

UNIVERSITY OF SOUTHERN QUEENSLAND

The effects of patch and landscape factors on the
resilience of poplar box (*Eucalyptus populnea* F.
Muell.) woodlands, southern Queensland.

Lisa Renee Fritz (B. Sc.)

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ABSTRACT

Eucalypt woodlands in agricultural landscapes in Queensland have significantly declined in extent and condition in recent years. One particular ecosystem, poplar box (*Eucalyptus populnea* F.Muell.) woodland, has experienced declining tree condition, function and recruitment, which threaten the ecosystem's on-going persistence. To aid the persistence of these systems, knowledge of how agricultural land uses affect their condition, biodiversity and provision of ecological function is required. Most studies of biodiversity and ecological function in agricultural landscapes focus on either the role of characteristics within a patch of vegetation, or occasionally the characteristics of the surrounding landscape, but rarely on these factors in combination. However, studies have shown that both patch and landscape factors can be important and may explain significantly more of the variation in biodiversity, condition and function than patch factors (or landscape factors) alone. This study examines trends in community (floristic composition and stand structure) and population (dominant tree condition and population structure) level processes in relation to both patch and landscape factors, and asks the question: Do both patch and landscape factors contribute to patterns in health of remnant vegetation in agricultural landscapes?

This study uses two complementary approaches; an *a priori* (natural) experiment and a correlative study. To examine the effects of agricultural land uses at the patch scale (patch grazing) and in the surrounding landscape (land use context), an *a priori* design was implemented. Patch grazing (3 levels) was defined by the current (at time of sampling) intensity of grazing at the patch, while land use context (3 levels) was defined by the proportion of cropping and grazing within 5 km of a site. The correlative study utilised various explanatory factors at both patch and landscape scales to explain patterns. These two approaches were used to examine patterns in both community and population factors.

Thirty-three sites were sampled in and around the Condamine Catchment, southern Queensland, according to the *a priori* design. Sites were sampled for floristic composition (including overall vascular plant composition, species richness,

functional group richness and richness transition ratios), stand structure (cover of strata) and disturbance factors. The diameter at breast height (DBH) of all trees, and the condition of adult trees were measured. Bulked soil samples were collected and a range of soil physical and chemical properties determined. ArcGIS was used to determine a range of spatial variables, including the proportions of cropping, grazing and remnant vegetation in the landscape surrounding the site, the distance to a river and groundwater depth.

Two-way analyses of variance were used to compare functional group richness, richness transition ratios, stand structure, tree condition and tree densities within size classes among patch grazing and land use context categories. Differences in overall floristic composition and stand structure among levels of patch grazing and land use context were examined using non-metric multidimensional scaling (nMDS) ordinations and analyses of similarity (ANOSIM) and explained using canonical correspondence analyses (CCA). Overall tree population structure was examined using the frequency distribution of multiple size (surrogate for age) classes. Potential environmental drivers of tree condition and densities within age classes were examined using generalised linear models (GLMs) in a model averaging framework.

Native species richness, C4 species richness, and the C4:C3 richness transition ratio differed among patch grazing categories, while overall floristic composition, exotic species richness and graminoid cover differed among levels of land use context. The interaction between patch grazing and land use context was important for the C4:C3 richness transition ratio and the total cover of trees. Patterns in overall floristic composition and stand structure were best explained by a mixture of environmental variables at both patch and landscape scales such as soil fertility and the proportions of cropping and grazing in the surrounding landscape.

The population age structure of *E. populnea* across the catchment suggests continuous rather than episodic recruitment of young trees. This pattern of recruitment differs to those found in other studies, suggesting changes in recruitment patterns. Mature tree density was the only measure to differ among levels of patch grazing intensity, while tree condition and tree densities within size classes did not differ among levels of land use context. The densities within size classes and tree

condition were mostly driven by both patch and landscape factors, such as soil organic carbon, patch size, the proportion of remnant vegetation in the surrounding landscape, and groundwater depth.

Overall, the study demonstrated that trends in floristic composition, stand structure within poplar box woodlands, and tree condition and population age structure of *E. populnea* may be driven by both patch and landscape factors. As the management of remnant vegetation in agricultural landscape mostly focuses on patch factors, this research highlights the need to take multi-scale factors into consideration.

CERTIFICATION OF DISSERTATION

I certify that the ideas, experimental work, results, analyses, software and conclusions reported in this dissertation are entirely my own effort, except where otherwise acknowledged. I also certify that the work is original and has not been previously submitted for any other award, except where otherwise acknowledged.

.....
Signature of Candidate

.....
Date

ENDORSEMENT

Supervisor:

Dr Andrew Le Brocque

Senior Lecturer in Ecology &
Sustainability, Faculty of Sciences,
University of Southern Queensland

Co-Supervisor:

Dr Martine Maron

Senior Lecturer in Environmental
Management, School of Geography
Planning and Environmental
Management, The University of
Queensland

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Signature

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Signature

.....
Date

.....
Date

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Chapter 1: General Introduction.

1.1 Overview

The modification of landscapes through agricultural intensification threatens biodiversity, ecosystem function and overall resilience of native vegetation (Seabrook et al. 2006). Woodland ecosystems have declined in health and extent across Australia, and globally (Moore et al. 2011; Yates & Hobbs 1997a). In Queensland, there has been significant loss and modification of woodlands due to landscape changes associated with agricultural intensification (Yates & Hobbs 1997b). Such changes have resulted in small patches and population sizes, altered landscape processes and reduced landscape connectivity (Lindenmayer & Fischer 2006), as well as alterations to structure, composition, soil properties and other ecological processes (McIntyre & Hobbs 1999).

Agriculture is an important and extensive industry in Australia, with 52% of the continent being devoted to agricultural uses (Australian Bureau of Statistics 2011a). In 1907, agriculture occupied an area of 9.5 million hectares across Australia (Trewin 2001). By 2008, this had expanded to 417 million hectares (Pink 2010b). The growth of agriculture in Australia has led to the loss of an estimated 13% of Australia's native vegetation cover since European settlement (Pink 2010b), with a further 40% significantly modified by 1980 (Wells et al. 1984). This has occurred in a non-random fashion in relation to the location of more fertile soils, which has reduced remnant vegetation and fragmented the landscape (Fensham & Fairfax 2003a).

In Queensland, native vegetation cover is widespread, with 82% of State land being occupied by remnant native vegetation (Department of Environment and Resource Management 2009b). Much of Queensland is utilised for agricultural activities such as cropping and grazing for livestock (Department of Natural Resources 2003). Such agricultural activities have significant ecological effects on nearby remnant vegetation. The impacts of land use at the patch scale, particularly grazing, have been well documented as affecting soil properties, species composition, tree recruitment

and woodland condition across Australia (e.g. Landsberg et al. 1990; Yates et al. 2000). In addition, agricultural land uses in the broader landscape enrich nearby soils, affect species richness and composition, and contribute to tree dieback (Duncan et al. 2008; Lindsay & Cunningham 2009; Reid et al. 2007).

In agricultural landscapes in Queensland, much of the native remnant vegetation has declined in extent and condition (Environmental Protection Agency 2008). One particular ecosystem which has been affected by landscape change due to agricultural growth is the poplar box (*Eucalyptus populnea* F.Muell.) woodland. This ecosystem has been severely reduced in extent, with just 10-30% of the pre-clearing extent now remaining in Queensland (Department of Natural Resources 2003; Sattler & Williams 1999). Native remnant woodlands in the Condamine Catchment, including poplar box woodlands, are commonly utilised for livestock grazing, and are often found within landscapes which are exploited extensively for cultivation and livestock grazing (Department of Natural Resources 2003). Remaining vegetation has shown signs of declining health and tree condition, woodland thickening, and declining landscape function (Chilcott et al. 2005). Poplar box woodlands in agricultural landscapes have also exhibited limited recruitment (e.g. Tunstall & Reece 2004). As a result, the future persistence of poplar box woodlands is under threat, with this ecosystem recognised by Queensland legislation, as ‘of concern’ under the *Vegetation Management Act 1999*.

Much research has been directed towards examining ecological processes at the patch level (Lee et al. 2002). As such, many systems are often researched and managed with only local factors in mind, overlooking the potential effects from the surrounding landscape (Boutin & Hebert 2002; Freudenberger & Harvey 2003). More recently, the broader landscape has been recognised as a significant influence on patch dynamics, particularly in fragmented landscapes (Haynes & Cronin 2004; Lindenmayer & Franklin 2002; Öckinger et al. 2011). It has been suggested that both patch and landscape factors are important for biodiversity and function (Dauber et al. 2003). As such, it is hypothesised that a combination of patch and landscape factors contribute to the declining state of remnant woodlands within agricultural landscapes.

This study examines community level responses (floristic composition, species richness, functional group richness, richness transition ratios and stand structure) and population level responses (tree condition and population structure) within poplar box (*Eucalyptus populnea*) woodlands in the production landscape of the Condamine Catchment, southern Queensland. These responses are examined to determine how environmental factors at the patch scale and in the surrounding landscape affect ecosystem health. The overarching general question examined in this study is: Do both patch and landscape factors contribute to patterns in health of remnant vegetation in agricultural landscapes?

This chapter discusses the consequences of agricultural intensification and extensification in relation to poplar box woodlands, a declining ecosystem in agricultural landscapes. The effects of agricultural intensification on native remnant vegetation are also reviewed more broadly. The importance of considering both patch and landscape factors when examining ecosystem processes is also discussed. The general questions of this study, and the approach used to examine them, are then presented, and a brief overview of the subsequent chapters is provided.

1.1.1 Poplar box woodlands in the Condamine Catchment

Remnant poplar box woodlands in the Condamine Catchment and other regions of Queensland and New South Wales have displayed signs of declining condition mostly due to rapid land use change and resulting pressures and disturbances (Chilcott et al. 2005). The health of poplar box trees has declined through dieback, which has increased in severity in recent years (Banks 2006; Chilcott et al. 2005). There is also evidence of inbreeding in maternal stock within remnant vegetation, which increases in severity with increasing isolation of patches (Chilcott et al. 2005). Ecosystem functions such as nutrient cycling and soil infiltration have also been declining in poplar box woodlands, possibly as a result of grazing management (Chilcott et al. 2005).

In particular, there has been little sign of recruitment of poplar box trees in these remnant woodlands (Wark 2000, cited in Reid et al. 2007). Such limited recruitment

was reported in poplar box woodlands in south-west Queensland, where tree recruitment was a rare event, and of those which emerged, none survived (Tunstall & Reece 2004). Poor recruitment and survival of seedlings could have major implications for the resilience of these ecosystems. Although the particular cause of reduced recruitment is unclear, Tunstall and Reece (2004) and Debuse et al. (2009) suggest that the abundance of mature poplar box trees has an effect, finding that recruitment is best at low densities of mature trees.

Poplar box woodlands are ecologically important systems, providing habitat for various native flora and fauna, along with other important ecosystem functions (see Chapter 2). The reduced condition and limited recruitment in poplar box woodlands are grounds for concern for the long-term persistence, and ecological function and ecosystem service provision values of these systems. Considering the significance of poplar box woodlands, it is vital that management actions be taken to conserve and improve these systems. To do this, it is necessary to have a deeper understanding as to how agricultural intensification and associated environmental factors affect the condition of these ecosystems.

1.2 Agriculture, patch and landscape change

The use and modification of land by humans alters the structure and functioning of ecosystems (Vitousek et al. 1997). Human enterprises, such as agriculture, directly and indirectly transform land, alter biogeochemical processes, and contribute to losses and additions of biota (Vitousek et al. 1997). Global studies have shown that agricultural intensification gradients have negative associations with the richness and diversity of various plant, animal and soil taxa (e.g. Culman et al. 2010; Flohre et al. 2011; Flynn et al. 2009; Kremen et al. 2002; Philpott et al. 2008). The modification of the landscape for the growth and intensification of agricultural production now threatens biodiversity, ecosystem function and overall ecosystem resilience (McAlpine et al. 2002; Saunders et al. 1991; Vitousek et al. 1997).

Agriculture is a significant and extensive industry in Queensland, with over 75% of the State utilised for agricultural production (Australian Bureau of Statistics 2011a);

however, agricultural development has not been without consequence. In Queensland, between 1990 and 1995, almost 1 million hectares of vegetation were cleared for agriculture and development, 83% of which was woodland and low woodland (Barson et al. 2000). Remnant woody vegetation in Queensland was once cleared at a rate of 505 000 ha/yr (1999-2000), but has since been reduced to 38 000 ha/year (2008/2009) (Department of Environment and Resource Management 2010a). Despite such a significant reduction in vegetation clearing, native vegetation is still being cleared at a faster rate than it can be replaced (Pink 2010b). A further two thirds of vegetation in Australia has been significantly modified in some way for human use, primarily by the grazing of livestock on natural vegetation (Pink 2010a). The growth of agriculture in Australia has led to the significant loss and modification of native vegetation, through changes at the patch and landscape levels.

1.2.1 Agricultural impacts at the patch scale

The extensive clearing of vegetation in Australia has created a modified and fragmented landscape, resulting in the reduction in size of many patches of remnant vegetation (Lindenmayer & Fischer 2006). The reduction in patch size can have a number of ecological consequences. For example, patch size may affect species composition, overall stand structure (e.g. Echeverria et al. 2007), and the structure of dominant species populations (e.g. Prentice & Leemans 1990). Patch size also influences stand structure, explaining trends in the density of trees of early and late maturity in *E. populnea* woodlands (Debus et al. 2009). Measures of tree condition in *E. populnea* woodlands, such as crown structure and crown dieback, may also be affected by patch size, with healthier trees found in larger patches (Batterham 2008).

Fragmentation not only reduces patch size, but it also increases the proportion of edge habitat in comparison to patch area (Fletcher 2005). Smaller patches have a higher ratio of edge to core habitat than larger patches (Collinge & Forman 2009). Similarly, patches of irregular or elongated shape will have more edge per unit area than patches of simpler shape (Collinge & Forman 2009). When two adjacent ecosystems are divided by an abrupt transition or edge, the resulting interaction between these two ecosystems are known as edge effects (Murcia 1995). The

processes at these edges are often very different to those towards the interior habitat (Donovan et al. 1997; Murcia 1995).

At patch edges, wind speeds, water fluxes, light fluxes, and nutrient levels may differ compared to that of the interior (Saunders et al. 1991; Weathers et al. 2001). For example, Weathers et al. (2001) found that the edge of forest patches may act as a significant trap for air-borne nutrients and pollutants from the surrounding environment, finding that nutrients and pollutants were concentrated at patch edges and decreased towards the interior. Furthermore, tree mortality rates have been found to be higher and more variable at forest edges than within forest interiors due to changes in microclimate and weather (such as windstorms) at edges (Laurance et al. 2007; Mesquita et al. 1999). The distribution and composition of species may also differ at habitat edges compared to the patch interior with the influence of edge effects (Murcia 1995). For example, in forests in Illinois, the richness of herbaceous species decreased with increasing distance from the forest edge (towards the interior) (Gehlhausen et al. 2000 1315).

While the clearing of vegetation is absolute and destructive, and the effects of changes to patch size and shape have been well-documented for remnant woodlands in agricultural landscapes (e.g. Batterham 2008; Debuse et al. 2009), remnant vegetation has also been significantly modified (McIntyre & Hobbs 1999). Modifying processes such as novel species introductions and the alteration of disturbance regimes are known to affect species composition, stand structure, and population level processes which have the potential to affect ecosystem function (e.g. Carey 2003; Tait 2004).

One of the most significant drivers modifying remnant vegetation at the patch scale is the grazing of livestock (Milchunas & Lauenroth 1993; Torre et al. 2007). Livestock grazing may influence floristic composition, by causing declines in native species cover, and increasing the cover of exotic species (e.g. Yates et al. 2000). Structural changes to vegetation stands may also occur with livestock grazing, by reducing and removing native shrubs and perennial herbs (Tunstall & Webb 1981), and generally simplifying overall stand structure (Lunt et al. 2007a). Livestock grazing may further affect stand structure and population dynamics by influencing

the condition and recruitment of dominant tree species (Calvert 2001; Davidson et al. 2007; Prober et al. 2011).

Ecosystems may be modified further by land use practices that alter soil properties. Vegetation remnants in agricultural landscapes often exhibit nutrient enriched soils, through fertiliser application, natural redistribution (by erosion and wind), and inputs from grazing livestock (through urination and defecation) (Duncan et al. 2008). Differences in soil fertility have been known to affect the composition of plants in various ecosystems (Clark et al. 1995; Keith & Myerscough 1993; Lindsay & Cunningham 2009). Furthermore, the increased soil fertility associated with agricultural landscapes also facilitates the growth of exotic plants (Allcock 2002). Conversely, the regeneration of some plant species, including eucalypts, may be hindered by higher soil nutrient concentrations (Ashton & Hall 1992; Colling et al. 2002; Fischer et al. 2009). The enrichment of nutrients in the soil has also been found to contribute to the decline in condition of eucalypts in pastoral regions (e.g. Landsberg 1990b).

More recent fire regimes in Australia, particularly the reduction in frequency of fire in agricultural landscapes (Fensham & Fairfax 2003a), have modified the composition and structure of remnant ecosystems. Differences in fire severity also affect eucalypt recruitment, with the recruitment of *E. delegatensis* seedlings in the Australian Capital Territory being significantly higher at high fire severity than at low fire severity (Vivian et al. 2008). The absence of fire has also been linked to changes in stand structure, and the declining condition of eucalypt species across Australia (Close et al. 2009; Ellis 1985; Lunt 1998).

Therefore, there are a number of factors associated with agriculture at the patch scale which may influence changes in remnant vegetation. However, factors in the surrounding landscape may also modify remnant ecosystems, affecting both community and population level processes.

1.2.2 Agricultural impacts from the broader landscape

While the influence of patch level factors is well studied, the surrounding landscape may also be of importance, particularly in fragmented landscapes. Landscape ecology recognises that patches of vegetation are part of the overall landscape mosaic, which also consists of corridors and a surrounding matrix (Forman 1995). In the mosaic landscape model developed by Forman (1995), patches are relatively homogenous areas that differ from their surroundings, corridors are strips of a particular patch type which connect patches, and the matrix is the dominant feature in the landscape. The mosaic model is an anthropocentric representation of the landscape; it is most evident in human-modified landscapes, where the spatial arrangement of vegetation and land uses are so bound by human activity that the land appears patchy and contrast between patches is high (e.g. remnant forest bordered by cropping fields) (Farina 2006).

Apart from influencing patch sizes and shapes, broad-scale clearing and modification of vegetation may also affect remnant vegetation through changes in the surrounding landscape. Through fragmentation, the distance between patches increases, leading to the isolation or creation of islands of vegetation (Saunders et al. 1991). The connectivity of the landscape has both a structural component, which considers the size, shape, and location of patches and corridors in the landscape (Collinge & Forman 2009); and an ecological or functional component, which considers the connectedness of ecological processes across scales (Lindenmayer & Fischer 2006). Such a loss of connectivity can impair many different components of an ecosystem's ecology. For example, isolation may impair the dispersal of plant constituents such as pollen, spores, and seeds (Lindenmayer & Fischer 2006). Forest connectivity has been identified as one of the more important factors driving trends in tree species richness in Brazil (Metzger 1997).

Land uses surrounding remnant vegetation may also affect patterns in species richness and composition. In a study of the effects of the land use matrix on grassland in Sweden, Öckinger et al. (2011) found that plant richness in grassland fragments was significantly lower in arable landscapes, than in forest landscapes.

Furthermore, in an urban-rural landscape of Michigan, USA, patterns of land use surrounding remnant forests were found to influence the survival of various plant seedlings, affecting overall plant composition (Barrie 2012). Additionally, in a study of native forest birds and the influence of surrounding landscape composition on Banks Peninsula, New Zealand, it was found that the abundance of most bird species was positively associated with the proportion of native forest in the surrounding area, while bird abundance was generally negatively correlated with the proportions of farmland or plantation forest in the surrounding landscape (Deconchat et al. 2009). Although these studies demonstrate the importance of surrounding landscape factors irrespective of patch factors, there is evidence from other systems that suggest a need to consider landscape factors in conjunction with patch factors.

1.3 The importance of both patch and landscape variables in agricultural landscapes

Patch variables have long been of greater research focus when examining the effects of landscape change and agricultural intensification (e.g. Chaneton et al. 2002; Milchunas & Lauenroth 1993; Torre et al. 2007). Although the preceding review shows that patch factors are undoubtedly important, focussing on patch factors alone often leads to certain elements of a heterogeneous landscape being ignored (Joyal et al. 2001, cited in Fischer et al. 2004; Haila 2002). Until recently, the matrix was largely overlooked in research into the ecology of remnant ecosystems; however, as suggested in Section 1.2.2, a number of studies have indicated that the composition of the surrounding landscape may be of significant ecological importance (Barrie 2012; Deconchat et al. 2009; Öckinger et al. 2011).

Since patch and landscape factors have been recognised as being important for trends in composition, structure and population processes in agricultural landscapes, it has been suggested that a combination of both these factors may be needed to adequately understand these trends (Dauber et al. 2003; Lee et al. 2002). The influence of patch and landscape factors on biodiversity has been examined by Dauber et al. (2003), showing variable responses in the species richness of ants, wild bees and vascular plants in managed grasslands in Germany. Patch variables examining habitat quality

were important for vascular plant richness, while matrix variables, such as the proportions of various land uses, were important for ant richness (Dauber et al. 2003). However, the species richness of bees was affected by both patch variables (such as soil type) and matrix variables (such as the proportion of arable land in the surrounding landscape) (Dauber et al. 2003).

Similar results to those found in Dauber et al. (2003) have been found for plant communities in Alpine hay meadows of Italy, where trends in plant species richness were better explained by both patch and landscape variables rather than patch or landscape variables alone (Marini et al. 2008). Both patch and landscape factors were important for wintering bird communities of Georgia, USA (Pearson 1993). In Pearson's (1993) study, statistical models found that while patch or matrix variables alone affected the abundance and richness of some species and subgroups, patterns of individual species and subgroups were mostly attributed to a combination of both patch and landscape factors. For example, factors such as shrubby vegetation and habitat diversity in the surrounding matrix were found to be important for generalist bird species (Pearson 1993).

Results comparable to those found in Pearson's (1993) study were also found for bird communities in an agricultural landscape in Ontario (Lee et al. 2002), and roadside transects in Vancouver (Melles et al. 2003). Similar to bird species, patterns of beetle diversity in Israel have also shown varied responses to patch and landscape factors in semi-arid agricultural landscapes (Yaacobi et al. 2007). In particular, landscape variables were most important for carabid diversity, while tenebrionids were affected by both patch and landscape variables, or patch variables alone, depending on patch size (Yaacobi et al. 2007).

In Australia, there are limited studies which examine the effects of both patch and landscape factors on remnant vegetation. Those studies which do examine both, often focus on the responses of animal populations (e.g. Brady et al. 2011; Fischer & Lindenmayer 2002; McAlpine & Eyre 2002; Westphal et al. 2003), with fewer examining patterns of vegetation (but see Chilcott et al. 2005; Lindsay & Cunningham 2009; Reardon-Smith 2011; Reid et al. 2007). The few vegetation studies which have considered patch and landscape factors have mostly found that

both are of importance (e.g. Chilcott et al. 2005; Lindsay & Cunningham 2009; Reid et al. 2007). In a study of remnant poplar box woodlands in the Maranoa-Balonne Catchment, Queensland, Chilcott et al. (2005) found that patterns in dieback of dominant trees and shrubs were driven by patch factors. Patterns in ant species and herbaceous plant species were driven by landscape factors, while reptile species' abundance and inbreeding in the dominant tree species were driven by both patch and landscape factors (Chilcott et al. 2005). Furthermore, the understorey vegetation in small remnant grassy woodlands in agricultural landscapes in New South Wales was also influenced by both patch and landscape factors, with patch variables such as patch grazing and soil fertility influencing exotic plant cover, and landscape variables such as surrounding land use affecting understorey vegetation and composition of the soil seed bank (Lindsay & Cunningham 2009).

The findings of these studies suggest that the relative importance of patch and landscape factors varies for different components of an ecosystem, but essentially both are important for the ecosystem as a whole. Therefore, efforts in conservation and management are likely to be more beneficial when the effects of both patch and landscape factors are considered (Lee et al. 2002).

1.3.1 Patch and landscape hypothesis

The above review demonstrates that patch factors are traditionally of greater interest in ecological studies of community and population processes (Figure 1.1a). Other studies have also examined the role of factors in the surrounding landscape (independent to patch factors); but this is a less common approach (Figure 1.1a). Both these views accept that community and population processes influence the overall health of the ecosystem, but may also, to some extent, influence one another. Here, ecosystem health is used as a broad term to represent the condition of the system in terms of biodiversity, ecological function and processes and resilience. An alternative view recognises that both patch and landscape factors may be of greater importance for community and population level processes than the independent effects of patch or landscape factors (Figure 1.1b). This 'patch and landscape' hypothesis, which recognises the importance of multi-scale considerations (Cushman

& McGarigal 2002), may be more appropriate to gain a better understanding of overall ecosystem health.

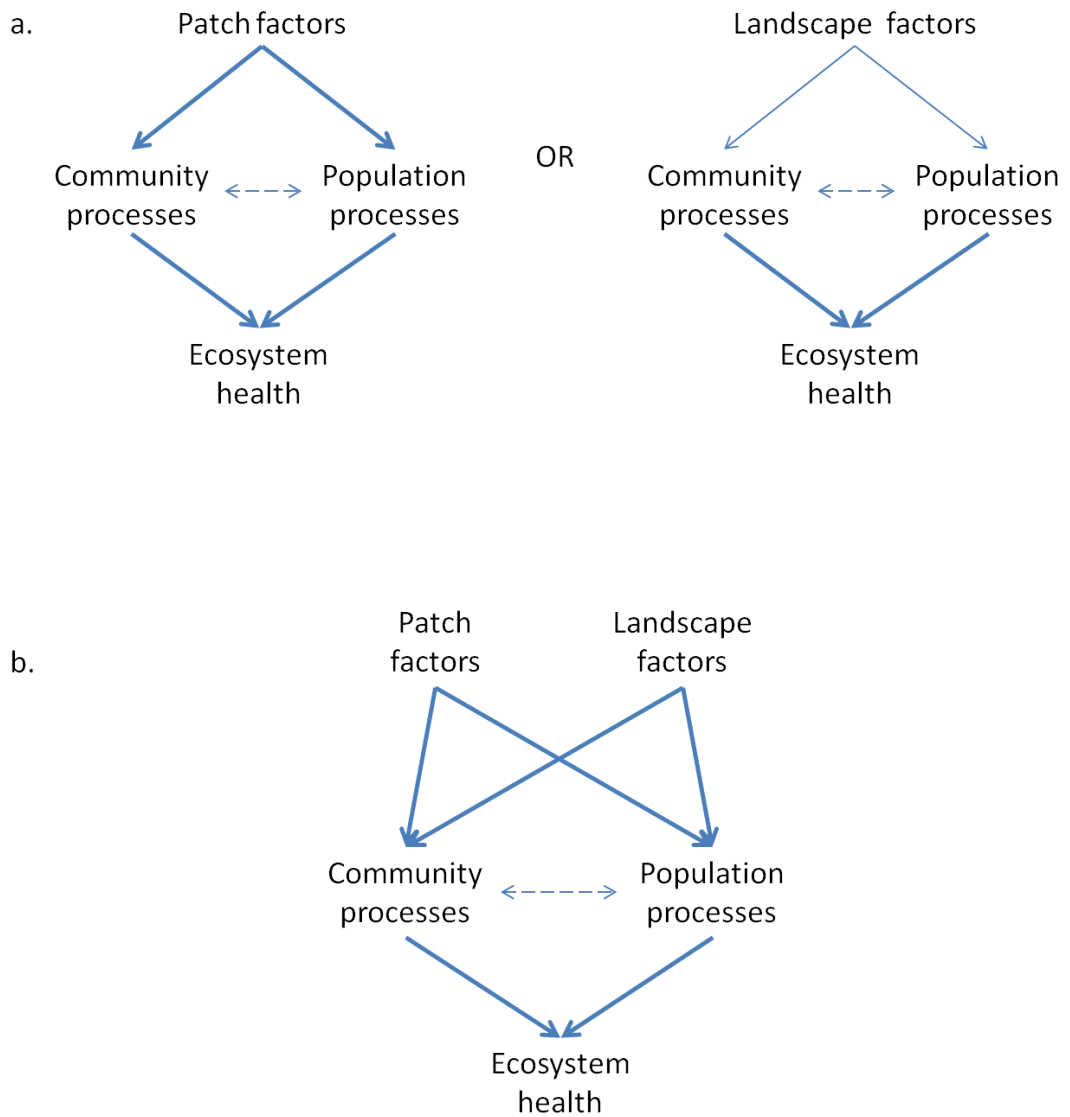


Figure 1.1 (a) Conceptual diagram of independent drivers of community and population processes from literature; (b) alternative 'patch and landscape' hypothesis

1.4 Study questions and hypotheses

From the literature, it can be hypothesised that both patch and landscape factors influence the health of remnant poplar box vegetation in agricultural landscapes in Queensland. This will be determined by examining the health of poplar box woodlands in the Condamine Catchment in relation to various patch and landscape factors, particularly in relation to land use. The effects of patch or landscape factors have been shown to manifest themselves as changes in a range of responses; many of the patch and landscape factors discussed earlier influence community level responses in floristic composition or stand structure, but may also influence changes in population dynamics and the condition of dominant species.

This study will examine the following questions:

1. Do patch grazing and land use context affect floristic composition, functional group richness, richness transition ratios, and stand structure in *Eucalyptus populnea* woodlands in the Condamine Catchment?
2. Do patch and landscape variables in combination explain variation in floristic composition and stand structure in *Eucalyptus populnea* woodlands in the Condamine Catchment?
3. Do patch grazing and land use context affect tree condition and tree population age structure in *Eucalyptus populnea* woodlands in the Condamine Catchment?
4. Do patch and landscape variables in combination explain variation in tree condition and population age structure in *Eucalyptus populnea* woodlands in the Condamine Catchment?

Questions 1 and 3 will be addressed using an *a priori* design (natural experiment). The design consists of 2 factors; patch grazing intensity (3 levels: no grazing, low grazing and high grazing); and land use context in terms of the proportion of cropping and grazing in the surrounding landscape (3 levels: cropping, mixed, grazing). This is applied to both community level responses (e.g. floristic composition, species richness, functional group richness, richness transition ratios

and stand structure) and population-level responses (tree condition and population age structure) of *E. populnea*.

Questions 2 and 4 will be addressed using a correlative approach. Drivers of community level responses (floristic composition, species richness, functional group richness, richness transition ratios and stand structure), and population level responses (tree condition and population age structure) are determined using canonical correspondence analyses and model averaging (generalised linear modelling) respectively, utilising a wide range of patch and landscape explanatory variables.

1.5 Thesis overview

Chapter 2 provides a detailed overview of the study area, including its history of development, vegetation management, soils and climate. The *a priori* experimental design is also explained. Chapter 3 examines how patch grazing and land use context affect poplar box woodlands in terms of floristic composition, species richness, functional group richness, richness transition ratios, and stand structure.

Environmental drivers of multivariate patterns in floristic composition and stand structure are also examined. Chapter 4 examines patterns in *E. populnea* tree condition and population structure in relation to patch grazing and land use context. Environmental drivers of tree condition and population structure of *E. populnea* are also analysed. Finally, Chapter 5 discusses the importance of key findings in relation to future persistence of poplar box woodlands in agricultural landscapes. The implications of these results, as well as future management and study directions, are also discussed.

Chapter 2: History of development, native vegetation in the Condamine Catchment, and experimental design

2.1 Study region:

2.1.1 The (Southern) Brigalow Belt Bioregion

The Brigalow Belt Bioregion occupies a large part of both Queensland and New South Wales, covering approximately 364 000 km² from Townsville to northern New South Wales (Sattler & Williams 1999). This bioregion has been an area of significant agricultural and pastoral growth, and also has important mining and forestry industries (Sattler & Williams 1999). This substantial development has been accompanied by extensive and rapid loss of vegetation and declines in species populations (Sattler & Williams 1999).

There have been two features of this bioregion which have made it so successful in terms of development. The moderate subtropical temperature regime allows for a wide range of crops to be grown, and soils associated with Brigalow vegetation, such as heavy grey and brown cracking clays, have moisture storage capabilities which are advantageous for the growth of crops as well as for sown pasture (Lloyd 1984b). Using these two features strategically and opportunistically has significantly increased the potential for and success of agricultural activities (Lloyd 1984b).

The Southern Brigalow Belt Bioregion occupies 279 496 km²; of this, 81% is in Queensland and 19% in New South Wales (Department of Urban Affairs and Planning 2000). The climate in Queensland's Southern Brigalow Belt Bioregion is subtropical and highly variable (Thornton et al. 2007). Rainfall is summer dominant, with an annual average of 720 mm (Thornton et al. 2007). Due to the highly variable rainfall in this region, extended dry periods are also common (Lloyd 1984b).

The Brigalow Belt takes its name from the Brigalow (*Acacia harpophylla*) vegetation which dominates the region (Johnson 1984). Apart from open *Acacia* forests and woodlands, other open forests and woodlands dominated by *Eucalyptus melanophloia*, *E. populnea*, *E. crebra*, *E. tereticornis* are also widespread throughout the Brigalow Belt (Johnson 1984). Other communities which make up the mosaic of vegetation comprise of open forests of *Casuarina* and *Callitris*, tussock grasslands and semi-evergreen vine thickets (Johnson 1984).

Although excessive native vegetation clearing is largely controlled by government legislation, such as the *Vegetation Management Act 1999* (Qld), clearing of vegetation has been a common practice. While the State as a whole has been significantly cleared, southern areas of the Brigalow Belt have been most affected, with over 60% of Queensland's clearing activities occurring in this region (Wilson et al. 2002). Previous annual clearing rates of remnant vegetation in the Brigalow Belt Bioregion averaged around 260 200 ha/yr (between 1997 and 1999) (Wilson et al. 2002). However, this has decreased to 12 348 ha/yr (between 2008 and 2009) (Department of Environment and Resource Management 2010a). Clearing of remnant vegetation has been so extensive in the southern Brigalow Belt Bioregion that in 1999 only 41% of the pre-clearing extent remained (Wilson et al. 2002).

2.1.2 The Condamine Catchment

This study was conducted in and around the Condamine Catchment, on the Darling Downs in the Southern Brigalow Belt, Queensland. The catchment lies west of the Great Dividing Range, and covers an area of 24 434 km² (McDougall et al. 2008). The Condamine Catchment centres around the town of Dalby (151° 16' 0'' E, 27° 11' 0'' S) and is at the headwaters of the Darling River, a tributary of the Murray-Darling Basin in Southern Queensland (Figure 2.1).

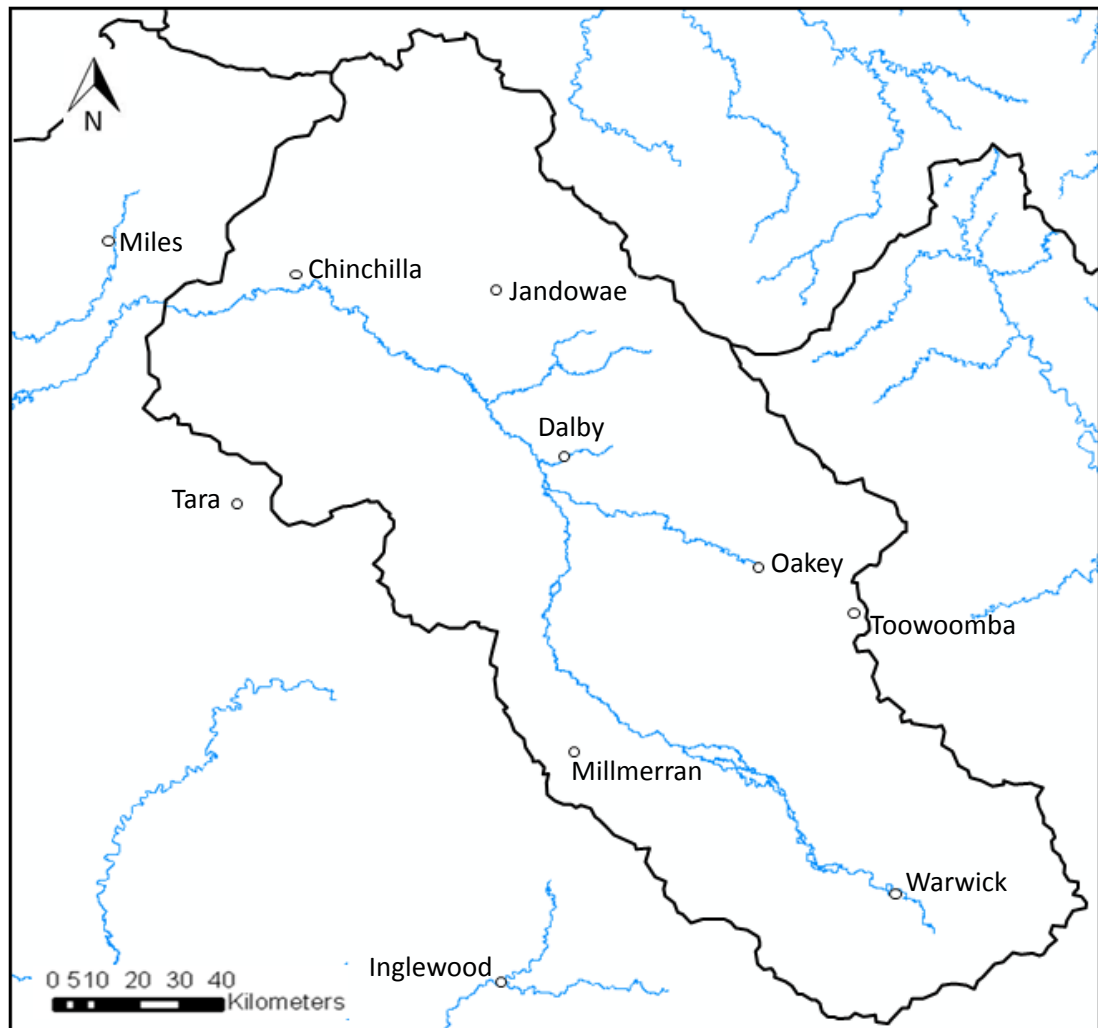


Figure 2.1 Map of the Condamine Catchment, showing the locations of major centres.

The climate in the region is subtropical, with mostly warm, wet summers, and cold, dry winters (Australian Bureau of Meteorology 2011). Average temperatures range from 25 °C to 11 °C in Warwick, 12 °C to 26 °C in Dalby, and 27 °C to 12 °C in Miles (Australian Bureau of Meteorology 2012). Frosts can also be quite frequent and severe in lower lying areas in winter (Biggs & Carey 2006). The region becomes more arid towards the NW, with average annual rainfalls of 700 mm in Warwick, 670 mm in Dalby, and 650 mm in Miles (Australian Bureau of Meteorology 2012). Although rainfall is summer dominant, significant falls may occur at any time throughout the year (Australian Bureau of Meteorology 2012). The region had been in prolonged drought; however, due to heavy rainfall and floods in March 2010

(National Climate Centre 2010), was declared no longer in drought in May 2010 (Department of Agriculture, Fisheries and Forestry 2010). There were further heavy rainfall and floods in the months leading up to sampling in early 2011.

The Condamine Catchment is bordered by the Great Dividing Range in the east, a feature created 19-23 million years ago in association with volcanic activity (Biggs & Carey 2006). In terms of topography, most of the Condamine Catchment is relatively flat, with over 50% of the catchment having a slope of <1% (McDougall et al. 2008). Many creeks flow from the Great Dividing Range, which converge on the Condamine River (Biggs & Carey 2006). Surrounding the Condamine River and its tributaries are floodplains which are subject to two types of flooding: overbank (or riparian) flooding from defined stream courses; and flooding resulting from overland or sheet flow of runoff moving across the floodplain as a result of localised rainfall (Condamine Alliance 2010).

There are various types of soils throughout the Condamine Catchment, with the most common being the highly fertile black, brown, grey and red Vertosols (cracking clays) which are formed in the alluvia on basalt and on sandstone (Biggs & Carey 2006). Some red, non-cracking clay soils may be found in the east of the catchment, while other sandy, gravelly and loamy soils of low fertility are scattered throughout the catchment (Biggs & Carey 2006). The soils of the Condamine Catchment are mostly considered to be fertile; however, the removal of nutrients through agricultural uses has led to a significant decline in fertility (Murray-Darling Basin Commission 2003, cited in Condamine Alliance 2010).

2.2 Background to the Condamine Catchment

2.2.1 Vegetation

Throughout the Condamine Catchment, the varying soils, climate and geological characteristics allow for a diverse range of ecosystems within the region (Johnson 1984). These communities, or regional ecosystems (REs), can be characterised by their geology, landform and soil (Sattler & Williams 1999).

In terms of vegetation, the Condamine Catchment is characterised by *Eucalyptus orgadophila*, *E. crebra* and *E. melanophloia* on basalt hills (Biggs & Carey 2006). In other areas, forests and woodlands of *Acacia harpophylla*-*Casuarina cristata*, *E. populnea*, *Allocasuarina luehmannii* and *Callitris* species are also common (Biggs & Carey 2006).

Remnant native vegetation is concentrated around the catchment boundary in upland areas, while much of the vegetation in the fertile lowlands in the catchment centre has been cleared (Condamine Alliance 2010) (Figure 2.2). In 2003, the remnant native vegetation remaining in the Condamine Catchment in comparison to pre-clearing extent was approximately 27% (Accad et al. 2006, cited in Condamine Alliance 2010). Furthermore, 32 regional ecosystems have a Biodiversity Status of 'endangered', and another 27 regional ecosystem are 'of concern' (Condamine Alliance 2010).

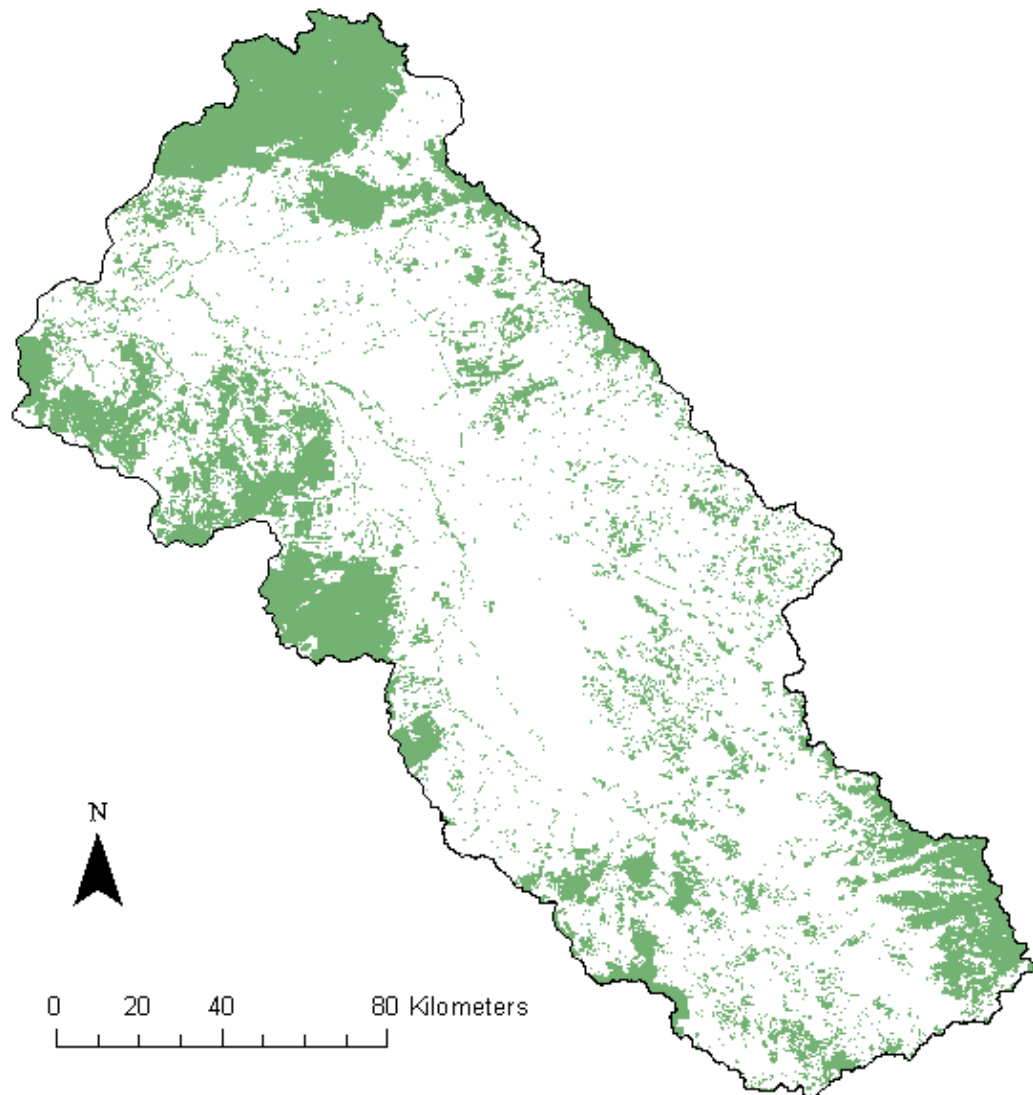


Figure 2.2 Remnant vegetation remaining in the Condamine Catchment (Data source: Department of Environment and Resource Management 2010b).

One particularly important ecosystem within the Condamine Catchment is poplar box (*Eucalyptus populnea*) woodland on alluvial soils (RE 11.3.2). This ecosystem consists of an open tree layer, a grassy understorey and occasional shrubs (Sattler & Williams 1999). The tree layer is dominated by *E. populnea*, but scattered shrubs such as *Geijera parviflora* and *Eremophila mitchellii* may also be present (Queensland Herbarium 2012). The understorey mostly consists of numerous native graminoids, including *Cyperus gracilis*, *Bothriochloa decipiens*, *Chloris* spp. and

Aristida spp., and various forbs such as *Brunoniella australis*, *Boerhavia dominii* and *Einadia* spp. (Wang et al. 2008).

Poplar box woodlands are found on fertile soils, and as such, have been a target of vegetation clearing for agriculture (Kaur & Stanley 2006). Agriculture has grown to cover over 80% of the Condamine Catchment (Table 2.1), leading to the drastic reduction in the cover of poplar box woodlands to accommodate such growth (Department of Natural Resources 2003). As a result, poplar box woodlands have been reduced to just 10% of their pre-clearing extent in the Condamine Catchment (Figure 2.3) (Department of Environment and Resource Management 2009b, 2010b).

Table 2.1 Summary of major land uses in the Condamine Catchment as of 2006
(Data source: Department of Science 2012)

Land use	Area of use (%)
Grazing of natural vegetation	48.41
Cropping	26.84
Production forestry	11.75
Irrigated cropping	7.01
Other minimal use	1.94
Residential	1.29
Nature Conservation	0.92
Intensive animal production	0.25
Mining	0.19

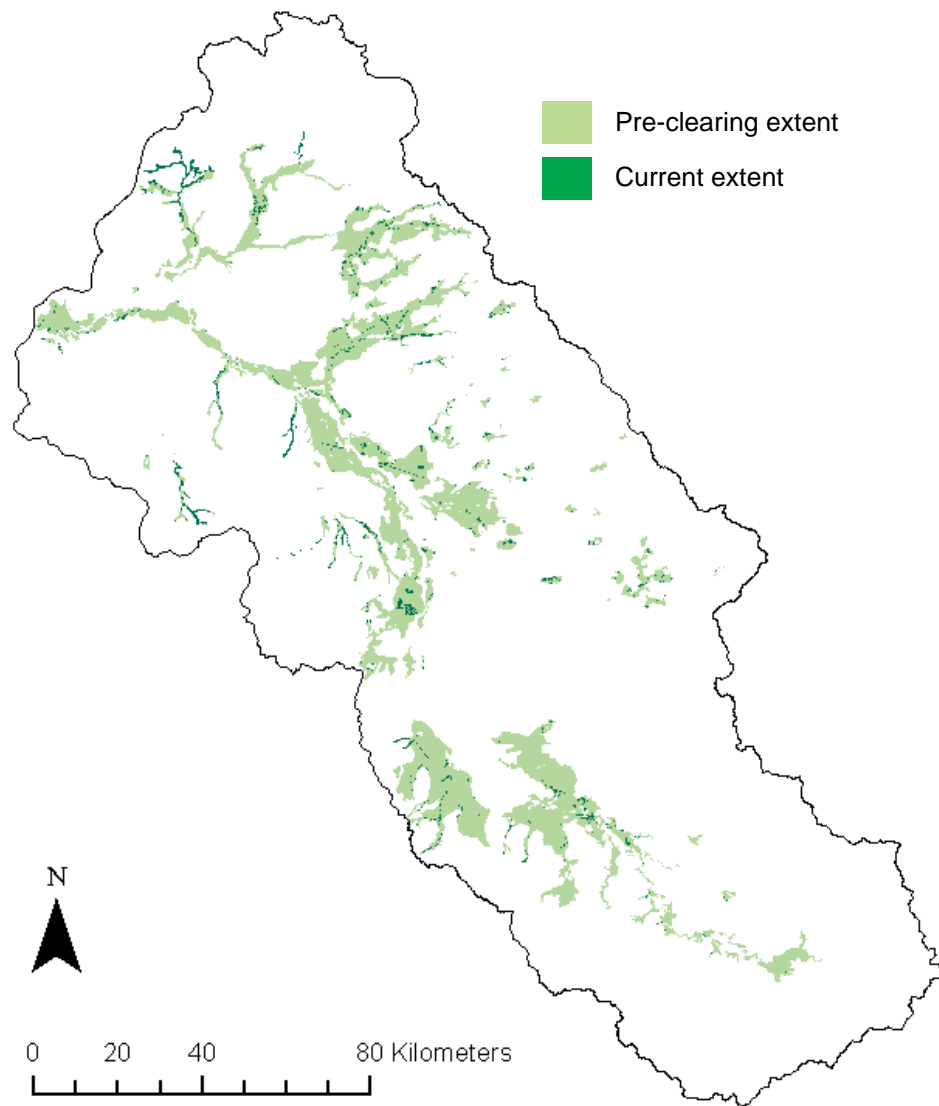


Figure 2.3 Pre-clearing extent and current extent of poplar box woodlands in the Condamine Catchment (Data source: Department of Environment and Resource Management 2009b, 2010b).

Poplar box woodlands play a number of ecologically significant roles, and provide habitat for a diverse range of woodland birds, reptiles and arboreal mammals (Curtis et al. 2012; Lundie-Jenkins & Lowry 2005). Poplar box trees are a significant hollow forming tree, and are important for providing habitat for a diverse range of native fauna (Dorricott et al. 1999). The grass layers of poplar box woodlands provide protection for fauna such as the narrow-nosed planigale (*Planigale tenuirostris*) and

the fat-tailed dunnart (*Sminthopsis crassicaudata*) (Queensland Parks and Wildlife Service (QPWS) 2001). The woodlands themselves are essential habitat for rare and threatened flora and fauna species, including the square-tailed kite (*Lophoictinia isura*), the lesser long-eared bat (*Nyctophilus geoffroyi*) (QPWS 2001) and the endangered bridled nailtail wallaby (*Onychogalea fraenata*) (Lundie-Jenkins & Lowry 2005). Poplar box woodlands are also important habitat for rare and threatened flora, such as *Homopholis belsonii* (Sattler & Williams 1999).

Poplar box woodlands also provide other ecosystem services which benefit agricultural production and the economy. The deep rooted trees bring nutrients from lower horizons in the soil up to the surface (QPWS 2001). Being a eucalypt, poplar box trees can extract more water from the soil and maintain higher moisture tension than many other plants, and so contribute to the regulation of groundwater depth, and prevent the salinisation of soil (QPWS 2001). Furthermore, woodland trees such as *Eucalyptus populnea* are known to act as ecosystem wicks, increasing the infiltration of water through soil (Eldridge & Freudenberger 2005).

Poplar box woodlands provide further services such as nutrient cycling, when plant, animal and other organic matter in the soil are broken down (QPWS 2001). Materials are broken down by inhabitants of eucalypt woodlands, such as mycorrhizal fungi, ants and termites, which also contribute to the loosening of the soil and increase moisture penetration (QPWS 2001). The trees, shrubs and other groundcover of the woodlands also aid soil quality by reducing wind and water erosion (QPWS 2001).

In the Condamine Catchment, *E. populnea* woodlands are mostly found within highly agriculturally productive landscapes. Approximately 95% of poplar box woodlands are utilised for the grazing of livestock in the Condamine Catchment (Department of Environment and Resource Management 2010b; Department of Natural Resources 2003). These ecosystems may therefore be subject to significant disturbance from livestock grazing and associated management practices such as fire exclusion (Fensham & Fairfax 2003a).

2.2.2 History of development

Prior to European colonisation, Indigenous Australians occupied the Condamine Catchment and the Southern Brigalow Belt Bioregion for some 40 000 years (Walsh 1999, cited in Seabrook et al. 2006). During their occupancy in the region, they altered the landscape through their use of various flora and fauna (Seabrook et al. 2006). The landscape was modified further through the use of fire to facilitate new plant growth, increase grass cover and manipulate fauna (Fensham 1997). However, it has been suggested that the extent to which fire was used is much lower than originally thought (Fensham 1997).

During initial European settlement, cultivation was limited on the Darling Downs, and land was mostly utilised for pastoral activity, such as the production of wool (Seabrook et al. 2006). However, the success of stock grazing significantly diminished at the turn of the century, due to prolonged drought and other environmental factors such as the spread of prickly pear (*Opuntia* spp.) (Seabrook et al. 2006). The vegetation in the area was originally viewed as ‘useless scrub’; grazing and the spread of prickly pear only promoted this view (Lloyd 1984a).

The eventual control of the prickly pear and clearing of trees on grazing properties resulted in an increase in stock capacity, making these lands more economically viable (Seabrook et al. 2006). In 1884, the ‘improvement’ of pastoral lands by the clearing of trees was encouraged through legislation and compensation (Seabrook et al. 2006). Early stages of vegetation clearing (until the 1930s) destroyed 15-20% of forest cover in the region; however, the rate and extent of tree clearing greatly increased with the introduction of mechanical removal methods, often involving dragging a large chain between two bulldozers (Fensham & Fairfax 2003a).

There have been dramatic changes in land use in Australia, the steadily growing demand on production making it necessary for land to be used for the grazing of livestock and cultivation of crops (Fensham & Fairfax 2003a). The extensive use of the Condamine Catchment for agriculture has led to significant losses of native vegetation (Condamine Alliance 2007). In the Condamine Catchment alone, clearing

rates of remnant woody vegetation reached a high of 4 356 ha/yr for the period between 1999 and 2000 (Department of Natural Resources and Water 2008). While these rates varied dramatically in following years, annual clearing rates have since been reduced, with only 822 hectares of remnant woody vegetation being removed from 2005-2006 (Department of Natural Resources and Water 2008). Such activity has unfortunately led to the loss of over 70% of remnant vegetation within the Condamine Catchment (Accad et al. 2006, cited in Condamine Alliance 2010).

The loss and modification of vegetation has allowed for the extensive growth of the agricultural industry in southern Queensland. As a result, approximately 31% of land throughout the catchment is utilised for the cultivation of crops (both dryland and irrigated) (Figure 2.4) (McDougall et al. 2008). In winter, crops such as wheat, barley, oats, and chick peas are cultivated, while the summer climate is ideal for the growth of sorghum, sunflower, maize, cotton and mung beans (Biggs & Carey 2006).

In Queensland, the agricultural industry contributes over \$9 billion to Australia's GDP, 9.4% of which comes from production in the Condamine Catchment (Australian Bureau of Statistics 2011b). Of this, 51% is contributed by crop production throughout the catchment (Australian Bureau of Statistics 2011b). Land used for cropping occurs mostly in the east of the catchment area, in relation with more fertile soils and higher average rainfalls (Department of Natural Resources 2003).

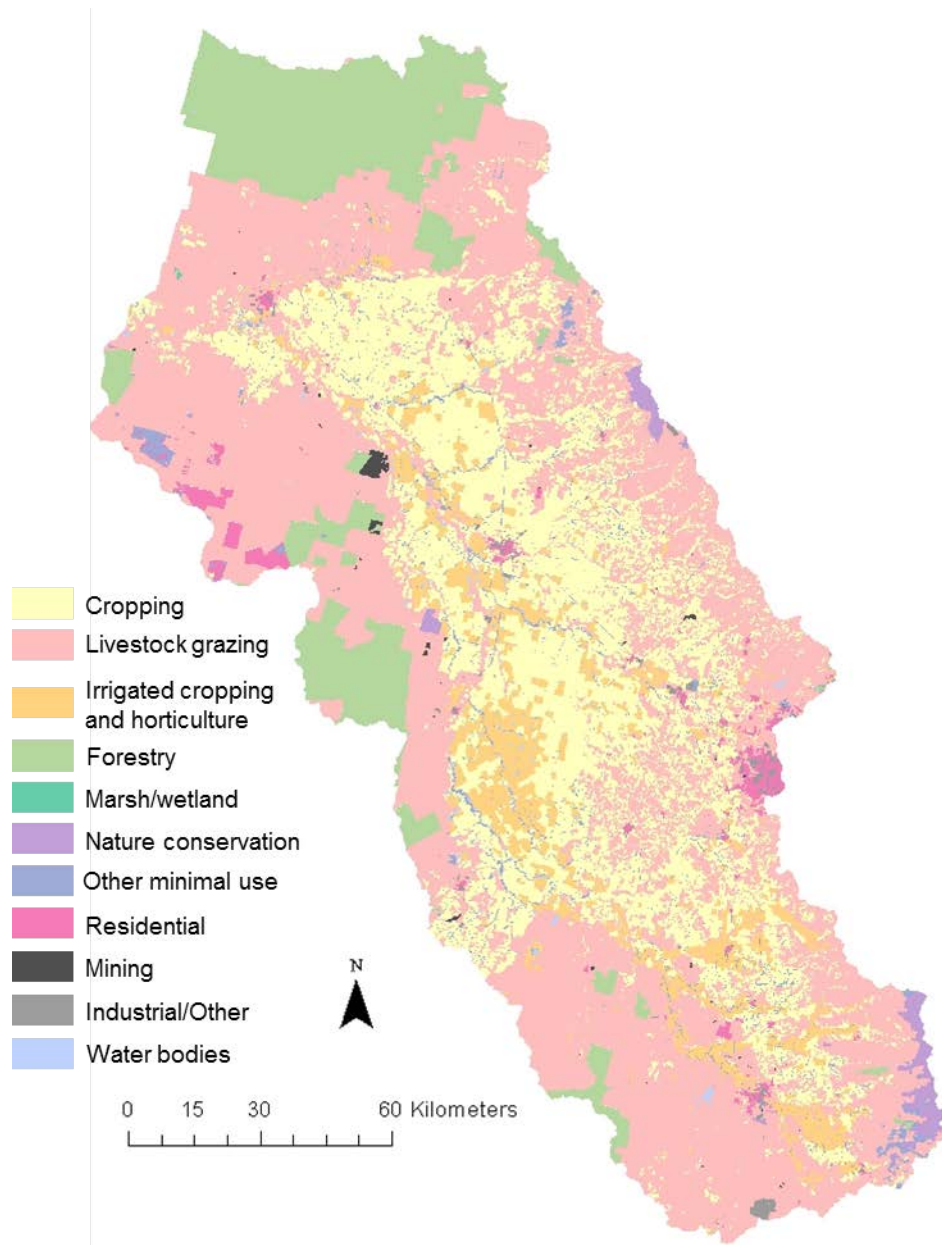


Figure 2.4 Map of land uses in the Condamine Catchment (Department of Science 2012).

Where cropping is not feasible or economical, land is often used for the grazing of sheep and cattle (Weston et al. 1980). Vegetation modification has allowed for the growth of the livestock industry, with 1.5 million hectares or 59.7% of the Condamine Catchment currently being used for the grazing of livestock (Figure 2.4) (McDougall et al. 2008). Livestock grazing is most dominant in the west of the Condamine Catchment (Department of Natural Resources 2003), as the drier

conditions in this region are less suited to cropping. In economic terms, livestock production contributes to 30% of agricultural commodities in the Condamine Catchment (Australian Bureau of Statistics 2011b).

Other industries are present in the Condamine Catchment; however, they are not as extensive as the agricultural industry. Forestry is practiced on 11% of land in the Condamine Catchment (Department of Science 2012). It is the second largest contributor of native vegetation clearing in the catchment (Department of Environment and Resource Management 2010a). Mining is another growing industry in the Condamine Catchment, with various coal mines from Killarney to Acland; coal seam gas and basalt rock are also extracted in the catchment (Biggs & Carey 2006). Mining contributed to 28% of non-agricultural clearing (2004-05); making the expansion of mining in the catchment a growing threat (Condamine Alliance 2010).

Across the Condamine Catchment, industry and community utilise on average, 315 000 ML of water per year, 40% of which is obtained via groundwater and the remainder from surface water (Wolfenden & Evans 2004). However, most of this water (90%) is consumed for rural uses, mostly irrigation (Wolfenden & Evans 2004). Current groundwater extraction exceeds water recharge over much of the Central Condamine Alluvium (White et al. 2010), with extraction often being 2 or even 5 times the average potential rainfall recharge (Commonwealth Scientific and Industrial Research Organisation (CSIRO) 2008). A heavy reliance on groundwater extraction has led to most of the Condamine River (its North Branch and the Oakey Creek tributary) to be under maximum losing conditions, meaning that the river is largely disconnected from the underlying aquifer (CSIRO 2008). There are many ecosystems in Australia which are dependent on groundwater resources. When these groundwater dependent ecosystems experience changes to the quantity, quality, timing, or distribution of groundwater, it may affect transpiration by vegetation, recruitment of seedlings, and may result in the significant loss of overstorey and understorey species (Groom et al. 2000; Murray et al. 2003).

Coal seam gas developments in the region also have potential to cause groundwater drawdown and contamination (Arrow Energy 2012). Water resource management in the Condamine Catchment has caused significant reductions in the frequency of high

and low river flows, reductions in low flow duration, and increases in the frequency of no flow periods in comparison to natural flows (Sheldon et al. 2000).

2.3 Experimental design

Site selection in this study focussed on remnant vegetation classed as regional ecosystem (RE) 11.3.2. This RE encompasses woodlands or open woodlands occurring on Cainozoic soils, dominated by *Eucalyptus populnea*. The understory is mostly composed of grasses; however, low scattered trees and shrubs may be present at times (Sattler & Williams 1999).

To determine how patch grazing intensity and land use in the surrounding landscape affect poplar box woodlands, a mixed design was employed. As the most common use of poplar box woodlands is for the grazing of livestock (Department of Natural Resources 2003), three categories of grazing intensity (no grazing, low grazing and high levels of grazing) at the patch level were implemented (Table 2.2, Figure 2.5). The level of grazing at each patch was determined based on the presence and intensity of grazing. Patches which had no recent history of grazing, and were classified as having minimal use (Department of Natural Resources 2003) were deemed to be ungrazed. Those patches which had not been grazed for 2 months or more but had been grazed in the past, were classified as having a low level of grazing, as were those which were currently grazed to a low level (subjectively determined). Patches which were currently grazed to a subjectively medium or high level, or had been grazed in the previous 2 months were classified as having a high level of grazing. The subjective judgement of the grazing intensity at each site was determined using a combination of landowner knowledge, presence of livestock, and visual factors such as cowpat density, cropped grass and bareground.

The most common land uses in the Condamine Catchment are grazing and cropping (Table 2.1, Figure 2.4), with the proportions of cropping and grazing in the surrounding landscape mostly being inversely related for sites studied within the Condamine Catchment. Land used for irrigated cropping was combined with dryland cropping, to get a summed percentage of cropping in the surrounding landscape (see

Chapter 2.4). Similarly, grazing of natural vegetation and intensive animal production were also summed to produce a measure of grazing in the surrounding landscape (see Chapter 2.4). The effect of the surrounding landscape was tested using categories based on the amount and type of land use within a 5 km radius. A 5 km radius was chosen as this has been used in other studies to examine the effect of the matrix on vegetation (Debusse et al. 2009; Williams et al. 2006). Based on the proportions of cropping and grazing in the surrounding landscape, three categories of land use context were utilised; cropping landscapes, mixed landscapes and grazing landscapes (Table 2.2, Figure 2.5, and Figure 2.6). As cropping land uses are generally more intensive than grazing land uses (Martin et al. 2006), this categorisation may also represent a gradient of agricultural intensification.

Table 2.2 *A priori* treatments and their levels.

Treatment	Level	Description
Patch grazing	No grazing (N)	No recent history of grazing.
	Low grazing (L)	Subjectively low levels of patch grazing currently practiced or 2+ months prior.
	High grazing (H)	Subjectively medium to high levels of patch grazing currently practiced or in 2 months prior.
Land use context	Cropping (C)	70% or more of land used for cropping within a 5 km radius.
	Mixed (M)	40-60% of land used for cropping and 40-60% of land used for grazing within a 5 km radius.
	Grazing (G)	70% or more land used for grazing within a 5 km radius.

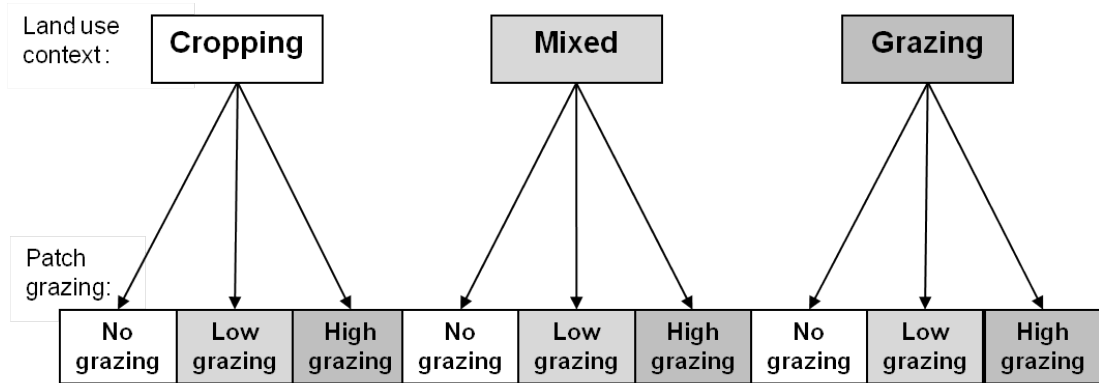


Figure 2.5 Experimental design showing main factors - patch grazing (3 levels) and land use context (3 levels).

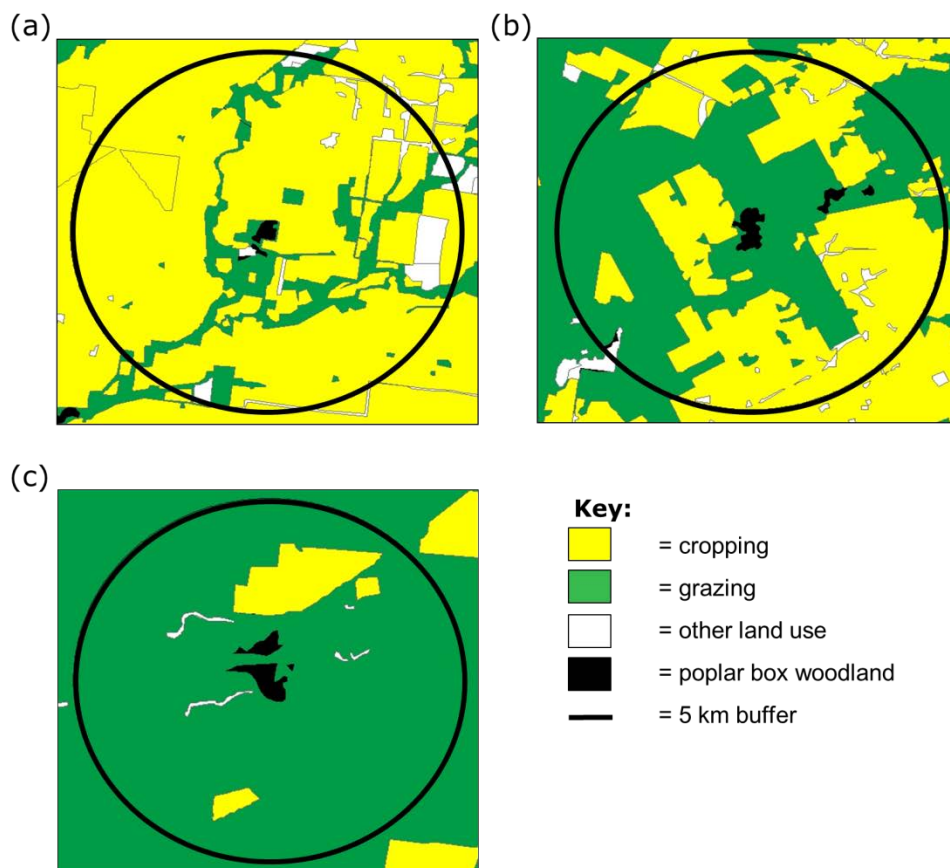


Figure 2.6 Diagrammatic examples of the matrix composition in a) cropping landscapes, b) mixed landscapes and c) grazing landscapes.

2.4 Site selection

ArcMap version 10.0 (Environmental Systems Research Institute (ESRI) 2011) and groundtruthing were used extensively to determine which patches of poplar box woodland were a suitable fit for the experimental design. Using ArcMap (ESRI 2011) and Regional Ecosystems mapping data (Department of Environment and Resource Management 2010b), all patches of poplar box woodland (classified as RE 11.3.2) 5 ha or greater in size were selected. Patches of poplar box woodland were limited to a region within or near the boundaries of the Condamine Catchment. 5 km buffer zones were then created around the centroid of each patch of suitable vegetation. The area within this buffer occupied by each different land use was summed for each potential site (Department of Natural Resources 2003). Sites were short-listed based on the proportion of land used for cropping and grazing within this buffer zone, being suitable if they met any of the criteria for categories of land use context outlined in Table 2.2.

Those sites that met any of the above criteria were then examined based on the intensity of grazing at the patch. This was determined using a combination of Queensland land use mapping (Department of Natural Resources 2003), ground-truthing and land owner knowledge. A site was suitable for this study if it met any of the criteria relating to levels of patch grazing intensity outlined in Table 2.2.

When potential sites met these previous criteria, suitable sites were chosen based on geographical location. A minimum of 2 sites for each combined patch grazing/land use context category were then chosen to be well spread across the study area, with two sites of the same category being at least 5 km from one another to minimise spatial confounding effects. These were then ground-truthed to ensure correct classification. Based on this design and process, 33 sites were selected for field sampling (Table 2.3, Figure 2.7).

Although it was preferable to have more than 2 sites for each category, this was not always possible due to limited occurrence in the landscape, accessibility and issues with misclassification (mapping errors). Often a patch of vegetation mapped as RE

11.3.2 would be selected; however, when ground-truthed would be comprised of a completely different vegetation type. Further difficulties were encountered when suitable sites were located on private property. While many landholders were cooperative, a number of landholders were not responsive.

It should be acknowledged that at site 16, only 49% of the surrounding landscape was grazed (Department of Natural Resources 2003). However, 43% of surrounding land was used for forestry (Department of Natural Resources 2003), but was also largely grazed (Pers. Obs.). As such, it was considered appropriate to categorise this site as being within a grazing landscape.

It may be noted that there was some clustering of sites within certain treatments (Figure 2.7); particularly with those in grazing landscapes with low levels of patch grazing, and perhaps those in cropping landscapes with high levels of patch grazing. This was a sub-optimal design in relation to spatial autocorrelation, and appropriate caution has been taken with interpretation of results. Furthermore, some sites (sites 7, 15, 16 and 19) were located outside of the boundaries of the Condamine Catchment. These sites were considered suitable for use in this study as they have similar climate and topography to those within the boundaries of the Condamine Catchment.

Table 2.3 Categorisation of sites based on a priori treatments.

Patch grazing intensity	Land use context	Symbol	N	Site Numbers
No grazing	Cropping	△	6	4, 5, 6, 17, 27, 28
	Mixed	○	4	2, 3, 8, 14
	Grazing	□	7	7, 9, 10, 15, 19, 20, 23
Low grazing	Cropping	▲	4	1, 12, 18, 22
	Mixed	●	3	24, 29, 30
	Grazing	■	3	25, 31, 32
High grazing	Cropping	▲	2	13, 21
	Mixed	●	2	11, 26
	Grazing	■	2	16, 33

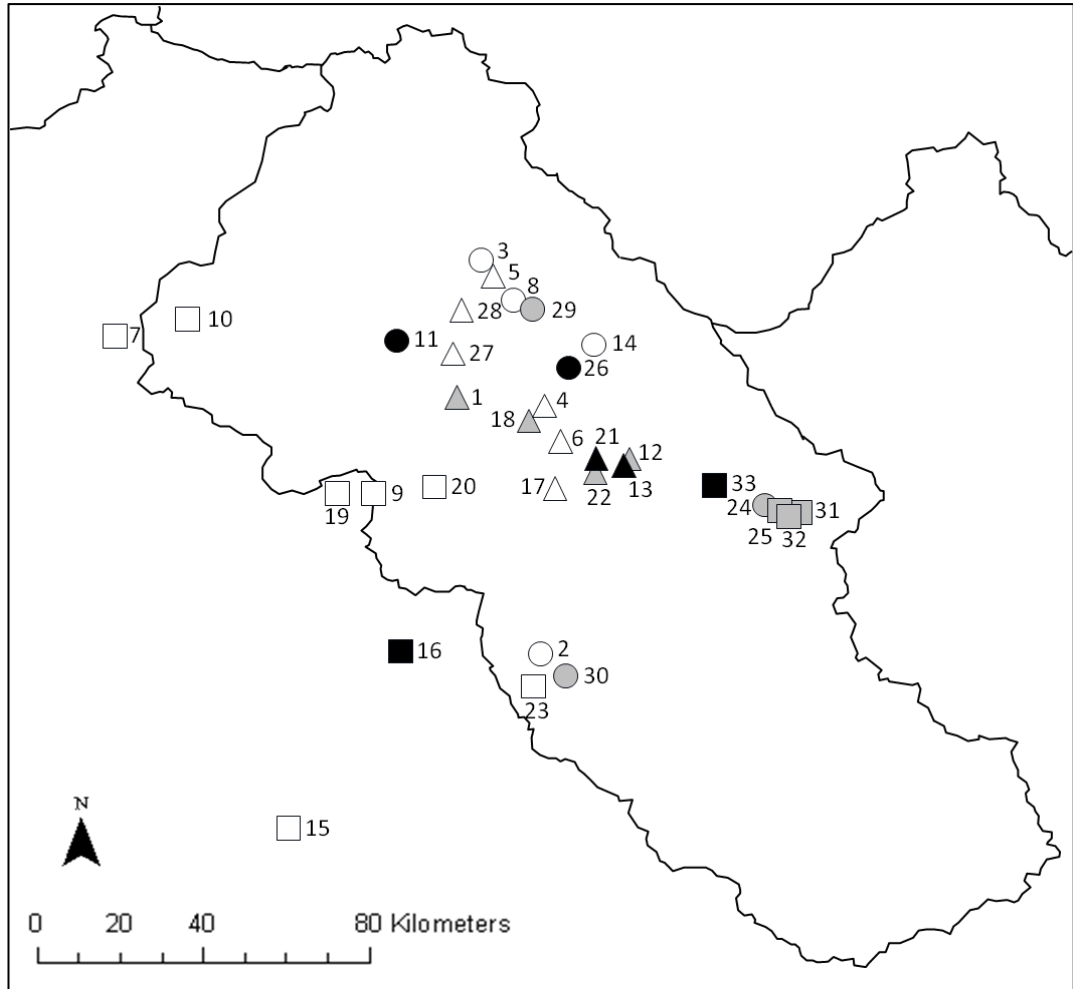


Figure 2.7 Site locations categorised by treatments.

Key: \triangle = No grazing, Cropping; \triangle = Low grazing, Cropping; \blacktriangle = High grazing, Cropping; \circ = No grazing, Mixed; \circ = Low grazing, Mixed; \bullet = High grazing, Mixed; \square = No grazing, Grazing; \square = Low grazing, Grazing; \blacksquare = High grazing, Grazing.

2.5 Approach

The experimental design described above was used to address Study Question 1 (Chapter 1), testing specific hypotheses regarding floristic composition and stand structure (Chapter 3) and population dynamics of *E. populnea* (Chapter 4). A correlative approach (Study Question 2) was used to examine potential drivers of community level and population responses.

Chapter 3: Composition and structure in remnant poplar box woodlands of the Condamine Catchment

3.1 Introduction

Woodlands in agricultural landscapes are declining in health due to various influences associated with agricultural development and management (Yates & Hobbs 1997a). A healthy ecosystem is one which is sustainable (i.e. resilient) in terms of structure and function when under stress (Mageau et al. 1995). Measures of plant diversity and vegetation stand structure have been widely related to ecosystem function (Covington et al. 1997; Ishii et al. 2004; Naeem et al. 1994; Tilman et al. 1997) and also to the stability and resilience of ecosystems (Cordonnier et al. 2008; Tilman et al. 1996; Walker et al. 1999). Therefore, changes in the composition and structure of a woodland may significantly impact the function of these systems and their ability to handle disturbance (Ruiz-Jaen & Aide 2005), which may have significant implications for broader ecosystem health.

The disturbances associated with agricultural management often manifest themselves as changes to floristic composition and stand structure (e.g. Clarke 2003; Ludwig & Tongway 2002). Patterns in floristic composition in woodlands in agricultural landscapes are influenced by numerous factors, such as livestock grazing, soil properties, and land use in the surrounding landscape (Fensham 1998a; Lindsay & Cunningham 2009). The stand structure of woodlands in agricultural landscapes has also been found to be influenced by numerous environmental factors, including vegetation clearing, altered fire regimes and livestock grazing (Fensham & Fairfax 2003b; Ludwig & Tongway 2002). Consequently, this study asks the general question: Do both patch and landscape factors contribute to variation in floristic composition and stand structure in poplar box woodlands in the agricultural landscapes of the Condamine Catchment?

3.1.1 Floristic composition and richness in agricultural landscapes

One of the more significant influences on floristic composition in agricultural landscapes is livestock grazing (e.g. Pettit & Froend 2001). In remnant *Eucalyptus marginata* woodlands in Western Australia, grazed and ungrazed woodlands have been found to differ distinctly in their floristic composition, while previously grazed fenced woodlands showed similarities to both (Pettit & Froend 2001). These differences were attributed to the loss of plants through continual grazing and the recovery after exclusion from grazing (Pettit & Froend 2001). Livestock grazing has also been found to influence floristic composition in the south and central rangelands of Australia, where the long-term impacts of livestock grazing along an intensity gradient (in relation to proximity to water) showed that plant assemblages closer to water (with higher grazing intensity) were distinctly different to those which were further away (lower grazing intensity) (Landsberg et al. 2003). Those sites further from water were composed of plants sensitive to grazing; while grazing sensitive plants decreased towards water sources, plants tolerant to grazing increased in dominance (Landsberg et al. 2003). Landsberg et al. (2003) suggest that these results indicate that changes in plant composition are likely related to the accumulated long-term impacts of water-centred grazing.

Management practices associated with livestock grazing may also affect floristic composition (Dorrough & Scroggie 2008). Management of grazing systems often involves reducing or removing the tree layer (McIvor & McIntyre 2002), which may affect species composition (Le Brocque et al. 2009). In a study of grazed eucalypt woodlands in southern Queensland, Le Brocque et al. (2009) found that different densities of overstorey trees affected plant species composition. Le Brocque et al. (2009) suggest that retaining trees in grazing landscapes provide significant landscape heterogeneity and important refuges for species that may be largely excluded from open grassland habitat. Floristic composition may also be affected by changes in the frequency of fire events, as found in open eucalypt forests on Fraser Island where the assemblage and abundance of plants at sites with a high frequency of fire events distinctly differed to those with a low frequency of fire events (Spencer & Baxter 2006).

Other studies have shown that factors in the surrounding landscape may also affect the composition of remnant vegetation (e.g. Lindsay & Cunningham 2009). In a study of remnant grassy woodlands in southern New South Wales, Lindsay and Cunningham (2009) found floristic composition in remnants adjacent to cropping and grazing land uses were distinctly different to those adjacent to less intensive land uses. Furthermore, Chilcott et al. (2005) found that in poplar box woodlands in the Maranoa-Balonne catchment, variation in the assemblage of herbaceous plant species are explained by factors in the surrounding landscape, such as the amount of clearing within 5 km of the patch.

Apart from examining overall composition, compositional differences may also be examined on the basis of functional groups. Functional groups are sets of species which have similar effects on ecosystems processes and have similar responses to environmental conditions (Walker et al. 1999). This is a widely used approach to examine composition which reduces the complexity of highly diverse systems (Domingues et al. 2007).

Disturbances associated with agriculture may also affect the richness of plants in specific functional groups. Trends in plant functional group richness may be significantly affected by livestock grazing (e.g. Prober & Thiele 1995). Prober and Thiele (1995) found that in grassy *Eucalyptus albens* woodlands in south-eastern Australia, total and native species richness responded negatively to grazing, while most exotic plant species were favoured by grazing. Livestock grazing was also found to positively affect the richness of C3 and C4 plants, with the richness of both these groups found to be significantly higher in grazed than ungrazed tall-grass prairie in Kansas (Collins & Calabrese 2012; Collins et al. 2002). Livestock grazing also affects patterns in floristic functional groups in *Eucalyptus coolabah* woodlands in eastern Australia (Good et al. 2011), *E. salmonophloia* woodlands in south-western Australia (Yates et al. 2000) and *E. melliodora*, *E. albens*, and *E. macrocarpa* woodlands of New South Wales (Prober & Thiele 2004).

Management practices associated with livestock grazing may also affect patterns in plant species richness within functional groups (Dorrough et al. 2011). In grazed

temperate grassy ecosystems of Australia, various native and exotic species responded differently to phosphorus enrichment (Dorrough et al. 2011). Soil phosphorus enrichment through fertiliser application, which is associated with intensification of livestock management practices, resulted in mostly negative responses in native plant species, while exotic species were generally tolerant, which may explain patterns in diversity (Dorrough et al. 2011). Furthermore, in temperate grassy vegetation of New South Wales, native species richness was negatively associated with soil fertility and water enrichment, while exotic species had a positive association with these variables (Mcintyre & Lavorel 1994). Soil fertility (nutrient enrichment) may also affect life-form richness, with soil enrichment having positive relationships with the richness of herbs, shrubs and trees (Grubb 1987, cited in Pausas & Austin 2001; Peet & Christensen 1988).

It has also been reported that fire exclusion may affect species richness in semi-arid poplar box woodlands, causing a shift in understorey species from mostly grasses to predominantly shrubs (Harrington et al. 1984, cited in Yates & Hobbs 1997b). Furthermore, in eucalypt woodlands of New South Wales, more frequent fire events have been found to have a positive effect on understorey plant species richness (Fox & Fox 1986).

Factors in the surrounding landscape may also affect trends in species richness (e.g. Liira et al. 2008). In a study of volcanic outcrops in modified landscapes in New Zealand, the plant community in the surrounding matrix had a large influence on the vegetation of habitat islands (Wiser & Buxton 2008). Wiser and Buxton (2008) found that 81% of the native flora and 90% of exotic flora present in native habitat islands, also occurred in the matrix. The richness of plants based on life-history also changes with land use intensity, which caused an increase in annual richness and decrease in perennial richness in various agricultural landscapes across Europe (Liira et al. 2008). Total plant species richness also differs with land uses in the surrounding landscape (as discussed in Chapter 1) (Dauber et al. 2003). Furthermore, the proportion of urban elements in the surrounding landscape has also been found to be important for patterns in species richness in Alpine hay meadows in Italy (Marini et al. 2008). However, trends in plant species richness in response to the surrounding landscape has not been widely studied in Australian ecosystems.

A novel way to examine trends in functional groups is through transition ratios, which involve examining the relative trends in reciprocal pairs of functional groups (e.g. increases and decreases in native and exotic richness) (Reardon-Smith 2011). Changes in functional groups and their reciprocal pairs (transition ratios) may be used to examine community responses to changes along environmental gradients (Reardon-Smith 2011).

A meta-analysis of plant trait responses to grazing in a broad range of biomes has shown that perennials mostly respond negatively while annuals respond positively to increasing grazing intensity (Diaz et al. 2007). Furthermore, the species composition of Mediterranean grassland systems has been shown to shift from taller perennial grasses to lower growing annual herbs along a gradient of grazing intensity (Noy-Meir et al. 1989). Similarly, in grassy white box woodlands of south east Australia, floristic composition has been reported to shift from a high richness of native plants to increasing richness of exotic plants along a grazing intensity gradient (Prober and Thiele 1995). C3 and C4 plants differ in their photosynthetic responses to temperature and altitude, and have different water and nitrogen use efficiencies (Cabido et al. 1997; Lattanzi 2010). For example, C4 plants are favoured at higher temperatures, higher light saturation, lower altitudes, and have higher water and nitrogen use efficiencies (Cabido et al. 1997; Johnston 1996; Lattanzi 2010). Therefore the relative richness of C3 and C4 plants may shift depending on the environmental and climatic conditions. These and other relationships may be classified as transition ratios (Reardon-Smith 2011) (Table 3.1).

Table 3.1 Functional group transitions and related environmental gradients (adapted from Reardon-Smith 2011).

Transition ratio	Environmental gradient	Reference
Exotic: Native	Dominance of exotic species with greater disturbance/grazing	(Ordonez et al. 2010; Prober & Thiele 1995)
Short-lived: Perennial	Dominance of short-lived plants with greater disturbance/grazing	(Bagstad et al. 2005; McIntyre et al. 1999; Noy-Meir et al. 1989)
C4:C3	Dominance of C4 plants under higher temperatures or light saturation, or lower altitudes, moisture or nutrients	(Cabido et al. 1997; Johnston 1996; Lattanzi 2010; Wand et al. 1999; Winslow et al. 2003)

Changes to species composition can have a number of implications. For example, the invasion of exotic weeds may have significant effects on ecosystem processes (Humphries 1994). Exotic annuals in Australian woodlands may outcompete with natives, and limit the recruitment (Hobbs & Atkins 1991) and survival (Semple & Koen 2003) of native woody species such as *Eucalyptus albens* and *Eucalyptus melliodora* (see Chapter 4). The species richness of vegetation has also been known to influence aspects of stand structure, with plant cover positively associated with species richness (Scherber et al. 2010; Vicca et al. 2007). Furthermore, the invasion of the exotic grass *Ehrharta calycina* alters characteristics of the fuel bed, increasing the frequency and intensity of fires (Wycherley 1984, cited in Yates & Hobbs 1997b). Through changes in species composition and related measures, the functional traits of biota may change, which can result in altered ecosystem processes (Chapin et al. 1997; Chapin et al. 2000).

3.1.2 Stand structure in agricultural landscapes

There are a number of disturbances associated with agricultural landscapes which may affect the stand structure of an ecosystem. In particular, the grazing of livestock is known to have a considerable effect on the stand structure of remnant vegetation in agricultural landscapes (Duncan et al. 2007; Ludwig & Tongway 2002). For

example, the cover of perennial grasses changes with livestock grazing, where native perennials decrease in cover, while exotic perennials increase in cover in grazed Queensland eucalypt savannas (Ludwig & Tongway 2002). Furthermore, the grazing of livestock has been identified as a factor limiting the recruitment of woody species in *Allocasuarina luehmannii*, *Eucalyptus largiflorens* and Mallee woodlands, with the recruitment of woody species being more common with exclusion from livestock grazing (Duncan et al. 2007). In poplar box woodlands, the grazing of livestock reduces forb and herbage biomass, increases the proportion of bare soil, and reduces the abundance of shrubs (Prober & Thiele 2004; Tunstall & Torsell 2004; Tunstall et al. 1981). The grazing of livestock may also interact with fire regimes to affect stand structure, reducing grass cover and fine fuel for fire, allowing for more successful establishment of woody plants, and giving existing trees and shrubs a competitive advantage (Burrows et al. 2002).

Management practices associated with grazing, such as tree clearing, may also affect stand structure. Selective tree clearing has been extensively practised in conjunction with agricultural development, and can contribute to an increase in the density of stems in the tree-shrub layer (thickening) (Scanlan 1988). Mature regrowth vegetation following tree clearing may have a higher cover of the overstorey canopy (Ludwig & Tongway 2002). Shrub and perennial grass cover also change with tree clearing events, having a higher percent cover in chain pulled regrowth ecosystems than unaltered ecosystems due to changes in canopy density (Ludwig & Tongway 2002). The deliberate removal of trees in poplar box woodlands is a common occurrence (Kaur & Stanley 2006), and has been found to result in increased grass density, herbage biomass, and twig litter biomass, along with decreases in leaf litter cover, and forb density (Tunstall & Cunningham 1990; Tunstall et al. 1981).

Other factors may also be important for vegetation stand structure. Different fire regimes have been found to alter shrub cover and height, and the cover of plants in the understorey (Fox & Fox 1986). Furthermore, fire suppression is known to cause thickening in grazed eucalypt woodlands of Queensland, primarily through the growth of surviving trees (Burrows et al. 2002).

Climatic factors may also define stand structure at a broader time scale (e.g. Fensham & Fairfax 2003b). Fensham et al. (2005) found that overstorey and understorey cover in *Acacia* forest and eucalypt woodlands in central Queensland increased over the latter half of the 20th century. Such patterns were associated with patterns in drought and rainfall, the latter half of the 20th century being relatively wet compared to the first half of the 20th century (Fensham et al. 2005).

Changes in stand structure may be directly associated with compositional changes (Dorrough et al. 2006). Dorrough et al. (2006) reported that tree cover generally has a positive influence on total and native species richness in grassy woodlands of southern Australia. The density of overstorey trees has also been found to influence overall floristic composition in woodlands of southern Queensland, by providing landscape heterogeneity and important refuges for species (Le Brocque et al. 2009). Goodhew and Le Brocque (Goodhew & Le Brocque 2006) have also found increasing mature tree density to negatively affect exotic and annual species richness, and positively affect shrub species richness in Ironbark woodlands of southern Queensland.

3.1.3 Study questions

The preceding review shows that patch factors, such as patch grazing intensity, and factors in the surrounding landscape can have significant effects on floristic composition and stand structure. However, very few studies have simultaneously examined the influence of patch and landscape factors on floristic composition (but see Lindsay & Cunningham 2009), and even fewer have examined their influence on stand structure (but see Wagner 2011).

In this chapter, floristic composition, functional group richness, richness transition ratios, and stand structure of poplar box woodlands are compared among different levels of patch grazing intensity (no, low and high levels of patch grazing) and among levels of land use context (cropping, mixed and grazed contexts). The overall patterns in floristic composition and stand structure in relation to a wide range of

patch and landscape variables are also examined to evaluate the patch and landscape hypothesis postulated in Chapter 1. The specific hypotheses examined are:

- Floristic composition, species richness, functional group richness, richness transition ratios, and stand structure differ among levels of patch grazing and land use context (*a priori*).
- Patch and landscape variables in combination explain variation in floristic composition and stand structure.

3.2 Methods

3.2.1 Field techniques

Sampling was undertaken between March and July 2011. At each site, attributes such as slope, aspect, location, landform and current land use were noted (Appendix A & C). The intensity of disturbances, such as clearing, grazing, compaction and erosion at the patch as whole, were subjectively measured on a scale of 0-3 (0 = no effect; 3 = severe effect) (Appendix A).

At each site, a 500 m² nested quadrat with nine concentrically arranged sub-quadrats (1 m², 2 m², 5 m², 10 m², 20 m², 50 m², 100 m², 200 m², 500 m²) (Figure 3.1) (after Morrison et al. 1995b) was erected in a location deemed to be subjectively representative of the entire patch. The quadrat was located at least 15 m from the patch edge to minimise edge effects. Within the quadrat, the percent cover of vegetation was estimated for 12 strata: trees (>30 m, 10-30 m, 10 m), shrubs (>2 m, <2 m), forbs and herbs, and graminoids as foliage projected cover, and cover of coarse litter, woody litter, leaf litter, bare ground, and cryptogams. These measures were then summed to give total foliage projected cover of trees (>30 m, 10-30 m, 10m), shrubs (>2 m, <2 m), understorey (herbs/forbs and graminoids), and cover of litter (coarse, woody and leaf litter). The number of living tree stems, stags (dead, standing trees) and cowpats in the 500 m² quadrat were counted directly. A direct measure of basal area was obtained using measurements of diameter (derived from

circumference) at breast height for all *E. populnea* trees in the quadrat (see Chapter 4), following the formula: $\pi(\text{DBH}/2)^2$. An alternative measure of basal area was obtained using the Bitterlich Variable Radius Method (Mueller-Dombois & Ellenberg 1974). Any other unique or noticeable characteristics were also noted, such as evidence of damage from fire and floods.

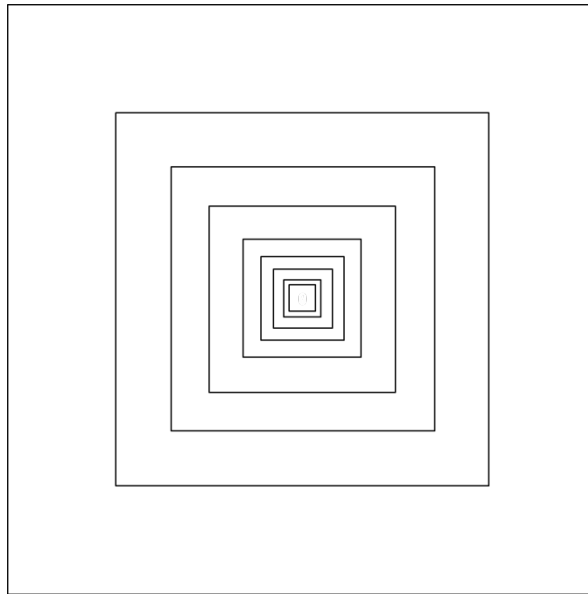


Figure 3.1 Diagrammatic representation of the 500 m² nested quadrat used in this study (after Morrison et al. 1995b).

Soil samples were taken at each site, at the 4 corner points of the 100 m² sub-quadrat. Leaf litter was removed at each point, before removing a small amount of soil to a depth of 10 cm using a clean stainless steel trowel. Sub-samples were bulked, mixed, and kept refrigerated until sent to a commercial laboratory (SGS Laboratories, Toowoomba) for analysis of pH, potassium, nitrogen, phosphorus, calcium, magnesium, organic carbon, sodium and cation exchange capacity using standard analytical techniques (Appendix A).

Species composition of vascular plants was examined using the importance score method (Morrison et al. 1995b). This method uses a nested quadrat with nine concentrically arranged sub-quadrats, where smaller sub-quadrats have a higher value (1m² being sub-quadrat 9). The sub-quadrat in which a plant first appears

denotes its importance score. Therefore, those species sighted in the first quadrat will be given the maximum score (9), with those initially encountered in progressively larger sub-quadrats will have smaller importance scores. This method is used to approximate the density of a species and has been shown to be functionally equivalent to frequency (Morrison et al. 1995b). The importance score method has been successfully used to study the effects of fire (Fynn et al. 2005; Morrison et al. 1995a), sand mining (Buckney & Morrison 1992) and clearing (Fensham et al. 2012) on vegetation. The quadrat size of 500 m² used to determine species composition was deemed efficient, with a preliminary study (Appendix B) confirming this sampling unit size adequately sampled the vegetation community.

Specimens of plants that could not be easily identified in the field were taken back to the laboratory and identified to the lowest taxonomic level possible using appropriate keys (Harden 1990-1993; Stanley & Ross 1983-1989), comparison to herbarium samples and expert assistance.

The origin (exotic or native), life history (annual, perennial, or short-lived perennial) and form (tree, shrub, herb, or graminoid) for each species was determined from the literature (Harden 1990-1993). Plant physiology (C3 or C4) was also determined for appropriate species (Bruhl & Perry 1995; Bruhl & Wilson 2007; Harden 1990-1993). Total species richness, and the richness of plants based on origin (native, exotic), life history (annuals and short-lived combined, perennial), growth form (tree, shrub, herb, graminoids), and physiology (C3 and C4) were established. A number of key functional group richness transitions were determined to examine potential shifts in response to disturbance gradients (Reardon-Smith 2011). The richness transition ratio of exotics to natives (E:N) (after Ordonez et al. 2010), and richness of short-lived plant to perennials (SL:P) (after McIntyre et al. 1999) were determined to examine whether there is any shift in response to disturbance gradients. In addition, the richness transition ratio of C4 plants to C3 plants (C4:C3) was used to examine responses to gradients in moisture or nutrients (after Wand et al. 1999; Winslow et al. 2003).

3.2.2 Spatial data

The location of each site was entered into the geographic information system, ArcMap version 10.0 (Environmental Systems Research Institute (ESRI) 2011). Regional ecosystem mapping (Department of Environment and Resource Management 2010b) was used to extract patch size (hectares) and perimeter length (km) from patches of sampled vegetation, which were also used to determine the perimeter to area ratio for each patch. A 5 km buffer was created around the centroid of each patch and the proportions of cropping and grazing land uses in the buffer region was calculated using land use mapping data (Department of Natural Resources 2003) (see Figure 2.6). A 2 km buffer was created around the centroid of each patch, and the proportion of remnant vegetation within the buffer region was calculated using regional ecosystem mapping data (Department of Environment and Resource Management 2010b). A 2 km buffer was used for this measure, as at a 5 km radius it was highly correlated with the proportions of cropping and grazing in the surrounding 5 km. The distance from each patch to the nearest remnant was also determined. Patch shape was determined using ArcMap version 10.0 (ESRI 2011), measuring the approximate length and width of each patch. Patches for which length was more than 2.5 times larger than the width were considered to be linear, and others categorised as block (after Batterham 2008).

A digital elevation model (DEM) of the Murray Darling Basin (Department of Environment and Resource Management 2001) and Queensland river data (Department of Environment and Resource Management 2010c) were used to calculate the distance from each patch to the nearest river source in ArcMap version 10.0 (ESRI 2011). The slope value for each data pixel of the DEM was calculated to create a cost raster. A cost-distance analysis was then performed using the aforementioned cost raster, and rasterised river data. The centroids of each patch were then overlaid with the cost-distance raster to extract the distance to the river, taking into account positive and negative slope.

Groundwater depths for 2009 were determined from the Queensland Groundwater Database (Department of Environment and Resource Management 2009a). Groundwater depth for 2009 was selected as it represents the recent groundwater

levels at a time when the region was in drought. Although the drought had broken the previous year, it was assumed that most components would not have fully responded by the time of sampling (Autumn-Winter 2011). Measured groundwater depths at monitoring bores for 2009 were extracted and interpolated in ArcMap version 10.0 (Figure 3.2) (ESRI 2011) (after Kath 2012). Site locations were then overlaid with the interpolated groundwater depth to give a measure of groundwater at each patch for 2009. It was acknowledged that the interpolated groundwater results may be less accurate in areas with fewer bore sites than those with more bore sites. However, this was deemed sufficient for this preliminary study. This variable was considered a landscape variable, as it is largely influenced by groundwater extraction activities in the surrounding landscape (Reardon-Smith 2011) (Appendix D), such as extraction associated with irrigated cropping.

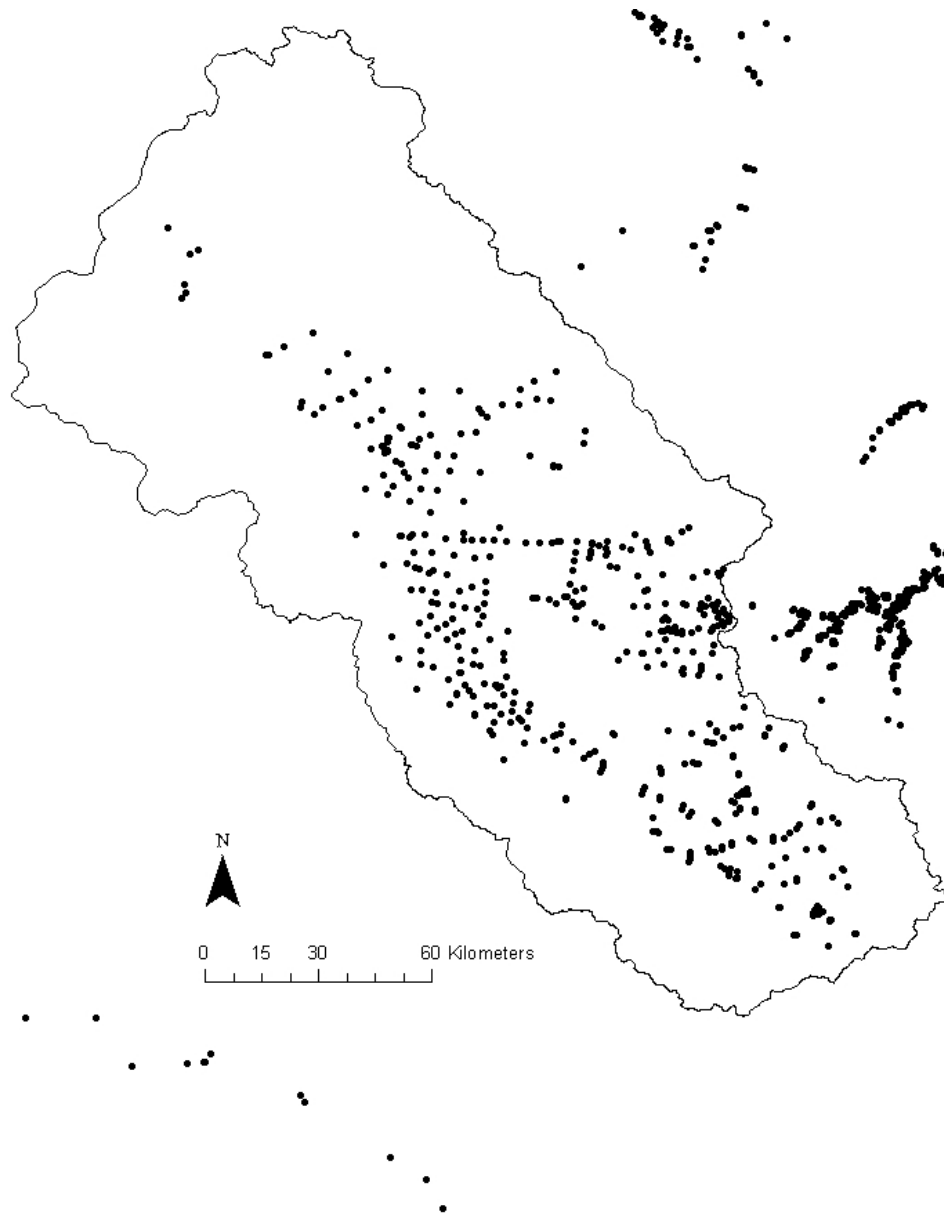


Figure 3.2 Map of bores sites used to interpolate groundwater depth (Department of Environment and Resource Management 2009a).

Mean annual rainfall data and mean maximum daily temperatures (Australian Bureau of Meteorology 2011) from 1987 to 2005 and the Data Visualisation and Analysis software FERRET version 6.02 (Hankin et al. 2007) were used to develop a gridded interpolated output of rainfall and temperature data of the study area (after Cottrill 2009). FERRET uses a linear interpolation procedure from the nearest non-missing

surrounding point (Hankin et al. 1996). This output was then converted to raster format in ArcMap (ESRI 2011), and specific site values extracted.

3.2.3 Statistical Analyses

3.2.3.1 Effects of patch grazing and land use context

Total species richness, functional group richness, richness transition ratios, and structural variables were analysed using two way analyses of variance (ANOVA) in SPSS version 19.0 (SPSS Inc. 2010) to determine whether they differed significantly ($p < 0.05$) with patch grazing intensity or land use context. All richness, transition ratio and structural variables were screened before analysis, examining normality using Shapiro-Wilk tests and homogeneity of variances using Levene's tests. All richness values (count data) were log transformed to improve normality (Bakus 2007), and all structural variables (percent data) were arcsine transformed (on proportions) to prevent boundedness effects (Quinn & Keough 2002). Some variables required the removal of outliers to attain normality and homogeneity of variances (Table 3.2). At times, samples within treatment levels were highly variable, making it necessary to exclude an entire treatment level to reach homogeneity (Table 3.2). Although homogeneity of variances was achieved for all variables, it was not always possible to reach normality. However, as the ANOVA is quite robust to violations of normality assumptions, the satisfaction of homogeneity was deemed sufficient to justify performing the ANOVA (Maxwell & Delaney 2004). Where significant differences ($p < 0.05$) were found, Tukey's post hoc tests were performed to examine pairwise comparisons (Stevens 2007).

Table 3.2 Summary of variables and excluded data in two-way analysis of variance.

Floristic composition	
Response	Excluded data
Total species richness	-
Exotic species richness	sites 16 (high patch grazing; grazing) and 20 (no patch grazing; grazing)
Native species richness	-
Short-lived species richness	-
Perennial species richness	-
C3 species richness	-
C4 species richness	-
Tree species richness	site 31 (low patch grazing; grazing)
Shrub species richness	-
Herb species richness	sites with low levels of patch grazing
Graminoid species richness	-
Exotic:Native	-
Shortlived:Perennial	-
C4:C3	site 19 (no patch grazing; grazing)
Stand structure	
Response	Excluded data
Foliage projected cover of trees 10-30m	sites with low levels of patch grazing
Foliage projected cover of trees <10m	-
Foliage projected cover of shrubs >2m	sites with low levels of patch grazing
Foliage projected cover of shrubs <2m	site 32 (low patch grazing; grazing)
Foliage projected cover of herbs	site 31 (low patch grazing; grazing)
Foliage projected cover of graminoids	site 11 (high patch grazing; mixed)
Cover of coarse litter	-
Cover of woody litter	-
Cover of leaf litter	sites 15,19 (no patch grazing; grazing) and 24low patch grazing; mixed)
Cover of cryptogams	sites with mixed land use context
Foliage projected cover of all trees	-
Foliage projected cover of all shrubs	-
Foliage projected cover of understorey	-
Cover of all litter	-

3.2.3.2. Multivariate analyses

Non-metric multidimensional scaling (nMDS) ordination (Clarke & Warwick 2000) with Bray-Curtis similarity of species composition data (importance scores) and stand structure were conducted using Primer version 5.2.9 (PRIMER-E Ltd 2000) to examine broad multivariate patterns in these datasets. Analyses of similarity (ANOSIM) (Clarke & Warwick 2000) were used to determine whether samples were significantly different ($p < 0.05$) based on their floristic composition or structural characteristics. SIMPER analysis (Clarke & Warwick 2000) was used to determine

species contributions to similarity and to identify those species which provide primary distinction between two categories.

Correspondence analyses (CA) (Canoco 4.5, Ter Braak & Smilauer 2002) of floristic composition and stand structure were undertaken to identify vegetation gradients and compare these to gradients constrained by environmental variables (see canonical correspondence analysis below). Prior to analysis, structure data were arcsine transformed to prevent bounded effects (Quinn & Keough 2002) and rare species (those which were only sighted once) were removed from the species composition dataset to reduced data noise (after Attwood 2009). Canonical correspondence analyses (CCA) (Canoco 4.5, Ter Braak & Smilauer 2002) of floristic composition and stand structure were used to determine what environmental (explanatory) variables may be driving broad patterns. This was performed using 29 and 23 environmental variables for composition and structure, respectively (Table 3.3). CCA, an extension of CA, is a multivariate direct gradient technique in which ordination axes were constrained to be linear combinations of environmental variables (Ter Braak 1986).

Environmental variables were chosen by examining collinearity diagnostics and Pearson's correlations (Appendix D). The final set of environmental variables was obtained in a stepwise fashion of removal based on levels of tolerance and the variance inflation factor (VIF). Those variables with the lowest tolerance and the highest VIF were removed until all variables included had tolerance and VIF values within an acceptable range (tolerance >0.1 and VIF <10) (Tuffery 2011). Grazing intensity (3 levels) was treated as a supplementary (passive) variable in the analyses, and as such, did not contribute to the constraining of axes (Ter Braak & Smilauer 2002) (Table 3.3). Stag density and the Bitterlich measure of basal area were also treated as supplementary variables. The density of stags was potentially more representative of a response rather than a driver of patterns, so it was not used to constrain axes. In addition, measures of basal area obtained using the Bitterlich method did not correlate with direct measures of stem density or basal area, so this measure was used with caution and was not used to constrain axes. As the proportions of cropping and grazing in the surrounding landscape were so strongly negatively correlated (Appendix D), trends in the proportion of cropping in the

surrounding landscape also inversely relate to proportions of grazing in the surrounding landscape.

Table 3.3 Environmental variables used in CCAs of floristic composition and stand structure.

Variable	Label
Patch shape	shape
Patch size (ha)	size
Proportion remnant vegetation in (2 km radius)	PRV2
Stem density (indiv./ha)	stems
Basal area (cm ² /ha)	basal
Foliage index (%)	FI
Clearing	clear
Erosion	erosion
Compaction	compact
Cowpat density (/500m ²)	cow
Soil pH	pH
Soil Nitrogen concentration (mg/kg)	N
Soil Phosphorus concentration (mg/kg)	P
Soil Calcium concentration (mg/kg)	Ca
Cropping land use in 5 km radius (%)	cropping
Average annual rainfall (mm)	rain
Distance to River (m)	river
Distance to nearest remnant (m)	rem
Bare ground (%)*	bare
Cryptogams (%)*	crypt
Foliage projected cover of trees (%)*	trees
Foliage projected cover of shrubs (%)*	shrubs
Foliage projected cover of understorey (%)*	under
Cover of litter (%)*	litter
No patch grazing [^]	no grazing
Low patch grazing [^]	low grazing
High patch grazing [^]	high grazing
Bitterlich [^]	bitter
Stags [^]	stags

* Variable only used in CCA of floristic composition

[^]Supplementary variables

3.3 Results

3.3.1 Patterns in floristic composition and species richness

3.3.1.1 General results

A total of 289 vascular plant species were sampled and identified across 33 sites. In total, there were 220 native species, 51 exotic species (18 of unknown origin), 204 perennial species, and 68 short-lived or annual species (17 of unknown perennality) (Appendix E1). There were also 13 tree species, 29 shrub species, 90 graminoid species, 153 herb species and 1 vine throughout the study area (Appendix E1).

Total species richness ranged from 27 to 66 per 500 m², with a mean of 50.5 per 500 m² (Appendix E2). Native plant richness ranged from 24 to 60 per 500 m², while the richness of exotic plants ranged from 1 to 16 per 500 m². The perennial plant richness had a range of 19 to 57 per 500 m², and short-lived plant (including annuals) ranged from 1 to 17 per 500 m² (Appendix E2). The richness of herbs, graminoids, shrubs and trees ranged from 9 to 36, 11 to 28, 1 to 9, and 1 to 6 per 500 m² respectively (Appendix E2). The C3 plant species richness had a range of 13 to 40 per 500 m², and the C4 plant species richness ranged from 9 to 31 per 500 m².

3.3.1.2 Effects of patch grazing and land use context

Native richness, C4 plant richness and the ratio of C4:C3 plants differed significantly among levels of patch grazing (Table 3.4). Native richness was significantly lower at low levels of patch grazing than at high levels of patch grazing, ungrazed being intermediate (Figure 3.3a). C4 plant richness and the richness transition ratio of C4 to C3 plants were significantly lower at low levels of patch grazing than at ungrazed sites, with high levels of patch grazing being intermediate (Figure 3.3b and c). Exotic richness was significantly different among levels of land use context (Table 3.4). Exotic richness was significantly lower in cropping landscapes, than in mixed or grazing landscapes (Figure 3.4).

There was a significant interaction between patch grazing and land use context for the ratio of C4:C3 plant richness (Table 3.4). The richness transition ratio of C4:C3 plants was lower under low levels of patch grazing than when grazing was absent or of high intensity, but only in grazing landscapes (Figure 3.5a). In grazing landscapes the C4:C3 richness ratio was higher at ungrazed patches, and lower under low levels of patch grazing, but did not differ from other levels of land use context at high levels of patch grazing (Figure 3.5b).

Table 3.4 Summary of two-way analyses of variance of functional group species richness (SR) and richness transition ratios, including means with standard errors in parentheses. Values in **bold** are significant at the 0.05 level. Superscript indicates results of Tukey's post hoc tests.

Response variable	Factor	Degrees of Freedom	Test Statistic (F)	Interaction Test Statistic (F)	Mean (S.E.)					
					No Grazing	Low Grazing	High Grazing	Cropping	Mixed	Grazing
Total SR	Patch grazing	2,24	1.191	0.697	51.00 (2.61)	45.20 (1.90)	54.17 (4.65)	48.42 (3.01)	50.44 (2.76)	50.75 (3.24)
	Land use context	2,24	0.020							
Exotic SR	Patch grazing	2,22	0.022	1.697	9.38 (0.78)	9.40 (1.16)	9.80 (0.97)	7.42 ^a (0.76)	11.33 ^b (0.78)	10.20 ^b (0.93)
	Land use context	2,22	3.889							
Native SR	Patch grazing	2,24	3.495	0.484	41.71 ^{ab} (2.29)	35.20 ^a (1.84)	47.00 ^b (3.12)	40.17 (2.46)	39.89 (2.22)	41.83 (3.25)
	Land use context	2,24	0.092							
Short-lived SR	Patch grazing	2,24	1.756	1.711	9.12 (0.89)	6.9 (0.98)	10.50 (1.82)	7.17 (1.21)	10.11 (1.05)	9.17 (1.01)
	Land use context	2,24	1.649							
Perennial SR	Patch grazing	2,24	0.658	0.623	41.82 (2.32)	37.80 (2.17)	42.83 (3.24)	40.50 (1.99)	40.11 (2.21)	41.58 (3.31)
	Land use context	2,24	0.196							
C3 SR	Patch grazing	2,24	0.932	1.274	27.06 (1.42)	26.80 (1.36)	31.00 (2.35)	27.42 (1.76)	28.67 (1.11)	27.25 (1.85)
	Land use context	2,24	0.114							

Response variable	Factor	Degrees of Freedom	Test Statistic (F)	Interaction Test Statistic (F)	Mean (S.E.)					
					No Grazing	Low Grazing	High Grazing	Cropping	Mixed	Grazing
C4 SR	Patch grazing	2,24	3.516	1.705	19.82 ^a (1.49)	14.40 ^b (1.06)	19.67 ^{ab} (2.23)			
	Land use context	2,24	0.197					16.83 (1.43)	18.78 (1.46)	19.00 (2.12)
Tree SR	Patch grazing	2,23	3.288	1.797	1.71 (0.27)	1.11 (0.11)	1.83 (0.31)			
	Land use context	2,23	3.957					1.25 ^a (0.13)	1.11 ^a (0.11)	2.27 ^b (0.36)
Shrub SR	Patch grazing	2,24	1.643	1.411	1.71 (0.27)	1.60 (0.50)	1.83 (0.31)			
	Land use context	2,24	0.938					4.00 (0.41)	3.89 (0.59)	4.92 (0.68)
Herb SR	Patch grazing	1,17	0.579	0.334	25.12 (1.77)	-	28.00 (2.34)			
	Land use context	2,17	0.364					26.50 (2.44)	24.67 (2.83)	26.83 (2.01)
Graminoid SR	Patch grazing	2,24	1.780	1.661	20.94 (1.21)	17.40 (1.25)	19.83 (2.02)			
	Land use context	2,24	0.176					18.92 (1.12)	20.33 (1.65)	19.92 (1.68)
Exotic: Native	Patch grazing	2,24	2.861	1.685	0.21 (0.02)	0.28 (0.04)	0.18 (0.03)			
	Land use context	2,24	2.554					0.19 (0.01)	0.29 (0.03)	0.22 (0.04)
Short-lived: Perennial	Patch grazing	2,24	0.334	0.972	0.28 (0.03)	0.23 (0.04)	0.29 (0.04)			
	Land use context	2,24	1.303					0.21 (0.03)	0.30 (0.03)	0.30 (0.04)
C4:C3	Patch grazing	2,23	3.830	3.073	0.71 ^a (0.04)	0.55 ^b (0.05)	0.63 ^{ab} (0.06)			
	Land use context	2,23	0.065					0.61 (0.03)	0.67 (0.08)	0.73 (0.09)

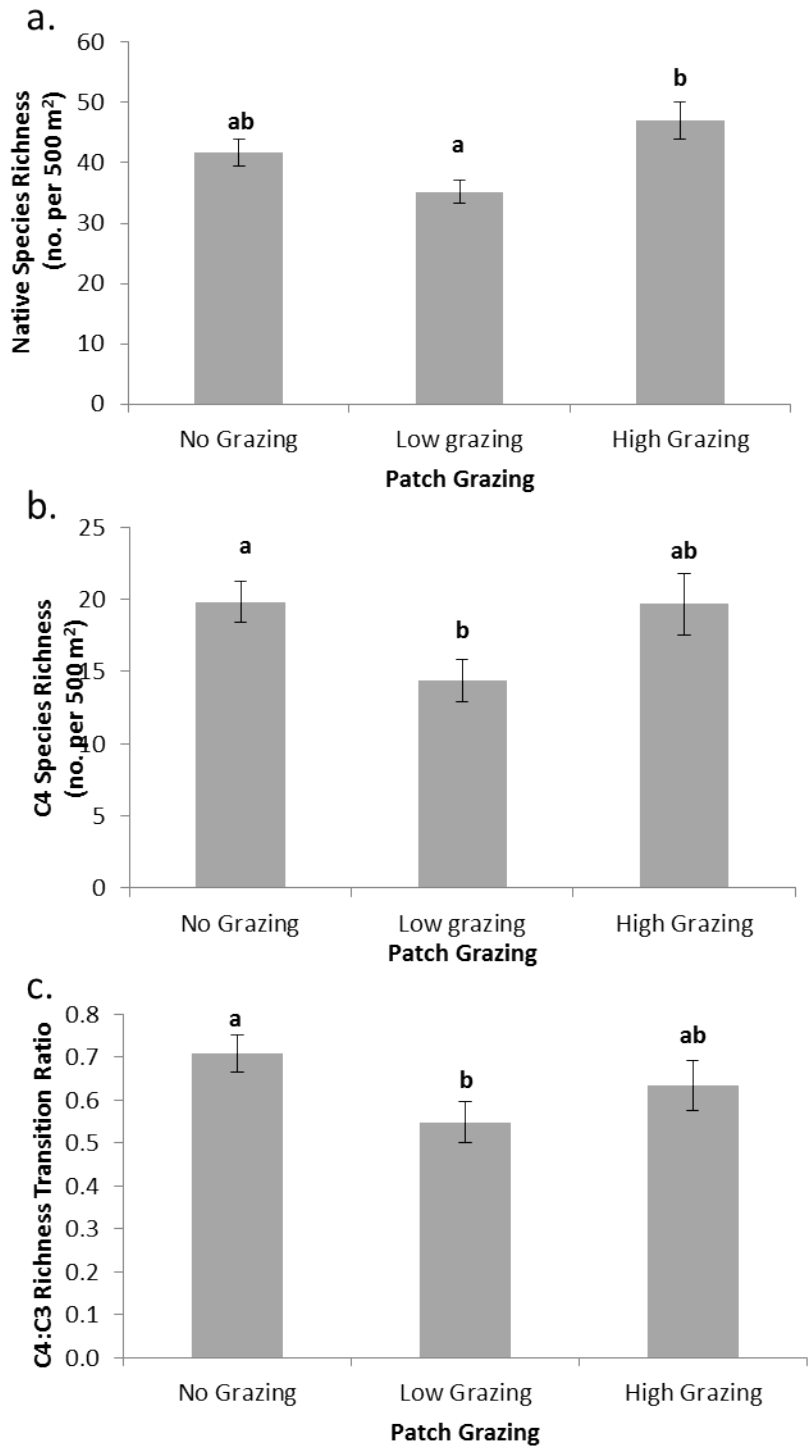


Figure 3.3 Mean richness of (a) native, (b) C4 plants; and (c) the mean ratio of C4:C3 by patch grazing intensity. Error bars are standard error. Means sharing the same letter are not significantly different (Tukey's post hoc tests, $p > 0.05$).

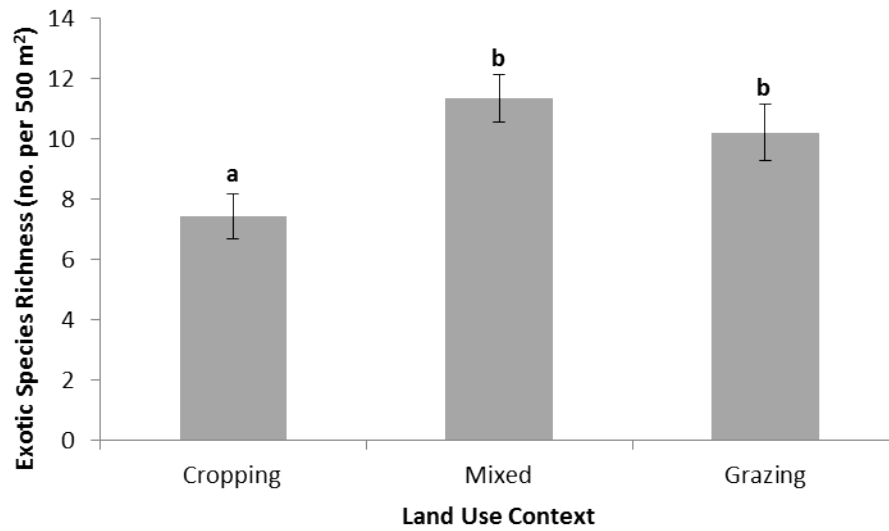


Figure 3.4 Mean richness of exotic plants by land use context. Error bars are standard error. Means sharing the same letter are not significantly different (Tukey's post hoc tests, $p > 0.05$).

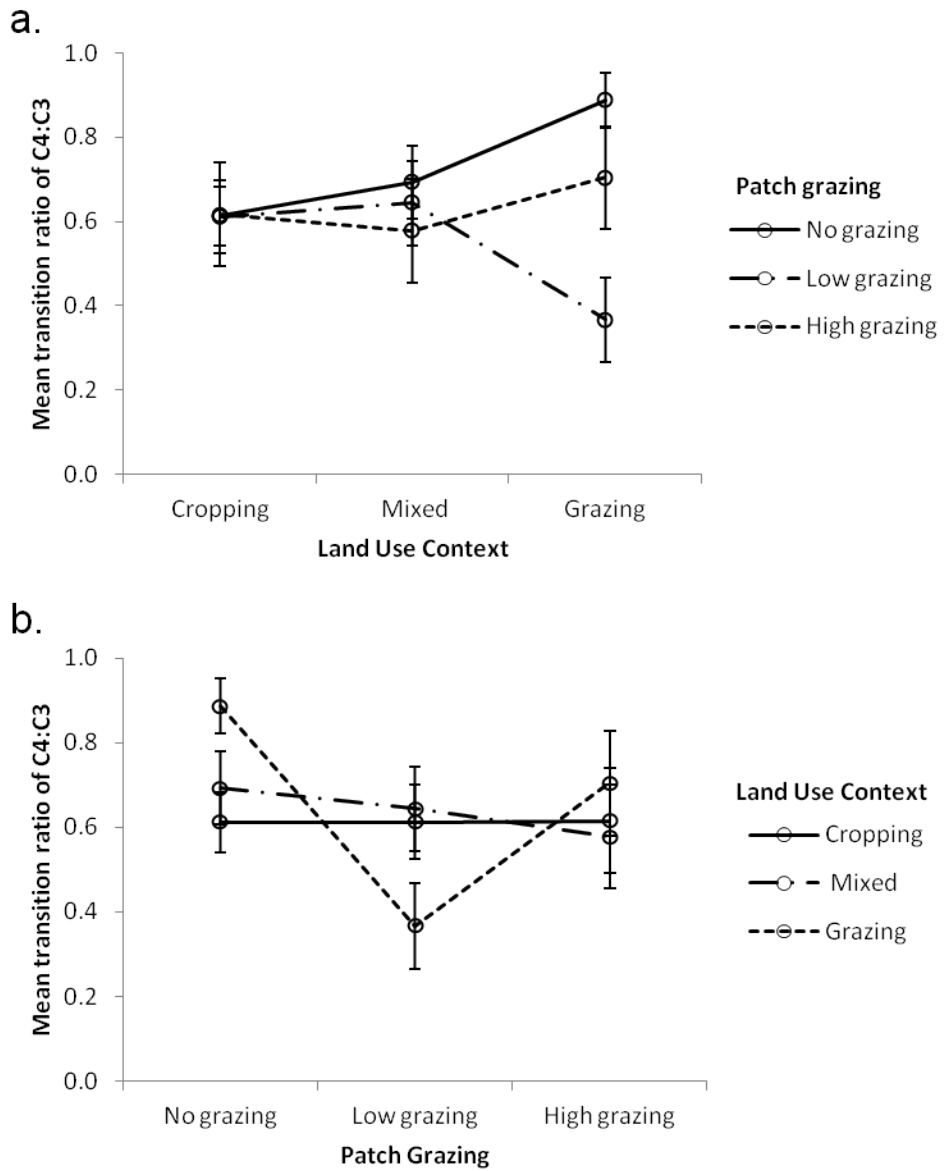


Figure 3.5 Interaction plots of the ratio between C4 and C3 plants: a) land use context by patch grazing and b) patch grazing by land use context. Error bars are standard error.

The non-metric multidimensional scaling (nMDS) ordination of floristic composition (Figure 3.6) shows that sites in cropping landscapes were very similar floristically, while sites in grazing landscapes were highly variable. Sites of mixed land use context were scattered among sites in cropping landscapes and sites in grazing landscapes. Of the two sites in grazing landscapes under high levels of patch grazing (sites 16 and 33), site 33 was more floristically similar to sites in cropping landscapes (Figure 3.6). Site 33 was considered to be an outlier due to higher proportions of cropping in the surrounding landscape and higher levels of soil calcium (in comparison to site 16) which were characteristic of other sites in cropping landscapes (Appendix H and I). Sites in grazing landscapes showed some floristic differences between ungrazed sites and sites with low grazing intensity. No other patterns were evident among the different levels of grazing.

The analysis of similarity (Table 3.5) shows a significant difference in floristic composition among levels of land use context, but not among levels of patch grazing intensity. The difference in composition due to land use context was significant between sites in cropping landscapes and sites in grazing landscapes, with sites in the mixed landscapes being intermediate between the two (Figure 3.6). No patterns among levels of patch grazing were statistically significant.

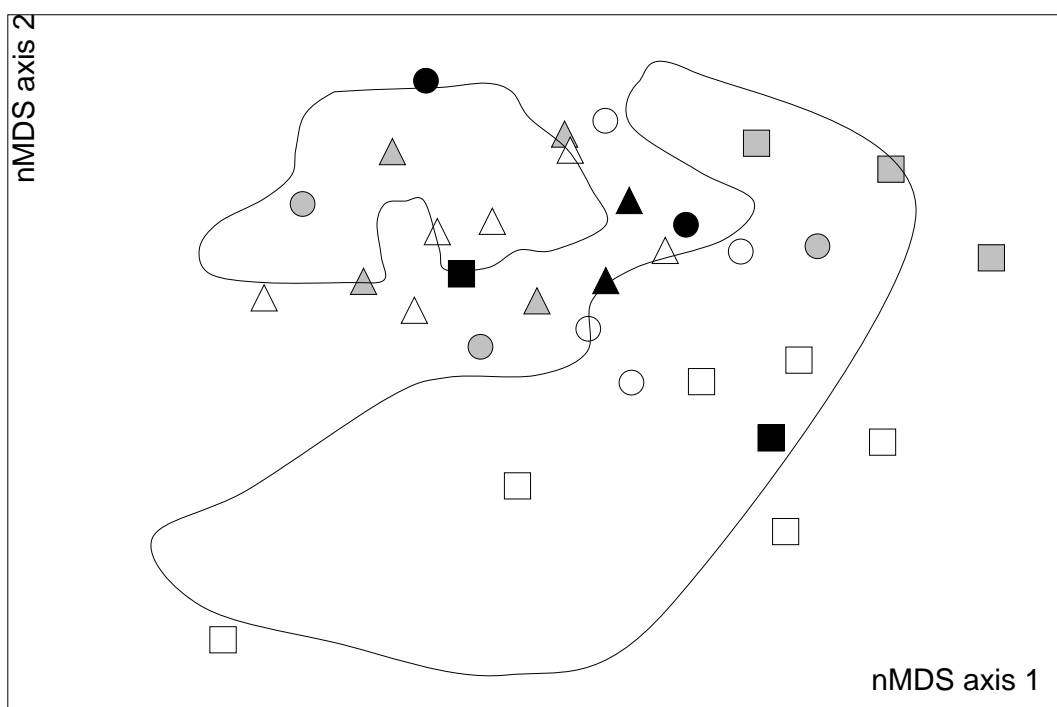


Figure 3.6 nMDS ordination of floristic composition. Enclosures show two significantly ($p < 0.05$) different groups (cropping land use context and grazing land use context). Key: \triangle = No grazing, Cropping; \triangle = Low grazing, Cropping; \blacktriangle = High grazing, Cropping; \circ = No grazing, Mixed; \circ = Low grazing, Mixed; \bullet = High grazing, Mixed; \square = No grazing, Grazing; \square = Low grazing, Grazing; \blacksquare = High grazing, Grazing.

Table 3.5 Two-way crossed ANOSIM of floristic composition by patch grazing and land use context. Values in **bold** are significant at the 0.05 level.

Groups	R statistic
<i>Patch grazing</i>	Global = 0.111
No grazing & Low grazing	0.098
No grazing & High grazing	0.118
Low grazing & High grazing	0.128
<i>Land use context</i>	Global = 0.249
Cropping & Mixed	0.155
Cropping & Grazing	0.477
Mixed & Grazing	0.026

Simper analysis (Table 3.6) shows that sixteen species contributed to 20% of the dissimilarity between sites in cropping landscapes and grazing landscapes. Native

perennials such as *Neptunia gracilis*, *Eriochloa crebra*, *Dichanthium sericeum*, *Carex inversa*, and *Leptochloa divaricatissima* were common in sites in cropping landscapes and relatively uncommon in sites in grazing landscapes. Native perennial grasses such as *Paspalidium caespitosum* and *Chloris ventricosa* were more abundant at sites in grazing landscapes (Table 3.6).

Table 3.6 Average dissimilarity between cropping and grazing landscapes with average abundance of plant species which contributed up to 20% of dissimilarity.

Species	Average Abundance		Cumulative %
	Cropping	Grazing	
Average dissimilarity: 75.91			
<i>Neptunia gracilis</i>	6.33	0.83	1.61
<i>Eriochloa crebra</i>	8.17	3.25	3.18
<i>Dichanthium sericeum</i>	8.58	3.92	4.74
<i>Carex inversa</i>	5.42	2.17	6.21
<i>Leptochloa divaricatissima</i>	5.50	1.42	7.67
<i>Alternanthera denticulata</i>	4.42	1.75	9.04
<i>Paspalidium caespitosum</i>	4.58	6.00	10.40
<i>Chloris ventricosa</i>	3.08	6.33	11.63
<i>Enteropogon acicularis</i>	4.33	0.75	12.80
<i>Einadia nutans var nutans</i>	4.25	2.25	13.97
<i>Glandularia aristigera</i>	3.92	6.17	15.11
<i>Glycine tabacina</i>	6.83	5.92	16.24
<i>Oxalis perennans</i>	4.17	1.83	17.36
<i>Phyllanthus virgatus</i>	4.25	1.08	18.48
<i>Asperula conferta</i>	3.08	3.75	19.56
<i>Bothriochloa bladhii</i>	3.75	1.00	20.65

3.3.1.3 Multivariate relationships between floristic composition and environmental variables

Correspondence analysis (CA) of floristic composition (Figure 3.7a) explained 16.1% of variation in the first two axes and showed very similar patterns to the nMDS ordination (Figure 3.6). There was a clear separation between sites with predominantly cropping or grazing in the surrounding landscape, with sites in mixed landscapes scattered among these two groups (Figure 3.7a).

Canonical correspondence analysis (CCA) of floristic composition constrained by spatial and environmental variables explained 29.6% of the variance in the two-dimensional representation (Figure 3.7b). Overall, the pattern in the constrained ordination (CCA) was the same as the un-constrained ordination (CA) (Figure 3.7a). The separation based on land use context (between sites in cropping and grazing landscapes) was best explained by the proportion of cropping in the surrounding landscape, soil calcium concentration, and to a lesser extent by average rainfall, patch shape, litter and the proportion of remnant vegetation in the surrounding 2 km (Figure 3.7b). The proportion of cropping in the surrounding landscape and soil calcium concentration were higher at sites in cropping landscapes, and rainfall, patch shape, litter and the proportion of remnant vegetation were higher at sites in grazing landscapes.

The variability within the grazing land use context was best explained by shrub cover, soil pH, distance to a river, soil phosphorus concentration, understorey cover and, to a lesser extent, patch grazing intensity (cowpat density) (Figure 3.7b). The passive grazing variables (no, low and high patch grazing) and basal area (Bitterlich) were correlated with this complex environmental gradient.

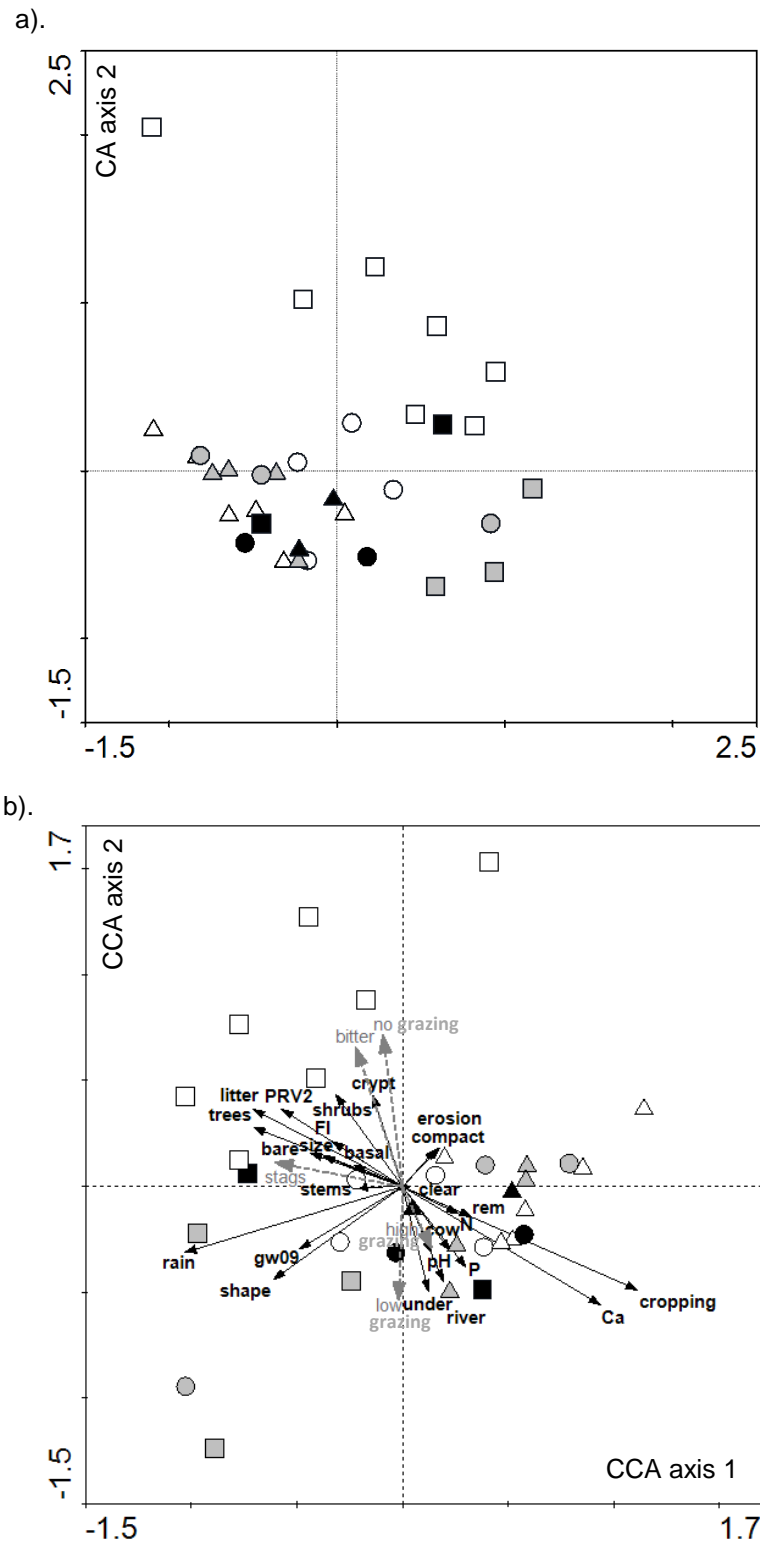


Figure 3.7 a) CA ordination of floristic composition (minus rare species) and b) CCA of floristic composition (minus rare species) environmental variables represented as vectors (Eigen values x-axis = 0.389, y-axis = 0.654; % variance of species-environment relation = 29.6). Grey arrows and text indicate supplementary variables. Key: \triangle = No grazing, Cropping; \blacktriangle = Low grazing, Cropping; \blacktriangle = High grazing, Cropping; \circ = No grazing, Mixed; \bullet = Low grazing, Mixed; \bullet = High grazing, Mixed; \square = No grazing, Grazing; \blacksquare = Low grazing, Grazing; \blacksquare = High grazing, Grazing.

3.3.2 Stand structure

3.3.2.1 General results

There were no trees over 30 m in height across sites sampled, while trees 10-30m and trees <10m had a mean foliage projected cover ranging between 0% to 35% and 1% to 35 % respectively (Appendix F). The total foliage projected cover of shrubs >2m and shrubs <2m both ranged from 0% to 8%. The most prominent layer in these systems was the understorey, where the cover of graminoids ranged from 40% to 95%, and the cover of herbs ranged from 2% to 20%. Leaf litter covered between 1% to 35% of the quadrat, while the cover of coarse and woody litter ranged from 0% to 10%, and 1% to 12% respectively. Cryptogams were scarce, which covered from 0% to 3% of the quadrat (Appendix F).

3.3.2.2 Effects of patch grazing and land use context

Graminoid cover was significantly different among levels of land use context (Table 3.7); the cover of graminoids was significantly lower in grazing landscapes than in mixed landscapes, with cropping landscapes being an intermediate (Figure 3.8). There were no significant differences ($p > 0.05$) among the levels of patch grazing for any of the structural variables (Table 3.7).

There was a significant interaction between patch grazing and land use context for total tree cover (Table 3.7). Total tree cover was lower at low levels of patch grazing but only in mixed landscapes, while tree cover was lower at high levels of patch grazing, but only in grazing landscapes (Figure 3.9a). Total tree cover was higher in grazing landscapes but only at low levels of patch grazing, while it was lower in cropping landscapes but only at ungrazed sites (Figure 3.9b).

Table 3.7 Summary of two-way analyses of variance of stand structural variables, including means with standard errors in parentheses. Values in **bold** are significant at the 0.05 level. Superscript indicates results of Tukey's post hoc tests.

(Foliage Projected) Cover (%)	Factor	Degrees of Freedom	Test Statistic (F)	Interaction Test Statistic (F)	Mean (S.E.)					
					No Grazing	Low Grazing	High Grazing	Cropping	Mixed	Grazing
Trees 10-30 m	Patch grazing	1, 17	0.002	1.179	15.18 (2.54)	-	15.00 (4.93)			
	Land use context	2, 17	0.105					12.50 (3.27)	17.17 (0.67)	16.11 (4.08)
Trees <10 m	Patch grazing	2, 24	0.189	0.387	13.76 (2.65)	12.10 (3.48)	10.67 (3.87)			
	Land use context	2, 24	0.341					10.42 (2.48)	13.00 (3.86)	14.75 (3.38)
Shrubs >2 m	Patch grazing	1, 17	0.282	0.144	1.24 (0.53)	-	0.33 (0.21)			
	Land use context	2, 17	0.984					0.25 (0.16)	0.67 (0.67)	1.89 (0.86)
Shrubs <2 m	Patch grazing	2, 24	1.357	0.394	1.00 (0.39)	1.22 (0.40)	1.00 (0.26)			
	Land use context	2, 24	1.067					1.08 (0.40)	0.89 (0.45)	1.18 (0.42)
Herbs	Patch grazing	2, 24	0.783	0.948	5.35 (0.95)	5.78 (1.38)	4.17 (1.25)			
	Land use context	2, 24	0.532					4.92 (1.12)	6.44 (1.21)	4.64 (1.15)
Graminoids	Patch grazing	2, 23	0.213	0.960	76.47 (2.77)	80.00 (2.58)	73.00 (6.04)			
	Land use context	2, 23	3.479					80.00 ^{ab} (2.89)	82.50 ^a (2.99) ^a	70.42 ^b (2.92)
Coarse litter	Patch grazing	2, 24	1.719	0.653	3.29 (0.55)	2.00 (0.42)	3.33 (0.61)			
	Land use context	2, 24	2.928					2.00 (0.33)	2.33 (0.47)	4.25 (0.66)

(Foliage Projected) Cover (%)	Factor	Degrees of Freedom	Test Statistic (F)	Interaction Test Statistic (F)	Mean (S.E.)					
					No Grazing	Low Grazing	High Grazing	Cropping	Mixed	Grazing
Woody litter	Patch grazing	2, 24	0.383	0.786	2.88 (0.62)	2.30 (0.33)	3.00 (0.45)			
	Land use context	2, 24	1.254					3.33 (0.84)	1.89 (0.20)	2.75 (0.35)
Leaf litter	Patch grazing	2, 21	0.793	0.593	8.33 (1.60)	5.89 (1.09)	9.17 (1.33)			
	Land use context	2, 21	0.277					7.50 (1.48)	8.00 (1.16)	7.90 (2.00)
Cryptogams	Patch grazing	2, 15	0.085	0.517	0.77 (0.20)	0.43 (0.43)	0.25 (0.25)			
	Land use context	1, 15	0.004					0.58 (0.26)	-	0.58 (0.23)
All trees	Patch grazing	2, 24	1.603	3.049	28.94 (2.20)	23.60 (3.56)	25.67 (3.77)			
	Land use context	2, 24	2.938					21.17 (2.20)	26.44 (3.58)	32.50 (2.37)
All Shrubs	Patch grazing	2, 24	0.099	0.094	2.24 (0.82)	1.90 (0.77)	1.33 (0.33)			
	Land use context	2, 24	1.401					1.25 (0.39)	1.33 (0.73)	3.17 (1.10)
Understorey	Patch grazing	2, 24	2.296	0.602	81.71 (3.08)	87.20 (1.91)	71.67 (6.85)			
	Land use context	2, 24	0.946					84.92 (3.10)	84.00 (4.99)	76.33 (3.53)
All litter	Patch grazing	2, 24	0.461	1.713	17.35 (2.73)	12.60 (2.45)	15.50 (1.48)			
	Land use context	2, 24	0.302					12.83 (2.08)	14.67 (2.43)	19.00 (3.42)

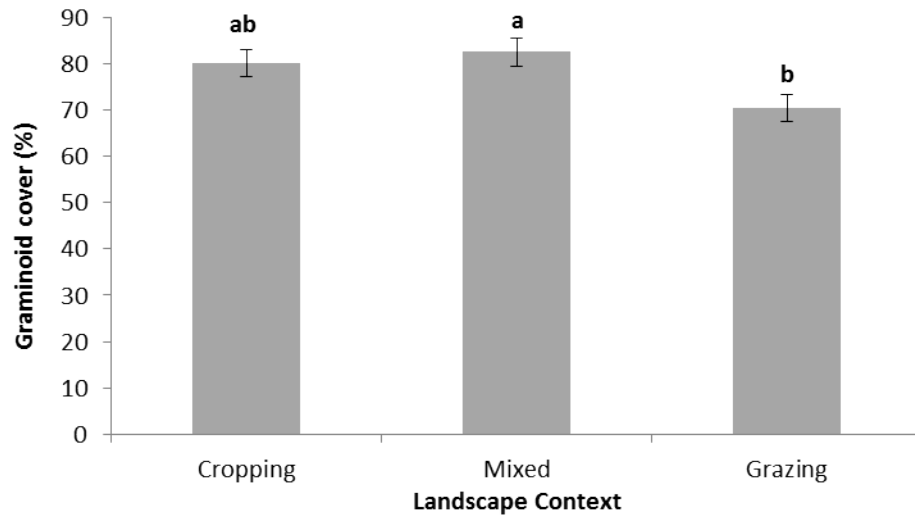


Figure 3.8 Mean cover of graminoids by land use context. Error bars are standard error. Means sharing the same letter are not significantly different (Tukey's post hoc tests, $p > 0.05$).

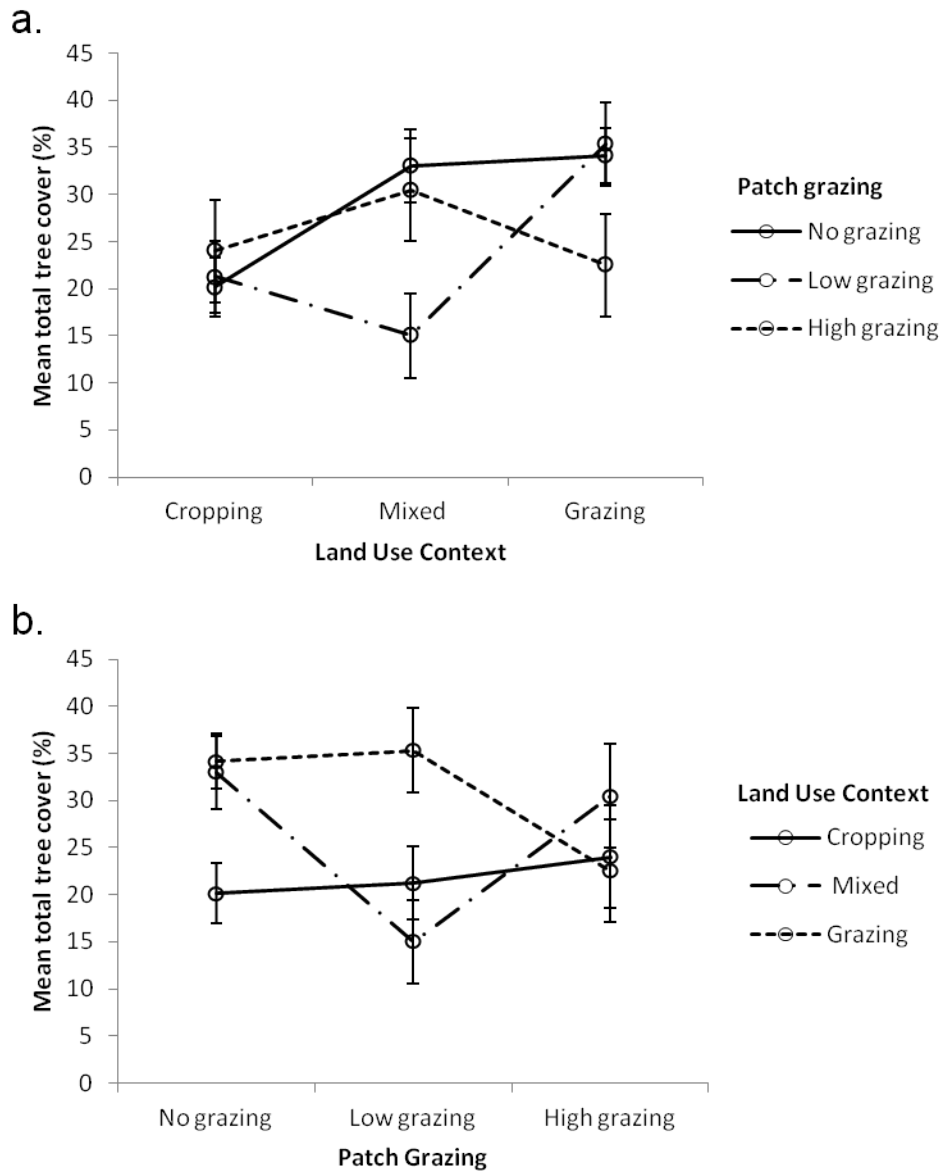


Figure 3.9 Interaction plot of total tree foliage projected cover (%) by: a) land use context by patch grazing and b) patch grazing by land use context. Error bars are standard error.

The non-metric multidimensional scaling (nMDS) ordination of stand structure (Figure 3.10) revealed no distinct pattern among levels of patch grazing or land use context. Analysis of similarity (Table 3.8) indicated that multivariate structural characteristics were not significantly different among levels of patch grazing or land use context.

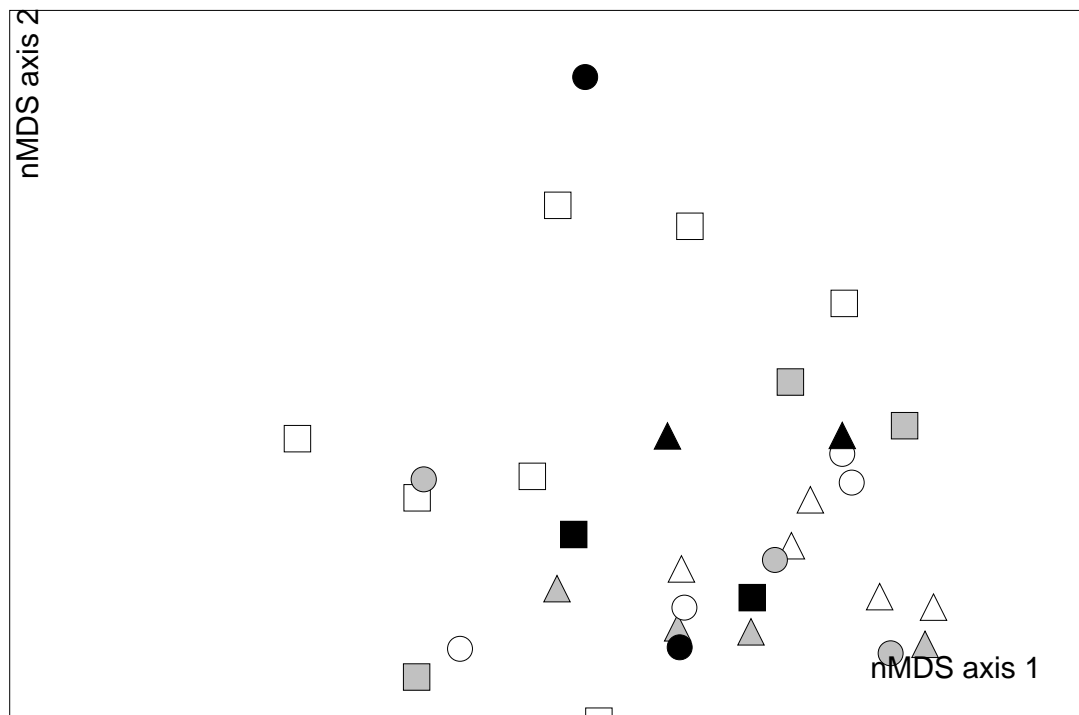


Figure 3.10 nMDS ordination of stand structure.

Key: \triangle = No grazing, Cropping; \triangle = Low grazing, Cropping; \blacktriangle = High grazing, Cropping; \circ = No grazing, Mixed; \circ = Low grazing, Mixed; \bullet = High grazing, Mixed; \square = No grazing, Grazing; \square = Low grazing, Grazing; \blacksquare = High grazing, Grazing.

Table 3.8 Two-way crossed ANOSIM of stand structure by patch grazing and land use context.

Groups	R statistic
<i>Patch grazing</i>	Global = -0.05
No grazing & Low grazing	0.027
No grazing & High grazing	0.117
Low grazing & High grazing	-0.225
<i>Land use context</i>	Global = 0.095
Cropping & Mixed	-0.071
Cropping & Grazing	0.216
Mixed & Grazing	0.066

3.3.2.3 Multivariate relationships between stand structure and environmental variables

Correspondence analysis (CA) of stand structure (Figure 3.11a) explained 66.4% of variance in the first two axes. Figure 3.11a showed considerable overlap between all types of patch grazing and land use context, with no noticeable pattern in relation to the *a priori* design.

Canonical correspondence analysis (CCA) of stand structure constrained by patch and landscape variables explained 43.4% of variance in two-dimensions (Figure 3.11b). Overall, the pattern in the constrained ordination (CCA) (Figure 3.11b) was similar to the un-constrained ordination (CA) (Figure 3.11a). No discernible pattern in relation to patch grazing and land use context was evident, with patch and landscape vectors spread across the ordination. However, it showed that patterns in stand structure (separate to the *a priori* design) were largely driven by the proportions or cropping (and grazing) in the surrounding landscape, the proportion of remnant vegetation in the surrounding landscape, and various patch factors such as soil calcium and nitrogen, stem density, and patch shape.

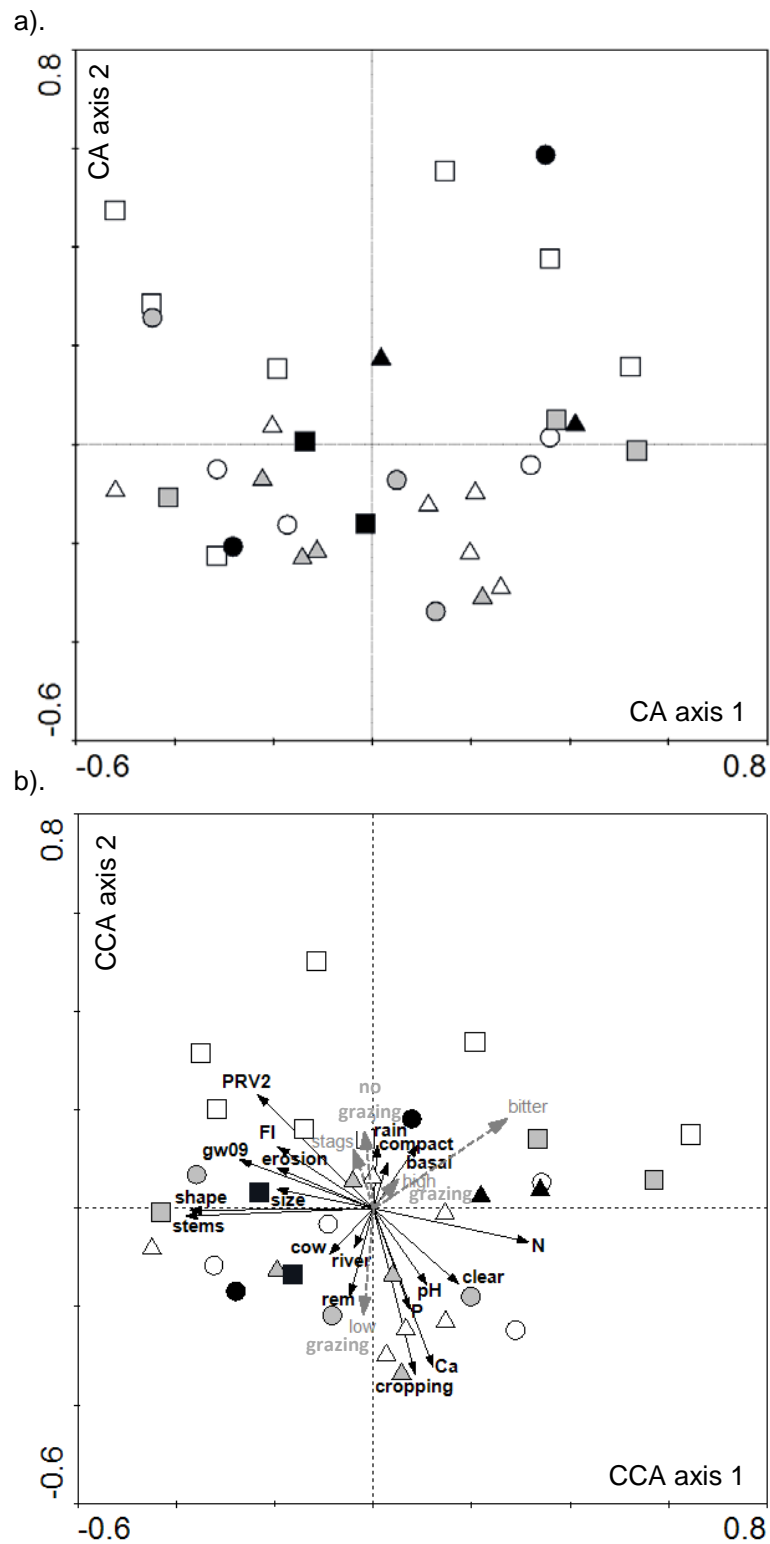


Figure 3.11 a) CA ordination of stand structure and b) CCA of stand structure with environmental variables represented as vectors (Eigen values x-axis = 0.362, y-axis = 0.448; % variance of species-environment relation = 66.4). Grey arrows and text indicate supplementary variables. Key: \triangle = No grazing, Cropping; \triangle = Low grazing, Cropping; \blacktriangle = High grazing, Cropping; \circ = No grazing, Mixed; \bullet = High grazing, Mixed; \circ = Low grazing, Mixed; \bullet = High grazing, Mixed; \square = No grazing, Grazing; \square = Low grazing, Grazing; \blacksquare = High grazing, Grazing.

3.4 Discussion

3.4.1 Effects of patch grazing and land use context (*a priori* design)

Analysis using the *a priori* design demonstrated that the dominant land uses in the surrounding landscape have a significant influence on the floristic composition of poplar box woodlands. Specifically, overall floristic composition differed distinctly between cropping and grazing landscapes, with mixed landscapes an intermediate. The vegetation in the surrounding landscape can be very important for biodiversity of remnants, the composition and structure of the vegetation matrix having a large influence on the composition and structure of remnants which it surrounds (Wiser & Buxton 2008). Land use in the surrounding landscape has been shown to affect plant species richness in semi natural grassland in south-eastern Sweden (Öckinger et al. 2011), and overall species composition in *Eucalyptus melliodora* and *E. albens* woodlands of New South Wales (Lindsay & Cunningham 2009).

The difference in composition between sites in cropping and grazing landscapes was contributed by significant differences in the abundance of certain species. A number of native perennials, such as *Neptunia gracilis*, *Eriochloa crebra*, *Dichanthium sericeum*, *Carex inversa*, and *Leptochloa divaricatissima* were much more abundant in cropping landscapes than in grazing landscapes (Table 3.8). Trends in abundance between cropping and grazing landscapes may be explained by their tolerance to grazing. For example, *Eriochloa crebra*, *Einadia nutans* var. *nutans*, and *Oxalis perennans*, which are more abundant in cropping landscapes, are known to respond negatively to grazing pressure (Dorrough et al. 2011; McIntyre et al. 2003).

Exotic plant richness also differed with land use context: lower at sites in cropping landscapes than those in grazing or mixed landscapes. The spread of exotic plants is a significant focus of grazed native vegetation studies (Fensham 1998a). It has been well established that an increase in livestock grazing pressure is associated with losses in native species which are replaced by exotic species (Prober & Thiele 1995; Yates et al. 2000). Furthermore, exotic species may be more prevalent in grazing landscapes, as it is a common practice to introduce exotic pasture species to improve

pasture and cattle productivity (Kaur et al. 2005). A greater prevalence of exotic species in the surrounding landscape may increase the potential for the propagation of exotic species in nearby patches of remnant vegetation, as shown in the present study. Although weeds are often associated with cropping landscapes they tend to occur at remnant patch edges (Beer & Fox 1997), so this association may not relate to the remnant core where sampling occurred. Alternatively, livestock may disperse plant matter past the edge and into the remnant core. Therefore, a high prevalence of exotic species in grazing landscapes may increase exotic richness at the patch, regardless of whether grazing occurs at the patch, but further research may be required to determine this.

Although land use context was important for floristic composition, it did not have any significant effects on overall stand structure. There are very few studies which document the effects of the surrounding landscape on stand structure. This is possibly because in most studies which examine both these factors use stand structure as an explanatory variable rather than a response variable (e.g. Lawler & Edwards 2002). The only study reviewed which examined structure in response to factors in the surrounding landscape was undertaken in Brigalow (*Acacia harpophylla*) vegetation, which found that the proportion of land uses in the surrounding landscape was not overly important in explaining patterns in stand structure (Wagner 2011).

Similar to land use context, patch grazing intensity was also found to have some influence on floristic composition, but little effect on stand structure. Patterns in overall floristic composition were observed in relation to patch grazing intensity, with a distinction between ungrazed sites and those with low grazing intensity in grazing landscapes; however, differences among patch grazing levels were not globally significant (Table 3.5). Livestock grazing has been known to affect floristic composition in other Australian ecosystems (e.g. Clarke 2003; Pettit & Froend 2001). The difference in floristic composition between ungrazed sites and those with low levels of grazing may be due to the potential loss of tussock species with low levels of grazing, which may then allow for greater recruitment and survival of inter-tussock species.

Patch grazing was found to influence some other measures of floristic composition, despite having no significant impact on overall floristic composition. Sites with low levels of grazing had a significantly lower richness of native species than sites with high levels of patch grazing, with intermediate levels of richness at ungrazed sites. This result is similar to that found by Fensham et al. (1998a), where native species richness was significantly lower under low levels of grazing than under high levels of grazing in poplar box woodlands of the Darling Downs. However, Fensham et al. (1998a) noted that these results differed to those of other studies, but did not suggest why. The results of Fensham et al. (1998a) and the present study are contrary to findings from Dorrough et al. (2006), who found that native richness was highest at moderate levels of grazing disturbance, and lower when grazing is regular or absent, but only when soil phosphorus levels were low. However, in a meta-analysis of plant species richness responses to grazing impacts, Proulx and Mazumder (1998) found that there was a reversal of this trend under contrasting soil nutrient concentrations. In nutrient poor or non-enriched systems, species richness was significantly lower under high levels of grazing than under low levels of grazing (Proulx & Mazumder 1998). In contrast, species richness in nutrient rich or enriched systems was significantly higher under high levels of grazing than under low levels of grazing (Proulx & Mazumder 1998). Although the current study sites may not be enriched *per se*, the fertile soils which these poplar box woodlands are associated with (see Appendix H) may be sufficiently rich in nutrients in comparison to the systems studied by Dorrough (2006) to produce contrasting patterns. Therefore, as suggested by Proulx and Mazumder (1998), in nutrient-rich ecosystems the increase in native species richness under high grazing may be due to regrowth, or perhaps an increase in the dominance of inedible plant species.

The richness of C4 plants, along with the C4:C3 richness transition ratio was also associated with differences in patch grazing intensity. C4 richness and the ratio of C4 to C3 plant richness were significantly lower under low levels of patch grazing than at ungrazed sites, and intermediate under high levels of patch grazing (Figure 3.3b and c). These results differ to those of other studies which have found that the richness of C3 and C4 plants respond positively to the presence of livestock grazing (Collins & Calabrese 2012; Collins et al. 2002). Such contrasting results may be due to the methods used to categorise patch grazing intensity (see discussion below).

In this study, the intensity of patch grazing had no significant effect on overall stand structure, or individual structural variables. This finding is contrary to much of the literature, which shows that the grazing of livestock can have significant effects on the structure of vegetation in eucalypt ecosystems (e.g. Duncan et al. 2007; Ludwig & Tongway 2002). More specifically in poplar box woodlands, previous research of grazing impacts have shown alterations in forb biomass, herbage biomass, and shrub abundance (Prober & Thiele 2004; Tunstall et al. 1981). Such contrasting results may be due to the methods used to categorise patch grazing intensity (see discussion below).

A large component of the structural make up of poplar box woodlands is its understorey. Studies have shown that these short-lived components are responsive to rainfall events (Heady 1958). Furthermore, Lunt et al. (2007b) noted that seasonal rainfall was found to have a greater impact than grazing activity on plant cover, especially the cover of exotic annuals (Lunt et al. 2007b). As sampling in this study was preceded by above average rainfall, there may have been changes to grass and herb cover which could have affected results relating to overall stand structure and patch grazing. In addition, results relating to long-lived components of the stand structure may have been affected by the methods used to categorise patch grazing intensity (see discussion below).

Many of the results observed here in relation to patch grazing intensity do not agree with the results of other studies. Furthermore, there is uncertainty as to why the overall floristic composition only differed among the levels of patch grazing in grazing landscapes, and not in mixed or cropping landscapes. These varied responses to patch grazing may be explained by the methods used to categorise patch grazing intensity. As detailed, historical knowledge of sites was lacking, evidence of more recent grazing activity was used to categorise patch grazing intensity. However, it is possible that patterns in floristic composition and stand structure are more reflective of long-term grazing trends, than recent grazing activity. For example, Pettit and Froend (2001) found that *Eucalyptus marginata* woodlands excluded from grazing for 6 years were still in transition floristically towards similarity to ungrazed vegetation, showing that recovery from livestock grazing may take many years.

Furthermore, long-term livestock grazing may act as a filter, which removes grazing sensitive species and sustains grazing tolerant species (Lunt et al. 2007a). Therefore, methods used to identify patch grazing intensity may not have been optimal, resulting in few responses and unexpected patterns (see Chapter 5).

As indicated by these analyses, some measures of floristic composition and stand structure, including overall floristic composition, the richness of native, exotic and C4 plants, the C4:C3 transition ration and the cover of graminoids, have responded to either patch grazing or land use context. This shows that both patch and landscape factors are important for overall ecosystem health in terms of floristic composition and stand structure. This finding was strengthened by significant interactions between both patch grazing and land use context. The interaction between patch grazing and land use context was significant for the C4:C3 richness transition ratio and the total foliage projected cover of trees.

Analyses using the *a priori* design have indicated that patch grazing and land use context are important for overall floristic composition and some measures of floristic composition and stand structure. This finding is supported by results of canonical correspondence analyses.

3.4.2 Relationships with the environment

Canonical correspondence analysis revealed that a combination of patch and landscape factors drive patterns in floristic composition of poplar box woodlands. This supports earlier findings which indicated that the overall floristic composition of poplar box woodlands differed with patch grazing and land use context (*a priori* design). The few studies which have examined the effects of both patch and landscape factors simultaneously have also found that both patch and landscape factors drive patterns in floristic composition (e.g. Chilcott et al. 2005; Lindsay & Cunningham 2009).

Canonical correspondence analysis revealed that both patch and landscape variables also explained a large degree of variation in overall stand structure, despite there

being no discernible patterns in relation to the *a priori* design. As indicated previously, this combined approach of examining both patch and landscape variables in relation to stand structure is uncommon in eucalypt woodlands. However, one of the few studies which examined stand structure as a response variable in relation to both patch and landscape factors found that both were important to explain trends in the stand structure of remnant and regrowth *Acacia harpophylla* vegetation in southern Queensland (Wagner 2011). In Wagner's (2011) study, patterns in stand structure in *Acacia harpophylla* woodlands were explained by the proportions of regrowth and grazing in the surrounding landscape, patch area, grazing intensity, and, to a lesser extent, soil factors.

Patterns in overall floristic composition and overall structure in poplar box woodlands were predominantly explained by similar factors. The factor which best explained patterns of floristic composition was the proportion of cropping (and grazing) in the surrounding landscape. This supports earlier findings which indicated that the overall floristic composition of poplar box woodlands differed significantly with land use context. The proportion of cropping (and grazing) in the surrounding landscape also explained a significant amount of variation in overall stand structure. The only other study found which examined the influence of the proportions of agricultural land uses in the surrounding landscape was a study of *Acacia harpophylla* woodlands in southern Queensland (Wagner 2011). Similarly, Wagner's (2011) study also showed that surrounding agricultural land uses have some influence on overall floristic composition.

Patterns in overall floristic composition in relation to land use context, and patterns in stand structure were also explained by the proportion of remnant vegetation in the surrounding landscape. As reported by Wagner (2011), the proportion of remnant vegetation in the surrounding landscape has also been found to explain some differences in floristic composition and stand structure in remnant and regrowth *Acacia harpophylla* woodlands. Higher proportions of vegetation in the surrounding landscape may provide a greater opportunity for the dispersal of seeds etc., thus the proportion of remnant vegetation surrounding a patch may influence its floristic composition and structure.

Soil variables also largely explained the patterns in floristic composition and stand structure. Although specific soil variables, such as calcium, were shown to be more important in the canonical correspondence analysis, a number of soil variables were removed prior to testing as many of these soil variables were highly correlated (Appendix D). It was therefore necessary to interpret this result more broadly as general soil fertility (soil nutrients collectively), rather than specific nutrients driving patterns in floristic composition and stand structure. Variation in floristic composition has also been associated with soil nutrients in remnant grassy eucalypt woodlands in agricultural landscapes of New South Wales (Lindsay & Cunningham 2009). In Lindsay and Cunningham's (2009) study, structural components were also significantly affected by soil nutrient concentrations, finding that the cover of exotic plant species was positively associated with soil nitrate concentration, and native plant species were negatively associated with total nitrogen and phosphorous in the soil. Native plant species may be lost through toxicity of soils with high levels of nutrients and the inability to compete with exotic species (Lindsay & Cunningham 2009), which are generally favoured in higher nutrient soils (Hobbs & Atkins 1988).

These canonical correspondence analyses therefore provide evidence to support previous findings from the *a priori* design. Both these approaches, for the most part, support the hypothesis that patch and landscape factors are important for community level responses in poplar box woodlands in agricultural landscapes.

3.5 Limitations

The understanding of trends in floristic composition was limited by the number of replicates in the *a priori* categories, particularly for those with high levels of patch grazing. If more replicates were used, it would have been possible to determine if sites 16 or 33 (high levels of patch grazing in a grazing landscape) were outliers in the nMDS ordination, or whether they simply represent a highly variable group. There were also limitations with the *a priori* design. Firstly, categories of land use context and the proportions of cropping and grazing in the surrounding landscape were found to be associated with a number of environmental variables, which could

have confounded results. Also, the variability of factors among patch grazing levels was, in some cases, counter-intuitive based on findings of other studies. This may be due to recent grazing levels being used to categorise levels of patch grazing intensity, while a long-term, historical measure of grazing intensity may be more appropriate.

Further limitations may have arisen due to recent weather events. For example, results relating to stand structure, particularly for short-lived components, may have been affected by the above average rainfalls prior to sampling.

3.6 Significance

Through examination of trends in floristic composition and stand structure, this study has demonstrated the importance of both patch and landscape factors. Conservation of vegetation in Queensland mostly focuses on managing patch factors to maintain biodiversity, structure and function (e.g, Tait 2004). However, as a number of factors in the surrounding landscape are largely ignored, some conservation and management efforts could, in part, be unsuccessful. It is imperative that future management of remnant vegetation in agricultural landscapes incorporates landscape factors with the various patch factors already considered. This aspect will be discussed more fully in Chapter 5.

3.7 Conclusion

The hypothesis that floristic composition, species richness, functional group richness and richness transition ratios, and stand structure differ among levels of patch grazing and land use context, as used in the *a priori* design, may only be partially accepted. Analyses of variance showed that native species richness was higher at high levels of patch grazing than at low levels of patch grazing, while C4 richness and C4:C3 richness transition ratio were higher at ungrazed patches than at low levels of patch grazing. Exotic plant richness was greater in mixed landscape than in cropping landscapes. The interaction between patch grazing intensity and land use context was important for the C4:C3 richness transition ratio and the total cover of

trees. Graminoid cover was higher in mixed landscapes than in grazing landscapes, although no other structural variables showed differences. Analysis of similarity showed that floristic composition differed between cropping and grazing landscapes. Therefore, it is concluded that patch grazing and land use context affect some attributes of floristic composition, but have a limited effect on stand structural variables.

The correlative approach using canonical correspondence analyses showed that multivariate trends in floristic composition and stand structure were driven by a combination of various patch and landscape factors, such as soil fertility and the proportions of cropping and grazing in the surrounding landscape. Therefore, the hypothesis that patch and landscape variables explain variation in floristic composition and stand structure is accepted.

The results of both the *a priori* and correlative studies show that both patch and landscape variables were important for overall floristic composition and stand structure, which agrees with the limited literature available (e.g. Lindsay & Cunningham 2009). Examining the effects of patch and landscape factors on floristic composition, species richness, functional group richness, richness transition ratios and stand structure has contributed to the understanding of remnant ecosystem health in agricultural landscapes. However, overall ecosystem health may also be related to the population dynamics and condition of structurally dominant species.

Chapter 4: Condition and structure of *Eucalyptus populnea* populations in the Condamine Catchment

4.1 Introduction

Many eucalypt populations in agricultural landscapes throughout Australia have exhibited declining condition (Wylie et al. 1992) and experienced reductions or changes in the recruitment of seedlings (e.g. Spooner et al. 2002). This decline in condition has been attributed to numerous factors, including changes in climate, insect attack, livestock grazing, nutrient deficiencies and toxicities, and changes in land use in the surrounding landscape (Calvert 2001; Reid et al. 2007; Wylie et al. 1992). In addition, the regeneration of eucalypt populations in agricultural landscapes has also been affected by factors such as livestock grazing, soil properties, and land use in the surrounding landscape (Dorrrough & Moxham 2005; Fischer et al. 2009; Wark 2000, cited in Reid et al. 2007; Tunstall & Reece 2004).

The overstorey of some eucalypt woodlands may be dominated by one or two species; for example, poplar box woodland generally consists of a monospecific canopy of *Eucalyptus populnea*. These structurally dominant species may have a significant effect on ecosystem resilience, as they are essentially the only species performing the functions of canopy trees (Walker 1995). It is therefore important to examine tree condition, and population structure for these structurally and functionally dominant species, to give an indication of overall ecosystem health (Yates & Hobbs 1997b) and long-term viability of these threatened ecosystems. This study examines the following general question: Do both patch and landscape factors contribute to patterns in *Eucalyptus populnea* tree condition and population structure in the agricultural landscapes of the Condamine Catchment, Queensland?

4.1.1 Tree population structure

Tree population structure is often used as an indicator of the future viability of a population (Fensham & Bowman 1992). Population structure can be variable; it may be composed of a number of rather indistinguishable age classes, or distinct cohorts (Oliver 1996). In terms of maintaining ecosystem health, it is generally considered optimal that the arrangement of stem size classes within a population should resemble the inverse J-shape typical of the log normal function (Smith et al. 1997, cited in George et al. 2005) (Figure 4.1a). Such an arrangement, comprised of many small stems with a gradually declining frequency of larger stems, indicates that population regeneration is a continuous process, with a sustained input of seedlings (Hett & Loucks 1976). This population structure has been observed in stands of *Eucalyptus camaldulensis* in the Murray floodplain in South Australia (George et al. 2005) (Figure 4.1b). This general pattern of population structure may also reflect attrition through mortality (George et al. 2005), with a gradual natural decline in trees, rather than events of mass mortality.

Deviations from the typical J-curve show changes in population structure which may indicate changes in underlying ecological processes (e.g. Fensham & Bowman 1992). For example, population structure may deviate from the typical J-curve when there are peaks and troughs in abundance of size classes (e.g. Fensham & Bowman 1992; George et al. 2005). This represents a loss of juveniles, or cohorts of individuals from previous regeneration events through disturbances (Fensham & Bowman 1992) (Figure 4.1c and d). The population structure of *Eucalyptus tetradonta* in tropical eucalypt forest on Melville Island showed a peak in the density of small trees in response to recent favourable conditions after removal of the overstorey (Fensham & Bowman 1992) (Figure 4.1c). The population structure of stands of *Eucalyptus largiflorens* on the Murray floodplain in South Australia also suggests episodic recruitment and opportunistic survival, showing peaks and troughs in tree abundance in relation to flooding regimes (George et al. 2005) (Figure 4.1d).

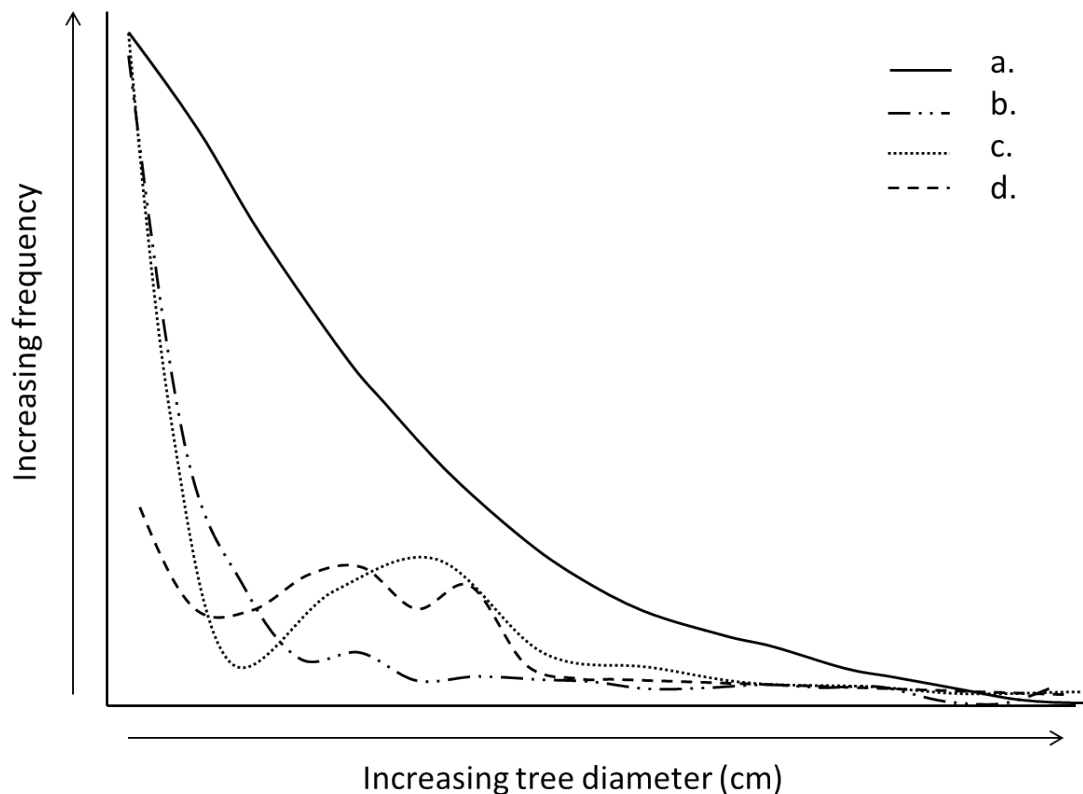


Figure 4.1 Varied trends in tree densities within age classes: a) typical inverse J-curve (adapted from George et al. 2005); b) continuous recruitment in *Eucalyptus camaldulensis* (adapted from George et al. 2005); c) peak in recruitment in *E. tetradonta* (adapted from Fensham & Bowman 1992); and d) episodic recruitment in *E. largiflorens* (adapted from George et al. 2005).

Ultimately, the persistence of a population is only possible if mortality of older trees does not exceed the growth of young trees (George et al. 2005). The persistence of a eucalypt population therefore relies on longevity of individuals and regeneration, with the latter perhaps having greater importance (Ashton 2000). Recruitment of seedlings may be affected by numerous factors. Some species, which have adapted to highly variable conditions, may have a slow background rate of recruitment, which relies on disturbance events for the episodic recruitment of seedlings (e.g. Spooner et al. 2004). Disturbance events such as flooding are essential for the episodic recruitment of *E. camaldulensis* (Di Stefano 2002), *E. largiflorens*, and *E. coolabah* (Capon 2002, cited in Capon & Dowe 2007), and are part of their natural ecology. Furthermore, the recruitment of *E. populnea* is known to be episodic in relation to heavy rainfall, but recent recruitment has been limited in the Maranoa-Balonne

Catchment and south-west Queensland (Chilcott et al. 2005; Tunstall & Reece 2004). Modifications in relation to agricultural management and development have altered disturbance regimes, including fire regimes and hydrological regimes (Lunt & Spooner 2005; Sheldon et al. 2000), which may affect the natural ecology of such ecosystems.

There are a number of other factors associated with agricultural landscapes which may affect tree recruitment. In Australia, the grazing of livestock in native remnant vegetation in agricultural landscapes is thought to be a key factor limiting eucalypt recruitment (Reid & Landsberg 2000, cited in Dorrough & Moxham 2005). For example, grazing limits tree recruitment in woodlands of Western Australia, with fenced woodlands containing a greater frequency of *E. loxophleba* and *Acacia acuminata* recruits than grazed woodlands (Prober et al. 2011). Furthermore, a survey of natural eucalypt regeneration in grassy eucalypt forests and woodlands in central Victoria, Australia, revealed that regeneration was only observed in 27% of sites surveyed, with the probability of regeneration primarily determined by the frequency of livestock grazing (Dorrough & Moxham 2005). The presence of grazing also negatively affects tree regeneration in *E. melliodora* woodlands, *E. macrocarpa* woodlands and *Callitris glaucophylla* woodlands in southern New South Wales (Spooner et al. 2002).

Soil properties have also been associated with limited eucalypt recruitment at the patch level. For example, Fischer et al. (2009) reported that in a study of remnant grassy eucalypt woodlands and dry forests in south-eastern Australia, tree recruitment was more likely at sites where soil nutrients were relatively low. Similarly, Venning (1985, cited in Semple & Koen 2003) found that the recruitment of eucalypt trees in South Australia was present in unfertilised paddocks, but rare in adjacent fertilised paddocks. Soil nutrient enrichment can also indirectly affect tree regeneration, by increasing the cover of annual plants (Lindsay & Cunningham 2009) which can outcompete emerging tree seedlings (Dorrough & Moxham 2005; Semple & Koen 2003; Spooner et al. 2002).

Tunstall and Reece (2004) identified that the recruitment of poplar box trees in south west Queensland may vary inversely with mature tree density, finding that the

germination of poplar box seedlings declined with increasing mature tree density and foliage cover. A similar relationship was also found by Debuse et al. (2009) in poplar box woodlands of south east Queensland with trees of early and late maturity. Mature trees may further limit the recruitment of eucalypt seedlings by producing fewer flowers and seeds when in poor health (Landsberg 1988; Zhang et al. 2005).

The literature relating tree regeneration to factors in the surrounding landscape is limited. A study of fragmented forests of Brazil, examined tree recruitment in relation to two different matrix types, *Vismia* dominated regrowth, and *Cecropia* dominated regrowth (Nascimento et al. 2006). Nascimento et al. (2006) found that tree recruitment was significantly higher in plots surrounded by *Vismia* than in plots surrounded by *Cecropia*. This was attributed to the higher rates of tree mortality near the edges of *Vismia*-surrounded fragments due to strong microclimate gradients (Nascimento et al. 2006).

Few studies were found which directly examine tree recruitment in relation to the surrounding landscape in Australian ecosystems. Wark (2000, cited in Reid et al. 2007) found that adjoining land uses may affect the health of *Eucalyptus camaldulensis* and *E. populnea* regeneration in the Namoi Catchment in New South Wales, with regeneration adjoining grazed areas being of poorer health than that adjoining cropland (including cotton). However, Debuse et al. (2009) found that factors in the surrounding landscape had very little bearing on patterns of tree recruitment in *E. populnea* woodlands in southern Queensland.

The recruitment of *E. populnea* is limited in areas of both Queensland and New South Wales (Wark 2000, cited in Reid et al. 2007; Tunstall & Reece 2004), which may have implications for the long-term persistence of poplar box woodlands in these regions. It is therefore necessary to study the age structure of populations to examine the likelihood of long-term persistence and determine drivers of population dynamics.

4.1.2 Decline of Tree Condition

The survival and longevity of individuals is also important for tree population structure (Ashton 2000). The survival and longevity of young trees may be affected by climate, herbivory, pathogens and competitors (Clarke 2002). However, the longevity of more mature trees may be largely affected by declining condition or dieback (Mackay et al. 1984).

In disturbed rural or agricultural lands, tree decline (dieback) is relatively rapid and premature (Wiley et al. 1993). Dieback is a common process, occasionally being an adaptive response to disturbance (Podger 1981, cited in Wiley et al. 1993).

Individuals under the stress of dieback often display symptoms of varying severity characterised by thinning of the crown and defoliation continuing along the branches to the trunk (Heatwole & Lowman 1986). After considerable crown dieback, epicormic shoots may be produced, which, in the absence of further stress, may allow for recovery (Heatwole & Lowman 1986).

Dieback has been a recognised occurrence throughout Australia for over 100 years (Wiley et al. 1993). However, the extent and severity of tree decline in rural woodlands has increased, causing concern for their future condition and persistence (Wiley et al. 1993). Dieback has been a serious concern for many eucalypt species across the New England Tablelands in New South Wales, particularly for *E. nova-anglica* and *E. blakelyi* (Jurskis & Turner 2002; Mackay et al. 1984).

Eucalyptus populnea trees are one of the many eucalypt species which are declining in condition across Australia (e.g. Chilcott et al. 2005; Reid et al. 2007). Dieback of *E. populnea* on the New England floodplains is generally considered moderate to severe (Reid et al. 2007). A comparison of studies in the Maranoa-Balonne catchment shows that dieback in poplar box trees has increased over time, from showing patchy, light to moderate dieback in the 1980s (Wylie et al. 1992) to widespread, moderate to severe dieback in recent times (Chilcott et al. 2005).

As dieback has become an increasingly prevalent issue, there has been extensive research into the dieback of eucalypt trees (Wardell-Johnson et al. 2006). As such, numerous causes of early and accelerated decline have been identified, including insect attack, nutrient deficiencies and toxicities, and extremes in climate and weather (Wylie et al. 1992). For example, the declining health of a number of eucalypt species typical of the New England Tablelands (otherwise known as New England dieback) has been attributed to defoliation by insects (Mackay et al. 1984; White 1986).

White (1986) proposed that the primary cause of tree decline and dieback is the alteration of rainfall patterns and stresses relating to water availability. Results of other studies have agreed with this hypothesis (e.g. Jensen 2011; Potts et al. 2001). For example, tree dieback has intensified in systems of chronic water deficit where water and river flow have been regulated (Reid et al. 2007). Furthermore, the poor condition of *Eucalyptus camaldulensis* has been recently attributed to groundwater decline in the Condamine Catchment (Reardon-Smith 2011) and in the Namoi Valley, New South Wales (Reid et al. 2007). Similarly, in the Namoi Valley, New South Wales, the distance to groundwater bores is an important factor for the condition of *E. populnea* in relation to water extraction activities (Reid et al. 2007).

Livestock grazing may also affect tree condition (e.g. Calvert 2001; Davidson et al. 2007). In a study of semi-arid rangelands in north Queensland, livestock grazing was associated with dieback symptoms of *Eucalyptus crebra* (Calvert 2001). Although the mechanism causing this response was not directly examined, it was suggested that decreased retention of soil surface water and infiltration rates, and increased runoff and topsoil loss caused by grazing resulted in poorer tree condition (Calvert 2001). Furthermore, Davidson et al. (2007) found that factors relating to livestock management, such as time since exclusion, grazing intensity and ease of stock access, affected eucalypt tree condition in Tasmania.

Soil properties, while influential on tree recruitment (Fischer et al. 2009), may also be important for tree condition (e.g. Landsberg 1990a). The relative availability of nutrients affects the production and palatability of leaves, thus encouraging defoliation by insects and vertebrates (Landsberg 1990c). High soil nutrient

concentrations have been negatively associated with the condition of *Eucalyptus saligna* in New South Wales (Stone & Simpson 2006) and older *E. populnea* trees in southern Queensland (Chilcott et al. 2005). However, neither Stone and Simpson (2006) nor Chilcott et al. (2005) explored the mechanism behind these relationships.

Factors in the surrounding landscape also influence the health of floodplain species such as *Eucalyptus camaldulensis* and *E. populnea*. In a study of tree decline in the Namoi Valley, New South Wales, the crown health of *E. populnea* was significantly related to the broad mix of land uses in the surrounding landscape (Reid et al. 2007). In particular, Reid et al. (2007) found that crown health was better at sites surrounded by areas used for grazing and timber, or grazing and dryland farming, than by areas used for irrigated farming, presumably in relation to groundwater extraction activities and its effects on groundwater depth. Furthermore, herbicides associated with cultivation have been shown to contribute to dieback through direct contact with leaves and soil (Banks 2006). Fertiliser application and general agricultural activity locally and in the surrounding landscape can also influence nutrient enrichment in remnant vegetation (Duncan et al. 2008) which may indirectly promote dieback (Landsberg 1990c).

The condition and longevity of eucalypt trees across Australia are of concern due to the effects of tree dieback (Reid et al. 2007). Dieback of poplar box trees has been found to be moderate to severe in areas of Queensland and New South Wales (Chilcott et al. 2005; Reid et al. 2007); however, the degree of declining tree health in the Condamine Catchment is uncertain. Dieback and general poor tree condition may be caused by a wide variety of patch and landscape factors, including insect attack, climate and weather, livestock grazing and factors in the surrounding landscape (Calvert 2001; Reid et al. 2007; Wylie et al. 1992). Therefore, it is possible that both patch and landscape influence the condition of *E. populnea* trees in agricultural landscapes.

4.1.3 Study Questions

The current state of poplar box populations in the Condamine Catchment in terms of condition and structure is uncertain. The condition and population structure of other dominant eucalyptus species have been found to be influenced mostly by patch factors (e.g. Debuse et al. 2009), but sometimes also landscape factors (e.g. Reid et al. 2007). This study asks: do patch and landscape factors contribute to patterns in *Eucalyptus populnea* condition and population structure in the agricultural landscape of the Condamine Catchment, Queensland?

In this chapter, the overall population structure of poplar box trees across the Condamine Catchment is studied. The population structure (examined using tree densities within size classes) and condition of poplar box trees are examined across patch grazing and land use context (*a priori* design). The responses of tree condition and population age structure to various patch and landscape variables are also examined, to evaluate the patch and landscape hypothesis postulated in Chapter 1. The specific hypotheses examined here are:

- The population age structure of poplar box trees demonstrates non-continuous recruitment across the Condamine Catchment;
- Tree condition and population age structure differ with patch grazing and land use context; and
- Patch and landscape variables in combination explain variation in tree condition and population age structure.

4.2 Methods

4.2.1 Field techniques

In the 500 m² quadrat used in Chapter 3, the diameter (at breast height (DBH) (1.3 m)) of each individual poplar box (*Eucalyptus populnea*) tree within the quadrat was measured. Those trees below 1.3 m in height (seedlings and some saplings) were measured at one third of their height. Some trees were found to be suckering, in

which case the DBH was calculated using two different methods. Firstly, the DBH of the main bole was determined: those trees where additional stems were of similar size to the main bole (<4 cm difference in DBH), an average was taken; those trees where additional stems had a considerably smaller circumference than that of the main bole (>4 cm DBH difference), the DBH of the main bole only was used.

The measurements of tree size (DBH) were used as a surrogate for age. It is a method which has been successfully used on a wide range of species, including many eucalypt species (Fensham & Butler 2004; George et al. 2005; Prior et al. 2006; Woodall et al. 2005). There are a number of potential issues in using size as a surrogate for age, such as differences in bark thickness unrelated to growth, and shrinkage and swelling associated with changes in moisture storage (Clarke et al. 2007; Prior et al. 2006). For the purposes of this study, the exact age of trees was not needed, but simply an estimate of age as an indication of its approximate stage of development. Therefore, it was deemed appropriate to use tree size as a surrogate for approximate age.

Using measures of DBH, each tree within the quadrat was placed in one of 4 size (surrogate for age) classes: seedlings (<2 cm DBH), saplings (2-10 cm DBH), mature trees (10-30 cm DBH) and matriarch trees (>30 cm DBH) (after Batterham 2008). The number of trees within each size class was used to determine the density of trees within each size class (per 500 m²). Mature trees are those trees which are well developed, but were mostly recruited in the last 50 years. In contrast, matriarch trees are those larger older trees, which have likely been in the landscape prior to major landscape development. Thus, this distinction between mature and matriarch trees was made using differing measures of DBH.

The condition of each mature tree was examined using a measure of foliage index (FI) (after Banks 2006). This measure involves the estimation of the percentage foliage remaining, compared to that of a healthy tree (Banks 2006). This index has been shown to be effective in assessing the condition of *E. populnea* and *E. camaldulensis* (Banks 2006; Batterham 2008; Reardon-Smith 2011; Reid et al. 2007), and is equally as informative as other measures of tree condition (Reid et al. 2007). Using the Foliage Index as a measure of tree condition, Banks (2006) has

suggested 5 classes of tree health (Table 4.1). Trees ≥ 19 cm in diameter (~60 cm circumference) were used exclusively for measures of condition (FI). Younger trees, such as saplings, were not included in this analysis as they have the potential to self-thin as they age (Souter et al. 2009). Foliage index was averaged across each site.

Table 4.1 Definition of health classes according to Foliage Index (after Banks 2006).

Tree Health Class	Definition
(i) very healthy	$\geq 95\%$ FI – vigorous; full habit; few or no stags.
(ii) healthy	75-94% FI –vigorous; few stags; little epicormic growth.
(iii) dieback: moderate to severe	30-74% FI –loss of vigour; stags; generally epicormic regrowth present moderate to poor health.
(iv) dieback: very severe	$\leq 30\%$ FI –loss of vigour; recent epicormic shoots along trunk and branches from main canopy; Stags; very poor condition.
(v) dead	No foliage; apparently dead crown.

A variety of explanatory variables were also examined. For each site, factors relating to soil properties, disturbance and stand structure, and spatial variables relating to the structure and composition of the surrounding landscape (Table 4.2 and Appendix A) were determined using the methods outlined in Chapter 3.

Table 4.2 Summary of explanatory variables examined in relation to tree condition and population stand structure.

Explanatory variables		
<u>Disturbance:</u>	Basal area (direct)	<u>Spatial factors:</u>
Clearing	Basal area (Bitterlich)	Remnant Vegetation Cover (%)
Erosion	<u>Soil factors:</u>	Area
Compaction	pH	Shape
Cowpats	Potassium	Perimeter:Area
Stags	Nitrogen	Average annual rainfall
<u>Structural factors:</u>	Phosphorus	Average maximum daily temperature
Total tree cover	Calcium	Distance to river
Total shrub cover	Magnesium	Distance to Nearest remnant
Total understory cover	Sodium	Proportion of Cropping
Total litter cover	Organic carbon	Proportion of Grazing
Stem density	Cation Exchange Capacity	Depth to groundwater
Adult stem density		

4.2.2 Statistical analyses

4.2.2.1 Overall population structure

The overall population structure of *E. populnea* across the Condamine Catchment was examined using a histogram showing the frequency distribution of trees within 5 cm size classes. The shape of the histogram was examined and compared to the shape of an inverse J-curve. This method of examining population dynamics has been successfully used in a wide range of studies (e.g. Fensham & Bowman 1992; George et al. 2005; Spooner et al. 2004). However, caution may be needed when making conclusions on population age structure and recruitment patterns made using the simple J-curve. Comparison of population age structure to an inverse J-curve may not be accurate, as it concentrates on recruitment, but does not appropriately account for mortality. This makes it difficult to determine whether peaks and troughs in structure represent episodes in recruitment or high mortality events (or both).

4.2.2.2 Effects of patch grazing and land use context

Two-way analyses of variance (ANOVA) (SPSS Inc. 2010) were used to determine if the density of trees in each size class (seedling, sapling, mature, and matriarch) and foliage index were significantly different ($p < 0.05$) among levels of patch grazing and land use context. Prior to analyses, the data on tree densities within size classes and foliage index were screened for normality and homogeneity of variances, using Shapiro-Wilk tests and Levene's tests respectively. To improve normality, data on tree densities within size classes (count data) was $\log_{10}(\text{density} + 1)$ transformed (Bakus 2007) and foliage index was arcsine transformed (on proportions) to prevent boundedness effects (Quinn & Keough 2002). Normality and homogeneity were improved by excluding site 16 from analysis of mature tree density, and site 33 from analysis of matriarch tree density. Although homogeneity of variances was achieved for all variables, it was not always possible to reach normality. However, the ANOVA is quite robust to violations of normality assumptions, so the satisfaction of homogeneity was deemed sufficient (Maxwell & Delaney 2004). Where significant differences ($p < 0.05$) were found, Tukey's post hoc tests were performed to determine which groups differ (Stevens 2007).

4.2.2.3 Modelling of *Eucalyptus populnea* condition and population age structure

Generalised linear models (GLMs) were used to model tree condition (FI) and tree densities within age classes as a function of various patch and landscape factors. The explanatory variables used in generalised linear modelling were selected using prior testing. The full complement of variables (Appendix A) was used initially; however, only the most ecologically relevant variables were retained. Variables displaying multi-collinearity were removed, retaining at least one representative of related factors. This was achieved by examining collinearity diagnostics in SPSS (SPSS Inc. 2010), and removing the variable with the highest variance inflation factor (VIF). This was repeated until the tolerance and VIF for each variable remaining were within acceptable limits (>0.1 and <10 respectively) (Tuffery 2011).

Each remaining explanatory variable was modelled separately with each of the response variables using generalised linear modelling in R (R Development Core Team 2011) to identify possible relationships. The tree densities within each size class were of an overdispersed Poisson distribution and as such, were modelled using a Quasi-Poisson distribution with a log link (Zuur et al. 2009). The foliage index was arcsine transformed prior to analysis (Quinn & Keough 2002), and was analysed using a Gaussian (normal) distribution with an identity link (linear predictor) (Hardin & Hilbe 2007). The Akaike’s Information Criterion adjusted for small sample sizes (AICc; QAICc for Quasi models), which is a measure of the relative goodness of fit of a statistical model (Hardin & Hilbe 2003), was used to decide variable inclusion. Those variables which had a smaller (Q)AICc than the null model were included in further generalised linear modelling using a model averaging framework in R (R Development Core Team 2011) (Table 4.3).

Table 4.3 Subset of environmental variables used in model averaging (See Appendix A. for codes).

Response variables	Explanatory variables
Seedling tree density (indiv./ha)	groundwater, shrubs, P
Sapling tree density (indiv./ha)	adult stems, trees
Mature tree density (indiv./ha)	size, N, P, OC, cropping %
Matriarch tree density (indiv./ha)	size, PRV2, N, P, OC, nearest remnant
Foliage Index (%)	groundwater, pH, OC

The package “MuMIn” (Barton 2012) was used to perform the model averaging in R (R Development Core Team 2011). Akaike’s Information Criterion adjusted for small samples sizes was used to compare models and calculate averaged parameter estimates (Barton 2012). Averaging was performed using models within 4 delta values (for seedlings, saplings and foliage index) and 2.5 delta values (for mature and matriarch trees) of the best model. The package “pscl” (Jackman et al. 2011) in R (R Development Core Team 2011) was used to calculate Madden’s pseudo- R^2 (an analogue of R^2) for each model to determine the proportion of variability each model explains.

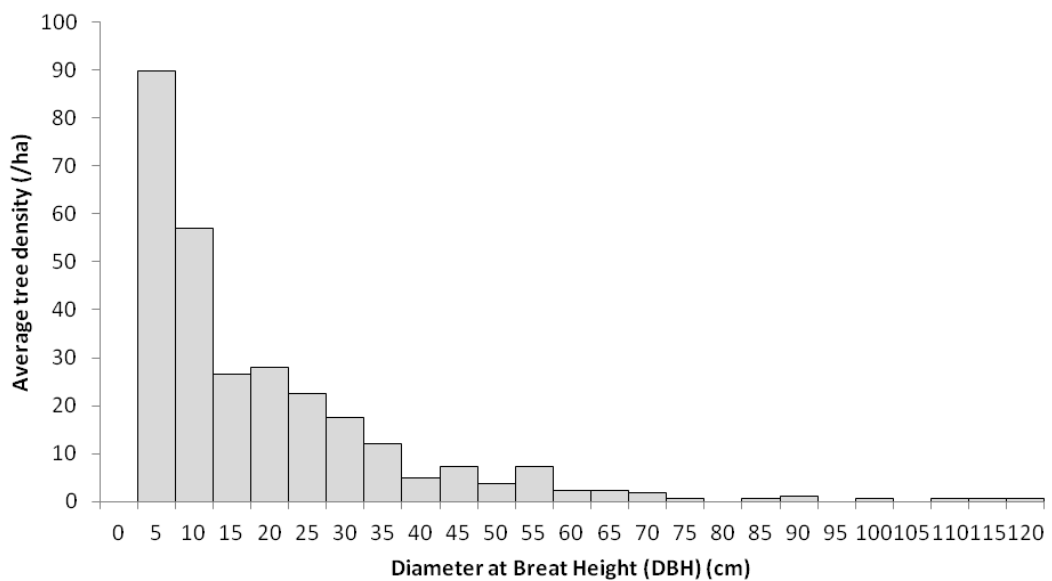
4.3 Results

4.3.1 General trends

A total of 475 *E. populnea* individuals were sampled across 33 sites. Most sites contained individuals in at least 3 size classes; however, some sites contained individuals in only 1 or 2 size classes (Appendix G). Evidence of recent recruitment was present at 58% of sites. Tree condition and densities within size classes were variable across the study region (Appendix G).

The frequency distribution of average tree densities across the range of tree diameters (5 cm classes) shows an inverse J-shaped curve (Figure 4.2), with a higher frequency of smaller trees and fewer larger trees. There were some minor peaks or troughs in tree density for larger size classes.

Figure 4.2 Histogram of tree diameter in 5 cm size classes.



4.3.2 Effects of patch grazing and land use context

Mature tree density (individuals/ha) was significantly higher at ungrazed patches than patches with low levels of patch grazing, with high levels of patch grazing intermediate between the two (Table 4.4, Figure 4.3). There were no significant differences in foliage index or tree densities in any size class across land use context types and no significant interaction between treatments (Table 4.4).

Table 4.4 Summary of two-way analyses of variance of tree densities within size classes and foliage index, including means with standard errors in parentheses. Values in **bold** are significant at the 0.05 level. Superscript indicates results of Tukey's post hoc tests.

Dependent	Factor	Degrees of Freedom	Test Statistic (F)	Interaction Test Statistic (F)	Mean (S.E.)					
					No Grazing	Low Grazing	High Grazing	Cropping	Mixed	Grazing
Seedling Density (indiv./ha)	Patch grazing	2, 24	1.847	0.132	41.05 (27.34)	75.0 (30.88)	50.0 (21.13)			
	Land use context	2, 24	0.189					28.3 (9.68)	97.8 (58.16)	38.3 (18.50)
Sapling Density (indiv./ha)	Patch grazing	2, 24	0.267	0.344	77.89 (24.75)	200.0 (84.68)	70.0 (16.93)			
	Land use context	2, 24	1.137					135.0 (46.65)	133.3 (67.33)	56.7 (20.13)
Mature Density (indiv./ha)	Patch grazing	2, 23	3.841	0.253	98.9 ^a (17.54)	60.0 ^b (20.70)	84.0 ^{ab} (32.50)			
	Land use context	2, 23	0.282					71.7 (16.23)	68.9 (19.47)	118.2 (27.03)
Matriarch Density (indiv./ha)	Patch grazing	2, 23	1.540	2.700	60.0 (9.30)	65.0 (9.82)	48.0 (10.20)			
	Land use context	2, 23	0.396					46.7 (7.91)	55.6 (10.42)	76.4 (12.30)
Foliage Index (%)	Patch grazing	2, 24	0.901	0.815	65.0 (2.89)	57.7 (4.52)	65.8 (5.57)			
	Land use context	2, 24	0.502					60.1 (3.83)	66.2 (4.83)	63.3 (3.51)

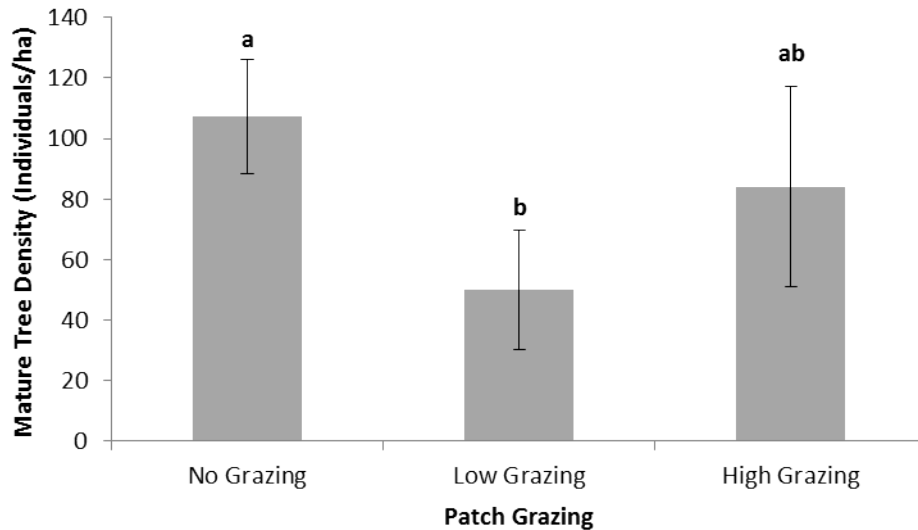


Figure 4.3 Mean levels of mature tree density (individuals/ha) by patch grazing categories. Error bars are standard error. Means sharing the same letter are not significantly different (Tukey's post hoc tests, $p < 0.05$).

4.3.3 Modelling of tree condition and densities within size classes

Most models show that a combination of patch and landscape variables explained significantly more variation in tree condition and tree densities in size classes than models including solely patch or landscape variables. Seedling density was best explained by groundwater depth and the cover of shrubs (Table 4.5). Groundwater depth appeared in all models in the confidence set ($\sum \omega_i = 1.00$) (Table 4.6). The density of adult (mature and matriarch) poplar box trees best explained trends in sapling density, but this had poor explanatory power. Mature tree density was best explained by the combination of patch size and the concentration of nitrogen and phosphorus in the soil. Patch size was present in all models, with a sum of Akaike weights ($\sum \omega_i$) of 1.00. Patch size, remnant vegetation cover and soil organic carbon best explained trends in matriarch tree density. Patch size and the proportion of remnant vegetation were consistently in all models ($\sum \omega_i = 1.00$ for each). The average foliage index of adult trees was best explained by both groundwater depth and the soil organic carbon level (Table 4.5). Groundwater depth was consistently in

all models in the confidence set for foliage index ($\sum \omega_i = 1.00$ for each), while organic carbon was less important, with a sum of Akaike weights of 0.92 (Table 4.6).

Table 4.5 Summary of results from generalised linear models of densities within size classes and tree condition (foliage index). Terms are explained in Appendix A.

Response	Model	(Q)AICc	Δ	ω_i^*	R^2
Seedling Density (indiv./ha)	gw09 + shrubs	40.78	0.00	0.44	0.3254
	gw09 + shrubs + P	41.88	1.10	0.25	0.3617
	gw09	42.43	1.65	0.19	0.2338
	gw09 + P	43.31	2.54	0.12	0.2707
Sapling Density (indiv./ha)	astems	37.76	0.00	0.29	0.0840
	trees	38.15	0.39	0.24	0.0724
	-	38.17	0.41	0.24	0.0000
	astems + trees	38.24	0.48	0.23	0.1468
Mature Density (indiv./ha)	size + N + P	46.46	0.00	0.27	0.4736
	size + N	47.02	0.56	0.21	0.4220
	size + P	47.02	0.57	0.21	0.4219
	size + N + crop	48.07	1.61	0.12	0.4488
	size + N + P + crop	48.28	1.83	0.11	0.4917
	size + P + crop	48.75	2.29	0.09	0.4383
Matriarch Density (indiv./ha)	size + PRV2 + OC	54.62	0.00	0.29	0.3776
	size + PRV2 + P	54.68	0.06	0.28	0.3768
	size + PRV2	54.90	0.28	0.25	0.3325
	size + PRV2 + N	55.66	1.05	0.17	0.3622
Foliage Index (%)	gw09 + OC	-39.53	0.00	0.47	-0.3379
	gw09 + OC + pH	-39.46	0.07	0.45	-0.4124
	gw09	-35.92	3.61	0.08	-0.1681

* ω_i = Akaike weight

Table 4.6 Averaged coefficients from model averaging of tree densities within size classes and tree condition (foliage index), and sum of Akaike weights ($\Sigma\omega_i$) for each variable based on (Q)AICc. Variables with significant sum of Akaike weights (>0.95) are in **bold**. Terms are explained in Appendix A.

Response		Explanatory variables						
		gw09	P	shrubs				
Seedling Density (indiv./ha)	Coefficient	0.144	-0.186	0.002				
	$\Sigma\omega_i$	1.00	0.37	0.69				
Sapling Density (indiv./ha)			astems	trees				
	Coefficient	-0.049	0.014					
	$\Sigma\omega_i$	0.52	0.47					
Mature Density (indiv./ha)			size	N	P	OC	crop	
	Coefficient	0.004	-0.043	-0.007	-	-0.002		
	$\Sigma\omega_i$	1.00	0.71	0.67	0.00	0.32		
Matriarch Density (indiv./ha)			size	PRV2	N	P	OC	nearrem
	Coefficient	-0.010	0.028	0.004	0.001	0.052	-	
	$\Sigma\omega_i$	1.00	1.00	0.17	0.28	0.29	0.00	
Foliage Index (%)			gw09	pH	OC			
	Coefficient	0.013	-0.051	-0.051				
	$\Sigma\omega_i$	1.00	0.45	0.92				

4.4 Discussion

4.4.1 General population structure of *Eucalyptus populnea* across the Condamine Catchment

The population structure resembling an inverse J-curve demonstrates that recruitment was a more or less continuous occurrence across the sites sampled. Batterham (2008), also reported an inverse J-curve trend in *E. populnea* population structure for the eastern Darling Downs, but showed a disproportionately higher number of poplar box seedlings and saplings per hectare. The population structure of poplar box trees holds some similarities to other Australian floodplain eucalypt species, such as *Eucalyptus camaldulensis*, which also shows the inverse J-curve trend (Figure 4.1b)(George 2004). Therefore, the overall population structure for poplar box trees across the study area does not indicate episodic recruitment, but instead indicates a steadier, continuous recruitment of seedlings. This contrasts with literature which

indicates that poplar box woodlands generally regenerate in episodes or pulses (e.g. Tunstall & Reece 2004).

Like other Australian native plants in variable environments (e.g. Spooner et al. 2004), *E. populnea* may have a regular, background rate of recruitment, but may rely on disturbance events for opportunistic episodes in recruitment. Studies have noted the episodic nature of *E. populnea* recruitment in relation to disturbance (heavy rainfall) (e.g. Tunstall & Reece 2004), which may be an adaptive response to sustain populations in unpredictable environments. However, the current study found that *E. populnea* recruitment in the Condamine Catchment is continuous, rather than episodic. This may indicate that the processes which cause episodes of recruitment have been restricted in the region, and in the absence of these events, recruitment of poplar box trees has been more continuous. The 15 year study by Tunstall and Reece (2004) in southern Queensland reported that episodes of poplar box recruitment coincided with heavy rainfall events. However, a combination of extended drought conditions (Chapter 2), the extraction of water and restriction of flows in the Darling Basin (Hankinson & Soutar 2008) have led to the alteration and reduction in environmental flows in the Condamine Catchment leading to significant reductions in the size and frequency of floods (Sheldon et al. 2000). It is possible that conditions were not conducive in recent years to facilitate episodic recruitment of seedlings.

Apart from rainfall and flooding, fire events have also been known to cause episodic recruitment in some eucalypt species (Tunstall & Reece 2004). Although the relationship between fire and poplar box trees has not been extensively studied, it is possible that fire has an important role in poplar box tree recruitment. Other studies have noted that fire may promote episodic eucalypt recruitment by releasing canopy seed stores (O'Dowd & Gill 1984), preparing the seed bed (Florence & Crocker 1962; Raison 1979), and reducing competition with other plants (Florentine 1999). The latter may be of particular importance for poplar box trees, as studies have found poplar box tree recruitment to be reliant on existing stem density (Debus et al. 2009; Tunstall & Reece 2004). However, as fire has been largely infrequent in, or excluded from ecosystems such as poplar box woodlands in recent history (Fensham & Fairfax 2003a), it is difficult to ascertain the role of fire in the episodic recruitment of poplar box trees in the region.

The continuous rate of recruitment found in this study may be sufficient to sustain the population. However, continued disturbance, loss of trees through dieback, and potential changes in climate, may increase rates of *E. populnea* mortality, making the mass recruitment episodes necessary for the long term survivorship of the population. Due to possible changes in disturbance regimes in relation to climate, water flow and fire, it is possible the poplar box woodlands in the Condamine Catchment have not experienced the disturbance events necessary to trigger episodes of mass recruitment of *E. populnea*. In the absence of significant disturbances, it is possible that conditions in poplar box woodlands have only allowed for the slow background rate of recruitment, resulting in more continuous recruitment. This may indicate that in the absence of natural disturbance regimes, poplar box woodlands are dysfunctional systems; therefore, management of these ecosystems should focus on reintroducing natural disturbance regimes (see Chapter 5). Further research on the roles of flooding and fire as stimuli for the episodic recruitment in *E. populnea* in the study region is needed to better understand the long-term viability of remnant populations in this agricultural landscape.

It is also possible that poplar box woodlands in the region have experienced episodic recruitment since the heavy rainfall and flooding events in 2010 and early 2011. As sampling was undertaken soon after such events, a response may not yet have been observable. However, further sampling at a later date may have revealed an event of episodic recruitment

4.4.2 Effects of patch grazing and land use context

Tree condition did not differ significantly across patch grazing levels. Although a number of studies have shown association with livestock grazing and eucalypt dieback (e.g. Calvert 2001; Davidson et al. 2007), there is also evidence that eucalypts are relatively resilient to the effect of livestock grazing (e.g. Banks 2006; Fensham 1998b). Fensham (1998b) found that the death of *Eucalyptus xanthoclada* in northern Queensland savanna woodlands was not associated with livestock grazing, and instead was a natural decline of the system in response to drought

conditions. Furthermore, Banks (2006) found that the health of *E. populnea* was not associated with livestock grazing in the Northern Liverpool Plains. As such, *E. populnea* may be somewhat resilient to patch grazing pressure.

The density of mature trees was significantly higher at ungrazed patches than patches with low levels of patch grazing, with high grazing intensity an intermediate between the two. There is little documentation of livestock grazing affecting the density of more developed trees (but see Debuse et al. 2009). However, the pattern in the present study could be related to management practices associated with grazing. In Queensland, it is a common practice to remove or reduce the tree layer to increase pasture productivity (Kaur et al. 2006). Such practices have been targeted towards communities such as poplar box woodlands in more recent times (Kaur & Stanley 2006); however, it is uncertain to what extent this occurs in the Condamine Catchment. Tree clearing may explain why mature tree density was higher in ungrazed patches than in patches with low grazing intensity; although it does not explain why the density of mature trees was intermediate under high levels of patch grazing. Mature trees are likely to have responded to historical disturbance; however, as this information was largely unavailable, it is uncertain whether historical disturbance influenced patterns of mature tree density.

In contrast to mature tree density, seedling, sapling, and matriarch tree densities did not differ across patch grazing (Table 4.2.). Although the density of more established eucalypt trees have shown little association with grazing in other studies (but see Debuse et al. 2009), younger, less developed eucalypt trees have been shown to be adversely affected by livestock grazing (Dorrough & Moxham 2005; Spooner et al. 2002). However, poplar box seedlings are not palatable to livestock and are only occasionally consumed during drought (Central West Catchment Management Authority 2010), which may explain why seedlings and sapling densities did not differ significantly with patch grazing intensity.

However, a lack of response to livestock grazing intensity in tree condition and tree densities within most age classes may also have occurred as a result of methods used to categorise patch grazing intensity. As historical information was difficult to obtain, categorisations were based on more recent knowledge. However, Pettit and

Froend (2001) found that the recovery of vegetation from livestock grazing in *Eucalyptus marginata* woodlands may take many years. Therefore, although recent grazing activity was used to categorise patch grazing intensity, the historical use at the patch may be more relevant (see Chapter 5).

Similar to patch grazing intensity, land use context had no significant effect on tree condition or tree density in any age class. Patterns in tree condition observed here contrast with findings from studies in agricultural landscapes which show that tree condition is influenced by land use in the surrounding landscape (e.g. Reid et al. 2007). For example, the crown condition of poplar box trees in the Liverpool Plains, New South Wales, had a highly significant relationship with the broad mix of land uses in the surrounding area at multiple scales (Reid et al. 2007). Reid et al. (2007) found that the crown health of *E. populnea* was better in areas dominated by grazing and timber, or grazing and dryland farming than in areas within 500 m, 1 km or 2 km of irrigated farming. Furthermore, limited literature shows that the health of *E. populnea* recruits in agricultural landscapes may be affected by nearby land use (Wark 2000, cited in Reid et al. 2007), which may affect seedling numbers. Contrasting results to those of Reid et al. (2007) and Wark (2000, cited in Reid et al. 2007) may represent a limitation of the *a priori* design and the methods used to categorise patch grazing and land use context, rather than the inadequacy of the patch and landscape hypothesis (see limitations below). In addition, it is possible that Reid et al. (2007) and Wark (2000, cited in Reid et al. 2007) may not have accounted for the possibility of variation in groundwater resources confounding results. In the current study, groundwater depth was found to be a significant factor influencing *E. populnea* condition and seedling density. However, as groundwater depth may be largely affected by nearby land use and associated activities (i.e. groundwater extraction on cropping land) (Reardon-Smith 2011) (Appendix D) this may have influenced results.

4.4.3 Modelling of patch and landscape factors

In this study, those models containing both patch and landscape factors explained a greater amount of variance than the model with landscape variables alone, showing

that *E. populnea* condition in the agricultural landscapes of the Condamine Catchment is driven by a combination of patch and landscape factors. This supports earlier findings which indicated the importance of both patch and landscape factors for floristic composition and stand structure (Chapter 3). The results concur with findings of Reid et al. (2007), who found that differences in *E. populnea* tree condition were due to factors relating to land use in the surrounding landscape, but also patch factors such as soil type and soil group (Reid et al. 2007).

Tree densities within age classes were also generally best explained by models which included both patch and landscape factors. Seedling, mature and matriarch tree densities were explained by a combination of patch and landscape factors, while saplings density was explained by patch factors. This generally supports earlier findings which indicated the importance of both patch and landscape factors for floristic composition and stand structure (Chapter 3) and tree condition. These results have similarities to those found in a study of tree density patterns (multiple species, including *E. populnea*) in poplar box woodlands in eastern Australia (Debus et al. 2009). In their study, Debus et al. (2009) found that the density of trees at different growth stages differed due to both patch and landscape characteristics. However, their study showed that juveniles and saplings were predominantly affected by patch variables, while the densities of trees of early and late maturity were mostly affected by landscape variables (Debus et al. 2009).

Groundwater depth was the most important variable present in all of the models of tree condition. In particular, adult tree condition had a strong positive relationship with groundwater depth, i.e. when groundwater is closer to the soil surface, adult tree condition is better. Such a result is not unexpected, as poplar box trees are able to utilise water resources from both the soil surface as well as groundwater reserves (Anderson & Hodgkinson 1997). These findings are similar to those of Reardon-Smith (2011), who also found groundwater depth to be a key feature influencing the condition of *E. camaldulensis* in the Condamine Catchment. Similarly, the condition of *E. populnea* in the Namoi Valley, New South Wales, was found to improve with increasing distance from groundwater bores, in relation to groundwater extraction associated with irrigated cropping (Reid et al. 2007).

The density of seedlings in this study also had a strong positive relationship with groundwater depth, with groundwater levels closer to the surface being associated with higher seedling densities. It is unlikely that groundwater depth directly affects seedling density, as they presumably would not be able to directly access this resource. However, mature and matriarch *E. populnea* trees may exercise hydraulic lift (Anderson & Hodgkinson 1997). Hydraulic lift involves the cycling of deep soil-water by trees and shrubs, where it is released into dry surface soils from roots at night, which may then be utilised other plants (Hodgkinson & Freudenberger 1997) such as *E. populnea* seedlings.

It is also possible that the relationship between groundwater depth and seedling density relates to mature and matriarch tree condition. It has been established that the condition of eucalypts may affect its ability to flower and set seed (Landsberg 1988; Zhang et al. 2005). As such, it is possible that a higher groundwater level improves adult tree condition, as evidenced by the positive relationship between foliage index and groundwater depth (Table 4.6), which may allow for greater flower and seed production, leading to a higher recruitment of seedlings. Further research is needed to examine this relationship in poplar box populations.

The density of saplings was partially explained by adult (mature and matriarch) stem density, where sapling density was negatively associated with the density of adult trees. A number of studies have noted the negative relationship between the densities of young poplar box trees and the density of other individuals (Debus et al. 2009; Tunstall & Reece 2004). Although adult stem density only partially explained trends in sapling densities, this may be showing competitive effects of self-thinning. Self-thinning is the process whereby tree density decreases with stand maturity as a result of the death of competitively disadvantaged trees (Johnson et al. 2009). Stands of trees self-thin to a density which allows all adult trees to obtain sufficient water, light and nutrients to survive. Therefore, saplings are perhaps also responding to demographic processes, in addition to environmental factors.

Matriarch tree density was partly explained by the proportion of remnant vegetation in the surrounding landscape, with a higher density of matriarch trees being found in areas with more remnant vegetation in the surrounding landscape. The proportion of

remnant vegetation in the surrounding landscape may give an indication of disturbance in the broader landscape, as it is possible that areas with lower proportions of remnant vegetation have been exposed to more vegetation clearing in the past. Remnant vegetation in the surrounding landscape has also been found to drive patterns in the density of early mature poplar box trees in southern Queensland (Debus et al. 2009).

Patch size was also a significant factor explaining trends in mature and matriarch tree densities. Similarly, Debus et al. (2009) also found patch size (area) to be an important factor explaining trends in early and overmature tree densities poplar box woodlands in agricultural landscapes of southern Queensland.

The concentration of organic carbon in the soil was also found to be of some importance for tree condition, with tree condition increasing with reduced levels of organic carbon in the soil. This result is similar to that found by Stone and Simpson (2006), who found that the canopy condition of *Eucalyptus saligna* in New South Wales had a weak negative association with organic carbon. These results are also similar to those of Landsberg (1990a), who found that the soil collected under healthier *Eucalyptus blakelyi* trees was less fertile, containing less organic carbon and nitrogen. In Landsberg's (1990a) studies, soil enrichment was associated with stressed trees with higher nutrient levels in foliage, which was associated with increased insect attack. It is possible that this same relationship exists for *E. populnea* trees in the Condamine Catchment. However, as herbivory was not measured in this study, this is uncertain, and further research is required to confirm this. Further study into the relationship between soil organic carbon, leaf nutrient concentrations and herbivory in *E. populnea* is needed to explain the relationship between soil organic carbon and tree condition (foliage index) observed here.

4.5 Limitations

The methods used to examine overall population age structure compare structure to an inverse J-curve, indicating continuous recruitment of seedlings. However, as discussed, this method is limited in what information it can provide; therefore, it is

necessary to conduct further research to examine phenology, recruitment and mortality more thoroughly. This may include examining the relationships between groundwater, tree condition, flower and seed production and seedling recruitment in greater depth, and long-term studies of recruitment and mortality.

Although the foliage index was sufficiently informative for this study, the use of more complex measures of tree condition may have been beneficial. For instance, a greater understanding of the potential causes of altered condition may have been achieved by examining other measures such as herbivory.

The *a priori* design did not effectively identify patterns of tree condition and population structure in relation to patch grazing intensity and land use context. Furthermore, variability of mature tree density across patch grazing levels was partially counterintuitive, with high levels of patch grazing being an intermediate between ungrazed patches and patches with low levels of patch grazing. This may be due to recent grazing levels being used to categorise levels of grazing intensity, while a long-term, historical measure of grazing intensity may be more appropriate. Therefore, future research related to patch livestock grazing should involve more reliable measures of patch grazing, or perhaps utilise grazing trials where grazing intensity can be more carefully controlled. Furthermore, the measures of land use context may have been inadequate, as the proportions of both cropping and grazing in the surrounding landscape are significantly correlated with a number of environmental variables (Appendix D).

In addition, use of the *a priori* design limited the availability of suitable sites, resulting in a limited sample size. The sample size used in statistical analyses is important as it affects the ability to detect relationships (McComb et al. 2010), influencing results and their interpretation. Although this could not be avoided when using the *a priori* design, more sample sites, which did not necessarily fit with the *a priori* design, may have been used when analysing the poplar box population age structure and modelling tree condition and tree densities within age classes in *E. populnea*.

4.6 Significance

This study has indicated that both patch and landscape factors are important for the population dynamics of the structurally dominant poplar box trees. Conservation of vegetation in Queensland mostly focuses on managing patch factors to maintain biodiversity, structure and function (e.g. Tait 2004). However, as a number of factors in the surrounding landscape are largely ignored, this means that some conservation efforts may be sub-optimal. It is imperative that future management plans incorporate various environmental factors from both patch and landscape scales. This will be discussed more in Chapter 5.

This study has also shown that tree condition and population age structure may have been affected by land management practices. The resulting decline in condition and potentially altered population structure could have significant implications for the future viability of the poplar box population. If climate and human management continue to restrict the presumably natural episodic recruitment of seedlings, and tree condition continues to decline, *E. populnea* populations in the Condamine Catchment may not be viable in the long term. This will also be discussed more in Chapter 5.

4.7 Conclusions

Analysis of population age structure across the catchment indicated that recruitment of *E. populnea* trees is continuous, with no evidence of episodic recruitment. However, this finding contrasts with published literature for *E. populnea* and may indicate possible changes (reduction or exclusion) to underlying disturbance regimes related to hydrology and/or fire. It is concluded that in the sites studied, recruitment of *E. populnea* trees is continuous, although research into *E. populnea* recruitment processes in the Condamine Catchment is required to better understand potential effects on population structure in the longer-term, and hence, persistence of this structurally and functionally dominant species in these production landscapes.

Analyses of variance showed patch grazing and land use context were not important for tree condition and for most densities within age classes, although mature tree density was significantly higher in ungrazed patches than low grazing patches. Therefore, it is concluded that tree condition and population age structure do not differ across patch grazing and land use context generally, although these patterns may represent limitations to the methodology employed.

Generalised linear modelling generally showed both patch and landscape factors were important for models of *E. populnea* condition and population age structure, although the densities of saplings and mature trees were explained by patch factors alone. These findings are consistent with the limited literature available for *E. populnea*. Therefore, it is concluded that tree condition and population age structure differ with patch and landscape factors.

The findings of this chapter mostly agree with those found in Chapter 3 in relation to floristic composition and stand structure. The significance and implications of these findings will be discussed in Chapter 5.

Chapter 5: Synthesis and Conclusions

5.1 Overview

Native remnant vegetation in Australia, as elsewhere, has undergone significant changes due to the growth in extent and intensity of agricultural practices (Sattler & Williams 1999). Patch factors have long been a focus of ecological studies (e.g. Lee et al. 2002) and it is well established that patch level factors can have significant impacts on ecological patterns and processes (e.g. Echeverria et al. 2007; Keith & Myerscough 1993; Yates et al. 2000). However, broader landscape factors have also gained recognition as significantly influencing patch dynamics, particularly in fragmented landscapes (e.g. Öckinger et al. 2011). While patch or landscape factors have mostly been studied separately, Dauber et al. (2003) suggested that a combination of both patch and landscape factors are important for biodiversity and function. This patch and landscape hypothesis is less common in ecological research in agricultural systems in Australia. This study addressed the general question: Do both patch and landscape factors influence remnant *Eucalyptus populnea* woodland health in the Condamine Catchment, southern Queensland?

In this chapter, a summary of key findings from previous chapters and a discussion of the broader ecological significance of findings are provided. The implications for the management of poplar box woodlands and other remnant vegetation in agricultural landscapes are considered. Limitations of this study are discussed, as well as potential future research directions. Finally, it is concluded that both patch and landscape factors are important for ecosystem processes in agricultural landscapes.

5.1.1 Summary of results.

This thesis examined the roles of both patch and landscape factors in relation to remnant vegetation in agricultural landscapes. Many native ecosystems in Australia, including poplar box woodlands, have shown a decline in extent and condition in relation to agricultural expansion and intensification (Chilcott et al. 2005; Yates &

Hobbs 1997a). Disturbances associated with agricultural activity may occur at both patch and landscape scales, and may manifest themselves as community level responses (changes in floristic composition and stand structure) (e.g. Dorrough et al. 2011; Prober & Thiele 2004) and population level responses (changes in tree condition and population structure) (e.g. Prober et al. 2011; Reid et al. 2007). The influences of both patch and landscape factors on community and population level responses in remnant vegetation in Queensland have not been widely examined. This thesis aimed to determine whether both patch and landscape factors drive trends in community and population levels processes in *E. populnea* woodlands. This was examined using two approaches: an *a priori* study and a correlative study.

Chapter 3 examined whether both patch and landscape factors were important for trends in floristic composition (including overall floristic composition, species richness, functional group richness and richness transition ratios) and stand structure. The *a priori* design showed that both patch grazing intensity and land use context were important for some measures of floristic composition. Patch grazing was important for native plant richness, C4 plant richness and the ratio of C4 to C3 plant richness, while land use context was important for overall composition, and the richness of exotic plants. The interaction of patch grazing intensity and land use context was important for the ratio of C4 to C3 plant richness. In contrast, patch grazing intensity had no effect on stand structural variables, and land use context was only important for graminoid cover, with the interaction of patch grazing intensity and land use context influencing the total cover of trees. Canonical correspondence analyses revealed that trends in both floristic composition and stand structure were largely driven by a combination of various patch and landscape factors, such as the proportions of cropping, grazing and remnant vegetation in the surrounding landscape, and soil fertility.

Chapter 4 studied overall *E. populnea* population age structure to provide an indication of recruitment and potential long-term population viability. This was achieved by examining the frequency distribution of *E. populnea* trees in multiple size (age) classes. Results of this study show that the population age structure of *E. populnea* trees resembles an inverse J-curve. This suggests that recruitment of poplar box trees is a relatively continuous process, which contrasts with other studies which

suggest poplar box recruitment is episodic (e.g. Tunstall & Reece 2004). The population studied appears to indicate that only background recruitment is present in poplar box woodlands of the Condamine Catchment, while recent episodes of mass recruitment are absent due to a reduction of significant disturbance events in the study area.

Chapter 4 also examined trends in population level processes, as indicated by overall population age structure and tree condition of *E. populnea* trees. Examination of the effects of patch grazing and land use context, as defined by the *a priori* design, showed that neither patch grazing intensity, nor land uses in the surrounding landscape had any significant effect on tree condition or population age structure, except for the density of mature trees, which differed with patch grazing intensity. Generalised linear models revealed that, for the most part, a combination of patch and landscape factors explain tree condition and tree densities within size (age) classes; however, some factors such as sapling and mature tree densities were driven solely by patch factors. Tree condition was driven by groundwater depth and soil organic carbon. Seedling density was driven by groundwater levels and the cover of shrubs, while sapling density was best explained by the density of adult (mature and matriarch) *E. populnea* trees. Mature tree density was driven by patch size, and soil fertility, while matriarch tree density was best explained by patch size, the proportion of remnant vegetation in the surrounding landscape, and soil organic carbon.

5.2 Ecological implications of patch and landscape drivers of community and population dynamics

This study therefore found that both patch and landscape factors were important drivers of trends in floristic composition, stand structure, and *E. populnea* condition and population age structure. A number of other studies have also shown that multi-scale factors are important for understanding ecological patterns. In Australia, both patch and landscape factors have been found to be important for patterns in bird (Martin et al. 2006), reptiles (Chilcott et al. 2005), and mammals species (McAlpine et al. 2006), and for the condition of *Eucalyptus* trees (e.g. Reid et al. 2007).

Although the importance of patch and landscape factors is already well established in landscape ecology, this has been rarely recognised for Australian production landscapes. Furthermore, this research strengthens the argument that these multi-scale factors need to be examined to gain a better understanding of ecological patterns in disturbed agricultural landscapes. This could have significant implications for the overall function and resilience of these native remnant ecosystems.

Multi-scale factors and their effects on floristic diversity are known to influence the provision of key ecosystem services as indicated by changes in net primary productivity of plants (Costanza et al. 2007). They may also affect the resilience and resistance of dominant plant species to disturbance (e.g. Godfree et al. 2011). Furthermore, the response of stand structure to patch and landscape factors may have implications for ecosystem recovery and resilience (e.g. Cordonnier et al. 2008). Patch and landscape effects on tree condition and population structure, particularly the recruitment of dominant tree seedlings, may also have implications for the resilience and long-term persistence of poplar box populations and communities. If tree condition were to decline further, there could also be a decline in seed production and thus recruitment. This, combined with other effects on recruitment, and potential tree loss with declining condition, could indicate that populations may not persist into the future.

5.3 Recruitment and condition of Eucalyptus populnea populations

This study has indicated that the long-term persistence of poplar box populations may be at risk. Previous studies of this ‘of concern’ ecosystem have noted that the recruitment of poplar box seedlings is episodic in response to environmental disturbances such as heavy rainfall (Tunstall & Reece 2004). However, the results of this study suggest that recent recruitment in the study area has occurred in a more continuous manner, with no evidence of recent episodes or pulses. This is possibly a response to altered disturbance regimes associated with rainfall, flooding and potentially fire.

Poplar box woodlands exist in highly variable landscapes in terms of climate and hydrology. As such, poplar box woodlands, like other floodplain communities, have possibly evolved to adapt to variability in their environment (Di Stefano 2002). It is therefore possible that these systems rely on such variability for normal ecological function. Natural disturbance regimes in the Condamine Catchment have changed with more recent agricultural development (in the last 50 years or so) (Sheldon et al. 2000; White et al. 2010), and may be altered further with future climate change (CSIRO 2008). Modelling shows that under different climate change scenarios, the Condamine-Balonne Catchment may experience decreases of up to 27% in runoff and 10% in rainfall, and may also experience reductions in groundwater recharge and the frequency, size and volume of floods (CSIRO 2008). Such changes could have significant implications for the overall functioning of these ecosystems. It is uncertain how such changes in the regenerative processes of poplar trees will affect the long-term sustainability and viability of poplar box populations and communities, but it is possible that the slow background rate of recruitment alone may not be sufficient to account for losses through mortality, particularly with ongoing disturbance and potential changes in climate. Further changes in disturbance regimes may also affect tree condition and other attributes such as floristic composition (Stokes et al. 2010). Continued change in the Condamine Catchment may reduce the resilience of poplar box woodlands, which may undermine their ability to persist in the long-term to provide important ecosystem services.

The declining condition of mature trees may also contribute to further concerns of long-term population viability. Current trends in *E. populnea* tree condition indicate moderate to severe levels of dieback. In the future, groundwater extraction could potentially increase (Wolfenden & Evans 2004), and climate change in the Condamine Catchment may reduce groundwater recharge (White et al. 2010). Such changes mean that groundwater extraction could continue to exceed recharge, leading to further groundwater decline (Barnett & Muller 2008), which could potentially intensify poplar box dieback and further reduce recruitment. Further declines in tree condition may increase mortality, which coupled with low recruitment rates, add to concerns of long-term population viability. It is therefore vital that groundwater resources are monitored and managed so that extraction does not exceed recharge. Such management of groundwater resources would possibly

benefit the condition of other eucalypt species such as *E. camaldulensis* (Kath 2012; Reardon-Smith 2011) and other groundwater dependent ecosystems.

5.4 Management implications

The importance of both patch and landscape factors may also be of significance for the management of *E. populnea* woodlands and other similar ecosystems in agricultural landscapes. It has been suggested that no further clearing of poplar box woodlands should occur, pressures from livestock grazing should be managed and reduced, feral animals and weeds controlled, and that some woodlands are placed under secure, long-term property agreements (Namoi Catchment Management Authority 2012; Queensland Government 2011). Although these suggestions are valid and important, this study has indicated the importance of patch *and* landscape factors; therefore the management of patch factors alone may not be appropriate or successful in poplar box woodlands. When patch factors alone are managed in poplar box woodlands (e.g. Queensland Government 2011), it is possible that poplar box tree condition and recruitment will not improve greatly, if at all, as surrounding landscape factors influencing groundwater depth will continue to have a significant negative influence on tree condition and seedling numbers (as found in this study). Therefore, it is important that factors in the broader surrounding landscape, as well as those at the patch, are recognised and managed. Apart from benefiting floristic composition, stand structure, and dynamics of *E. populnea* populations, management of both patch and landscape factors would possibly benefit many other components of the ecosystem (see Section 5.2).

Successful management of poplar box woodlands may involve the management of groundwater resource (particularly extraction rates) to prevent further declines in groundwater depth; this may potentially prevent further reductions in the condition and recruitment of poplar box trees. Although the broader landscape may be more difficult to manage than factors at the patch scale, it may be managed through effective implementation of broad-scale management plans, highlighting the importance of effective integrated catchment management.

This study has indicated that episodic recruitment of *E. populnea* has not occurred in recent history, and is instead more continuous, which may affect long-term population viability. As discussed previously, this may be due to a lack of significant disturbances and alterations to disturbance regimes in the study region. To improve the long-term persistence of *E. populnea* populations, the episodic nature of poplar box recruitment may need to be re-established; this may be achieved through the re-introduction or simulation of more natural disturbance regimes in relation to hydrology (overland flows and rainfall) and fire. The re-introduction of natural disturbance regimes has been widely recognised as a tool to improve ecosystem health (e.g. Withers 1978). Natural disturbance regimes may improve ecosystem health by increasing seedling regeneration (Dexter 1970, cited in Di Stefano 2002) and survival (Withers 1978), reducing the effects of dieback related pests and generally improving ecological imbalances (Jurskis 2005).

5.5 Limitations and future directions

There are a number of factors in this study which may have limited the full understanding of results. Some issues are discussed briefly in Chapters 3 and 4; however, there are some broader limitations which may have arisen in relation to the experimental approach taken, which may guide the design of future research. Furthermore, this study has raised a number of questions which may indicate important directions for future research.

5.5.1 Limitations of experimental approach

This study was limited by the methods used to categorise patch grazing intensity and land use context. Use of the *a priori* design showed that few response variables differed with patch grazing. As historical information was limited, knowledge of more recent grazing activity was used. However, it has been noted by Pettit and Froend (2001) that the recovery from the effects of grazing can take many years. Furthermore, a study by Lindsay and Cunningham (2009) found that soils in eucalypt woodlands enriched by livestock were not affected by grazing exclusion, indicating

that it may take many years for soil nutrients to return to normal levels after livestock removal. These studies suggest that it is perhaps more important to take note of historic or long-term grazing activity, rather than current or recent activity. Although a patch may currently be very lightly and infrequently grazed, it could have been much more heavily grazed in the past. Therefore, while the current grazing regime of a patch may be classified as being of low intensity impact, the state of the system may be more indicative of moderate or high intensity. As such, measures of recent grazing activity used here were possibly not ideal to examine trends in grazing intensity. It is suggested that future research which examines patch grazing intensity utilises grazing trials, by experimentally manipulating grazing regimes. Grazing trials have been successfully utilised in a number of studies of native remnants in agricultural landscapes (Le Brocque & Cockfield 2008; Pettit & Froend 2001; Wimbush & Costin 1979). This also highlights the importance of obtaining detailed historic information of patch disturbance, particularly in relation to historic livestock grazing regimes, to categorise patch grazing more appropriately.

The *a priori* design was also potentially spatially confounded, as a number of environmental factors were significantly correlated with the proportions of cropping and grazing in the surrounding landscape (Appendix D). However, some of these relationships are somewhat unavoidable. For example, land which is situated close to a river and receives higher levels of annual rainfall is much more valuable for cropping activities than livestock grazing. Furthermore, certain land uses, such as irrigated cropping, may have a higher need for extracted groundwater, and as such may have more of an influence on local groundwater depth than other land uses. Therefore, the correlation of cropping (and grazing) in the surrounding landscape is naturally and unavoidably associated with rainfall, river distance and groundwater depth. As such, results obtained using the *a priori* design should be interpreted with some caution.

5.5.2 Broadening the study

This study examined responses of the poplar box woodlands to land use at a broad catchment scale. However, the patterns shown may not be directly transferrable to other regions. Other regions may have different climate, geology, topography etc., which may govern the suitability of certain land uses, and may influence how land uses at the patch and in the surrounding landscape affect native ecosystems (e.g. Collins & Calabrese 2012). Furthermore, other regions may also have other land uses, such as forestry or urban development, which are dominant in the surrounding landscape. This study solely examined cropping and grazing land uses in the surrounding landscape, which was appropriate for the Condamine Catchment.

This study also focussed on responses from the one ecosystem type; however, species and ecosystems may differ in their response to factors in the environment (Fu et al. 2006; Wiens 2002). Therefore, the broader application of findings from one ecosystem to another may not be appropriate. It may therefore be necessary to expand upon this research by examining a broader range of land uses, in a different region, and in association with different ecosystems.

5.5.3 Influence of climate

Another factor which may have limited broader interpretation of results is climate. In the 12 months prior to sampling, a long drought had broken, with above average rainfall and flooding in the weeks preceding sampling (Chapter 2). Flooding may promote the germination and growth of some riparian plants (Robertson et al. 2001), but may also cause a decline in flood intolerant species which establish in floodplains during dry periods (Van der Valk & Davis 1976). Flooding events may therefore alter floristic composition and stand structure, and it is possible that results obtained in this study could have differed had sampling occurred while the region was in drought. Furthermore, the drought may have altered grazing pressures, by either lowering stocking rates while cattle were fed with supplementary feed, or increasing pressure on remaining pastures. It would therefore be advantageous to undertake this study under varied climatic conditions to examine how climatic factors affect

responses to patch and landscape factors, particularly in relation to land use. For example, further examination of trends in floristic composition, stand structure, and *E. populnea* tree condition and population structure may be undertaken when poplar box woodlands have experienced extended drought conditions.

Flooding events also have the potential to promote the episodic recruitment of floodplain or riparian eucalypt species (Capon & Dowe 2007). This may aid the recruitment of *E. populnea* seedlings; however, this has not been confirmed in this study. The flooding event immediately prior to sampling may have been too recent for an observable response in recruitment to have occurred; however, a follow up study may be capable of determining whether there was an increase in recruitment in response to the flooding event. Such research would be important in relation to climate change and long-term population viability, as climate change is expected to produce a hotter, drier climate with more severe extreme weather events and altered flooding regimes in the Condamine Catchment (Condamine Alliance 2011; CSIRO 2008).

5.5.4 Examination of population dynamics

Understanding of patterns in *E. populnea* population age structure was limited by the methods used. Comparison of population age structure (frequency distribution within age classes) to an inverse J-curve assumes a constant rate of mortality to understand patterns in recruitment. As mortality rates are unknown, the comparison of population structure to the inverse J-curve may not give an accurate representation of population dynamics (Chapter 4). Furthermore, the choice of age classes used to examine the shape of the *E. populnea* frequency distribution may have had some bearing on results and subsequent interpretation. It is therefore essential that a more complex approach is taken with future research which examines trends in recruitment or future population viability. This would require examination of trends in flowering, seeding, germination and survival of the target species' new growth. Results of the current study suggest poplar box tree condition, groundwater depth and disturbance regimes are important for patterns in poplar box recruitment. As such, it may be particularly important to undertake long-term studies which examine poplar box

phenology, recruitment and mortality, perhaps in relation to groundwater, disturbance regimes (hydrological and fire), and adult tree condition.

This study has found that the hydrology of the study area is of ecological significance to *E. populnea* populations, affecting tree condition and population dynamics. As such, future research may seek to better understand the hydrology of the Condamine Catchment and the hydrological needs of *E. populnea* trees. This may help to develop more appropriate management plans for water resources, identify future risks to *E. populnea* populations, and identify the ideal hydrological regime necessary to optimise the condition and longevity of *E. populnea* trees and woodlands.

5.6 Conclusions

Native remnant vegetation in Australia has undergone significant changes due to agricultural development. In most studies of disturbed systems, such as agricultural landscapes, the influence of patch or landscape factors is often examined independently; however, some studies have found that a combination of both patch and landscape factors may be important to adequately understand trends in such landscapes. Therefore, this study examined the question: Do patch and landscape factors contribute to patterns in health of remnant poplar box vegetation in the agricultural landscape of the Condamine Catchment, southern Queensland?

Analyses using the *a priori* design showed that some attributes of floristic composition and stand structure differed with levels of patch grazing or land use independently, as well as the interaction between patch grazing and land use context. In contrast, only mature tree density differed with patch grazing, while tree condition and other densities within age classes did not differ with levels of patch grazing or land use context, or their interaction. Therefore, the hypothesis that both patch grazing and land use context are important for remnant vegetation in agricultural landscapes is partially accepted in this study.

Canonical correspondence analyses showed that patterns in floristic composition and stand structure were explained by various environmental factors relating to both patch and landscape scales. Generalised linear models showed that tree condition and the densities of seedlings and matriarch trees were best explained by models containing both patch and landscape factors, although the densities of saplings or mature trees were best explained by patch factors alone. Therefore it is concluded that both patch and landscape factors are important for remnant vegetation in agricultural landscapes. Further research is needed to provide a more definitive explanation as to the relative contributions of patch and landscape factors, particularly patch grazing, which may be tested more accurately through experimental manipulation.

The population structure of the dominant species in this ecosystem, *E. populnea*, resembled an inverse J-curve, which shows that *E. populnea* recruitment may be a continuous process. This suggests that presumably natural episodic recruitment patterns may be inhibited by altered disturbance regimes in relation to flooding and/or possibly fire. In the future, long-term studies may be required to study trends in phenology, recruitment and mortality in relation to adult tree condition, disturbance regimes and varying climatic conditions.

This study indicates the importance of both patch and landscape variables on both community and population level processes in ecosystems in the disturbed agricultural landscape of the Condamine Catchment. These findings may be of relevance to other ecosystems in similar disturbed, agricultural landscapes and may have significant implications for future management strategies for remnant native ecosystems. As remnant vegetation in agricultural landscapes is mostly managed for patch factors, current management and conservation action may not be completely successful. More effective management should consider both patch and landscape drivers for the long-term persistence and health of remnant ecosystems in production landscapes.

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APPENDICES

Appendix A Table of variables collected at each site, their units, code used, and description of sampling method.

Variable	Units	Code	Method description
Location	degrees (°), minutes (')	-	Co-ordinates (Degrees and minutes) Southing and Easting.
Slope	%	-	The incline of the ground surface.
Aspect	-	-	The compass direction that the slope faces.
Land form	-	-	Subjective notes on land form.
Present use	-	-	Subjective notes on present land use.
Clearing	Score 0-3	clearing	(0) no disturbance; (1) minor; (2) moderate; (3) severe. Examined using visual evidence such as stumps, lying trees.
Erosion	Score 0-3	erosion	(0) no disturbance; (1) minor; (2) moderate; (3) severe. Examined using visual evidence.
Compaction	Score 0-3	compact	(0) no disturbance; (1) minor; (2) moderate; (3) severe. Examined using visual evidence.
Cowpats	density/ 500m ²	cowpats	Direct count of cowpats in the quadrat.
Stem density	density/ 500m ²	stems	Direct count of all <i>E. populnea</i> trees in the quadrat.
Adult stem density	density/ 500m ²	astems	Direct count of mature and matriarch <i>E.</i> <i>populnea</i> trees in the quadrat.
Stags	-	stags	Direct count of stags (dead, standing trees) visible in the patch.
Foliage index	%	FI	Site average of foliage remaining (compared to that of a healthy tree) of mature trees.
Cover trees >30m	%	tree1	Foliage projected cover of trees >30 m in quadrat.
Cover trees 10-30m	%	tree2	Foliage projected cover of trees 10-30 m in quadrat.
Cover trees <10m	%	tree3	Foliage projected cover of trees <10 m in quadrat.
Cover shrubs >2m	%	shrub1	Foliage projected cover of shrubs >2 m in quadrat.
Cover shrubs <2m	%	shrub2	Foliage projected cover of shrubs <2 m in quadrat.
Cover herbs	%	herbs	Cover of herbs and forbs in quadrat.
Cover graminoids	%	gram	Cover of grasses and sedges in quadrat.
Cover of coarse litter	%	coarse	Cover of logs and branches (>10 cm circumference) in quadrat.
Cover of woody litter	%	woody	Cover of woody litter (<10 cm circumference) in quadrat.
Leaf litter	%	leaf	Cover of leaf litter in quadrat.

Variable	Units	Code	Method description
Cryptogams	%	crypto	Cover of cryptogams in quadrat.
Bare ground	%	bare	% area of bare ground in quadrat.
Total tree cover	%	trees	Summed foliage projected cover of trees (>30 m, 10-30 m, 10 m).
Total shrub cover	%	shrubs	Summed foliage projected cover of shrubs (>2 m, <2 m).
Total understory cover	%	under	Summed cover of herbs and graminoids.
Total litter cover	%	litter	Summed cover of litter cover (logs and branches, woody, and leaf).
pH	pH units	pH	Method 4A1 of Rayment and Lyons (2010).
Potassium	mg/kg	K	Method 15D3 of Rayment and Lyons (2010).
Nitrogen	mg/kg	N	Method 7B1 of Rayment and Lyons (2010).
Phosphorus	mg/kg	P	Method 9B1 of Rayment and Lyons (2010).
Calcium	mg/kg	Ca	Method 15D3 of Rayment and Lyons (2010).
Magnesium	mg/kg	Mg	Method 15D3 of Rayment and Lyons (2010).
Sodium	mg/kg	Na	Method 15D3 of Rayment and Lyons (2010).
Organic content	%	OC	Method 6A1 of Rayment and Lyons (2010).
Cation Exchange	meq/100g	CEC	Method 15D3 of Rayment and Lyons (2010).
Remnant Vegetation Cover (%)	%	PRV2	Percent of remnant vegetation in a 2 km radius.
Area	ha	area	Area of sampled patch.
Shape	-	shape	Linear= \geq length 2.5 x width; block= length < 2.5 x width.
Perimeter:Area	-	P:A	Ratio of patch perimeter (m) to size (ha).
Average annual rainfall	mm	rain	Interpolated Average annual rainfall.
Average maximum daily temperature	°C	temp	Interpolated Average maximum daily temperature.
Distance to river	m	riverdis	Distance from patch centre to nearest river edge.
Distance to Nearest remnant	m	nearrem	Distance from patch edge to edge of nearest remnant patch.
Cropping	%	crop	% area utilised for cropping in 5 km radius.
Grazing	%	graz	% area utilised for grazing in 5 km radius.
Depth to groundwater	m	gw09	Depth to groundwater at site as at 2009.

Appendix B Testing of quadrat size for floristic composition.

The 500 m² quadrat was used on the first few sites, and the results from initial sampling were used to test efficiency. An average cumulative species curve was produced in relation to distance from the centre of the quadrat, showing that the number of new species sightings starts to plateau before the area reaches 500 m² (Figure B.1). This indicates a very low chance of new species sightings beyond this point, meaning that the 500 m² quadrat is efficient in capturing the diversity of plants within the entire patch. This assumption was tested by sampling up to 1000 m² at 2 sites (from different treatments).

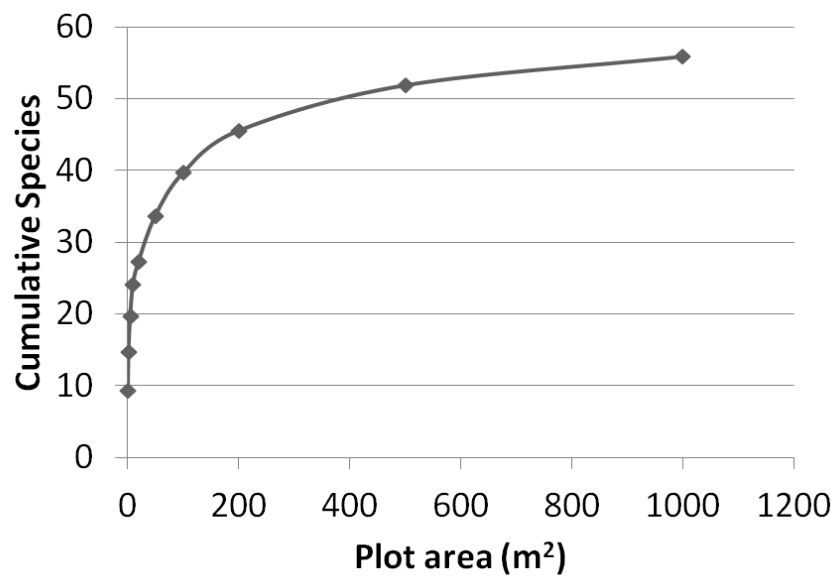


Figure B.1 Cumulative species curve for first 9 sites at 500 m² and two sites at 1000 m².

Appendix C Summary table of general site data including date visited, co-ordinates, altitude, soil type, and management notes for each site.

Site	Date visited	Co-ordinates S	Co-ordinates E	Altitude (m)	Soil type	Management	Other notes
1	17/02/2011	27°6.255'	151°1.940'	320.127	vertisol	grazing (roadside)	
2	7/03/2011	27°44.519'	151°13.319'	376.637	sodosol	minimal (roadside)	Marks on <i>E. populnea</i> from fire?
3	71/03/2011	26°45.519'	151°5.768'	359.080	vertisol	minimal (roadside)	
4	24/03/2011	27°7.647'	151°14.597'	342.612	vertisol	minimal (roadside)	
5	1/07/2011	26°48.211'	151°07.584'	357.945	vertisol	minimal (roadside)	Pooled water in quadrat.
6	12/04/2011	27°13.064'	151°16.981'	341.916	vertisol	minimal (roadside)	
7	21/4/2011	26°55.736'	150°11.813'	289.138	vertisol	minimal	Pooled water in quadrat.
8	29/04/2011	26°51.679'	151°10.394'	354.935	vertisol	minimal (roadside)	
9	1/05/2011	27°20.164'	150°49.447'	374.812	sodosol	minimal (roadside)	
10	05/05/2011	26°53.666'	150°22.834'	289.005	sodosol	minimal	Marks on <i>E. populnea</i> from fire? More open, less grass cover.
11	7/05/2011	26°57.340'	150°53.359'	316.031	sodosol	grazing (cattle movement)	Deep hoof marks in soil, less grass cover.
12	12/05/2011	27°16.209'	151°26.268'	363.240	vertisol	grazing (1 horse)	-
13	12/05/2011	27°16.270'	151°26.185'	362.567	vertisol	grazing (2 cattle current, more in past), no pasture improvement	-
14	13/05/2011	26°58.434'	151°21.756'	396.557	vertisol	minimal	-
15	16/05/2011	28°10.014'	150°35.978'	257.312	chromosol	minimal (roadside)	-
16	17/05/2011	27°43.594'	150°52.840'	333.669	sodosol	grazed forestry	-

Site	Date visited	Co-ordinates S	Co-ordinates E	Altitude (m)	Soil type	Management	Other notes
17	19/05/2011	27°20.179'	151°16.171'	342.931	vertisol	minimal (roadside)	-
18	20/05/2011	27°09.651'	151°12.394'	339.299	vertisol	minimal (roadside) *	-
19	24/05/2011	27°20.140'	150°44.064'	345.186	sodosol	minimal	
20	24/05/2011	27°19.497'	150°58.303'	347.753	sodosol	minimal	Pooled water in quadrat. (rained heavily 23/5/11)
21	26/05/2011	27°15.574'	151°22.032'	354.708	vertisol	grazed (heavily)*	-
22	26/05/2011	27°17.493'	151°21.735'	352.617	vertisol	minimal *	Hoof marks in soil.
23	31/05/2011	27°49.381'	151°12.228'	398.180	sodosol	minimal	-
24	31/05/2011	27°22.501'	151°46.390'	421.903	chromosol	minimal *	-
25	31/05/2011	27°23.342'	151°48.719'	414.358	chromosol	minimal *	-
26	2/06/2011	27°01.779'	151°18.255'	379.494	vertisol	grazed (recently)	-
27	2/06/2011	26°59.802'	151°1.623'	321.515	vertisol	minimal	Some <i>E. populnea</i> regrowth, flood damage/debris, older trees unhealthy/suckering.
28	6/06/2011	26°53.191'	151°2.879'	331.254	vertisol	minimal (roadside)	Other tree species present.
29	9/06/2011	26°52.926'	151°12.937'	362.202	vertisol	minimal (roadside) *	-
30	21/06/2011	27°47.755'	151°17.275'	378.803	vertisol	minima	A slight gully
31	21/06/2011	27°23.560'	151°50.818'	428.462	vertisol	minimal (conservation) *	-
32	28/06/2011	27°23.265'	151°49.363'	415.628	chromosol	grazing (of chickens - intermittent)	-
33	5/07/2011	27°19.506'	151°39.191'	403.426	chromosol	grazing (250 head of cattle on rotation)	Fallen/cleared trees have left some contouring (where root mass was).

*Grazed in preceding year when sampled by Batterham (2008).

Appendix D Pearson's r correlations between explanatory factors. Values in **bold** are significant at the 0.05 level; * indicates significance at the 0.01 level.

	pH	K	N	P	Ca	Mg	OC	Na	CAC	crop	graz	PRV2	rain	temp	gw09	riverdis	nearrem	area
pH	1.00																	
K	0.11	1.00																
N	-0.08	0.64*	1.00															
P	-0.08	0.75*	0.45*	1.00														
Ca	0.19	0.82*	0.49*	0.64*	1.00													
Mg	0.21	0.72*	0.31	0.40	0.77*	1.00												
OC	-0.22	0.73*	0.57*	0.71*	0.65*	0.47*	1.00											
Na	0.28	0.43	0.19	0.12	0.58*	0.65*	0.38	1.00										
CAC	0.22	0.83*	0.45*	0.57*	0.96*	0.92*	0.61*	0.67*	1.00									
crop	0.23	0.44	0.13	0.22	0.49*	0.66*	0.28	0.26	0.59*	1.00								
graz	-0.30	-0.34	-0.08	-0.16	-0.42	-0.61*	-0.21	-0.27	-0.52*	-0.96*	1.00							
PRV2	-0.17	0.00	0.05	0.09	-0.05	-0.32	-0.10	-0.14	-0.17	-0.52*	0.58*	1.00						
rain	-0.02	-0.22	-0.04	-0.08	-0.29	-0.53*	-0.10	-0.23	-0.41	-0.53*	0.45*	0.16	1.00					
temp	0.05	-0.19	-0.03	-0.30	-0.02	0.20	-0.31	0.00	0.06	0.10	-0.14	0.16	-0.61*	1.00				
gw09	-0.29	0.07	0.06	0.18	0.09	-0.17	0.20	0.05	-0.02	-0.42	0.45*	0.34	0.20	-0.27	1.00			
riverdis	0.19	0.23	0.21	0.00	0.20	0.15	0.13	0.20	0.20	0.42	-0.41	-0.31	0.00	-0.27	-0.38	1.00		
nearrem	0.28	-0.16	-0.39	-0.26	-0.07	0.22	-0.34	0.15	0.05	0.27	-0.39	-0.25	-0.19	0.31	-0.23	-0.11	1.00	
area	0.34	-0.31	-0.25	-0.30	-0.25	-0.20	-0.34	0.05	-0.24	-0.22	0.03	0.12	0.19	0.20	-0.06	-0.05	0.39	1.00
stems	-0.03	-0.13	-0.27	-0.04	-0.11	-0.13	-0.03	0.00	-0.12	-0.14	0.06	0.07	0.09	-0.13	0.30	0.07	0.03	0.32

Appendix E Summary of floristic data

Appendix E1 Summary of plant species frequency (0-9) for sites and functional groups; Origin (Native or Exotic); Life Cycle (Perennial, Short-Lived Perennial, or Annual); Life form (Tree, Shrub, Herb/forb, Graminoid, or Vine), and Photosynthetic pathway (C3 or C4). *indicates sighted in the 10th sub-quadrat.

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Abutilon malvifolium</i>	N	P	S	C3	0	0	0	0	0	2	0	0	0	0	0	0	3	0	3	6	0
<i>Acacia excelsa</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia salicina</i>	N	P	S	C3	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia sp 1.</i>	N	P	S	C3	0	1	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
<i>Acacia sp 2.</i>	N	P	S	C3	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia stenophylla</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Acacia tenuinervis</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Alectryon diversifolius</i>	N	P	S	C3	0	0	0	0	0	0	2	0	0	0	0	2	0	0	0	0	0
<i>Alternanthera denticulata</i>	N	A	H	C3	0	7	4	4	9	6	0	0	0	5	0	9	1	4	7	0	0
<i>Alternanthera nana</i>	N	P	H	C3	0	0	0	0	0	2	0	0	0	7	0	0	0	9	0	7	0
<i>Alternanthera nodiflora</i>	N	A	H	C3	0	0	0	0	0	0	9	0	0	2	0	0	0	0	0	0	0
<i>Ammannia multiflora</i>	N	A	H	C3	8	0	2	0	6	0	0	0	0	0	0	0	0	0	0	0	0
<i>Arabidella eremigena</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	8	4	0	0	0	0	0	0	0
<i>Arabidella nasturtium</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Aristida calycina</i>	N	P	G	C4	0	0	0	0	0	5	0	0	9	0	0	0	5	9	9	7	0
<i>Aristida caput-medusae</i>	N	P	G	C4	0	0	0	0	0	0	0	0	3	0	0	0	0	0	1	4	0
<i>Aristida gracilipes</i>	N	P	G	C4	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0
<i>Aristida latifolia</i>	N	P	G	C4	0	0	0	0	0	2	0	0	6	0	0	0	0	0	0	0	0
<i>Aristida leptopoda</i>	N	P	G	C4	0	0	0	9	0	8	0	8	0	0	0	3	0	0	0	1	0

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Aristida queenslandica</i>	N	P	G	C4	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Aristida ramosa</i>	N	P	G	C4	0	8	4	0	0	0	0	3	0	0	3	2	9	0	7	9	0
<i>Asperula conferta</i>	N	P	H	C3	9	0	9	0	9	0	0	8	9	0	0	0	4	2	9	0	0
<i>Asteraceae sp. 1</i>	-	-	H	-	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0
<i>Asteraceae sp. 2</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Asteraceae sp. 3</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Asteraceae sp. 4</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Asteraceae sp. 5</i>	-	-	H	-	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Atriplex muelleri</i>	N	A	H	C4	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
<i>Atriplex semibaccata</i>	N	P	H	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Austrostipa verticillata</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
<i>Bidens pilosa</i>	E	A	H	C3	0	0	0	0	0	0	0	5	0	0	0	0	7	0	0	0	0
<i>Boerhavia diffusa</i>	N	P	H	C4	0	0	7	0	0	1	0	0	3	2	5	2	4	0	9	2	4
<i>Bothriochloa biloba</i>	N	P	G	C4	0	0	0	1	0	0	0	9	0	0	0	0	0	0	0	0	0
<i>Bothriochloa bladhii</i>	N	P	G	C4	0	0	0	3	2	2	5	4	7	0	9	0	2	0	0	0	0
<i>Bothriochloa decipiens</i>	N	P	G	C4	0	9	8	9	7	9	9	9	9	0	0	9	9	9	7	4	9
<i>Brachyachne convergens</i>	N	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brachychiton populneus</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
<i>Brachyscome basaltica</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brachyscome ciliaris</i>	N	P	H	C3	9	0	0	7	0	4	0	0*	3	0	0	0	4	5	1	9	0
<i>Brachyscome curvicarpa</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brachyscome debilis</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Brachyscome multifida</i>	N	A	H	C3	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brunoniella australis</i>	N	P	H	C3	8	9	0	9	4	6	8	9	4	0	8	7	0	5	1	9	5
<i>Bryophyllum tubiflorum</i>	E	P	H	CAM	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Bulbine alata</i>	N	A	H	C3	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0
<i>Bulbine bulbosa</i>	N	P	H	C3	0	0	0	4	3	0	0	0	0	0	0	0	0	0	0	0	0
<i>Bursaria spinosa</i>	N	P	S	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Callitris columellaris</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Calotis cuneifolia</i>	N	P	H	C3	0	0	0	0	0	0	0	3	7	3	0	0	0	0	0	0	0
<i>Calotis lappulacea</i>	N	P	H	C3	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	8	0
<i>Calotis scabiosifolia</i>	N	P	H	C3	0	9	5	8	0	7	0	4	9	0	0	5	7	0	2	0	3
<i>Calotis scapigera</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0
<i>Camptacra barbata</i>	N	P	H	C3	0	4	0	9	4	0	0	0*	0	0	0	0	0	0	0	0	0
<i>Carex inversa</i>	N	P	G	C3	8	6	8	3	8	0	3	1	0	2	8	8	8	7	1	0	0
<i>Cassinia laevis</i>	N	P	S	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
<i>Casuarina cristata</i>	N	P	T	C3	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	2	0
<i>Cenchrus ciliaris</i>	E	P	G	C4	0	0	0	0	0	0	0	0	0	4	0	0	0	0	9	0	0
<i>Centipeda minima</i>	N	A	H	C3	0	0	0	0	2	0	0	0	0	8	0	0	0	0	0	0	0
<i>Chamaesyce drummondii</i>	N	P	H	C4	0	0	8	0	5	2	0	0	9	0	0	0	0	0	0	5	0
<i>Cheilanthes sieberi</i>	N	P	H	-	0	8	0	0	0	0	0	0	8	0	0	0	0	0	3	5	0
<i>Chenopodium cristatum</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Chloris divaricata</i>	N	P	G	C4	0	0	7	0	0	7	0	0	0	0	0	7	0	9	0	0	0
<i>Chloris gayana</i>	E	P	G	C4	2	2	1	0	3	0	0	9	0	2	0	0	4	4	0	0	7
<i>Chloris truncata</i>	N	P	G	C4	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Chloris ventricosa</i>	N	P	G	C4	0	1	6	0	0	5	9	2	6	2	0	7	9	8	8	6	9
<i>Chloris virgata</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	4	1	0	0	0
<i>Chrysocephalum apiculatum</i>	N	P	H	-	3	0	0	6	0	2	0	3	5	0	0	0	0	0	3	1	2
<i>Chrysopogon fallax</i>	N	P	G	C4	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
<i>Ciclospermum leptophyllum</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0

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					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Cirsium vulgare</i>	E	SL	H	C3	0	1	0	0	0	0	0	0	0	0	0	0	3	5	0	0	3
<i>Commelina cyanea</i>	N	P	H	C3	0	9	7	0	2	0	6	0	0	8	0	0	0	0	3	6	0
<i>Convolvulus arvensis</i>	E	P	H	C3	0	0	0	0	0	3	0	0	0	0	9	0	2	0	0	0	4
<i>Convolvulus erubescens</i>	N	P	H	C3	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Conyza bonariensis</i>	E	A	H	C3	0	1	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0
<i>Conyza sumatrensis</i>	E	A	H	C3	1	0	2	0	0	6	0	0	0	0	0	0	0	0	2	0	0
<i>Corymbia tessellaris</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Cullen tenax</i>	N	P	H	C3	0	0	0	4	0	0	0	2	0	1	2	0	0	0	0	0	0
<i>Cymbopogon refractus</i>	N	P	G	C4	0	8	7	8	4	9	1	3	9	0	0	0	3	6	2	4	0
<i>Cynodon dactylon</i>	N	P	G	C4	0	6	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus concinnus</i>	N	P	G	C3	0	0	7	0	8	0	0	0	0	0	0	0	3	0	4	0	0
<i>Cyperus cyperoides</i>	N	P	G	C4	0	0	6	0	2	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus difformis</i>	N	A	G	C3	9	0	0	0	3	0	0	0	0	0	0	0	6	0	0	0	0
<i>Cyperus flavus</i>	E	P	G	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus fulvus</i>	N	P	G	C4	0	0	0	0	0	0	0	0	2	0	0	0	6	0	0	0	0
<i>Cyperus gracilis</i>	N	P	G	C3	0	9	9	0	0	1	0	0	7	2	3	7	7	9	3	9	6
<i>Cyperus iria</i>	N	A	G	C4	2	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0
<i>Cyperus sp 1.</i>	N	P	G	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus sp 2.</i>	N	P	G	-	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus spp. 1 (triangle)</i>	N	P	G	-	9	1	0	0	5	0	0	8	0	0	0	4	0	0	0	0	0
<i>Cyperus spp. 2</i>	N	P	G	-	0	0	0	6	0	2	1	0	0	0	0	0	6	0	5	0	6
<i>Cyperus subulatus</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Desmodium brachypodum</i>	N	P	H	C3	0	2	0	0	3	0	0	8	7	0	0	0	0	0	0	0	0
<i>Desmodium varians</i>	N	P	H	C3	3	1	4	0	0	7	0	8	0	0	0	0	3	5	3	2	0
<i>Deyeuxia decipiens</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

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<i>Dianella caerulea</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0
<i>Dianella longifolia</i>	N	P	H	-	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
<i>Dianella rara</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dianella revoluta</i>	N	P	H	-	0	0	0	0	0	0	0	6	6	0	0	0	0	e	0	0	0
<i>Dichanthium sericeum</i>	N	P	G	C4	9	0	7	8	9	5	9	9	9	2	7	9	9	5	0	5	9
<i>Dichelachne micrantha</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dichondra repens</i>	N	P	H	C3	0	1	0	0	0	2	0	0	0	4	0	0	0	0	3	2	0
<i>Digitaria brownii</i>	N	P	G	C4	0	0	0	0	0	9	0	0	6	0	0	0	2	3	0	0	0
<i>Digitaria diffusa</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Digitaria divaricatissima</i>	N	P	G	C4	0	0	0	6	0	0	0	8	4	0	0	0	0	8	0	0	3
<i>Dodonaea viscosa</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	0
<i>Echinochloa colona</i>	N	A	G	C4	0	0	2	0	0	2	0	0	0	0	0	9	0	0	0	0	0
<i>Echinochloa esculenta</i>	E	A	G	C4	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
<i>Einadia hastata</i>	N	P	H	C3	0	0	0	0	0	0	3	0	2	0	0	0	0	0	0	0	0
<i>Einadia nutans v linifolia</i>	N	P	H	C3	0	0	0	0	0	0	9	0	0	0	0	1	2	0	2	0	0
<i>Einadia nutans v nutans</i>	N	P	H	C3	5	0	0	6	0	8		0	0	0	0	5	8	0	0	0	7
<i>Einadia polygonoides</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis acuta</i>	N	P	G	C3	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis blakeana</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis cylindrostachys</i>	N	P	G	C3	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis pallens</i>	N	P	G	C3	0	0	6	0	8	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleusine tristachya</i>	E	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Enchylaena tomentosa</i>	N	P	S	C3	0	0	0	0	0	8	4	0	0	0	6	3	4	0	9	0	0
<i>Enneapogon nigricans</i>	N	P	G	C4	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0
<i>Enneapogon sp. 1</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

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<i>Enteropogon acicularis</i>	N	P	G	C4	0	9	6	9	7	0	0	0	5	0	7	6	4	0	0	0	7
<i>Enteropogon ramosus</i>	N	P	G	C4	0	0	9	0	0	0	9	0	0	4	0	9	0	0	3	0	0
<i>Epaltes australis</i>	N	P	H	-	0	0	0	0	0	0	2	0	0	0	0	0	0	0	5	0	0
<i>Eragrostis brownii</i>	N	P	G	C4	0	0	6	0	0	1	0	0	0	0	0	0	0	9	1	3	0
<i>Eragrostis cilianensis</i>	E	A	G	C4	0	0	0	0	0	0	0	0	0	3	0	0	0	9	0	0	0
<i>Eragrostis curvula</i>	E	P	G	C4	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0
<i>Eragrostis elongata</i>	N	SL	G	C4	6	9	0	0	2	4	5	0	7	0	0	0	0	9	1	0	0
<i>Eragrostis lacunaria</i>	N	P	G	C4	0	0	0	0	0	0	0	0	5	0	0	0	0	0	5	0	0
<i>Eragrostis leptostachya</i>	N	P	G	C4	0	8	0	0	0	0	0	0	6	0	0	0	3	5	7	5	0
<i>Eragrostis megalosperma</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Eragrostis parviflora</i>	N	A	G	C4	0	4	0	0	7	4	5	0	7	0	0	0	0	0	4	8	0
<i>Eragrostis spartinoides</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eragrostis trachycarpa</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eremophila debilis</i>	N	P	S	C3	4	1	4	8	1	4	0	3	2	0	0	0	4	0	0	0	0
<i>Eremophila longifolia</i>	N	P	S	C3	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
<i>Eremophila mitchelli</i>	N	P	S	C3	0	0	0	0	0	0	7	0	4	0	0	0	0	0	2	4	0
<i>Eriochloa crebra</i>	N	P	G	C4	9	9	9	9	9	6	0	3	8	0	9	6	8	8	0	0	9
<i>Eriochloa procera</i>	N	P	G	C4	0	0	0	3	0	0	4	0	0	9	2	0	0	0	2	0	0
<i>Eriochloa pseudoacrotricha</i>	N	SL	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	4	0
<i>Eryngium plantagineum</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus camaldulensis</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus crebra</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus melliodora</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus populnea</i>	N	P	T	C3	3	3	4	3	4	4	3	4	4	4	5	4	3	6	3	5	3
<i>Eucalyptus tereticornis</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

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					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Eulalia aurea</i>	N	P	G	C4	3	5	1	1	8	1	7	0	5	0	0	3	2	0	0	5	0
<i>Euphorbia sp. 1</i>	E	P	H	-	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Eustrephus latifolius</i>	N	P	V	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Evolvulus alsinoides</i>	N	P	H	C3	0	0	0	0	0	4	0	0	3	1	0	0	0	0	8	0	0
<i>Fimbristylis dichotoma</i>	N	P	G	C4	0	7	0	0	0	3	0	0	8	0	0	0	0	0	5	0	0
<i>Fimbristylis neilsonii</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Flaveria trinervia</i>	N	A	H	C4	4	4	0	0	4	0	4	0	0	3	0	9	0	0	0	0	9
<i>Geijera parviflora</i>	N	P	T	C3	0	0	0	0	0	0	2	0	1	0	0	0	0	0	2	2	0
<i>Geijera salicifolia</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Glandularia aristigera</i>	E	P	H	C3	2	9	2	5	1	9	3	7	9	8	5	6	7	9	5	0	0
<i>Glossocardia bidens</i>	N	P	H	C4	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
<i>Glycine latifolia</i>	N	P	H	C3	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Glycine tabacina</i>	N	P	H	C3	9	9	9	9	7	8	9	9	9	0	0	9	9	9	0	9	9
<i>Glycine tomentella</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	4	2	1	4	0	9	0	0
<i>Gnaphalium pennsylvanicum</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
<i>Gnaphalium sphaericum</i>	N	A	H	C3	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gomphocarpus physocarpus</i>	E	P	S	C3	0	1	0	0	0	0	0	1	0	4	0	1	0	2	0	0	0
<i>Gomphrena celosioides</i>	E	SL	H	C4	1	6	0	4	0	4	0	0*	9	1	0	0	1	0	8	0	0
<i>Goodenia bellidifolia</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Goodenia fascicularis</i>	N	P	H	C3	0	1	0	0	3	0	0	0	0	0	3	0	0	0	0	0	0
<i>Goodenia glabra</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Goodenia sp. 1</i>	N	P	H	C3	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0
<i>Goodenia sp. 2</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Goodenia sp. 3</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0
<i>Haloragis stricta</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

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					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Heliotropium amplexicaule</i>	E	P	H	C3/C4	0	0	0	6	0	0	0	4	0	0	0	0	0	0	0	0	0
<i>Heteropogon contortus</i>	N	P	G	C4	0	0	1	0	3	0	0	5	0	0	0	0	0	0	0	0	0
<i>Hibiscus trionum</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Iseilema membranaceum</i>	N	A	G	C4	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
<i>Ixiolaena brevicompta</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
<i>Ixiolaena leptolepis</i>	N	P	H	C3	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Jasminum suavisissimum</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Juncus sp.</i>	-	P	G	C3	3	0	6	0	0	0	0	0	0	0	0	0	0	0	2	1	0
<i>Lactuca serriola</i>	E	SL	H	C3	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
<i>Lepidium pseudohyssopifolium</i>	N	SL	H	C3	0	0	0	3	0	0	0	0	3	2	0	0	7	7	1	2	0
<i>Leptochloa divaricatissima</i>	N	P	G	C3	7	0	6	0	9	6	8	0	0	0	3	9	5	0	0	0	9
<i>Lomandra confertifolia</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0
<i>Lomandra filiformis</i>	N	P	H	-	0	0	0	0	0	0	0	0	9	9	0	0	0	0	1	0	0
<i>Lomandra longifolia</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
<i>Lomandra multiflora</i>	N	P	H	-	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	5
<i>Lycium ferocissimum</i>	E	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
<i>Maireana aphylla</i>	N	P	S	C3	0	0	0	6	0	3	0	0	0	0	0	0	0	0	0	0	5
<i>Maireana microphylla</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Malvastrum americanum</i>	E	SL	H	C3	0	0	4	1	2	2	0	3	0	0	5	0	0	3	0	0	0
<i>Malvastrum coromandelianum</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	3	0	3	0	0	0	0
<i>Marsilea costulifera</i>	N	P	H	C3	0	5	4	0	9	0	1	0	0	0	0	0	0	0	5	0	0
<i>Marsilea drummondii</i>	N	P	H	C3	5	0	3	0	7	2	0	0	0	0	0	0	0	0	0	0	3
<i>Medicago minima</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Medicago sp 1.</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Megathyrsus maximus</i>	E	P	G	C4	7	0	6	0	0	0	4	0	0	5	0	5	0	0	1	0	0

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					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Melinis repens</i>	E	SL	G	C4	0	0	0	0	0	0	0	6	5	3	0	0	0	0	2	0	0
<i>Mentha satureioides</i>	N	P	H	C3	8	0	0	0	8	0	0	4	5	0	0	0	0	0	0	0	0
<i>Minuria integerrima</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Monachather paradoxus</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Neobassia proceriflora</i>	N	SL	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
<i>Neptunia gracilis</i>	N	P	H	C3	7	0	4	7	7	7	0	7	0	0	9	9	6	4	0	0	2
<i>Notelaea linearis</i>	N	p	S	C3	2	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	4
<i>Nyssanthes erecta</i>	N	SL	H	C3	0	0	7	0	0	0	0	0	8	0	0	5	5	0	2	1	0
<i>Olearia sp.</i>	N	A	S	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Opuntia aurantiaca</i>	E	P	H	CAM	0	2	0	0	0	3	0	3	5	0	0	1	0	0	0	0	5
<i>Opuntia tomentosa</i>	E	P	S	CAM	2	6	1	1	1	8	7	2	9	0	0	8	4	6	9	0	4
<i>Oxalis corniculata v. corniculata</i>	E	P	H	C3	0	9	7	6	0	3	0	6	0	7	0	0	0	0	0	0	0
<i>Oxalis perennans</i>	N	P	H	C3	0	0	0	0	4	0	5	0	0	0	0	5	5	0	0	0	9
<i>Panicum buncei</i>	N	P	G	C4	0	0	6	0	0	4	9	0	7	0	0	2	0	0	1	0	0
<i>Panicum decompositum</i>	N	P	G	C4	0	0	0	0	0	3	0	8	0	0	5	0	0	0	0	0	5
<i>Panicum effusum</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	8	7	0	0
<i>Panicum queenslandicum</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
<i>Panicum simile</i>	N	P	G	C4	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0
<i>Panicum spp.</i>	N	P	G	C4	7	1	0	9	6	6	0	8	0	4	0	2	0	0	0	0	0
<i>Parsonsia ventricosa</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Paspalidium caespitosum</i>	N	P	G	C4	0	8	9	0	0	0	9	1	9	9	0	9	9	0	0	9	9
<i>Paspalidium constrictum</i>	N	P	G	C4	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0
<i>Paspalidium distans</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Paspalidium globoideum</i>	N	P	G	C4	0	0	0	4	0	0	4	2	0	0	0	4	0	0	0	0	0
<i>Paspalum dilatatum</i>	E	P	G	C4	0	0	0	0	2	0	0	0	0	9	0	0	0	0	0	0	0

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Phyla canescens</i>	E	P	H	C3	1	8	0	0	1	0	0	0	0	0	4	4	3	0	0	0	2
<i>Phyllanthus virgatus</i>	N	P	H	C3	0	5	5	4	9	0	0	9	0	4	7	0	3	0	4	0	4
<i>Physalis lanceifolia</i>	E	A	H	C3	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Physalis longifolia</i>	E	P	H	C3	0	0	0	0	0	0	0	0	0	3	0	0	2	0	0	0	0
<i>Pimelea neo-anglica</i>	N	P	S	C3	0	2	0	0	1	0	0	7	3	0	0	0	4	0	2	0	0
<i>Pittosporum angustifolium</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0
<i>Plantago lanceolata</i>	E	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Poaceae sp 1</i>	-	-	G	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Polygonum aviculare</i>	E	SL	H	C3	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0
<i>Portulaca filifolia</i>	N	A	H	C4	0	3	1	0	0	0	0	0	0	0	0	0	0	5	5	0	0
<i>Portulaca oleracea</i>	N	SL	H	C4	0	3	1	4	0	9	9	0	9	0	9	8	5	0	0	0	0
<i>Portulaca pilosa</i>	E	A	H	C4	0	8	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
<i>Pratia purpurascens</i>	N	A	H	C3	0	8	0	0	1	0	0	4	0	0	0	0	2	0	0	0	1
<i>Pycnosorus chrysanthes</i>	N	SL	H	-	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Racosperma striatifolium</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Ranunculus inundatus</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Rapistrum rugosum</i>	E	SL	H	C3	0	0	0	0	0	0	0	0	0	0	3	0	1	0	0	0	0
<i>Rhynchosia minima</i>	N	P	H	C3	0	4	3	0	1	6	0	8	0	0	8	0	3	4	0	0	0
<i>Rostellularia adscendens</i>	N	P	H	C3	0	6	0	7	0	0	0	1	6	0	0	0	0	0	0	6	0
<i>Rumex brownii</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
<i>Sclerolaena birchii</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Sclerolaena muricata v muricata</i>	N	P	H	C3	0	0	0	4	0	0	0	0	0	0	0	8	0	0	0	0	2
<i>Sclerolaena muricata v semiglabra</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Senecio hispidulus</i>	N	P	H	C4	3	0	0	0	0	0	0	0	0	0	0	4	0	0	1	0	0
<i>Senecio quadridentatus</i>	N	P	H	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Sesbania cannabina</i>	N	A	S	C4	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0
<i>Setaria pumila</i>	E	A	G	C4	0	0	0	0	0	0	4	0	1	0	0	0	0	0	0	0	0
<i>Sida corrugata</i>	N	P	H	C3	0	0	4	8	0	7	0	1	1	0	0	4	0	0	0	4	0
<i>Sida cunninghamii</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	6
<i>Sida hackettiana</i>	N	P	H	C3	0	0	0	0	0	0	3	0	0	0	0	0	0	2	0	0	0
<i>Sida rhombifolia</i>	E	P	S	C3	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
<i>Sida rohlenae</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	7	0	0	0	3	0	0	0
<i>Solanum ellipticum</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0
<i>Solanum nodiflorum</i>	N	P	H	C3	0	0	7	0	2	2	0	0	0	5	0	0	4	6	0	0	0
<i>Solanum sp 1.</i>	N	P	H	C3	0	0	0	0	0	2	4	0	0	0	0	0	0	0	0	1	0
<i>Solanum sp 2.</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Sonchus oleraceus</i>	E	A	H	C3	0	0	0	0	0	7	0	0	0	8	6	0	3	0	0	0	9
<i>Sorghum halepense</i>	E	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Spergularia rubra</i>	E	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	6	0
<i>Sporobolus caroli</i>	N	SL	G	C4	0	0	0	0	0	7	0	0	0	0	0	4	0	0	2	0	0
<i>Sporobolus spp.</i>	N	P	G	C4	5	7	0	8	9	9	9	5	9	0	3	0	5	9	5	7	2
<i>Stellaria pallida</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0
<i>Swainsona galegifolia</i>	N	P	H	C3	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
<i>Swainsona sp. 1</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
<i>Swainsona sp. 3</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Symphyotrichum subulatum</i>	E	SL	H	C3	0	1	0	0	3	0	0	2	0	0	0	0	7	0	0	0	0
<i>Tetragonia eremaea</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	6	0	0	4	0	0	0	0
<i>Themeda avenacea</i>	N	P	G	C4	0	0	0	9	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Themeda triandra</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Triraphis mollis</i>	N	SL	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
<i>Vachellia farnesiana</i>	E	P	S	C3	5	0	0	0	0	0	0	2	0	0	0	2	5	0	0	0	0
<i>Verbena bonariensis</i>	E	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0
<i>Verbena litoralis</i>	E	P	H	C3	0	0	0	0	5	0	0	5	0	0	0	0	0	0	0	0	0
<i>Verbena officinalis</i>	E	P	H	C3	2	0	0	0	2	0	0	9	0	0	5	0	0	6	0	0	0
<i>Vittadinia pustulata</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0
<i>Vittadinia sulcata</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	9	0
<i>Wahlenbergia gracilis</i>	N	P	H	C3	0	0	3	9	4	4	0	4	5	0	0	0	1	0	4	9	2
<i>Xanthium strumarium</i>	E	A	H	C3	0	0	0	0	0	0	1	0	0	0	1	0	3	0	0	0	0
<i>Zinnia peruviana</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Unknown sp. 1</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Unknown sp. 2</i>	-	-	H	-	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 3</i>	-	-	H	-	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 4</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0
<i>Unknown sp. 5</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
<i>Unknown sp. 6</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0
<i>Unknown sp. 7</i>	-	-	H	-	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 8</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 9</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0
<i>Unknown sp. 10</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	0
<i>Unknown sp. 11</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site															
					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Abutilon malvifolium</i>	N	P	S	C3	1	5	0	5	4	3	0	0	7	0	0	0	0	0	0	3
<i>Acacia excelsa</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
<i>Acacia salicina</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0
<i>Acacia sp 1.</i>	N	P	S	C3	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia sp 2.</i>	N	P	S	C3	0	2	0	5	0	0	0	0	0	0	0	0	0	0	3	0
<i>Acacia stenophylla</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0
<i>Acacia tenuinervis</i>	N	P	S	C3	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
<i>Alectryon diversifolius</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Alternanthera denticulata</i>	N	A	H	C3	0	0	9	0	8	0	0	1	7	9	7	0	0	0	0	
<i>Alternanthera nana</i>	N	P	H	C3	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Alternanthera nodiflora</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	6	8	0	0	0	
<i>Ammannia multiflora</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Arabidella eremigena</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Arabidella nasturtium</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	
<i>Aristida calycina</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	
<i>Aristida caput-medusae</i>	N	P	G	C4	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Aristida gracilipes</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Aristida latifolia</i>	N	P	G	C4	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	
<i>Aristida leptopoda</i>	N	P	G	C4	9	0	0	8	0	0	0	5	5	0	5	0	0	0	0	
<i>Aristida queenslandica</i>	N	P	G	C4	0	7	0	0	0	7	9	0	0	0	0	0	0	2	0	
<i>Aristida ramosa</i>	N	P	G	C4	0	8	0	9	0	9	9	4	6	0	0	0	4	9	1	
<i>Asperula conferta</i>	N	P	H	C3	0	0	0	7	3	9	0	7	0	9	0	3	3	3	0	
<i>Asteraceae sp. 1</i>	-	-	H	-	2	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Asteraceae sp. 2</i>	-	-	H	-	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	
<i>Asteraceae sp. 3</i>	-	-	H	-	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site															
					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Asteraceae sp. 4</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0
<i>Asteraceae sp. 5</i>	-	-	H	-	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0
<i>Atriplex muelleri</i>	N	A	H	C4	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0
<i>Atriplex semibaccata</i>	N	P	H	C4	0	0	0	0	4	0	0	3	9	0	0	0	0	0	0*	6
<i>Austrostipa verticillata</i>	N	P	G	C3	0	2	0	0	0	0	0	2	0	0	0	0	0	2	3	0
<i>Bidens pilosa</i>	E	A	H	C3	0	1	0	0	0	1	0	4	0	0	0	1	0	9	0*	0
<i>Boerhavia diffusa</i>	N	P	H	C4	0	5	0	0	3	0	0	0	0	0	0	0	0	0	0	0
<i>Bothriochloa biloba</i>	N	P	G	C4	4	0	0	0	0	0	4	0	0	0	0	0	9	9	0	4
<i>Bothriochloa bladhii</i>	N	P	G	C4	9	0	0	3	9	0	0	0	0	6	9	9	5	0	0	0
<i>Bothriochloa decipiens</i>	N	P	G	C4	7	9	0	9	0	9	9	6	4	6	0	0	9	3	9	0
<i>Brachyachne convergens</i>	N	A	G	C4	0	0	0	0	0	0	0	9	3	0	0	0	0	0	0	0
<i>Brachychiton populneus</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brachyscome basaltica</i>	N	P	H	C3	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brachyscome ciliaris</i>	N	P	H	C3	8	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0
<i>Brachyscome curvicarpa</i>	N	A	H	C3	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
<i>Brachyscome debilis</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brachyscome multifida</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Brunoniella australis</i>	N	P	H	C3	7	0	0	5	4	5	8	0	7	0	0	4	9	7	3	9
<i>Bryophyllum tubiflorum</i>	E	P	H	CAM	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Bulbine alata</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Bulbine bulbosa</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Bursaria spinosa</i>	N	P	S	-	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
<i>Callitris columellaris</i>	N	P	T	C3	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
<i>Calotis cuneifolia</i>	N	P	H	C3	0	6	0	0	0	5	1	0	0	0	0	0	0	0	5	0
<i>Calotis lappulacea</i>	N	P	H	C3	0	0	0	0	0	0	9	0	0	0	0	0	0	0	9	0

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					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Calotis scabiosifolia</i>	N	P	H	C3	9	0	0	5	0	2	0	0	9	0	0	0	6	0	0	7
<i>Calotis scapigera</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	8
<i>Camptacra barbata</i>	N	P	H	C3	0	0	0	0	2	0	0	0	6	0	0	0	9	0	0	0
<i>Carex inversa</i>	N	P	G	C3	4	0	9	2	9	0	0	0	0	6	9	1	7	3	0	8
<i>Cassinia laevis</i>	N	P	S	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Casuarina cristata</i>	N	P	T	C3	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
<i>Cenchrus ciliaris</i>	E	P	G	C4	0	1	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Centipeda minima</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0
<i>Chamaesyce drummondii</i>	N	P	H	C4	0	0	0	4	6	0	0	0	9	2	0	0	0	0	0	9
<i>Cheilanthes sieberi</i>	N	P	H	-	0	9	0	0	0	9	0	0	0	0	0	0	0	0	0	0
<i>Chenopodium cristatum</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0*	0
<i>Chloris divaricata</i>	N	P	G	C4	2	0	0	5	0	0	0	0	4	0	0	0	0	0	0	0
<i>Chloris gayana</i>	E	P	G	C4	0	3	0	0	0	1	0	1	2	0	3	3	3	4	0	0
<i>Chloris truncata</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Chloris ventricosa</i>	N	P	G	C4	0	8	3	7	0	9	9	6	7	0	0	0	7	9	8	2
<i>Chloris virgata</i>	N	P	G	C4	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
<i>Chrysocephalum apiculatum</i>	N	P	H	-	2	0	0	0	0	4	0	0	0	0	0	0	0	6	1	5
<i>Chrysopogon fallax</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Ciclospermum leptophyllum</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Cirsium vulgare</i>	E	SL	H	C3	0	0	0	9	0	0	0	3	0	0	0	0	0	0	2	0
<i>Commelina cyanea</i>	N	P	H	C3	0	0	0	0	0	1	0	0	7	0	0	0	0	0	0	0
<i>Convolvulus arvensis</i>	E	P	H	C3	8	0	0	3	0	0	0	0	0	5	0	0	0	0	0	0
<i>Convolvulus erubescens</i>	N	P	H	C3	0	0	0	0	0	0	5	1	0	0	0	0	0	0	0	0
<i>Conyza bonariensis</i>	E	A	H	C3	0	0	0	0	0	0	0	1	0	0	0	0	0	4	0	0
<i>Conyza sumatrensis</i>	E	A	H	C3	0	0	0	0	0	0	0	5	3	0	0	0	0	0	4	0

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					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Corymbia tessellaris</i>	N	P	T	C3	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0
<i>Cullen tenax</i>	N	P	H	C3	6	8	0	0	0	0	0	0	0	0	1	0	0	5	0	0
<i>Cymbopogon refractus</i>	N	P	G	C4	8	9	0	5	0	8	9	0	5	0	0	0	0	6	0*	0
<i>Cynodon dactylon</i>	N	P	G	C4	0	0	0	0	9	0	0	0	0	0	0	0	0	0	9	6
<i>Cyperus concinnus</i>	N	P	G	C3	0	0	0	0	9	0	0	0	0	0	7	1	5	0	0	0
<i>Cyperus cyperoides</i>	N	P	G	C4	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus difformis</i>	N	A	G	C3	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus flavus</i>	E	P	G	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus fulvus</i>	N	P	G	C4	9	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus gracilis</i>	N	P	G	C3	4	9	0	7	0	3	7	8	7	9	0	0	8	9	1	5
<i>Cyperus iria</i>	N	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus sp 1.</i>	N	P	G	-	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
<i>Cyperus sp 2.</i>	N	P	G	-	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus spp. 1 (triangle)</i>	N	P	G	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cyperus spp. 2</i>	N	P	G	-	0	0	2	0	2	0	0	0	8	9	8	0	6	0	0	1
<i>Cyperus subulatus</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0
<i>Desmodium brachypodum</i>	N	P	H	C3	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0
<i>Desmodium varians</i>	N	P	H	C3	0	0	0	0	0	9	0	0	0	0	0	0	7	0	0	0
<i>Deyeuxia decipiens</i>	N	P	G	C3	0	0	6	4	5	0	0	0	0	0	0	0	1	0	0	0
<i>Dianella caerulea</i>	N	P	H	-	0	0	4	0	0	8	0	0	0	0	0	0	0	0	0	0
<i>Dianella longifolia</i>	N	P	H	-	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dianella rara</i>	N	P	H	-	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dianella revoluta</i>	N	P	H	-	0	0	0	1	0	3	0	0	0	0	0	0	0	0	0	0
<i>Dichanthium sericeum</i>	N	P	G	C4	9	0	0	9	9	9	6	0	9	9	9	9	5	4	0	9
<i>Dichelachne micrantha</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0

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					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Dichondra repens</i>	N	P	H	C3	0	0	0	0	0	9	9	7	0	0	0	0	0	0	9	0
<i>Digitaria brownii</i>	N	P	G	C4	0	0	0	8	0	0	0	0	8	0	0	0	0	0	0	0
<i>Digitaria diffusa</i>	N	P	G	C4	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Digitaria divaricatissima</i>	N	P	G	C4	5	2	0	0	0	2	0	0	4	0	0	0	0	0	0	0
<i>Dodonaea viscosa</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
<i>Echinochloa colona</i>	N	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Echinochloa esculenta</i>	E	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Einadia hastata</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Einadia nutans v linifolia</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0
<i>Einadia nutans v nutans</i>	N	P	H	C3	0	0	0	5	7	0	0	9	8	0	0	6	0	9	9	0
<i>Einadia polygonoides</i>	N	SL	H	C3	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0
<i>Eleocharis acuta</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis blakeana</i>	N	P	G	C3	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis cylindrostachys</i>	N	P	G	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eleocharis pallens</i>	N	P	G	C3	0	0	0	0	2	0	0	0	0	0	9	4	0	0	0	0
<i>Eleusine tristachya</i>	E	A	G	C4	0	0	0	0	0	0	0	6	0	0	0	6	0	0	0	
<i>Enchylaena tomentosa</i>	N	P	S	C3	0	0	0	0	7	0	0	8	8	0	0	0	0	0	0	
<i>Enneapogon nigricans</i>	N	P	G	C4	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	
<i>Enneapogon sp. 1</i>	N	P	G	C4	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Enteropogon acicularis</i>	N	P	G	C4	0	0	0	0	6	0	7	0	0	6	7	0	7	0	0	4
<i>Enteropogon ramosus</i>	N	P	G	C4	0	2	0	9	5	0	0	0	0	0	0	5	5	0	9	
<i>Epaltes australis</i>	N	P	H	-	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Eragrostis brownii</i>	N	P	G	C4	4	0	0	0	0	6	6	6	5	0	0	0	0	4	0	0
<i>Eragrostis cilianensis</i>	E	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0
<i>Eragrostis curvula</i>	E	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

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					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Eragrostis elongata</i>	N	SL	G	C4	0	4	9	0	0	9	0	0	0	0	0	0	1	0	0	0
<i>Eragrostis lacunaria</i>	N	P	G	C4	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eragrostis leptostachya</i>	N	P	G	C4	0	8	0	0	0	6	0	0	0	0	0	0	6	0	9	0
<i>Eragrostis megalosperma</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eragrostis parviflora</i>	N	A	G	C4	0	0	2	1	0	6	0	0	0	0	0	0	9	0	0*	0
<i>Eragrostis spartinoides</i>	N	P	G	C4	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eragrostis trachycarpa</i>	N	P	G	C4	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0
<i>Eremophila debilis</i>	N	P	S	C3	1	0	0	9	0	0	4	7	0	0	0	0	3	2	0	0
<i>Eremophila longifolia</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eremophila mitchelli</i>	N	P	S	C3	0	2	0	0	0	0	0	0	9	0	0	0	0	0	0	0
<i>Eriochloa crebra</i>	N	P	G	C4	8	8	0	7	9	6	2	5	8	9	9	9	8	8	0	4
<i>Eriochloa procera</i>	N	P	G	C4	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eriochloa pseudoacrotricha</i>	N	SL	G	C4	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0
<i>Eryngium plantagineum</i>	N	SL	H	C3	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus camaldulensis</i>	N	P	T	C3	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus crebra</i>	N	P	T	C3	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus melliodora</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Eucalyptus populnea</i>	N	P	T	C3	4	4	6	3	3	4	4	4	3	4	3	2	6	6	5	4
<i>Eucalyptus tereticornis</i>	N	P	T	C3	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Eulalia aurea</i>	N	P	G	C4	1	2	4	0	1	0	0	0	2	4	9	6	0	4	0	4
<i>Euphorbia sp. 1</i>	E	P	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eustrephus latifolius</i>	N	P	V	-	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0
<i>Evolvulus alsinoides</i>	N	P	H	C3	0	5	0	0	0	1	2	0	7	0	0	0	0	0	0	0
<i>Fimbristylis dichotoma</i>	N	P	G	C4	0	9	0	0	0	0	0	0	0	0	1	0	0	0	0	8
<i>Fimbristylis neilsonii</i>	N	P	G	C4	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0

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<i>Flaveria trinervia</i>	N	A	H	C4	0	0	8	0	0	0	0	0	0	0	9	7	7	0	0	0
<i>Geijera parviflora</i>	N	P	T	C3	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0
<i>Geijera salicifolia</i>	N	P	T	C3	0	0	0	0	0	0	0	0	2	0	0	0	0	7	0	0
<i>Glandularia aristigera</i>	E	P	H	C3	9	9	0	0	0	9	9	9	9	8	0	0	0	6	8	8
<i>Glossocardia bidens</i>	N	P	H	C4	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
<i>Glycine latifolia</i>	N	P	H	C3	0	0	0	0	0	7	3	9	0	0	0	0	0	9	0	0
<i>Glycine tabacina</i>	N	P	H	C3	9	0	0	9	4	9	9	9	9	0	0	9	6	9	9	8
<i>Glycine tomentella</i>	N	P	H	C3	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gnaphalium pensylvanicum</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gnaphalium sphaericum</i>	N	A	H	C3	0	0	0	1	0	0	0	0	0	0	1	0	0	0	1	0
<i>Gomphocarpus physocarpus</i>	E	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0
<i>Gomphrena celosioides</i>	E	SL	H	C4	0	5	1	5	0	4	4	0	7	0	0	0	0	0	0	0
<i>Goodenia bellidifolia</i>	N	P	H	C3	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Goodenia fascicularis</i>	N	P	H	C3	0	0	2	7	0	0	0	0	0	8	0	0	0	0	0	0
<i>Goodenia glabra</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Goodenia sp. 1</i>	N	P	H	C3	0	0	0	9	0	0	0	0	0	0	0	0	5	0	0	0
<i>Goodenia sp. 2</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
<i>Goodenia sp. 3</i>	N	P	H	C3	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Haloragis stricta</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	2	0	4	0	0	0	0
<i>Heliotropium amplexicaule</i>	E	P	H	C3/C4	4	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
<i>Heteropogon contortus</i>	N	P	G	C4	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Hibiscus trionum</i>	N	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Iseilema membranaceum</i>	N	A	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Ixiolaena brevicompta</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Ixiolaena leptolepis</i>	N	P	H	C3	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

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					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Jasminum suavissimum</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0
<i>Juncus sp.</i>	-	P	G	C3	0	0	7	0	0	0	0	0	0	0	3	0	0	0	0	0
<i>Lactuca serriola</i>	E	SL	H	C3	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0
<i>Lepidium pseudohyssopifolium</i>	N	SL	H	C3	0	0	0	2	0	1	1	4	4	0	0	0	0	7	2	9
<i>Leptochloa divaricatissima</i>	N	P	G	C3	0	0	0	0	5	0	0	0	0	7	9	9	6	0	0	9
<i>Lomandra confertifolia</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lomandra filiformis</i>	N	P	H	-	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lomandra longifolia</i>	N	P	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lomandra multiflora</i>	N	P	H	-	0	0	0	0	0	1	0	0	0	2	0	0	3	4	9	0
<i>Lycium ferocissimum</i>	E	P	S	C3	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0
<i>Maireana aphylla</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Maireana microphylla</i>	N	P	S	C3	0	0	0	0	0	0	0	9	0	0	0	0	0	4	7	0
<i>Malvastrum americanum</i>	E	SL	H	C3	0	0	0	0	0	0	8	7	6	0	0	0	0	5	0	0
<i>Malvastrum coromandelianum</i>	N	SL	H	C3	0	0	0	0	0	0	5	6	3	3	0	0	0	0	0	1
<i>Marsilea costulifera</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
<i>Marsilea drummondii</i>	N	P	H	C3	0	0	8	3	9	0	0	0	0	0	8	7	0	0	0	0
<i>Medicago minima</i>	E	A	H	C3	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0
<i>Medicago sp 1.</i>	E	A	H	C3	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
<i>Megathyrsus maximus</i>	E	P	G	C4	0	9	0	0	0	3	4	7	0	0	0	1	0	1	7	0
<i>Melinis repens</i>	E	SL	G	C4	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Mentha satuireioides</i>	N	P	H	C3	3	0	0	0	0	0	0	0	0	0	1	0	2	0	0	9
<i>Minuria integerrima</i>	N	P	H	-	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
<i>Monachather paradoxus</i>	N	P	G	C3	0	0	0	0	0	0	0	0	7	0	0	0	0	9	3	0
<i>Neobassia proceriflora</i>	N	SL	S	C3	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0
<i>Neptunia gracilis</i>	N	P	H	C3	5	0	0	5	7	0	0	1	7	7	7	0	0	0	0	9

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site															
					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Notelaea linearis</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
<i>Nyssanthes erecta</i>	N	SL	H	C3	0	0	0	9	3	7	0	0	7	0	0	0	0	0	9	0
<i>Olearia sp.</i>	N	A	S	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0
<i>Opuntia aurantiaca</i>	E	P	H	CAM	0	5	0	0	7	2	1	0	0	0	0	0	7	4	2	3
<i>Opuntia tomentosa</i>	E	P	S	CAM	2	9	0	0	0	0	3	7	3	1	0	5	0	7	3	1
<i>Oxalis corniculata v. corniculata</i>	E	P	H	C3	0	0	0	9	0	1	3	8	1	4	7	0	0	0	0	0
<i>Oxalis perennans</i>	N	P	H	C3	0	9	0	9	6	0	0	0	0	9	3	9	6	8	0	9
<i>Panicum buncei</i>	N	P	G	C4	0	0	0	0	4	0	0	0	0	0	0	0	5	0	0	0
<i>Panicum decompositum</i>	N	P	G	C4	6	0	0	1	7	0	0	0	3	7	9	6	6	0	0	0
<i>Panicum effusum</i>	N	P	G	C4	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0
<i>Panicum queenslandicum</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
<i>Panicum simile</i>	N	P	G	C4	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Panicum spp.</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Parsonsia ventricosa</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0
<i>Paspalidium caespitosum</i>	N	P	G	C4	0	0	9	8	9	9	0	0	7	4	7	0	9	9	0	9
<i>Paspalidium constrictum</i>	N	P	G	C4	0	1	0	0	0	3	0	0	9	0	0	0	9	8	0	0
<i>Paspalidium distans</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0
<i>Paspalidium globoideum</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	0	8	0	0	0	0	9
<i>Paspalum dilatatum</i>	E	P	G	C4	0	0	0	0	0	0	0	0	0	0	7	2	0	0	0	0
<i>Phyla canescens</i>	E	P	H	C3	0	0	0	0	4	0	0	0	0	5	0	0	4	0	0	9
<i>Phyllanthus virgatus</i>	N	P	H	C3	8	0	0	4	3	0	0	0	0	8	8	5	5	0	0	5
<i>Physalis lanceifolia</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Physalis longifolia</i>	E	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pimelea neo-anglica</i>	N	P	S	C3	0	1	0	6	0	0	2	5	5	0	4	0	0	0	0	0
<i>Pittosporum angustifolium</i>	N	P	S	C3	0	0	0	0	3	0	0	0	0	0	0	0	0	3	0	2

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	
<i>Plantago lanceolata</i>	E	SL	H	C3	0	0	0	0	0	2	2	0	0	0	0	0	0	3	0*	0	
<i>Poaceae sp 1</i>	-	-	G	-	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	
<i>Polygonum aviculare</i>	E	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0	0	
<i>Portulaca filifolia</i>	N	A	H	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Portulaca oleracea</i>	N	SL	H	C4	0	0	0	5	0	0	0	9	0	0	0	0	0	0	0	0	
<i>Portulaca pilosa</i>	E	A	H	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Pratia purpurascens</i>	N	A	H	C3	4	0	2	2	0	0	0	4	1	3	2	6	0	0	0		
<i>Pycnosorus chrysanthes</i>	N	SL	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Racosperma striatifolium</i>	N	P	S	C3	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Ranunculus inundatus</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0		
<i>Rapistrum rugosum</i>	E	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0		
<i>Rhynchosia minima</i>	N	P	H	C3	9	0	0	9	0	0	7	0	7	0	0	0	8	0	0		
<i>Rostellularia adscendens</i>	N	P	H	C3	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0		
<i>Rumex brownii</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
<i>Sclerolaena birchii</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
<i>Sclerolaena muricata v muricata</i>	N	P	H	C3	0	0	0	0	0	0	5	0	0	0	0	2	1	0			
<i>Sclerolaena muricata v semiglabra</i>	N	P	H	C3	0	0	3	2	0	0	0	5	0	0	0	0	1	8			
<i>Senecio hispidulus</i>	N	P	H	C4	0	0	0	0	4	0	0	0	0	0	0	0	0	0			
<i>Senecio quadridentatus</i>	N	P	H	C4	0	0	0	0	0	0	0	0	0	0	1	0	0	0			
<i>Sesbania cannabina</i>	N	A	S	C4	0	0	0	3	0	0	0	0	0	0	0	0	0	0			
<i>Setaria pumila</i>	E	A	G	C4	0	0	0	0	9	0	0	0	0	1	3	0	0	0			
<i>Sida corrugata</i>	N	P	H	C3	0	7	0	0	0	5	0	4	4	0	0	0	0	0			
<i>Sida cunninghamii</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0			
<i>Sida hackettiana</i>	N	P	H	C3	0	5	0	0	0	0	0	0	0	0	0	0	9	0			
<i>Sida rhombifolia</i>	E	P	S	C3	0	0	0	0	0	1	3	0	0	4	0	9	3	0			

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site																
					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	
<i>Sida rohlenae</i>	N	P	S	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Solanum ellipticum</i>	N	P	H	C3	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0
<i>Solanum nodiflorum</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	
<i>Solanum sp 1.</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Solanum sp 2.</i>	N	P	H	C3	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	
<i>Sonchus oleraceus</i>	E	A	H	C3	0	0	0	5	0	0	0	0	8	8	0	9	0	0	0	3	
<i>Sorghum halepense</i>	E	P	G	C4	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	
<i>Spergularia rubra</i>	E	SL	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0	
<i>Sporobolus caroli</i>	N	SL	G	C4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Sporobolus spp.</i>	N	P	G	C4	7	9	3	8	4	9	6	1	8	9	0	0	0	0	0	0	
<i>Stellaria pallida</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Swainsona galegifolia</i>	N	P	H	C3	0	6	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
<i>Swainsona sp. 1</i>	N	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Swainsona sp. 3</i>	N	P	H	C3	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	
<i>Symphotrichum subulatum</i>	E	SL	H	C3	0	0	0	0	0	4	0	0	0	0	1	0	4	0	0	3	
<i>Tetragonia eremaea</i>	N	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Themeda avenacea</i>	N	P	G	C4	0	0	0	0	0	0	0	0	0	2	0	4	0	0	0	0	
<i>Themeda triandra</i>	N	P	G	C4	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	3	
<i>Triraphis mollis</i>	N	SL	G	C4	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Vachellia farnesiana</i>	E	P	S	C3	3	0	0	2	4	0	0	0	0	0	2	0	0	0	2		
<i>Verbena bonariensis</i>	E	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	
<i>Verbena litoralis</i>	E	P	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Verbena officinalis</i>	E	P	H	C3	0	0	0	8	0	0	1	0	8	0	0	7	6	5	0	3	
<i>Vittadinia pustulata</i>	N	A	H	C3	0	8	0	0	0	9	4	0	0	0	0	0	0	0	0	0	
<i>Vittadinia sulcata</i>	N	A	H	C3	0	0	0	0	0	4	8	9	4	0	0	0	0	0	7	0	

Scientific Name	Origin	Life Cycle	Life form	Photosynth. pathway	Site															
					18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
<i>Wahlenbergia gracilis</i>	N	P	H	C3	4	6	0	0	1	1	1	0	2	3	4	0	0	0	0	0
<i>Xanthium strumarium</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Zinnia peruviana</i>	E	A	H	C3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 1</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 2</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 3</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 4</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 5</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 6</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 7</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 8</i>	-	-	H	-	0	0	0	0	0	4	0	0	0	0	0	0	0	0	9	0
<i>Unknown sp. 9</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 10</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Unknown sp. 11</i>	-	-	H	-	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0

Appendix E2 Summary data of functional group species richness (SR) and richness transition ratios

Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	
Total SR	42	58	55	46	58	64	52	55	66	54	46	48	66	50	60	58	40	
Exotic SR	9	12	8	7	9	10	9	15	6	16	9	7	13	12	9	1	7	
Native SR	32	46	45	39	47	53	42	40	60	35	37	41	53	38	50	54	32	
Short-lived SR	7	14	9	5	12	12	9	6	9	17	8	5	15	11	12	10	4	
Perennial SR	28	32	39	35	37	44	33	41	47	31	26	36	45	35	39	33	30	
Tree SR	1	1	1	1	1	1	2	1	3	1	1	2	2	1	3	3	1	
Shrub SR	5	6	2	3	3	5	7	5	5	2	3	6	5	4	4	6	3	
Herb SR	19	31	25	25	32	32	18	29	31	36	21	20	35	22	27	27	23	
Graminoid SR	17	20	27	17	22	26	25	20	27	15	11	20	24	23	25	22	13	
C3 SR	25	31	29	24	36	32	24	31	27	31	25	25	40	29	34	27	22	
C4 SR	13	22	23	17	18	27	23	18	31	17	11	19	23	20	23	23	12	
Exotic:Native	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.4	0.1	0.5	0.2	0.2	0.2	0.3	0.2	0.0	0.2	
Shortlived:Perennial	0.3	0.4	0.2	0.1	0.3	0.3	0.3	0.1	0.2	0.5	0.3	0.1	0.3	0.3	0.3	0.3	0.1	
C4:C3	0.5	0.7	0.8	0.7	0.5	0.8	1.0	0.6	1.1	0.5	0.4	0.8	0.6	0.7	0.7	0.9	0.5	
Site	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	Mean	S.E.
Total SR	40	55	26	58	45	62	44	40	64	38	36	47	46	60	40	46	50.5	1.7
Exotic SR	5	9	1	8	3	10	12	13	11	6	5	14	9	13	9	8	8.9	0.6
Native SR	34	46	24	48	42	51	32	27	52	31	30	32	37	45	30	38	40.7	1.6
Short-lived SR	1	6	7	10	4	11	9	11	16	4	7	8	10	5	9	4	8.7	0.7
Perennial SR	34	39	13	37	36	40	30	23	39	29	20	33	31	39	27	34	33.8	1.2
Tree SR	1	2	2	1	1	5	1	1	2	1	2	1	1	6	1	2	1.7	0.2
Shrub SR	4	6	1	5	5	2	5	7	6	1	3	3	1	9	5	5	4.3	0.3
Herb SR	19	19	9	30	21	33	24	21	33	20	15	21	21	24	22	22	24.5	1.1
Graminoid SR	16	28	14	22	18	22	14	11	23	16	16	22	23	20	13	17	19.7	0.8
C3 SR	22	21	13	32	26	31	27	29	35	21	24	26	25	38	25	27	27.7	1.0
C4 SR	13	28	10	21	15	24	14	10	25	13	11	18	18	15	9	15	18.2	1.0
Exotic:Native	0.1	0.2	0.0	0.2	0.1	0.2	0.4	0.5	0.2	0.2	0.2	0.4	0.2	0.3	0.3	0.2	0.2	0.0
Shortlived:Perennial	0.0	0.2	0.5	0.3	0.1	0.3	0.3	0.5	0.4	0.1	0.4	0.2	0.3	0.1	0.3	0.1	0.3	0.0
C4:C3	0.6	1.3	0.8	0.7	0.6	0.8	0.5	0.3	0.7	0.6	0.5	0.7	0.7	0.4	0.4	0.6	0.7	0.0

Appendix F Summary of stand structural variables (cover %)

Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	
trees >30m	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
trees 10-30m	8	12	25	10	20	10	35	25	5	30	30	8	30	8	8	12	5	
trees <10m	20	35	1	5	8	3	3	6	30	8	3	20	3	20	25	20	12	
shrubs > 2m	0	4	0	0	0	0	8	0	3	0	0	0	0	0	3	1	0	
shrubs <2m	2	2	0	5	0	1	5	0	1	0	1	1	1	0	1	2	0	
herbs	15	5	8	8	8	2	3	3	2	5	10	5	3	12	3	2	2	
graminoids	75	70	85	80	85	85	80	85	75	55	40	75	85	90	60	65	80	
coarse litter	1	4	4	1	2	2	2	2	5	5	2	3	2	1	5	5	3	
woody litter	3	1	2	3	3	2	2	1	1	3	3	2	3	2	3	3	2	
leaf litter	8	5	5	8	3	5	3	5	3	20	12	3	6	10	35	8	20	
bare ground	10	15	10	8	8	5	3	2	20	1	4	4	4	1	8	12	3	
cryptogams	3	2	1	1	1	1	1	0	0	1	0	0	0	0	0	0	0	
Site	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	mean	S.E.
trees >30m	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0
trees 10-30m	5	5	30	10	8	15	3	35	3	0	15	5	10	10	23	5	14.0	1.8
trees <10m	15	20	5	5	1	20	15	1	25	28	5	5	7	35	2	8	12.7	1.8
shrubs > 2m	0	0	1	1	0	1	0	0	0	0	1	0	0	0	0	0	0.7	0.3
shrubs <2m	1	1	0	0	1	0	0	1	1	0	1	4	0	1	8	1	1.3	0.3
herbs	5	8	3	2	5	15	10	3	5	2	2	2	3	20	4	3	5.7	0.8
graminoids	90	70	60	60	85	80	70	90	90	65	95	90	80	70	75	65	75.9	2.2
coarse litter	2	3	10	4	1	4	0	4	2	3	0	2	4	1	2	5	2.9	0.3
woody litter	2	2	5	5	1	3	2	2	2	12	2	2	2	2	5	2	2.7	0.3
leaf litter	10	30	8	12	3	18	30	1	12	10	2	5	10	5	8	5	9.9	1.5
bare ground	3	10	2	8	2	1	10	1	2	5	1	2	8	2	12	12	6.0	0.8
cryptogams	0	2	1	1	0	2	0	0	0	0	0	0	0	0	0	0	0.5	0.1

Appendix G Summary of densities within size (age) classes and foliage index of *Eucalyptus populnea*.

Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	
Seedling (indiv.)/ha	80	0	0	0	80	20	0	0	0	0	100	60	0	520	40	120	20	
Sapling (indiv.)/ha	240	280	0	40	20	100	0	40	20	200	80	440	80	120	120	100	60	
Mature (indiv.)/ha	100	20	100	80	20	200	20	80	260	80	40	40	0	60	220	380	40	
Matriarch (indiv.)/ha	20	20	60	40	40	0	60	40	20	140	40	40	40	20	20	0	120	
Foliage index	70.0	65.0	51.4	72.5	52.5	67.5	77.0	57.8	73.8	75.0	68.3	50.0	85.0	86.7	74.6	50.9	43.1	
Site	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	mean	S.E.
Seedling (indiv.)/ha	80	0	0	60	0	80	20	0	20	0	0	20	240	200	40	20	55.2	17.7
Sapling (indiv.)/ha	400	20	20	120	0	120	20	0	20	20	40	0	60	220	0	20	91.5	19.6
Mature (indiv.)/ha	40	220	120	60	0	40	200	0	180	160	100	0	20	20	80	140	94.5	15.6
Matriarch (indiv.)/ha	20	20	80	40	60	40	0	120	60	20	40	100	80	80	60	0	46.7	6.3
Foliage index	60.0	63.0	53.1	50.0	46.7	68.3	88.9	45.8	65.7	73.3	50.0	67.0	45.0	46.0	57.1	75.0	62.9	2.3

Appendix H Summary of soil data

Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	
pH	6.3	6.4	6.7	6.5	6.4	6.2	6.0	6.4	6.3	6.3	6.5	6.6	6.1	6.0	5.6	6.8	6.2	
K (mg/kg)	279.0	88.0	261.0	305.0	240.0	94.0	275.0	301.0	161.0	146.0	409.0	494.0	350.0	363.0	144.0	130.0	516.0	
N (mg/kg)	3.0	4.0	7.0	5.0	2.0	2.0	26.0	9.0	3.0	6.0	6.0	15.0	22.0	9.0	2.0	6.0	10.0	
P (mg/kg)	15.0	4.0	35.0	8.0	9.0	12.0	12.0	10.0	3.0	25.0	76.0	53.0	61.0	117.0	5.0	7.0	147.0	
Ca (mg/kg)	2060.0	752.0	2830.0	2510.0	2210.0	851.0	1710.0	3320.0	1460.0	1250.0	4030.0	4070.0	2390.0	2470.0	743.0	1600.0	3550.0	
Mg (mg/kg)	1210.0	451.0	788.0	1470.0	843.0	427.0	810.0	1310.0	918.0	235.0	1540.0	1590.0	807.0	705.0	253.0	414.0	1880.0	
Na (mg/kg)	177.0	135.0	102.0	213.0	88.0	70.0	106.0	200.0	226.0	27.0	197.0	262.0	103.0	73.0	52.0	221.0	161.0	
OC (%)	1.6	1.1	1.7	1.4	1.1	1.7	1.9	1.8	1.2	1.1	2.2	3.3	>4.0	2.5	1.5	1.6	>4	
CEC (meq/100g)	21.8	8.3	21.9	26.5	19.1	8.4	16.5	29.2	16.3	8.7	34.9	36.0	20.0	19.5	6.4	12.8	35.4	
Site	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	mean	S.E.
pH	6.5	6.5	5.6	6.6	6.6	6.6	5.9	6.3	6.5	6.1	6.3	6.0	6.0	5.8	6.4	6.5	6.3	0.1
K (mg/kg)	230.0	184.0	174.0	531.0	400.0	106.0	127.0	434.0	506.0	279.0	286.0	555.0	233.0	449.0	396.0	264.0	294.2	24.2
N (mg/kg)	2.0	4.0	4.0	19.0	10.0	6.0	3.0	18.0	14.0	12.0	3.0	25.0	9.0	11.0	4.0	5.0	8.7	1.2
P (mg/kg)	8.0	7.0	5.0	35.0	127.0	4.0	5.0	62.0	37.0	30.0	15.0	150.0	21.0	98.0	79.0	19.0	39.4	7.7
Ca (mg/kg)	2420.0	1190.0	791.0	3100.0	2970.0	1020.0	1670.0	2060.0	3070.0	2840.0	2740.0	5400.0	3100.0	3080.0	2320.0	2960.0	2379.9	188.7
Mg (mg/kg)	1110.0	257.0	719.0	1570.0	1010.0	489.0	148.0	617.0	1170.0	1150.0	1560.0	1400.0	860.0	958.0	618.0	839.0	912.9	79.6
Na (mg/kg)	158.0	46.0	107.0	237.0	187.0	144.0	31.0	82.0	321.0	216.0	326.0	262.0	178.0	280.0	53.0	369.0	163.9	15.9
OC (%)	2.1	0.8	2.8	2.6	3.3	1.3	0.9	2.3	3.4	3.1	1.5	>4.0	3.8	>4	3.1	2.7	2.0	0.2
CEC (meq/100g)	22.7	8.8	10.9	31.0	25.1	10.1	10.0	16.9	27.8	25.5	28.9	41.2	24.0	25.7	18.0	24.1	21.0	1.6

Appendix I Summary of environmental and spatial data for each site

Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
Patch shape	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
Patch size (ha)	75.4	18.9	12.2	91.4	14.9	29.9	9.5	43.7	59.1	6.6	19.3	4.5	6.1	9.2	46.9	339.8	9.2
P:A	0.1	0.1	0.2	0.1	0.1	0.1	0.2	0.2	0.1	0.2	0.1	0.2	0.2	0.2	0.1	0.0	0.2
PRV2 (%)	7.4	10.9	3.9	5.2	4.5	1.0	8.0	11.1	7.1	40.7	11.8	1.0	1.0	7.8	19.2	10.5	5.6
Stem density (/500m²)	14.0	4.6	20.4	8.3	23.8	6.1	16.2	9.4	22.5	25.3	14.3	27.4	34.0	8.3	13.3	15.4	22.5
Clearing	1	1	2	1	0	1	0	0	1	1	0	1	1	0	0	0	1
Erosion	0	0	0	0	1	1	0	0	1	1	1	1	0	0	1	1	1
Soil Compaction	1	0	0	0	0	1	0	0	1	1	2	1	1	0	1	1	0
Cowpat density (/500m²)	0	0	0	0	0	0	0	0	0	0	0	4	2	0	0	6	0
Stags (/500m²)	0	5	0	0	0	0	2	0	0	0	0	2	2	0	1	2	0
Cropping (%)	72.5	40.9	47.9	87.5	77.3	57.2	11.1	53.2	12.6	8.8	56.0	91.8	91.5	40.2	24.8	8.0	80.9
Grazing (%)	21.1	46.0	44.9	5.5	14.4	20.2	76.4	41.9	80.6	90.8	37.6	4.9	5.2	56.4	73.7	48.6	9.3
Rainfall (mm)	593.5	637.5	611.8	603.4	611.8	603.4	625.7	611.8	617.2	625.7	593.5	603.4	603.4	603.4	599.7	661.3	607.6
Groundwater ('09) (m)	-17.6	-18.2	-29.3	-23.9	-28.9	-24.4	-21.4	-21.2	-22.2	-20.3	-13.4	-23.7	-23.7	-12.2	-20.0	-20.9	-22.6
River distance (m)	43.7	159.4	542.4	250.5	587.2	281.3	80.4	280.8	449.5	122.1	310.5	1087.5	1019.3	233.2	123.4	470.0	98.4
Nearest patch (m)	947.0	1526.8	1202.5	2666.4	1356.2	1437.6	524.0	541.7	1328.0	161.4	1095.0	177.3	177.3	359.8	1016.8	1702.3	1354.2

Site	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	mean	S.E.
Patch shape	0.0	1.0	0.0	0.0	1.0	1.0	1.0	1.0	1.0	1.0	0.0	0.0	0.0	1.0	0.0	1.0	0.3	0.1
Patch size (ha)	70.2	171.7	13.5	28.0	25.7	11.4	6.6	5.1	20.0	13.9	24.1	2.4	14.0	9.9	3.6	13.6	37.3	11.2
P:A	0.1	0.1	0.2	0.1	0.1	0.1	0.2	0.2	0.1	0.1	0.2	0.4	0.2	0.1	0.2	0.1	0.2	0.0
PRV2 (%)	5.1	35.7	12.3	8.2	8.1	5.8	14.1	10.2	22.7	5.1	4.5	29.8	7.8	20.9	9.5	8.5	11.1	1.6
Stem density (/500m ²)	5.8	10.2	12.3	14.1	10.5	35.8	11.6	26.6	11.3	26.1	10.1	15.1	19.9	17.1	16.5	5.4	16.2	1.4
Clearing	0	1	1	0	1	1	0	1	2	0	1	1	2	0	2	2	0.8	0.1
Erosion	1	0	3	1	1	1	2	0	1	2	1	1	1	1	1	1	0.8	0.1
Soil Compaction	1	1	1	2	1	0	1	0	0	0	1	1	1	0	1	2	0.7	0.1
Cowpat density (/500m ²)	0	0	0	18	0	0	1	0	19	0	0	0	2	0	0	35	2.6	1.3
Stags (/500m ²)	2	7	1	0	0	0	0	3	0	0	0	0	3	1	3	3	1.1	0.3
Cropping (%)	66.7	9.6	0.0	84.4	81.1	0.7	45.2	32.9	50.0	76.2	87.3	38.3	54.2	19.0	29.5	26.8	47.4	5.1
Grazing (%)	19.1	90.3	98.0	13.7	11.9	94.4	49.9	63.6	45.2	20.5	3.6	59.5	39.4	78.1	67.5	72.8	45.6	5.2
Rainfall (mm)	603.4	617.2	609.4	603.4	603.4	637.5	658.6	658.6	603.4	593.5	603.4	603.4	637.5	658.6	658.6	613.0	617.5	3.7
Groundwater ('09) (m)	-22.6	-21.9	-21.4	-31.0	-30.2	-20.4	-18.7	-17.8	-13.3	-17.8	-31.5	-15.8	-18.8	-16.8	-17.6	-14.0	-21.0	0.9
River distance (m)	261.5	71.2	152.7	1050.2	472.9	35.3	921.8	269.4	270.8	295.7	367.6	0.0	9.2	725.2	0.0	422.5	347.4	54.1
Nearest patch (m)	2795.3	678.2	466.2	882.7	472.1	248.8	656.3	675.4	1054.0	800.4	1515.6	813.1	144.9	320.0	675.4	725.4	924.2	112.1

Appendix J Summary of the diameter at breast height (DBH) of all trees and their foliage index.

Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)
1	6.0		3	19.7	70	8	5.1		10	9.5	
1	4.5		3	28.0	80	8	23.2	40	11	3.0	
1	82.8	70	3	23.6	40	8	21.3	97	11	2.7	
1	11.5		3	28.0	30	8	27.4	2	11	30.2	65
1	3.8		4	9.5		8	5.4		11	6.4	
1	4.1		4	43.0	65	8	52.5	65	11	21.3	90
1	4.8		4	4.1		8	19.1		11	86.3	50
1	12.4		4	11.1		9	88.8	20	11	12.7	
1	5.7		4	16.2		9	19.7	80	11	9.2	
1	8.3		4	14.0		9	13.7		11	0.4	
1	10.2		4	11.8		9	14.6		11	0.1	
1	1.9		4	50.9	80	9	22.0	75	11	0.2	
1	4.5		5	1.3		9	25.8	90	11	0.5	
1	4.1		5	106.0	65	9	26.4	95	11	0.4	
1	1.3		5	8.0		9	20.4	90	12	2.5	
1	8.3		5	60.8	40	9	26.1	90	12	8.0	
1	9.5		5	1.0		9	9.5		12	6.7	
1	15.3		5	0.6		9	10.2		12	18.1	
1	10.8		5	1.6		9	17.8		12	8.3	
1	2.9		5	12.1		9	11.5		12	5.1	
1	0.4		6	26.1	40	9	18.8		12	7.6	
1	0.5		6	1.7		9	21.3	50	12	2.5	
2	6.0		6	11.8		10	36.6	65	12	8.0	
2	5.4		6	4.9		10	29.3	80	12	5.4	
2	5.4		6	11.5		10	34.4	65	12	4.8	
2	4.5		6	7.6		10	36.9	75	12	8.3	
2	8.0		6	16.9		10	4.6		12	3.7	
2	7.0		6	2.9		10	36.3	80	12	1.0	
2	6.7		6	18.1		10	25.5	40	12	1.0	
2	5.4		6	18.1		10	18.1		12	7.6	
2	47.7	65	6	16.9		10	8.3		12	5.4	
2	3.8		6	6.5		10	9.2		12	5.1	
2	6.0		6	15.8		10	5.4		12	8.0	
2	8.0		6	18.5		10	8.3		12	9.5	
2	10.2		6	15.0		10	4.7		12	4.5	
2	4.8		6	2.9		10	31.5	90	12	7.0	
2	8.3		7	45.5	80	10	5.8		12	30.0	60
2	3.5		7	37.2	70	10	22.3	-	12	5.1	
3	55.7	60	7	46.5	75	10	30.2	85	12	1.7	
3	63.3	30	7	50.9	70	10	4.1		12	7.0	
3	17.8		7	21.0	90	10	44.2	95	12	4.1	
3	55.4	50	8	31.2	85	10	6.4		12	37.9	40

Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)
12	11.1		15	1.7		16	23.6	40	19	20.4	95
13	5.4		15	4.5		16	0.6		19	17.2	
13	4.1		15	2.9		16	0.3		19	25.1	30
13	2.9		15	19.1	95	16	16.9		19	11.1	
13	117.5	75	15	28.0	50	16	29.3	75	19	16.9	
13	55.1	90	15	13.4		17	37.2	30	19	20.7	70
13	3.2		15	13.7		17	1.0		19	17.5	
14	0.4		15	13.7		17	38.8	75	19	16.2	
14	0.4		15	23.2	50	17	12.7		19	30.2	60
14	0.3		15	30.6	70	17	73.8	40	19	12.7	
14	0.4		15	2.9		17	9.5		19	17.0	
14	5.4		15	22.3	90	17	6.4		19	24.2	60
14	19.4	95	15	6.4		17	33.4	40	19	5.7	
14	1.6		15	9.9		17	7.0		20	24.2	40
14	0.4		15	1.6		17	25.5	40	20	9.7	
14	0.5		15	20.4	95	17	43.0	85	20	33.4	70
14	0.4		15	16.6		17	31.5	5	20	19.4	40
14	0.5		15	18.5		18	7.0		20	33.1	65
14	60.8	70	15	19.7	80	18	51.2	60	20	39.5	75
14	5.1		16	2.9		18	8.9		20	31.2	85
14	0.3		16	21.0	90	18	1.4		20	17.8	
14	0.4		16	22.3	60	18	6.0		20	12.4	
14	0.3		16	10.5		18	5.7		20	27.4	40
14	0.4		16	23.9	30	18	11.1		20	23.2	10
14	0.4		16	12.1		18	2.2		21	51.9	50
14	0.3		16	23.6	70	18	0.6		21	7.6	
14	5.1		16	7.0		18	5.7		21	56.0	60
14	0.3		16	17.2		18	3.2		21	3.5	
14	0.3		16	8.9		18	3.5		21	4.1	
14	0.3		16	11.5		18	6.7		21	29.9	50
14	0.7		16	23.2	40	18	3.8		21	4.1	
14	0.5		16	2.5		18	4.5		21	3.8	
14	0.3		16	1.3		18	10.2		21	1.0	
14	0.5		16	1.6		18	8.0		21	6.0	
14	0.7		16	15.6		18	8.9		21	20.1	40
14	2.5		16	22.0	5	18	3.8		21	12.4	
14	9.9		16	1.0		18	6.7		21	0.3	
14	17.5		16	29.3	60	18	5.1		21	0.2	
14	3.2		16	3.5		18	6.4		22	42.3	50
14	0.7		16	18.5		18	3.5		22	46.8	40
14	0.5		16	0.6		18	9.2		22	51.9	50
14	0.2		16	19.4	5	18	0.3		23	8.6	
14	26.1	95	16	21.3	85	18	0.3		23	9.2	
15	3.5		16	14.6		18	9.9		23	0.6	

Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)	Site	DBH (cm)	FI (%)
23	9.2		27	17.5		31	1.0	
23	1.0		27	7.3		31	1.0	
23	0.6		27	98.0	50	31	6.4	
23	10.5		27	11.8		31	1.6	
23	8.0		27	23.6	80	31	54.1	30
23	2.5		27	12.7		31	41.4	85
23	24.8	95	27	10.8		31	2.5	
23	1.3		27	19.4	90	31	0.3	
23	68.4	60	27	10.2		31	0.3	
23	112.4	50	28	36.9	45	31	49.0	50
23	7.0		28	25.5	55	31	2.9	
24	26.4	75	28	6.4		31	24.8	10
24	20.7	97	28	19.7	45	31	53.8	55
24	27.7	95	28	16.2		31	9.2	
24	13.4		28	44.6	70	31	0.6	
24	19.4	90	28	28.0	40	31	0.6	
24	19.4	75	28	8.1		31	4.1	
24	18.5		28	26.1	55	31	1.3	
24	9.9		29	42.7	60	31	7.0	
24	21.3	95	29	30.9	70	31	1.3	
24	18.8		29	1.3		31	7.3	
24	24.5	95	29	49.0	40	31	2.2	
24	0.3		29	52.2	75	31	5.1	
25	65.6	25	29	41.4	90	31	2.9	
25	45.2	50	30	20.2	40	31	3.8	
25	43.6	60	30	42.0	45	31	1.0	
25	67.8	50	30	9.7		32	63.7	85
25	42.7	60	30	0.2		32	26.1	10
25	31.8	30	30	1.6		32	26.1	95
26	31.8	70	30	51.6	5	32	25.1	70
26	13.1		30	1.0		32	30.9	45
26	11.1		30	1.0		32	0.6	
26	6.4		30	0.2		32	0.3	
26	31.8	55	30	1.6		32	53.2	45
26	18.1		30	1.3		32	26.1	50
26	25.8	90	30	53.8	60	33	1.3	
26	21.0	80	30	42.3	75	33	18.8	
26	12.1		30	3.2		33	21.0	80
26	0.2		30	1.3		33	29.9	65
26	30.9	40	30	0.3		33	11.5	
26	22.9	90	30	1.3		33	28.0	80
26	17.8		30	6.0		33	17.8	
26	22.0	35	30	0.2		33	8.9	
27	15.0		30	0.1		33	13.1	