Moonie and Weir River Catchments (total area) based on Statewide landcover and Tree Studies (SLATS) 1988 – 2009

| | **Time Period** | **Moonie Catchment (1 391 ha)** | **Weir River Catchment (1 540 ha)** |
|---|---|---|---|
| **Catchment Woody and Remnant clearing** | 1988 – 1991 | 28.18 | 18.92 |
| | 1991 – 1995 | 5.00 | 4.40 |
| | 1995 – 1997 | 17.76 | 9.99 |
| | 1997 – 1999 | 30.97 | 15.92 |
| | 1999 – 2001 | 22.90 | 9.77 |
| | 2001 – 2003 | 22.41 | 10.03 |
| | 2003 – 2004 | 11.73 | 4.03 |
| | 2004 – 2005 | 8.18 | 4.33 |
| | 2005 – 2006 | 5.58 | 2.39 |
| | 2006 – 2007 | 7.41 | 2.67 |
| | 2007 - 2008 | 3.05 | 2.63 |

Brigalow regrowth communities were protected under the *Vegetation Management and Other Legislation Amendment Act 2009*, with many remnant acacia communities throughout the Brigalow Belt, particularly on fertile clays, remaining below the 30% threshold necessary for biological conservation efforts (McAlpine *et al.*, 2002b). Over-grazing, soil erosion, raised water tables and salinity also continue to degrade the quality of much of the remaining habitat (Seabrook *et al.*, 2006).

## *2.3 Study Design*

### 2.3.1 Site Selection

This study focussed on *Acacia harpophylla* and *Casuarina cristata* woodlands (Regional Ecosystem 11.4.3; Sattler & Williams, 1999). This endangered ecosystem is described as open forest dominated by *Acacia harpophylla* and in some instances co-dominant with *Casuarina cristata* (Sattler & Williams, 1999; DERM, 2009b). Other tree species, including *E. populnea, E. cambageana* and *E. orgadophila*, may also be present in the canopy (DERM, 2009b). The shrub layer consists predominantly of *Geijera parviflora* and *Eremophila mitchelli* with *Carissa ovata* in the lower shrub cover, the ground layer is sparse to open with some scattered grasses and forbs (DERM, 2009b).

Potential study sites were selected based on area of vegetation patch (20 – 50ha), specified using the Queensland Regional Ecosystem Data Version 5 2003 and Queensland Regrowth Data provided by the Queensland Herbarium. Reserve sites were those found within state forests and national reserves and are assumed to have experienced relatively minimal disturbance. Remnant sites were recognised as vegetation that had no previous history of clearing but may have been subjected to some form of extrinsic disturbance (sporadic grazing, minor selective logging, recreational activity etc.). While some Reserve sites were sampled within much larger blocks (e.g. Southwood National Park (7120 ha)), small blocks (<50 ha) were preferred and where possible, thin linear strips of vegetation were avoided. Sites containing water sources, such as dams and streams, were also excluded from sampling. Other factors, such as access to sites, landholder permission and the spatial distribution of sites, were also considered during the site selection process.  Sites also had to be visually representative of the local vegetative community.

The age of regrowth patches was determined based on aerial photographs of the area and oral history according to land holders and managers.  The four regrowth treatments were established based on the availability of replication of sites.  Due to the small number of available regrowth patches that met the established criteria, site randomisation was not a viable option.  The majority of sites were located on private property and along road sides with some situated in surrounding state forests and national parks.  Sites also encompassed a number of surrounding land-uses, including grazing, cropping and reserves. The method of clearing and land-use history for each site was recorded from discussion with landholders where possible.

### 2.3.2 Experimental Design

Thirty-eight sites were sampled encompassing both remnant ("Reserve" and "Remnant") and regrowth classes (Regrowth$_{>40}$, Regrowth$_{30\text{-}40}$, Regrowth$_{20\text{-}30}$, Regrowth$_{<20}$) in the study area (Figure 2.2).  Sites had to be a minimum of 1km from other sites within the landscape.  Quadrats were placed subjectively within in an area deemed representative of the vegetation within the patch. Precautions were taken to avoid areas of recent anthropogenic disturbance and potential edge effects, leaving at least a 10-15m buffer between the edge of the patch and the sample quadrat.  The effects of livestock grazing were partially reduced by incorporating both grazed and ungrazed sites in each treatment, except for Reserve remnant, in an attempt to reduce potential grazing influence on patterns observed.

The availability of suitable replicates within the six treatments created an uneven factorial design (Table 2.4; Appendix A). Apart from the Reserve sites, at least 5 replicates were recorded for each treatment.

**Table 2.4.** Description of treatment types, abbreviated label, site number and total number of sites for each treatment.

| Treatment | Label | Time since last cleared (years) | n | Site Number |
|---|---|---|---|---|
| Reserve | REF | - | 4 | 3, 9, 10, 15. |
| Remnant | REM | - | 8 | 5, 6, 7, 8, 11, 13, 14, 16. |
| Regrowth greater than 40 years since clearing | $R_{>40}$ | > 40 | 7 | 1, 2, 25, 30, 31, 33, 37. |
| Regrowth 30 to 40 years since clearing | $R_{30-40}$ | 30-40 | 7 | 4, 22, 24, 27, 28, 35, 38. |
| Regrowth 20 to 30 years since clearing | $R_{20-30}$ | 20 – 30 | 7 | 12, 20, 21, 26, 32, 34, 36. |
| Regrowth less than 20 years since clearing | $R_{<20}$ | < 20 | 5 | 17, 18, 19, 23, 29. |

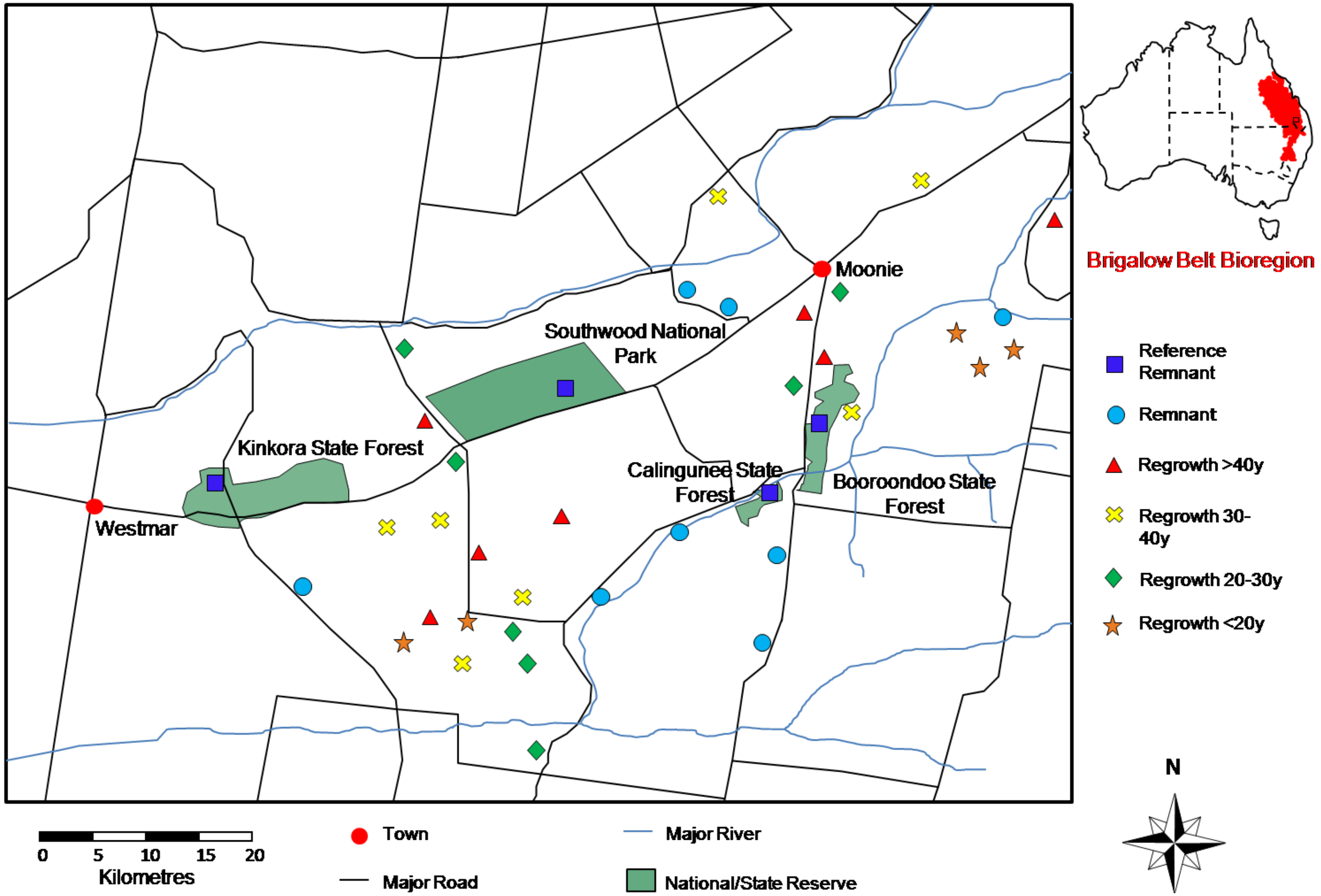


**Figure 2.2**. Map of study area showing the approximate location of study sites (n=38).

# Chapter 3: Composition and structure in remnant and regrowth Brigalow communities, western Darling Downs.

## 3.1 Introduction

The recent protection of high value regrowth, particularly Brigalow, in Queensland and throughout the east coast of Australia aims to increase biological diversity and ultimately, the extent of endangered ecosystems within highly fragmented agricultural landscapes (Dorrough and Moxham, 2005). Significant habitat disturbance and past and current land management practices can influence the succession and development of a community, affecting the composition and structural complexity and the ability of a system to recover to pre-disturbed states, resembling remnant vegetation (Aide *et al.*, 2000).

The protection of regrowth vegetation based on remotely sensed data as an indicator of vegetation type and function is yet to be proven effective, with the relative composition and condition of regrowth patches relatively unknown (Turner *et al.*, 2003; Bowen *et al.*, 2007). The recovery of certain environmental attributes through recovery processes does not necessarily suggest a return to identical pre-clearing levels. Traditionally, secondary succession follows a single pathway, whereby a cleared site, or partially cleared site, progresses through a number of transition states until it eventually comes to resemble a community similar in function, diversity and structure to its previous uncleared state (Figure 3.1) (Odum, 1969; Cattelino *et al.*, 1979). The emergence of multiple succession pathway models (see (Connell and Slatyer, 1977; Cattelino *et al.*, 1979) and alternate stable state theory (see (Scheffer *et al.*, 2001; Beisner *et al.*, 2003; Scheffer and Carpenter, 2003) has recognised that recovery from disturbances may not always follow a single pathway resulting in a similar community assemblage (‘climax’ community *sensu* Clements, 1916) prior to disturbance (Figure 3.1) (Cattelino *et al.*, 1979). State and Transition Models (Westoby *et al.*, 1989) are an effective method to conceptualise potential alternate states over time. These models differ from traditional plant succession models (progressing towards a single “climax” community) and describe vegetation as variety of alternate states over time, dependent on disturbances and how it affects particular state factors, including species populations, life histories and environmental variables (Cattelino *et al.*, 1979; Whalley, 1994; Garnier *et al.*, 2004)).

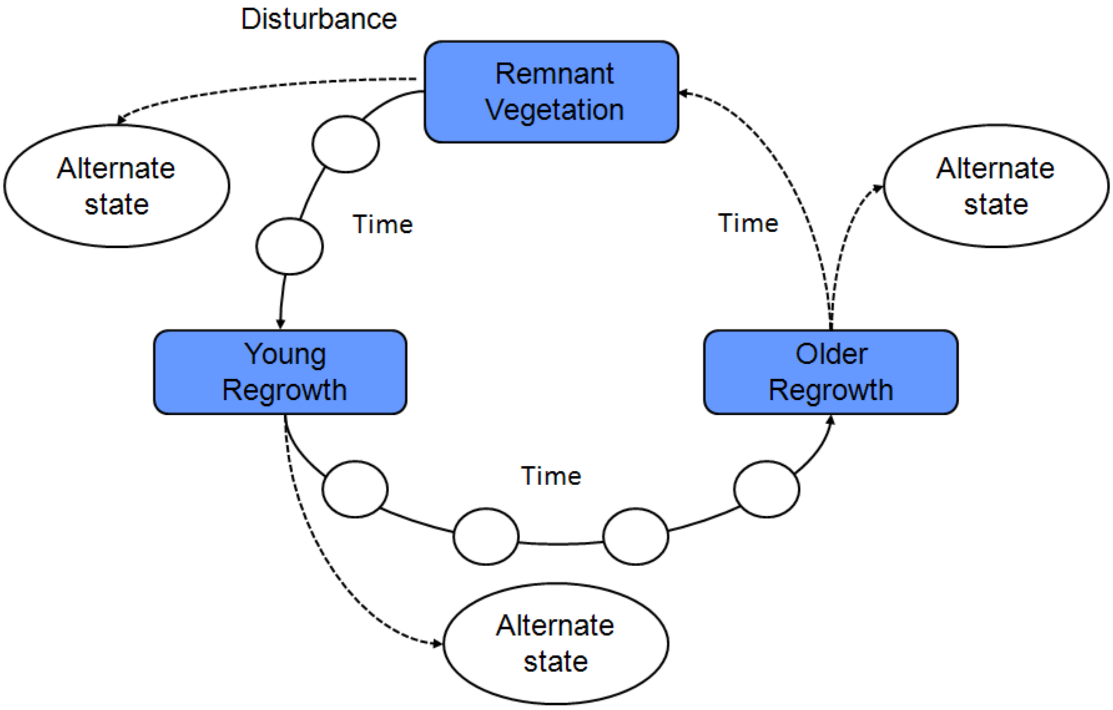


**Figure 3.1**. Concept model of secondary succession and the development of alternate stable states. (Note: Alternate stable state trajectories may divert at any given stage over time, similarly disturbances may occur at any time resulting in regression along the pathway. Circles represent other theoretical transition states over the course of succession (adapted from (Connell and Slatyer, 1977; Beisner *et al.*, 2003).

### 3.1.1 Composition and Structure in Determining Ecosystem Recovery

Species composition, species richness and community structure are important aspects that can influence the persistence and ultimate resilience of an ecosystem and can be employed to gauge the recovery of a system (Ruiz-Jaen and Aide, 2005). Ecosystem and community response to disturbance is dependent on the morphology, life histories, tolerances and physiology of the species within the ecosystem (Galindo-Jaimes *et al.*, 2002). During succession, further disturbances can significantly affect internal dynamics and successional pathways, ultimately impacting on the community's ability to recover and function to pre-clearing levels (Turner *et al.*, 1998; Hobbs *et al.*, 2007). By measuring different ecosystem attributes, such as vegetation structure, diversity and composition, the overall success of the recovery process can be assessed (Ruiz-Jaen and Aide, 2005).

Measures of vegetation structure and complexity are important attributes that can assist in providing information on habitat development and suitability at different stages of recovery (McElhinny *et al.*, 2005; Ruiz-Jaén and Aide, 2005). Assessment of structure can reveal the potential successional pathways of a particular community by the development of ecosystem processes and provide an indication of ecosystem productivity (Ruiz-Jaén and Aide, 2005). Stand structure can greatly influence the growth of plant species by contributing to the availability of nutrients and soil fertility, as well as affecting the microclimate (Buchmann *et al.*, 1997; Rhoades *et al.*, 1998; Chen *et al.*, 1999). The recovery of vegetation structure following major disturbance has also been linked with the return of fauna and ecosystem processes such as productivity (DeWalt *et al.*, 2003; Ruiz-Jaen and Aide, 2005). A number of studies have found a positive correlation between species diversity and the development of secondary forests in relation to fauna (e.g. (Fisher, 2000; Nichols and Grant, 2007; Woinarski *et al.*, 2009), particularly bird diversity (see Raman *et al.*, 1998; Stouffer and Bierregaard, 2007; Kanowski *et al.*, 2008).

Measuring the composition and richness of species can also provide an indication of the long-term stability of an ecosystem (Ruiz-Jaen and Aide, 2005). Species composition can provide information on trophic levels, functioning and dynamics within populations and communities, and can influence the ability of a community to withstand different disturbances and persist in the landscape (Walker, 1995; Ruiz-Jaen and Aide, 2005). Species richness can affect numerous aspects of an ecosystem through species interactions and individual tolerances (Hooper and Vitousek, 1997, 1998). For example, complementarity in nutrient uptake can impact ecosystem processes, such as productivity, with different species accessing various parts of the available nutrient reserves (Hooper and Vitousek, 1998). Analysing functional groups aids in providing an indication of the importance of species richness within an ecosystem, highlighting the diversity of species contributing to particular aspects of structure and ecosystem functioning (Hooper and Vitousek, 1997; Peterson, 1998). Measuring functional group diversity further assists in providing an indication of overall ecosystem resilience, showing the strength and robustness of groups to differing intensities and frequencies of disturbances (Peterson, 1998).

Recovery and succession following a significant disturbance can be influenced by various environmental factors, including individual species biology, competition, stochasticity and the interaction between biotic and abiotic components (Guariguata and Ostertag, 2001). In some instances, ecosystem function within regrowth vegetation may recover long before similar composition is achieved, if at all (Guariguata and Ostertag, 2001). In other circumstances, the availability and uptake of soil nutrients may be similar to the original system, but may be dominated by a single or very few species, typical of a monoculture (Hooper and Vitousek, 1998).

Long-term perturbations, such as livestock grazing, can also influence the recovery of regrowth communities. Grazing has been found to influence floristic composition and species richness in a variety of community types, interfering with soil dynamics and favouring particular plant species (McIntyre and Lavorel, 1994; Pettit *et al.*, 1995; Prober and Thiele, 1995). Similarly, continued disturbances may also promote invasion of exotic species, increasing competition, particularly during early stages of recovery or when coupled with other disturbance regimes (Hobbs and Huenneke, 1992).

Identifying changes in stand structure and species composition can be manifestations of altered ecological functioning within a system brought about by past and continued disturbance regimes (Turner *et al.*, 1998; Ruiz-Jaen and Aide, 2005). Assessment of these ecosystem components can be used to determine the development and potential trajectory of regrowth vegetation following disturbance, providing an indication of a community's general resilience (Aide *et al.*, 2000).

In this chapter, species composition and community stand structure were compared across different aged regrowth vegetation, including old regrowth (Regrowth >40y), intermediate regrowth (Regrowth 30-40y and Regrowth 20-30y) and recent regrowth (Regrowth <20y), and remnant (Reserve and Remnant) vegetation types to provide an indication of plant community regeneration since last disturbance. In addition, analysis of overall plant species richness and functional group species richness were used to describe any patterns observed across the treatment types.

The general question examined was: are there differences between Brigalow remnant and regrowth communities at different regeneration stages? The specific hypothesis examined are:

- Floristic composition, stand structure, compositional and structural functional group richness differ between Brigalow treatments (Reserve, Remnant, Regrowth$_{>40}$, Regrowth$_{30\text{-}40}$, Regrowth$_{20\text{-}30}$ and Regrowth$_{<20}$).

## 3.2 Methodology

### 3.2.1 Study Sites

The Brigalow vegetation sampled in the study area was identified as RE 11.4.3 under the Regional Ecosystem Mapping (Version 5, 2003; Sattler & Williams 1999) and regrowth mapping (Queensland Herbarium, 2008) provided by the Queensland Herbarium. The study contrasts the stand structure, species richness and floristic composition of 38 sites across the six treatment types:

1) Reserve remnant Brigalow vegetation ('Reserve') with an absence of clearing and minimal other disturbance (e.g. grazing), protected within reserves and national parks.
2) Remnant Brigalow vegetation ('Remnant') with an absence of clearing but some continued disturbance, mainly grazing.
3) Brigalow regrowth greater than 40 years since last cleared (Regrowth$_{>40}$).
4) Brigalow regrowth 30 – 40 years since last cleared (Regrowth$_{30\text{-}40}$).

5) Brigalow regrowth 20 – 30 years since last cleared (Regrowth$_{20-30}$).
6) Brigalow regrowth less than 20 years since last cleared (Regrowth$_{<20}$).

Most regrowth sites are currently grazed to varying intensities. More detail of study site characteristics and experimental design can be found in Section 2.3.

### 3.2.2 Field Techniques

Remnant and regrowth sites were sampled between 10 February and 20 May 2010. At each site a 500 m$^2$ quadrat was established covering an area that was deemed representative of the selected vegetation patch (see Section 2.3.2).

Within each quadrat, structural and general site condition was recorded. Structural characteristics were visually estimated based on the Specht (1981) structural classification scheme. Several vegetative strata were recognised within the study sites, including trees (> 30 m; 20-30 m; 10-20 m; < 10 m), shrubs (> 2 m; <2 m), graminoids and herbs/forbs (Table 3.1) (after (Le Brocque and Buckney, 1995b). Overhanging branches, falling within quadrat boundaries, from trees outside were included in cover assessments. Ground litter cover was also visually estimated within the sample quadrat, classified according to logs (< 20 cm circumference; 10-20 cm circumference), coarse leaf litter (branches and twigs 5-10 cm circumference), fine leaf litter (< 5cm circumference), cryptogams, bare ground and rock (Table 3.1).

Overall site condition was subjectively ranked based on evidence of disturbance within the sample quadrat. Livestock grazing, feral animal invasion, erosion, logging, clearing, pollution and exotic weed invasion were scored on a scale of 0-3 (0 = absent; 1 = minor; 2 = moderate; 3 = severe) (Table 3.1).  The presence or absence of gilgai within the site were also recorded and scored based on the degree of undulation (1 = Light (<50cm deep); 2 = moderate (50-100cm deep); 3 = heavy (>100cm deep) (Table 3.1) (Russell *et al.*, 1967).

**Table 3.1.** Summary of variables measured at each sample site

| Variable | Area | Description |
|---|---|---|
| **i) Structural Features** | | |
| Cover Trees > 30m | 500m$^2$ | Dominant species was also recorded |
| Cover Trees 20 – 30m | 500m$^2$ | Dominant species was also recorded |
| Cover Trees 10 – 20m | 500m$^2$ | Dominant species was also recorded |
| Cover Trees < 10m | 500m$^2$ | Dominant species was also recorded |
| Cover Shrubs > 2m | 500m$^2$ | Dominant species was also recorded |
| Cover Shrubs < 2m | 500m$^2$ | Dominant species was also recorded |
| Cover Graminoids | 500m$^2$ | Non-woody grasses and sedges |
| Cover Herbs / Forbs | 500m$^2$ | |
| Cover Large Logs | 500m$^2$ | Logs > 20cm in circumference |
| Cover Small Logs | 500m$^2$ | Logs 10 – 20cm in circumference |
| Cover Coarse Litter | 500m$^2$ | Twigs and branches 5 – 10cm in circumference |
| Cover Fine Litter | 500m$^2$ | Twigs and branches < 5cm in circumference |
| Cover Rock | 500m$^2$ | |
| Cover Cryptogams | 500m$^2$ | |
| Cover Bare Ground | 500m$^2$ | |
| Tree Stag Count | 500m$^2$ | |
| Tree Basal Area | m$^2$/ ha | |
| Tree Stem Density | 200m$^2$ | |
| **ii) Site Condition** | | |
| Weed Invasion | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Grazing Intensity | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Feral Animals | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Erosion | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Logging | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Clearing | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Pollution | 500m$^2$ | 4 levels scored: (0) absent; (1) minor; (2) moderate; (3) severe |
| Gilgai | 500m$^2$ | Present / absent; 3 levels scored: (1) light; (2) medium; (3) heavy |
| **iii) Floristic Composition** | | |
| Species Richness | 500m$^2$ | Total number of species found at each site |
| Species Frequency Score | 500m$^2$ | Number of quadrats species are present divided by the number of quadrats (9) |

Leaf litter biomass was also collected at four points along four transects that extended from the centre to each corner of the 500m$^2$ quadrat. Using a random number generator (between 0 -20), the number of steps from the centre outwards along each transect was determined. A 0.25m$^2$ square quadrat was placed to the right hand side of the transect line and all dead leaf litter collected, bulked and tagged. The procedure was repeated for the remaining three transects and bulked together. Sampling at the bottom of gilgai was avoided in order to reduce the effects of litter pooling (Facelli and Pickett, 1991). Biomass samples were dried in an oven at 40˚C for four days and dry weight recorded.

Floristic composition was evaluated, as a frequency, using the Nested Quadrat Method (Le Brocque and Buckney, 1995a; Morrison *et al.*, 1995). Nine concentric sub-quadrats were erected (1m$^2$; 2m$^2$; 5m$^2$; 10m$^2$; 20m$^2$; 50m$^2$; 100m$^2$; 200m$^2$; 500m$^2$) based on species area curves previously conducted. The presence of each vascular plant species was recorded for each sub-quadrat to obtain a frequency (Frequency score, *sensu* (Morrison *et al.*, 1995). Such scores have been found to be functionally equivalent to more traditional, randomly or contiguously positioned frequency (Morrison *et al.*, 1995). Unidentified species were collected and tagged for later identification (International Code of Botanical Nomenclature). Plant species were categorised within 3 key functional groupings based on life origin (exotic/introduced or native), life histories (annual or perennial) and growth form (tree, shrub, graminoid or herb) for further analysis (McIntyre *et al.*, 1995; Lavorel *et al.*, 1997).

### 3.2.3 Numerical Analyses

An unweighted pair-group using arithmetic averages clustering (UPGMA, (Clarke and Warwick, 2001) and non-metric multidimensional ordination (nMDS), using Bray Curtis Similarity Matrices derived from floristic composition (frequency) and stand structure data were created to clarify *a priori* groups (PRIMER for Windows V5.2.9 (PRIMER-E, 2000)). UPGMA clustering analyses the similarity between sites where least similar sites diverge early in the analysis, grouping sites that are most similar (Clarke and Warwick, 2001). Structural data (Foliage Projected Cover of strata) was transformed using an ArcSine function before analysis in order to prevent bounded effects on percentages (Quinn and Keough, 2002) (Table 3.2). An Analysis of Similarities (ANOSIM) (Clarke and Warwick, 2001) was used to identify significant differences between the six treatment groups based on floristic composition and structure data.

One-way Analysis of Variance (ANOVA) (SPSS Inc. V17, 2007) was used to analyse the cover of each strata and species richness across the six *a priori* groupings. Residual plots and tests of variance were used to assess data variance in order to meet the assumptions for the analysis. Tukey's *post hoc* tests were calculated ($p<0.05$) in order to identify differences between the groups (De Veaux and Velleman, 2004). One-way ANOVA was also performed on functional traits, categorising species richness based on origin (native/ exotic) and perenniality (annual/ perennial), and structural functional traits according to life form (trees, shrubs, groundcover; graminoids, herbs/forbs) (after (Lavorel *et al.*, 1997; McIntyre *et al.*, 1999). Raw data was transformed using a natural log function (Ln or Ln+1) to improve homoscedasticity (Table 3.2) and checked for skewed data using Q-Q Plots.

**Table 3.2.** Summary of response variables, analyses carried out and data transformations used.

| Analyses | Response Variable | Statistical Test | Transformation |
|---|---|---|---|
| Multivariate Analyses | Floristic Composition | UPGMA (Bray Curtis Similarity) Cluster Analysis | |
| | | nMDS/ ANOSIM | |
| | Stand Structure | UPGMA (Bray Curtis Similarity) Cluster Analysis | ArcSin |
| | | nMDS/ ANOSIM | ArcSin |
| Univariate Analyses | Species Richness | ANOVA | Ln |
| | Tree Species Richness | ANOVA | Ln |
| | Shrub Species Richness | ANOVA | Ln+ 1 |
| | Groundcover Species Richness | ANOVA | Ln |
| | Herb/Forb Species Richness | ANOVA | Ln |
| | Graminoid Species Richness | ANOVA | Ln |
| | Introduced Species Richness | ANOVA | Ln + 1 |
| | Native Species Richness | ANOVA | Ln |
| | Perennial Species Richness | ANOVA | Ln |
| | Annual Species Richness | ANOVA | Ln |
| | Tree 10m-20m Foliage Projected Cover | ANOVA | Ln + 1 |
| | Tree <10m Foliage Projected Cover | ANOVA | ArcSin |
| | Shrub >2m Foliage Projected Cover | ANOVA | Ln + 1 |
| | Shrub <2m Foliage Projected Cover | ANOVA | ArcSin |

| Analyses | Response Variable | Statistical Test | Transformation |
|---|---|---|---|
| Univariate Analyses | Graminoid Cover | ANOVA | Ln |
|  | Herb/Forb Cover | ANOVA | Ln |
|  | % Logs < 20cm circum. | ANOVA | Ln + 1 |
|  | % Logs 10cm – 20cm circum. | ANOVA | Ln |
|  | Coarse Litter Cover | ANOVA | ArcSin |
|  | Fine Litter Cover | ANOVA | ArcSin |
|  | % Bare Ground | ANOVA | Ln |
|  | Leaf Litter Biomass | ANOVA | Ln |

## *3.3. Results*

### 3.3.1 Patterns in Floristic Composition in Remnant and Regrowth Brigalow Communities

#### *3.3.1.1 General patterns in Floristic Composition*

A total of 148 plant species were identified across the 38 sites sampled (Appendix B). The plant species recorded included 6 tree species, 18 shrub species, 3 mistletoe species and 121 herbaceous species. A total of 49 species were endemic to regrowth sites compared to 26 species found only in the remnant sites sampled (Appendix B). Remnant sites had the highest total number of species with Regrowth$_{<20}$ exhibiting the least number of species (Table 3.3). Herbaceous plant species included 37 species of dicot perennials, 34 monocot perennials, 16 dicot annuals and 8 monocot annual species. Sixteen introduced species were also found within the sites sampled. *Homopholis belsonii* (Belson's Panic), classified as Vulnerable under the *EPBC Act 1999* and Endangered under Queensland's *Nature Conservation Regulation 2006,* was found in one site (site 13) in relatively low abundance.

**Table 3.3.** Cumulative native, exotic and total plant species recorded for each treatment. Values in parentheses are total species found exclusively in that treatment.

| Treatment | Native species | Introduced species | Total species |
|---|---|---|---|
| **Reserve** | 57 (6) | 5 (1) | 62 (7) |
| **Remnant** | 75 (13) | 7 (2) | 82 (15) |
| **Regrowth$_{>40}$** | 60 (6) | 5 (1) | 65 (7) |
| **Regrowth$_{30-40}$** | 72 (11) | 7 (1) | 79 (12) |
| **Regrowth$_{20-30}$** | 54 (3) | 5 (1) | 59 (4) |
| **Regrowth$_{<20}$** | 49 (6) | 9 (3) | 58 (9) |

#### *3.3.1.2 Multivariate Gradients in Floristic Composition*

UPGMA Cluster Analysis of frequency data for the six vegetation types shows the treatment groups (Reserve, Remnant, $\text{Regrowth}_{>40}$, $\text{Regrowth}_{30\text{-}40}$, $\text{Regrowth}_{20\text{-}30}$, $\text{Regrowth}_{<20}$) were generally well-recovered in the analysis although some overlap is evident (Figure 3.2). Remnant and Reserve sites were more similar to each other than to regrowth treatments, with the exception of sites 8 and 14. $\text{Regrowth}_{<20}$ were also well clustered compared to other regrowth sites. Intermediate regrowth treatments ($\text{Regrowth}_{30\text{-}40}$ and $\text{Regrowth}_{20\text{-}30}$) and older regrowth ($\text{Regrowth}_{>40}$ sites) were more dispersed throughout the dendrogram.

nMDS ordination showed a general gradient from Remnant and Reserve sites, through older regrowth ($\text{Regrowth}_{>40}$, $\text{Regrowth}_{30\text{-}40}$) to more recent regrowth from the left to right of the ordination diagram (Figure 3.3). The $\text{Regrowth}_{20\text{-}30}$, $\text{Regrowth}_{30\text{-}40}$ and $\text{Regrowth}_{>40}$ sites showed considerable overlap in terms of similarity. Site 4 ($\text{Regrowth}_{30\text{-}40}$) was recognised as an outlier and was excluded from the analysis. This was the only site in this treatment protected in a reserve (Booroondoo Reserve) and contained relatively high *Callitris glaucophylla* abundance.

Analysis of Similarity (ANOSIM) showed significant differences in floristic composition between the treatment types (Global R = 0.364, p = 0.001) (Figure 3.3b) (Appendix C). Reserve remnant and remnant sites were significantly different floristically to all regrowth types ($p < 0.05$). Recent regrowth ($\text{Regrowth}_{<20}$) was also significantly different to all other treatments ($p < 0.05$). There was no difference in floristic composition between intermediate (Regrowth$_{30\text{-}40}$ and Regrowth$_{20\text{-}30}$) and older (Regrowth$_{>40}$) regrowth treatments (Figure 3.3b).

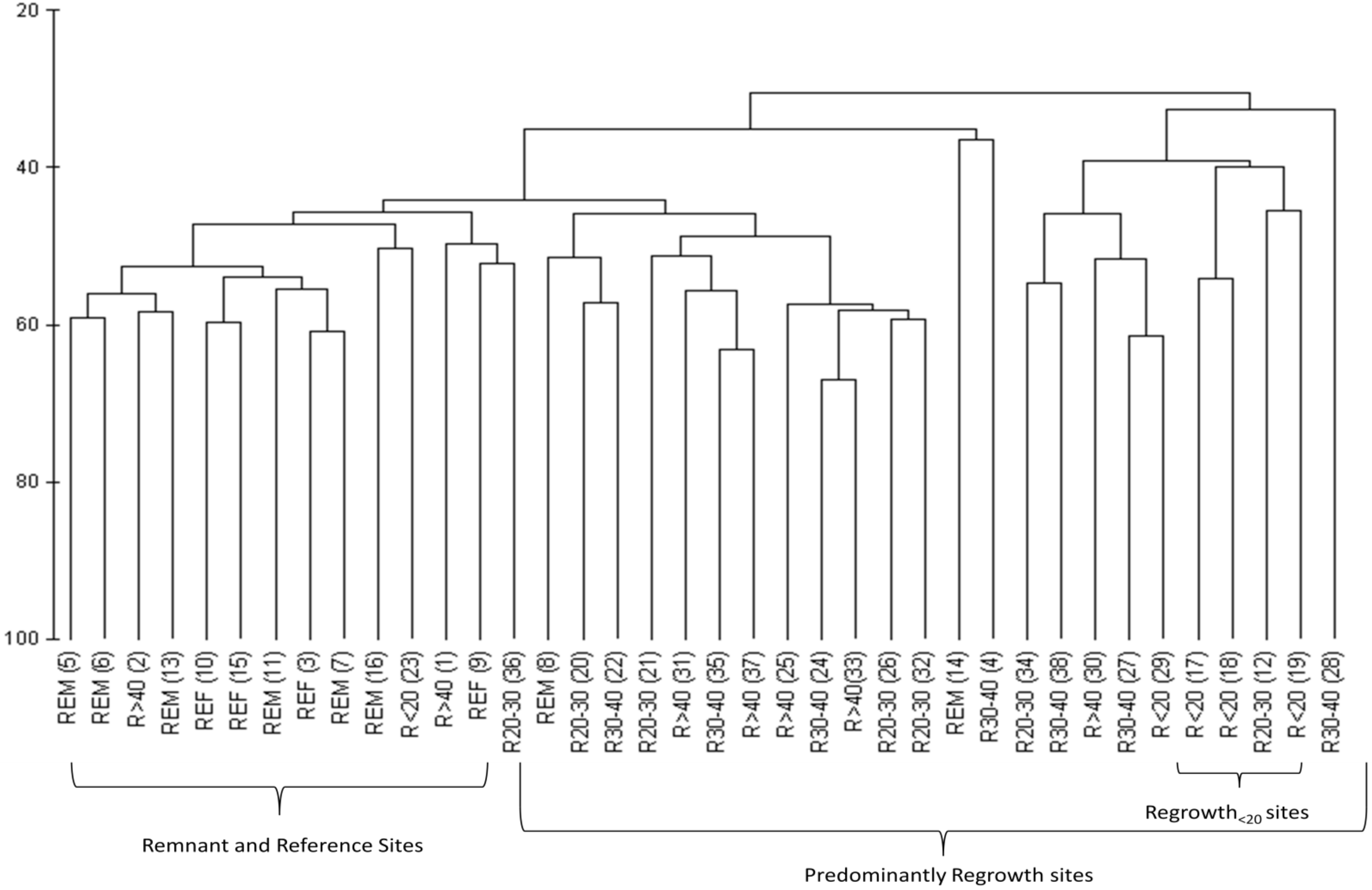


**Figure 3.2**. Dendrogram based on UPGMA cluster analysis of floristic composition (frequency data) showing sample sites.

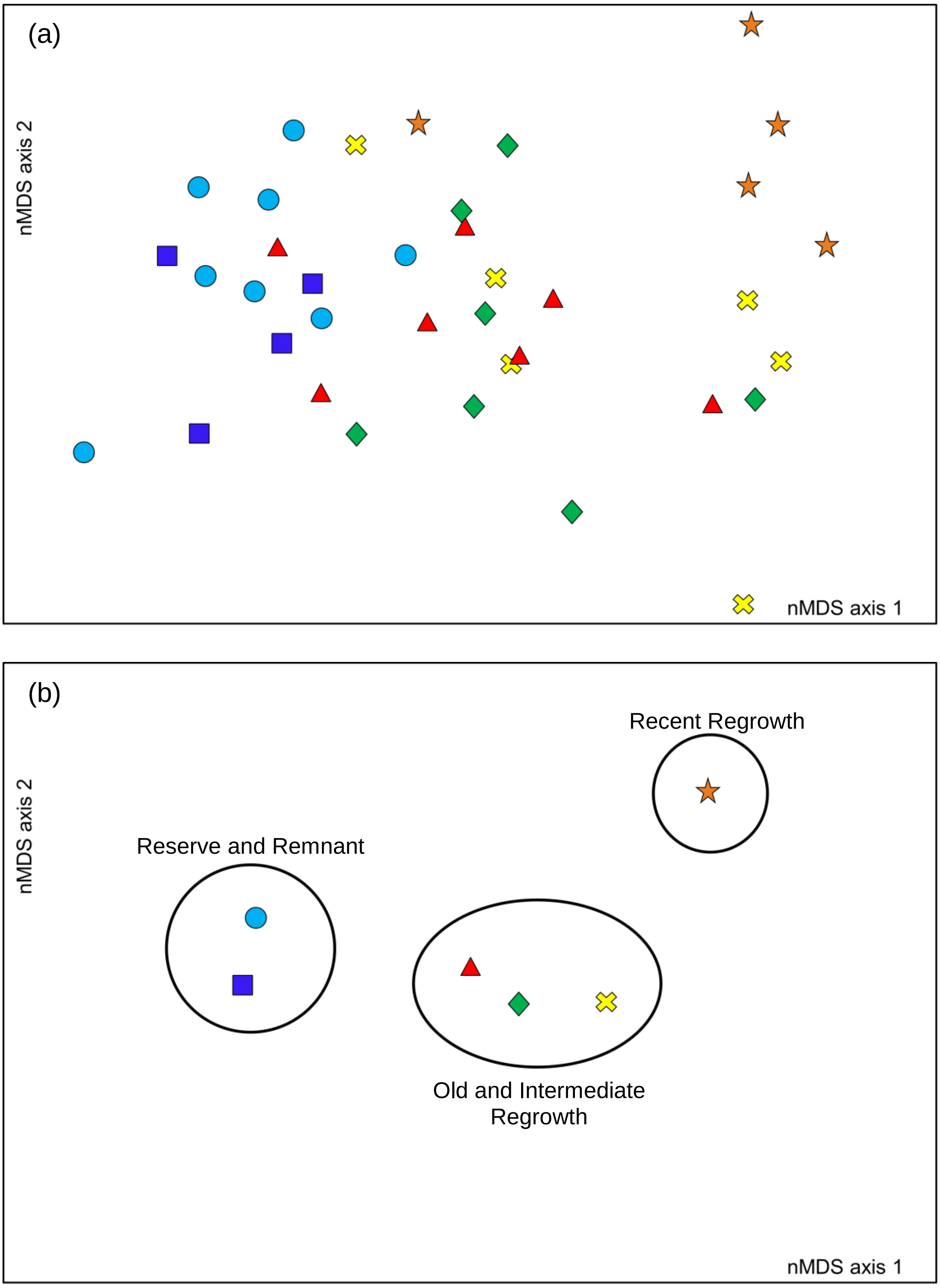


**Figure 3.3**. nMDS ordination of floristic composition (frequency data) of (a) site sample scores (excluding site 4) and (b) treatment centroids (Stress = 0.18). Pairwise comparisons of ANOSIM represented by circles, encapsulated treatments are not significantly different at p<0.05. ■ = Reserve remnant, ● = Remnant, ▲ = Regrowth$_{>40}$, ✖ = Regrowth$_{30\text{-}40}$, ◆ = Regrowth$_{20\text{-}30}$, ★ = Regrowth$_{<20}$.

### *3.3.1.3 Patterns in plant origin and perenniality species richness*

Species richness ranged between 16 to 36 species/500m$^2$ across the 38 sites sampled, with mean species richness varying between 20.3 and 29.6/500m$^2$. Species richness was significantly different between the treatment types (ANOVA, F = 4.730; p = 0.002), with higher species richness in the reserve and remnant sites than Regrowth$_{20\text{-}30}$ sites (Figure 3.4a; Appendix D). Remnant sites were also significantly higher in species richness than the Regrowth$_{<20}$ sites (Figure 3.4a).

Exotic species were recorded in 37 of the 38 sites sampled. There were no significant differences in introduced species richness between the six treatments (Figure 3.4b). Regrowth <30 years since disturbance had significantly less native species richness than remnant and reserve treatments (Figure 3.4c) ($p<0.05$). No significant differences in native species richness were observed between regrowth treatments.

Mean perennial species richness was significantly higher in remnant sites than both Regrowth$_{20\text{-}30}$ and Regrowth$_{<20}$ treatments (F= 3.185; p =0.019) although not different to other treatments (Figure 3.4d). Mean annual richness was significantly higher in Regrowth$_{<20}$ sites than Regrowth$_{>40}$ and Regrowth$_{20\text{-}30}$ (ANOVA F = 3.640, p = 0.010) although not different to other treatments (Figure 3.4e).

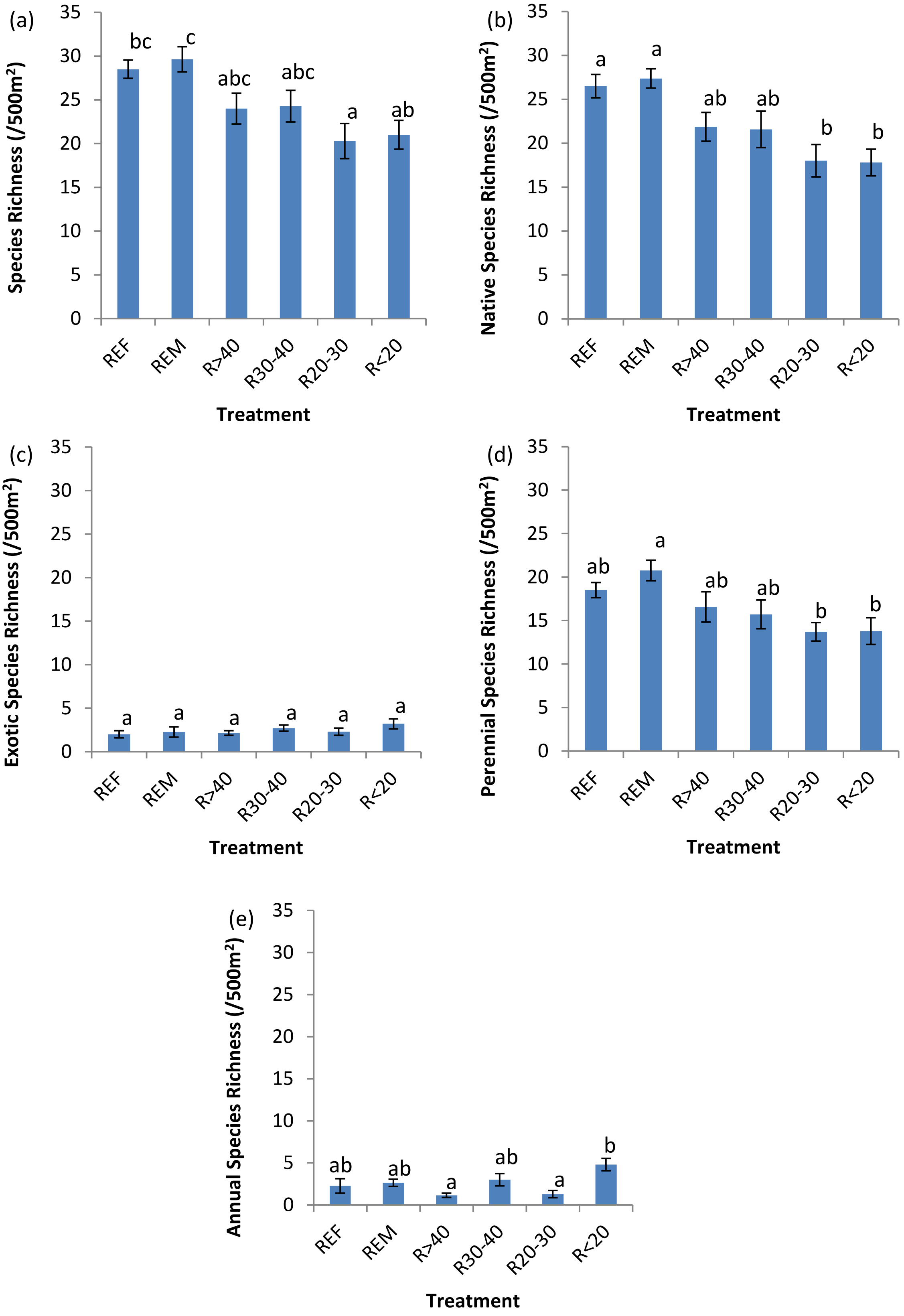


**Figure 3.4.** Mean (a) total species richness, (b) native species richness, (c) exotic species richness, (d) perennial and (e) annual species richness for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using natural log (Ln) transformation (Tukey's post hoc tests, $p < 0.05$).

## 3.3.2 Patterns in Stand Structure in Remnant and Regrowth Brigalow Communities

### 3.3.2.1 General Patterns in Stand Structure

There were no trees greater than 20m recorded in any of the regrowth treatments (Appendix E). Total tree and shrub foliage projected cover ranged between 33% to 52% and 3% to 21% respectively between Brigalow treatments (Table 3.4). Groundcover ranged between 17% with Regrowth>40 and 39% in Regrowth$_{<20}$. Cover of logs (>10cm) varied between 4% to 18% and litter (<10cm) ranged between 40% to 66% (Appendix E).

**Table 3.4.** Summary of total strata cover for each treatment. Standard errors are shown in parentheses.

| | **Reserve** | **Remnant** | **Regrowth$_{>40}$** | **Regrowth$_{30\text{-}40}$** | **Regrowth$_{20\text{-}30}$** | **Regrowth$_{<20}$** |
|---|---|---|---|---|---|---|
| Total Tree Cover | 52.0% (5.4) | 48.0% (2.2) | 40.9% (2.5) | 42.9% (3.6) | 49.0% (4.1) | 33.4% (3.8) |
| Total Shrub Cover | 21.0% (0.7) | 14.8% (2.4) | 12.0% (2.7) | 5.9% (1.2) | 4.6% (1.1) | 3.0% (1.5) |
| Groundcover | 21.3% (1.4) | 22.3% (2.9) | 16.7% (3.7) | 22.7% (4.4) | 17.3% (1.7) | 39.0% (7.5) |
| Total Log Cover | 18.3% (2.3) | 12.5% (1.9) | 7.8% (2.1) | 5.0% (1.7) | 4.1% (0.9) | 7.8% (2.7) |
| Total Litter Cover | 66.0% (6.3) | 64.8% (3.8) | 59.5% (3.6) | 60.1% (6.7) | 57.3% (6.6) | 39.6% (4.4) |
| Bareground | 7.0% (3.0) | 8.0% (1.7) | 19.4% (4.7) | 18.7% (3.8) | 23.3% (6.0) | 21.6% (3.4) |

### 3.3.2.2 Multivariate Gradients in Stand Structure

The UPGMA analysis of the structural cover data showed the majority of Remnant and Reserve sites were more similar to each other than to regrowth sites (Figure 3.5), with the exception of two Regrowth$_{>40}$ sites (Sites 1 and 25). Regrowth$_{<20}$ sites were more similar to each other in stand structure compared to other regrowth sites. Regrowth$_{>40}$, Regrowth$_{30\text{-}40}$, and Regrowth$_{20\text{-}30}$ sites overlapped in relation to structural cover similarity. $\text{Regrowth}_{30\text{-}40}$ and $\text{Regrowth}_{>40}$ were more similar to one another, as were $\text{Regrowth}_{20\text{-}30}$ and $\text{Regrowth}_{<20}$ sites. Site 19 was identified as an outlier and excluded from further analysis. This site was heavily grazed, had a more open tree canopy than other recent regrowth sites and contained a much higher cover of grasses (Appendix H).

Patterns observed in the UPGMA of structure data were reflected in the nMDS ordination, with the remnant sites most similar to one another (Figure 3.6a). The regrowth sites were more widely dispersed, particularly the older regrowth ($\text{Regrowth}_{>40}$, $\text{Regrowth}_{30\text{-}40}$). The $\text{Regrowth}_{<20}$ sites were also relatively similar structurally.

There were significant differences in the similarity of treatments in structural cover (ANOSIM; Global R = 0.354; p = 0.001) (Figure 3.6b) (Appendix F). A general gradient was observed in relation to age, with $\text{Regrowth}_{<20}$, $\text{Regrowth}_{20\text{-}30}$ and $\text{Regrowth}_{30\text{-}40}$ significantly different from remnant classes; $\text{Regrowth}_{20\text{-}30}$ and $\text{Regrowth}_{30\text{-}40}$ not significantly different from $\text{Regrowth}_{>40}$; and Remnant sites not significantly different from $\text{Regrowth}_{>40}$. The Reserve remnant sites, while not distinguishable from remnant sites, were significantly different from $\text{Regrowth}_{>40}$ sites (Figure 3.6b).

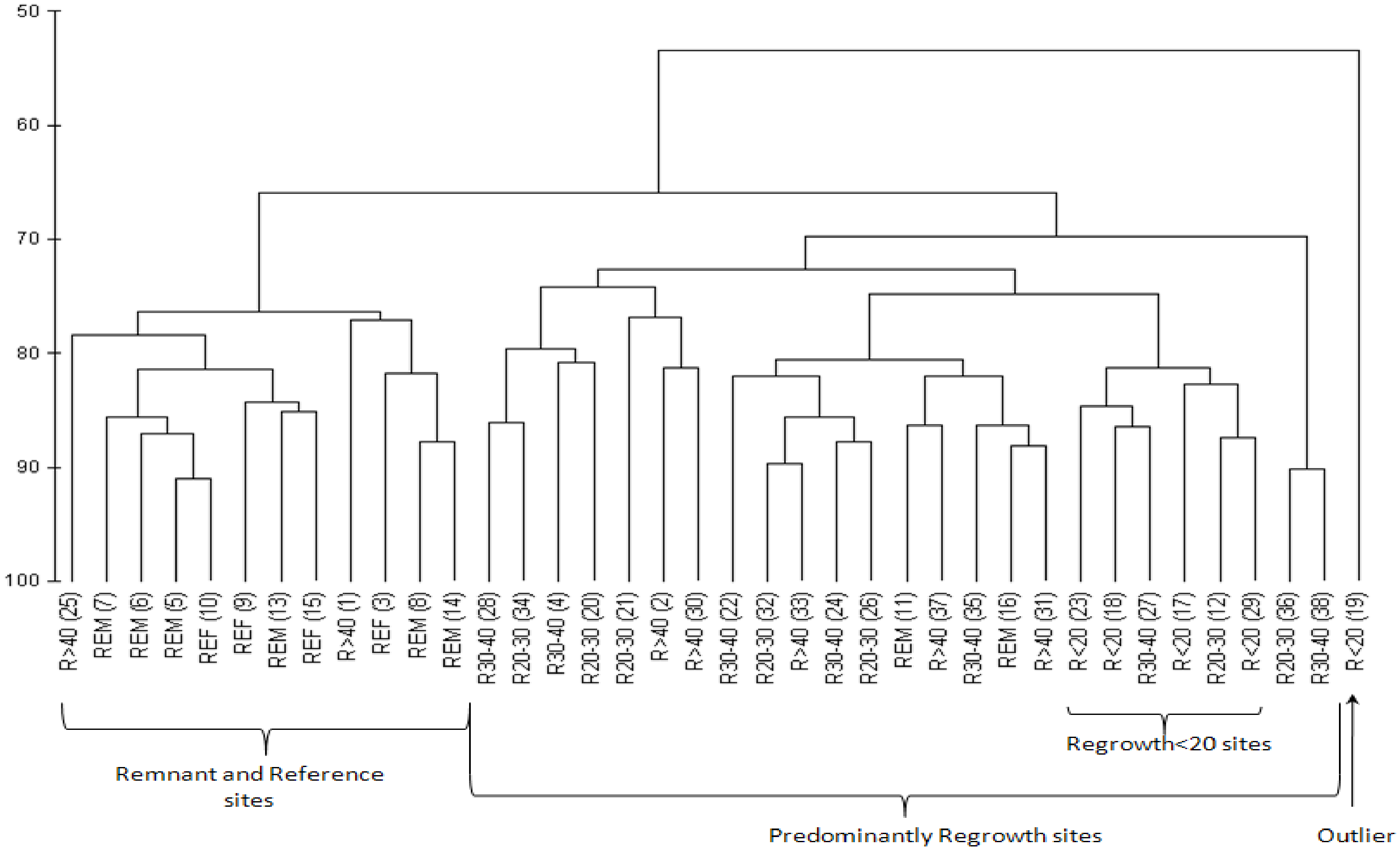


**Figure 3.5.** Dendrogram based on UPGMA Cluster Analysis of stand structure (cover data) showing sample sites.

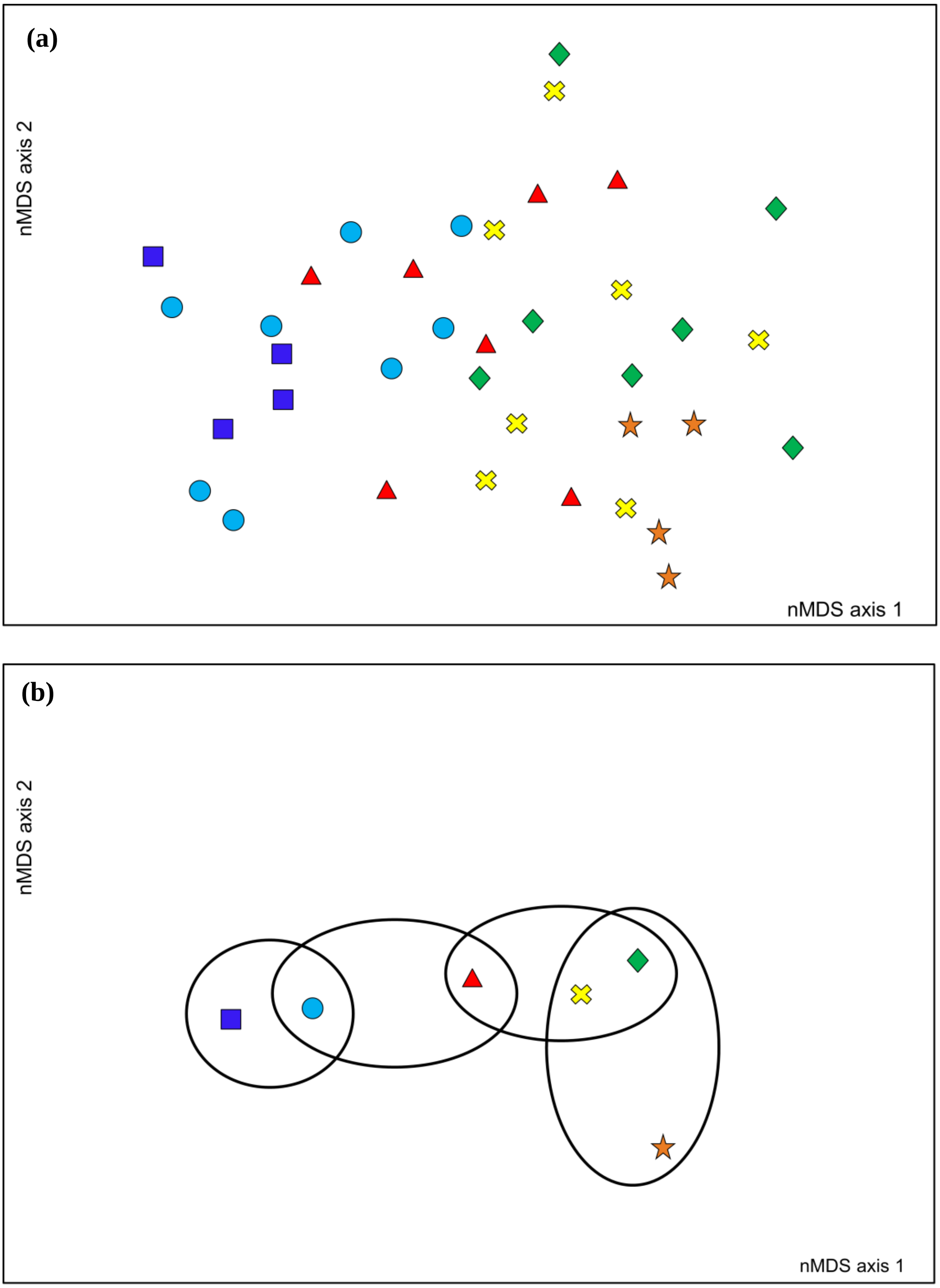


**Figure 3.6.** nMDS ordination of stand structure (cover data ArcSine transformed) of (a) sites (excluding site 19) and (b) treatment centroids (Stress = 0.15). Pairwise comparisons of ANOSIM represented by circles, encapsulated treatments are not significantly different at $p<0.05$. ■ = Reserve remnant, ● = Remnant, ▲ = Regrowth$_{>40}$, ✖ = Regrowth$_{30\text{-}40}$, ◆ = Regrowth$_{20\text{-}30}$, ★ = Regrowth$_{<20}$.

### *3.3.2.3 Patterns in Strata Cover*

There were significant differences between treatments in relation to tree strata (ANOVA, $p<0.05$; Figure 3.7a, b) (Appendix G). Both remnant treatments had significantly higher cover of trees between 10m – 20m than $\text{Regrowth}_{30\text{-}40}$ and $\text{Regrowth}_{<20}$ ($p<0.05$; Figure 3.7a) (recent regrowth excluded). $\text{Regrowth}_{30\text{-}40}$ and $\text{Regrowth}_{20\text{-}30}$ had a significantly higher cover of trees less than 10m in height compared with remnant and reserve sites ($p<0.05$; Figure 3.7b).

Remnant and Reserve sites were significantly higher in shrub cover than the $\text{Regrowth}_{<20}$ ($p<0.05$; Figure 3.7c,d). Reserve sites exhibited a higher shrub cover (>2m) than both $\text{Regrowth}_{30\text{-}40}$ and $\text{Regrowth}_{20\text{-}30}$ ($p<0.05$). $\text{Regrowth}_{20\text{-}30}$ also had significantly less shrub cover (>2m) than the remnant treatment ($p<0.05$; Figure 3.7c). Reserve and remnant communities were also higher in shrub cover (<2m) in comparison to $\text{Regrowth}_{<20}$ ($p<0.05$; Figure 3.7d).

Total groundcover was significantly higher in $\text{Regrowth}_{<20}$ sites compared to $\text{Regrowth}_{20\text{-}30}$ and $\text{Regrowth}_{>40}$ ($p<0.05$; Figure 3.7e). Graminoid cover was significantly higher in $\text{Regrowth}_{<20}$ in comparison to $\text{Regrowth}_{20\text{-}30}$ and $\text{Regrowth}_{>40}$ ($p<0.05$) (Figure 3.8a). Herb and forb cover were not significantly different among treatments ($p>0.05$; Figure 3.8b).

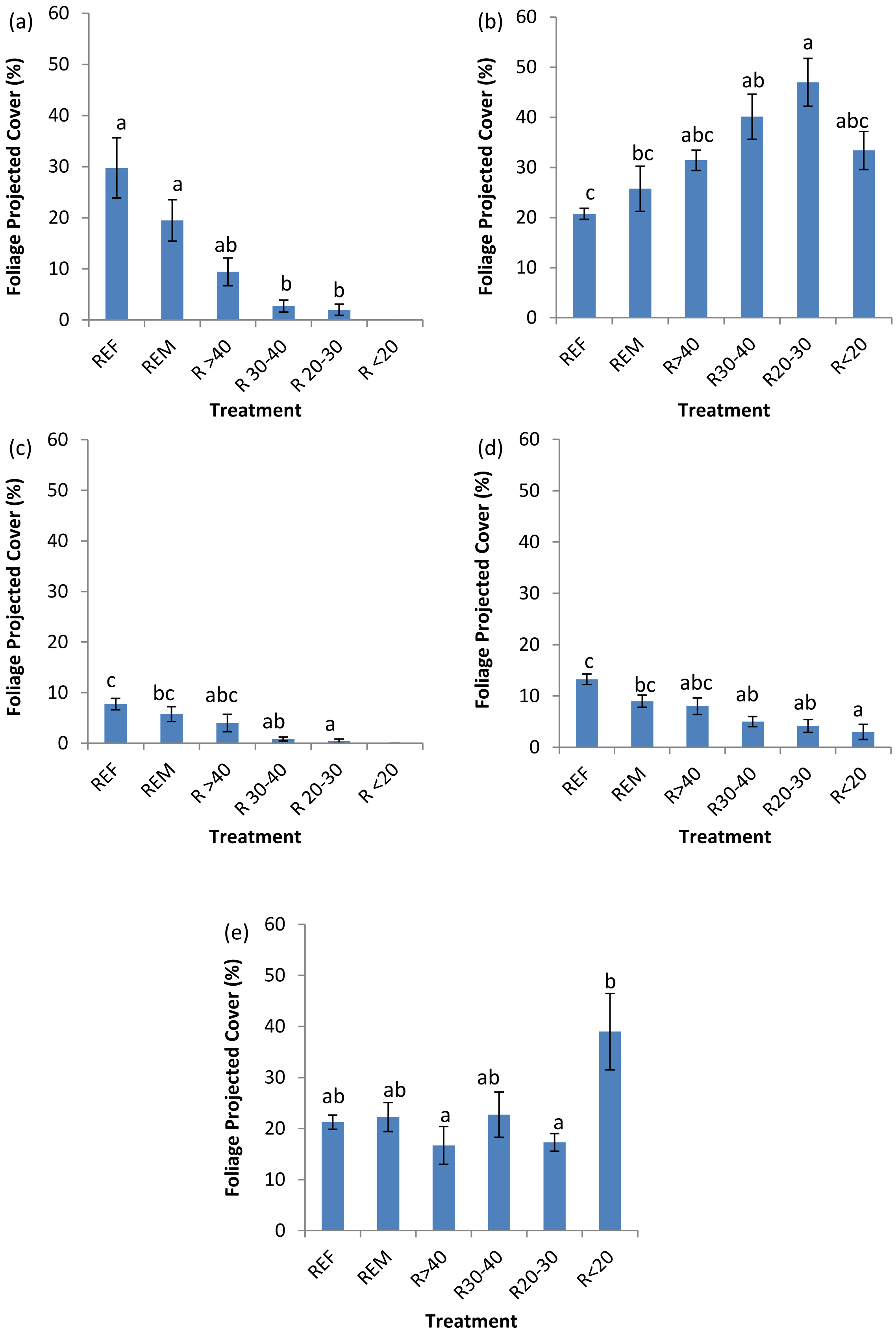


**Figure 3.7.** Mean FPC of (a) trees 10-20m, (b) trees <10m, (c) Shrubs >2m, (d) Shrubs <2m and (e) groundcover for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using natural log (Ln) transformation. (Tukey's post hoc tests, $p < 0.05$).

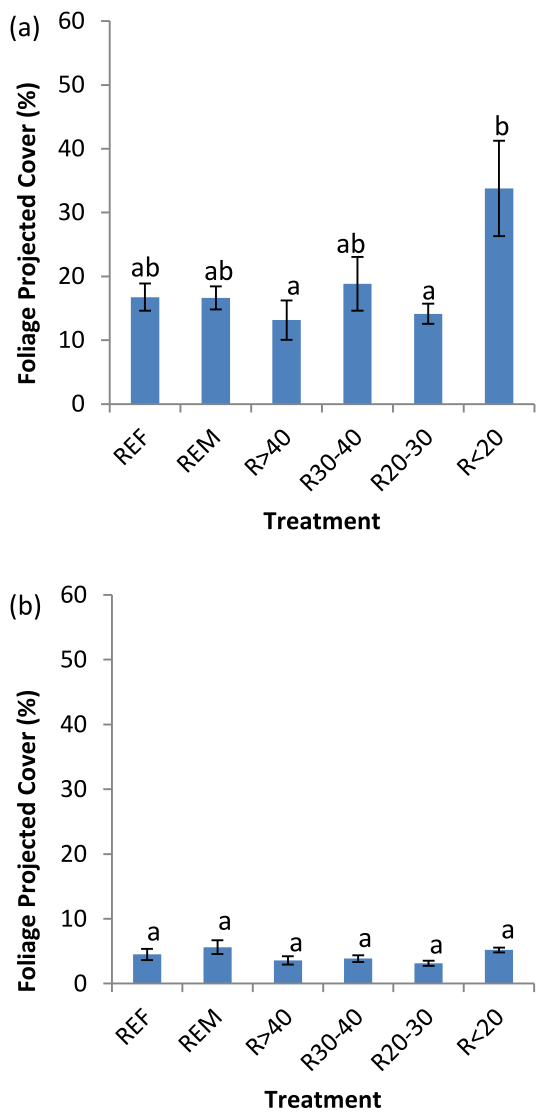


**Figure 3.8.** Mean percentage cover of (a) graminoid and (b) forbs/herbs for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using natural log (Ln) transformation (Tukey's post hoc tests, $p < 0.05$).

Remnant and Reserve sites had a significantly higher percentage cover of large logs (>20cm circumference) compared to regrowth sites less than 40 years since clearing (Figure 3.9a). Log cover, 10-20cm circumference, was also higher in the Reserve sites than $\text{Regrowth}_{20\text{-}30}$ and $\text{Regrowth}_{30\text{-}40}$ ($p<0.05$).

There were no differences in coarse leaf litter ($p>0.05$). Fine leaf litter (<5cm circumference) was significantly higher in both remnant sites and Regrowth$_{>40}$ than Regrowth$_{<20}$ ($p<0.05$) (Figure 3.9d). Cover of bare ground was significantly higher in all regrowth treatments than reserve sites ($p<0.05$), although not significantly different to remnant sites ($p>0.05$) (Figure 3.9e).

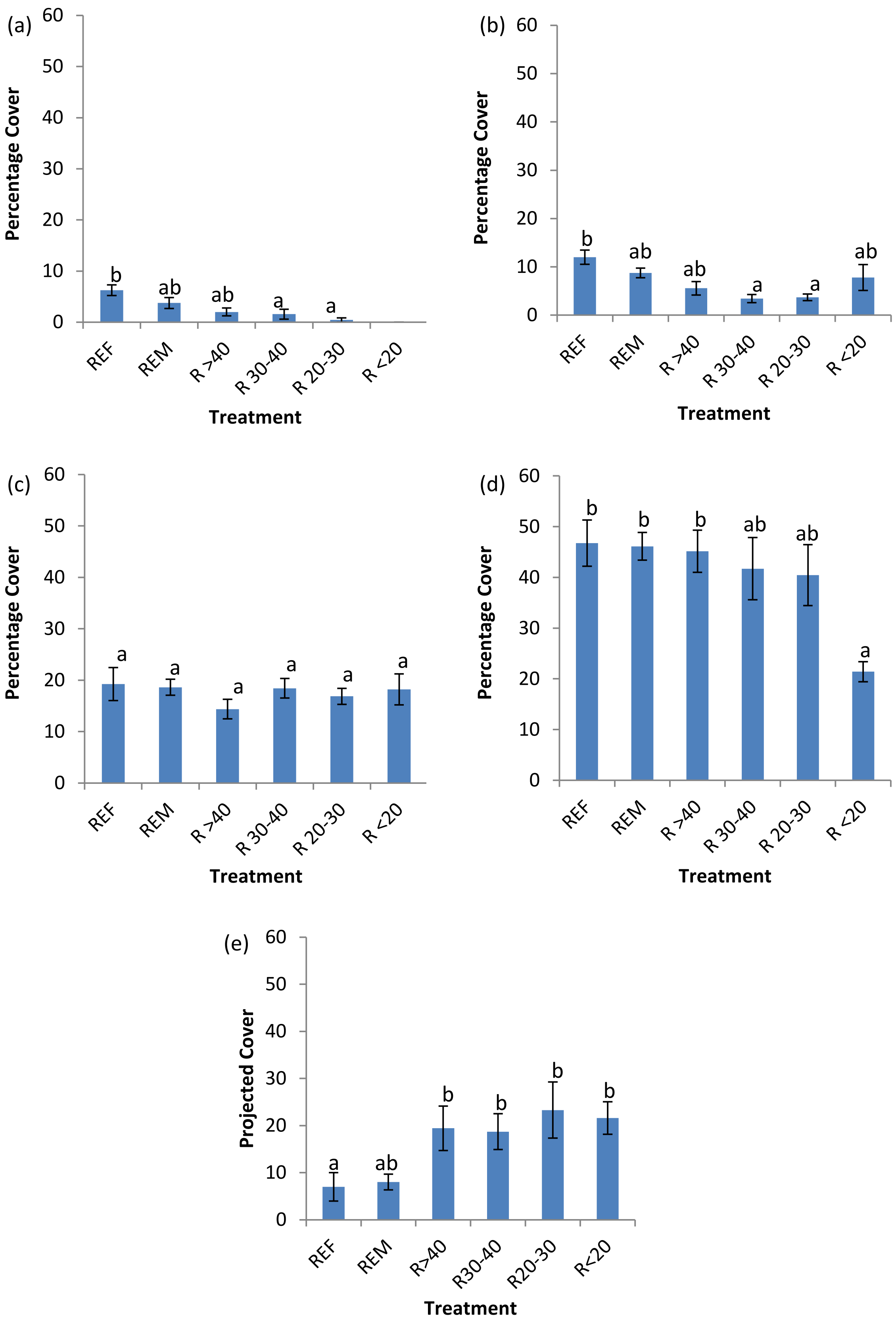


**Figure 3.9.** Mean percentage structural cover for (a) logs >20cm, (b) logs 10 - 5cm, (c) coarse leaf litter (d) fine leaf litter and (e) bareground for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using arcsine transformation (a) and natural log (Ln) transformation (b) (Tukey's post hoc tests, $p < 0.05$).

Leaf litter biomass was significantly lower in Regrowth$_{<20}$ than all other treatments except Regrowth$_{20-30}$ (Figure 3.10). Old regrowth (Regrowth$_{>40}$, Regrowth$_{30-40}$), with the highest estimated mean dry weight, had more leaf litter biomass/m$^2$ than Regrowth$_{20-30}$.

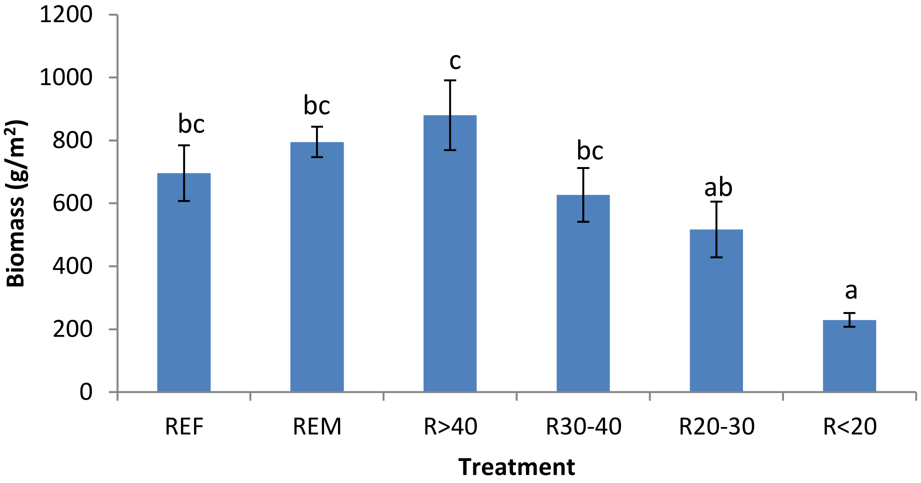


**Figure 3.10.** Mean leaf litter biomass (g/m2) for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using natural log (Ln) transformation (Tukey's post hoc tests, p< 0.05).

### *3.3.2.4 Patterns in structural functional group richness*

Reserve sites had a significantly higher richness of tree species than Regrowth$_{<20}$ (F= 2.721; p<0.05; Figure 3.11a) although no other differences between treatments (Appendix D). Shrub species richness was significantly lower in Regrowth<20 than all other treatments (F= 4.092; p>0.05; Figure 3.11b). Remnant sites were significantly higher in groundcover species richness than Regrowth$_{20-30}$ sites (F= 3.744; p<0.05; Figure 3.11c).

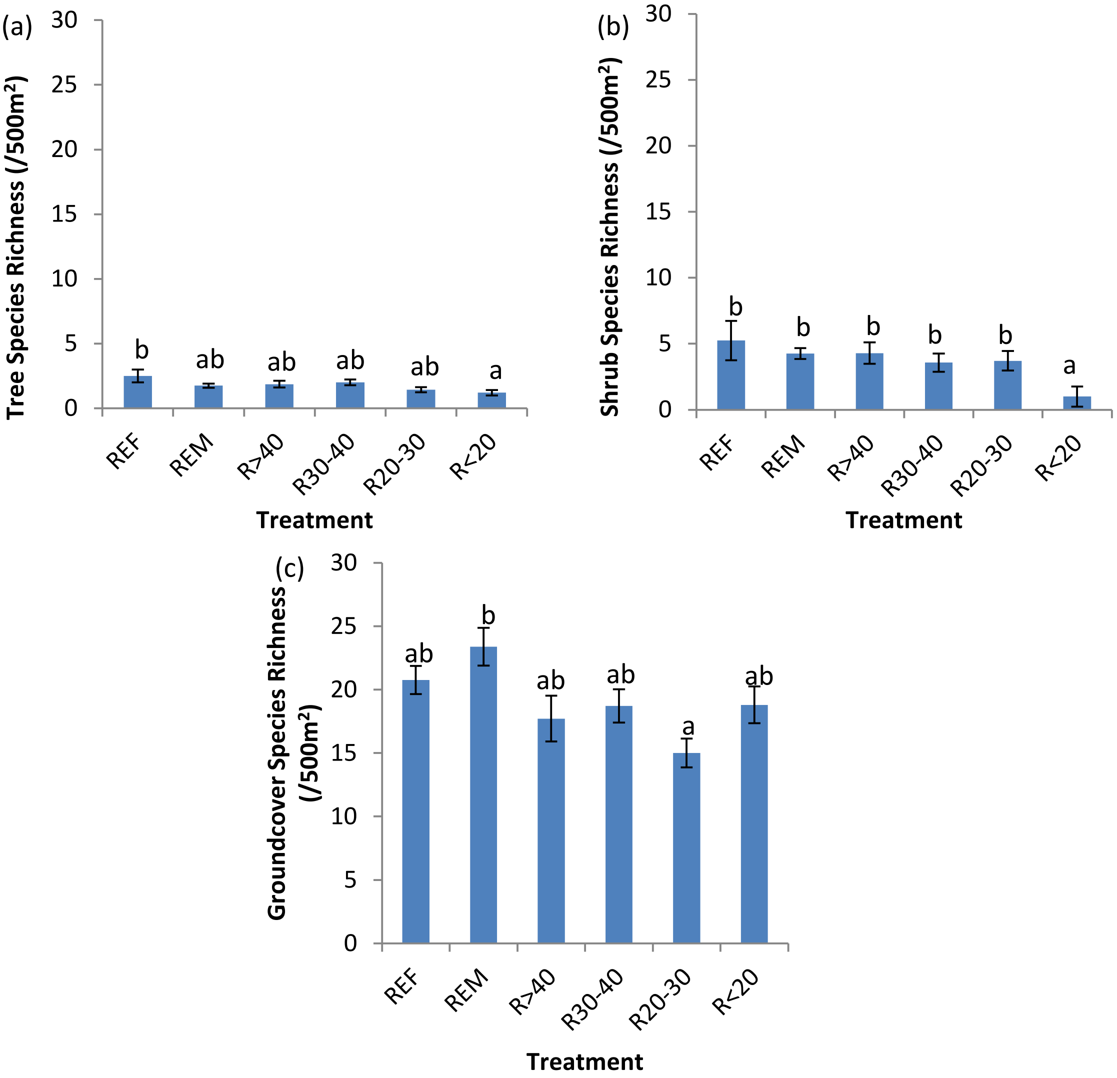


**Figure 3.11.** Mean species richness for tree (a), shrub (b) and ground (c) layer for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using natural log (Ln) transformation (Tukey's post hoc tests, $p < 0.05$).

There were no significant differences between the treatments in relation to herb/forb species richness (F = 1.898; p = 0.122) (Figure 3.12b). Recent regrowth had significantly higher graminoid species richness than Regrowth$_{20\text{-}30}$ sites (F = 2.833; $p < 0.05$; Figure 3.12a), although no further differences between treatments.

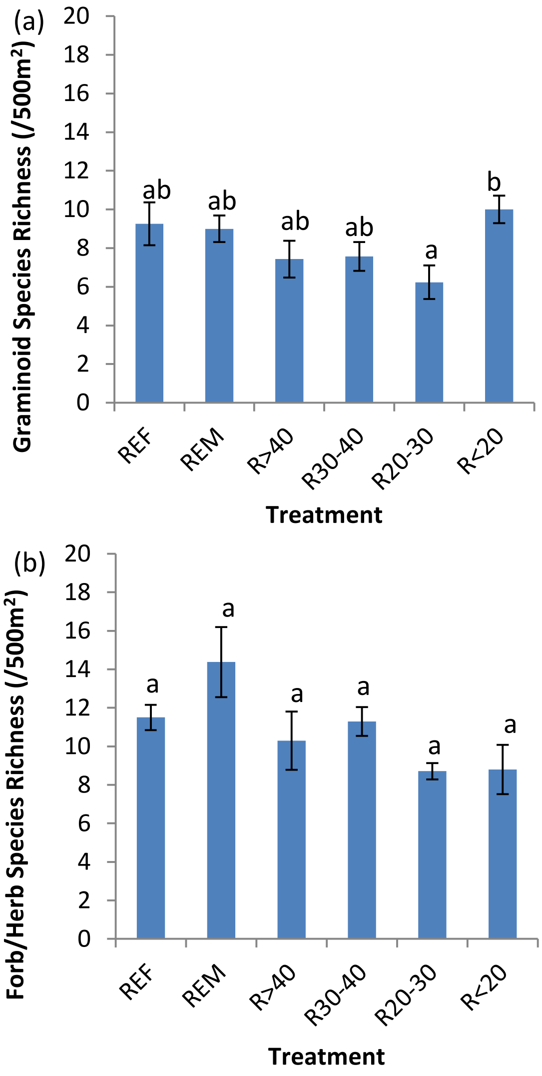


**Figure 3.12.** Mean species richness for graminoid (a) and forb and herb (b) layers for each of the treatments. Error bars are standard error. Means sharing the same letter (e.g. a, b and c) are not significantly different using natural log (Ln) transformation (Tukey's post hoc tests, $p < 0.05$).

## 3.4 Discussion

### 3.4.1 Patterns in Floristic Composition

Plant species recorded across the remnant and regrowth communities in this study were consistent with other studies in Brigalow vegetation in the region (Litchfield, 1975; Johnson, 1984b; Johnson and Burrows, 1994; Bradley *et al.*, 2010). Differences were observed between remnant and regrowth Brigalow vegetation treatments, in relation to floristic composition, with significant differences in the overall species composition (multivariate) between remnant and regrowth treatments and between intermediate and older regrowth and more recent regrowth. This is consistent with other patterns observed in secondary succession in relation to plant composition. For example, Bradley (2006) examined the composition and species richness of woody plant species in Brigalow vegetation and found that while 50% of woody plant species were present shortly after a disturbance (~4yr), it would take over 50 years for regrowth stands to reach 90% of species present in remnant Brigalow communities. Similarly, Aide *et al*. (2000) found that despite recovery of stand structure in secondary tropical forests, species composition was not matched with that of old growth forests (greater than 80 years). Other studies have also found that despite long periods of recovery (~ 100 years), many regrowth communities still differ in species composition and diversity compared with remnant and intact communities (Grau *et al.*, 1997; Turner *et al.*, 1997; Foster *et al.*, 1998).

Total and native species richness generally increased with the age of regrowth, with species richness within regrowth communities over 30 years since disturbance (Regrowth$_{>40}$ and Regrowth$_{30\text{-}40}$) similar to that recorded in remnant communities. This result is supported by Standish *et al*. (2007) who found some remnant Eucalypt woodlands in Western Australia had plant species richness almost double that of regrowth (>40yrs) vegetation patches. The low species richness within recent regrowth may be associated with early successional trends and the low structural complexity compared to older regrowth that allows niche creation and species specialisation (Bazzaz, 1996; Aide *et al.*, 2000). This pattern is also likely to be at least partly driven by the development of stand structural complexity, replacing open grassland species with more shade tolerant species as the overstorey develops (Veblen, 1992), discussed further in section 3.4.2.

Little response in relation to exotic species richness was observed between regrowth, remnant and reserve sites. Although 16 introduced species were observed within the study, the exotic species richness at each site was relatively low. McIntyre & Martin (2001) also experienced limited exotic species response in grassy woodland ecosystems following the removal of tree species. Fensham (1998) notes most exotic species do not significantly affect the integrity of native species colonisation in grassy woodlands on the Darling Downs; however, the conversion from native to exotic colonisation can occur when mechanical disturbance increases. Despite the disturbance experienced during clearing, the relatively rapid recovery of structure and the early dominance of some native species may increase the resilience of these communities to exotic invasion. Some exotic species found during this study (e.g. *Panicum maximum* and *Eragrostis cilianensis*) have also been shown to be poor competitors and can be kept in low abundances with light grazing (Fensham, 1998), which was observed at most regrowth sites (see Chapter 4).

No significant differences in annual richness were observed between remnant, reserve and regrowth 20 years since disturbance. Annual plant species are commonly favoured by perturbations, particularly grazing, compared to perennial species (Lavorel *et al.*, 1997). Despite higher annual richness in regrowth$_{<20}$ compared with $R_{20\text{-}30}$ and $R_{>40}$, the richness of annual plants in all regrowth treatments was not significantly different from remnant and reserve sites. In contrast, perennial plant species required a longer time, 30-40 years, in order to attain equivalent richness to remnant and reserve sites. This is consistent with the literature whereby perennial richness tends to increase over time as they replace annual species that can dominate early successional stages (Bazzaz, 1996; Foster and Tilman, 2000). Due to the ability to delay reproduction until favourable conditions are met, perennial plant species are able to buffer seasonal variation, accumulate growth and store resources, maintaining diversity over the long term (Bazzaz, 1996). Consequently, perennial species are usually indicative of stable conditions within terrestrial ecosystems (Frank, 1968; Aronson *et al.*, 1993).

However the patterns observed with respect to species composition and richness between the treatments does not necessarily apply to stand structure.

### 3.4.2 Patterns in Stand Structure

Following clearing of woody vegetation, a number of factors can influence the regeneration of trees and woody shrubs, impacting community structure development and composition. The cover of grasses and herbaceous species can significantly influence woody plant recruitment following vegetation removal (Zimmerman *et al.*, 2000; Skinner *et al.*, 2010). Graminoid species frequently have buds near the surface and following disturbance, can have a competitive advantage over woody plant species (Wilson, 1998).  Tree recruits compete with herbs and grasses for light, water and nutrients and can result in high tree seedling mortality (Gordon and Rice, 2000; Skinner *et al.*, 2010).  In this study, the relatively rapid re-establishment of tree and shrub structure could be dependent on the combination of high groundcover and the suckering nature of Brigalow following disturbance (Chandler *et al.*, 2007).

Recent regrowth sites had relatively high groundcover, attributed mostly to increased graminoid cover.  This may be due to the rapid growth and development of Brigalow suckers creating a greater area of shade over time.  Following initial clearing, grasses (and forbs/herbs) and the Brigalow suckers would be actively competing for resources including light, water and nutrients.  However, over time (after 20 years, but possibly earlier given the high variability in grass cover in Regrowth$_{<20}$), the Brigalow suckers have reached a height capable of increasing shade over an area.  When overstorey structure is present, the dominance of less shade tolerant grass species, can result in a decrease in grass cover (Tremont and McIntyre, 1994).  The reduction in groundcover may open up potential niches in the understorey for other herbaceous and woody species, more shade tolerant, to germinate and establish themselves within the community (Grubb, 1977).  The reduction in graminoid cover, not surprisingly, coincided with an increase in shrub and tree species richness 20 years since disturbance.  Despite little variation in the richness of ground cover species across treatments, with the exception of regrowth sites 20-30yrs old, the assemblage of species may have changed with the development of the overstorey, reflected in the multivariate comparisons on floristic composition and stand structure. Clarke (2003), for instance, found that shade created by the tree canopy decreased the competitive ability of the herbaceous layer, allowing the recruitment of other woody shrubs and tree species. In this study, similar tree and shrub cover in regrowth and remnant was reached within 20 years; however, a longer period of time (30 to 40 years) was needed to match the cover found in reserve sites.

Assessment of litter cover also identified a number of patterns between the *a priori* community types. The cover of fine leaf litter was equivalent to remnant communities within 20 years following disturbance. Similar leaf litter biomass in comparison to reserve sites was reached within 20-30 years since disturbance. As the cover and size of trees and shrubs increase with time since initial clearing, the aggregation of leaf litter would be expected to rise and the available bare ground to decrease. The accumulation of litter is characteristic of Brigalow communities, with the decreased rate of composition attributed to the seasonably dry conditions typical of these communities (Tunstall and Connor, 1981). Litter is important in changing the physical and chemical properties of soil, as well as manipulating the microclimate within an ecosystem (Facelli and Pickett, 1991; Yates, 1992). Responsible for the release of nutrient back into the soil following decomposition, litter can also influence the evaporation rates of soil moisture and intercept light, affecting the germination and survival of seeds and seedlings (Facelli and Pickett, 1991). The increase in litter cover since disturbance and slow decomposition rate (discussed in Chapter 2) may also have contributed to the reduction in the cover of graminoids over time. Facelli and Pickett (1991b) found increased litter accumulation to significantly decrease the density of *Setaria faberii* and *Panicum dichotomiflorum*, which also impacted seed production.

The presence of larger logs and debris, as would be expected, required a longer time frame, achieving similar levels to remnant and reserve sites after approximately 40 years. The lack of differences observed in relation to coarse litter and logs between regrowth<20 and other age classes would be a result of the clearing method employed. A number of recent regrowth sites had been cleared repeatedly in the past and most recently were pulled but not burnt, including sites 23 and 24 (observation and pers. comm.). This has more than likely resulted in the higher levels of medium sized logs and large standard error (size of new regrowth stems not mature trunks) within Regrowth<20 compared to other regrowth treatments that had been burnt after clearing.

The return of similar structure within Brigalow regrowth appears to be dependent on the development of the shrub and tree layers. As these layers grow and develop, cover of grass decreases and the accumulation of leaf litter and logs increases to levels that resemble remnant vegetation.

### 3.4.3 Alternate states and ecological implications

Based on the age of regrowth assessed in this study, the return of similar plant dynamics and functioning, in terms of structure and composition, to remnant vegetation is yet to be observed. Recent studies by Ngugi *et al.* (2010), using spatial community simulation models, found other community factors, such as stem density, may be influencing vegetative patterns in regrowth. Models estimated Brigalow regrowth communities in central Queensland may require up to 95 years before tree stem densities are naturally thinned to levels similar to remnant Brigalow vegetation (Ngugi *et al.*, 2011). As community structure and diversity return following disturbance, feedback controls driven by environmental properties, such as soil nutrients and biomass, and interactions between component plant species help regulate the development and composition of the community (Tilman *et al.*, 1997).

Multivariate analysis showed general stand structure attributes and complexity to recover to levels resembling remnant vegetation after approximately 40 years since initial clearing (Figure 3.13 *cf* Figure 3.1). While studies on the development of regrowth structure, excluding fire disturbances, is sparse in relation to Australian woodland and forest communities, other studies elsewhere have identified similar time frames in relation to the return of stand structure in regrowth communities following clearing (Guariguata *et al.*, 1997; Aide *et al.*, 2000). Aide *et al.* (2000) assessed secondary wet and dry tropical forests and found, with little soil degradation from past agricultural practices and limited fire disturbance, regrowth achieved similar structure to old growth forests (greater than 80 years) after 40 years of pasture abandonment. Chazdon (2003) also reviewed a number of studies assessing the recovery of tropical forests and found that patches post-agriculture obtained stand structure, soil nutrient stores and species richness more rapidly than species composition.

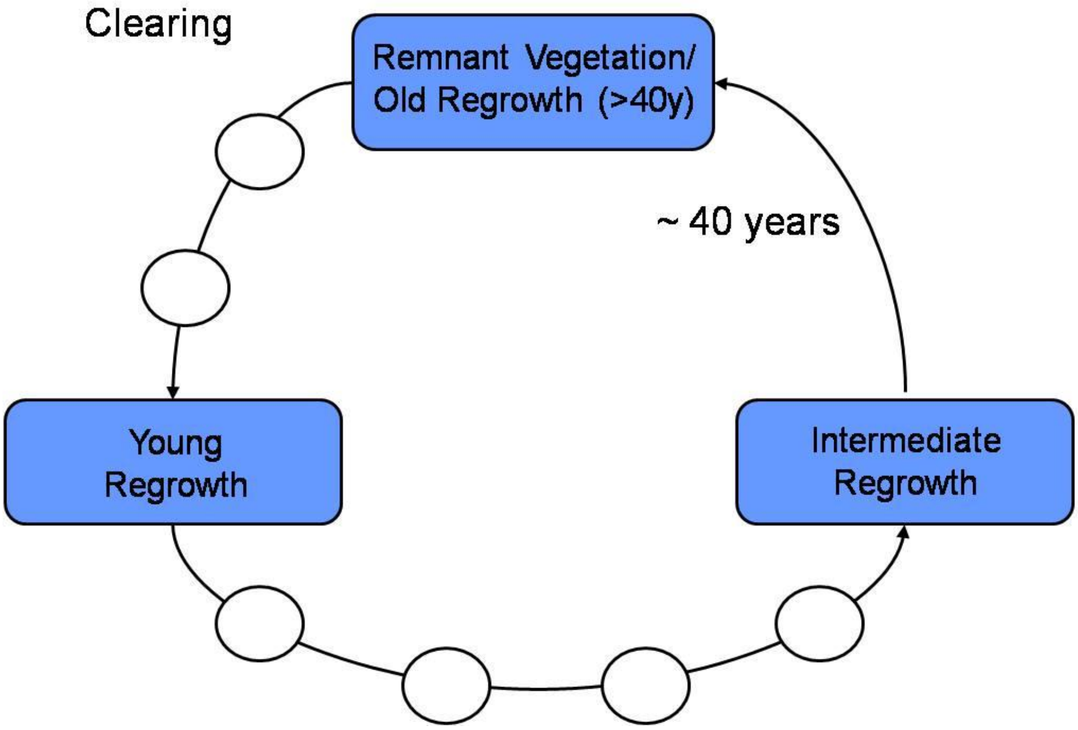


**Figure 3.13.** Sucessional pathway in relation to stand structure within Brigalow regrowth communities. Equivalent structure in regrowth compared with remnant and reserve was reached after approximately 40 years since last disturbed. Circles represent other theoretical transitional states during the recovery process.

In contrast, Kanowski *et al*. (2003) in North Queensland found stand structure in mixed 30 to 40 year old regrowth tropical and subtropical forests, when left to regenerate naturally, had not achieved equivalent structure to that of intact forests. Similar structure was only achieved in actively reforested sites 20-30 years after abandonment (Kanowski *et al*., 2003). Differences in return of 'remnant' structure may be attributed to the life histories of dominant plant species in a given community (Glenn-Lewin and van der Maarel, 1992). The rapid response of resprouting species, such as Brigalow, following disturbance may result in a faster recovery of stand structure compared to seeding species, particularly when dealing with more arid environments (Bazzaz, 1996; Midgley, 1996).

Despite recovery of stand structure observed in this study, floristic composition in regrowth after 40 years (and up to 60 -70y in some sites; Appendix A) still remains significantly different to remnant vegetation (Figure 3.14). Although Brigalow regrowth progresses through a number of transition states, in comparison to Figure 3.13, Brigalow regrowth is not floristically the same as remnant vegetation. Consequently it cannot be determined whether old regrowth represents an alternate stable state or is still in another transition state. As previously discussed, Bradley (2010) also found woody understorey composition within Brigalow regrowth to be significantly different to remnant vegetation. Other studies within forest regrowth fragments (Grau *et al.,* 1997; Turner *et al.,* 1997; Aide *et al.,* 2000) have also reported significant differences in species richness compared to undisturbed remnant forests. So while some ecological components may return to regrowth communities, the ability to recover entire ecosystem functioning evident prior to clearing remains unknown. Determining if Brigalow regrowth >40y is still in a transition phase progressing towards a community similar in composition to remnant, or is in an alternate state, with its successional pathway altered by past anthropogenic disturbance needs to be further investigated.

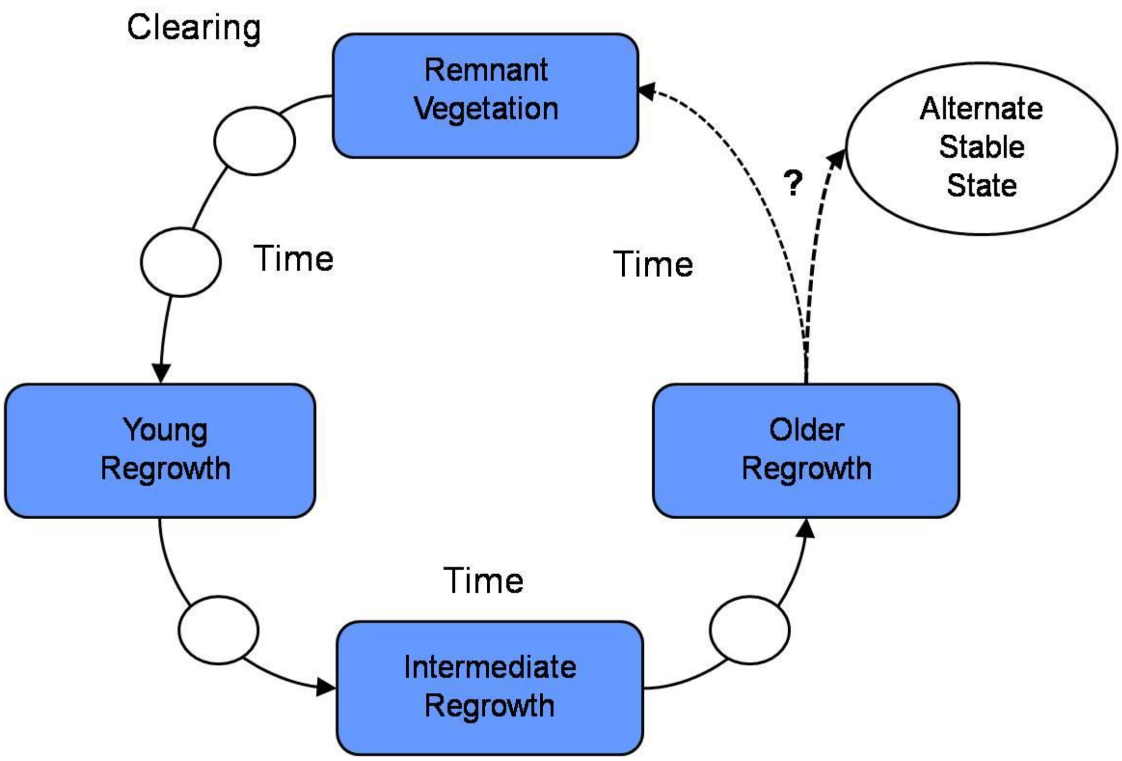


**Figure 3.14.** Successional pathway in relation to floristic composition within Brigalow regrowth communities. Regrowth >40 years is yet to resemble that of reserve or remnant communities. Circles represent other theoretical transitional states during the recovery process.

The confirmation of an alternate stable state, particularly within a short ‘snapshot’ study, can be difficult to determine as not all stable equilibriums can be clearly demonstrated (Suding *et al.*, 2004). Further assessment of species and functional groups present, as well as community structure, are required along different periods of succession to further understand the potential direction of regrowth communities and their contribution to their remnant counterparts and overall landscape resilience. A long term investigation may help determine the exact trajectory of Brigalow regrowth and the community’s resilience under continued management practices. An enhanced understanding of these communities may also assist land managers to guide communities along a desired trajectory to meet an ultimate goal of conservation (Hobbs & Norton, 1996).

Furthermore, other factors may be affecting the observed patterns in flora and structure within Brigalow communities. Environmental variables, such as soil nutrients and surrounding landuses can significantly affect recovery processes following disturbance. For example, the position of a patch within a landscape can influence factors such as hydrology, fluxes of nutrients and patch connectivity, in relation to other surrounding landscape elements, affecting species populations and dynamics (Saunders *et al.*, 1991; Simberloff *et al.*, 1992). Factors such as livestock grazing intensity and the intensity of agricultural development in the matrix can also affect abundance and composition of species in a particular patch of vegetation (Yates *et al.*, 2000; Williams *et al.*, 2006). So, despite the return of similar stand structure creating niches for specialised native species, other environmental factors may be inhibiting seed dispersal and species recruitment between remnant and regrowth communities. The question if patch factors and spatial characteristics influence the floristic and structural patterns observed in this study will be investigated in Chapter 4.

## 3.5 Conclusion

A number of significant patterns in floristic composition and stand structure were identified between the remnant and regrowth Brigalow treatments. The majority of compositional functional groups reached similar levels of richness to remnant after approximately 30 years after clearing. Despite considerable mechanical disturbance, no significant exotic invasion was found across the treatment types, possibly inhibited by the rapid recovery of native species. Structural functional group richness recovered within a shorter time period, with most plant form richness similar to remnant after 20 years.  While regrowth communities greater than 40 years recovered similar stand structure to remnant and reserve communities, floristic composition in old regrowth was still significantly different to both remnant and reserve sites, resulting in a partial rejection of the initial hypothesis.  The redevelopment of similar stand structure to remnant vegetation does not assure the eventual return of equivalent floristic composition.

Whether older regrowth patches are still in transition towards a state equivalent to “remnant” status in relation to composition, or the communities are in fact in an alternate state remains uncertain.  Further investigation is required in order to ascertain the trajectory of older Brigalow regrowth communities within the landscape.

Other potential ecosystem drivers also need to be assessed in order to further understand the floristic patterns observed within regrowth Brigalow communities following significant disturbance. The impact of surrounding spatial attributes and current disturbance regimes along with environmental factors may provide further insight into the patterns observed within these remnant and regrowth communities.

# Chapter 4: Spatial and patch drivers of stand structure and composition in Brigalow remnant and regrowth communities.

## *4.1 Introduction*

Further investigation is required in order to determine if environmental factors may help explain the differential patterns in composition and structure observed across the Brigalow treatments in Chapter 3. Spatial and environmental variables can be important drivers affecting the plant community dynamics and structural complexity within highly fragmented landscapes (Saunders *et al.*, 1991; McIntyre and Hobbs, 1999). External disturbances and inputs, as well as in-patch attributes, can significantly affect the persistence of ecological processes critical in the development of plant species and community dynamics within an ecosystem (Tilman, 1987; Saunders *et al.*, 1991). The ability of disturbed communities to regenerate can be significantly impacted by the type, frequency, intensity and duration of continued disturbance regimes (Denslow, 1980; Hobbs *et al.*, 2007). Identifying and measuring potential ecosystem drivers, both local and extrinsic, is critical in identifying ecosystem thresholds and establishing successful conservation strategies (Groffman *et al.*, 2006). This chapter will investigate the relationship of environmental variables, both spatially and in-patch, to the patterns in floristic composition and stand structure across remnant and regrowth Brigalow treatments in order to help determine why old regrowth remains distinctly compositionally different to remnant.

### 4.1.1 Patch drivers influencing plant community dynamics

There are a number of local habitat characteristics, or patch variables, that can drive changes in biotic composition and structure within an ecosystem. Measures of ecosystem processes assist in providing information on biogeochemical cycles and nutrient cycling necessary for long-term stability of an ecosystem (Ruiz-Jaen and Aide, 2005; Ruiz-Jaén and Aide, 2005). Direct assessment of ecosystem processes is rarely undertaken due to their slow recovery following disturbances and they require multiple measurements of different ecosystem components, resulting in a costly and time consuming process (Ruiz-Jaen and Aide, 2005). Assessment of patch variables, such as soil and nutrient pools, are an indirect measure that can help efficiently assess one aspect of ecosystem processes and have been extensively utilised within studies of community dynamics (Doran and Zeiss, 2000).

Changes within plant community development are highly dependent on soil nutrient availability (Roem and Berendse, 2000). The interdependence between soil chemistry and plant composition, involves numerous complex interactions and feedbacks over time, particularly following a disturbance (Eviner & Chapin, 2003). Soil nutrients and structure interact with plant biogeophysical properties, influencing growth, development and ultimately plant survival and reproduction (Eviner and Chapin, 2003). As the composition of plant species change over time, so too can the levels of nutrients in the soil, leading to further changes in plant recruitment processes (Hooper and Vitousek, 1997). Individual soil characteristics can also influence the richness and diversity of species within a community. Tilman (1987), for example, assessed different aged native oak regrowth and existing remnants in southern Minnesota in the United States and established plots of varying nitrogen gradients using commercial fertiliser ($NH_4NO_3$). Over time, a significant decrease in species richness was observed with increased levels of nitrogen (Tilman, 1987).

Other soil characteristics, such as soil pH and phosphorus, have also been found to be good indicators of plant species diversity and composition (Roem and Berendse, 2000). Gould & Walker (1999) found a significant positive correlation between soil pH and the richness of vascular plants in riverine communities in northern Canada. Dorrough *et al*. (2006) found species richness within Eucalypt grassy woodlands in central Victoria decreased with increased input of phosphorus. The decrease in plant richness, in relation to increased nutrient availability, has been attributed to increased competition between dominant plant species as well as tolerance limitations of some species to high nutrient availability (Dorrough *et al.*, 2006).

Continued local scale disturbances can also play a pivotal role in the composition and development of plants in a community. Plants respond in a variety of ways to different disturbances, as well as disturbance intensities and frequencies (Lavorel *et al.*, 1997). Altered fire regimes can significantly change plant community dynamics, triggering invasion of exotic species and altering abundances of resprouters and seeders across the landscape (Fisher *et al.*, 2009). Watson *et al*. (2009) found reduced fire frequency in Eucalypt woodland (*E. moluccana*, *E. tereticornis* & *E. crebra*) on the eastern coast of Australia transformed these systems into shrublands dominated by Blackthorn (*Bursaria spinosa*).

Livestock grazing can also impact upon plant populations and composition. Fensham *et al*. (1999) found differences in plant functional group diversity in

Queensland Bluegrass (*Dichantheum sericeum*) pastures in response to grazing pressure in central Queensland. A decrease in many native perennial species was observed in relation to increased grazing intensity, favouring a number of more tolerant species, particularly exotics such as *Parthenium hysterophorus* (Parthenium Weed) (Fensham *et al.*, 1999). Livestock grazing can also impact community structure within woodlands, reducing the biomass of understorey grasses, herbs and litter deposits, resulting in further changes in species composition and diversity (Bakker, 1998; Landsberg *et al.*, 2002). Robertson & Rowling (2000) surveyed riparian Eucalypt woodlands on the Murrumbidgee floodplain in southern New South Wales and found significantly less groundcover biomass and leaf litter within grazed sites compared to ungrazed sites. As well as containing significantly higher cover of bare ground, grazed sites also had three times fewer seedlings and saplings compared to ungrazed sites (Robertson and Rowling, 2000).

Other factors, such as patch geometry may also affect plant community composition and structure within fragmented landscapes (Ewers and Didham, 2007). Positive relationships between patch size and species richness have been observed within a number of habitat types with a tendency of larger patches to contain more heterogeneity than small habitat areas, enabling the creation of niches for specialist species (Cagnolo *et al.*, 2006). A nation-wide survey of lowland woodlands in Great Britain, for example, found larger patches had significantly higher herbaceous plant species richness compared to smaller patches (Petit *et al.*, 2004). Similarly, Krauss *et al*. (2004) found species richness of habitat specialist and generalists increased with area within calcareous grasslands in central Germany.

The shape of a patch can also affect community organisation and interactions, through associations with 'edge effects', within the landscape (Saunders *et al.*, 1991; Ewers and Didham, 2007). Laurance *et al*. (1998), for example, found increased tree mortality and turnover rates within 60 m of rainforest patches in the central Amazon, attributed to changed forest microclimate and wind turbulence. Young & Mitchell (1994) found plant composition to differ between edges and interior of Podocarp-Broadleaf forests in northern New Zealand. Changes in microclimate resulted in differences in the abundance and occurrence of specific plant species between edge and interior sites (Young and Mitchell, 1994). Changes to plant species dynamics associated with edge effects are greatest within small irregular patches where the ratios of edge to area are higher (Laurance *et al.*, 1998). While considerable studies have assessed the effects and importance of edges on vertebrate (Sisk *et al.*, 1997; Schlaepfer and Gavin, 2001; Fletcher, 2005) and invertebrate (Dauber and Wolters, 2004; Ewers and Didham, 2008) species composition and richness, little is known in relation to the condition and viability of plant communities of Australian woodlands (Batterham, 2008). While some authors have noted no differences in tree density or recruitment patterns in Eucalypt woodlands in relation to patch shape (Norton *et al.*, 1995; Batterham, 2008), evidence suggests reproductive cycles can be impacted (Cunningham, 2000). Cunningham (2000) surveyed *Acacia brachybotrya* and *Eremophila glabra* populations within mallee woodlands in central New South Wales. Individual plants within linear remnant strips of vegetation were found to receive less pollen than those in nearby reserves, resulting in a reduced fruit set, highlighting the potential threats to population viability and biodiversity within fragmented landscapes (Cunningham, 2000).

To fully understand the effects of local scale factors on the variability of species responses, one also needs to investigate surrounding landscape level variables that may be influencing ecosystem processes (Mazerolle and Villard, 1999).

### 4.1.2 Spatial drivers influencing plant community dynamics

Spatio-temporal characteristics can also play an important role in the development of plant species and community structure within fragmented landscapes (Waldhardt and Otte, 2003). The diversity and abundance of plant species within agricultural landscapes is highly dependent on the current and previous land-uses and intensities, as well as the patterns and dynamics in the surrounding matrix and landscape (Waldhardt and Otte, 2003). Changes in land-use, land-use intensity and decreases in surrounding habitat can, for instance, result in a loss of rare and specialist species, at times reducing species richness and favouring more generalist species within an environment (Saunders *et al.*, 1991; Vellend *et al.*, 2007). Vellend *et al.* (2007) analysed worldwide studies of alpha diversity within regrowth and remnant forests and found that within regrowth, surrounding land uses modified for agricultural production acted as ecological filters, reducing plant diversity by impeding the dispersal of different species.

The expansion and intensification of the agricultural industry can also result in changed fire regimes, increased erosion, from both wind and water, soil salinisation, and increased nutrient inputs, impacting on remnant and other vegetation patches in the landscape (Saunders *et al.*, 1991; Williams *et al.*, 2006). Williams *et al*. 2006 found increased plant population extinctions in remnant grasslands, attributed to increased exogenous disturbance from a rise in urban and rural development in the surrounding matrix over a 20 year period.  Similarly, Guirado *et al*. (2006) found forest fragments surrounded by urban development in north eastern Spain contained significantly higher richness of common forest plant species compared to similar sized forest patches surrounded by cropping.

Different surrounding land uses can also increase the possibility of exotic invasions into remaining vegetation patches within the landscape.  Borgmann & Rodewald (2005) examined riparian forests along a rural – urban gradient and found increased cover of *Lonicera* spp., an exotic shrub, in vegetation surrounded by urban development compared to areas of agriculture.

The arrangement and structure of landscape features, or landscape configuration, can also have a significant effect on local diversity and community structure (Dauber *et al.*, 2003).  The quality and different characteristics of the matrix can significantly influence the dispersal success of species between patches, as well as factors such as nutrient fluxes and hydrology (Saunders *et al.*, 1991; Simberloff *et al.*, 1992; Debinski, 2006).  Distance between patches and matrix permeability is crucial in the long-term persistence of vegetation patches and the recovery of highly disturbed communities (Suding *et al.*, 2004; Debinski, 2006).  Dzwonko (1993) found distance to remnant oak-pine woodlands significantly influenced the floristic composition within regrowth communities.  The fragmentation and loss of vegetation within landscapes reduces the regional pool of propagules that can assist in recolonisation and development following disturbances (Suding *et al.*, 2004).  Landscape configuration can also influence ecosystem processes, such as primary productivity. Ludwig *et al*. (1999) also assessed landscape patchiness using a simulation model

(SEESAW – Simulation of the ecology and economics of semi arid woodlands) and found patch shape to influence soil water availability directly impacting annual primary productivity.

Agricultural land use affects a significant proportion of the landscape and its contribution to regional biodiversity is imperative within conservation initiatives, not only for biota, but the continuance of ecosystem services (Tscharntke *et al.*, 2005). Identifying matrix parameters and drivers of community change are important in determining the impacts on rates of recovery of fragmented communities following significant disturbance, and their ultimate persistence in the landscape (Altieri, 1999; Dauber *et al.*, 2003).

### 4.1.3 Study aims

In this chapter, I examine the relationship between environmental factors and floristic composition and stand structure of remnant and regrowth Brigalow vegetation in order to help explain trends observed in Chapter 3. The correlation of soil chemistry, landscape connectivity and proportion surrounding landuse on vegetative composition and structure are analysed. How these factors relate to species richness, native and exotic richness as well as strata cover classes across Brigalow vegetation in general, encompassing both remnant and regrowth communities, are also be assessed.

The hypotheses assessed in this chapter are:

- Spatial and patch variables explain the differential patterns in stand structure and floristic composition across reserve, remnant and different aged Brigalow regrowth vegetation (Regrowth$_{>40}$, Regrowth$_{30\text{-}40}$, Regrowth$_{20\text{-}30}$ and Regrowth$_{<20}$).

- Spatial and patch variables explain patterns in total species richness, life origin functional group richness or strata cover classes in Brigalow remnant and regrowth communities.

## 4.2 Methodology

### 4.2.1 Study Sites

The Brigalow sites sampled in Chapter 3 for composition and structure were further examined in terms of habitat and landscape (spatial) relationships across the six treatments:

1) Reserve remnant Brigalow vegetation never been cleared and protected within reserves and national parks (Reserve).
2) Remnant Brigalow vegetation never been cleared (Remnant).
3) Brigalow regrowth cleared over 40 years ago (Regrowth$_{>40}$)
4) Brigalow regrowth cleared between 30 – 40 years ago (Regrowth$_{30\text{-}40}$)
5) Brigalow regrowth cleared between 20 – 30 years ago (Regrowth$_{20\text{-}30}$)
6) Brigalow regrowth cleared < 20 years ago (Regrowth$_{<20}$)

More detail of the study site characteristics and design can be found in section 2.3 of Chapter 2.

### 4.2.2 Field techniques

Sites were sampled between 10 February and 20 May 2010. Plant species frequency score was calculated using nested quadrats and foliage projected cover of strata was estimated over a 500m$^2$ plot. Further detail on the collection of species data is provided in Chapter 3.

A complete list of habitat factors investigated is provided in Table 4.1. Livestock grazing was scored on from 0-3 (0 = absent; 1 = minor; 2 = moderate; 3 = severe) based on evidence of understorey herbivory and stock faeces (cow ‘pats’). The presence or absence of gilgai (see section 2.1.2) within the site was also recorded and scored based on the degree of undulation (1 = light (<50cm deep); 2 = moderate (50-100cm deep); 3 = heavy (>100cm deep). Overstorey tree stem density was also recorded in a 10m x 20m plot by direct counting.

Soil samples were collected at each site sampled using four transects, intersecting at the centre of a 500m$^2$ plot. Following the collection of leaf litter biomass (Chapter 3.2.2), a 125g soil sample, consisting of the first 10 cm of topsoil (see (Adams *et al.*, 1994), was taken at the centre of the biomass quadrat for each of the four transects using a PVC soil corer. The four soil samples acquired at each site were bulked together, labelled and stored in a freezer for later analysis. Soils were analysed commercially by SGS Food and Agriculture Laboratory, in Toowoomba, for pH, nitrogen, phosphorus, potassium, calcium, magnesium, sodium, organic carbon and cation exchange capacity (Table 4.1). Sampling at the base of gilgai was excluded to avoid potential variation in soil nutrient concentration (Russell *et al.*, 1967).

Land-use data for each site was derived using Land-use imagery (1999) provided by the Queensland Department of Environment and Resource Management (DERM). Using ArcGIS V9 (ESRI, 2008), a 2km buffer was produced around each site and intersected with the Queensland Land Use and Management Mapping (1999). Land-use was classified into five land-use categories; 1) grazing in natural vegetation; 2) cropping; 3) water bodies, 4) production forestry and 5) national park cover (Table 4.1). Other minimal land-uses, such as public services and recreation, were omitted from the analysis. Surrounding area of remnant and regrowth Brigalow (RE 11.4.3) was also calculated for each site using ArcGIS (ESRI, 2008) buffer tool (2km) on the Queensland Regional Ecosystem Mapping (Version 5, 2003) and Regrowth Mapping (Queensland Herbarium, 2008). The area, in hectares, was recorded for each land-use category and log transformed.

Other landscape factors, including site area and distance to nearest remnant (measured from centroid to centroid), were also recorded for each site. Perimeter to core ratio, used to determine how much of the patch is influenced by edge effects (Saunders *et al.*, 1991; Ewers and Didham, 2007), was also calculated for each site (Table 4.1).

**Table 4.1**. Patch and spatial variables recorded and associated unit of measurement, transformation performed and abbreviated code.

| Variable | Transformation | Code | Method |
|---|---|---|---|
| **Patch Variables** | | | |
| Grazing Intensity (score) | - | Grazing Int. | Score |
| Gilgai Presence/ Depth (score) | - | Gilgai | Score |
| Tree Stem Density (/200m$^{2)}$ | Log10 | Stem Density | Direct counting |
| pH | - | pH | 1:5 – Soil: Water (Rayment and Higginson, 1992) |
| Cation Exchange (meq/100g) | Log10 | Cation Exch. | Ammonium acetate, pH7 extraction no adjustment for soluble salts (Rayment and Higginson, 1992) |
| Nitrate/ Nitrogen (mg/kg) | Log10 | N | 1:5 – Soil: Water, Colourimetric determination (Rayment and Higginson, 1992) |
| Phosphorus (mg/kg) | Log10 | P | Colwell – 0.5M bicarbonate, pH8.5 (Colwell, 1963) |
| Potassium (mg/kg) | Log10 | K | Ammonium acetate, pH7 extraction no adjustment for soluble salts (Rayment and Higginson, 1992) |
| Calcium (mg/kg) | Log10 | Ca | Ammonium acetate, pH7 extraction no adjustment for soluble salts (Rayment and Higginson, 1992) |
| Sodium (mg/kg) | Log10 | Na | Ammonium acetate, pH7 extraction no adjustment for soluble salts (Rayment and Higginson, 1992) |
| Magnesium (mg/kg) | Log10 | Mg | Ammonium acetate, pH7 extraction no adjustment for soluble salts (Rayment and Higginson, 1992) |
| Organic Carbon (%) | - | OC | Walkley Black Method (Walkley and Black, 1934) |

| Variable | Transformation | Code | Method |
|---|---|---|---|
| **Spatial Variables** | | | |
| Forestry area (ha) | Log10 (log10 +1) | Forestry | |
| Cropping area (ha) | Log10 (log10 +1) | Cropping | |
| Grazing area (ha) | Log10 (log10 +1) | Grazing | |
| National Parks (ha) | Log10 (log10 +1) | Nat. Parks | |
| Remnant Area (ha) | Log10 (log10 +1) | Remnant 2km | |
| Regrowth Area (ha) | Log10 (log10 +1) | Regrowth 2km | |
| Water Bodies (ha) | Log10 (log10 +1) | Water | |
| Patch Area (ha) | Log10 (log10 +1) | Site Area | |
| Perimeter: Area | - | P:A | |
| Distance to Closest Remnant (km) | - | Distance 2 Rem. | |

### 4.2.3 Numerical Analyses

Correspondence Analyses (CA) was performed on the floristic composition data and arcsine transformed structure data to identify patterns evident within the data, using CANOCO V4.5 (Ter Braak and Smilauer, 1991). Canonical Correspondence Analysis (CCA) was performed on composition/structure data and environmental variables (Patch and Spatial variables) in order to explain the variation in community composition evident in the CA (Leps and Smilauer, 2003). Canonical correspondence analysis is a multivariate direct gradient technique whereby the axes of an ordination are constrained to linear combinations of explanatory environmental variables (represented by vectors) (Leps and Smilauer, 2003). Highly correlated explanatory variables, if evident, were removed from the analysis and the process repeated.

Correlation analysis was performed on all spatial and habitat variables, using Pearson's correlation in SPSS V17 (SPSS Inc., 2007). Mann-Whitney U test was performed on the spatial and patch variables between remnant and Regrowth$_{>40}$ to determine which environmental factors may be responsible for differences in floristic composition. This non-parametric test compares the medians across the sites and then ranks the scores across the two groups to determine the significance (Pallant, 2011).

Stepwise multiple regression, using SPSS V17 (SPSS Inc.,2007), following the standardisation of spatial variables, was used to identify spatial and patch factors influencing total, native and exotic species richness and strata cover classes (tree cover, shrub cover, groundcover, herb/forb cover and graminoid cover) across the 38 remnant and regrowth sites. Analyses were performed with the GIS-derived spatial factors and patch factors as explanatory variables. Some highly correlated factors were removed in order to create a more parsimonious set of parameters. However, inferences from these analyses will only be made in relation to broad patterns relating to variables used.

## *4.3 Results*

### 4.3.1 General patterns in environmental variables.

No evidence of livestock grazing was recorded within reserve sites, with mean intensity ranging between 0.4 and 2.0 of the remaining five treatments (Table 4.2; Appendix H). Mean soil nitrogen and phosphorous levels ranged between 3.6 to 6.6 mg/kg and 4.8 to 16.4 mg/kg respectively. Organic carbon varied between 1.2% and 1.6% within the Brigalow treatments. Mean stem density was highest in Regrowth$_{<20}$, averaging around 207 stems/200m$^2$ and lowest within reserve sites at approximately 53 stems/ 200m$^2$ (Table 4.2).

No cropping or water bodies were recorded in the 2km buffer zone around Reserve sites (Appendix H). The mean area of cropping and grazing ranged between 609 to 984ha and 121 to 419ha respectively, between the six treatments. Regrowth$_{<20}$ had the highest mean distance from the closest remnant patches of 3.7km compared to Regrowth$_{30-40}$ which had the shortest mean distance of 1.9km to the nearest remnant patch. The highest mean area of regrowth vegetation in the surrounding landscape, approximately 155ha, was within Regrowth$_{<20}$ and averaged lowest around Reserve sites at 1.6ha.

**Table 4.2.** Summary of mean environmental variables for each treatment. Standard errors are shown in parentheses.

| Variable | Reserve | Remnant | Regrowth$_{>40}$ | Regrowth$_{30\text{-}40}$ | Regrowth$_{20\text{-}30}$ | Regrowth$_{<20}$ |
|---|---|---|---|---|---|---|
| **Patch Variables** | | | | | | |
| Grazing Intensity (score) | 0.0 (0.0) | 0.4 (0.2) | 0.6 (0.4) | 1.3 (0.4) | 1.0 (0.6) | 2.0 (0.3) |
| Gilgai Presence/ Depth (score) | 1.3 (0.5) | 1.0 (0.3) | 1.1 (0.3) | 1.6 (0.5) | 1.0 (0.5) | 0.6 (0.2) |
| Tree Stem Density (/200m$^{2)}$ | 53.3 (10.5) | 56.5 (6.5) | 60.9 (3.7) | 114.7 (36.5) | 192.2 (45.2) | 206.8 (39.4) |
| pH | 7.0 (0.2) | 7.1 (0.2) | 7.0 (0.3) | 7.1 (0.3) | 7.4 (0.2) | 7.4 (0.4) |
| Cation Exchange (meq/100g) | 18.1 (2.2) | 28.4 (2.5) | 26.4 (3.5) | 26.7 (4.4) | 27.5 (3.1) | 28.5 (4.7) |
| Nitrate/ Nitrogen (mg/kg) | 5.5 (1.3) | 5.3 (0.5) | 6.3 (1.5) | 3.6 (0.7) | 4.2 (1.6) | 8.6 (1.7) |
| Phosphorus (mg/kg) | 4.8 (1.1) | 6.4 (1.5) | 6.9 (1.0) | 16.4 (6.9) | 9.8 (2.0) | 6.2 (1.1) |
| Potassium (mg/kg) | 285.3 (30.8) | 357.6 (23.2) | 406.0 (59.3) | 432.9 (92.7) | 333.3 (52.0) | 353.0 (63.8) |
| Calcium (mg/kg) | 2257.5 (175.5) | 3968.8 (500.0) | 3962.9 (601.6) | 4052.9 (698.6) | 4028.3 (609.6) | 3944.0 (868.7) |
| Sodium (mg/kg) | 160.5 (57.8) | 225.3 (57.4) | 128.7 (46.8) | 133.9 (67.1) | 165.3 (23.0) | 364.4 (40.7) |
| Magnesium (mg/kg) | 640.3 (165.9) | 796.1 (79.1) | 600.3 (45.7) | 569.1 (104.6) | 696.5 (32.7) | 808.6 (55.4) |
| Organic Carbon (%) | 1.6 (0.1) | 1.6 (0.1) | 1.5 (0.1) | 1.4 (0.2) | 1.2 (0.1) | 1.3 (0.1) |

| **Variable** | **Reserve** | **Remnant** | **Regrowth$_{>40}$** | **Regrowth$_{30-40}$** | **Regrowth$_{20-30}$** | **Regrowth$_{<20}$** |
|---|---|---|---|---|---|---|
| **Spatial Variables** | | | | | | |
| Forestry area (ha) | 252.9 (147.1) | 0.0 (0.0) | 57.0 (52.8) | 123.8 (123.8) | 0.0 (0.0) | 21.7 (14.5) |
| Cropping area (ha) | 0.0 (0.0) | 120.8 (39.7) | 351.0 (88.2) | 175.3 (81.5) | 418.8 (106.8) | 179.4 (117.3) |
| Grazing area (ha) | 609.4 (203.1) | 983.7 (41.2) | 634.4 (47.0) | 809.5 (121.8) | 669.0 (98.2) | 907.9 (110.5) |
| National Parks (ha) | 248.0 (248.0) | 0.0 (0.0) | 66.7 (66.7) | 0.0 (0.0) | 6.2 (6.2) | 0.0 (0.0) |
| Remnant Area (ha) | 1056.4 (985.1) | 55.5 (10.5) | 618.5 (541.8) | 48.6 (21.4) | 27.1 (13.9) | 5.7 (2.6) |
| Regrowth Area (ha) | 1.6 (1.6) | 16.2 (9.4) | 61.0 (16.2) | 107.5 (38.8) | 74.1 (30.1) | 154.7 (30.7) |
| Water Bodies (ha) | 0.0 (0.0) | 0.6 (0.6) | 0.9 (0.7) | 0.6 (0.4) | 9.85 (7.3) | 0.3 (0.3) |
| Patch Area (ha) | 995.3 (952.8) | 42.1 (16.4) | 25.3 (3.8) | 65.7 (36.7) | 17.5 (2.4) | 50.7 (15.6) |
| Perimeter: Area | 11.5 (3.9) | 16.0 (2.8) | 17.1 (2.6) | 13.9 (2.6) | 16.0 (1.3) | 13.5 (1.5) |
| Distance to Closest Remnant (km) | 2.2 (0.6) | 2.0 (0.3) | 2.2 (0.5) | 1.9 (0.4) | 2.5 (0.5) | 3.7 (0.8) |

### 4.3.2 Correlation of spatial and habitat variables on floristic composition and stand structure within Brigalow communities.

Correspondence Analysis (CA) of floristic species composition (Figure 4.1a) reflected a similar pattern to the nMDS ordination produced in Chapter 3 (Figure 3.2a). A gradient from reserve and remnant sites through older regrowth to more recent sites can be observed. However, Regrowth$_{<20}$ sites are more dispersed within the CA ordination (Figure 4.1a). Sites 4 and 14 were identified as outliers and were removed from further analysis.

CCA of composition constrained to the spatial and patch variables, explained 22% of the variance in the first two axes (Axis 1= 0.341; Axis 2 = 0.217; F= 1.316; p= 0.002). Each of the vectors in the ordination from the CCA represent an environmental axis (extends from either end of the arrow). The length of each vector is equal to the rate of change of the related variable across the ordination, providing an indication of its importance (Le Brocque and Buckney, 1995a). The main gradient across Reserve/ Remnant through older regrowth to recent regrowth was associated with increased grazing intensity and proportion of regrowth in the landscape (Regrowth 2km), high soil nutrients (Phosphorus, Potassium, Cation Exchange and to a lesser extent magnesium) and stem density, low proportion of forestry in the landscape and smaller patch area (Figure 4.1b). Regrowth$_{<20}$ sites were also characterised by higher distance to remnants in the landscape and stem density. Variation within Regrowth$_{<20}$ was explained by the proportion of water bodies in the landscape and soil nitrogen.

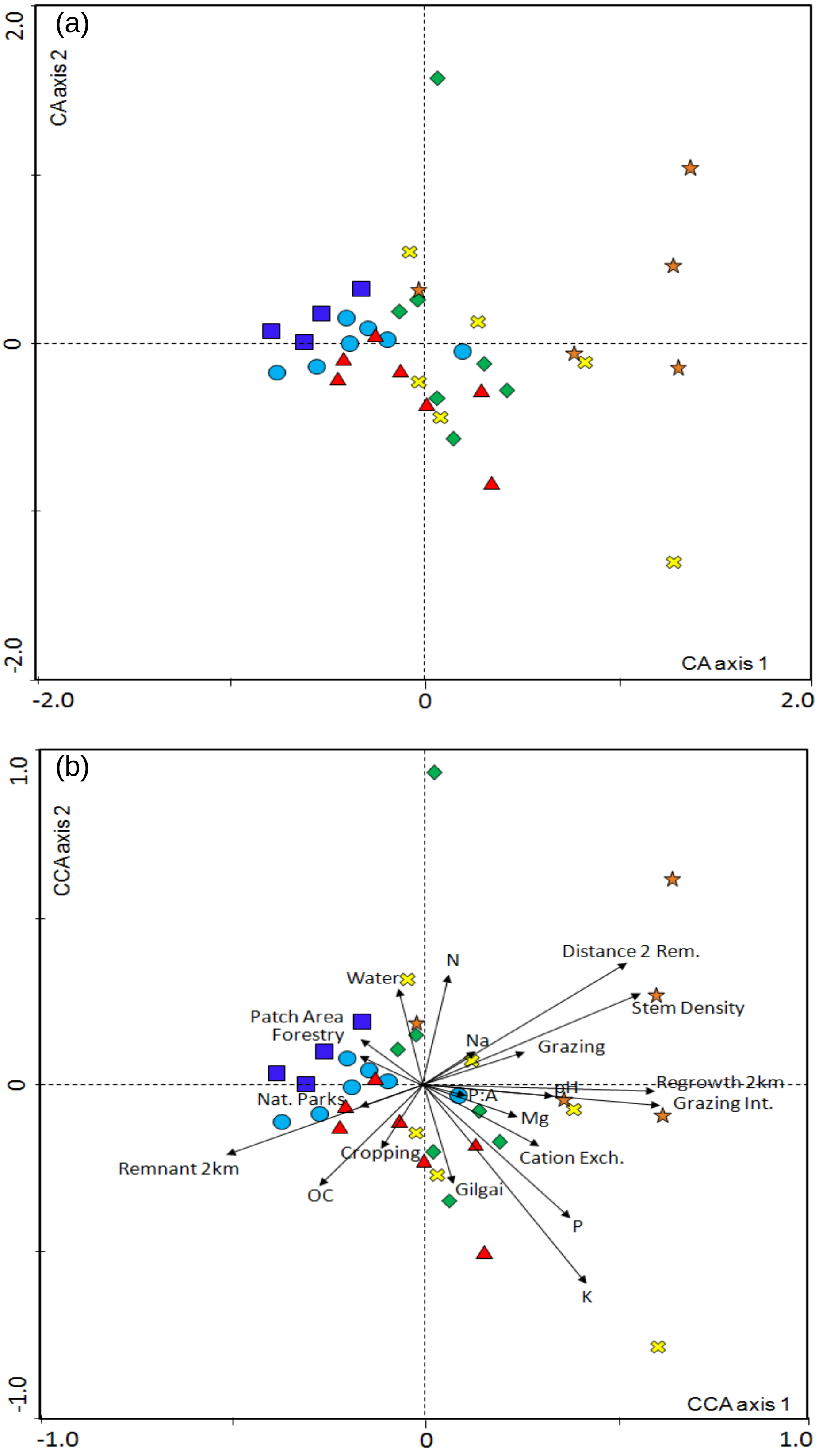


**Figure 4.1**. a) CA ordination of sample floristic composition and b) CCA ordination of sample floristic composition and patch and spatial variables (Eigenvalues x-axis = 0.347, y-axis = 0.212; F=1.316; p= 0.002). ■ = Reserve remnant, ● = Remnant, ▲ = Regrowth$_{>40}$, ✖ = Regrowth$_{30\text{-}40}$, ◆ = Regrowth$_{20\text{-}30}$, ★ = Regrowth$_{<20}$.

CA of stand structure showed a gradient in relation to the six treatment types, from remnant and reserve through to more recent regrowth (right to left; Figure 4.3a). Forty-six percent of the variation in stand structure was explained by the first two axes of the ordination. While Reserve sites were distinguished from all regrowth treatments, considerable overlap was evident between remnant and regrowth sites. Regrowth$_{<20}$ was discernible from Remnant sites but not from older regrowth (Regrowth$_{>40}$, Regrowth$_{30\text{-}40}$) (Figure 4.3a).

CCA of stand structure constrained by spatial and patch variables increased the explained variance between sites to 71.7% within the first two axes (F= 1.581; p= 0.05). Spatial factors were strong correlates explaining structural differences in remnant and reserve sites compared to regrowth sites (Figure 4.2b). A gradient can be seen across axis 1 from Reserve through to Remnant and older regrowth to recent regrowth sites (right to left). This gradient is associated with low proportion of regrowth and grazing in the landscape in Reserve sites and a higher proportion of natural vegetation in the landscape (Remnant 2km, National Parks and Forestry) and patch area. Grazing intensity and, to a lesser extent, soil phosphorus are also associated with this gradient. Soil factors (Nitrogen, Cation exchange) and the proportion of water bodies and cropping in the landscape explained the variation within the older regrowth sites (Regrowth$_{>40}$, Regrowth$_{30\text{-}40}$) (Figure 4.2b).

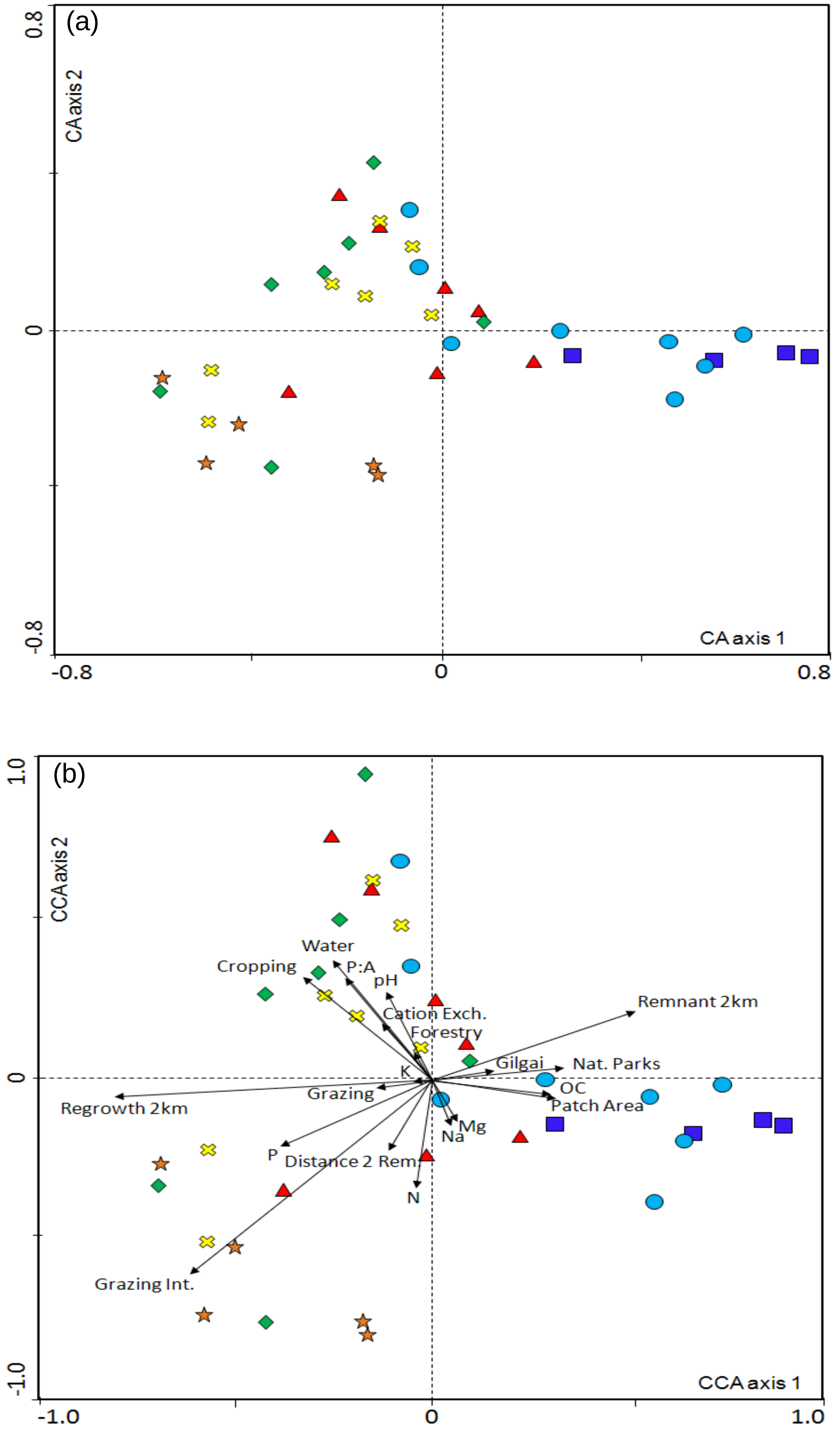


**Figure 4.2.** a) CA ordination of stand structure and b) CCA of site stand structure (ArcSine transformed) and patch and spatial variables (Eigenvalues x-axis = 0.133, y-axis = 0.036; F=1.481, p = 0.05). ■ = Reserve remnant, ● = Remnant, ▲ = Regrowth$_{>40}$, ✖ = Regrowth$_{30\text{-}40}$, ◆ = Regrowth$_{20\text{-}30}$, ★ = Regrowth$_{<20}$.

Remnant sites contained a significantly higher soil magnesium concentration than regrowth>40 (Mann-Whitney U Test, $p<0.05$). Regrowth>40 also had a significantly higher proportion of regrowth, cropping and grazing in the surrounding landscape compared to remnant sites ($p<0.05$). No other variables were different between Remnant and old regrowth (Table 4.3).

**Table 4.3.** Summary Table of Independent Sample Mann-Whitney U Tests for mean spatial and patch variables (Standard error) between Remnant and Regrowth$_{>40}$ treatments.

| Patch and Spatial Variables | Remnant | Regrowth $_{>40}$ | p- value |
|---|---|---|---|
| Grazing Intensity (score) | 0.38 (0.18) | 0.57 (0.43) | 0.889 |
| Gilgai Presence/ Depth (score) | 1.00 (0.27) | 1.14 (0.26) | 0.702 |
| Tree Stem Density (/200m$^2$) | 1.73 (0.05) | 1.78 (0.03) | 0.487 |
| pH | 7.05 (0.20) | 6.97 (0.27) | 0.643 |
| Cation Exchange (meq/100g) | 1.44 (0.04) | 1.40 (0.05) | 0.324 |
| Nitrate/ Nitrogen (mg/kg) | 0.71 (0.05) | 0.72 (0.11) | 0.907 |
| Phosphorus (mg/kg) | 0.73 (0.10) | 0.81 (0.07) | 0.558 |
| Potassium (mg/kg) | 2.55 (0.03) | 2.58 (0.06) | 0.728 |
| Calcium (mg/kg) | 3.55 (0.05) | 3.57 (0.06) | 0.954 |
| Sodium (mg/kg) | 2.25 (0.12) | 1.94 (0.17) | 0.093 |
| Magnesium (mg/kg) | 2.89 (0.04) | 2.77 (0.03) | 0.032* |
| Organic Carbon (%) | 1.55 (0.12) | 1.54 (0.14) | 1.000 |
| Forestry area | 0.57 (0.39) | 0.00 (0.0) | 0.118 |
| Cropping area | 1.88 (0.17) | 2.39 (0.18) | 0.037* |
| Grazing area | 2.99 (0.02) | 2.80 (0.03) | 0.002** |
| Remnant Area | 1.64 (0.15) | 1.57 (0.5) | 0.728 |
| Regrowth Area | 0.78 (0.26) | 1.71 (0.10) | 0.011* |
| Water Bodies | 0.10 (0.10) | 0.17 (0.11) | 0.457 |
| Patch Area | 1.49 (0.11) | 1.37 (0.07) | 0.562 |
| Perimeter: Area | 15.95 (2.76) | 17.08 (2.64) | 0.563 |
| Distance to Closest Remnant | 2.05 (0.30) | 2.15 (0.45) | 0.908 |

### 4.3.3 Correlation of spatial and habitat variables on species richness and strata cover classes in Brigalow communities.

Tree stem density and proportion of remnant vegetation in the surrounding area (2 km buffer) explained approximately 49% of the variation in species richness in Brigalow regrowth and remnant vegetation (Step-wise Regression, F= 18.699; p <0.001; Table 4.4). These factors also explained 53% of the variation of native species richness (F= 22.357; p<0.001; Table 4.4). No predictor variables significantly explained exotic species richness in the Brigalow sites assessed (p>0.05).

The proportion of regrowth in the surrounding landscape explained 16.5% of the variation in tree cover (Step-wise Regression F= 8.332; p<0.05; Table 4.4). Over 59% of the variation in shrub cover within Brigalow vegetation was explained by a combination of tree stem density, surrounding regrowth vegetation and soil organic carbon (Table 4.4). No predictor variables were identified for total groundcover, herb/forb cover or graminoid cover.

**Table 4.4.** Summary of stepwise regression results for the dependent variables total, native, exotic species richness and strata cover classes for Brigalow sites.

| Dependent Variable | Explanatory Variables in Final Model | d.f. | Adjusted $R^2$ | F | p-value | β | β Coefficient (SE) | p-value |
|---|---|---|---|---|---|---|---|---|
| Species Richness | Stem Density + Remnant Area (2km Buffer) | 2, 35 | 0.489 | 18.699 | 0.000 | Constant | 1.661 (0.089) | 0.000 |
| | | | | | | Stem Density | -0.170 (0.040) | 0.000 |
| | | | | | | Remnant Area | 0.036 (0.014) | 0.017 |
| Native SR | Stem Density + Remnant Area (2km Buffer) | 2, 35 | 0.536 | 22.357 | 0.000 | Constant | 1.644 (0.095) | 0.000 |
| | | | | | | Stem Density | 0.193 (0.043) | 0.000 |
| | | | | | | Remnant Area | 0.046 (0.016) | 0.006 |
| Exotic SR | No variables significant in the model | | | | | | | |
| Tree Cover | Regrowth (2km Buffer) | 1, 36 | 0.165 | 8.332 | 0.007 | Constant | 0.554 (0.036) | 0.000 |
| | | | | | | Regrowth Area | -0.64 (0.22) | 0.007 |
| Shrub Cover | Stem Density + Regrowth Area (2km buffer) + Organic Carbon | 3, 34 | 0.596 | 19.203 | 0.00 | Constant | 0.236 (0.71) | 0.002 |
| | | | | | | Stem Density | -0.92 (0.28) | 0.002 |
| | | | | | | Regrowth Area | -0.44 (0.11) | 0.000 |
| | | | | | | OC | 0.72 (0.25) | 0.006 |
| Ground-cover | No variables significant in the model | | | | | | | |
| Forb/Herb cover | No variables significant in the model | | | | | | | |
| Graminoid cover | No variables significant in the model | | | | | | | |

## 4.4 Discussion

A number of environmental and spatial factors were correlated with the patterns in plant community composition, richness and stand structure identified across Brigalow remnant and regrowth vegetation in Chapter 3. Consistent with other studies (see (Pettit *et al.*, 1995; Boutin and Jobin, 1998; Dorrough *et al.*, 2004), composition and structure were explained by a combination of spatial and environmental factors, particularly those related to agricultural intensification.

The spatial and habitat factors assessed were correlated with floristic composition between the sites, with spatial features explaining the gradient between remnant and older regrowth, and recent regrowth. Remnant and Reserve sites contained a higher proportion of vegetation in the surrounding areas while an increasing trend in soil nutrient concentrations was evident from regrowth sites to remnant sites. Regrowth sites also exhibited increased grazing intensity and a larger proportion of surrounding regrowth vegetation compared to remnant sites. The proportion of grazing and regrowth in the surrounding area was also a significant difference between Regrowth$_{>40}$ and Remnant sites, suggesting spatial factors may be influencing vegetation patterns within patches. Similar spatial and landscape variables were found to explain the gradient in relation to stand structure within the treatments. Univariate results identified similar environmental variables as the multivariate analyses, with spatial factors, particularly the proportion of remnant and regrowth in the surrounding landscape and stem density explaining patterns in relation to functional group richness and structural cover classes. However, some caution should be used in the generalisations made from the regression analysis as the age of regrowth may be strongly correlated with stem density. The rates of self thinning of overstorey trees within a patch can be dependent on stand age and other spatial factors (Chen *et al*., 2008; Dwyer *et al*., 2010). Patch geometry, such as size and perimeter to core ratio, for instance, while related to some of the patterns in composition and structure, was not found to affect species richness or tree cover classes.

### 4.4.1 Spatial and Patch factors impacting stand structure, composition and richness in Brigalow Communities

The proportion of different land-use within the surrounding landscape was associated with the differences in stand structure and floristic composition between the sites sampled. The different floristic composition within regrowth sites was related to the area of surrounding agricultural management. Metzger (2000) found surrounding matrix complexity, modified for coffee bean plantations, to be significantly related to the diversity of tree species found within Brazilian rainforest fragments. Surrounding agricultural has also been found to impact on the health and quality of remnant fragments within the landscape. Increased tree mortality was found within rainforest fragments bordering cattle grazing pastures compared to those surrounded by forest regrowth in the Amazon Basin (Mesquita *et al*., 1999). These factors might also be preventing older regrowth from reaching similar composition to remnant.

Intensification in the surrounding matrix has also been linked to increased vulnerability to invasive species (Saunders *et al*., 1991). Although there was no relationship recognised in regards to exotic species richness, the abundance of introduced species may have contributed to the patterns found in the CCAs.

Disturbances, coupled with increased nutrients can also increase the invasibility of non-native species (McIntyre *et al.*, 1995; Lake and Leishman, 2004). Nutrient fluxes from surrounding agricultural practices, particularly cropping, can affect community composition and ecological processes within adjacent communities (McIntyre and Lavorel, 1994). Fertiliser drift from surrounding agricultural practices can alter the soil chemistry within landscape fragments influencing plant communities. Burke & Grime (1996) found increases in nutrient availability increased the invasion success of exotic species following a disturbance in limestone grasslands. Surrounding landuse, mainly proportion of cropping, may have also contributed to influxes of nutrients within some sites, resulting in differences in abundance and composition of plant species, particularly in the herbaceous understorey.

Stem density was also associated with not only the patterns in composition but also shrub cover, total and native species richness. Overstorey tree stem density has been found to influence the composition and diversity of understorey species within woody thickened communities. Price & Morgan (2008) found reduced groundcover and understorey diversity within *E. camaldulensis* open woodlands suffering encroachment by *Leptospermum scoparium*. Thickening of particular overstorey species can out-compete understorey plant species and other woody species for light and nutrients, increasing homogeneity within communities (Gordon and Rice, 2000; Vellend *et al.*, 2007). This may in-turn impact the total cover and richness of understorey strata and species. Over time the gradual thinning of the overstorey, via natural processes, would allow other species, such as woody shrubs, to establish themselves (Grubb, 1977). Dwyer *et al*. (2010) identified a relationship between

Brigalow tree stem density and the recovery rate within regrowth communities. Selective thinning was shown to increase woody species diversity and influence groundcover species within 29 year old Brigalow regrowth (Dwyer *et al.*, 2010). As an active management intervention, the thinning of younger regrowth stands may assist in the understorey recovery of disturbed Brigalow patches in the study area, enabling the communities to reach more similar composition and richness to older regrowth and remnant patches.

Soil chemistry explained some variation in composition and stand structure between the remnant and regrowth treatments. Soil properties relating to plant growth and development can be highly dynamic over time, relating to complex interactions between soil and plant components and feedback mechanisms within an ecosystem (Eviner and Chapin, 2003). Christensen & Peet (1984) found soil chemistry, particularly soil pH, calcium, and magnesium, to be significant factors explaining the variance in floristic composition in old field and different aged secondary Pine forests in south eastern America. Fluxes of soil nutrients, such as nitrogen, calcium and magnesium are primarily controlled by biological processes while potassium and sodium levels are largely dominated by physical processes, including leaching and atmospheric deposition (Laskowski *et al.*, 1995; Eviner and Chapin, 2003). Fluxes in phosphorus are strictly reliant on a combination of physical and chemical processes (Laskowski *et al.*, 1995). The large standard error for the mean phosphorus for Regrowth$_{30\text{-}40y}$ (Table 4.2) was attributed to high levels recorded in some sites, particularly sites 24 and 28, and may have been a result of the use of fertiliser to increase pasture productivity or run-off from surrounding properties. Also, while levels of soil organic carbon recorded were on the lower end of the scale found within some Brigalow studies (e.g. Dalal *et al*., 1986; Collard & Zammit, 2006), they are still within the range recorded within other studies in both Brigalow and Eucalypt woodlands with similar soil types and climate (e.g. Graham *et al*., 1981; Jackson & Ash, 1998).

The differences in soil nutrient concentrations observed between the treatments may be a result of the disturbance history of sites and changes in biological processes within regrowth patches at different stages of development. The removal of vegetation followed by periods of grazing or cropping can affect soil nutrient concentrations (Saunders *et al.*, 1991; Murty *et al.*, 2002; Sangha *et al.*, 2005). Due to limited site history, possible changes in land use following clearing and prior to abandonment may impact on available soil nutrients, in-turn influencing differences in species composition and general recovery. Investigation into clearing techniques and prior land uses may provide further information on the recovery of Brigalow regrowth both within and between treatments.

The soil nutrients measured did not influence species richness or the foliage projected cover of strata. This, along with the small variation explained in the CCAs, may be due to all sites being based on the same soil classification under Queensland’s Regional Ecosystem classification, resulting in limited variation across the sites assessed. Graham *et al*. (1981) also found no significant differences in soil nitrogen in relation to Brigalow regrowth age. However, some differences were observed in relation to changes in land-use, with an organic carbon and nitrogen reduction with increased agricultural production (cropping and grazing) (Graham *et al*., 1981). This may be attributed to fluxes in nutrients associated with inputs from agricultural activity discussed earlier. Species richness may not always be a good indicator of biodiversity within different systems (see (Reyers *et al.*, 2000; Gotelli and Colwell, 2001). As a result, while small changes in soil chemistry may not influence the number of species found, the composition of species may be different (Hooper and Vitousek, 1997). So, while the number of species may not be influenced significantly by soil chemistry, the composition of species may be dependent on particular site nutrient availability and properties.

In contrast, Dauber *et al*. (2003) found patch factors, chiefly soil type and aspect, to solely explain species richness in regression analysis in grasslands in Germany. However, the surrounding matrix and other spatial factors may be more critical in impacting plant recolonisation and diversity following significant environmental disturbances (Kupfer *et al.*, 2006).

The distance and connectivity between regrowth and remnant vegetation explained some of the differences in structure and composition between treatments. Distance to surrounding mature vegetation patches can affect seed dispersal and recolonisation rates within plant populations (Brunet and Von Oheimb, 1998; Metzger, 2000; Reed *et al.*, 2000). Grashof-Bokdam & Geertsema (1998) found increased distance to old forest remnants to significantly reduce the occurrence of species within younger patches of evergreen forests. Similarly, species richness within Brigalow sites was also found to be highly dependent on area of remnant vegetation in the surrounding landscape. Limited proximity to remnant vegetation may impact the recovery time of floristic composition, particularly native species, to that of ‘remnant’ status, with seeds having to cross longer distances before reaching suitable habitat for germination. Dorough & Moxham (2005) also found surrounding tree cover to be critical in successful regeneration in Eucalypt forests and grassy woodlands (*E goniocalyx, E. polyanthemos, E. macrorhyncha, E. camaldulensis* and *E. macrocarpa*) in central Victoria. Simulations showed decreases (30%) in surrounding tree cover significantly reduced the area of regeneration by over 50% (Dorrough and Moxham, 2005). Although Regrowth$_{>40}$ did not have a significantly different proportion of remnant vegetation in the landscape compared to remnant sites, a significantly higher proportion of regrowth was found in the surrounding area compared to remnant. Seed dispersal between regrowth patches may be contributing to the difference in composition between older regrowth and remnant. Regrowth patches would be more likely to receive seeds and propagules from other regrowth patches as opposed to the few remnant communities remaining in the surrounding area.

Patch grazing intensity can also impact on stand structure and floristic composition. Yates *et al*. (2000) found heavily grazed sites had reduced cover of shrubs and herbs in comparison to lightly grazed and ungrazed sites in *E. salmonophloia* woodlands in the central wheatbelt of Western Australia, suggesting reductions in grazing intensity can assist in ecosystem recovery. Similarly, Spooner *et al*. (2002) found reduced recruitment of understorey species, particularly herbs and forbs, in unfenced grassy Eucalypt woodlands compared to fenced sites in southern New South Wales. Grazing intensity has also been found to affect community regeneration over time. Dorrough & Moxham (2005) found Eucalypt regeneration under frequent livestock management was practically non-existent, while, the highest levels of plant regeneration were found in ungrazed sites, and some evidence of regeneration was also exhibited in sites under intermittent grazing levels (Dorrough and Moxham, 2005). Other studies have also found reduced regeneration ability under livestock grazing (Walker *et al.*, 1981; Milchunas and Lauenroth, 1993; Pettit *et al.*, 1995).

Patterns in stand structure were also related to increased grazing intensity within the patch. Livestock grazing can impact the cover of grasses and the woody understorey, reducing germination via trampling and favouring grazing tolerant species. For example, Robertson & Rowling (2000) found livestock grazing and trampling to significantly impact the recruitment success of Eucalypt tree species in riparian woodlands in southern New South Wales. Areas of high livestock grazing had significantly higher seedling and sapling mortality compared with sites that were ungrazed (Robertson and Rowling, 2000). Overgrazing and trampling by livestock can also significantly reduce the amount of dead leaf litter and logs, resulting in an increase in bare ground (Belsky and Blumenthal, 1997; Robertson and Rowling, 2000). The relatively high grazing intensity within regrowth, particularly Regrowth$_{<20}$ (high percentage of bare ground and low shrub cover; Appendix H), may have contributed to the differences in overall stand structure when compared with remnant Brigalow sites.

Livestock grazing can also influence soil chemistry characteristics. Long-term grazing can result in decreased soil phosphorus and potential loss of magnesium, potassium and calcium (Asner *et al.*, 2004). While the loss of some soil nutrients, mainly phosphorus, remains relatively uncertain, erosion and leaching further into the soil profile have identified within grazing systems (Asner *et al.*, 2004). Turner

(1998) similarly found long-term grazing to impact the availability of plant nutrients, particularly phosphorous, in central Mali.

A number of patch variables were not found to be related with the composition or richness of plant species across the Brigalow sites sampled. Perimeter to core ratio assists in determining the amount of potential edges effects of a given patch (Ewers and Didham, 2007). The extent of edge effects can influence the germination, development and composition of species present via changes to microclimate, light intensity, soil moisture and evaporation rates (Saunders *et al.*, 1991; Jules and Rathcke, 1999; De Blois *et al.*, 2002). However, this was not significant within this study and may be a result of sampling sites away from the edge of the patch and the exclusion of long, narrow shaped patches.

Patch size has also been shown to affect plant composition, with large patches generally exhibiting greater environmental heterogeneity than smaller patches (De Blois *et al.*, 2002). A number of studies have found fragment area to be significantly related to patterns observed in plant richness and diversity (Soulé *et al.*, 1992; Kohn and Walsh, 1994). However, patch size may not always influence the composition or richness of plants within fragmented landscapes. Holt *et al.* (1995) for example, found that patch size did not influence the pattern or rate of secondary succession in varying sized plots (up to 0.5ha) in disturbed oak woodlands in central America: However, the present study was not designed to test the effect of patch size as sites were restricted to much larger patches of <50ha. The few noted exceptions, mainly site 9 and 13 may also have strongly influenced the results obtained in relation to patch size and were identified as extremes within the CCA analysis (see ter Braak, 2002).

The magnitude of gilgai also had limited effect on structure and composition within remnant and regrowth in relation to time since disturbance. There has been some literature in relation to the effects microrelief can have on vegetation, particularly annual shrubs and grass species (see (Wilson and Leigh, 1964; Mott and McComb, 1974; Malik *et al.*, 1976). Microrelief can also effect ecological processes, particularly decomposition, and play an important role in soil moisture dynamics (Dickson and Wilsey, 2009). However, Eldridge *et al*. (1991) found that microrelief had minimal affect on shrub seedling survival, noting other environmental variables, such as soil nutrients and seasonal climate variation to be more influential on plant development. The inclusion of soil moisture levels may help in further explaining some of the patterns found within these systems (Russell *et al.*, 1967; Tunstall and Connor, 1981).

### 4.4.2 Spatial autocorrelation, limitations and implications for Brigalow communities in Production Landscapes

The significant relationship between some spatial and patch factors may limit the inferences deduced within this study. Some significant associations between variables may be a result of confounding effects relating to historical and landscape management effects. For instance, areas recently cleared and modified for grazing and cropping are more likely to be surrounded by areas of further cultivation and livestock grazing, due to similarity in soil chemistry and landscape morphology. Under these circumstances, neighbouring vegetation patches are more likely to be other patches of recent regrowth as opposed to remnant vegetation. Significant differences in shrub and tree cover in relation to time since disturbance have already been identified between recent regrowth and remnant and reserve sites (see Chapter 3). Consequently the results obtained in this study may be confounded by the fact that recent regrowth sites are more likely to be surrounded by similarly disturbed/modified vegetation. The subjective selection of sites within patches, aimed at reducing potential error, may also slightly constrain the generalisation made. These confounding variables may also be contributing to patterns observed in relation to stand structure and floristic composition.

Distance to Brigalow remnants and the proportion of vegetation (National Parks, Remnant vegetation) in the surrounding landscape were found to be key factors associated with the patterns in similarity of composition and stand structure. Reserve, Remnant and some Regrowth$_{>40}$ sites were spatially closer to other remnant Brigalow patches and surrounded by a higher area of remnants compared to more recent regrowth. In contrast, recent regrowth (Regrowth$_{<20}$ and some Regrowth$_{20\text{-}30}$) tended to be further from remnant patches and surrounded by other patches of regrowth vegetation. Within the context of the study area, this spatial autocorrelation appears to identify two ‘contrasting’ landscapes occurring in regards to Brigalow communities, one containing high proportions of remnant vegetation and the other more intensively agriculturally managed, containing a high proportion of regrowth in the surrounding area. However, while autocorrelation may constrain the results observed, it may be a result of small scale pattern analysis. Discerning true spatial gradients from artificial ones is a common challenge within ecological studies and are significantly dependent on spatial scale (Legendre, 1993). While the two landscape types were identified on a relatively small scale (2 km buffer), over a larger extent (e.g. Figure 2.2) regrowth and remnant vegetation appears to be more interspersed, potentially minimising the significance of this finding. Time constraints prevented further assessment of the spatial arrangement of remnant and regrowth communities in the study area; however, future assessment of land-uses at a variety of scales may provide additional information on the spatial dynamics and effects of surrounding disturbance on Brigalow communities in the landscape, and the implications for natural restoration.

However, despite autocorrelation, the study findings do highlight the importance of landscape connectivity and disturbance, both in-patch and in the surrounding matrix, in community recovery following vegetation clearing. Incorporating the matrix and adjacent vegetation patches is vital in understanding the interactions and effects between species and the environment needed for conservation on a landscape scale (Hersperger and Forman, 2003). While some differences between Remnant and Regrowth$_{>40}$ was observed, other patch or spatial factors may be influencing vegetation patterns and preventing the recovery or return of similar floristic composition in older regrowth. Future studies need to investigate how these patch factors and 'matrix effects' influence recruitment processes within Brigalow vegetation in order to determine if different management actions are required to help restore regrowth to a state resembling remnant communities.

## *4.5 Conclusion*

Spatial and environmental factors were found to help explain the differential patterns in floristic composition, stand structure and species richness within Brigalow remnants and regrowth patches observed in Chapter 3. The proportion of surrounding land-use type and connectivity as well as patch variables, including stem density and grazing intensity, were found to explain the presence and abundance of plant species, resulting in the acceptance of the study hypotheses.

Differences in species composition within regrowth appeared to be inhibited by isolation from remnant patches as well as exogenous disturbance associated with agricultural management. Based on the scale analysed, regrowth communities experienced higher proportion of agricultural intensity, such as grazing and cropping and a higher percentage of regrowth in the surrounding matrix, compared to remnant communities. Patch factors, including high stem density and grazing intensity as well as soil phosphorus and potassium, were also found to explain the patterns in plant composition between regrowth and remnant treatments. Stem density and the proportion of remnant vegetation within the surrounding landscape was also found to help clarify patterns of species richness in Brigalow communities in the study area.

Similar spatial and patch variables were associated with the patterns in stand structure between remnant and regrowth communities, with lower proportions of regrowth and grazing in the landscape, and a higher proportion of remnant vegetation in Reserve and Remnant sites compared to regrowth communities. Patch grazing and soil properties, including cation exchange capacity and nitrogen, were also related to structural differences within regrowth treatments.

Despite potential spatial autocorrelation in relation to the location of regrowth and remnant sites, highlighting two contrasting landscapes in the region, spatial and environment features and disturbances were found to explain compositional and structural differences in regenerating Brigalow communities. Furthermore, differences in spatial features, particularly the proportion of grazing, cropping and regrowth, may be affecting recovery processes and influencing the significant differences in composition between old regrowth and remnant communities. Consequently, it is important to incorporate not only local site habitat factors into conservation strategies, but also management of surrounding landscape and spatial connectivity of landscape elements.

# Chapter 5: General Discussion and Conclusions

## *5.1 Study Summary*

The protection of regrowth vegetation on the premise of natural regeneration, presumes similar ecosystem structure and function to pre-clearing states will be achieved over time (Aide *et al.*, 2000). Consequently, current management in Queensland considers old regrowth the same as remnant vegetation, and has been protected from future clearing in order to allow the communities to naturally recover over time (Butler, 2009; DERM, 2009b). However, minimal scientific investigation of plant communities within Brigalow vegetation has been undertaken in order to determine if these communities could passively regenerate to a state similar to that of remnant vegetation. This study found while overall structure and total species richness exhibited evidence of recovery within regrowth communities after 40 years since clearing, old Brigalow regrowth (40-60 years since clearing) still remains floristically different to remnant communities, questioning the resilience of these communities in light of disturbance history.

Furthermore, continued perturbations within regrowth patches and changes in the surrounding landscape may be affecting the recovery of these communities in relation to plant establishment and dispersal. Agricultural modification and intensification in the surrounding matrix as well as patch connectivity was strongly related to the vegetative patterns in floristic composition and stand structure. Despite possible spatial autocorrelation, the increased isolation of Brigalow regrowth communities from less disturbed remnant communities may be a contributing factor responsible for the differences in floristic composition found between treatments.

Habitat factors, including tree stem density, grazing intensity and to a lesser extent soil nutrient concentrations, were also found to be related to the patterns in stand structure and floristic composition.

### 5.1.1 Overview

This chapter discusses the key findings from previous chapters outlining the potential trajectory of regrowth communities in relation to remnant communities within production landscapes as well as the relationships of surrounding spatial attributes and local scale environmental factors. In consideration of study findings the relative value, persistence and viability of regrowth within highly fragmented communities are discussed. Finally, the broader management implications are discussed in relation to Brigalow communities and other vegetation types within highly fragmented production landscapes.

## *5.2 Impact of Disturbance on Brigalow Regrowth Vegetation Recovery*

The Southern Brigalow Belt Bioregion has experienced significant landscape fragmentation and modification following European settlement (Bowen *et al*, 2009b). Since the early 1900s, changes to state legislation and the agricultural market have dictated the modifications in management, land use and conservation within these landscapes shifting from one dominated by Briglow/Belah and Eucalypt woodlands, to one extensively cleared for pastoral grazing and cropping (Johnson, 1984b; Biggs *et al.*, 2005). The extent of much of the remnant vegetation in the landscape has been reduced by over 90% compared to pre-clearing levels and has resulted in the protection of regrowth vegetation in an attempt to increase the area of this endangered community in fragmented landscapes (Seabrook *et al.*, 2007; Bowen *et* *al.*, 2009b; Butler, 2009). However, little research has examined the condition and relative composition of regrowth communities and how they contribute to biodiversity in the greater landscape (Bowen *et al.*, 2007; Bradley *et al.*, 2010).

In this study, stand structure, total species richness and functional group richness of the regrowth communities recovered to similar levels as remnant communities after 40 years following clearing (Chapter 3). No significant invasion of exotic species was observed within any remnant or regrowth sites. Native and perennial species richness increased with time since disturbance, reaching equivalent richness to remnant after 30 to 40 years. While overall stand structure within regrowth resembled that of remnant after approximately 40 years, individual strata differed in the time required to recover, suggesting much of the variation in understorey cover in regrowth treatments to be driven by shrub and tree development.

However, the species composition of old regrowth still remains significantly different from that of remnant communities. This finding is supported by other literature that has found unique assemblages of species within highly disturbed vegetation (McIntyre and Lavorel, 1994; Fensham, 1998; Dauber *et al.*, 2003). Species typically exhibit different response patterns to physical disturbance, resulting in differences in composition and abundance over time (Fensham *et al.*, 1999; Walker *et al.*, 1999). The composition of species within a community can significantly influence ecological functioning and can potentially result in what is recognised as an alternate stable state (Beisner *et al.*, 2003).

While older Brigalow regrowth (as defined here) represents a novel ecosystem in the landscape (Hobbs *et al.*, 2006), there is insufficient data to determine if these communities are in, or progressing towards, an alternate state or are still developing into a community resembling remnant vegetation. Determining the possibility of an alternate stable state in an ecosystem can be difficult over small areas and short time periods (Petraitis and Latham, 1999; Suding *et al.*, 2004). Other regrowth communities have been found to take extended periods of time, in some instances centuries, to recover to similar composition to remnant ecosystems despite similar structure and biochemical attributes returning (Turner *et al.*, 1997; Guariguata and Ostertag, 2001). Furthermore, dealing with a system with slow transient successional change, as may be the case for Brigalow regrowth communities (see (Ngugi *et al.*, 2011), determining internal recovery and relative stability can be problematic, highlighting the importance of continued long term research (Suding *et al.*, 2004).

However, recognising regrowth, or a degraded system, as an alternate stable state can be beneficial for conservation management. Acknowledging that the population and community dynamics within regrowth are different from those in a less disturbed or more ‘natural’ state, such as remnant vegetation, recognises the trajectory to recovery will be different (Suding *et al.*, 2004). Consequently, different management tools may need to be applied if the degraded system is ever to recover the same function, structure and composition of the ‘target’ state (Young *et al.*, 2001; Suding *et al.*, 2004). While old regrowth may not be the same as remnant, employing more active restoration techniques may accelerate the recovery and help guide the community to a state resembling remnant. With a significant proportion of remnant Brigalow vegetation cleared from the Southern Brigalow Belt, representing a relictual landscape (*sensu* (McIntyre and Hobbs, 1999), the restoration of regrowth, particularly older patches, should be an important priority. Without effective management, this nationally recognised endangered ecosystem may become further degraded, impacting regional biodiversity, or in some instances, disappear entirely from the landscape.

Increased vegetative suckering may be a key factor inhibiting the return and establishment of other plant species within Brigalow communities (Dwyer *et al.*, 2010; Ngugi *et al.*, 2011). Dwyer *et al*. (2010) conducted selective thinning trials within regrowth Brigalow vegetation (29 years old) in southern Queensland over a two year period. Thinned treatments were found to contain a significantly higher woody species diversity and grass cover compared to non-thinned sites (Dwyer *et al.*, 2010). Within the present study, recent (regrowth$_{<20}$) and some intermediate regrowth (Regrowth$_{20\text{-}30}$) had higher tree stem density, and significantly lower total species richness compared with remnant sites (Chapter 4). Selective thinning may be an effective tool in assisting the recovery and establishment of species within Brigalow regrowth allowing the communities to continue to develop towards similar composition compared to remnant.

However, old regrowth did not contain significantly high stem density compared with remnant vegetation (Chapter 4), indicating other factors, potentially patch and spatial characteristics, may be driving the differences in composition.

### 5.2.1 Impacts of Patch and Spatial Variables on Communities

Spatial and environmental variables helped explain the differential patterns observed in floristic composition and stand structure between regrowth and remnant vegetation. The proportion of vegetation in the surrounding matrix as well as land use, were identified as key factors correlated with floristic patterns between Brigalow remnant and regrowth communities. Remnant communities were found to have a reduced proportion of agricultural land within the surrounding matrix compared to regrowth communities. While an autocorrelation suggests essentially two landscapes operating in the study region in relation to Brigalow vegetation; one with a higher proportion of remnant vegetation in the surrounding area and one containing a higher proportion of regrowth, on a broader scale, these two distinct landscape types may become less apparent at different scales of assessment. Further investigation using a larger study area and buffer zone may provide further understanding on surrounding matrix effects and the position of landscape features on Brigalow regrowth and remnant communities.

However, despite spatial autocorrelation of management intensity in the surrounding matrix and the location of isolated regrowth communities in the landscape, the findings highlight the importance of the matrix within community succession and recovery processes. The relative isolation of regrowth from mature remnant vegetation may affect the long term community development and persistence of these Brigalow communities within the landscape. The implications of the extent of other landscape elements, such as cropping and grazing, and their relative location in the landscape also needs to be considered within management decisions. Other studies (see (Ricketts, 2001; Dauber *et al.*, 2003; Murphy and Lovett-Doust, 2004; Debinski,

2006) have also noted the importance of including the matrix within conservation initiatives.

In Australia, intensive agricultural practices, such as grazing and cropping, have been recognised as key drivers altering the composition and abundance of plant species in a variety of woodland types (Yates *et al.*, 2000; McIvor, 2001; Spooner *et al.*, 2002; McIntyre *et al.*, 2004). In the case of Brigalow regrowth, land use changes and effects from the surrounding landscape appear to be strongly correlated with the ability of these communities to recover similar floristic characteristics to remnants.

Patch variables, including grazing intensity and soil chemistry, were also found to partially explain the differential patterns between regrowth and remnant vegetation. Recent and intermediate regrowth were found to exhibit higher grazing intensity and increased soil nutrient concentrations, particularly soil phosphorus compared to older regrowth and remnant communities. Older regrowth exhibited significantly higher proportion of grazing and regrowth in the surrounding matrix compared to remnant communities, suggesting isolation and continued exogenous disturbance may be impacting species recruitment and establishment, particularly in the understorey.

Differences in environmental variables, particularly patch factors, including soil chemistry, may also be indicative of reduced habitat condition and quality within regrowth as a result of past clearing and continued disturbance regimes (Lindenmayer *et al.*, 2005). Further research into the spatial and environmental differences and individual species dynamics between old regrowth and remnant communities may help identify specific factors, reducing or potentially inhibiting the recovery of old Brigalow regrowth.

## *5.3 Ecological Value and Importance of Continued Research of Regrowth Vegetation*

While it was evident that remnant and Reserve Brigalow communities support significantly different floristic composition compared to regrowth Brigalow vegetation, it does not imply regrowth communities are of little conservation value. Regrowth communities can still provide critical habitat for both flora and fauna at different stages of regeneration, as well as improve connectivity and matrix permeability across highly fragmented systems (Chandler *et al.*, 2007; Bowen *et al.*, 2009b; Dwyer *et al.*, 2010; Michael *et al.*, 2011). These communities can also assist in the provision of other ecosystem services within the landscape. Regrowth vegetation can act as shelterbelts, synonymous with shade-lines or wind breaks, particularly surrounding crops, protecting against direct solar radiation and harness long-wave radiation influencing functional processes, such as evaporation and plant growth (Cleugh, 1998; Vandermeer *et al.*, 1998). Young Brigalow regrowth has also been recognised as a significant source for carbon sequestration, with high stem densities and increased biomass storing large amounts of carbon within a relatively short to medium term (Chandler *et al.*, 2007).

However, despite evidence that regrowth currently provides important ecosystem services, it does not imply that these services will continue into the future. Ecological processes impacted by past disturbance regimes may not fully recover within some systems, affecting long term, regional biodiversity and functioning (Folke *et al.*, 2004). Reduced rates of recruitment within a patch, for example, may significantly impact the long term stability and persistence of the community (Bond and Midgley, 2001). Recruitment processes can be impacted by a variety of historical and on-going disturbances. For example, Yates *et al*. (1994) suggested the reduction in recruitment processes of *E. salmonophloia* may be a result of continued fragmentation and agricultural development in the surrounding landscape, disrupting natural disturbance regimes necessary for normal recruitment.

Increased grazing pressures may also be affecting woody recruitment rates within highly disturbed vegetation patches (Hobbs, 2001). Tiver & Andrew (1997) found past and present grazing significantly decreased rates of recruitment and regeneration of 10 of the 18 shrub and tree species examined within woodlands of eastern South Australia. Similarly, Scanlan *et al*. (1996) found livestock grazing within *E. drepanophylla* and *E. erythrophloia* dominated woodlands significantly reduced recruitment and increased mortality of woody plant population in north eastern Queensland. Reliance on surrounding vegetation for propagule recruitment, particularly within a highly fragmented and regrowth dominated landscape, may affect the persistence of a community in a landscape, especially if connectivity further decreases (Standish *et al.*, 2007).

Stand structure and floristic composition can also be crucial in determining habitat suitability and long term persistence of faunal species within a landscape (DeWalt *et al.*, 2003). Variation in these ecosystem components, associated with habitat, food and resource availability, can significantly influence the return and persistence of animal species in the area (DeWalt *et al*., 2003). Coppedge *et al*. (2001), for example, investigated avian species richness within disturbed and undisturbed native grasslands in the southern United States. The disturbed grasslands had experienced woody plant invasion (mostly *Juniper virginianus*) attributed to changes in fire regimes and surrounding land-use (Coppedge *et al.*, 2001). Although the disturbed sites contained higher avian richness due to an increase in resource base for avian communities associated with the shrub encroachment, over the long term, increased abundances of exotic and habitat generalists could possibly occur, displacing many specialist and obligate bird species endemic to the grasslands (Coppedge *et al.*, 2001). While Brigalow communities may be useful conduits and habitat for some generalist species, further understanding on the long term response and implications for both common and rare specialist species of a variety of taxa needs to be addressed (Bowen *et al.*, 2007; Michael *et al.*, 2011).

Future climatic changes to the study region may further impact on the recovery of regrowth communities and the continued services provided by these landscape elements. Vegetation within Australia has adapted to specific nutrient and water limitations and considerable uncertainty remains in relation to the response of vegetation dynamics and productivity to future climate change (Hughes, 2003; Eamus and Palmer, 2007). With up to 90% of the productivity within Australian rangelands and grasslands dependent on water availability, these communities can be highly sensitive to changes in hydrology (Campbell and Stafford, 1997). Debuse *et al*. (2009) found a combination of spatial, site factors and rainfall to explain differences in tree density and composition within different aged *Eucalyptus populnea* woodlands in southern Queensland. Similarly, Fensham *et al*. (2005)

analysed the changes in overstorey and understorey cover of vegetation in central Queensland over a 50 year period, and found rainfall to be a critical factor determining structural cover. In both instances increased rainfall events were found to positively influence the cover and density of vegetation over time (Fensham *et al.*, 2005; Debuse *et al.*, 2009). Long term patterns in water availability may be influencing the abundance and occurrence of plant species within regrowth.

El Nino events can impact plant productivity resulting in changed biodiversity and ecosystem function throughout many ecosystem types (Holmgren *et al.*, 2001). The long period of drought experienced throughout eastern Australia, coupled with continued agricultural related disturbance, may be affecting recovery processes within Brigalow regrowth communities. Rainfall events may also play a critical role in germination and recruitment processes impacting the regeneration and recovery rates of species and the community in general. Pulse recruitment events have been strongly linked with seasonal precipitation and soil moisture, triggering the flowering and seeding of many Australian tree and shrub species (Yates *et al.*, 1996; Clarke and Davison, 2001; Batterham, 2008). Climate modelling (medium emissions scenario, IPCC, 2007) predicts the Brigalow Belt region could experience an increase in temperature of 1.5˚C to 2˚C and a reduction of 2 - 5% in annual rainfall by 2050 (Commonwealth Scientific and Industrial Research Organisation (CSIRO) and Meteorology, 2010). Reductions in water availability may be further exacerbated by a 4 to 8% increase in potential evaporation by 2050 (Commonwealth Scientific and Industrial Research Organisation (CSIRO) and Meteorology, 2010). These changes may affect the recruitment and successful establishment of propagules within Brigalow communities in production landscapes, particularly when coupled with necessary water and nutrient inputs for continued agricultural production. Continued investigation of recruitment processes in light of recent rainfall events, associated with a La Nina event (Bureau of Meteorology, 2010), may increase our understanding on the recovery within Brigalow regrowth communities.

## *5.4 Implications for Management and Recommendations*

The results obtained within this study are significant for management of fragmented communities within production landscapes. Natural regeneration has been adopted as a cost effective method of restoring disturbed and regrowth vegetation in order to increase the extent of ecosystems within highly fragmented landscapes (Aide *et al.*, 2000). While evidence does suggest some ecosystems components may recover following disturbance (Aide *et al.*, 2000; Dunn, 2004; Martin *et al.*, 2004; Ruiz-Jaén and Aide, 2005), there is insufficient evidence to suggest older Brigalow regrowth communities are, or will eventually, obtain similar floristic composition to remnant communities. However, in relation to management implications, this study has emphasised the importance and role of multiple landscape elements in affecting the structural and compositional development of Brigalow regrowth in production landscapes.

Conservation initiatives concerning Brigalow communities need to incorporate the surrounding spatial and environmental characteristics in order to recognise the potential impacts of surrounding land-uses on community dynamics. With entire landscape revegetation and restoration highly unlikely, managers need to acknowledge production landscapes are made up of a variety of landscape elements, including areas of cropping and grazing as well as patches of remnant vegetation and regrowth at a variety of different ages and condition. These elements do not act independently but interact at a variety of scales in a landscape, providing ecosystem services and disservices, ultimately affecting regional biodiversity and functioning and resilience (Fahrig and Merriam, 1994; Forman, 1995). The recognition that Brigalow regrowth tends to occur within areas of higher agricultural intensity and proportion of regrowth compared to remnant communities suggests different communities may require different management strategies. Effective conservation needs to maintain balance between a variety of landscape elements in order to sustain continued agricultural productivity as well as conserve ecological communities and functioning (Forman, 1995; Fahrig, 2003).

Although recent regrowth was shown to exhibit significantly different composition and structural complexity and is generally subjected to higher exogenous disturbance in the way of agricultural activity, regrowth retention in the landscape still remains important. Under Queensland's legislation (*Vegetation Management and other legislation Amendment Act 2009*) regrowth patches created after 1980 are not protected. However, even the retention of young regrowth can be an effective method in landscape restoration (Bradley *et al.*, 2010). The monitoring and management of all regrowth types can assist in improving the coverage and persistence of this ecologically endangered ecosystem within agricultural landscapes. Clear and specific guidelines and targets are required within conservation initiatives in order to clearly define which ecological processes and components are essential within regrowth vegetation to improve the functionality of Brigalow communities and ensure long term persistence in the greater landscape. While a particular regrowth patch may exhibit different ecological components, such as floristic composition, and represents an alternate state, the patch may still maintain and provide particular services complementary or similar to less disturbed remnant communities and still be recognised as “of value” within the context of conservation goals. Alternatively, regrowth contributing little to regional biodiversity or providing disservices to surrounding vegetation patches and populations, such as sites containing prolific suckering impeding further community development, may be deemed dysfunctional and require a more active management approach.

As well as maintaining the condition and functioning within remnant communities, incorporating more active management into conservation initiatives may assist the recovery of regrowth Brigalow communities in the region. For example, attempts to reduce the management intensity within some regrowth patches, particularly older regrowth, may assist in recovery of Brigalow communities following disturbance. Enabling the development of structural complexity and function within regrowth patches can increase the number of species supported in the landscape, as opposed to one consisting of less complex features in a landscape (Fischer *et al.*, 2006). Grazing exclusion has been identified as a useful method of improving habitat quality and plant development in some communities (Pettit *et al.*, 1995; Belsky and Blumenthal, 1997). Spooner & Briggs (2008), for instance, found grazing exclusion to significantly increase overstorey tree regeneration and native ground cover richness across a variety of woodland types in southern New South Wales. However, other trials (Tremont, 1994; Lunt and Morgan, 1999; Le Brocque and Cockfield, 2008) have shown mixed responses in relation to the effectiveness of grazing exclusion, highlighting the importance of continued monitoring and adaptive management. Due to the thickening response of Brigalow to disturbance, the inclusion of other techniques, such as selective thinning, may also assist in re-establishing species composition and increasing rates of recovery within regrowth (Chandler *et al.*, 2007; Dwyer *et al.*, 2010).

Continued investigation into the recovery and trajectory of regrowth communities is required in order to maintain and continue to improve their conservation value within production landscapes.  If regrowth communities are in fact developing towards a dysfunctional or alternate state, continued long term monitoring is necessary in order to guarantee the continuance of these ecological services.  Without continued monitoring and investigation the effectiveness of broad-scale passive restoration within this landscape will remain relatively unknown.

## *5.4 Limitations and Future Directions*

There are a number of other potential factors and limitations that may have contributed to the patterns in composition, structure and richness observed within this study and may be incorporated in future studies to further support the patterns observed.  While this investigation was essentially a ‘snapshot study’, increasing the sample size may further separate the age classes, identifying patterns over a longer temporal scale.  The selection of regrowth sites in the area was restricted based on the availability of aerial photography and accessibility, expanding the study area may identify other potential sites.  Identifying older regrowth classes (e.g. 60 – 80 years since clearing) may also shed further light on the trajectory of regrowth.  Similarly, a more prolonged investigation may provide further detail in relation to seasonal variation and longer term climatic change in community composition within regrowth and remnant Brigalow communities (Block *et al.*, 2001).

The disturbance history, including clearing frequency and changes in land use, may further assist in explaining the patterns in composition and structure across treatment types. The type of disturbance experienced within a patch can affect the response of species and processes, resulting in differences in overall diversity of the community through successional stages (Denslow, 1980). The availability of patch history of regrowth and remnant sites was in some cases incomplete, associated with changes in land ownership over time and limited availability of aerial photography. A number of sites may have potentially been cleared multiple times or may have been heavily grazed or cropped prior to abandonment. Blocking of sites based on management history may further explain the variation in floristic composition and stand structure within treatment types; however, determining an accurate management history, particularly for old regrowth, may prove problematic.

The method of clearing utilised may also have affected the assemblages of plant species. Technological changes throughout the 1900s revolutionised the methods and effectiveness of clearing Brigalow vegetation in Queensland (Anderson, 1984). Initial methods employed the use of ringbarking and axing, but with the introduction of mechanical machinery, “pulling” became readily employed within the region (Anderson, 1984). Some plant species may require certain types of disturbances in order to grow and develop or trigger reproductive cycles (Clarke and Davison, 2001). The degree of suckering of Brigalow, for instance, is significantly influenced by the level of mechanical disturbance (Johnson and Burrows, 1994). Some older regrowth sites, according to land managers, were cleared via ringbarking and axing while more recent regrowth were pulled and in some instances burned afterwards. Differences between older and more recent regrowth treatments, particularly in relation to stem density, may be contributing to the differences observed in floristics and ultimately the rate of recovery. However, the ability to control for clearing methods, without experimentation, was strongly limited based on the availability of patch history and replication, particularly for older regrowth patches.

Investigation into specific species assemblages, abundances and recruitment processes may also help identify individual species or groups of species responsible for the significant differences between older regrowth and remnant vegetation stands. For example, the analysis of the soil seed banks may help shed some light on the composition, recruitment and recovery of plant species within Brigalow regrowth following disturbance. Identification of seed viability and species can assist in providing an indication of the importance of surrounding vegetation as a source of propagules and which species can survive significant and on-going disturbances (Thompson, 2000; Wills and Read, 2007). An in-depth study of overstorey recruitment processes of shrub and tree species as well as the genetic diversity within Brigalow populations may also provide information on the recovery and ultimate persistence of Brigalow communities. Improved tree regeneration and population turnover can be a useful indicator of healthy ecosystem functioning (Gibbons and Freudenberger, 2006; Spooner and Briggs, 2008).

Finally, spatial analysis of the study region may help clarify the correlation of elements in the landscape and the relative implications for Brigalow communities in the region. Spatial patterns are a result of complex interactions between physical, biological and social drivers and need to be investigated in order to fully understand landscape dynamics (Turner, 1989). Continued analysis could help identify areas requiring further restoration in order to maintain biodiversity over the broader landscape.

## 5.6 Conclusion

The research presented in this dissertation provides a significant contribution to understanding of recovery processes and the role of Brigalow regrowth in the study landscape and production systems in general. However, although the existing literature generally supports the benefits of regrowth vegetation for faunal populations, significant differences in floristic composition were identified in Brigalow regrowth compared with remnant communities despite a relatively lengthy recovery period. Furthermore, this study emphasizes the importance of the surrounding matrix in the recovery processes of regrowth vegetation and maintaining ecological functioning across landscapes.

Brigalow regrowth communities were found to recover similar stand structure and plant species richness after approximately 40 years since being cleared. However, species composition within old regrowth remains significantly different to remnant vegetation, indicating the possibility of an alternate stable state. Population and recruitment studies may further help in identifying specific barriers and issues affecting the re-establishment of floristic composition prior to disturbance.

The location of regrowth within the landscape is largely a reflection of management history, and has resulted in potentially higher exogenous disturbance from grazing and cropping in the surrounding landscape as well as reduced connectivity with remnant patches. Continued grazing and changes in soil nutrient availability also explained differences in the development of stand structure and species composition in regrowth communities. The presence and influence of landscape elements, both production and ecological, needs to be examined and incorporated into the management of regrowth communities in production landscapes in order to maintain not only agricultural productivity but also environmental sustainability.

Despite differences in composition, regrowth communities still provide important ecosystem services within production landscapes, enabling faunal migration as well as benefiting agricultural practices, through shelterbelts and carbon sinks. However, further research is required on the recovery of older regrowth patches in order to determine their relative trajectory and identify key spatial and patch factors affecting plant recruitment, in order to ensure the long term persistence of Brigalow ecosystems in these landscapes. More active management, via grazing exclusion or selective thinning, may need to be employed in order to accelerate recovery patterns and improve the relative conservation value of regrowth communities.

While the full ramifications of habitat clearing on the capacity of Brigalow regrowth to regenerate and recover were not comprehensively assessed, this research does provide important information for land managers on Brigalow communities in highly fragmented, agricultural landscapes. However, further investigation is required in order to determine if old Brigalow regrowth is still in a transition state or is progressing towards an alternate stable state and which specific compositional elements and patch and landscape variables are preventing old regrowth communities from resembling Brigalow remnants.

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## *Appendix A: Summary of site data*

**Table A1.1.** Summary table of general site data including site number, treatment, area, perimeter, time since last cleared and added notes for each site.

| Site | Treatment | Co-ordinates | Area (ha) | Perimeter (km) | Time since last cleared (yrs) | Clearing Method | Notes |
|---|---|---|---|---|---|---|---|
| 1 | Regrowth$_{>40}$ | 27.75799˚S, 150.36362˚E | 38.65 | 5.75 | >47 | Unknown | |
| 2 | Regrowth$_{>40}$ | 27.67031˚S, 150.56650˚E | 21.05 | 2.98 | >47 | Unknown | Boondandillla State Forest |
| 3 | Reserve | 27.84505˚S, 150.35865˚E | 14.00 | 2.57 | - | - | Booroondoo State Forest |
| 4 | Regrowth$_{30\text{-}40}$ | 27.85269˚S, 150.39014˚E | 280.31 | 13.51 | 30-35 | Unknown | Booroondoo State Forest |
| 5 | Remnant | 28.04516˚S, 150.32574˚E | 25.00 | 3.53 | - | - | |
| 6 | Remnant | 27.97885˚S, 150.33765˚E | 22.00 | 3.44 | - | - | |
| 7 | Remnant | 27.73795˚S, 150.24946˚E | 16.00 | 2.99 | - | - | |
| 8 | Remnant | 27.75834˚S, 150.29993˚E | 43.00 | 8.01 | - | - | |
| 9 | Reserve | 27.83196˚S, 150.10567˚E | 3853.00 | 120.60 | - | - | Southwood National Park |
| 10 | Reserve | 27.90702˚S, 149.83533˚E | 97.00 | 6.27 | - | - | Kinkora State Forest |
| 11 | Remnant | 27.99533˚S, 149.89526˚E | 155.00 | 11.71 | - | - | |
| 12 | Regrowth$_{20\text{-}30}$ | 28.09900˚S, 150.16225˚E | 28.23 | 5.01 | 20-25 | Unknown | |
| 13 | Remnant | 27.96916˚S, 150.19400˚E | 20.00 | 2.23 | - | - | |
| 14 | Remnant | 27.95761˚S, 150.20823˚E | 25.00 | 2.34 | - | - | |
| 15 | Reserve | 27.92765˚S, 150.26133˚E | 17.00 | 3.04 | - | - | Calingunee State Forest |
| 16 | Remnant | 27.75939˚S, 150.51870˚E | 31.00 | 10.07 | - | - | |
| 17 | Regrowth$_{<20}$ | 27.80430˚S, 150.46859˚E | 26.19 | 4.36 | 10-15 | Pulled & burnt | Cleared 2-3 times prior in 1950s &60s |
| 18 | Regrowth$_{<20}$ | 27.82003˚S, 150.47941˚E | 41.15 | 4.76 | 14-15 | Pulled & burnt | |
| 19 | Regrowth$_{<20}$ | 27.81688˚S, 150.50228˚E | 35.27 | 6.23 | 10-15 | Pulled & burnt | |

| Site | Treatment | Co-ordinates | Area (ha) | Perimeter (km) | Time since last cleared (yrs) | Clearing Method | Notes |
|---|---|---|---|---|---|---|---|
| 20 | Regrowth$_{20\text{-}30}$ | 27.97905˚S, 150.08695˚E | 11.92 | 2.35 | 20-23 | Pulled & burnt | |
| 21 | Regrowth$_{20\text{-}30}$ | 27.98747˚S, 150.10897˚E | 12.55 | 2.35 | 20-25 | Pulled | |
| 22 | Regrowth$_{30\text{-}40}$ | 27.98259˚S, 150.07300˚E | 73.73 | 5.23 | 33-47 | Rim-barked | |
| 23 | Regrowth$_{<20}$ | 27.96934˚S, 150.07589˚E | 112.23 | 11.30 | 10-12 | Pulled | |
| 24 | Regrowth$_{30\text{-}40}$ | 27.95459˚S, 150.12133˚E | 36.15 | 5.10 | 30-35 | Rim-barked & burnt | |
| 25 | Regrowth$_{>40}$ | 27.93665˚S, 150.14830˚E | 34.00 | 3.79 | 60-70 | Unknown | |
| 26 | Regrowth$_{20\text{-}30}$ | 27.74095˚S, 150.36850˚E | 17.98 | 2.40 | 26-33 | Unknown | |
| 27 | Regrowth$_{30\text{-}40}$ | 27.90613˚S, 150.04234˚E | 12.98 | 1.90 | 40 | Pulled & burnt | |
| 28 | Regrowth$_{30\text{-}40}$ | 27.91390˚S, 150.02420˚E | 11.43 | 2.70 | 30-35 | Pulled & burnt | |
| 29 | Regrowth$_{<20}$ | 27.97629˚S, 150.04316˚E | 38.76 | 4.50 | 15-20 | Pulled | |
| 30 | Regrowth$_{>40}$ | 27.96078˚S, 150.06101˚E | 32.71 | 2.95 | 60-70 | Rim-barked & burnt | |
| 31 | Regrowth$_{>40}$ | 27.93871˚S, 150.06561˚E | 14.31 | 2.71 | 45-50 | Unknown | |
| 32 | Regrowth$_{20\text{-}30}$ | 27.87789˚S, 150.04706˚E | 17.66 | 2.26 | 26 | Pulled | Cleared twice prior |
| 33 | Regrowth$_{>40}$ | 27.85412˚S, 150.03458˚E | 23.08 | 6.78 | 35-47 | Unknown | |
| 34 | Regrowth$_{20\text{-}30}$ | 27.79341˚S, 149.98551˚E | 11.64 | 1.56 | 20-25 | Pulled & burnt | Cleared once prior |
| 35 | Regrowth$_{30\text{-}40}$ | 27.65707˚S, 150.25469˚E | 21.48 | 2.55 | 40 | Unknown | |
| 36 | Regrowth$_{20\text{-}30}$ | 27.81243˚S, 150.35605˚E | 22.68 | 3.98 | 26-30 | Unknown | |
| 37 | Regrowth$_{>40}$ | 27.78753˚S, 150.36142˚E | 13.51 | 2.97 | 35-40 | Unknown | |
| 38 | Regrowth$_{30\text{-}40}$ | 27.64886˚S, 150.44867˚E | 23.61 | 4.93 | 33 | Unknown | |

# *Appendix B: Species frequency data and functional groups*

**Table B1.1** – Summary table of plant species (ICBN), species frequency (0-9) for sites and functional groups; Life origin (Native or Exotic); Life form (Tree; Shrub and Groundcover (Herb/forb or Graminoid)); Perenniality (Annual or Perennial). Species number for each site denotes frequency score recorded using Nested Quadrats (500m$^2$).

| Species | Life origin | Life form | Perenniality | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Abutilon oxycarpum* | N | G/H | P |  |  |  |  | 6 | 1 |  |  |  |  |  |  |  | 3 |  |  |  |  |  |
| *Acacia harpophylla* | N | T | - | 6 | 5 | 5 | 6 | 7 | 7 | 8 | 5 | 7 | 5 | 5 | 6 | 6 | 5 | 6 | 8 | 9 | 6 | 7 |
| *Acacia* sp. | - | - | - |  |  |  |  |  |  |  |  |  |  |  |  |  | 3 |  |  |  |  |  |
| *Alectryon diversifolius* | N | S | - | 3 |  |  |  | 1 | 3 |  |  | 1 |  |  |  | 3 | 3 |  | 3 |  |  |  |
| *Alternanthera denticulata* | N | G/H | A |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 1 |  |  | 2 |  |
| *Amyema cambagei* | N | - | - |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Amyema quandang* | N | - | - |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Amyema* sp. | N | - | - |  |  |  |  |  |  |  |  |  |  |  |  |  | 3 |  |  |  |  |  |
| *Ancistrachne uncinulata* | N | G/G | P |  | 4 | 8 | 1 | 4 | 3 | 2 | 1 | 3 | 3 | 5 |  | 6 | 4 | 2 | 1 |  |  |  |
| *Apophyllum anomalum* | N | S | - |  |  |  |  | 2 |  |  |  | 2 | 1 |  |  | 2 |  |  |  |  |  |  |
| *Aristida calycina* | N | G/G | P |  |  |  |  |  |  |  |  |  |  |  |  |  | 6 |  |  |  |  |  |
| *Aristida ramosa* | N | G/G | P |  |  | 1 | 7 |  |  | 1 |  | 1 | 1 |  |  | 3 |  |  |  |  |  |  |
| *Asteraceae* 1 | N | G/H | - |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 1 |  |  |
| *Atriplex muelleri* | N | G/H | A |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Atriplex semibaccata* | N | G/H | P |  | 3 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |

| Species | Life origin | Life form | Perenniality | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Austrochloris dichanthoides* | N | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Boerhavia dominii* | N | G/H | P | | 5 | | | 1 | 5 | 1 | 4 | | | 2 | | | | | | 3 | | |
| *Boronia* sp. | N | S | - | | | | 3 | | | | | | | | | 1 | | | | | | |
| *Bothriochloa decipiens* | N | G/G | P | | | | | | | | | | | | | | 9 | | | | | |
| *Brachyachne convergens* | N | G/G | A | | | | | | | | | | | | | | | | | | | |
| *Brachyscome aculeata* | N | G/H | P | | | | | | | | | | | | | | | 6 | | | | |
| *Brunoniella australis* | N | G/H | P | 7 | 9 | 8 | 9 | 9 | 8 | 9 | 8 | 9 | 9 | 9 | | 9 | 7 | 9 | 7 | 1 | | |
| *Bryophyllum delagoense* | E | G/H | P | | | | | 9 | | | | | | | | | | | | | | |
| *Callitris glaucophylla* | N | T | - | | | | 2 | | | | | | | | | | | | | | | |
| *Calotis scabiosifolia* | N | G/H | P | | | | | | | | | | | 2 | | | | | | | | |
| *Capparis canescens* | N | T | - | | | 1 | | | | | | | | | | | | | | | | |
| *Capparis lasiantha* | N | S | - | | | | | | | | | 4 | | | | | | | | | | |
| *Capparis sarmentosa* | N | S | - | | | | | | | | | | | | | | | | | | | |
| *Carex inversa* | N | G/H | P | | | | | | | | | | 1 | | | | | | 2 | 1 | | |
| *Carissa ovata* | N | S | - | 2 | | | | | | | 2 | 4 | 1 | 1 | | | | | | | | |
| *Cassine australis* | N | S | - | 2 | | | | | | | | 2 | | | | | | | | | | |
| *Casuarina cristata* | N | T | - | 4 | 5 | 3 | 1 | 4 | 1 | 3 | | 4 | 5 | 1 | | 5 | 5 | 5 | | | | |
| *Cenchrus ciliaris* | E | G/G | P | | | | | | 1 | | | | 1 | 3 | 3 | | | | | 6 | 4 | 2 |

| **Species** | **Life origin** | **Life form** | **Perenniality** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13** | **14** | **15** | **16** | **17** | **18** | **19** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Centipeda cunninghamii* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Cheilanthes* sp. | N | G/H | - | 5 | 2 | 4 | 8 | | 1 | 5 | 1 | | 2 | 9 | | | | 2 | | | | 1 |
| *Chenopodium* 1 | N | G/H | - | | | | | | | | | | 4 | | | | | | | 2 | | |
| *Chenopodium* 2 | N | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Chenopodium desertorum* | N | G/H | P | | 2 | | 3 | 4 | 2 | | | | 3 | 4 | 2 | | | | | | | |
| *Chloris divaricata* | N | G/G | P | 4 | 9 | 5 | 2 | 7 | 9 | 5 | 9 | 3 | 8 | 7 | 6 | 9 | 4 | 5 | 8 | 6 | 6 | 8 |
| *Chloris truncata* | N | G/G | P | 1 | | | | | | | | | | | | | | | | | | |
| *Chloris unispiceus* | N | G/G | P | | | | | | | 3 | | 4 | 2 | 4 | 5 | | 6 | | | | | |
| *Citrus glauca* | N | S | - | | | | | | | | 1 | 2 | | | 2 | | | | 1 | | | 4 |
| *Commelina cyanea* | N | G/H | P | | 3 | 7 | | | 6 | 7 | 1 | | 2 | 5 | | 1 | | 4 | | | | |
| *Convolvulus clementii* | N | G/H | P | | 2 | | | | | | | | | | | | | | | | | |
| *Cymbopogon refractus* | N | G/G | P | | | 2 | | | | 7 | 3 | 2 | | 5 | | 1 | 2 | | 2 | | | |
| *Cyperus brevifolius* | E | G/G | P | | | | | | | | | | | | | | | 6 | | | 4 | |
| *Cyperus gracilis* | N | G/G | P | | 6 | 3 | | 3 | 4 | 5 | 6 | | | | | 4 | | | 4 | | | |
| *Cyperus rigidellus* | N | G/G | A | | | | | | | | | | | | | | | | | | 2 | |
| *Cyperus rotundus* | E | G/G | P | | | | | | | | | | | | | | | | | 2 | | |
| *Cyperus vaginatus* | N | G/G | P | | | | | | | | | | | | | | | | | | | 1 |
| *Dactyloctenium radulans* | N | G/G | A | | | | | | | | | | | | | | 2 | | | | | |
| *Damasonium minus* | N | G/H | A | | | | | | | | | | | | | | | 6 | 1 | | | |
| *Danthonia tenuior* | N | G/G | P | 4 | | 3 | 3 | | | | | | 1 | | | | 1 | | | 5 | | |

| Species | Life origin | Life form | Perenniality | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Desmodium varians* | N | G/H | P |  |  |  | 4 |  | 2 |  | 4 | 3 |  | 1 |  | 3 | 3 |  | 1 |  |  |  |
| *Dianella revoluta* | N | G/H | P |  |  |  | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Dianella* sp. | N | G/H | P |  | 1 |  |  | 1 | 3 | 1 | 1 |  |  |  |  |  |  |  |  |  |  |  |
| *Dichantheum sericeum* | N | G/G | P |  |  |  |  |  |  |  | 1 |  |  |  |  |  |  |  |  |  |  | 5 |
| *Echinochloa colona* | N | G/G | A |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 2 | 8 | 4 | 2 |
| *Eclipta platyglossa* | N | G/H | A |  |  | 5 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Einadia hastata* | N | G/H | P |  | 3 | 1 |  |  | 7 |  |  |  |  |  |  | 2 |  | 2 | 5 |  |  |  |
| *Einadia nutans* | N | G/H | P |  |  |  |  | 2 |  |  |  |  |  |  |  |  |  |  |  |  |  | 2 |
| *Eleocharis* sp. | - | G/H | - |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 3 |  |  |  |  |
| *Enchylaena tomentosa* | N | G/H | P | 4 | 6 | 7 |  | 7 | 6 | 4 | 2 | 6 | 7 | 7 | 2 | 5 |  | 4 | 8 |  |  | 3 |
| *Ennaepogon polyphyllus* | N | G/G | A |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Eragrostis brownii* | N | G/G | P |  |  |  |  |  |  |  |  |  |  |  |  |  | 4 |  |  |  |  |  |
| *Eragrostis cilianensis* | E | G/G | A |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | 2 |  |  |
| *Eragrostis lacunaria* | N | G/G | P |  |  | 5 | 1 | 8 | 9 | 3 |  | 2 | 9 | 6 |  | 6 |  | 7 | 5 |  |  |  |
| *Eremophila debilis* | N | S | - | 3 | 3 |  |  |  | 1 | 3 |  |  |  |  |  | 1 |  | 1 | 2 |  |  |  |
| *Eremophila glabra* | N | S | - |  |  |  |  |  | 3 |  |  |  | 6 |  |  |  |  |  |  |  |  |  |
| *Eremophila mitchelli* | N | S | - |  |  |  |  | 3 |  |  |  |  |  |  | 2 |  |  |  |  |  |  |  |
| *Eremophila* sp. | N | S | - | 2 | 2 |  |  |  |  | 4 |  | 3 |  | 3 |  |  |  | 3 |  |  |  |  |
| *Eremophila* sp.2 | N | S | - |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| *Eriochloa procera* | N | G/G | P |  |  |  |  |  |  |  |  |  |  |  |  |  | 1 |  |  |  |  |  |
| *Eucalyptus populnea* | N | T | - |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |

| Species | Life origin | Life form | Perenniality | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Evolvulus alsiniodes* | N | G/H | P | | | | | | | | | 1 | | 1 | | | | | | | | |
| *Geijera parviflora* | N | S | - | 4 | 3 | 5 | 4 | 5 | 6 | 5 | 4 | 7 | 5 | 3 | | 6 | 5 | 5 | 4 | | | |
| *Gomphocarpus fruiticosa* | E | S | - | | | | | | | | | 2 | | | | | | | | | | |
| *Goodenia bellidifolia* | N | G/H | - | | | | | 8 | 2 | | 3 | | | | | | | | | | | |
| *Homopholis belsonii* | N | G/G | P | | | | | | | | | | | | | 6 | | | | | | |
| *Juncus* sp. | N | G/H | - | | | | | | | | | | | | | | 3 | | | | | |
| *Justicia procumbens* | N | G/H | P | | 1 | | | | | | | 1 | | | | | | | | | | |
| *Justicia* sp. | N | G/H | P | | | | | | | | | | | | | | | | 1 | | | |
| *Lomandra leucocephala* | N | G/H | P | | | | | | | | | 1 | | | | | | | 2 | | | |
| *Maireana villosa* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Marsdenia* sp. | N | G/H | - | | 1 | | | | | | | | | | | | | | | | | |
| *Marsdenia* sp. 2 | N | G/H | - | | | | | | | | | | | | 1 | | 4 | | | | | |
| *Marsilea* sp. | N | G/H | - | | | | | | | | | | | | | | | | | | | 1 |
| *Maytenus cunninghamii* | N | S | - | | | | 1 | | | | | | | | | | | | | | | |
| *Melaleuca* sp. | N | T | - | 1 | | 4 | | | | | | | | | | | | | | | | |
| *Neobassia proceriflora* | N | G/H | A | | | | | 1 | | | 1 | | | | | | | | | | | |
| *Nicotiana megalosiphon* | N | G/H | P | | | | | | | | | | | | | | | | | | | 2 |
| *Nyssanthes diffusa* | N | G/H | A | | | 2 | | 1 | 2 | 2 | | | | | | 2 | | | | | | |
| *Opuntia aurantiaca* | E | G/H | - | | 8 | 4 | | 4 | | | 1 | | | 2 | | | | 1 | | | | |
| *Opuntia tomentosa* | E | S | - | 2 | 3 | 1 | | 7 | 7 | 6 | 1 | | 4 | 2 | 4 | 2 | | 4 | | | | |

| **Species** | **Life origin** | **Life form** | **Perenniality** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13** | **14** | **15** | **16** | **17** | **18** | **19** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Oxalis* sp. | - | G/H | - | | | | 3 | | 3 | | | | | 3 | | | 1 | | 2 | | | |
| *Panicum antidotale* | E | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Panicum buncei* | N | G/G | P | | | | | | | | | | | | | | | | | | 2 | |
| *Panicum maximum* | E | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Panicum queenslandicum* | N | G/G | P | | | 1 | 3 | | 4 | 5 | 2 | 3 | 5 | | 3 | | 5 | 6 | 4 | | 2 | 3 |
| *Parsonsia lanceolata* | N | G/H | - | | | | | | | | | 4 | | 4 | | | | | | | | |
| *Parsonsia* sp. | N | G/H | - | | | | 3 | | | | | | | | | | | | | | | |
| *Paspalidium caespitosum* | N | G/G | P | 5 | 5 | 9 | 8 | 8 | 6 | 9 | 5 | 7 | 7 | 9 | 7 | 5 | 8 | 6 | 5 | 5 | 4 | |
| *Paspalidium constrictum* | N | G/G | P | | | | | 6 | | | | | | | | | | | | 6 | 3 | |
| *Phyllanthus virgatus* | N | G/H | P | | | | 3 | | | | | | | | | | | | | | | |
| *Physalis lanceifolia* | E | G/H | A | | | | | | 1 | | 1 | | | | | | | | | | | |
| *Pimelea* sp. | - | G/H | - | | | | 1 | | | | | | | | | | | | | | | |
| *Plectranthus parviflora* | N | G/H | P | | | | | 3 | 3 | | | | | | | | | | | | | |
| *Plectranthus* sp. | - | G/H | - | | | | | | | | | | | | | | | | | 1 | | |
| *Poaceae* sp. | - | G/G | - | | | | | | | | | | | | | | | | | | | |
| *Portulaca filifolia* | N | G/H | A | | | | 6 | | | 1 | | | | | | | | | | | | |
| *Portulaca oleracea* | N | G/H | A | | 3 | 1 | 1 | 3 | 8 | | | | 3 | 1 | 6 | | | 6 | 5 | 7 | 6 | 5 |
| *Pseudoranthemum variabile* | N | G/H | P | | | | 5 | | | | | | | | | | | | | | | |
| *Salsola kali* | N | G/H | A | | | | | | | | | | | | | | | | | 2 | 7 | |

| Species | Life origin | Life form | Perenniality | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Santalum lanceolatum* | N | S | - | | | | | | | | | | 3 | | 4 | | | | | | | |
| *Sarcostemma australe* | N | G/H | P | | | | | | | | | 3 | 3 | | | | | | | | | |
| *Sclerolaena birchii* | N | G/H | P | | | | | | | | | | | | | | | | 2 | | | |
| *Sclerolaena calcarata* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Sclerolaena muricata* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Sclerolaena tetracuspis* | N | G/H | P | | | | | | | | | | | 5 | 8 | | | 3 | 3 | 2 | | 6 |
| *Sclerolaena tricuspis* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Sida cordifolia* | N | G/H | P | | | | | | | | | | | | | | | | | 9 | | |
| *Sida corrugata* | N | G/H | P | | | | | | 7 | | | | | 7 | | | | | 3 | 9 | | |
| *Sida pleiantha* | N | G/H | P | | | | | | | | | | | | | 1 | | | | | | |
| *Sida rohlenae* | N | G/H | P | | | | | | 3 | | | 3 | | | | | | | | | | |
| *Sida* sp. | N | G/H | P | | | 6 | 9 | 4 | | 1 | 5 | | 1 | | | 1 | | 1 | 2 | | | 5 |
| *Sida trichopoda* | N | G/H | P | | | | | | | | | 1 | 3 | 3 | 3 | | | | 3 | | | 3 |
| *Sigesbeckia orientalis* | - | G/H | P | | | | | | | | | | | | | | | | 2 | | | |
| *Solanum ellipticum* | N | G/H | P | | 1 | | | 4 | 4 | | | | | | | 2 | | | | | | |
| *Solanum esuriale* | N | G/H | P | | | | 3 | | | | 3 | | | 1 | | | | | | | | |
| *Solanum* sp. | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Solanum stelligerum* | N | G/H | P | | | | | 5 | 4 | 4 | 2 | | 1 | | | 3 | | | 1 | | | |
| *Soliva* sp. | E | G/H | - | | | | 1 | | | | | | | | | | | | | | | |
| *Sonchus oleraceus* | E | G/H | A | | | | | | 1 | | 4 | | | | | | | | | | | |

| Species | Life origin | Life form | Perenniality | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Spartothamnella juncea* | N | S | - | | | | | | | | | | | | | | | | | | | |
| *Sporobolus caroli* | N | G/G | P | 2 | 2 | | 1 | | 2 | 1 | 4 | | 3 | 2 | 1 | | | | 4 | 7 | 7 | 8 |
| *Stipa aristiglumis* | N | G/G | P | | | 5 | | | | | | | | 4 | | | | | | | | |
| *Stipa verticillata* | N | G/G | P | | | | | | | | | | | | | | | | | 3 | | |
| *Tetragonia tetragonioides* | E | G/H | A | | | | | | | | | | | 5 | | | | | 5 | 7 | 7 | 6 |
| *Tragus australianus* | N | G/G | A | | | | 4 | | | 1 | | | 1 | | | | | | | 8 | | |
| *Trianthema triquetra* | N | G/H | A | | | | | | | | | | | | 6 | | | | | | 2 | |
| *Tribulus terrestris* | E | G/H | A | | | | | | | | | | | | 5 | | | | | | | |
| Unknown 1 | - | G/H | - | | | | | | | | | | | | | | | | | | | |
| Unknown 2 | - | G/H | - | | | | | 1 | | | | | | | | | | | | | | |
| *Urochloa mosambicensis* | E | G/G | P | 1 | | | | | | | | | | | | | | | | | | 7 |
| *Vittadinia pterochaeta* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Vittadinia sulcata* | N | G/H | A | | | 4 | | | | | | | | | | | | | | | | |
| *Xanthium occidentale* | E | G/H | A | | | | | | | | | | | | | | | | | 1 | | 1 |
| *Zygophyllum apiculatum* | N | G/H | P | | | | | | | | | | | | | | | 4 | | | | |

| Species | Life origin | Life form | Perenniality | 20 | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 | 31 | 32 | 33 | 34 | 35 | 36 | 37 | 38 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Abutilon oxycarpum* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Acacia harpophylla* | N | T | - | 7 | 8 | 6 | 8 | 7 | 6 | 7 | 7 | 9 | 9 | 6 | 8 | 8 | 6 | 8 | 6 | 7 | 6 | 9 |
| *Acacia* sp. | - | - | - | | | | | | | | | | | | | | | | | | | |
| *Alectryon diversifolius* | N | S | - | | | | | 3 | 3 | 3 | 2 | 1 | | | 1 | | 1 | | | | | |
| *Alternanthera denticulata* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Amyema cambagei* | N | - | - | | | | | | | | | | | 2 | | | | | | | | |
| *Amyema quandang* | N | - | - | | | | | | | 1 | | | | | | | | | | | | |
| *Amyema* sp. | N | - | - | | | | | | | | | | | | | | | | | | | |
| *Ancistrachne uncinulata* | N | G/G | P | | | 8 | | | | 2 | | | | | 1 | | | | | | | |
| *Apophyllum anomalum* | N | S | - | 1 | | | | 1 | 1 | | | | | | | 1 | 2 | | | 1 | | |
| *Aristida calycina* | N | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Aristida ramosa* | N | G/G | P | | | | 4 | | | | | | | | | | | | 1 | | | |
| *Asteraceae* 1 | N | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Atriplex muelleri* | N | G/H | A | | | | | | | | 3 | | | 3 | | | | | | | | |
| *Atriplex semibaccata* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Austrochloris dichanthoides* | N | G/G | P | | | | 4 | | | | | | | | | | | | | | | |
| *Boerhavia dominii* | N | G/H | P | | | | | | 2 | 1 | | | | | | 3 | | | | | | |
| *Boronia* sp. | N | S | - | | | | | | | | | | | | | | | | | | | |
| *Bothriochloa decipiens* | N | G/G | P | | | | | | | | | | | | | | | | | | | |

| Species | Life origin | Life form | Perenniality | 20 | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 | 31 | 32 | 33 | 34 | 35 | 36 | 37 | 38 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Brachyachne convergens* | N | G/G | A | | | | | | | | | | | | | 1 | | | | | | |
| *Brachyscome aculeata* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Brunoniella australis* | N | G/H | P | 7 | 5 | 6 | 8 | 6 | 2 | 2 | | | | | 5 | 3 | 3 | | 3 | 7 | 4 | |
| *Bryophyllum delagoense* | E | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Callitris glaucophylla* | N | T | - | | | | | | | | | | | | | | | | | | | |
| *Calotis scabiosifolia* | N | G/H | P | | | | | | | | | | | | | | | | 1 | | | |
| *Capparis canescens* | N | T | - | | | | | | | | | | | | | | | | | | | |
| *Capparis lasiantha* | N | S | - | | | | | | | | | | | | | | | | 1 | | | |
| *Capparis sarmentosa* | N | S | - | | | | | | | | | | | | | | | | | | 2 | |
| *Carex inversa* | N | G/H | P | | 3 | | | | | | | 3 | 1 | 1 | 1 | | | | | | | |
| *Carissa ovata* | N | S | - | | | 1 | | | | 5 | | | | | | | | | | | | |
| *Cassine australis* | N | S | - | | | | | | | 2 | | | | | | | | | | | | |
| *Casuarina cristata* | N | T | - | | | | | 1 | 2 | 4 | 1 | 1 | | 4 | | | 2 | | 2 | 1 | | |
| *Cenchrus ciliaris* | E | G/G | P | 1 | | 2 | | 2 | 4 | 3 | 6 | 3 | 8 | 4 | 3 | | 2 | 7 | 5 | 3 | 4 | 4 |
| *Centipeda cunninghamii* | N | G/H | P | | | | | | | | | 1 | | | | | | | | | | |
| *Cheilanthes* sp. | N | G/H | - | 4 | 6 | 3 | | | 3 | 4 | | | | | | 4 | | 2 | 2 | | 3 | |
| *Chenopodium* 1 | N | G/H | - | 3 | | | | | | | | | | | 2 | | 1 | | | | | 2 |
| *Chenopodium* 2 | N | G/H | - | | 3 | | | | 2 | | | | | | | | | 1 | 1 | 3 | | |
| *Chenopodium desertorum* | N | G/H | P | 2 | | 2 | | 7 | 5 | 7 | 9 | 1 | 3 | 7 | 3 | 4 | 5 | 2 | | | | 3 |
| *Chloris divaricata* | N | G/G | P | 8 | 6 | 9 | 7 | 3 | 7 | 3 | 4 | 2 | 4 | 5 | 4 | 5 | 6 | 4 | 3 | 4 | 5 | |

| **Species** | **Life origin** | **Life form** | **Perenniality** | **20** | **21** | **22** | **23** | **24** | **25** | **26** | **27** | **28** | **29** | **30** | **31** | **32** | **33** | **34** | **35** | **36** | **37** | **38** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Chloris truncata* | N | G/G | P | | | | | | 4 | 1 | | | | | | | | | | | | |
| *Chloris unispiceus* | N | G/G | P | | | | | | | | | | | | | | | | | 4 | | |
| *Citrus glauca* | N | S | - | 2 | | | 2 | | | 2 | | 2 | | | 2 | 4 | | 1 | 1 | 4 | | |
| *Commelina cyanea* | N | G/H | P | | 1 | | | | | | | | | | 4 | | | | 2 | | 2 | |
| *Convolvulus clementii* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Cymbopogon refractus* | N | G/G | P | | | | | | 1 | 1 | | | | | 1 | | | | 1 | 3 | 2 | |
| *Cyperus brevifolius* | E | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Cyperus gracilis* | N | G/G | P | 5 | | 4 | | 1 | | 1 | | | | | 5 | 1 | 2 | | 2 | | 5 | |
| *Cyperus rigidellus* | N | G/G | A | | | | | | | | | | | | 2 | | | | | | | |
| *Cyperus rotundus* | E | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Cyperus vaginatus* | N | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Dactyloctenium radulans* | N | G/G | A | | | | 2 | | | | | | | | | | | | | | | |
| *Damasonium minus* | N | G/H | A | | 3 | | | | | | | 2 | | | | | | | | | | |
| *Danthonia tenuior* | N | G/G | P | | | | 3 | | 2 | | | | 1 | | | | 3 | | | | | |
| *Desmodium varians* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Dianella revoluta* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Dianella* sp. | N | G/H | P | | | | | | | | | | | | 1 | | | | 3 | | | 1 |
| *Dichantheum sericeum* | N | G/G | P | | | | | | | | | | 1 | | | | | | | | | |
| *Echinochloa colona* | N | G/G | A | | | | | | | 3 | | 2 | | | | | | | | | | |
| *Eclipta platyglossa* | N | G/H | A | | | | | | | | | | | | | | | | | | | |

| Species | Life origin | Life form | Perenniality | 20 | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 | 31 | 32 | 33 | 34 | 35 | 36 | 37 | 38 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Einadia hastata* | N | G/H | P | | | | | 3 | | | | | | | | | 6 | | | 1 | | |
| *Einadia nutans* | N | G/H | P | | | | | 3 | | | | | | | | | | | 4 | | | |
| *Eleocharis* sp. | - | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Enchylaena tomentosa* | N | G/H | P | 4 | 3 | 2 | 2 | 6 | 4 | 5 | 5 | 1 | 4 | 1 | 3 | 5 | 6 | 1 | 5 | 2 | 3 | 4 |
| *Ennaepogon polyphyllus* | N | G/G | A | | | | | | | | | 1 | | | | | | | 1 | | | |
| *Eragrostis brownii* | N | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Eragrostis cilianensis* | E | G/G | A | | | | | | | | | | | | | | | | | | | |
| *Eragrostis lacunaria* | N | G/G | P | | | 2 | 9 | 1 | | 3 | 2 | | | 4 | 1 | 2 | | | 1 | 3 | 2 | |
| *Eremophila debilis* | N | S | - | | | | | | | | | | | | | | | | 1 | | 4 | 3 |
| *Eremophila glabra* | N | S | - | | | | | | | | | | | | | | | | | | | |
| *Eremophila mitchelli* | N | S | - | | | 4 | | | 3 | | | | | | | | | | | | | |
| *Eremophila* sp. | N | S | - | | | | | 1 | | 1 | | | | | 2 | 5 | 3 | | 2 | | | |
| *Eremophila* sp.2 | N | S | - | | | | 2 | | | | | | | | | | | | | | | |
| *Eriochloa procera* | N | G/G | P | | | | | | | | | 1 | | | | | | | | | | |
| *Eucalyptus populnea* | N | T | - | | 1 | 2 | 1 | | | | | | | | | | | | | | | |
| *Evolvulus alsiniodes* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Geijera parviflora* | N | S | - | | | 1 | 2 | 2 | 2 | 4 | | | | | 4 | 3 | 3 | | 1 | 2 | 2 | 1 |
| *Gomphocarpus fruiticosa* | E | S | - | | | | | | | | | | | | | | | | | | | |
| *Goodenia bellidifolia* | N | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Homopholis belsonii* | N | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Juncus* sp. | N | G/H | - | | | | | | | | | | | | | | | | | | | |

| **Species** | **Life origin** | **Life form** | **Perenniality** | **20** | **21** | **22** | **23** | **24** | **25** | **26** | **27** | **28** | **29** | **30** | **31** | **32** | **33** | **34** | **35** | **36** | **37** | **38** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Justicia procumbens* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Justicia* sp. | N | G/H | P | | | | | | 1 | | | | | | | | | | | | | |
| *Lomandra leucocephala* | N | G/H | P | | | | | | | | | | | | | | | 1 | | | | |
| *Maireana villosa* | N | G/H | P | | | | | 1 | 1 | | | | | 1 | | 2 | 3 | | | | | |
| *Marsdenia* sp. | N | G/H | - | | | | | | | | | | | | | | | | | | 2 | |
| *Marsdenia* sp. 2 | N | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Marsilea* sp. | N | G/H | - | | 2 | | | | | | | | | | | | | | | | | |
| *Maytenus cunninghamii* | N | S | - | | | | | | | | | | | | | | | | | | | |
| *Melaleuca* sp. | N | T | - | | | | | | | | | | | | | | | | | | | |
| *Neobassia proceriflora* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Nicotiana megalosiphon* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Nyssanthes diffusa* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Opuntia aurantiaca* | E | G/H | - | | | 1 | | | | | 1 | | | | | | | 2 | | | 1 | |
| *Opuntia tomentosa* | E | S | - | 2 | 1 | 4 | 4 | 3 | 2 | 1 | | | | | 4 | 3 | 5 | 2 | 4 | 3 | 3 | 3 |
| *Oxalis* sp. | - | G/H | - | | | | | | | | | 1 | | | 2 | | | | | | | |
| *Panicum antidotale* | E | G/G | P | | | | | | | | | | | | 1 | | | | | | | |
| *Panicum buncei* | N | G/G | P | | | | 5 | | | | | | | | | | | 2 | | | | 3 |
| *Panicum maximum* | E | G/G | P | | | | | | | 3 | | | | | | | | | | | | |
| *Panicum queenslandicum* | N | G/G | P | | 2 | | 8 | | 4 | | 2 | 2 | | | 4 | 3 | 2 | | 2 | | | |
| *Parsonsia lanceolata* | N | G/H | - | | | | | | | | | | | | | | | | | | | |

| **Species** | **Life origin** | **Life form** | **Perenniality** | **20** | **21** | **22** | **23** | **24** | **25** | **26** | **27** | **28** | **29** | **30** | **31** | **32** | **33** | **34** | **35** | **36** | **37** | **38** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Parsonsia* sp. | N | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Paspalidium caespitosum* | N | G/G | P | 3 | 4 | 9 | 7 | 5 | 6 | 3 | 2 | 1 | | | 6 | 4 | 3 | | 4 | 6 | 5 | |
| *Paspalidium constrictum* | N | G/G | P | | | | | | | | | | | | | | | | | | | |
| *Phyllanthus virgatus* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Physalis lanceifolia* | E | G/H | A | | | | 1 | | | | | 1 | | | | | | | | | | |
| *Pimelea* sp. | - | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Plectranthus parviflora* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Plectranthus* sp. | - | G/H | - | 1 | | | | | | | | | | | | | | | | | | |
| *Poaceae* sp. | - | G/G | - | | | | | | | | 1 | | | | | | | | | | | |
| *Portulaca filifolia* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Portulaca oleracea* | N | G/H | A | 1 | | | 2 | 4 | 2 | | 5 | 1 | 6 | 3 | | | | | 3 | | 3 | 3 |
| *Pseudoranthemum variabile* | N | G/H | P | | | 2 | | | | | | | | | | | 1 | | | | | |
| *Salsola kali* | N | G/H | A | | | | | | | | 2 | | 6 | | | | 4 | | | | | 4 |
| *Santalum lanceolatum* | N | S | - | | | | | | | | | | | | | | | | | | | |
| *Sarcostemma australe* | N | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Sclerolaena birchii* | N | G/H | P | | | | | | 2 | | 3 | | 1 | | | | | 1 | | | | |
| *Sclerolaena calcarata* | N | G/H | P | | | | | | | | 4 | | 3 | | | | | | | | | |
| *Sclerolaena muricata* | N | G/H | P | | | | | | | | | | | | | | | | 1 | | | |

| Species | Life origin | Life form | Perenniality | 20 | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 | 31 | 32 | 33 | 34 | 35 | 36 | 37 | 38 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Sclerolaena tetracuspis* | N | G/H | P | | | | | 5 | 4 | 2 | 4 | | | 3 | 2 | 3 | 5 | 4 | | | | 2 |
| *Sclerolaena tricuspis* | N | G/H | P | | | | | 3 | | | | | | | | | | | | | | |
| *Sida cordifolia* | N | G/H | P | | | 4 | | | 1 | | | | 1 | | | | | | | | | |
| *Sida corrugata* | N | G/H | P | | 4 | | 5 | | 5 | 1 | | 3 | | | | 2 | | | 4 | | 4 | |
| *Sida pleiantha* | N | G/H | P | | | | | | | | | | | | 3 | | | | 1 | | | |
| *Sida rohlenae* | N | G/H | P | | | | | | | | | | 1 | | | 1 | | | | | | |
| *Sida* sp. | N | G/H | P | | | | | 3 | | 2 | | | | | | | 4 | 4 | | | 1 | 2 |
| *Sida trichopoda* | N | G/H | P | | | 5 | 2 | | | | | | | 1 | | | | | | 1 | | |
| *Sigesbeckia orientalis* | - | G/H | P | | | | | | | | | | | | | | | | | | | |
| *Solanum ellipticum* | N | G/H | P | | | | | | | | | | | | | | | | | 1 | 7 | |
| *Solanum esuriale* | N | G/H | P | | | | 1 | | | | | | | | | | | | | | | |
| *Solanum* sp. | N | G/H | P | | | | | | | | | | | 2 | | | | | | | | |
| *Solanum stelligerum* | N | G/H | P | 1 | | 2 | | | | | | | | | | | | | | 2 | | |
| *Soliva* sp. | E | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Sonchus oleraceus* | E | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Spartothamnella juncea* | N | S | - | | | 1 | | | | | | | | | | | | | | | | |
| *Sporobolus caroli* | N | G/G | P | 4 | 5 | 4 | 9 | 4 | 9 | 5 | 9 | | 9 | 6 | 3 | 6 | 5 | 3 | 4 | | 5 | 3 |
| *Stipa aristiglumis* | N | G/G | P | | | | 1 | | | | | | | | | | | | | | | |
| *Stipa verticillata* | N | G/G | P | | | 5 | | | 2 | | | | 2 | | | | | | | | | |
| *Tetragonia tetragonioides* | E | G/H | A | 1 | | | | 4 | | 1 | 3 | 3 | | | | | | | | | | 1 |
| *Tragus australianus* | N | G/G | A | | | | | | | | | | 1 | | | | | | | | 2 | 2 |

| Species | Life origin | Life form | Perenniality | 20 | 21 | 22 | 23 | 24 | 25 | 26 | 27 | 28 | 29 | 30 | 31 | 32 | 33 | 34 | 35 | 36 | 37 | 38 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Trianthema triquetra* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Tribulus terrestris* | E | G/H | A | | | | | | | | | | | | | | | | | | | |
| Unknown 1 | - | G/H | - | | | | | | | | | | | 1 | | | 1 | | | | | |
| Unknown 2 | - | G/H | - | | | | | | | | | | | | | | | | | | | |
| *Urochloa mosambicensis* | E | G/G | P | | | | | | | | 3 | | 5 | | | | | | | | | |
| *Vittadinia pterochaeta* | N | G/H | A | | | | | | | | | | 1 | | | | | | | | | |
| *Vittadinia sulcata* | N | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Xanthium occidentale* | E | G/H | A | | | | | | | | | | | | | | | | | | | |
| *Zygophyllum apiculatum* | N | G/H | P | | | 1 | | 6 | 8 | 5 | | 6 | | | | 1 | 4 | | | | | |

# *Appendix B-1: Plant Species List*

**Table B1.2** – Summary table of plant species names and authorities.

*Abutilon oxycarpum* F. Muell.
*Acacia harpophylla* F. Muell.
*Acacia* sp.
*Alectryon diversifolius* (F. Muell.) S. T. Reynolds
*Alternanthera denticulata R. Br.*
*Amyema cambagei* (Blakely) Danser
*Amyema quandang* (Lindl.) Tiegh.
*Amyema* sp.
*Ancistrachne uncinulata* (R. BR.) S. T. Blake
*Apophyllum anomalum* F. Muell.
*Aristida calycina* R. Br.
*Aristida ramosa* R. Br.
*Asteraceae* 1
*Atriplex muelleri* Benth.
*Atriplex semibaccata* R. Br.
*Austrochloris dichanthoides* (Everist) Lazarides
*Boerhavia dominii* Meikle & Hewson
*Boronia* sp.
*Bothriochloa decipiens* (Hack.) C. E. Hubb.
*Brachyachne convergens* (F. Muell.) Stapf.
*Brachyscome aculeata* (Labill.) Less.
*Brunoniella australis* (Cav.) Bremek.
*Bryophyllum delagoense* (Eckl. & Zeyh.) Schinz.
*Callitris glaucophylla* Joy Thomps. & L. A. S. Johnson
*Calotis scabiosifolia* Sond. & F. Muell.
*Capparis canescens* Banks ex DC.
*Capparis lasiantha* R. Br. ex DC.
*Capparis sarmentosa* A. Cunn. ex Benth
*Carex inversa* R. Br.
*Carissa ovata* R. Br.
*Cassine australis* (Vent.) Kuntze
*Casuarina cristata* Miq.
*Cenchrus ciliaris* L.
*Centipeda cunninghamii* (DC.) A. Braun & Asch.
*Cheilanthes* sp.
*Chenopodium 1*
*Chenopodium* 2
*Chenopodium desertorum* J. M. Black
*Chloris divaricata* R. Br.
*Chloris truncate* R. Br.
*Chloris unispiceus* (F. Muell.) Clayton
*Citrus glauca* (Lindl.) Burkill
*Commelina cyanea* R. Br.
*Convolvulus clementii* Domin.
*Cymbopogon refractus* (R. Br.) A. Camus
*Cyperus brevifolius* (Rottb.) Hassk.
*Cyperus gracilis* R. Br.
*Cyperus rigidellus* (Benth.) J. M. Black
*Cyperus rotundus* L.
*Cyperus vaginatus* R. Br.
*Dactyloctenium radulans* (R. Br.) P. Beauv.
*Damasonium minus* (R. Br.) Buchenau
*Danthonia tenuior* (Steud.) Conert
*Desmodium varians* (Labill.) G. Don.
*Dianella revoluta* R. Br.
*Dianella* sp.
*Dichantheum sericeum* (R. Br.) A. Camus
*Echinochloa colona* (L.) Link
*Eclipta platyglossa* F. Muell.
*Einadia hastata* (R. Br.) A. J. Scott
*Einadia nutans* (R. Br.) A. J. Scott
*Eleocharis* sp.
*Enchylaena tomentosa* R. Br.
*Ennaepogon polyphyllus* (Domin) N. T. Burb.
*Eragrostis brownii* (Kunth) Nees
*Eragrostis cilianensis* (All.) Vignolo ex Janch.
*Eragrostis lacunaria* F. Muell. Ex Benth.
*Eremophila debilis* (Andrews) Chinnock
*Eremophila glabra* (R. Br.) Ostenf.
*Eremophila mitchelli* Benth.

*Eremophila sp.*
*Eremophila* sp.2
*Eriochloa procera* (Retz.) C. E. Hubb.
*Eucalyptus populnea* F. Muell.
*Evolvulus alsiniodes* (L.) L.
*Geijera parviflora* Lindl.
*Gomphocarpus fruiticosa* (L.) W. T. Aiton
*Goodenia bellidifolia* Sm.
*Homopholis belsonii* C. E. Hubb.
*Juncus* sp.
*Justicia procumbens* L.
*Justicia* sp.
*Lomandra leucocephala* (R. Br.) Ewart
*Maireana villosa* (Lindl.) Paul G. Wilson
*Marsdenia* sp.
*Marsdenia* sp. 2
*Marsilea* sp.
*Maytenus cunninghamii* (Hook.) Loes.
*Melaleuca* sp.
*Neobassia proceriflora* (F. Muell.) A. J. Scott
*Nicotiana megalosiphon* Van Heurck & Mull. Arg.
*Nyssanthes diffusa* R. Br.
*Opuntia aurantiaca* Lindl.
*Opuntia tomentosa* Salm-Dyck
*Oxalis* sp.
*Panicum antidotale* Retz.
*Panicum buncei* F. Muell. Ex Benth.
*Panicum maximum* Jacq.
*Panicum queenslandicum* Domin.
*Parsonsia lanceolata* R. Br.
*Parsonsia* sp.
*Paspalidium caespitosum* C. E. Hubb.
*Paspalidium constrictum* (Domin) C. E. Hubb.
*Phyllanthus virgatus* G. Forst.
*Physalis lanceifolia* Nees
*Pimelea* sp.
*Plectranthus parviflora* Willd.
*Plectranthus* sp.
*Poaceae* sp.
*Portulaca filifolia* F. Muell.
*Portulaca oleracea* L.
*Pseudoranthemum variabile* (R. Br.) Radlk.
*Salsola kali* L.
*Santalum lanceolatum* R. Br.
*Sarcostemma australe* R. Br.
*Sclerolaena birchii* (F. Muell.) Domin
*Sclerolaena calcarata* (Ising) A. J. Scott
*Sclerolaena muricata* (Moq.) Domin
*Sclerolaena tetracuspis* (C. T. White) A. J. Scott
*Sclerolaena tricuspis* (F. Muell.) Ulbr.
*Sida cordifolia* L.
*Sida corrugata* Lindl.
*Sida pleiantha* F. Muell. Ex Benth.
*Sida rohlenae* Domin
*Sida* sp.
*Sida trichopoda* F. Muell.
*Sigesbeckia orientalis* L.
*Solanum ellipticum* R. Br.
*Solanum esuriale* Lindl.
*Solanum* sp.
*Solanum stelligerum* Sm.
*Soliva* sp.
*Sonchus oleraceus* L.
*Spartothamnella juncea* (A. Cunn. ex Walp.) Briq.
*Sporobolus caroli* Mez.
*Stipa aristiglumis* F. Muell.
*Stipa verticillata* Spreng.
*Tetragonia tetragonioides* (Pall.) Kuntze
*Tragus australianus* S. T. Blake
*Trianthema triquetra* Willd.
*Tribulus terrestris* L.
Unknown 1
Unknown 2
*Urochloa mosambicensis* (Hack.) Dandy
*Vittadinia pterochaeta* (F. Muell. Ex Benth.) J. M. Black
*Vittadinia sulcata* N. T. Burb.
*Xanthium occidentale* Bertol.
*Zygophyllum apiculatum* F. Muell.

# *Appendix C: Floristic composition ANOSIM*

Floristic Composition ANOSIM Results

## Global Test

Sample Statistic (Global R) = 0.364

Significance Level of Sample Statistic = 0.1% ($p<0.05$)

Number of Permutations = 999 (Random sample from a large number)

Number of Permuted Statistics Greater than or equal to Global R = 0

**Table C1.1.** Summary table of ANOSIM Pair-wise Tests for floristic composition of treatments.

| Pair-wise Groups | R Statistic | p-value | Possible Permutations | Actual Permutations | Number ≥ Observed |
|---|---|---|---|---|---|
| REF, REM | -0.040 | 0.570 | 495 | 495 | 282 |
| REF, $R_{>40}$ | 0.339 | 0.030 | 330 | 330 | 10 |
| REF, $R_{30-40}$ | 0.468 | 0.014 | 210 | 210 | 3 |
| REF, $R_{20-30}$ | 0.497 | 0.003 | 330 | 330 | 1 |
| REF, $R_{<20}$ | 0.781 | 0.016 | 126 | 126 | 2 |
| REM, $R_{>40}$ | 0.328 | 0.004 | 6435 | 999 | 3 |
| REM, $R_{30-40}$ | 0.545 | 0.002 | 3003 | 999 | 1 |
| REM, $R_{20-30}$ | 0.509 | 0.001 | 6435 | 999 | 0 |
| REM, $R_{<20}$ | 0.806 | 0.002 | 1287 | 999 | 1 |
| $R_{>40}$, $R_{30-40}$ | 0.014 | 0.382 | 1716 | 999 | 381 |
| $R_{>40}$, $R_{20-30}$ | -0.001 | 0.487 | 1716 | 999 | 486 |
| $R_{>40}$, $R_{<20}$ | 0.620 | 0.003 | 792 | 792 | 2 |
| $R_{30-40}$, $R_{20-30}$ | 0.036 | 0.316 | 1716 | 999 | 315 |
| $R_{30-40}$, $R_{<20}$ | 0.275 | 0.050 | 462 | 462 | 23 |
| $R_{20-30}$, $R_{<20}$ | 0.549 | 0.004 | 792 | 792 | 3 |

Note: Pair-wise tests significant at $p< 0.05$

# *Appendix D: Total species richness and functional group richness Analysis of Variance*

**Table D1.1.** Summary table of ANOVAs for mean species richness and mean functional group species richness for each treatment. ANOVA significant at $p<0.05$. Values in parentheses are standard error.

| Response Variable | Reserve | Remnant | Regrowth $_{>40y}$ | Regrowth $_{30-40y}$ | Regrowth $_{20-30y}$ | Regrowth $_{<20y}$ | Homogeneity | F-score (df1; df2) | p-value |
|---|---|---|---|---|---|---|---|---|---|
| Species Richness | $28.5^{bc}$ (1.0) | $29.6^{c}$ (1.4) | $24.0^{abc}$ (1.8) | $24.3^{abc}$ (1.8) | $20.3^{a}$ (2.0) | $21.0^{ab}$ (1.6) | Y 0.466 | 4.730 (5; 32) | 0.002 |
| Tree Species Richness | $2.5^{b}$ (0.5) | $1.8^{ab}$ (0.2) | $1.9^{ab}$ (0.3) | $2.0^{ab}$ (0.2) | $1.4^{ab}$ (0.2) | $1.2^{b}$ (0.2) | Y 0.673 | 2.721 (5; 32) | 0.037 |
| Shrub Species Richness | $5.3^{b}$ (1.5) | $4.3^{b}$ (0.4) | $4.3^{b}$ (0.8) | $3.6^{b}$ (0.7) | $3.7^{b}$ (0.8) | $1.0^{a}$ (0.8) | Y 0.374 | 4.092 (5; 32) | 0.006 |
| Groundcover Species Richness | $20.8^{b}$ (1.1) | $23.4^{ab}$ (1.5) | $17.7^{ab}$ (1.8) | $18.7^{ab}$ (1.3) | $15.0^{a}$ (1.1) | $18.8^{ab}$ (1.5) | Y 0.415 | 3.744 (5; 32) | 0.009 |
| Graminoid Species Richness | $9.3^{ab}$ (1.1) | $9.0^{ab}$ (0.7) | $7.4^{ab}$ (1.0) | $7.6^{ab}$ (0.8) | $6.2^{a}$ (0.9) | $10.0^{b}$ (0.7) | Y 0.775 | 2.833 (5; 32) | 0.032 |
| Herb & Forb Species Richness | 11.5 (0.7) | 14.4 (1.8) | 10.3 (1.5) | 11.3 (0.8) | 8.7 (0.4) | 8.8 (1.3) | Y 0.085 | 1.898 (5; 32) | 0.122 |
| Native Species Richness | $26.5^{b}$ (1.3) | $27.4^{b}$ (1.1) | $21.9^{ab}$ (1.6) | $21.6^{ab}$ (2.1) | $18.0^{a}$ (1.8) | $17.8^{a}$ (1.5) | Y 0.273 | 5.441 (5; 32) | 0.001 |
| Introduced Species Richness | 2.0 (0.4) | 2.3 (0.6) | 2.1 (0.3) | 2.7 (0.4) | 2.3 (0.4) | 3.2 (0.6) | N 0.012 | 0.732 (5; 32) | 0.605 |
| Perennial Species Richness | $18.5^{ab}$ (0.9) | $20.8^{a}$ (1.2) | $16.6^{ab}$ (1.7) | $15.7^{ab}$ (1.6) | $13.7^{b}$ (1.1) | $13.8^{b}$ (1.5) | Y 0.240 | 3.185 (5; 32) | 0.019 |
| Annual Species Richness | $2.3^{ab}$ (0.9) | $2.6^{ab}$ (0.4) | $1.1^{a}$ (0.3) | $3.0^{ab}$ (0.7) | $1.3^{a}$ (0.4) | $4.8^{b}$ (0.7) | Y 0.477 | 3.640 (5; 32) | 0.010 |

# *Appendix E:* ***Site stand structure data***

**Table E1.1.** Summary table of stand structure (Cover %) and Leaf Littler Biomass data for each site.

| **Site** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13** | **14** | **15** | **16** | **17** | **18** | **19** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Treatment | $R_{>40}$ | $R_{>40}$ | REF | $R_{30-40}$ | REM | REM | REM | REM | REF | REF | REM | $R_{20-30}$ | REM | REM | REF | REM | $R_{<20}$ | $R_{<20}$ | $R_{<20}$ |
| Trees > 30m | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Trees 20 - 30m | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 2 | 0 | 0 | 0 | 11 | 5 | 4 | 0 | 0 | 0 | 0 |
| Trees 10 - 20m | 23 | 12 | 38 | 0 | 11 | 19 | 14 | 37 | 38 | 13 | 8 | 2 | 25 | 34 | 30 | 8 | 0 | 0 | 0 |
| Trees < 10m | 27 | 35 | 20 | 55 | 30 | 23 | 40 | 12 | 18 | 23 | 32 | 48 | 13 | 12 | 22 | 44 | 35 | 37 | 20 |
| Shrubs > 2m | 6 | 0 | 5 | 2 | 5 | 7 | 11 | 2 | 10 | 7 | 2 | 3 | 9 | 10 | 9 | 0 | 0 | 0 | 0 |
| Shrubs < 2m | 5 | 8 | 16 | 5 | 13 | 12 | 13 | 7 | 13 | 13 | 7 | 0 | 9 | 7 | 11 | 4 | 0 | 0 | 2 |
| Graminoids | 6 | 5 | 22 | 14 | 15 | 23 | 18 | 15 | 12 | 15 | 25 | 17 | 10 | 12 | 18 | 15 | 40 | 18 | 60 |
| Herbs / Forbs | 1 | 2 | 2 | 2 | 6 | 9 | 6 | 4 | 6 | 5 | 11 | 4 | 3 | 3 | 5 | 3 | 6 | 6 | 5 |
| Logs > 20cm | 4 | 0 | 4 | 7 | 4 | 6 | 8 | 2 | 8 | 8 | 0 | 0 | 7 | 0 | 5 | 3 | 0 | 0 | 0 |
| Logs 10 - 20cm | 6 | 2 | 12 | 7 | 8 | 13 | 10 | 6 | 13 | 15 | 4 | 5 | 10 | 10 | 8 | 9 | 7 | 3 | 4 |
| Coarse Litter | 12.5 | 6 | 10 | 28 | 20 | 17 | 25 | 13 | 24 | 20 | 13 | 20 | 19 | 18 | 23 | 24 | 23 | 18 | 7 |
| Fine Litter | 50 | 52 | 45 | 37 | 40 | 48 | 53 | 38 | 60 | 40 | 48 | 28 | 50 | 35 | 42 | 57 | 20 | 26 | 15 |
| Rock | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Cryptogams | 2 | 0 | 2 | 2 | 1 | 1 | 1 | 2 | 2 | 1 | 2 | 0 | 2 | 1 | 1 | 0 | 0 | 0 | 0 |
| Bare Ground | 15 | 34 | 14 | 20 | 7 | 3 | 3 | 10 | 3 | 1 | 9 | 14 | 7 | 18 | 10 | 7 | 16 | 30 | 15 |
| Biomass ($g/m^2$) | 1392 | 1011 | 546 | 726 | 628 | 893 | 949 | 660 | 952 | 628 | 862 | 275 | 937 | 797 | 657 | 633 | 230 | 252 | 196 |

| **Site** | **20** | **21** | **22** | **23** | **24** | **25** | **26** | **27** | **28** | **29** | **30** | **31** | **32** | **33** | **34** | **35** | **36** | **37** | **38** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Treatment | $R_{20-30}$ | $R_{20-30}$ | $R_{30-40}$ | $R_{<20}$ | $R_{30-40}$ | $R_{>40}$ | $R_{20-30}$ | $R_{30-40}$ | $R_{30-40}$ | $R_{<20}$ | $R_{>40}$ | $R_{>40}$ | $R_{20-30}$ | $R_{>40}$ | $R_{20-30}$ | $R_{30-40}$ | $R_{20-30}$ | $R_{>40}$ | $R_{30-40}$ |
| Trees > 30m | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Trees 20 - 30m | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Trees 10 - 20m | 0 | 5 | 8 | 0 | 5 | 9 | 7 | 2 | 0 | 0 | 9 | 4 | 0 | 9 | 0 | 4 | 0 | 0 | 0 |
| Trees < 10m | 46 | 38 | 30 | 32 | 29 | 22 | 32 | 30 | 50 | 43 | 32 | 32 | 38 | 34 | 60 | 34 | 67 | 38 | 53 |
| Shrubs > 2m | 0 | 0 | 2 | 0 | 2 | 12 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 3 | 0 | 0 | 0 | 0 | 0 |
| Shrubs < 2m | 3 | 2 | 6 | 6 | 8 | 15 | 8 | 7 | 0 | 7 | 11 | 3 | 9 | 10 | 2 | 5 | 5 | 4 | 4 |
| Graminoids | 18 | 14 | 38 | 24 | 17 | 17 | 12 | 30 | 7 | 27 | 7 | 16 | 19 | 13 | 7 | 16 | 12 | 28 | 10 |
| Herbs / Forbs | 2 | 5 | 4 | 5 | 4 | 5 | 3 | 6 | 3 | 4 | 3 | 4 | 3 | 4 | 2 | 3 | 3 | 6 | 5 |
| Logs > 20cm | 0 | 0 | 2 | 0 | 0 | 3 | 3 | 0 | 2 | 0 | 2 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Logs 10 - 20cm | 3 | 4 | 3 | 18 | 2 | 8 | 5 | 3 | 5 | 7 | 6 | 12 | 4 | 3 | 0 | 4 | 5 | 2 | 0 |
| Coarse Litter | 15 | 10 | 15 | 24 | 20 | 17 | 23 | 12 | 17 | 19 | 18 | 20 | 15 | 17 | 17 | 18 | 18 | 10 | 19 |
| Fine Litter | 37 | 20 | 35 | 21 | 32 | 29 | 38 | 28 | 30 | 25 | 30 | 53 | 47 | 47 | 43 | 65 | 70 | 55 | 65 |
| Rock | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Cryptogams | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Bare Ground | 25 | 45 | 12 | 30 | 18 | 13 | 12 | 28 | 35 | 17 | 39 | 7 | 15 | 20 | 45 | 12 | 7 | 8 | 6 |
| Biomass (g/m$^2$) | 560 | 397 | 631 | 171 | 529 | 684 | 840 | 658 | 387 | 299 | 922 | 647 | 411 | 521 | 311 | 405 | 829 | 984 | 1053 |

# *Appendix F: Stand structure ANOSIM*

Stand Structure ANOSIM Results

## Global Test

Sample Statistic (Global R) = 0.362

Significance Level of Sample Statistic = 0.1% (p=0.001)

Number of Permutations = 999 (Random sample from a large number)

Number of Permuted Statistics greater than or equal to Global R = 0

**Table F1.1.** Summary table of ANOSIM Pair-wise Tests for stand structure of treatments.

| Pair-wise Groups | R Statistic | p-value | Possible Permutations | Actual Permutations | Number ≥ Observed |
|---|---|---|---|---|---|
| REF, REM | -0.110 | 0.752 | 495 | 495 | 372 |
| REF, $R_{>40}$ | 0.426 | 0.015 | 330 | 330 | 5 |
| REF, $R_{30-40}$ | 0.780 | 0.003 | 330 | 330 | 1 |
| REF, $R_{20-30}$ | 0.852 | 0.003 | 330 | 330 | 1 |
| REF, $R_{<20}$ | 1.000 | 0.029 | 35 | 35 | 1 |
| REM, $R_{>40}$ | 0.169 | 0.064 | 6435 | 999 | 63 |
| REM, $R_{30-40}$ | 0.462 | 0.002 | 6435 | 999 | 1 |
| REM, $R_{20-30}$ | 0.547 | 0.002 | 6435 | 999 | 1 |
| REM, $R_{<20}$ | 0.803 | 0.002 | 495 | 495 | 1 |
| $R_{>40}$, $R_{30-40}$ | 0.095 | 0.207 | 1716 | 999 | 206 |
| $R_{>40}$, $R_{20-30}$ | 0.192 | 0.051 | 1716 | 999 | 40 |
| $R_{>40}$, $R_{<20}$ | 0.574 | 0.003 | 330 | 330 | 1 |
| $R_{30-40}$, $R_{20-30}$ | -0.13 | 0.933 | 1716 | 999 | 932 |
| $R_{30-40}$, $R_{<20}$ | 0.108 | 0.194 | 330 | 330 | 64 |
| $R_{20-30}$, $R_{<20}$ | 0.212 | 0.112 | 330 | 330 | 37 |

Note: Pair-wise tests significant at $p < 0.05$

# *Appendix G: Strata Cover Analysis of Variance*

**Table G1.1.** ANOVA Summary table for stand strata classes showing mean Foliage Projective Cover (SE) for each treatment type. ANOVA significant at $p<0.05$. Values in parentheses are standard errors.

| Stratum | Reserve Remnant | Remnant | Regrowth $_{>40y}$ | Regrowth $_{30\text{-}40y}$ | Regrowth $_{20\text{-}30y}$ | Regrowth $_{<20y}$ | Homogeneity | F-Score (df1; df2) | P-Value |
|---|---|---|---|---|---|---|---|---|---|
| **Trees 10-20m** | $29.7^{b}$ (1.4) | $19.5^{b}$ (4.0) | $9.4^{ab}$ (2.7) | $2.7^{a}$ (1.2) | $2.0^{a}$ (1.1) | NA | N 0.007 | 15.486 | <0.001 |
| **Trees <10m** | $20.8^{c}$ (1.1) | $25.8^{bc}$ (4.5) | $31.4^{abc}$ (2.0) | $40.1^{ab}$ (4.5) | $47.0^{a}$ (4.8) | $33.4^{abc}$ (3.8) | N 0.009 | 5.049 | 0.002 |
| **Shrubs >2m** | $7.8^{c}$ (1.1) | $5.6^{bc}$ (1.5) | $4.0^{abc}$ (1.7) | $0.9^{ab}$ (0.4) | $0.4^{a}$ (0.4) | NA | N 0.000 | 7.657 | <0.001 |
| **Shrubs <2m** | $13.3^{c}$ (1.0) | $9.0^{bc}$ (1.2) | $8.0^{abc}$ (1.6) | $5.0^{ab}$ (1.0) | $4.1^{ab}$ (1.3) | $3.0^{a}$ (1.5) | Y 0.345 | 6.513 | <0.001 |
| **Graminoids** | $16.8^{ab}$ (2.1) | $16.6^{ab}$ (1.8) | $13.1^{a}$ (3.1) | $18.9^{ab}$ (4.2) | $14.1^{a}$ (1.6) | $33.8^{b}$ (7.5) | Y 0.135 | 3.036 | 0.024 |
| **Herbs / Forbs** | 4.5 (0.9) | 5.6 (1.1) | 3.6 (0.7) | 3.9 (0.5) | 3.1 (0.4) | 5.2 (0.4) | Y 0.147 | 1.736 | 0.155 |
| **Logs >20cm** | $6.3^{b}$ (1.0) | $3.8^{ab}$ (1.1) | $2.0^{ab}$ (0.8) | $1.6^{a}$ (1.0) | $0.4^{a}$ (0.4) | NA | N 0.002 | 5.489 | 0.001 |
| **Logs 10-20cm** | $12.0^{b}$ (1.5) | $8.8^{ab}$ (1.0) | $5.6^{ab}$ (1.4) | $3.4^{a}$ (0.8) | $3.7^{a}$ (0.7) | $7.8^{ab}$ (2.7) | Y 0.594 | 4.359 | 0.004 |
| **Coarse Leaf Litter** | 19.3 (3.2) | 18.6 (1.6) | 14.4 (1.9) | 18.4 (1.9) | 16.9 (1.6) | 18.2 (3.0) | Y 0.922 | 0.779 | 0.572 |
| **Fine Leaf Litter** | $46.8^{b}$ (4.5) | $46.1^{b}$ (2.7) | $45.1^{b}$ (4.2) | $41.7^{ab}$ (6.1) | $40.4^{ab}$ (6.0) | $21.4^{a}$ (2.0) | Y 0.106 | 2.987 | 0.025 |
| **Bare Ground** | $7.0^{b}$ (3.0) | $8.0^{b}$ (1.7) | $19.4^{b}$ (4.7) | $18.7^{b}$ (3.8) | $23.3^{ab}$ (6.0) | $21.6^{a}$ (3.4) | Y 0.116 | 4.436 | 0.004 |
| **Biomass** | $695.9^{bc}$ (88.7) | $794.9^{bc}$ (48.2) | $880.1^{c}$ (110.5) | $626.9^{bc}$ (85.7) | $517.4^{ab}$ (88.6) | $229.6^{a}$ (22.3) | Y 0.107 | 13.263 | <0.001 |

# *Appendix H: Site spatial and patch data*

**Table H1.1.** Summary table of patch and spatial data for each site.

| **Site** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13** | **14** | **15** | **16** | **17** | **18** | **19** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH | 6.0 | 6.4 | 6.7 | 6.7 | 6.2 | 6.8 | 7.0 | 8.0 | 7.3 | 7.2 | 6.6 | 7.6 | 7.6 | 6.9 | 6.7 | 7.3 | 7.4 | 7.4 | 7.9 |
| Nitrogen (mg/kg) | 9.0 | 13.0 | 5.0 | 2.0 | 5.0 | 5.0 | 4.0 | 6.0 | 3.0 | 9.0 | 6.0 | 12.0 | 5.0 | 3.0 | 5.0 | 8.0 | 7.0 | 8.0 | 8.0 |
| Phosphorus (mg/kg) | 6.0 | 8.0 | 2.0 | 2.0 | 9.0 | 9.0 | 4.0 | 3.0 | 4.0 | 6.0 | 15.0 | 12.0 | 3.0 | 3.0 | 7.0 | 5.0 | 8.0 | 9.0 | 3.0 |
| Potassium (mg/kg) | 285.0 | 375.0 | 214.0 | 115.0 | 298.0 | 424.0 | 334.0 | 368.0 | 311.0 | 260.0 | 270.0 | 234.0 | 329.0 | 367.0 | 356.0 | 471.0 | 315.0 | 386.0 | 318.0 |
| Magnesium (mg/kg) | 534.0 | 630.0 | 413.0 | 270.0 | 631.0 | 690.0 | 736 | 666 | 357 | 711 | 549 | 697 | 867 | 1220 | 1080 | 1010 | 711 | 684 | 994 |
| Sodium (mg/kg) | 45.0 | 185.0 | 124.0 | 39.0 | 110.0 | 190.0 | 117.0 | 50.0 | 57.0 | 135.0 | 126.0 | 200.0 | 380.0 | 519.0 | 326.0 | 310.0 | 147.0 | 277.0 | 335.0 |
| Organic Carbon (%) | 1.8 | 1.4 | 1.5 | 1.2 | 2.0 | 1.9 | 1.6 | 0.9 | 1.9 | 1.5 | 1.4 | 0.8 | 1.5 | 1.4 | 1.6 | 1.7 | 1.4 | 1.3 | 1.1 |
| Cation Exchange (meq/100g) | 19.2 | 19.6 | 14.4 | 12.1 | 19.9 | 27.9 | 31.1 | 40.7 | 16.6 | 16.9 | 19.4 | 35.7 | 27 | 27.1 | 24.3 | 33.9 | 23.8 | 25.2 | 31.3 |
| Grazing Intensity | 0 | 1 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 2 | 3 | 2 |
| Gilgai (score) | 1 | 1 | 0 | 0 | 0 | 1 | 1 | 1 | 2 | 1 | 0 | 0 | 2 | 1 | 2 | 2 | 1 | 1 | 1 |
| Stem Density (/200m$^2$) | 55 | 52 | 23 | 76 | 45 | 61 | 86 | 35 | 68 | 55 | 68 | 145 | 40 | 43 | 67 | 74 | 179 | 195 | 84 |
| SiteArea (ha) | 39 | 21 | 14 | 280 | 25 | 22 | 16 | 43 | 3853 | 97 | 155 | 28 | 20 | 25 | 17 | 31 | 26 | 41 | 35 |
| Perimeter:Area | 14.9 | 14.2 | 18.4 | 4.8 | 14.1 | 15.7 | 18.7 | 18.6 | 3.1 | 6.5 | 7.6 | 17.8 | 11.1 | 9.4 | 17.9 | 32.5 | 16.6 | 11.6 | 17.7 |
| Grazing (ha) | 469 | 704 | 560 | 240 | 866 | 1035 | 1085 | 912 | 119 | 649 | 1015 | 912 | 1082 | 1093 | 1110 | 783 | 1111 | 1074 | 1031 |
| Forestry (ha) | 0 | 373 | 550 | 867 | 0 | 0 | 0 | 0 | 0 | 462 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 36 | 73 |
| Cropping (ha) | 642 | 35 | 0 | 0 | 203 | 74 | 24 | 200 | 0 | 0 | 94 | 146 | 27 | 17 | 0 | 328 | 0 | 0 | 7 |
| Water (ha) | 0 | 0 | 0 | 3 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 45 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| National Parks (ha) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 992 | 0 | 0 | 0 | 0 | 0 | 0 | 0. | 0 | 0 | 0 |
| Regrowth (ha) | 88 | 31 | 6 | 280 | 28 | 0 | 6 | 0 | 0 | 0 | 6 | 34 | 13 | 0 | 0 | 78 | 141 | 91 | 222 |
| Remnant (ha) | 0 | 332 | 50 | 1 | 54 | 76 | 105 | 51 | 4011 | 139 | 3 | 17 | 33 | 57 | 26 | 66 | 0 | 1 | 11 |
| Distant to Remnant (km) | 3.7 | 0.5 | 1.0 | 3.7 | 0.9 | 1.6 | 2.7 | 3.6 | 3.1 | 1.6 | 2.4 | 4.1 | 1.9 | 1.9 | 3.3 | 1.4 | 6.1 | 4.1 | 2.9 |

| **Site** | **20** | **21** | **22** | **23** | **24** | **25** | **26** | **27** | **28** | **29** | **30** | **31** | **32** | **33** | **34** | **35** | **36** | **37** | **38** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH | 6.6 | 6.8 | 6.2 | 5.9 | 6.7 | 7.5 | 7.9 | 6.7 | 7.8 | 8.4 | 6.6 | 7.0 | 8.0 | 8.3 | 7.7 | 8.6 | 7.6 | 6.8 | 7.3 |
| Nitrogen (mg/kg) | 3.0 | 3.0 | 3.0 | 15.0 | 5.0 | 8.0 | 3.0 | 6.0 | 5.0 | 5.0 | 3.0 | 4.0 | 2.0 | 2.0 | 2.0 | 1.0 | 3.0 | 5.0 | 3.0 |
| Phosphorus (mg/kg) | 14.0 | 10.0 | 2.0 | 5.0 | 25.0 | 5.0 | 5.0 | 45.0 | 35.0 | 6.0 | 11.0 | 3.0 | 15.0 | 7.0 | 3.0 | 2.0 | 1.0 | 8.0 | 4.0 |
| Potassium (mg/kg) | 242.0 | 207.0 | 272.0 | 177.0 | 475.0 | 693.0 | 503.0 | 860.0 | 600.0 | 569.0 | 468.0 | 278.0 | 467.0 | 488.0 | 347.0 | 423.0 | 361.0 | 255.0 | 285.0 |
| Magnesium (mg/kg) | 596.0 | 699.0 | 203.0 | 853.0 | 446.0 | 573.0 | 771.0 | 801.0 | 573.0 | 801.0 | 606.0 | 583.0 | 618.0 | 836.0 | 798.0 | 930.0 | 441.0 | 440.0 | 761.0 |
| Sodium (mg/kg) | 112.0 | 148.0 | 43.0 | 199.0 | 56.0 | 94.0 | 263.0 | 72.0 | 62.0 | 364.0 | 89.0 | 90.0 | 131.0 | 383.0 | 138.0 | 530.0 | 111.0 | 15.0 | 135.0 |
| Organic Carbon (%) | 1.3 | 1.6 | 1.5 | 1.5 | 2.2 | 1.7 | 1.3 | 1.7 | 1.2 | 1 | 2 | 0.9 | 1.3 | 1.3 | 1.1 | 1 | 1.7 | 1.7 | 1.3 |
| Cation Exchange (meq/100g) | 18.7 | 24.2 | 12.3 | 17.3 | 29.1 | 36.4 | 27.2 | 26.2 | 37.6 | 45 | 24.6 | 26 | 37.2 | 41.9 | 22.1 | 43.4 | 32.3 | 17.3 | 26.1 |
| Grazing Intensity (score) | 3 | 3 | 1 | 1 | 2 | 0 | 0 | 2 | 3 | 2 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Gilgai (score) | 0 | 2 | 0 | 0 | 3 | 2 | 2 | 1 | 3 | 0 | 2 | 1 | 2 | 1 | 0 | 3 | 1 | 0 | 1 |
| Stem Density (/200m$^2$) | 234 | 241 | 47 | 258 | 45 | 54 | 77 | 65 | 192 | 318 | 66 | 53 | 88 | 70 | 368 | 76 | 328 | 76 | 302 |
| SiteArea (ha) | 11.9 | 12.6 | 73.7 | 112.2 | 36.2 | 34.0 | 18.0 | 13.0 | 11.4 | 38.8 | 32.7 | 14.3 | 17.7 | 23.1 | 11.6 | 21.5 | 22.7 | 13.5 | 23.6 |
| Perimeter:Area | 19.7 | 18.7 | 7.1 | 10.1 | 14.1 | 11.2 | 13.4 | 14.6 | 23.6 | 11.6 | 9.0 | 19.0 | 12.8 | 29.4 | 13.4 | 11.9 | 17.6 | 22.0 | 21.0 |
| Grazing (ha) | 650 | 482 | 492 | 522 | 837 | 830 | 301 | 1108 | 1033 | 802 | 519 | 682 | 885 | 561 | 784 | 1006 | 693 | 675 | 950 |
| Forestry (ha) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 205 | 26 | 0 |
| Cropping (ha) | 459 | 627 | 617 | 586 | 264 | 280 | 803 | 0 | 77 | 304 | 589 | 424 | 167 | 83 | 311 | 107 | 207 | 405 | 162 |
| Water (ha) | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 15 | 0 | 0 | 0 | 5 | 5 | 0 |
| National Parks (ha) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 43 | 467 | 0 | 0 | 0 | 0 | 0 |
| Regrowth (ha) | 66 | 218 | 198 | 231 | 149 | 54 | 66 | 45 | 30 | 89 | 145 | 21 | 53 | 51 | 8 | 21 | 23 | 36 | 29 |
| Remnant (ha) | 0 | 12 | 17 | 13 | 20 | 82 | 87 | 46 | 160 | 4 | 0 | 30 | 47 | 3858 | 0 | 84 | 52 | 27 | 12 |
| Distant to Remnant (km) | 3.2 | 0.6 | 1.3 | 1.1 | 2.0 | 1.2 | 1.8 | 2.0 | 1.6 | 4.4 | 3.0 | 1.5 | 2.8 | 3.3 | 2.6 | 0.7 | 1.0 | 1.9 | 1.8 |

# *Appendix I: Krukall-Wallis correlations of site patch and spatial data*

## **Correlations between patch and spatial variables**

Numerous significant correlations were identified between the various spatial and patch factors assessed. Remnant area inside the buffer zone (2km) was negatively correlated with spatial factors such as area of surrounding regrowth and distance to remnant patches, as well as in patch factors, grazing intensity and stem density. Patch geometry (P:A) was significantly correlated with site area, but also landscape grazing, soil magnesium, cation exchange capacity and calcium.

Soil chemistry measures were highly correlated with a number of spatial and patch variables. Grazing in the surrounding landscape was positively correlated with all soil properties excluding nitrogen and organic carbon. Increased surrounding forestry was also associated with decreased potassium, calcium and cation exchange capacity within the soil. Soil pH was positively correlated with potassium, magnesium, sodium, calcium and cation exchange and negatively correlated with soil nitrogen and organic carbon (Table I1.1)

**Table I1.1.** Summary table of correlation analysis showing Pearson statistic (*r*) and significance.

| | Crop. | Graz. | Fore. | Water | Rem. 2km | Reg. 2km | Dist. to Rem. | Site Area | P:A | Graz. Int. | Gilgai | Stem Dens. | pH | N | OC | P | K | Mg | Na | Ca |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cropping | - | | | | | | | | | | | | | | | | | | | |
| Grazing | 0.003 | - | | | | | | | | | | | | | | | | | | |
| Forestry | -0.469$^{**}$ | -0.182 | - | | | | | | | | | | | | | | | | | |
| Water | 0.127 | -0.005 | 0.103 | - | | | | | | | | | | | | | | | | |
| Remnant 2km | -0.136 | -0.186 | 0.030 | -0.049 | - | | | | | | | | | | | | | | | |
| Regrowth 2km | 0.333$^{*}$ | -0.117 | 0.018 | 0.183 | -0.420$^{**}$ | - | | | | | | | | | | | | | | |
| Distance to Remnant | -0.305 | 0.022 | -0.136 | 0.082 | -0.419$^{**}$ | 0.080 | - | | | | | | | | | | | | | |
| Site Area | -0.305 | -0.639$^{**}$ | 0.122 | -0.034 | 0.173 | -0.134 | 0.187 | - | | | | | | | | | | | | |
| P:A | 0.244 | 0.334$^{*}$ | -0.139 | -0.022 | 0.167 | 0.110 | -0.042 | -0.595$^{**}$ | - | | | | | | | | | | | |
| Grazing Intensity | -0.036 | 0.102 | -0.033 | -0.140 | -0.435$^{**}$ | 0.489$^{**}$ | 0.189 | -0.094 | -0.065 | - | | | | | | | | | | |
| Gilgai | 0.014 | 0.052 | -0.273 | -0.257 | 0.337$^{*}$ | -0.029 | -0.200 | -0.079 | 0.106 | 0.070 | - | | | | | | | | | |
| Stem Density | 0.202 | 0.064 | -0.108 | 0.125 | -0.374$^{*}$ | 0.329$^{*}$ | 0.221 | -0.121 | 0.146 | 0.334$^{*}$ | -0.145 | - | | | | | | | | |
| pH | -0.004 | 0.170 | -0.096 | -0.011 | 0.248 | -0.073 | 0.253 | -0.128 | 0.227 | -0.161 | 0.344$^{*}$ | 0.242 | - | | | | | | | |
| Nitrogen | -0.117 | 0.224 | 0.162 | 0.037 | -0.075 | 0.039 | 0.075 | 0.063 | 0.062 | 0.083 | -0.208 | -0.084 | -0.343$^{*}$ | - | | | | | | |
| OC | -0.007 | -0.177 | -0.050 | -0.059 | 0.111 | -0.015 | -0.271 | 0.158 | -0.112 | 0.068 | 0.229 | -0.282 | -0.552$^{**}$ | 0.105 | - | | | | | |
| P | 0.066 | 0.266 | -0.247 | 0.260 | -0.042 | 0.160 | 0.089 | -0.222 | 0.183 | -0.120 | 0.213 | 0.211 | -0.120 | 0.248 | 0.304 | - | | | | |

|  | Crop. | Graz. | Fore. | Water | Rem. 2km | Reg. 2km | Dist. to Rem. | Site Area | P:A | Graz. Int. | Gilgai | Stem Dens. | pH | N | OC | P | K | Mg | Na | Ca |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| K | 0.109 | 0.363$^{*}$ | -0.382$^{*}$ | -0.185 | 0.296 | -0.062 | -0.037 | -0.306 | 0.200 | 0.087 | 0.527$^{**}$ | -0.063 | 0.432$^{**}$ | -0.031 | 0.141 | 0.424$^{**}$ | - |  |  |  |
| Mg | -0.017 | 0.547$^{**}$ | -0.308 | -0.170 | 0.045 | -0.206 | 0.052 | -0.410$^{*}$ | 0.332$^{*}$ | -0.019 | 0.227 | 0.167 | 0.336$^{*}$ | 0.144 | -0.221 | 0.080 | 0.349$^{*}$ | - |  |  |
| Na | -0.113 | 0.353$^{*}$ | -0.126 | -0.207 | 0.121 | -0.110 | -0.004 | -0.206 | 0.108 | 0.489$^{**}$ | -0.037 | 0.142 | 0.404$^{*}$ | -0.026 | -0.289 | -0.173 | 0.214 | 0.760$^{**}$ | - |  |
| Ca | 0.315 | 0.384$^{*}$ | -0.363$^{*}$ | 0.104 | 0.228 | 0.052 | 0.136 | -0.317 | 0.421$^{**}$ | -0.035 | 0.460$^{**}$ | 0.161 | 0.757$^{**}$ | -0.166 | -0.278 | 0.216 | 0.630$^{**}$ | 0.312 | 0.242 | - |
| Cation Exchange | 0.258 | 0.501$^{**}$ | -0.404$^{*}$ | 0.034 | 0.235 | -0.033 | 0.112 | 0.387$^{*}$ | 0.437$^{**}$ | -0.037 | 0.480$^{**}$ | 0.180 | 0.753$^{**}$ | -0.097 | -0.299 | 0.207 | 0.672$^{**}$ | 0.574$^{**}$ | -0.458$^{**}$ | 0.953$^{**}$ |

Note: $^{*}$ $p<0.05$, $^{**}$ $p<0.001$.