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Riparian woodland condition in relation to lippia (*Phyla canescens* (Kunth) Greene) and fire management, southern Queensland

A thesis submitted by

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ABSTRACT

The implementation of inappropriate management regimes encourages alien species invasion into native ecosystems. Disturbances, such as fire, flooding and grazing, create a unique window of opportunity for invasive species to colonise areas not previously invaded. Fire regimes can be changed through the introduction of foreign species and may induce detrimental ecosystem effects including increased tree mortality, an increase in bare ground and further invasion by foreign species.

Phyla canescens (lippia) is an invasive introduced species covering over 5.3 million hectares of the Murray-Darling Basin. Many potential control methods have been attempted. However, the response of lippia to fire as a potential management tool has not previously been studied.

The general question of this research was: what is the effect of a prescribed fire on lippia, the vegetation structural components and the soil seed bank of a vegetation remnant in a highly modified agricultural landscape? This research examined the following specific hypotheses: that there is no difference in the abundance of lippia between burnt and unburnt treatments, six months after a prescribed burn; that there is no difference in the abundance of other vegetative structural components between burnt and unburnt woodlands, six months after a prescribed burn; that the fire has resulted in no change in the seedling emergence of lippia seeds stores in the soil seed bank; and, that the fire has resulted in no change of pseudo-species (not taxonomically identified species) richness within the seed bank.

Ten sites, (5 burnt, 5 unburnt) were sampled within each of the two woodlands: *Eucalyptus camaldulensis* and *Eucalyptus populnea*, six months after a prescribed burn. At each site, a 500 m^2 quadrat was established to determine general disturbance measures (using a 0–4 scoring method) and the abundance and percentage cover of vegetative structural components. Within each 500 m^2 quadrat, the density of the following vegetative structural components were measured: trees > 20 m; trees 10–20 m; trees < 10 m; standing stags > 20 m; standing stags 10–20 m; standing stags < 10 m; stumps; logs 6–13 cm; logs 13–35 cm; logs 35–90 cm; and logs 90–150 cm. The percent cover of each of the following vegetative structural components was also measured: trees; stags; stumps; logs; grasses/sedges; herbs/forbs; lippia; lippia litter; fine litter; bare ground. Within each 500 m^2 quadrat, eight 70 cm^2 sub-quadrats were placed along a 16 m transect to determine the cover abundance of lippia, grasses and forbs at a finer scale. Soil samples were also collected at each site for the glasshouse seed germination trial. Samples were kept in the glasshouse for 14 weeks to determine total germination and pseudo-species richness of each site and treatment. Independent T-tests determined whether there were any significant differences of variables between treatments. The Levene's Test for Homogeneity was used to determine homogeneity.

There were no significant differences in the abundance and cover of lippia or vegetation structural components between burnt and unburnt woodlands, six months after a prescribed burn at either the broad (500 m^2) or finer scale (70 cm^2) (T-test, $p > 0.05$). The only exception to this finding was a significant difference in forb cover between burnt and unburnt *Eucalyptus populnea* woodlands (T-test, $p < 0.05$). There were no significant differences in seedling emergence or pseudo-species richness between burnt and unburnt treatments within each woodland (T-test, $p > 0.05$).

This research shows that fire may not be a suitable control method for lippia invasion. The results found for structural components and the seedling germination trial were consistent with previous literature. Lippia's response to fire in this study and the lack of positive effect that fire had on the invasive weed at St Ruth Reserve, has begun to fill an identified knowledge gap in the control methods for lippia. This study shows that fire may not be a useful management tool for lippia.

DECLARATION

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

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Signature of Candidate

…………………………
Date

Endorsement

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Chapter 1: INTRODUCTION

1.0 General Introduction

Inappropriate management regimes may adversely impact the native vegetation through composition and structural changes (Lindenmayer & Burgman 2005). The implementation of inappropriate management regimes can increase the potential for an ecosystem invasion, especially if the system is highly stressed or sensitive, such as some riparian woodlands (Fisher *et al.* 2009).

Invasion occurs when a species establishes in an ecosystem that is not in its native home range. These species have the potential to have a significant impact on both natural and managed ecosystems because of the lack of native predators (Hobbs & Mooney 2005). Often, new colonising species are disturbance specialists and are able to thrive under disturbed conditions. The r- and K- selection theory and the life histories associated with each selection trait, helps to explain why invaders are successful colonists (MacArthur & Wilson 1967; Pianka 1970), particularly in disturbed ecosystems. Disturbances can create a unique window for invasive species to colonise if the conditions are favourable (Hobbs 1991). Environmental disturbances such as fragmentation (Laurance *et al.* 2000), grazing (Earl 2003), flooding (Elderd 2003) and fire (Masocha *et al.* 2011) often enhance invasion by alien species. This thesis will predominantly focus on fire as a management tool in modified remnant vegetation.

Many Australian plants and ecosystems have evolved to withstand the effects of fire (Pausas & Schwilk 2012). The reproduction of many Australian species, especially eucalypts, requires a particular fire regime to initiate the reproduction process (Penman *et al.* 2009). A fire regime is determined by five main components of fire: intensity, frequency, season, type and extent (Gill 1975; Whelan 1995). If this fire regime is changed through the suppression or introduction of fire, an ecosystem can experience species diversity decline (Bond & Keeley 2005). The change of a fire regime can also have other detrimental ecosystem effects such as increased tree mortality (Pettit & Naiman 2007), increase in bare ground (Kutt & Woinarski 2007) and increased invasion by foreign species (Masocha *et al.* 2011).

When a fire occurs an ideal window of opportunity is created for invasive grasses, forbs, shrubs and woody species to colonise areas previously not invaded. This opportunity is created by the enhancement of light and nutrient availability in the recently burnt ecosystem (Hester & Hobbs 1992). This initiation for invasion can create vast ecosystem changes from just one fire. Invasive species can either increase or decrease the frequency and intensity of a fire due to their ecological properties such as moisture content of plants, chemical compositions and fuel load (Brooks *et al.* 2004). One invasive that is reported to decrease the risk of fire is lippia (*Phyla canescens* (Kunth) Greene) (State Government of Victoria 2011).

Lippia is an invasive species that originated from Argentina and was introduced to Australia in the 1920s as a ‘no-mow grass’ (Leigh & Walton 2004). It has now invaded over 5.3 million hectares of the Murray-Darling Basin as well as much of eastern Australia (Earl 2003). The prostrate species has an extensive root system, which forms a mat that does not allow the emergence of most species (Earl 2003; Crawford 2008). The decline of natural riparian woodland regeneration has been linked to lippia cover, which is now threatening many ecological communities including *Eucalyptus camaldulensis* (Denh.) and *Eucalyptus populnea* (F. Muell.) woodlands (Earl 2003).

Control methods for lippia have been varied and research has included the use of grazing, mechanical disturbance and flooding. The search for a biological control identified four insects, two fungi and a mite as potential controls for lippia (van Klinken & Julien 2010). It has been proposed that *Kuschelina bergi*, a beetle, be introduced to Australia under quarantine conditions for research purposes.

Although there is no evidence of the use of fire to control lippia, the Western Downs Regional Council, in southern Queensland, applied a cool, low intensity controlled fire in an attempt to reduce the impact of lippia within a remnant patch of riparian woodland. This research addresses the general question: What is the effect of a prescribed fire on lippia, vegetation structural components and the soil seed bank of a vegetation remnant in a highly modified agricultural landscape?

This thesis will explore current literature on alien species, theories that help to explain why alien species successfully invade and thrive in disturbed environments, fire and its effects on the ecosystem, *Phyla canescens*' physiology and control methods previously researched. It will then provide a detailed description of two studies into the response of vegetative structural components and the soil seed bank to fire, six months post fire.

1.1 Alien Species Invasion

Biological invasions have caused more extinctions to date than human-induced climate change, second only to land use change (D'Antonio & Vitousek 1992). The increased movement of plants and animals across the globe has led to species being exposed to much higher exchanges of biological material in a relatively short space of time (Mooney 2005). This movement of organisms has led to the invasion of native areas by alien species. A significant issue is that although an invasive species may appear innocent in their natural environment, it can cause ecological chaos when introduced to a new area, due to the absence of native predators and competitors (Mooney 2005).

Invasive plant species in Australia cost the agricultural industry $3.4 billion annually in impact and control costs, with farmers spending $1.6 billion during the 2006–2007 year on the control and prevention of foreign species (Pink 2013). This is more than the combined annual cost of controlling animal pests and alleviating land and soil problems ($1,717 million) (Pink 2013).

A number of theories and frameworks have been proposed or applied to alien species to explain why these species colonise new environments in such an aggressive way. One theory that has been applied to invasive species is the r- and K- selection theory (MacArthur & Wilson 1967; Pianka 1970). The r- and K- selection theory has revealed that successful colonists across taxa have r-selected life histories with an ability to switch between r- and K- selection strategies (Sakai *et al.* 2001). No single organism is clearly r- or K-selected (Pianka 1970). See Table 1 for a summary of r- and K- selection traits.

Table 1: Pianka's correlates of r- and K- selection (Pianka, 1970)

	r-Selection	**K-Selection**
Climate	Variable and/or unpredictable	Fairly constant and/or predictable
Mortality	Often catastrophic, density independent	Density-dependent
Population size	Variable in time, non-equilibrium, well below carrying capacity	Fairly constant in time, equilibrium, at or near carrying capacity
Intra- and interspecific competition	Variable, often lax	Usually keen
Relative abundance	Does not fit broken stick model	Frequently fits broken stick model
Length of life	• Rapid development • High r_{max} • Early reproduction • Small body size • Single reproduction	• Slower development, greater competitive ability • Lower resource thresholds • Delayed reproduction • Larger body size • Repeated reproduction
Leads to…	Productivity	Efficiency

1.1.1 Disturbance and Invasive species

Alien species invasion is generally enhanced by environmental and anthropogenic disturbances (Hobbs 1991). However, there is a careful balance between too many and too few disturbances (Connell 1978). The intermediate disturbance hypothesis examines this phenomena (Connell 1978). If disturbances occur too frequently or infrequently, species diversity decreases (Connell 1978). If a disturbance, such as a fire, happens too frequently, reproduction may take longer than the interval between disturbances, with the ecosystem losing species due to a lack of reproductively mature individuals (Hobbs & Huenneke 1992). If the interval between disturbance events is too long, community diversity will be reduced by the lack of disturbances to create new recruitment (Hobbs & Huenneke 1992; Penman *et al.* 2009).

Therefore, it would be expected that there is a stable point at which diversity would be at a maximum. However, Connell (1978) hypothesised that a community will never achieve equilibrium because disturbances, however large or small, are so common that the ecosystem rarely reaches an ordered state. Species diversity is highest when disturbances are at an intermediate level in terms of frequency and intensity (Connell 1978).

Disturbances can provide undesirable impacts, such as creating a unique window for invasive species to colonise if the environment present is amenable (Hobbs 1991). Quite often new colonising species are disturbance specialists who are able to thrive under disturbed conditions, even when native species are well adapted to natural disturbance regimes (Hobbs & Mooney 2005).

The disturbance of an ecosystem can occur in a number of different ways including land transformation. Land transformation frequently brings about an opportunity for biological invasion, as ecological remnants are often surrounded by potential invaders (Brothers & Spingarn 1992). The fragmentation of natural ecosystems is often subject to alien species invasion, as the changed microclimate of fragmented edges often promotes a decrease of native species and an increase in alien species (Laurance *et al.* 2000).

Brothers and Spingarn (1992) suggest that forest fragmentation could encourage invasions in two ways: firstly, fragmentation increases the ratio of forest to non-forest and of forest edge to interior; and secondly, the microclimate changes at fragment edges, which provides entry points for invasion due to the more open nature of these disturbed habitats. This hypothesis is currently under extended testing at the Biological Diversity Forest Fragments Program in the Amazonian rainforest (Laurance *et al.* 2011). The project has found that both of the potential invasion points that Brothers and Spingarn (1992) suggested are true (Laurance *et al.* 2011). Land transformation can also make way for changes in the usage of land, particularly the introduction of agricultural practices such as grazing, which may encourage invasion.

Grazing has constituted part of the global landscape over an evolutionary time frame. However, grazing can still be a disturbance within an ecosystem (Hobbs 2001). Hobbs and Huenneke (1992) hypothesised that in any situation, a significant change in grazing, either introduced or withdrawn, will create a disturbance within that particular landscape. Grazing constitutes a disturbance as it increases the spaces between grass tufts and light penetration to the soil level (Earl 2003). Grazing can also help invasions when other disturbances may be present within the ecosystem.

Disturbance by flooding generally occurs in riparian areas. Plants in these ecosystems are able to survive the repeat impacts of inundation (Elderd & Doak 2006). The flooding of riparian areas alters site conditions where new species can colonise due to the alteration of micro-ecosystem conditions (Bendix 1997). This alteration can affect species composition due to woody species being washed away, allowing additional light penetration to the soil where forbs, both invasive and native, have an enhanced opportunity to establish (Elderd 2003). However, fragmentation, flooding and grazing are not the only disturbances to encourage foreign species invasion.

Fire is a well-known ecosystem disturbance. It acts as a short-term fertilising agent where nutrient and light availability is increased post fire (Hobbs & Huenneke 1992). This creates ideal conditions for invasive species to colonise (Hobbs & Huenneke 1992). The effects of fire will be further explored in Section 1.2.

1.2 Fire, Management & Invasives

Natural and anthropogenic burning is an ecological disturbance that has led to the diversification of many landscapes. Human–induced burning of the landscape has been utilised throughout human evolution as a way to control the natural environment (D'Antonio 2000; Bond & Keeley 2005).

Plants directly reflect the landscapes that fire has changed throughout evolutionary history. It has been argued that plants have not adapted to fire *per se*, but have been selected through fire regimes to have traits that align with being able to withstand the conditions a fire creates, both during and after a burn (Keeley *et al.* 2011; Keeley & Brennan 2012; Pausas & Schwilk 2012).

Therefore, fire is seen as an evolutionary pressure shaping plant traits under particular fire regimes (Keeley *et al.* 2011). Examples of plant traits that have ensured the survival of many species appearing in areas with active fire regimes include:

- epicormic and basal sprouting (e.g. *Eucalyptus*);
- thick bark for protection from heat and fire (e.g. *Sequoia* and *Eucalyptus*);
- a dense seed bank that is only activated when smoke or heat triggers germination (e.g. *Acacia*); and

- holding seed in the canopy of a tree until a fire stimulates the release of seed, also known as serotiny (e.g. *Banksia*) (Atwell *et al.* 1999; Dawson & Lucas 2005; Keeley *et al.* 2011).

The introduction or suppression of fire within an ecosystem can lead to species diversity loss (Bond & Keeley 2005). In tropical rainforests where fire is rare, it has been documented that a single fire can alter the composition and structure of the forest, where both common and rare species die (Cochrane 2003). The possibility for rainforest regeneration by seed germination is lost due to much of the seed being stored in the litter layer, which is destroyed in the fire (Cochrane 2003). Alternatively, when fire is suppressed in tall grass prairies it can lead to loss of species diversity (Leach & Givnish 1996). Leach and Givnish (1996) found that after fire suppression over a 32 to 52 year period, between 8 and 60 percent of the original plant species had vanished from 54 prairie remnants.

If an ecosystem is not adapted to fire then the introduction of fire can also have detrimental ecological effects including an increase in the mortality of trees, bare ground cover and forbs. A study by Pettit and Naiman (2007) showed that 33% of riparian trees in a 1.5 km transect had been killed by fire, with 48% killed if there were accumulations of wood within 5 m of the base of the tree. Kutt and Woinarski (2007) reported that bare ground was far more extensive in burnt grazed quadrats of their experiment in north-central Queensland. It was also found that there was a significant difference in forb cover post fire, with the most extensive cover found in burnt and grazing quadrats. This increase of forbs post fire is well documented with many studies presenting similar data (Laughlin *et al.* 2004; Pettit & Naiman 2007; Masocha *et al.* 2011).

Fire not only has an effect on the above ground vegetation but also on the below ground soil seed bank. It has been found that many seeds experience mortality during fire in the first centimetre of soil, with most emergence occurring from soil 1 to 3 cm into the soil profile (Auld & Denham 2006). It appears that the effect of fire on seeds is dependent on the species. Auld and Denham

(2006) found that there was a significant difference in germination post fire in three woody Australian species (*Acacia suaveolens*, *Grevillea speciosa* and *Grevillea buxiflora*), with fire having a positive effect on germination and therefore a decline of the seed bank. In contrast, Davies *et al.* (2013) found that there was no significant difference in germination between untreated and heat plus smoke treated soil samples from fragmented Mallee habitat. The reproduction requirements and maturation times of species should be taken into account when prescribed burning is being implemented as a management tool, to ensure the highest amount of biodiversity is retained within the ecosystem.

1.2.1 Fire as a Native Vegetation Management Tool

Vegetation remnants are often located within urban and agricultural landscapes, which can prevent or stop natural fire regimes. Some ecosystems require fire to maintain growth and regeneration (Penman *et al.* 2009). Many Australian ecosystems rely on fires at a particular frequency, intensity and season to ensure their long term health, continued recruitment and survival (COAG Standing Council on Environment and Water 2012).

Prescribed burning can often assist in restoring a more natural fire regime to an area (Brockway *et al.* 2002). Fires may also promote preferred species such as pasture grasses for grazing (Border Rivers-Gwydir Catchment Management Authority 2008). This disturbance can be a valuable native vegetation management tool where an appropriate fire regime is implemented to maintain the biodiversity of these areas.

An example of fire being used as a native vegetation management tool to maintain the biodiversity of an area, is the implemented management regime for the endangered grassy balds in the Bunya Mountains, Queensland. This geographically isolated area exhibits 134 medium to high altitude grasslands (grassy balds) (Fensham & Fairfax 1996; Queensland Government:

Department of National Parks Recreation Sport and Racing 2012). Historic indigenous burning patterns in the area have created and helped to maintain the balds for a long period of time (Queensland Government: Department of National Parks Recreation Sport and Racing 2012).

Approximately one-quarter of the Bunya Mountains balds have been degraded over the past fifty years due to encroachment from surrounding woodland or rainforest (Queensland Government: Department of National Parks Recreation Sport and Racing 2012). Fire regimes have been implemented within the area to maintain the existing balds and to help control advances by woody native species (Queensland Government: Department of National Parks Recreation Sport and Racing 2012; Queensland Government: Department of Environment and Heritage Protection 2014). The current implemented fire regime includes a low to high intensity fire at any time of the year after rain, with biennial to triennial burns in areas where woody invasion has occurred (Queensland Government: Department of Environment and Heritage Protection 2014). A lack of fire encourages the growth of shrubs/vines as well as encouraging a change in ecosystem type to a *Eucalyptus* woodland, notophyll or microphyll vine forest (Queensland Government: Department of Environment and Heritage Protection 2014). However, sometimes the introduction or re-introduction of fire to an ecosystem, can have detrimental effects for the invasion of new species.

1.2.2 Fire and Alien Species Invasion

Fire provides an ideal window of opportunity for invasive grasses, forbs, shrubs and woody species to colonise areas not previously invaded (Hobbs & Huenneke 1992). This opportunity is created by the short term enhancement of both light and nutrient availability (Hester & Hobbs 1992). In studies reviewed by D'Antonio (2000), fire reduced or eliminated an introduced species in some 20% of cases.

Much research has documented the vast changes that occur when an alien species invades the landscape after a fire regime has been altered (Brooks *et al.* 2004; Pierson *et al.* 2011; Keeley & Brennan 2012). A well-documented study of post fire landscape change, is that of the Hawaiian submontane area. This area was invaded post fire by three alien grass species, *Melinis minutiflora*, *Andropogon virginicus* and *Schizachyrium condensatum* (Hughes *et al.* 1991; D'Antonio *et al.* 2010). These invasive grasses maintain an extremely high dead to live biomass ratio throughout most of the year, resulting in high amounts of moisture (80-90%) and fine litter particles which causes increased fire frequency (Hughes *et al.* 1991). These introduced species also exhibit rapid post fire recovery rates (Hughes *et al.* 1991). Hughes *et al.* (1991) found that post-fire, the total cover of exotic grasses increased while the cover and diversity of native vegetation sharply declined as the native species germinants were not able to grow past the dense stand of exotic grasses established soon after the fire (Hughes *et al.* 1991). D'Antonio *et al.* (2010) conducted a study 17 years after the Hughes *et al.* (1991) study to observe native ecosystem recovery. Sites that were burnt 17 and 37 years earlier, had little native species recovery, with no evidence of recruitment of the two dominant native tree species that are found at unburned sites (D'Antonio *et al.* 2010).

Alien species invasions can change an ecosystem to the extent that it can alter the natural fire regime of the area (D'Antonio & Vitousek 1992). The modification of the fire regime will depend on properties possessed by the plants, both native and introduced. The most important factors that will change a fire regime are the properties of the fuel, both intrinsic and extrinsic (Brooks *et al.* 2004). Researchers have identified five factors that can change the fuel of a fire and consequently the fire regime of the area: 1. moisture content of plant tissues; 2. chemical composition of the plant tissues; 3. fuel loads; 4. fuel continuity; and 5. fuel packing ratio (Brooks *et al.* 2004). These five properties could eventually change the natural fire regime of an area through either the suppression or enhancement of fire.

The promotion of fire has also been documented in invasive woody species such as *Hakea sericea* and *Acacia saligna* due to higher fuel loads and decreased moisture content (Mack & D'Antonio 1998). However, not all invasive species promote fire. The suppression of fire by alien species is not well documented, with research involving invasive woody species such as *Mimosa pigra* (Mack & D'Antonio 1998). The invasive forb *Phyla canescens* (lippia) has been listed in the 'Victorian Weed Risk Assessment' as an invasive weed that is able to decrease the intensity of fire and potentially reduce fire frequency (State Government of Victoria 2011).

Limited literature has been published on the use of fire to manage weeds that are not of a woody nature. Based on this finding, the current literature does not explore the use of fire on invasive forbs, in particular lippia. This lack of understanding motivates the contributions that this project will make to the current literature.

1.3 Characteristics of Lippia

Lippia is an invasive prostrate forb originally from Argentina that has been introduced and recorded in over thirty-three countries including Australia (Leigh & Walton 2004). The species was introduced to Australia through multiple introductions (Xu *et al.* 2010) in the early nineteenth century as an ornamental plant (Sosa *et al.* 2008). Lippia's current distribution covers all states and territories in Australia, excluding the Northern Territory (Earl 2003). This current distribution covers an estimated 5.3 million hectares of the Murray-Darling basin alone, with expansion likely under climate change (Figure 1). Lippia is most abundant in frequently flooded areas such as wetlands, floodplains and the riparian zone of waterways which includes several Ramsar and other protected wetlands (van Klinken & Julien 2010). This invasive species predominately affects pastoral production and mixed farming enterprises (van Klinken & Lawrence 2008).

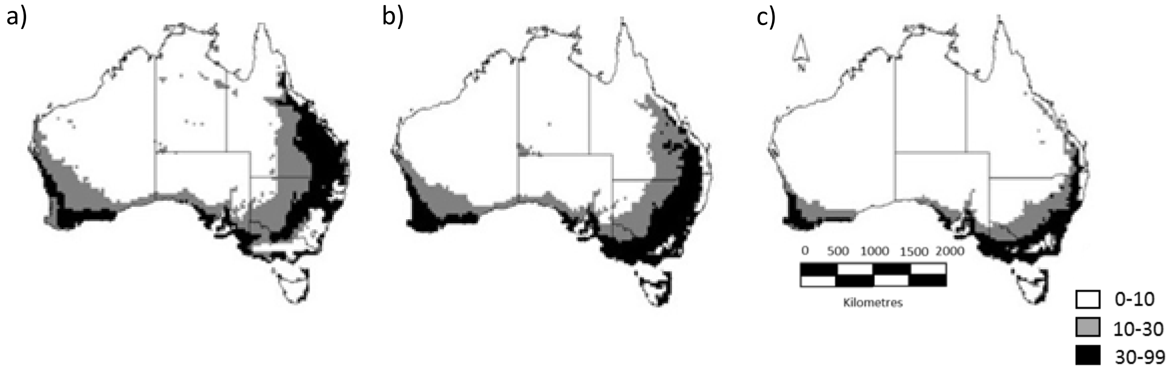

Figure 1: Lippia distribution under (a) current climate, (b) 2070 future climate using a wetter climate scenario, and (c) 2070 future climate using a drier climate scenario
(Murray *et al* 2012)

The decline of natural woodland regeneration in riparian areas of Queensland (Leigh & Walton 2004) has been suggested to be also linked with lippia cover. Lippia's competitive nature threatens many ecological communities such as *Eucalyptus camaldulensis* and *Eucalyptus populnea* woodlands, *Dicanthium* grasslands and *Cyperus* sedgelands (Earl 2003).

The economic impact on the livestock industry in the Murray-Darling Basin has been estimated to be $38 million per year, excluding the cost of herbicidal control, clearing and infrastructure damage caused by lippia (Julien *et al.* 2004; van Klinken & Julien 2010).

1.3.1 The Physiology of Lippia

Lippia is a serious environmental threat in Australia due to its current invaded areas (Figure 1a) and the area of potential invasions in the future (Figure 1b&c). The invasive species grows creeping stems up to 1 m long, with fibrous roots that develop at each node (Julien *et al.* 2004) (Figure 2). These extra root nodes allow lippia to grow into a thick, mat groundcover (Crawford

2008) which does not allow other ground covers, grasses, shrubs or trees to establish (Earl 2003). Lippia also grows a deep tap-root, up to 80 cm long (Figure 2), which has the ability to reach deep moisture reserves (Julien *et al.* 2004). This extensive root system dries out floodplain clay soils and causes soil, banks, dam walls and roads to crack and become unstable (Leigh & Walton 2004). This increased erosion may have a flow-on effect in increasing the eutrophication in river systems as a result of higher nutrient levels in the water (Leigh & Walton 2004).

Figure 2: 14 week old lippia seedling (source: Lucy Galea)

Lippia reproduces both vegetatively and sexually, which allows it to gain an advantage over other plants in the area (Xu *et al.* 2012). Both seed and vegetative fragments have the capability of floating and spreading with floodwaters (Julien *et al.* 2012b), which helps to distribute lippia throughout its current range (Taylor & Ganf 2005). Despite vegetative reproduction being a favoured strategy within local colonies (Price *et al.* 2011b), field, genetic and morphological data suggests that sexual reproduction is favoured in the Australian distribution of lippia compared to the native Argentinian range (Macdonald 2008; Price *et al.* 2010; Xu *et al.* 2010). In Australia, lippia seed bank estimates have been approximated to be 25 439 ± 4901 /m^2 seeds directly underneath adult plants (Price *et al.* 2011b).

Recent research has shown that lippia seeds require water, light and fluctuating temperatures for germination (Macdonald *et al.* 2012). Diurnal temperatures provide enough fluctuation in temperature to help create the correct germination environment (Macdonald 2008).

Table 2: Summary of lippia traits

	Lippia trait	**References**
Above-ground growth form	- Creeping stems up 1m long	(Julien *et al.* 2004)
	- Thick, mat groundcover	(Crawford 2008)
Below-ground growth form	- Up to 80cm-long taproot	(Julien *et al.* 2004)
	- Fibrous roots at each node	(Crawford 2008)
Reproduction	- Vegetative fragments dispersed by floodwaters	(Taylor & Ganf 2005; Julien *et al.* 2012b)
	- Sexual reproduction	
Flowers	- Spring to Autumn	(CSIRO 2007)
	- Small tubular white, pink or lilac	(Julien *et al.* 2004; Crawford 2008)
Seeds	- Two seeds – dispersed by natural falling & floodwaters	(CSIRO 2007)
Seed Bank	- 25 439 ± 4901 m^{-2}	(Price *et al.* 2011b)
	- Very high density in Australia	(Macdonald *et al.* 2012)
Pollination	- Pollinator needed	(Gross *et al.* 2010)
	- Introduced honeybee (*Apis mellifera*)	
Germination	- Requires free water, light & fluctuating temperatures (diurnal temperatures enough)	(Macdonald 2008; Macdonald *et al.* 2012)
Environmental Impacts	- Soil drying to several meters	(Julien *et al.* 2004)
	- Erosion	(Crawford 2008)
	- Slumping of banks, dam walls & roads	
	- All caused by deep taproot	

1.3.2 Control Methods for Lippia

The current and potential ranges of lippia highlight the need for an effective lippia control solution. This control method may potentially involve a combination of multiple control types to allow managers the best possible opportunity to successfully reduce the invasive species. Current methods of eradication include pesticides such as Glyphosate, 2,4-D amine and Dichlorprop, which are not only harmful for the environment but also for animals (Dellow *et al.* 2010). Many of these herbicides are not suitable for application due to the potential contamination of sensitive riparian areas, where much of lippia's range occurs. (Dellow *et al.* 2010). Herbicides are also not economically feasible to treat the large areas of land that lippia has invaded. Therefore, it is necessary that more environmentally friendly methods of restriction are found. The following sections will examine four previously researched potential control methods.

A potential control method for lippia could be a biological control. Biological control agents have previously worked for the control of some species, such as the Cactoblastis beetle (*Cactoblastis cactorum*) for prickly pear bushes (*Opuntia stricta*), but not others, such as the cane toad (*Rinella marina*) for cane beetles (*Dermolepida albohirtum*). It is only after careful research, trialling and testing that a suitable biological control can be released into a new environment. Currently, research is occurring in Australia to find a suitable biological control for lippia.

A study was initiated by CSIRO, the United States Department for Agriculture (USDA) and the University of Bahia Blanca in the native home range of *Phyla canescens* to examine potential biological controls for introduction in Australia (Lawrence & van Klinken 2008). Sixteen species of arthropods and 17 species of fungi were found to be eating or growing on lippia plants throughout four ecological regions of Argentina over a three and a half year period (van Klinken & Julien 2010). Out of these species, four insects, two fungi and a mite have been identified as potential biological control agents (van Klinken & Julien 2010). It has been proposed that the leaf-feeding beetle, *Kuschelina bergi*, be tested under quarantine conditions in Australia (van Klinken & Julien 2010). Australia's native lippia, *Phyla nodiflora*, poses a potential problem in the quest to find a suitable biological control, as it could also be impacted by the introduction of a lippia eating species (Julien *et al.* 2012a). It is unclear whether or not funding was reallocated to this project after its 2012 Phase 2 completion.

If *Kuschelina bergi* or another biological control is to be found effective in the control of lippia in Australia, it may need to be coupled with other methods of control. Other potential control methods that have previously been studied include grazing, flooding and mechanical disturbance.

Grazing has been promoted as a lippia control method (Earl 2003). However, it has been found that grazing is not a suitable control method due to lippia's lack of palatability to grazers (Price *et al.* 2011b). Grazing in fact encourages lippia abundance in invaded pastures/grasslands due to the reduction of grass root biomass that occurs during grazing (Earl 2003). Earl (2003) suggests that the most effective method of reducing lippia on a grazing property would be to concentrate grazing for short periods of time in one area and allowing larger areas to be periodically rested from grazing, allowing for the regeneration of native grasses and root reserves. However, recent research suggests that periodic rest from grazing has a limited effect, at least in the short term (< 3 years) (Price *et al.* 2011a). Seasonally determined rest (rest periods that were determined by the growing period of the plant) had little to no impact on the biomass of native species over three different locations (Price *et al.* 2011a). This shows that resting and grazing may not be effective solution to reducing lippia cover in grazing areas.

Another method of lippia control that has been investigated is the flooding of areas to encourage wetland species growth to outcompete lippia. Research in the Gwydir Wetlands, New South Wales, found that *Phyla canescens* prefers drier areas, suggesting that a flooding regime may work to reduce the competitiveness of the species against native wetland plants (Mawhinney 2003). This finding was supported by Price *et al.* (2010) through field studies in the Gwydir Wetlands and Macquarie Marshes, New South Wales. However, it should be noted that although evidence indicates that flooding may reduce lippia abundance, the germination strategies of lippia (Macdonald 2008) need to be accounted for before any management decision is made to flood lippia infested areas.

The response of lippia to mechanical disturbance was investigated in the PhD thesis of Macdonald (2008). One of four replicates at each site was disturbed each season with a rotary hoe, with 5-10 cm of the soil profile being disturbed (Macdonald 2008) to determine the impact it would have on lippia growth. No seedlings were found in any of the disturbance treatments at three sites, with emergence observed at the fourth site after a flood (Macdonald 2008). The cover of lippia recovered relatively quickly in spring and summer from the crowns of surviving roots, buried plant fragments and vegetative expansion from outside the treatment site (Macdonald 2008). This research indicates that mechanical disturbance is not a suitable control method for lippia.

A method of control that has not been investigated in relation to lippia is that of prescribed burning. Some grey literature (Grey 2005; Crawford 2008) lists fire as being used to control lippia; however, neither author reports on how successful this method was in preventing the growth and expansion of the lippia. This uncertainty reinforces the need for rigorous, peer reviewed research into the control of lippia using fire. This knowledge gap will be explored through this research.

The implementation of inappropriate fire regimes can be detrimental to biodiversity (Lindenmayer & Burgman 2005), especially in already highly disturbed ecosystems. A study by Fisher *et al.* (2009), assessed the use of an altered fire regime in a poor condition, highly invaded banksia woodland in Western Australia. The study found strong interaction between the number of fire events, tree canopy cover, community composition and function, and invasion (Fisher *et al.* 2009). The continual implementation of inappropriate fire regimes has led to significantly less ($p < 0.01$) native species present in sites of poor condition (Fisher *et al.* 2009). This study shows how detrimental the implementation of an inappropriate fire regime can be in highly disturbed ecosystems, such as St Ruth Reserve.

1.4 Research Hypothesis and Questions

A review of current literature has indicated that fire as a control method for *Phyla canescens* has not been adequately tested. The literature suggests that fire may be detrimental to native species within the area as fire encourages the invasion and growth of invasive species, particularly forbs, grasses (D'Antonio *et al.* 2010) and woody species (Mack & D'Antonio 1998), through changed fire regimes, increased nutrient, light availability and bare ground (Hobbs & Huenneke 1992).

This research is investigating the following general question:

What is the effect of a prescribed fire on lippia, vegetation structural components and soil seed bank in a vegetation remnant in a highly modified agricultural landscape?

Specifically, this research project will examine lippia, vegetative structural components and soil seed banks in burnt and unburnt remnant *Eucalyptus camaldulensis* and *Eucalyptus populnea* woodland at St Ruth Reserve, South East Queensland.

This research project will examine the following specific hypotheses:

1. *That there is no difference in the abundance of lippia between burnt and unburnt treatments, six months after a prescribed burn.*
2. *That there is no difference in the abundance of other vegetative structural components between burnt and unburnt woodlands, six months after a prescribed burn.*

The project will also examine seedling emergence of the post fire soil seed bank of both unburnt and burnt *Eucalyptus camaldulensis* and *Eucalyptus populnea* woodlands through the following hypotheses:

1. *That the burn has resulted in no change in the seedling emergence of lippia seeds stored in the soil seed bank.*
2. *That the burn has resulted in no change of pseudo-species richness within the seed bank.*

This research will help to fill the knowledge gap of whether fire is a useful tool in the control and management of lippia whilst contributing to the growing lippia knowledge bank. This study will help inform farmers and natural resource managers of the type of impact burning may have on the abundance of lippia. It will also increase our knowledge of the impacts that burning may have on sensitive riparian ecosystems.

Chapter 2: METHODOLOGY

2.1 Study Area

The study was undertaken at St Ruth Reserve, situated along the banks of the Condamine River, approximately 15 km south of Dalby, southern Queensland (Figure 3). The average minimum and maximum temperatures for Dalby, the closest town to St Ruth Reserve, are 11.9^{o}C and 26.2^{o}C respectively (Bureau of Meterology 2014). Dalby is 344 m above sea level and receives a mean annual rainfall of 676.4 mm (Bureau of Meterology 2014).

Figure 3: Map showing the location of the St Ruth Reserve (Source: Google Maps)

The Western Downs Regional Council (WRDC) currently manages St Ruth Reserve, a 150 ha state recreational reserve (Grey 2005). In November 2013, WRDC burnt approximately 50% of the reserve in a cool, low intensity control (prescribed) burn (Figure 4).

The reserve is a patch of remnant riparian and floodplain grassy woodland which is predominantly Regional Ecosystem types 11.3.25, 11.3.4 and 11.3.2 (Sattler & Williams 1999). The predominant species of the reserve are *Eucalyptus camaldulensis* (River Red Gum)/*Eucalyptus tereticornis* (forest red gum), *Eucalyptus populnea* (poplar box), *Acacia salicina* (native willow) and *Acacia stenophylla* (shoestring acacia) (Crawford 2008).

Until recently, the reserve was held under a grazing lease for grazing cattle (Crawford 2008). Continual stocking of the reserve resulted in significant changes in the groundcover composition and structure, resulting in a change from native perennial tussock grasses to a low growing, lippia-dominant ‘carpet’ (Reardon-Smith 2011). By the late 1990s the reserve was heavily degraded with gully and river erosion, eucalypt dieback and lippia groundcover dominance (Crawford 2008). The grazing lease was revoked in 1999 when the reserve was reclassified as a state recreational reserve (Crawford 2008). Since the cessation of grazing, the reserve has shown signs of recovery (QMDC 2005). Lippia cover had decreased significantly post grazing exclusion with an increase in the relative abundance of other species including native perennial grasses (Reardon-Smith 2011). However, this decline in lippia abundance coincided with a prolonged drought, which is believed to have contributed to the decrease in abundance (Reardon-Smith 2011). Since then, the reserve has flooded several times, including 2014, which has seen lippia return of to all areas of the reserve (pers. obs.). Some small scale trials had been initiated within the reserve to help reduce lippia cover, but once discontinued the experiments did not provide an observable, overall impact on lippia cover at St Ruth Reserve (QMDC 2005). One of the small scale trials implemented was a cool fire in 2001 (QMDC 2005). Other than the 2001 and 2013 fires, there are no records of other fires at St Ruth Reserve.

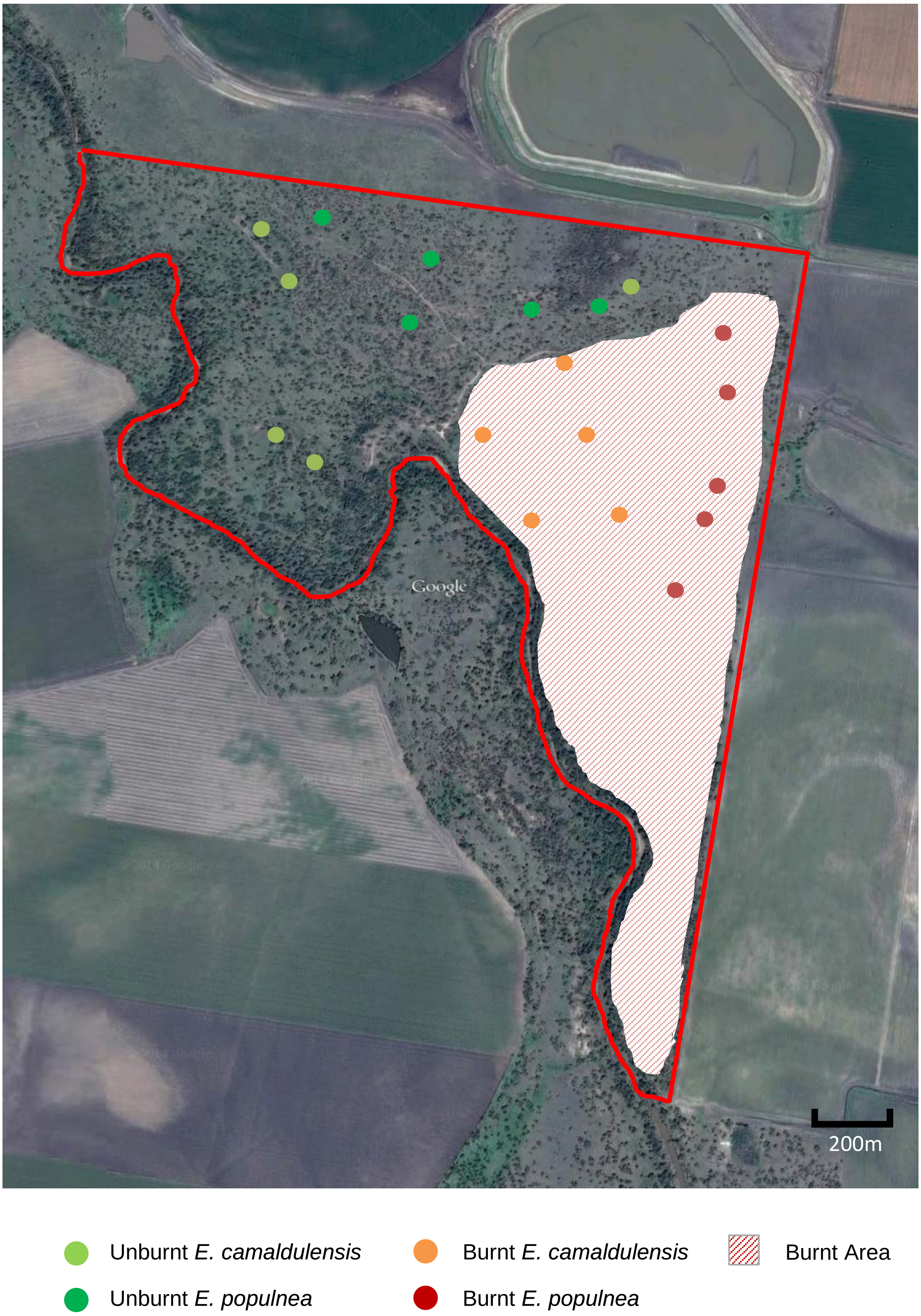

Figure 4: Aerial photograph of St Ruth Reserve showing burnt/unburnt areas in November 2013 and the position of survey sites (Source: Lucy Galea/Google Maps)

The Darling Downs region is recognised as Queensland's most fertile agricultural district and was predominantly used for cattle and sheep grazing from 1840 until about the 1960s from which time and agriculture and cultivation replaced grazing as the dominant land use (Fensham 1998).

In 2010–2011, the Darling Downs and Maranoa region had a gross agricultural production value of $2.5 billion (Trestrail *et al.* 2013). This equated to 26% of the total gross value of agricultural production in Queensland for 2010−2011 (Trestrail *et al.* 2013). The region's main agricultural commodities are cotton (worth $656 million), cattle and calves (worth $505 million) wheat (worth $297 million) and sorghum (worth $181 million) (Trestrail *et al.* 2013).

The region is composed of four broad remnant vegetation types: a) red gum woodland; b) grassland; c) poplar box woodland; d) hill woodland. Red gum woodlands are dominated by *E. camaldulensis* or *E. tereticornis* growing on the floodplains of major water systems (Fensham 1998). Treeless grassland occurs on clay soil plains (Fensham 1998). Poplar box woodland is defined by *E. populnea* woodland occupying rock free clay loam soils that are not prone to flooding (Fensham 1998). Hill woodland is dominated by *E. albens, E. crebra, E.. melliodora* or *E. orgadophila* species on clay loam soils on rocky hilly terrain (Fensham 1998).

Prior to modification the region was dominated by Queensland bluegrass (*Dichanthium sericeum*) (Fensham 1998). However, this has been reduced to approximately 1% of its former area (Fensham 1998). *E. tereticornis*/*E. populnea* grassy open woodland has been reduced to 11.39% of its original area since clearing began in the area (Fensham 1997). Remnant native areas that are remaining on the Darling Downs are mostly in poor condition (Reardon-Smith 2011).

2.2 Experimental Design

Study sites were located within two vegetation types within the Reserve:

1. *E. camaldulensis/E. tereticornis* (river red gum/forest red gum) grassy woodland (from here referred to as *E. camaldulensis* woodland)
2. *E. populnea* (poplar box) grassy woodland

The experimental design was a nested design with burnt/unburnt plots nested within woodland type (Figure 5). This gave four experimental combinations of: burnt *E. camaldulensis* (BC), burnt *E. populnea* (BP), unburnt *E. camaldulensis* (UC) and unburnt *E. populnea* (UP). The unburnt plots formed the control sites for the experiment. Five replicates of each treatment and control were used (Figure 4 & 5). This gave a total of 20 total sampling sites for the experiment.

Sites were selected two weeks after the prescribed burn with GPS locations recorded (see Appendix A). The selection of sites considered surrounding vegetation and external factors to ensure that confounding effects were minimised. These included avoiding obvious tracks and gullies within the reserve and selecting sites at least 30 m away from fences and being at least 100 m away from sites of the same type.

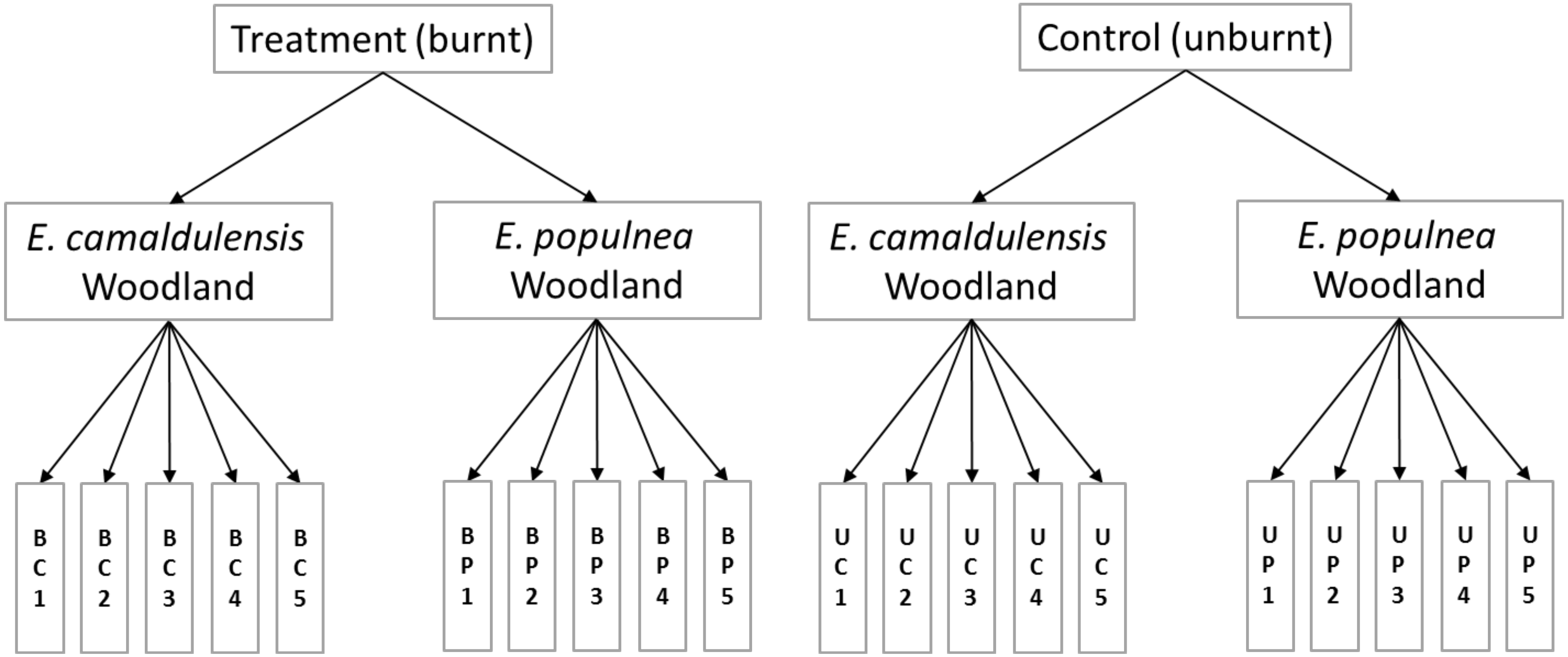

Figure 5: Diagram of experimental design used, showing burnt/unburnt plots nested within each woodland

2.3 Field Experiment: Patterns in Vegetation Across Burnt/Unburnt Woodlands

Field work was undertaken between May and June 2014, six months after the controlled burn. A brief site description was recorded to account for any nearby potentially confounding influences such as agricultural land, obvious tracks and the river, as well as how far away these were from the site. It was also noted if there was evidence of flooding, such as debris in trees, from March 2014.

General site condition was subjectively measured on a five point ‘disturbance’ scale: 0 (no evidence of disturbance) to 4 (substantial disturbance) at each site. Disturbance variables measured were grazing, clearing, logging, erosion, weeds, feral animals, soil compaction and evidence of dieback (as per Batterham 2008) (Table 3).

Table 3: Summary of variables measured at each site

Variable	Measure	Description
Site Condition		
Grazing	0–4	Cattle, sheep grazing etc: evidence of scats, hoof prints
Clearing	0–4	Pushing, chaining etc: presence of logs, mechanical soil disturbance
Logging	0–4	Selective logging, firewood gathering: presence of stumps, chopped logs
Erosion	0–4	Sheet, Gully
Weeds	0–4	Introduced pest species, native pests
Feral Animals	0–4	Pigs, goats, horses, foxes, cats etc: evidence of scats, burrows, diggings (pigs)
Soil Compaction	0–4	Physical compaction of soil
Evidence of Dieback	0–4	Stags
500m^2 Quadrat		
Trees >20m	No. and %cover	Trees above 20 m high
Trees 10-20m	No. and %cover	Trees between 10 m and 20 m high
Trees <10m	No. and %cover	Trees less than 10 m high
Standing Stag >20m	No. and %cover	Stags above 20m high
Standing Stag 10-20m	No. and %cover	Stags between 10 m and 20 m high
Standing Stag <10m	No. and %cover	Stags less than 10 m high
Stumps	No. and %cover	Stumps
Logs: Thumb – Two Fingers	No. and %cover	Logs and branches between 6 cm and 13 cm
Logs: 2 Fingers – 2 Hands	No. and %cover	Logs and branches between 13 cm and 35 cm
Logs: 2 Hands – 1 Arm	No. and %cover	Logs and branches between 35 cm and 90 cm
Logs: 1 Arm – 2 Arms	No. and %cover	Logs and branches between 90 cm and 150 cm
Grasses/Sedges	%cover	Subjective cover of total grasses present
Herbs/Forbs	%cover	Subjective cover of total herbs/forbs present
Lippia	%cover	Subjective cover of total alive lippia present
Fine Litter: Leaf Litter/Twigs	%cover	Subjective cover of total fine litter (leaf litter/twigs) present
Lippia Litter	%cover	Subjective cover of total fine litter (brown lippia) present
Bare ground	%cover	Subjective cover of bare ground exposed
70cm^2 Quadrat		
Lippia	%cover	Approximate % of quadrat that has green lippia
Lippia Litter	%cover	Approximate % of quadrat that has brown lippia
Lippia Frequency	/64	Squares of quadrat that has lippia growth
Lippia Frequency	/81	Point intersections that have lippia underneath them
Grass Frequency	/64	Squares of quadrat that has rooted grasses growth
Forbs Frequency	/64	Squares of quadrat that has rooted forbs growth

At each site a 500 m^2 (22.3 m x 22.3 m) quadrat was set out with the initial site marker approximately in the centre of the quadrat. The number of trees within the quadrat were counted within three height categories: > 20 m, 10–20 m and < 10 m. A clinometer and tape measure was used to measure trees thought to be taller than 10 m. The number of standing stags within the quadrat were also measured using the same height categories. The number of stumps within the quadrat were also counted. Fallen logs and branches were categorised into four diameter categories: 6 cm to 13 cm; 13 cm to 35 cm; 35 cm to 90 cm; 90 cm to 150 cm. The percentage cover of the above variables plus the cover of grasses/sedges, herbs/forbs, lippia, fine litter: leaf litter/twigs, lippia litter and bare ground were also subjectively assessed.

A 16 m transect was run diagonally across each 500 m^2 from the south eastern corner. Along this transect, a 70 cm^2 sub-quadrat was placed at every second metre (8 samples total across transect) to determine finer scale patterns in lippia cover and other selected variables. The quadrat was laced with 4 ropes equal distance apart on each side to create a network of 64 squares and 81 point intercepts. Lippia variables measured within each 70 cm^2 quadrat included: a subjective percent cover of lippia and lippia litter (lippia litter is brown/dead lippia); frequency (presence/absence) for lippia within each of the 64 squares (reported as a proportion); intercept lippia cover (presence/absence) of lippia at each of the 81 points (reported as a proportion). A count was made when any part of the lippia plant was found in a square or directly under a point intercept, respectively. These three estimates were used to determine if there were any differences in the ability of methods to account for the variable being measured. Frequency at 64 squares (reported as a proportion) was also recorded for grasses and forbs, with the grass or forb having to be rooted within the square to be counted.

2.4 Glasshouse Experiment: Soil Seed Bank Germination Across Burnt/Unburnt Woodlands

Soil samples were collected at each of the sites sampled in Section 2.3 to determine the germinable seed bank of burnt/unburnt treatments within each woodland. Soil sub-samples were collected from four points within the 500 m^2 quadrat; which were determined using random numbers between 1 and 10, previously generated by a Random Number Generator application. A 15 cm^2 square was outlined using a shovel, and soil was taken from this area to a depth of approximately 2 cm. The four soil samples for each site were bulked, mixed and placed in paper bags. After collection, any large root matter, live plant particles and rocks were removed to ensure that any emergent germinants were only from the seed bank.

In the glasshouse, each bulked soil sample was placed on top of 1 cm of washed river sand in seed trays. Two trays of sand only were also prepared as a control for the experiment to identify any contamination from the sand or glasshouse during the experiment. Sample number was written on each tray for identification. Trays were placed in the greenhouse and watered until saturation point. Samples were left in the greenhouse for a period of 14 weeks so that germination could occur. Watering occurred every one to four days, and was dependent on the dryness of the soil. Trays were situated in two rows within the greenhouse. Rotation of trays (both swapping of rows and within rows) occurred once a week to minimise any bias due to variations in sunlight and temperature within the greenhouse. A table of randomised numbers between 1 and 11, generated by a Random Number Generator application, provided a different tray rotation within a row each week.

Once a week, trays were checked for new germinants. New germinants were labelled with a coloured toothpick to indicate the week of emergence (Appendix B). Week one and two emergence was combined due to field work also being carried out at this time. Germinants of species that were distinguishable at Week 4 were removed, while others that could not be determined were left in the tray to grow further. Removal of identifiable germinants reduced competition for light and nutrients, which may have inhibited further germination if these germinants had not been removed. After Week 14, all seedlings were removed and counted. Seedlings were separated into monocots and dicots. Species, apart from lippia, were not identified taxonomically due to time constraints and pseudo species names were given to distinguish between species and to enable ‘pseudo-species’ richness of each sample to be determined (Appendix C).

2.5 Statistical Analyses

All analyses were performed using IBM SPSS ® for Windows version 20.0. Independent T-tests (Sedgewick 2010) were used for all analyses following any necessary data transformations (see below). The Levene’s Test for Homogeneity was also used to determine whether the samples were homogeneous (Lim & Loh 1996).

2.5.1 Field Experiment: Patterns in Vegetation Across Burnt/Unburnt Woodlands

Independent T-tests were used to determine if there were any significant differences ($p < 0.05$) in measured variables at the 500 m^2 quadrat scale between burnt and unburnt plots within each woodland type. Variables tested were the proportion of trees (total), stags (total), stumps, logs (total), grasses/sedges, herbs/forbs, lippia, fine litter (leaf litter/twigs), lippia litter and bare ground. All proportion values were arcsine transformed before analysis to overcome boundedness (e.g. Masocha *et al*. 2011). All count data was $\log_{10}(1+x)$ transformed prior to analysis to also overcome non-normality (e.g. Price *et al*. 2011b)..

Any potential significant differences ($p < 0.05$) between burnt and unburnt sites within each woodland were also tested by comparing lippia, grass and forb growth within woodlands from the 70 cm^2 quadrat data. Frequency data from the eight 70 cm^2 sub-quadrats were converted to proportions and averaged to obtain mean proportion cover data for the analyses. All proportion data were arcsine transformed and Independent T-tests were used to determine if there were any significant differences between means.

2.4.2 Glasshouse Experiment: Soil Seed Bank Germination Across Burnt/Unburnt Woodlands

Independent T-tests were used to test whether there were any significant differences ($p < 0.05$) in seedling emergence, between burnt and unburnt treatments within woodland types. Numbers of ‘monocots’, ‘dicots’, ‘dicots minus lippia’, ‘lippia’, ‘total germinants’ and ‘total germinants minus lippia’ were all tested for significant differences between treatments.

Independent T-tests were also used to determine any differences ($p < 0.05$) in pseudo-species richness between burnt and unburnt treatments from each woodland. As the data was count data, all values were $\log_{10}(x+1)$ transformed prior to analysis (e.g. Price *et al.* 2011b). The $(x+1)$ transformation was used to account for weeks where there was no new emergence (i.e. zeros in the data) (Price *et al.* 2011b). Data from both Week 3 and Week 14 were analysed. Week 3 data were analysed because this time period signalled the end of the immediate emergence flush. Week 14 data were analysed as this was at the end of the experiment.

Chapter 3: RESULTS

3.1 General Results

Measures of disturbance were similar across all sites irrespective of treatment or woodland (Table 4). There was no evidence of erosion or soil compaction, with minimal feral animal disturbance at any of the sites at either woodland (Table 4). High levels of weed infestation and dieback were seen across all sites and woodlands (Table 4) (Appendix D).

Table 4: Site disturbance score (0-4) ranges and means in burnt and unburnt *E. camaldulensis* and *E. populnea* woodland at St Ruth Reserve, south-eastern Queensland

Disturbance type	Burnt			Unburnt		
	Min	**Max**	**Mean**	**Min**	**Max**	**Mean**
E. camaldulensis						
Grazing	1	2	1.6	1	2	1.4
Clearing	1	2	1.2	1	2	1.6
Logging	1	3	1.8	1	3	2
Erosion	0	0	0	0	0	0
Weeds	2	3	2.4	2	4	3
Feral animals	0	0	0	0	1	0.2
Soil compaction	0	0	0	0	0	0
Evidence of dieback	1	3	2	2	3	2
E. populnea						
Grazing	0	1	0.8	1	2	1.6
Clearing	1	3	2	1	3	1.8
Logging	2	3	2.2	0	3	2
Erosion	0	0	0	0	0	0
Weeds	2	3	2.6	1	3	1.8
Feral animals	0	1	0.2	0	1	0.2
Soil compaction	0	0	0	0	0	0
Evidence of dieback	2	3	2.4	1	2	1.2

3.2 Patterns in Vegetation Across Burnt/Unburnt Woodlands

3.2.1 Cover and density of structural components (at 500 m^2 scale)

Across all sites sampled, total tree cover ranged from 0% to 10%. Across the 10 unburnt sample sites, tree density per 500 m^2 ranged from 1 to 21 with an average of 7.5 trees per 500 m^2 (SE ± 1.1). The 10 burnt treatment sites had a range of tree density per 500 m^2 of 0 to 11 with an average of 3.8 trees per 500 m^2 (SE ± 2.2). Stag and stump density was similar across all treatments and woodlands, with a range from 0 to 3. The average stag density was 0.6, with the average stump density of 0.95. There was large variation in log density across the two treatments and woodlands. Log density ranged from 5 to 187 with an average of 44.25. There were no major patterns or trends to report from this data.

Density of trees, stags and stumps were relatively similar across both burnt and unburnt treatments in *E. camaldulensis* and *E. populnea* woodlands. Tree density was highest in unburnt woodlands. *E. camaldulensis* woodlands experiences high log density in burnt patches, while the highest log density in *E. populnea* woodlands was observed in unburnt treatments (Table 5; Figures 6 & 7; Appendix E)

Table 5: Means and standard errors of density data in burnt and unburnt *E. camaldulensis* and *E. populnea* woodland on a 500 m^2 scale

Density	Burnt		Unburnt		T-test (df=8)	
	Mean	**± SE**	**Mean**	**± SE**	**F**	**p-value**
E. camaldulensis						
Trees	5.40	2.15	9.00	3.98	0.11	0.53
Stags	1.00	0.63	0.40	0.24	10.04	0.55
Stumps	1.00	0.45	1.60	0.68	1.70	0.64
Logs	53.60	9.28	20.00	4.93	0.48	0.12
E. populnea						
Trees	2.20	0.58	5.80	2.27	2.85	0.20
Stags	0.40	0.24	0.60	0.40	1.32	0.78
Stumps	0.60	0.40	0.60	0.40	0.000	1.00
Logs	35.40	10.53	68	30.44	0.04	0.33

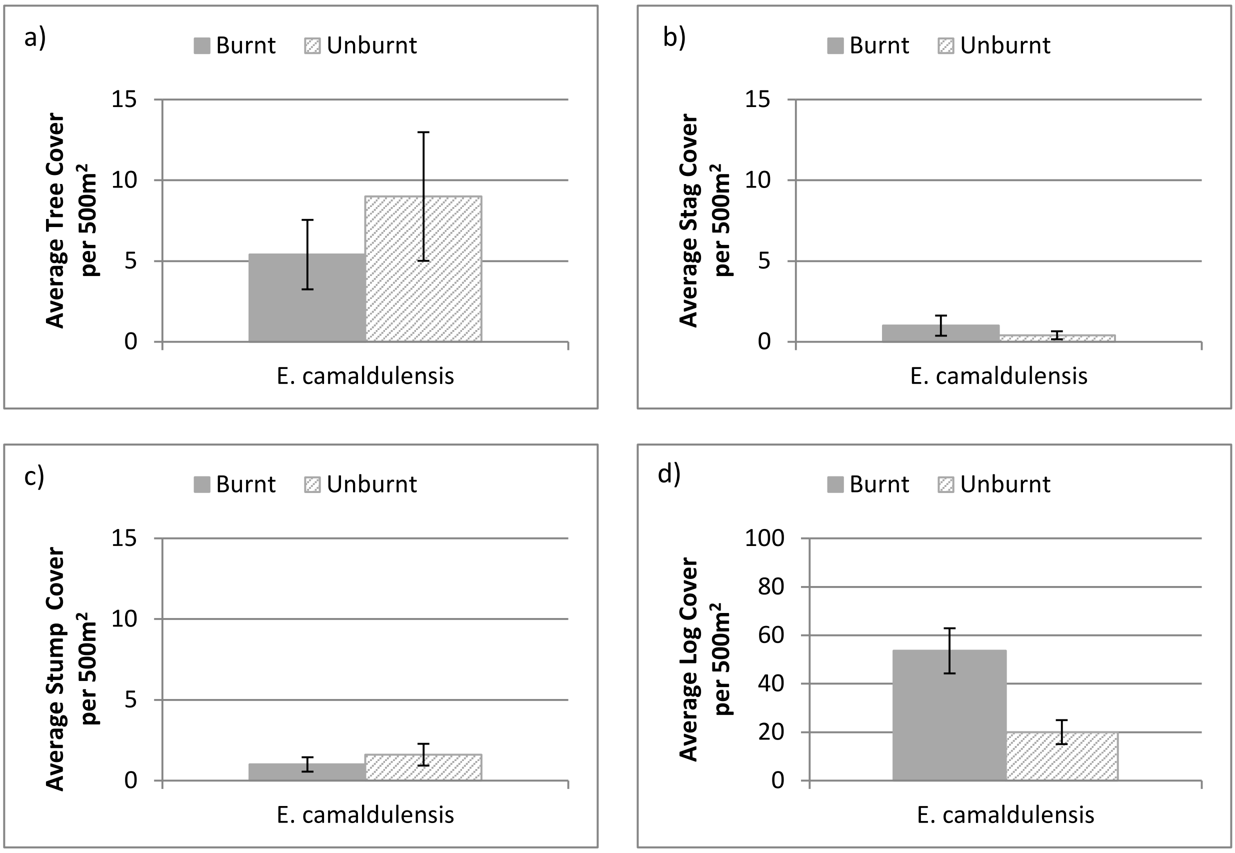

Figure 6: *E. camaldulensis* average count of: a) trees; b) stags; c) stumps; d) logs (note different scale)

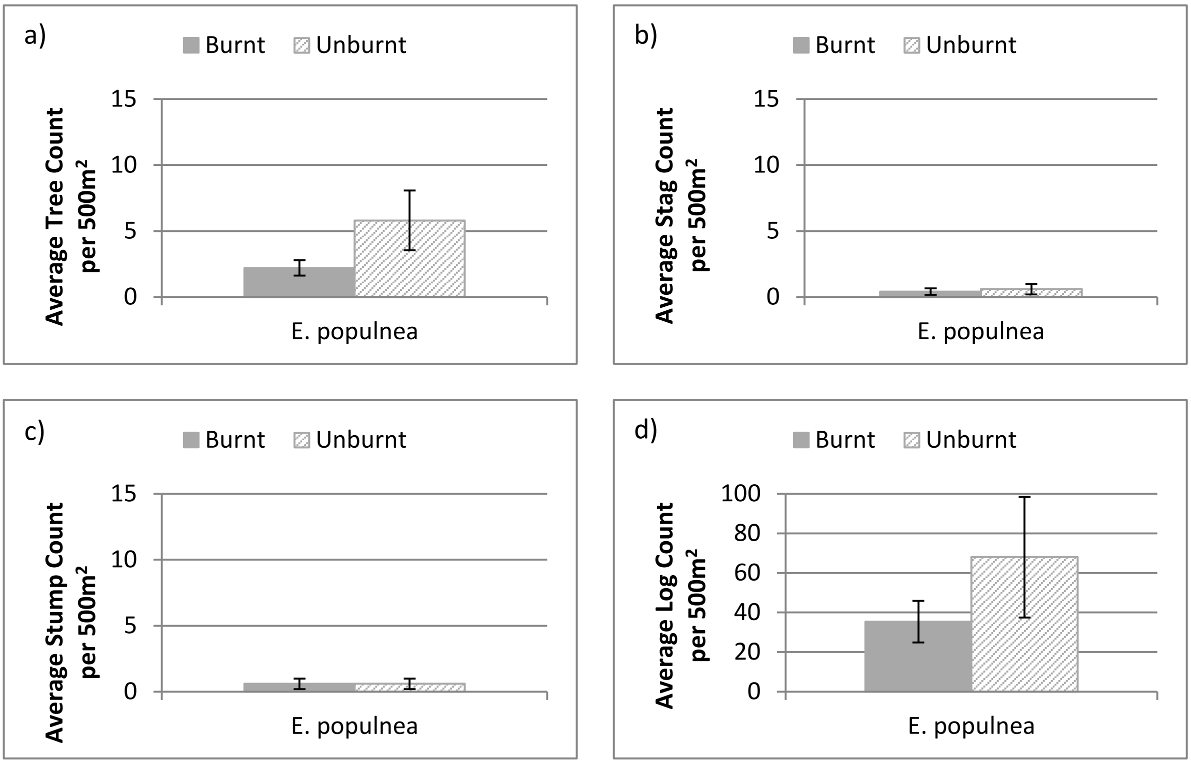

Figure 7: *E. populnea* average count of: a) trees; b) stags; c) stumps; d) logs (note different scale)

Across all treatments and woodlands, percent tree cover ranged from 0% to 10% with an average cover of 3.5% (SE ± 0.6). Average stag cover was 0.2% (SE ± 0.05), with an average stump cover of 0.17% (SE ± 0.05). The average log cover across all treatments and woodlands was 1.2% (SE ± 0.8) with a range of 0.5% to 3%. Across both woodlands and treatments, the average percentage cover of grass/sedge (70 cm^2 quadrats) was 53.5% (SE ± 6.1), with a range of 5% to 90%. Fine litter cover ranged from 0.1% to 3%, with an overall average of 0.9% (SE ± 0.2). The average lippia cover of all treatments was 35.2% (SE ± 6.7) and a range of 0.5% to 85% with lippia litter having an observed average of 0.1% (SE ± 0.1), ranging from 0% to 1% cover. Bare ground had an overall average cover of 0.7% (SE ± 0.3), with a range from 0% to 5% (Appendix E).

Tree cover was highest in unburnt *E. populnea* treatments, with bare ground cover highest in burnt treatments. The average cover of grass was highest in *E. camaldulensis* unburnt treatments, with percent lippia consistent over both treatment types. Lippia litter was minimal (Table 6; Appendix E).

No significant differences (T-tests; $p > 0.05$) were found between burnt and unburnt treatments for the average density (Appendix F) or percentage cover of trees, stags, stumps or logs in 500 m^2 quadrats (Table 7; Appendix G) within either *E. camaldulensis* or *E. populnea* woodlands.

Table 6: Percentage cover average of field variables at 500 m^2 scale

% Cover	Burnt		Unburnt	
	Mean	**± SE**	**Mean**	**± SE**
E. camaldulensis				
Trees	2.80	1.24	2.40	0.93
Stags	0.20	0.12	0.20	0.12
Stumps	0.3	0.12	0.22	0.12
Logs	1.00	0.27	0.70	0.122
Grasses/sedges	47.00	17.07	59.00	9.66
Herbs/forbs	8.1	6.73	4.6	2.61
Lippia	36.2	19.92	38.8	14.7
Lippia litter	0.00	0.00	0.20	0.20
Fine litter	0.82	0.24	0.60	0.10
Bare ground	0.60	0.10	0.12	0.09
E. populnea				
Trees	3.60	1.63	4.12	1.26
Stags	0.20	0.12	0.12	0.09
Stumps	0.12	0.09	0.04	0.02
Logs	1.78	0.49	1.30	0.30
Grasses/sedges	60.00	12.14	48.00	12.00
Herbs/forbs	8.6	4.38	1.72	1.32
Lippia	29	6.40	37.1	13.12
Lippia litter	0.00	0.00	0.34	0.18
Fine litter	1.02	0.52	1.3	0.44
Bare ground	1.72	0.88	0.42	0.18

Table 7: Summary data for field variables at a 500 m^2 scale
(all proportions were arcsine transformed prior to analysis)

	Levene's Test for Homogeneity		T-Value	df	Significance
	F stat	Sig			
***E. camaldulensis* Woodland**					
Trees	0.267	0.619	1.096	8	0.305
Standing stags	1.833	0.213	-0.512	8	0.622
Stumps	35.200	0.000	-1.521	8	0.167
Logs	8.395	0.02	1.852	5.297[1]	0.120
Grasses	0.311	0.592	-0.588	8	0.573
Herbs/forbs	1.336	0.281	-0.982	8	0.355
Fine litter	3.467	0.100	1.565	8	0.156
Lippia	0.391	0.549	-0.147	8	0.887
Lippia litter	0.003	0.958	0.513	8	0.622
Bare ground	1.580	0.244	1.485	8	0.176
***E. populnea* Woodland**					
Trees	0.116	0.742	0.391	8	0.706
Standing stags	0.000	1.000	0.000	8	1.000
Stumps	1.833	0.213	-1.152	8	0.282
Logs	7.334	0.027	1.372	6.213[1]	0.218
Grasses	0.725	0.419	0.539	8	0.604
Herbs/forbs	0.609	0.458	0.050	8	0.962
Fine litter	1.152	0.314	0.352	8	0.734
Lippia	30.448	0.001	-0.526	4.637[1]	0.623
Lippia litter	0	0	0	0	0
Bare ground	5.914	0.041	1.265	4.103[1]	0.273

[1] T-test statistic for non-homogeneous variances as indicated by Levene's Test

3.2.2 Fine Scale Patterns of Frequency and Cover (70 cm^2 sub-quadrats)

Fine scale patterns of lippia, grasses/sedges and herbs/forbs were determined using the 70 cm^2 sub-quadrats within each 500 m^2 quadrat. Subjective lippia cover ranged from 7.6% to 95.6% over all treatments and woodlands, with an overall percent average of 44.1% (SE ± 5.1). Sixty-four square frequency of lippia ranged from 38.7% to 100% with an average of 75% (SE ± 3.9). The point-intersect method for lippia had a range of 17.9% to 93.7% with an average of 51.6% (SE ± 4.8). The average grass frequency on a fine scale was 66.92% (SE ± 4.1), with a range from 22.9% to 95.5%. Average forb frequency was 47.7% (SE ± 3.9) with a range from 20.5% to 81.4%.

Lippia cover was consistently high over the three study methods, with minimal lippia litter present over both treatments and woodlands. Grass and forb cover were relatively similar over both *E. camaldulensis* and *E. populnea* woodlands (Table 8; Figures 8 & 9; Appendix H).

Table 8: Average percent cover at a 70 cm^2 scale

Ground Cover (%)	Burnt		Unburnt		T-test (df=8)	
	Mean	**± SE**	**Mean**	**± SE**	**F**	**p-value**
E. camaldulensis						
Lippia	56.40	13.21	51.84	11.36	1.21	0.72
Lippia litter	0.06	0.04	2.79	2.08	6.77	0.15 (df=4.105)
Lippia (/64)	80.59	7.37	82.19	6.92	0.13	0.93
Lippia (/81)	72.03	15.76	77.15	11.80	1.11	0.91
Grasses/sedges (/64)	61.21	10.34	68.55	6.88	0.49	0.59
Forbs/herbs (/64)	52.15	9.13	39.49	5.99	1.84	0.26
E. populnea						
Lippia	39.88	5.49	28.39	7.58	0.59	0.26
Lippia litter	0.03	0.02	0.37	0.19	15.55	0.154 (df=4.002)
Lippia (/64)	67.03	9.54	71.76	7.05	0.46	0.75
Lippia (/81)	62.73	7.21	49.14	11.88	1.109	0.599
Grasses/sedges (/64)	73.59	6.16	64.34	10.21	1.95	0.03*
Forbs/herbs (/64)	61.37	5.80	37.77	6.72	0.00	0.389

* Denotes significance difference ($p < 0.05$)

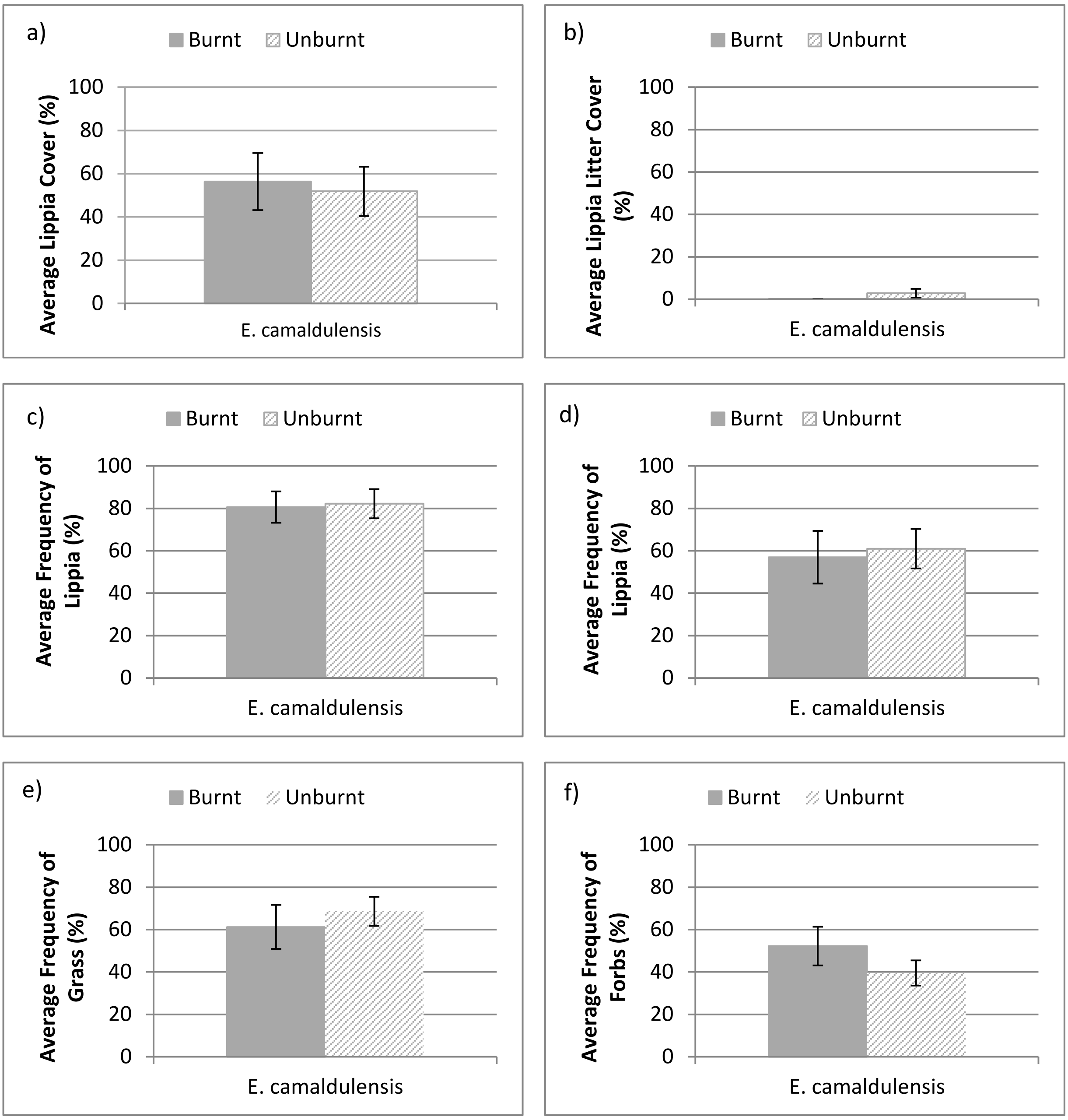

Figure 8: *E. camaldulensis* percent average cover 70 cm^2: a) lippia b) lippia litter; c) lippia (/64) ; d) lippia point intersect; e) grass; f) forbs;
values are means; error bars are standard errors

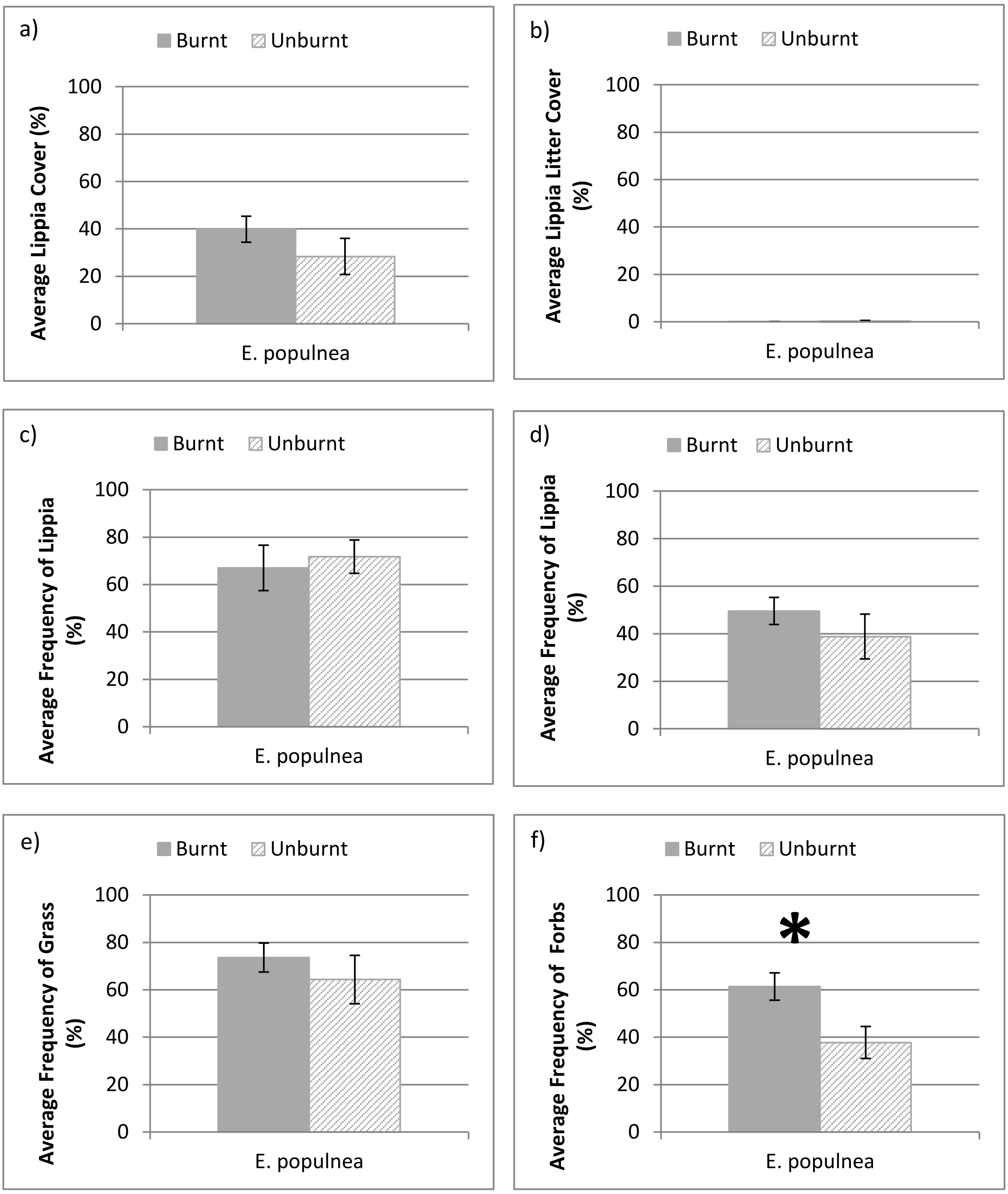

Figure 9: *E. populnea* percent average cover 70 cm^2: a) lippia b) lippia litter; c) lippia (/64); d) lippia point intersect; e) grass; f) forbs;
values are means; error bars are standard errors; *significant difference $p<0.05$

There was a significant difference ($p < 0.05$) in forb frequency between burnt and unburnt treatments within *E. populnea* woodland (Figure 9f; Table 7) in the 70cm^2 sub-quadrats, with forb frequency significantly higher in the burnt *E. populnea* treatment (Figure 9f). There were no other significant differences ($p > 0.05$) in frequency and cover variables between burnt and unburnt treatments of *E. camaldulensis* and *E. populnea* woodlands when compared at a fine scale of 70 cm^2 (Table 9; Appendix I).

Table 9: Summary Data for Field Variables at a 70 cm^2 scale
(all proportions were arcsine transformed prior to analysis)

	Levene's Test for Homogeneity		T-Value	df	Significance
	F stat	Sig			
***E. camaldulensis* Woodland**					
Lippia cover (subjective)	1.212	0.303	0.376	8	0.716
Lippia litter cover (subjective)	6.770	0.032	-1.314[1]	4.002	0.259
Lippia frequency	0.131	0.726	-0.086	8	0.933
Grass frequency	0.488	0.505	-0.569	8	0.585
Forbs frequency	1.836	0.212	1.209	8	0.268
Lippia cover (point intersect)	1.109	0.323	-0.125	8	0.903
***E. populnea* Woodland**					
Lippia cover (subjective)	0.590	0.464	1.213	8	0.260
Lippia litter cover (subjective)	15.552	0.004	-1.745[1]	4.105	0.154
Lippia frequency	0.457	0.518	-0.327	8	0.752
Grass frequency	1.954	0.200	0.547	8	0.599
Forbs frequency	0.000	0.990	2.579	8	0.033*
Lippia cover (point intersect)	1.797	0.217	0.912	8	0.389

[1] T-test statistic for non-homogeneous variances as indicated by Levene's Test; * significant difference ($p < 0.05$)

3.3 Soil Seed Bank Germination Across Burnt/Unburnt Woodlands

3.3.1 Germinant Emergence

The average total germinant emergence over the 14 weeks for all treatment and woodland combinations was 108.6 germinants (SE ± 16.1) with a range from 25 to 305 germinants. Monocot germination ranged from 0 to 30 germinants across all treatment and woodland combinations (Appendix J).

Total germinants were higher in unburnt plots than burnt plots over both woodlands. The germination and emergence of lippia was low over both *E. camaldulensis* and *E. populnea* woodlands and burnt and unburnt treatments. Lippia did not have any significant effect on the total sample germination, with total germinants minus lippia being minimally less than total germinants (Table 10; Figures 10 & 11).

Table 10: Mean and standard error of average germination

Average Germination	**Burnt**		**Unburnt**	
	Mean	**± SE**	**Mean**	**± SE**
E. camaldulensis				
Monocots	1.20	0.49	4.80	1.85
Dicots	102.00	26.84	141.40	44.33
Lippia	1.80	0.97	3.20	0.97
Total germinants	103.20	26.86	146.20	45.90
Total germinants minus lippia	101.4	27.50	143.00	45.36
E. populnea				
Monocots	7.80	5.60	9.8	2.92
Dicots	65.6	13.85	101.6	30.96
Lippia	2.40	1.25	5.00	2.00
Total germinants	73.40	18.48	111.4	33.40
Total germinants minus lippia	71.00	17.74	106.4	34.30

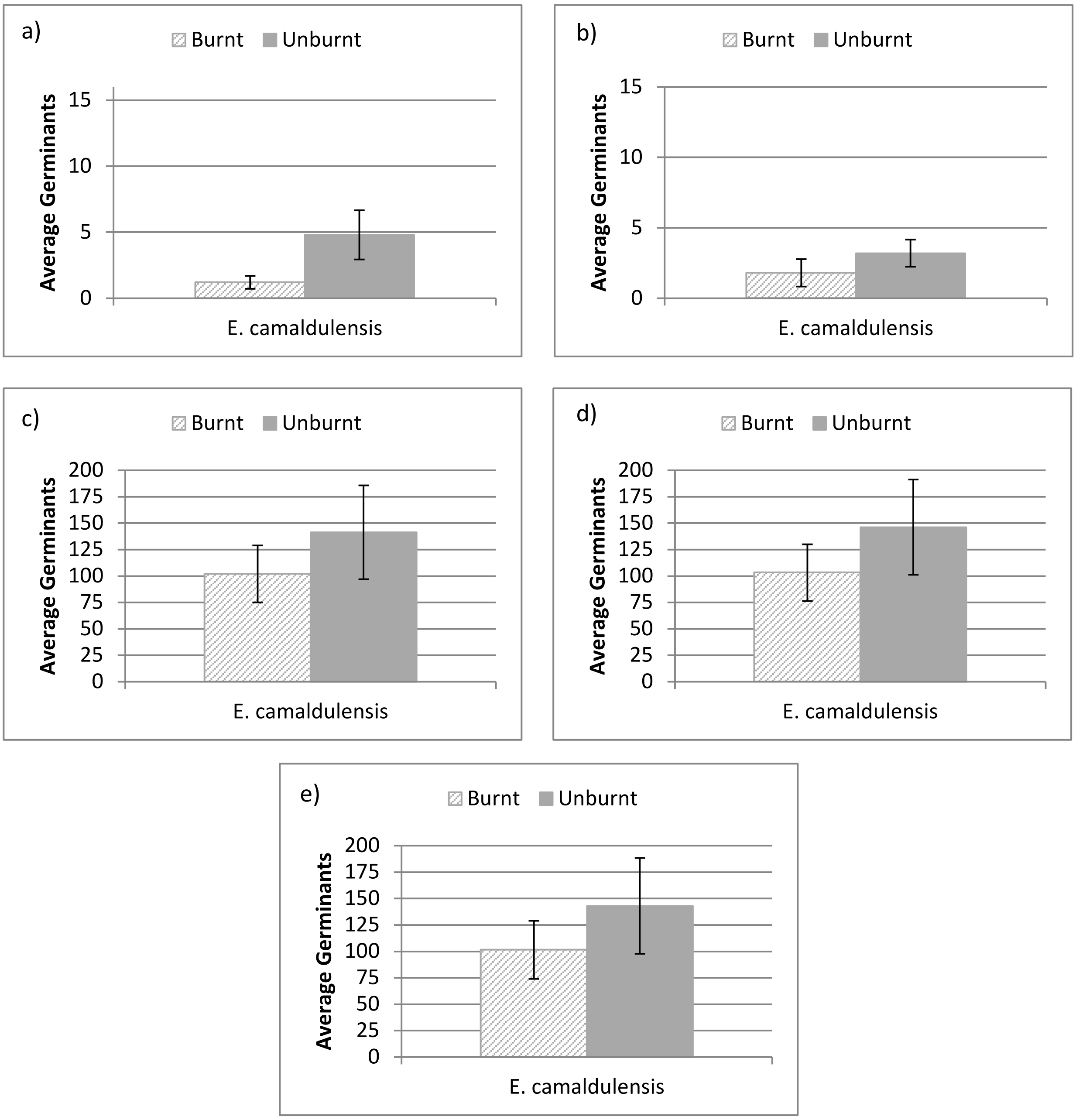

Figure 10: *E. camaldulensis* average germinants of: a) monocots; b) lippia; c) dicot; d) total; e) total minus lippia (please note scale on all graphs); values are means; error bars are standard errors

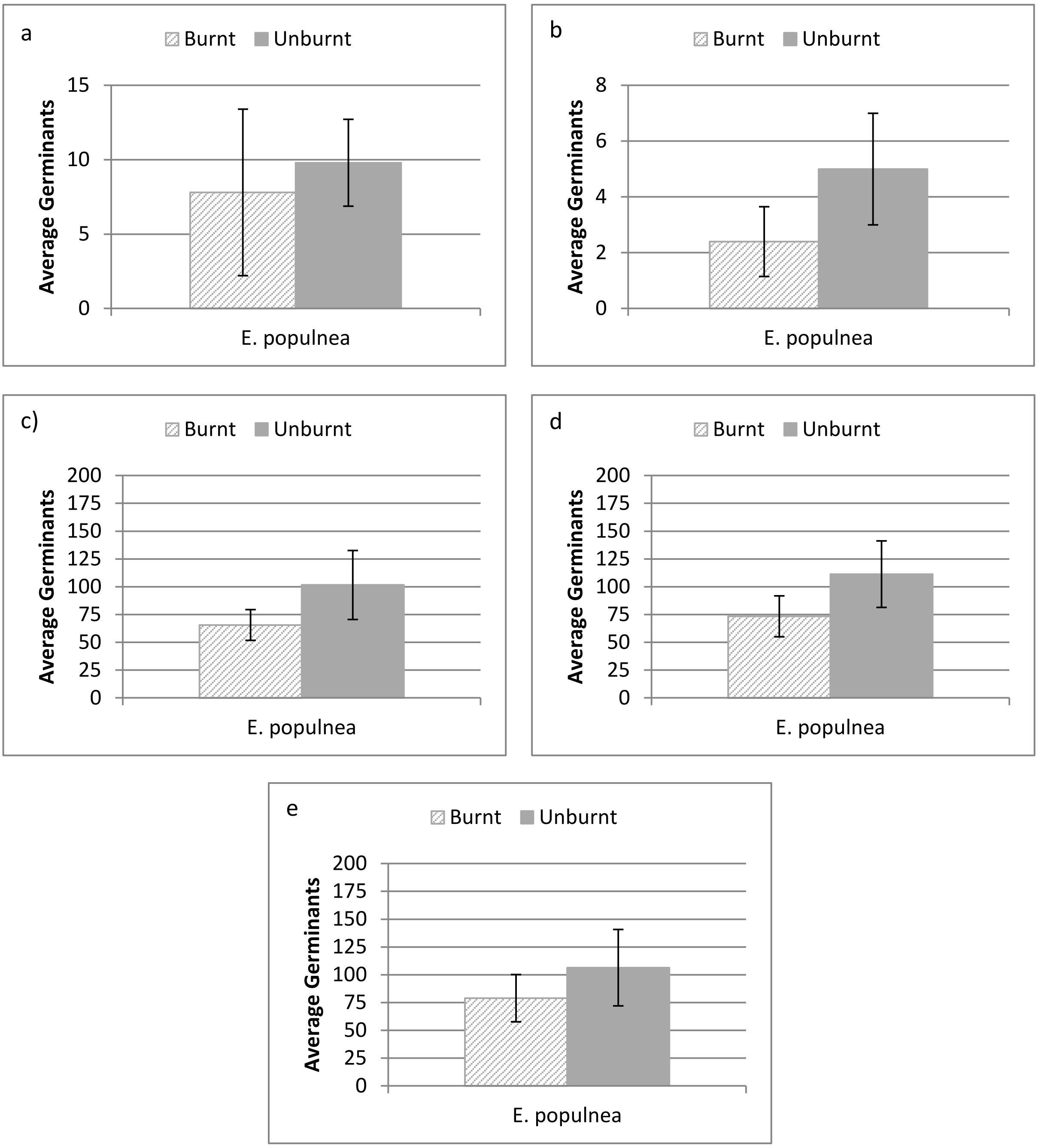

Figure 11: *E. populnea* average germinants of: a) monocot; b) lippia; c) dicot; d) total; e) total minus lippia (please note scale on all graphs) values are means; error bars are standard errors

Average accumulation curves for both woodland types show that the average number of germinants per tray plateaued after the initial flush of seedling emergence in the first three weeks of the trial (Figure 12). There were no significant differences (T-tests; p > 0.05) in seedling emergence between burnt and unburnt treatments in either woodland type This was true for seedling emergences for both Week 3 and Week 14 (Table 11; Appendix K).

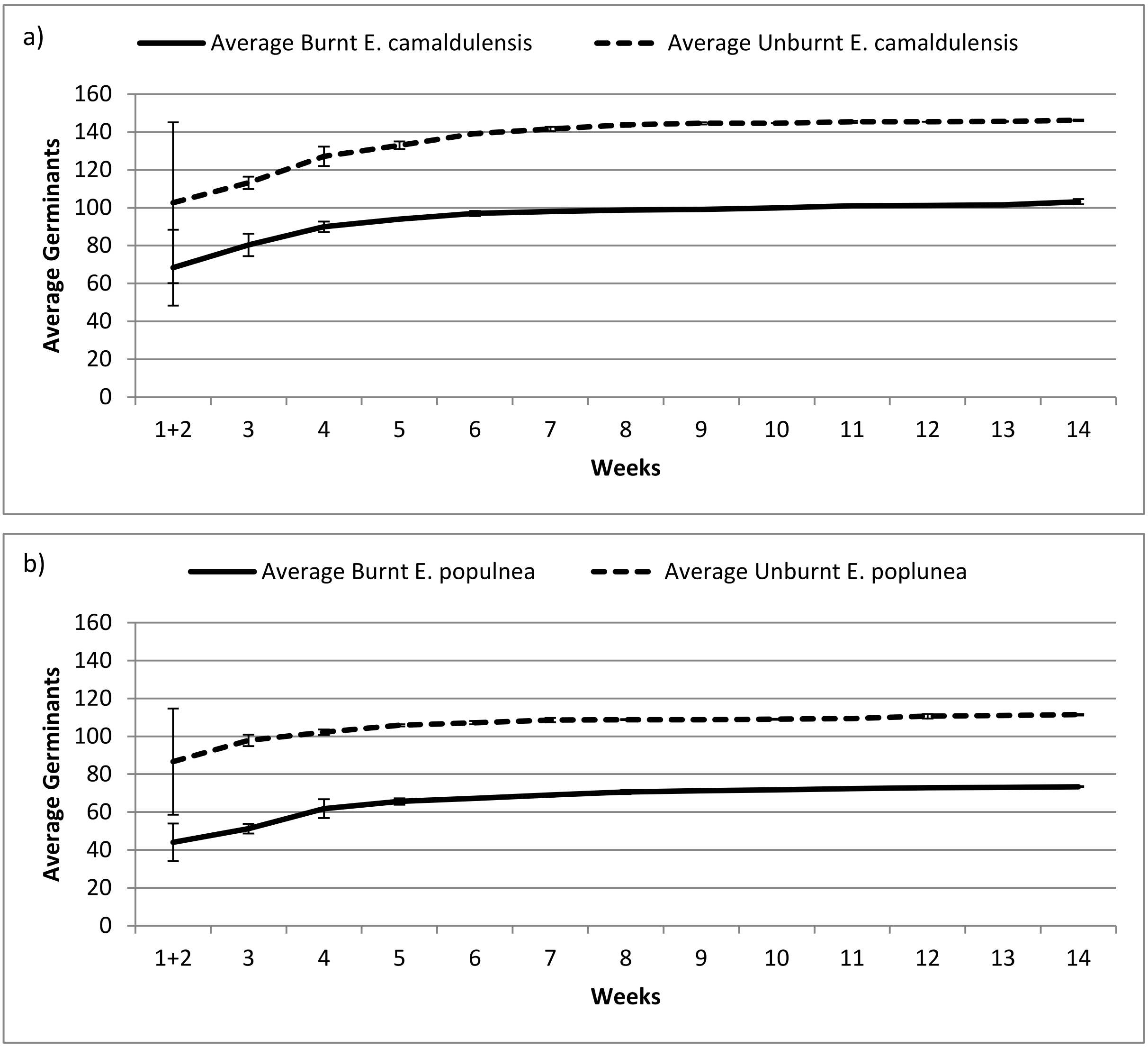

Figure 12: Average cumulative number of germinants in a) *E. camaldulensis* and b) *E. populnea* woodland soil samples; values are means; error bars are standard errors

Table 11: Summary data for glasshouse germinants in seedbank trial
(all means were *$\log_{10}(1+x)$* transformed before analysis)

	Levene's Test for Homogeneity		T-Value	df	Significance
	F stat	**Sig**			
***E. camaldulensis* Woodland (T=3)**					
Total Germinants	0.13	0.913	-1.617	8	0.145
Total Germinants (minus lippia)	0.013	0.912	-0.521	8	0.617
Lippia Germinants	0.174	0.687	-1.416	8	0.195
***E. populnea* Woodland (T=3)**					
Total Germinants	0.007	0.933	-0.600	8	0.565
Total Germinants (minus lippia)	0.44	0.840	-1.243	8	0.249
Lippia Germinants	0.054	0.822	-0.964	8	0.363
***E. camaldulensis* Woodland (T=14)**					
Total Germinants	0.007	0.935	-0.636	8	0.542
Total Germinants (minus lippia)	0.008	0.930	-0.619	8	0.396
Lippia Germinants	0.298	0.600	-1.000	8	0.347
***E. populnea* Woodland (T=14)**					
Total Germinants	0.040	0.847	-1.086	8	0.309
Total Germinants (minus lippia)	0.318	0.588	-0.729	8	0.487
Lippia Germinants	0.202	0.665	-0.899	8	0.395

T = weeks since start of germination trial

Average accumulation of lippia emergence shows a similar pattern to that of the total germinants (Figure 13). Burnt sites in both woodland types had approximately half the number of lippia seedlings compared to the unburnt sites. However, when statistically analysed this difference was not significant ($p > 0.05$) for either woodland (Table 11). In all treatments and woodland types there was an eventual plateauing of lippia germination.

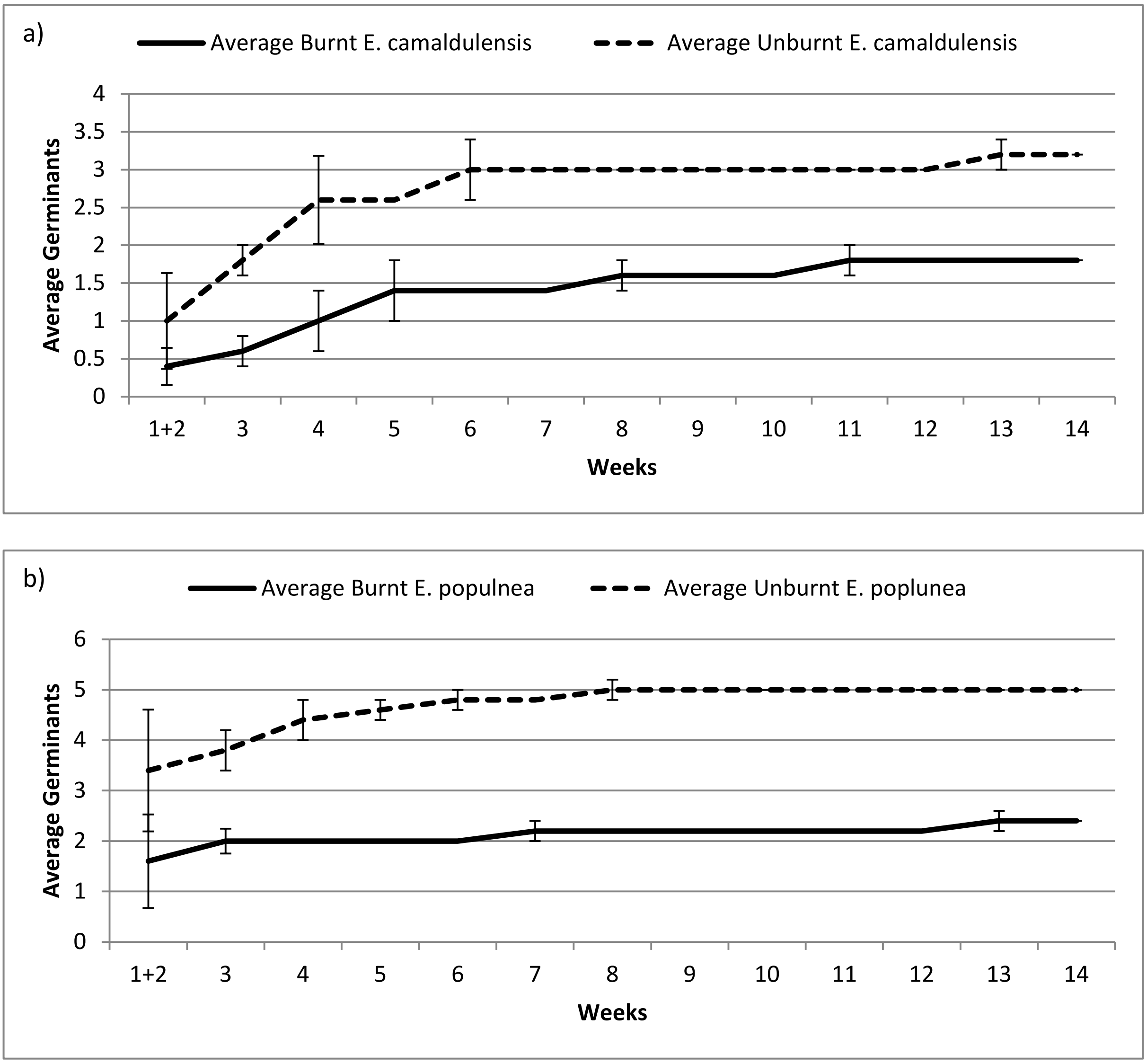

Figure 13: **Average accumulation curves for lippia in a) *E. camaldulensis* and b) *E. populnea* woodlands**; values are means; error bars are standard errors

3.3.2 Pseudo-Species Richness

Pseudo-species richness ranged from 5 pseudo-species to 25 pseudo-species across both woodlands and treatments. The average pseudo-species richness of burnt *E. camaldulensis* was 10.2 pseudo-species (SE ± 2.1) compared to the average pseudo-species richness of unburnt treatments of 17.8 pseudo-species (SE ± 2.6) (Figure 16a). The average pseudo-species richness of burnt treatments in *E. populnea* woodland was 15.4 pseudo-species (SE ± 1.9) compared to the unburnt richness of 12.6 pseudo-species (SE ± 2.2) (Figure 16b) (Appendix L).

The pseudo-species richness between the burnt and unburnt samples from the *E. camaldulensis* woodland was significantly different ($p < 0.05$), with burnt treatment plots exhibiting lower pseudo-species richness (see Figure 16a). However, when plot BC5 ($n = 5$) was removed from the data and the analysis re-run, the difference between the treatments was no longer significant ($p = 0.091$; $p > 0.05$). Pseudo-species richness was not significantly different ($p > 0.05$) between burnt and unburnt samples of the *E. populnea* woodland (Figure 16b and Appendix L).

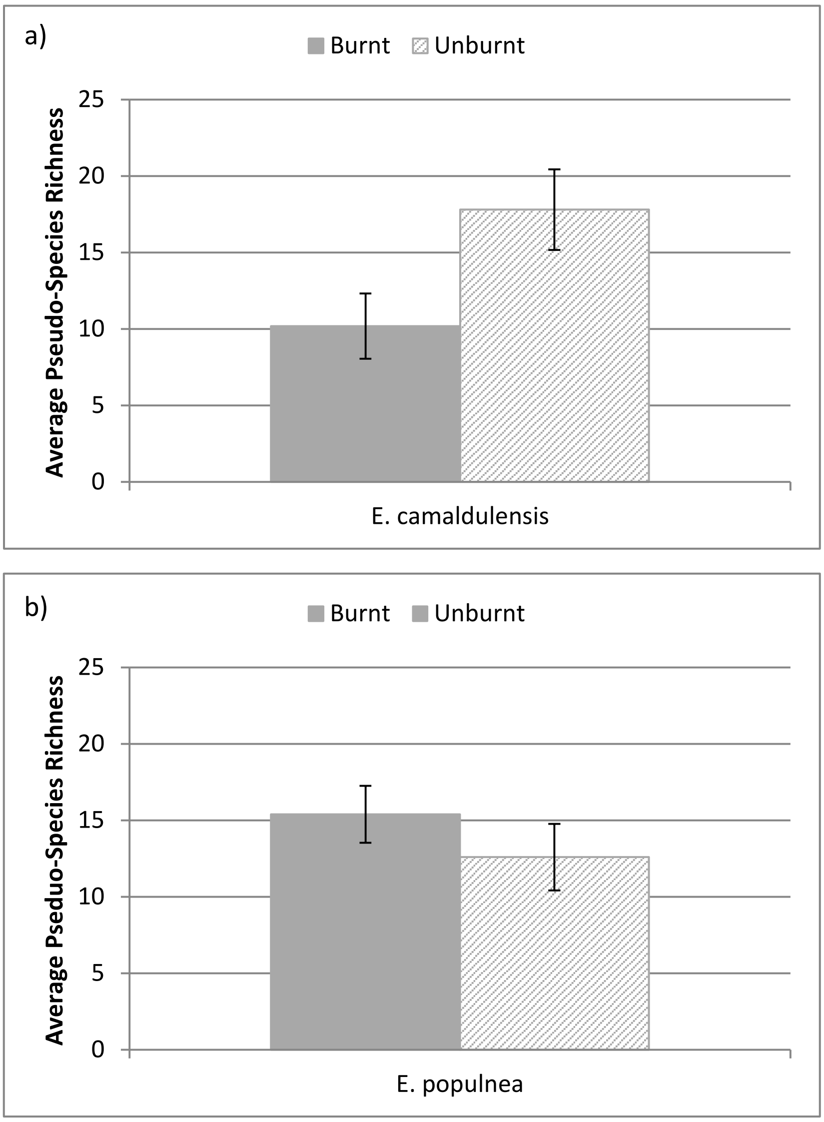

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Figure 14: Average pseudo-species richness of each woodland a) *E. camaldulensis*; b) *E. populnea*; values are means; error bars are standard error

Chapter 4: DISCUSSION

4.1 Similarity of Burnt/Unburnt Plots Prior to Fire

Disturbance (as determined by the subjective disturbance scores) was relatively uniform across both burnt and unburnt treatment types of both *Eucalyptus camaldulensis* and *Eucalyptus populnea* woodlands. The broad structural similarity of each site clearly shows that all treatments and woodlands were structurally similar. The uniformity of disturbance and dominant stand structure across all sites shows that any potential confounding effects have been reduced and that any main patterns can be attributed to fire, not experimental errors.

4.2 Vegetation Response

Structural components of the vegetation in both woodlands were not significantly different across the treatments. Personal observations soon after the fire indicated that tree mortality, stag mortality and fire scars had increased in burnt woodlands and therefore it was expected that a significant difference may be found in tree and stag structure. In a study of riparian woodlands in Kruger National Park, showed that two months after a low intensity fire in a riparian landscape, 33% of riparian trees had been killed by the burn. Overall, when the data was collected at St Ruth Reserve, trees were of a higher average frequency in unburnt plots over both woodlands, with minimal variation in stag cover between burnt and unburnt treatment in either woodland. No significant difference was found for either variable. The relatively small quadrat sizes used in the field (500 m^2), as well as the low number of replicates for each treatment type, could be influencing the lack of difference between the treatment types found here. An increase in the number of replicates, as well as an increase in the quadrat size could help capture any differences in vegetation response between burnt and unburnt treatments in either woodland.

An increase in bare ground in burnt plots was expected, as Kutt and Woinarski (2007) had found that bare ground increased significantly in burnt, grazed sites and burnt, ungrazed sites.

Schmalzer and Hinkle (1992) found that a prescribed burn in Palmetto scrub in Florida, increased mean bare ground cover from a 0 to 23 percent. Bare ground cover at St Ruth Reserve did not increase in burnt treatments as was observed in these other studies: there was no significant difference between burnt and unburnt treatments in either woodland. This lack of change could be due to St Ruth Reserve already being a highly disturbed system and therefore the ecological processes occurring within the reserve, such as increased edge effect in the remnant patch and fast lippia growth (Macdonald 2008), may have been influential on the cover of bare ground when the study was taken.

Forb cover has been observed in many studies to increase post fire. Laughlin *et al.* (2004) found that annual and biennial forb richness increased post fire with a significantly greater change rate than unburnt old-growth ponderosa pine forest sites. In a study observing the impact of grazing and fire on tropical savanna woodland vegetation, Kutt and Woinarski (2007) also found that there was a significant difference in forb cover over the four treatment combinations of: no fire and grazing; fire and grazing; no fire and ungrazed; and fire and ungrazed. Forb cover was most extensive in burnt and grazed treatments (Kutt & Woinarski 2007).

In the present study an increase in forb cover at burnt sites was observed at both the coarse 500 m^2 scale and the fine 70 cm^2 scale. This change was significant in *E. populnea* woodlands on a fine scale but not a coarse scale. The change was not significant on either scale in *E. camaldulensis* woodlands even though the average frequency ranged from 12.6 to 48.4 on a fine scale. It is important to note that lippia was not included in any forb count during this study and therefore analyses of forb frequency and cover is of forbs excluding lippia. This result agrees with the literature that shows that fire can change the ground cover abundance of forbs. The difference between burnt and unburnt forb abundance may have been increased further if Nardoo was included in the forb cover count rather than being included in the grass cover count. Nardoo, a fern, was originally included in the grass count due to its rhizomal growing form, similar to many grasses. However, on reflection it is now believed that this plant would have been better classified as a forb given its possible similar ecological function to that growth form.

A recent study in grasslands at Yarralumla, Australia, showed that there was a significant decline in native grass cover four months after more than 10% of the area was burnt (Ross & Macris 2012). White and Loftin (2000), whose study was conducted in the semiarid grasslands of New Mexico, found that although grass cover did decline post fire, it was not a significant decrease. In contrast, some studies show that there are no significant differences in grass cover between unburnt and burnt vegetation of sagebrush rangeland and grasslands, regardless of time since fire (Nelle *et al.* 2000; Drewa & Havstad 2001). The results of Nelle *et al.* (2000) and Drewa and Havstad (2001) are consistent with the results found at St Ruth Reserve as there was no significant difference in grass cover between burnt and unburnt treatments in either woodland. The contradictory findings of these studies may be due to the different ecosystem types being studied, seasonality of study or the type of data collection method being used.

The response of lippia to disturbance was documented in Macdonald (2008) where the soil was mechanically disturbed to a depth of 10 cm and vegetation cover, including lippia, was removed. Each season, one replicate at each site was mechanically disturbed. It was found that following spring disturbances, lippia reached 50% cover within 85 days of disturbance. This was not significantly different from the control plots (Macdonald 2008). Summer disturbance plots returned to control levels after 216 days (Macdonald 2008). However, winter disturbed plots remained significantly lower than control plots, indicating that lippia does not grow well in cooler conditions (Macdonald 2008). The disturbance in the current study occurred in late spring, so therefore the results could be correlated with the spring results of Macdonald (2008).

It was noted by Macdonald (2008) that the recovery of lippia cover in disturbed plots came from the surviving tap-roots, buried plant fragments and vegetative expansion from outside the disturbance areas. It is likely that these influences were present post fire and helped lead the regeneration of lippia after the November 2013 fire. Lippia is known to reduce the fire risk of invaded areas (Leigh & Walton 2004). This means that the prescribed fire may not have burnt the entire landscape as hoped it would, owing to the lack of flammable ground cover species, and therefore the burn may have resulted in patches of live lippia remaining in the landscape. The potential for propagule survival would have ensured that enough vegetative material was left behind for vegetative reproduction to commence as soon as the plant had recovered from the effects of the fire. The added potential for lippia seed germination could also contribute to the recovery of lippia cover post fire.

4.3 Seed Germination Response

Research from small fragmented Mallee habitat on Kangaroo Island that had not been burnt for more than thirty years, found that germination both native and introduced shrubs, grasses, forbs and non-grass graminoids, was not significantly different between unburnt and heat plus smoke treated samples (Davies *et al.* 2013). Although different vegetation types, the broad site ecology, highly fragmented remnant vegetation in an agricultural matrix, and previous use, grazing and other land uses, at the Kangaroo Island study are similar to that of St Ruth Reserve and therefore the findings are reflective of those found at St Ruth Reserve. A study by Auld and Denham (2006) found that there was a significant difference between the seed bank before and after a fire. However, the species studied in this study, *Acacia suaveolens, Grevillea speciosa* and *Grevillea buxiflora* (Auld & Denham 2006) were very different to those studied in the current study and in the Davies *et al.* (2013) study. These studies show that although there is some similarity throughout the previous literature to the current study, there are also conflicting studies which may indicate that seed response is dependent on the species' requirements for germination.

The post fire species richness of a site is important for the future vegetative composition of the ecosystem. A previous study found that there was no overall significant trend in species richness as a function of fire regimes or site effects in heathland vegetation (Bradstock *et al.* 1997). This result was supported by a study by Pettit and Naiman (2007), who also found no significant difference in the species richness between burnt and unburnt quadrats in a semi-arid riparian ecosystem in South Africa.

The current study partially followed this trend, with the pseudo-species richness of *E. populnea* plots not significantly different between burnt and unburnt treatments. The initial difference between burnt and unburnt treatment pseudo-species richness in *E. camaldulensis* samples was found to be significant. However, this significance is likely due to an outlier of very low pseudo-species richness in one replicate (BC5), which had a pseudo-species richness of five. High levels of lippia (average of 95%; Appendix M) at this site may have prevented other plants from emerging and setting seed within the area. Ironically, lippia was not one of the species that emerged in this tray. When the outlier was removed from the dataset and the data was reanalysed, the *E. camaldulensis* pseudo-species richness was not significantly different between burnt and unburnt plots. This result then follows the current literature on species richness post fire.

Overall, it does not appear that the prescribed burn had any significant effect on either the germination or pseudo-species richness of the soil seed bank at St Ruth Reserve, consistent with previous literature.

4.4 General Discussion

The results from this research show that there were no significant effects of the prescribed fire on lippia cover, vegetation structural components and the soil seed bank in a vegetation remnant in a highly modified agricultural landscape. The only exception to this conclusion is the significant increase of forbs in burnt *E. populnea* woodlands which was anticipated from the prevailing literature. In particular, this study shows that fire may not be a successful management tool to control lippia abundance.

This study showed that the vegetative structure of both woodlands and treatments at St Ruth Reserve were the same, even though it was anticipated that the tree and stag cover may have declined in burnt areas. This result allows the first hypothesis to be accepted, that there was no difference in vegetation structure post fire.

This study also shows that bare ground, grass or lippia cover were not significantly different in burnt woodlands compared to unburnt woodlands. The result for bare ground was inconsistent with previous literature. However, this could be attributed to the already highly disturbed state of the vegetation at St Ruth Reserve and therefore a difference may not have been detected using the preliminary study applied here. The literature is generally inconclusive as to whether fire induces a significant change in grass cover, although the result observed here for lippia was consistent with other lippia disturbance studies (e.g. Macdonald 2008).

The results of the ground cover study allow the second sub hypothesis, that there is *no difference in the abundance of other ground cover species between burnt and unburnt woodlands,* six months after a prescribed burn to be partially accepted in terms of grass cover. Forb cover at the fine scale of 70cm^2 was significantly different in *E. populnea* woodland but not in *E. camaldulensis* woodland, even though there was a large range of cover. The significant result was consistent with prevailing literature on forb growth post fire (Masocha *et al.* 2011). However, further research is needed on the apparent differential response of components of the ground cover and woodland differences in the *Eucalyptus* woodlands at St Ruth Reserve.

These results were reflected in the *ex situ* germination study. Germination trials showed that burning did not have any significant impact on the species richness or germination of soil seed banks, consistent with previous literature (Pettit & Naiman 2007; Davies *et al.* 2013). However, this could be extended by taxonomically identifying the species that germinated and comparing them to species found in the field at the same time post fire. The results of the germination trial allow the third and fourth sub-hypotheses of that there is *no significant difference in the seedling emergence or species richness of seeds stored within the soil seed bank* to be accepted.

The low germination of lippia seeds in the glasshouse trials was consistent with Macdonald (2008), who found that the most successful germination method could only germinate approximately 27% of seeds being tested. Adding to this hypothesis, approximately half the seeds (from two sites) used by Macdonald (2008) were not viable. This shows that the quick regeneration of lippia in summer, in the *in situ* study is most likely from vegetative fragments and taproots left behind after the fire (as per MacDonald 2008) and allows the hypothesis: that vegetative regeneration may be the preferred method of regeneration at St Ruth Reserve, to be drawn. These results could have an impact on the management decisions to be made regarding lippia's control (see Section 4.4.2).

Although this study has provided expected results, there are some experimental limitations. The study design could be improved by sampling both before and after the fire to achieve the greatest comparison. A design that would help to improve the current design of the present study is that of the MBACI design – multiple before, after, control, impact (Keough & Quinn 2000). The MBACI design increases statistical rigour and power through the use of multiple, randomly selected control and treatment locations (Miao *et al.* 2009). In turn, the ability to detect impacts from the treatments are increased and the confounding effects are minimised (Miao *et al.* 2009).

However, the opportunistic nature of the present study meant that sampling prior to burning was not possible. The comparatively simple spatial 'snap-shot' approach taken here is relatively common in fire ecology studies (Gosper *et al.* 2011; Lewis *et al.* 2012) although it does limit the generalisations that can be made (Williams *et al.* 1999). The nature of this study also meant that there were many time constraints and therefore there are temporal limitations to the study (see below).

4.4.1 Limitations and Future Directions

The independence of study plots in this study was limited due to each treatment (control and burnt) being nested within one large treatment plot (St Ruth Reserve). Ideally, the study would employ the MBACI design (see Section 4.4), multiple, independent sites where half the sites are burnt and half are kept as controls. More study sites would increase the statistical power and rigour of the study whilst enabling any differences between treatments to detected (Miao *et al.* 2009). Vegetative structure, ground cover and floristic surveys would be conducted at all sites both before and after the fire. As the study was opportunistic, it was not possible to have a control site, independent to the burnt site of St Ruth Reserve. Therefore, conclusions made from this study are specifically for St Ruth Reserve and may limit generalisations (Woinarski *et al.* 2004). This improved experimental design would help to further identify whether fire is not a successful control method for lippia and whether there are any changes in vegetation structure and ground cover.

Time-since-fire has an important role in the change of vegetation structure. Morrison *et al.* (1995) found that within the first year after a fire, early-colonising annuals comprised half the vegetation in a Mediterranean-climate chaparral and scrub. Herbs, annuals and small shrubs have been found to decrease in abundance with increasing-time-since fire, with vegetation moving back toward the vegetation structure of an unburnt site (Morrison *et al.* 1995; Keeley *et al.* 2005). These results show that the time since fire does impact upon the vegetative community. The ‘snapshot study’ here sampled only six months post fire and therefore may only be indicative of early colonisers, not the full spectrum of present species or richness. This small timeframe limits the conclusions able to be made about the full complement of species and changes that may occur within a lippia invaded ecosystem. However, invasive species such as lippia could potentially stop the natural vegetative progression of the site due to the aggressive mat growth form and extensive root system of the invasive weed (Earl 2003). This study is a good indicator of the changes that can occur six months after fire, especially as the results found in this study were consistent with the previous literature.

The extension of the monitoring time frame would help researchers and managers to understand the vegetative succession of the reserve post fire. Some studies have extended to a twenty-three year period (Woinarski *et al.* 2004), with other studies re-examining sites twelve months (Pettit & Naiman 2007) or thirty months post fire (Milberg & Lamont 1995). Follow up studies allow the natural vegetation succession to be followed over a longer period of time, rather than the snapshot provided in this study.

A successor project would be able to document the changes in lippia growth as well as vegetation cover and ground cover in respect to time since fire at St Ruth Reserve. The impact of other environmental factors such as drought and flood, which appear to influence lippia cover (Macdonald 2008), could also be included in a longer time frame.

The short time frame of this study also limited the length of time that the glasshouse trials could be run. In previous literature, seedling emergence has been monitored for 24 weeks (Gashaw *et al.* 2002) and 30 weeks (Hill & French 2003). After the 24 weeks, Gashaw *et al.* (2002) removed all germinants, disturbed the soil and then left the tray for another month for further germination to occur. Even though germination did plateau in this study, the effect of the season, in particular the strength of light over the autumn/winter period in which the study was run, may mean that with changed conditions in spring/summer there may be a period of renewed germination in the new season. Either 24 or 30 weeks would allow for multiple seasons to occur and for seeds with varying germination conditions to emerge.

To encompass the wider effects of the fire, such as larger changes in ecosystem structure, for example burnt and fallen stags, a larger quadrat size would be required to encompass all potential changes post fire. This meant that in this research study, the broad scale vegetation effects were not able to be determined and are therefore their response to fire is unknown. The size quadrat used in this study (500 m^2) was large enough to encompass coarse changes in this small area. A common size used in fire studies is 1000 m^2 (0.1 ha) (Keeley *et al.* 2005; Ainsworth & Kauffman 2013). Larger quadrat sizes have been used including 2500 m^2 (0.25 ha) (Woinarski *et al.* 2004) and 3600 m^2 (0.278ha) (White & Loftin 2000). To enable future studies to assess as much vegetative structural and broad scale vegetation change post fire as possible, it is recommended that a much larger study quadrat, such as those used by Woinarski *et al.* (2004) and White and Loftin (2000), be used. However, the 500 m^2 quadrat used in this study was sufficient to measure all variables required.

A future study into the floristic composition of the ground cover would allow more subtle differences between burnt and unburnt treatments to be explored. Floristics studies are often included in fire ecology studies to determine the full extent and impact of the fire in the field (e.g. Milberg & Lamont 1995; Bramston 2003)

Taxonomic identification could also be completed if the glasshouse trials were able to continue beyond the 14 weeks of this study. The continuation of the trial would greatly assist in determining whether many of the germinated species are in fact native or invasive. After monitoring seedling emergence for 30 weeks, Hill and French (2003) allowed the trays to be left in the glasshouse for another eight months to allow plants to flower. This allowed further identification of the species to occur (Hill & French 2003). If the time was available, this type of extended glasshouse trial would help to identify what species have emerged as well as whether the germinants are native or invasive species. The continuation of the glasshouse trials would then create a broader picture of the seed bank at St Ruth Reserve. The taxonomic identification of the species would also create an opportunity to be able to compare this data to the published literature more closely than has been possible in this study. This would allow the species richness and composition of the field study to be compared to the species richness and composition of the glasshouse trials.

The soil samples used in the germination trials were collected six months after the prescribed burn, which meant that some species may have had an opportunity to recolonise the seed bank before the soil was sampled. In a future study it would be beneficial to sample the seed bank immediately after the fire and then each season after the fire (Gashaw *et al.* 2002), to determine if there is a difference in post fire response in relation to time since fire. Given time constraints for this study and that the vegetation field study was taken at the same time as the soil for the glasshouse trial, it can be seen that the seed bank is a reflection of the current state of the above ground vegetation at St Ruth Reserve.

4.4.2 Management Implications

Prescribed fires may not always enhance the native species of the area. Penman and Towerton (2008) reported that in an autumn fire, only two plots out of forty, reached between 80°C and 100°C, which is considered to be the necessary trigger for germination in many Australian fire-adapted plants. It has been found that the mean number of species in a study in a Cumberland Plain Woodland near Sydney, is significantly different in treatments that have been exposed to 80°C heat compared to unheated samples with litter left on top of the seed trays (Hill & French 2003). Grasses, graminoids, herbs and shrubs all germinated in the Hill and French (2003) study. These findings may provide an insight into the future native species succession or lack thereof, at St Ruth Reserve. If managers were to continue to burn the reserve, the potential lack of native germinants due to the low temperature of the fire, may increase the number of invasive species in the area, in particular species that can successfully reproduce vegetatively, such as lippia.

The results for lippia cover at St Ruth Reserve indicate that the use of fire has not been successful in controlling lippia. The fast growth rate from vegetative reproduction and previous literature also showing fast vegetative regrowth after disturbance (Macdonald 2008) helps to support the hypothesis that disturbance such as fire, may not be effective in the management of this invasive species. Therefore, it is not recommended that the Western Downs Regional Council continue to pursue the burning of St Ruth Reserve, with the sole purpose of controlling lippia. The broader ecological impacts that may not have been enumerated in this study, including the decline of trees and stags post fire (Pettit & Naiman 2007), also need to be considered if burning is contemplated again.

4.5 Conclusion

This research has shown that there are no significant effects of a prescribed fire on lippia, vegetation structural components (except for a significant increase of forbs in burnt *Eucalyptus populnea* woodlands) or the soil seed bank at St Ruth Reserve. There have been no studies to date that have documented the change in lippia cover and abundance post fire in Australia. These results are consistent with previous studies that have examined the effects of fire on vegetation cover. This study also shows that fire does not have any significant impact on the seedling emergence or pseudo-species richness of the soil seed bank, six months after a prescribed burn. These conclusions are significant in that they show that fire is not an effective control method for lippia and therefore it is not recommended to be used as a management tool within the St Ruth Reserve in the future until other possible detrimental ecosystem effects are researched. Further research needs to be conducted to expand this experiment both temporally and spatially so that recommendations and conclusions can be expanded to the wider distribution of lippia and vegetative remnants.

Chapter 5: REFERENCES

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Appendices

Appendix A: GPS data points for sites

		South	East
BC1	Burnt *E. camaldulensis* rep 1	27o20'0.3722"	151o14'38.037"
BC2	Burnt *E. camaldulensis* rep 2	27o20'0.140"	151o14'43.766"
BC3	Burnt *E. camaldulensis* rep 3	27o20'8.91"	151o14'46.6"
BC4	Burnt *E. camaldulensis* rep 4	27o19'58.577"	151o14'34.784"
BC5	Burnt *E. camaldulensis* rep 5	27o19'49.711"	151o14'45.254"
UC1	Unburnt *E. camaldulensis* rep 1	27o19'45.522"	151o14'47.527"
UC2	Unburnt *E. camaldulensis* rep 2	27o19'42.642"	151o14'11.198"
UC3	Unburnt *E. camaldulensis* rep 3	27o19'51.845"	151O14'13.55"
UC4	Unburnt *E. camaldulensis* rep 4	27o19'59.232"	151o14'17.177"
UC5	Unburnt *E. camaldulensis* rep 5	27o20'3.230"	151o14'20.346"
BP1	Burnt *E. populnea* rep 1	27o20'8.5"	151o14'50.83"
BP2	Burnt *E. populnea* rep 2	27o20'6.621"	151o14'54.819"
BP3	Burnt *E. populnea* rep 3	N/A	N/A
BP4	Burnt *E. populnea* rep 4	27o19'52.867"	151o14'59.619"
BP5	Burnt *E. populnea* rep 5	27o19'46.995"	151o14'59.08"
UP1	Unburnt *E. populnea* rep 1	27o19'43.94"	151o14'49.089"
UP2	Unburnt *E. populnea* rep 2	27o19'47.255"	151o14'38.830"
UP3	Unburnt *E. populnea* rep 3	27o19'44.471"	151o14'33.142"
UP4	Unburnt *E. populnea* rep 4	27o19'46.039"	151o14'28.957"
UP5	Unburnt *E. populnea* rep 5	27o19'37.853"	151o14'23.142"

Appendix B: Toothpick colours for week of emergence

Toothpick Colour	Week
No Colour	1&2
Purple	3
Purple & Red	4
Watermelon	5
Red	6
Fluoro Orange	7
Devine	8
Green	9
Nude	10
Gold	11
Gold & Purple	12
Red & Green	13
No Toothpick	14

Appendix C: Pseudo-species list

Pseudo-species name		
Stumpy Bright Green Weed	Chive Thing	Big Segmented Fern
Lippia	Eucalypt	Frilly Weed
Segmented Leaf	Upright Woolley	Milk Weed Look-a-Like
Red/Green Tree	Fuzzy Thin Leaf	Little Bright Green
Carrot Top	Weird Succulent	Woolley Tree
Milk Weed	Thick Stem Dark Green Weed	4 o'clock
Small Clover	Daisy	Large Clover
Lime Green Weed	Singled Leaf	Thick Dark Green Carrot Top
Dew Rooty Tall	Sword Plant	Thorny Weed
Unidentified Dicot	4 o'clock	Woolley Weed
Purple Backed Thistle	Sage	Purple Carrot Top
Wahlenbergia	Small Clover	Smells Like Rosemary
Rose Weed	Thick Carrot Top	Thin Woolley
Upright Woolley with Flower	Fluffy with Weird Flower	Red/Green Succulent
Grasses	Tall Skinny Leaf	Upright Woolley with Flower
Thick Woolley	Segmented Leaf	Purple Single Leaf
Thin Woolley with Flowers	Purple Backed Thistle	Woolley (dead)
Smells Like Rosemary	Furry Small Tree	Thorned Strawberry
Heavily Segmented Fern	Glossy Green Weed	Cabbage Weed
Maple Carrot	Rambling Oak	Chive Thing
Upright Woolley	Rose Weed	Little Bright Green
Dark Green Fuzzy (with Flowers)	Fluffy Carrot Top with Weird Flower	

Appendix D: Raw data from disturbance scores

Site	Grazing	Clearing	Logging	Erosion	Weeds	Feral Animals	Soil Compaction	Evidence of Dieback
BC1	2	2	2	0	3	0	0	2
BC2	1	1	2	0	2	0	0	2
BC3	2	1	3	0	2	0	0	3
BC4	2	1	1	0	2	0	0	2
BC5	1	1	1	0	3	0	0	1
UC1	2	2	3	0	3	0	0	1
UC2	1	1	3	0	3	0	0	3
UC3	1	2	1	0	2	0	0	3
UC4	1	2	1	0	4	1	0	2
UC5	2	1	2	0	3	0	0	1
BP1	1	3	2	0	2	0	0	3

BP2	1	1	2	0	3	0	0	3
BP3	0	2	2	0	3	1	0	2
BP4	1	2	3	0	2	0	0	2
BP5	1	2	2	0	3	0	0	2
UP1	2	2	3	0	1	0	0	1
UP2	2	3	3	0	3	1	0	2
UP3	2	2	2	0	2	0	0	1
UP4	1	1	2	0	1	0	0	1
UP5	1	1	0	0	2	0	0	1

Appendix E: Raw data from 500 m^2 quadrat - Raw Data

	Trees >20m	Trees 10-20m	Trees <10m	Total Trees	Tree Cover	Standing Stag >20m	Standing Stag 10-20m	Standing Stag <10m	Total Stags	Stumps	Stag Cover	Stump Cover	Logs: T-2F^1	Logs: 2F-2H^1	Logs: 2H-1A^1	Logs: 1A-2A^1	Total Logs	Log Cover	Grasses/ Sedges	Herbs/Forbs	Lippia	Fine Litter	Lippia Litter	Bare ground
Units	Density (no. per 500 m^2)				%	Density (no. per 500 m^2)					%	%	Density (no. per 500 m^2)					%	%	%	%	%	%	%
BC1	0	1	8	9	4	1	0	1	2	0	0.5	0	20	21	19	4	64	2	10	1	85	1.5	0	0.5
BC2	0	0	0	0	0	0	1	2	3	2	0.5	0.5	23	18	8	0	49	0.5	55	35	4	0.1	0	1
BC3	0	1	0	1	1	0	0	0	0	0	0	0	5	5	2	1	13	1	90	2	4	1	0	0.5
BC4	1	0	5	6	2	0	0	0	0	1	0	0.5	17	2	3	1	23	0.5	75	2	3	0.5	0	0.5
BC5	0	3	8	11	7	0	0	0	0	2	0	0.5	89	24	5	1	119	1	5	0.5	85	1	0	0.5
UC1	0	1	0	1	1	1	0	0	1	3	0.5	0.1	2	5	12	3	22	1	30	1	85	0.5	1	0
UC2	1	2	0	3	2	0	0	0	0	3	0	0.5	5	2	4	4	15	0.5	50	3	25	0.5	0	0.1
UC3	0	0	4	4	1	0	0	0	0	2	0	0.5	2	1	1	1	5	0.5	85	2	4	1	0	0.5
UC4	0	1	20	21	6	0	0	1	1	0	0.5	0	4	14	2	3	23	1	55	2	60	0.5	0	0
UC5	0	1	15	16	2	0	0	0	0	0	0	0	24	6	3	2	35	0.5	75	15	20	0.5	0	0
BP1	1	0	0	1	2	0	0	1	1	0	0.5	0	12	6	0	4	22	3	60	10	25	0.5	0	1
BP2	1	0	0	1	1	0	0	1	1	1	0.5	0.1	4	1	0	1	6	1	85	2	10	1	0	2
BP3	1	2	0	3	10	0	0	0	0	0	0	0	19	13	1	2	35	3	85	1	30	3	0	5
BP4	2	0	0	2	2	0	0	0	0	0	0	0	34	6	4	2	46	0.9	50	5	30	0.1	0	0.1
BP5	0	1	3	4	3	0	0	0	0	2	0	0.5	42	23	2	1	68	1	20	25	50	0.5	0	0.5
UP1	0	1	0	1	2	0	0	0	0	0	0	0	12	6	0	0	18	0.5	60	7	60	0.5	0.5	0.1
UP2	1	0	1	2	3.1	0	0	0	0	1	0	0.1	10	16	4	2	32	1	30	0.5	70	1	0.1	1
UP3	0	2	11	13	4	0	0	0	0	0	0	0	159	20	7	1	187	2	30	0.5	15	1	0.1	0.5
UP4	0	1	3	4	2.5	0	0	1	1	2	0.1	0.1	19	13	16	8	56	2	90	0.5	0.5	1	0	0
UP5	0	1	8	9	9	0	0	2	2	0	0.5	0	25	13	6	3	47	1	30	0.1	40	3	1	0.5

[1] **These measurements have now been converted to the following: T–2F = 6 –13 cm; 2F–2H = 13–35 cm; 2H – 1A = 35–90 cm 1A– 2A = 90 – 150cm**

Appendix F: Statistics from 500m^2 density data

		Levene's test for equality of variances		T-test for Equality of Means						
		F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	95% Confidence Interval of the Difference	
									Lower	Upper
Tree – *E. camaldulensis*	Equal variances assumed	.105	.754	.659	8	.528	.18992	.28815	-.47456	.85441
Trees – *E. populnea*	Equal variances assumed	2.846	.130	1.410	8	.196	.24861	.17631	-.15796	.65517
Stags – *E. camaldulensis*	Equal variances not assumed	10.036	.013	-.625	6.229	.554	-.09542	.15263	-.46560	.27475
Stags – *E. populnea*	Equal variances assumed	1.317	.284	.285	8	.783	.03522	.12367	-.24997	.32041
Stumps – *E. camaldulensis*	Equal variances assumed	1.703	.228	.485	8	.641	.08519	.17579	-.32018	.49057
Stumps – *E. populnea*	Equal variances assumed	.000	1.000	.000	8	1.000	.00000	.14041	-.32379	.32379
Logs – *E. camaldulensis*	Equal variances assumed	.477	.509	-1.747	8	.119	-.36738	.21025	-.85222	.11746
Logs – *E. populnea*	Equal variances assumed	.044	.839	1.039	8	.329	.24689	.23758	-.30097	.79476

Appendix G: Statistics for 500m^2 cover data

	Levene's Test for Equality of Variances	T-test for Equality of Means

		F	Sig.	t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference	95% Confidence Interval of the Difference	
									Lower	Upper
Trees – *E. camaldulensis*	Equal variances assumed	.267	.619	1.096	8	.305	.01722	.01571	-.01900	.05344
Trees – *E. populnea*	Equal variances assumed	.116	.742	.391	8	.706	.00802	.02053	-.03931	.05535
Stags – *E. camaldulensis*	Equal variances assumed	1.833	.213	-.512	8	.622	-.00080	.00156	-.00440	.00280
Stags – *E. populnea*	Equal variances assumed	.000	1.000	.000	8	1.000	.00000	.00173	-.00399	.00399
Stumps – *E. camaldulensis*	Equal variances not assumed	35.200	.000	-1.521	4.357	.197	-.00180	.00118	-.00498	.00138
Stumps – *E. populnea*	Equal variances assumed	1.833	.213	-1.152	8	.282	-.00180	.00156	-.00540	.00180
Logs – *E. camaldulensis*	Equal variances not assumed	8.395	.020	1.852	5.297	.120	.00600	.00324	-.00219	.01419
Logs – *E. populnea*	Equal variances not assumed	7.334	.027	1.372	6.213	.218	.00780	.00569	-.00600	.02160
Grass – *E. camaludlensis*	Equal variances assumed	.311	.592	-.588	8	.573	-.11947	.20319	-.58802	.34908
Grass – *E. populnea*	Equal variances assumed	.725	.419	.539	8	.604	.14001	.25952	-.45845	.73847
Herbs/Forbs – *E. camaldulensis*	Equal variances assumed	1.336	.281	-.982	8	.355	-.02890	.02945	-.09681	.03900
Herbs/Forbs – *E. populnea*	Equal variances assumed	.609	.458	.050	8	.962	.00406	.08190	-.18480	.19292
Lippia – *E. camaldulensis*	Equal variances assumed	.391	.549	-.147	8	.887	-.03351	.22840	-.56020	.49318
Lippia – *E. populnea*	Equal variances not assumed	30.448	.001	-.526	4.637	.623	-.13123	.24931	-.78751	.52504
Lippia litter – *E. camaldulensis*	Equal variances assumed	.003	.958	.513	8	.622	.00140	.00273	-.00490	.00770
Lippia litter – *E. populnea*						N/A				
Fine Litter – *E. camaldulensis*	Equal variances assumed	3.467	.100	1.565	8	.156	.00700	.00447	-.00331	.01732
Fine Litter – *E. populnea*	Equal variances assumed	1.152	.314	.352	8	.734	.00200	.00568	-.01110	.01510
Bare ground – *E. camaldulensis*	Equal variances assumed	1.580	.244	1.485	8	.176	.00300	.00202	-.00166	.00766
Bare ground – *E. populnea*	Equal variances not assumed	5.914	.041	1.265	4.103	.273	.01120	.00885	-.01314	.03555

Appendix H: Raw data from 70 cm^2 quadrats

Quadrat	Q1	Q2	Q3	Q4	Q5	Q6	Q7	Q8
BC1								
Lippia %	95	60	99	98	85	95	90	15

Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	64	63	64	64	64	64	64	45
Lippia (/81)	69	53	76	71	68	76	65	27
Grasses (/64)	46	26	31	11	24	40	60	64
Forbs (/64)	15	41	16	11	27	45	28	10
BC2								
Lippia %	10	30	30	80	15	0.1	25	40
Lippia litter %	0	0.1	0	0	0.1	0	0.5	0.5
Lippia (/64)	26	42	37	64	31	6	59	59
Lippia (/81)	10	25	23	55	15	1	22	29
Grasses (/64)	30	25	41	43	61	57	62	64
Forbs (/64)	59	55	51	59	14	67	26	56
BC3								
Lippia %	25	10	80	75	30	65	55	10
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	44	24	56	60	34	59	63	41
Lippia (/81)	29	15	65	45	27	59	52	16
Grasses (/64)	61	56	42	34	45	25	30	51
Forbs (/64)	13	29	30	62	61	57	60	28
BC4								
Lippia %	50	25	6	3	0	10	85	95
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	52	38	45	40	0	51	64	64
Lippia (/81)	38	15	18	9	0	30	59	75
Grasses (/64)	42	35	44	59	64	64	61	52
Forbs (/64)	60	52	46	48	42	6	17	12
BC5								
Lippia %	98	98	97	99	99	100	99	75
Lippia litter %	0.1	0	0	0	0	0	0	1
Lippia (/64)	64	64	64	64	64	64	64	64
Lippia (/81)	79	79	78	78	73	79	76	65
Grasses (/64)	7	5	23	5	9	8	16	44
Forbs (/64)	2	1	17	20	28	17	16	
UC1								
Lippia %	95	40	90	95	90	70	98	70
Lippia litter %	0	3	5	2	7	3	0.1	5
Lippia (/64)	64	64	64	64	64	64	64	64
Lippia (/81)	71	60	70	67	62	60	78	61
Grasses (/64)	33	45	36	38	8	34	9	12
Forbs (/64)	19	2	32	21	8	3	6	14
UC2								
Lippia %	85	25	25	30	45	40	5	0.1
Quadrat	**Q1**	**Q2**	**Q3**	**Q4**	**Q5**	**Q6**	**Q7**	**Q8**
Lippia litter %	0	40	15	15	10	1	5	0
Lippia (/64)	63	32	59	56	57	57	18	20
Lippia (/81)	69	28	50	51	52	47	12	8
Grasses (/64)	29	32	42	45	55	64	64	64
Forbs (/64)	19	48	36	44	33	29	20	2
UC3								

Lippia %	45	1	25	60	0	35	8	2
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	55	23	57	64	0	59	39	28
Lippia (/81)	45	10	31	68	0	45	21	19
Grasses (/64)	36	55	41	29	57	49	49	47
Forbs (/64)	41	43	36	20	20	25	44	64
UC4								
Lippia %	95	100	75	45	98	90	70	10
Lippia litter %	0	0	0	0	0.5	0	0	0
Lippia (/64)	64	64	64	62	64	64	64	39
Lippia (/81)	78	80	72	51	77	79	72	35
Grasses (/64)	24	30	64	64	28	30	64	64
Forbs (/64)	35	37	21	7	17	46	32	0
UC5								
Lippia %	95	6	95	65	50	5	0.5	95
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	64	64	64	64	54	40	6	64
Lippia (/81)	79	39	67	47	30	8	0	76
Grasses (/64)	33	57	57	53	45	64	62	43
Forbs (/64)	4	3	18	27	48	19	37	31
BP1								
Lippia %	50	85	0.5	0.1	3	45	25	50
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	57	61	11	1	35	48	51	46
Lippia (/81)	55	66	5	0	30	46	38	46
Grasses (/64)	36	59	57	57	63	55	44	35
Forbs (/64)	44	35	36	45	37	25	54	55
BP2								
Lippia %	2	75	1	99	0	0	6	0.5
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	18	64	29	64	0	0	17	6
Lippia (/81)	13	64	20	80	0	0	16	6
Grasses (/64)	59	39	50	60	59	54	41	40
Forbs (/64)	41	37	59	41	62	62	59	56
BP3								
Lippia %	0	0	0.1	97	25	100	40	100
Lippia litter %	0	0	0	0	0	0	0	0
Lippia (/64)	0	0	1	63	51	64	60	64
Lippia (/81)	0	0	1	65	50	78	56	81
Quadrat	**Q1**	**Q2**	**Q3**	**Q4**	**Q5**	**Q6**	**Q7**	**Q8**
Grasses (/64)	63	62	64	52	51	49	59	13
Forbs (/64)	35	41	50	25	49	25	33	0
BP4								
Lippia %	35	30	50	97	30	75	15	25
Lippia litter %	0	0.1	0.1	0	0	0.1	0	0
Lippia (/64)	58	49	62	64	58	64	40	54

Lippia (/81)	57	42	52	80	34	68	22	34
Grasses (/64)	57	54	58	44	55	35	56	53
Forbs (/64)	28	15	24	8	29	30	57	63
BP5								
Lippia %	75	10	80	40	65	99	50	15
Lippia litter %	0.1	0.1	0	0.5	0.1	0	0	0.1
Lippia (/64)	64	28	63	58	61	64	63	55
Lippia (/81)	68	9	67	42	61	81	49	24
Grasses (/64)	32	46	36	31	26	14	30	36
Forbs (/64)	43	55	20	20	36	52	35	50
UP1								
Lippia %	11	0.1	40	1	5	20	25	80
Lippia litter %	0	0	0.5	0	0	0	0.1	0.5
Lippia (/64)	40	14	57	21	39	53	51	64
Lippia (/81)	24	4	35	12	22	26	34	56
Grasses (/64)	53	63	47	48	39	41	51	53
Forbs (/64)	26	20	30	57	48	26	52	45
UP2								
Lippia %	75	35	10	90	50	95	25	30
Lippia litter %	0.1	0.1	1	1	2	0.5	0	0
Lippia (/64)	63	60	54	64	64	64	50	49
Lippia (/81)	67	56	40	64	62	71	43	45
Grasses (/64)	32	16	17	7	2	17	56	57
Forbs (/64)	31	24	24	31	18	2	10	2
UP3								
Lippia %	3	15	2	5	1	65	55	25
Lippia litter %	0	0	0	0	0	0.1	0	0.1
Lippia (/64)	19	33	19	20	9	64	64	58
Lippia (/81)	9	6	2	8	2	52	43	31
Grasses (/64)	38	44	47	43	38	33	45	37
Forbs (/64)	61	43	41	43	17	5	4	13
UP4								
Lippia %	12	6	3	0.5	4	8	12	15
Lippia litter %	0	0	0	0	0	0	0	0.5
Lippia (/64)	41	52	26	6	28	40	52	62
Lippia (/81)	16	11	6	1	9	9	22	42
Grasses (/64)	63	62	60	64	63	59	62	56
Forbs (/64)	46	16	10	17	0	3	1	13
UP5								
Lippia %	60	28	28	80	25	15	75	1
Lippia litter %	2	1	2	1	1	0.1	1	0
Lippia (/64)	62	49	59	64	57	52	64	30
Lippia (/81)	55	29	41	56	47	26	69	5
Quadrat	**Q1**	**Q2**	**Q3**	**Q4**	**Q5**	**Q6**	**Q7**	**Q8**
Grasses (/64)	17	29	35	25	22	37	17	52
Forbs (/64)	44	44	29	24	31	10	3	3

Appendix I: Statistics for 70 cm^2 data

		Levene's Test for Equality of Variances		T-test for Equality of Means						
		F	Sig.	t	df	Sig (2-tailed)	Mean Difference	Std. Error Difference	95% Confidence Interval of the Difference	
									Lower	Upper
Lippia - *E. camaldulensis*	Equal variances assumed	1.212	.303	.376	8	.716	.08837	.23484	-.45318	.62991
Lippia – *E. populnea*	Equal variances assumed	.590	.464	1.213	8	.260	.12136	.10008	-.10941	.35214
Lippia litter – *E. camaldulensis*	Equal variances not assumed	6.770	.032	-1.314	4.002	.259	-.02733	.02080	-.08506	.03041
Lippia litter – *E. populnea*	Equal variances assumed	15.552	.004	-1.745	4.105	.154	-.00335	.00192	-.00863	.00193
Lippia (/64) – *E. camaldulensis*	Equal variances assumed	.131	.726	-.086	8	.933	-.02026	.23503	-.56224	.52171
Lippia (/64) – *E. populnea*	Equal variances assumed	.457	.518	-.327	8	.752	-.05726	.17521	-.46129	.34677
Lippia (/81) – *E. populnea*	Equal variances assumed	1.797	.217	.912	8	.389	.11382	.12487	-.17412	.40177
Lippia (/81) – *E. camaldulensis*	Equal variances assumed	1.109	.323	-.125	8	.903	-.02681	.21390	-.52006	.46644
Grass/sedge – *E. camaldulensis*	Equal variances assumed	.488	.505	-.569	8	.585	-.08754	.15376	-.44212	.26704
Grass/sedge – *E. populnea*	Equal variances assumed	1.954	.200	.547	8	.599	.09626	.17587	-.30929	.50181
Forbs/herbs – *E. camaldulensis*	Equal variances assumed	1.836	.212	1.209	8	.261	.15352	.12693	-.13918	.44622
Forbs/herbs – *E. populnea*	Equal variances assumed	.000	.990	2.579	8	.033	.27883	.10813	.02948	.52818

Appendix J: Raw data from germination trial

Plot	Week	Plant	Number
BC1	**1+2**	Stumpy Bright Green Weed	1
		Lippia	1
		Segmented Leaf	1
		Red/Green Tree	1
		Carrot Top	75
		Milk Weed	5
		Small Clover	6
		Lime Green Weed	1
	3	Dew Rooty Tall	1
		Milk Weed	1
		Carrot Top	5
		Lippia	1
	4	Small Clover	1
		Carrot Top	15
		Unidentified Dicot	1
	5	Carrot Top	3
	6	Carrot Top	4
		Segmented Leaf	1
	7	Carrot Top	1
	8	Milk Weed	1
	10	Carrot Top	4
	11	Lippia	1
	13	Unidentified Dicot	1
		Total Germinants	**132**
		Total Species Count	**9**
BC2	**1+2**	Purple Backed Thistle	1
		Wahlenbergia	1
		Rose Weed	1
		Unidentified Dicot	2
		Carrot Top	99
		Upright Woolley with Flower	5
		Grasses	1
		Thick Woolley	3
		Thin Woolley with Flowers	3
		Smells Like Rosemary	6
		Heavily Segmented Fern	5
		Small Clover	2
		Maple Carrot	2
		Milk Weed	1
	3	Smells Like Rosemary	3
		Carrot Top	20
		Red/Green Tree	2
		Unidentified Dicot	1
		Upright Woolley	1
		Dew Rooty Tall	1
		Grasses	1
	4	Carrot Top	9
		Red/Green Tree	2
		Sage	1
		Unidentified Dicot	3
	5	Carrot Top	1
		Sage	1
	7	Carrot Top	1
	8	Carrot Top	1
	11	Carrot Top	1

		Unidentified Dicot	1
		Total Germinants	**182**
		Total Species Count	**18**
BC3	**1+2**	Segmented Leaf	1
		Lippia	1
		Chive Thing	1
		Wahlenbergia	7
		Unidentified Dicot	1
	4	Cabbage Weed	1
		Wahlenbergia	1
		Red/Green Tree	2
		Carrot Top	4
	5	Carrot Top	5
	6	Chive Thing	1
		Carrot Top	2
		Unidentified Dicot	3
		Cabbage Weed	1
	7	Unidentified Dicot	1
	12	Unidentified Dicot	1
	14	Unidentified Dicot	1
		Total Germinants	**34**
		Total Species Count	**9**
BC4	**1+2**	Stumpy Bright Green Weed	1
		Segmented Leaf	1
		Thick Woolley	1
		Carrot Top	52
		Small Clover	2
		Milk Weed	2
	3	Sage	1
		Red/Green Tree	1
		Carrot Top	2
		Grasses	2
		Small Clover	2
		Carrot Top	15
	4	Carrot Top	6
	5	Carrot Top	3
		Milk Weed	1
	6	Carrot Top	3
	7	Carrot Top	1
	8	Carrot Top	1
	9	Carrot Top	2
	11	Carrot Top	2
	13	Unidentified Dicot	1
	14	Unidentified Dicot	7
		Total Germinants	**109**
		Total Species Count	**10**
BC5	**1+2**	Eucalypt	1
		Grasses	2
		Carrot Top	11
		Small Clover	1
	4	Lippia	2
	5	Small Clover	1
		Carrot Top	3
Plot	**Week**	**Plant**	**Number**
		Lippia	2
	7	Carrot Top	1
	8	Lippia	1

		Total Germinants	**25**
		Total Species Count	**5**
BP1	**1+2**	Upright Woolley	5
		Upright Woolley with Flower	2
		Wahlenbergia	11
		Milk Weed	1
		Small Clover	1
		Carrot Top	44
	3	Thick Woolley	7
		Wahlenbergia	1
		Fuzzy Thin Leaf	1
		Grasses	1
		Carrot Top	1
	4	Unidentified Dicot	1
	5	Carrot Top	1
	6	Weird Succulent	1
		Unidentified Dicot	1
	7	Unidentified Dicot	1
	13	Lippia	1
		Total Germinants	**81**
		Total Species Count	**13**
BP2	**1+2**	Wahlenbergia	1
		Thin Woolley with Flowers	2
		Smells Like Rosemary	3
		Sage	1
		Upright Woolley with Flower	1
		Thick Woolley	2
		Upright Woolley	1
		Cabbage Weed	2
		Thick Stem Dark Green Weed	1
	4	Carrot Top	4
		Upright Woolley	2
		Unidentified Dicot	1
		Grasses	1
	5	Upright Woolley	3
	7	Carrot Top	1
	11	Carrot Top	1
		Unidentified Dicot	1
	12	Unidentified Dicot	2
		Total Germinants	**30**
		Total Species Count	**12**
BP3	**1+2**	Stumpy Bright Green Weed	3
		Red/Green Tree	4
		Weird Succulent	6
		Daisy	3
		Lippia	5
		Smells Like Rosemary	5
		Wahlenbergia	5
		Segmented Leaf	3
		Grasses	3
		Singled Leaf	2
		Sword Plant	1
		Dark Green Fuzzy (with Flowers)	1
Plot	**Week**	**Plant**	**Number**
		4 o'clock	1
		Small Clover	1
		Carrot Top	18

		Thick Carrot Top	1
		Milk Weed	1
	3	Grasses	3
		Fluffy Carrot Top with Weird Flower	1
		Stumpy Bright Green Weed	1
		Carrot Top	3
		Small Clover	1
		Red/Green Tree	3
		Lippia	1
	4	Carrot Top	4
		Smells Like Rosemary	1
		Red/Green Tree	2
		Tall Skinny Leaf	1
	5	Chive Thing	1
		Grasses	2
		Red/Green Tree	4
		Carrot Top	1
		Unidentified Dicot	1
	7	Lippia	1
		Grasses	1
		Segmented Leaf	1
		Unidentified Dicot	1
	8	Unidentified Dicot	1
	11	Unidentified Dicot	1
	14	Carrot Top	1
		Total Germinants	**100**
		Total Species Count	**22**
BP4	**1+2**	Lippia	2
		Thick Woolley	2
		Smells Like Rosemary	3
		Wahlenbergia	3
		Tall Skinny Leaf	2
		Unidentified Dicot	1
		Grasses	1
	3	Weird Succulent	1
		Grasses	4
		Carrot Top	1
		Lippia	1
	4	Weird Succulent	2
		Small Clover	1
		Stumpy Bright Green Weed	1
		Thick Dark Green Carrot Top	1
		Unidentified Dicot	2
		Red/Green Tree	1
		Wahlenbergia	2
		Grasses	5
		Carrot Top	15
	5	Grasses	4
		Dark Green Carrot Top	1
		Wahlenbergia	1
	6	Grasses	4
	7	Grasses	5
	8	Grasses	1
Plot	**Week**	**Plant**	**Number**
		Unidentified Dicot	1
	9	Grasses	2
		Dark Green Carrot Top	1

		Carrot Top	1
	10	Unidentified Dicot	1
		Total Germinants	**73**
		Total Species Count	**13**
BP5	**1+2**	Lippia	1
		Daisy	2
		Purple Backed Thistle	1
		Furry Small Tree	1
		Stumpy Bright Green Weed	1
		Glossy Green Weed	1
		Smells Like Rosemary	15
		Thin Woolley with Flowers	3
		Thick Woolley	13
		Unidentified Dicot	4
		Grasses	1
	3	Thick Woolley	1
		Unidentified Dicot	1
	4	Carrot Top	1
		Small Clover	1
		Grasses	1
		Unidentified Dicot	2
		Red/Green Tree	1
	6	Red/Green Tree	1
		Milk Weed	1
	8	Carrot Top	1
	14	Grasses	1
		Total Germinants	**55**
		Total Species Count	**17**
UC1	**1+2**	Smells Like Rosemary	1
		Small Furry Tree	1
		Red/Gree Tree	2
		Thin Woolley	3
		Segmented Leaf	2
		Unidentified Dicot	1
		Thick Dark Green Carrot Top	1
		Small Clover	3
		Carrot Top	8
	3	Lippia	1
		Grasses	1
		Red/Green Tree	2
		Unidentified Dicot	1
		Small Clover	1
	4	Lippia	3
		Red/Green Tree	1
		Grasses	1
	5	Grasses	1
	6	Lippia	2
		Grasses	1
		Thick Woolley	1
		Total Germinants	**38**
		Total Species Count	**12**
UC2	**1+2**	Stumpy Bright Green Weed	1
		Carrot Top	31
Plot	**Week**	**Plant**	**Number**
		Dark Green Carrot Top	2
		Heavily Segmented Fern	1
		Purple Backed Thistle	2

		Thick Carrot Top	6
		Milk Weed	3
		Small Clover	1
	3	Lippia	1
		Weird Succulent	1
		Carrot Top	9
		Segmented Leaf	1
		Dark Green Carrot Top	1
		Grasses	1
		Milk Weed	1
		Thick Carrot Top	1
	4	Lippia	1
		Stumpy Bright Green Weed	1
		Carrot Top	21
		Dark Green Carrot Top	1
	5	Carrot Top	7
		Stumpy Bright Green Weed	1
	6	Carrot Top	4
		Unidentified Dicot	1
	8	Carrot Top	2
	13	Lippia	1
	14	Unidentified Dicot	1
		Total Germinants	**104**
		Total Species Count	**13**
UC3	**1+2**	Thin Woolley with Flowers	2
		Carrot Top	64
		Stumpy Bright Green Weed	1
		Segmented Leaf	1
		Red/Green Tree	2
		Rambling Oak	1
		Upright Woolley	1
		Sage	3
		Smells Like Rosemary	7
		Unidentified Dicot	1
		Small Clover	2
		Milk Weed	1
		Large Clover	3
		Grasses	1
		Thick Dark Green Carrot Top	13
	3	Carrot Top	6
		Sage	1
		Thick Woolley	1
		Red/Green Tree	1
	4	Smells Like Rosemary	2
		Red/Green Tree	2
		Unidentified Dicot	4
		Carrot Top	19
	5	Carrot Top	9
		Thick Dark Green Carrot Top	1
	6	Carrot Top	6
		Red/Green Tree	1
		Grasses	1
	7	Carrot Top	2
Plot	**Week**	**Plant**	**Number**
	8	Carrot Top	2
	9	Carrot Top	2
		Thick Dark Green Carrot Top	1

	11	Unidentified Dicot	1
	14	Unidentified Dicot	2
		Total Germinants	**167**
		Total Species Count	**16**
UC4	**1+2**	Rambling Oak	3
		Rose Weed	4
		Purple Carrot Top	5
		Little Bright Green	1
		Carrot Top	27
		Segmented Leaf	3
		Red/Green Tree	2
		Stumpy Bright Green Weed	1
		Lippia	2
		Heavily Segmented Fern	2
		Thorned Strawberry	1
		Big Segmented Fern	1
		Dew Rooty Tall	1
		Thick Carrot Top	26
		Frilly Weed	4
		Small Clover	2
		Grasses	2
		Thick Dark Green Carrot Top	1
		Milk Weed	1
		Milk Weed Look-a-Like	1
	3	Little Bright Green	1
		Rambling Oak	1
		Lippia	1
	4	Carrot Top	2
		Purple Carrot Top	1
		Rambling Oak	2
		Grasses	4
		Milk Weed	1
		Woolley Tree	1
		Small Clover	1
		Little Bright Green	1
	5	Rambling Oak	1
	6	Small Clover	1
		Carrot Top	2
		Rambling Oak	1
		Grasses	2
		Purple Carrot Top	1
		Milk Weed	1
		Unidentified Dicot	1
	7	Little Bright Green	2
		Rambling Oak	1
		Grasses	1
	8	Red/Green Tree	1
		Carrot Top	1
		Little Bright Green	1
		Purple Carrot Top	3
	9	Thick Dark Green Carrot Top	1
	11	Grasses	1
		Carrot Top	1
Plot	**Week**	**Plant**	**Number**
		Unidentified Dicot	1
		Total Germinants	**130**
		Total Species Count	**23**

UC5	**1+2**	4 o'clock	2
		Thin Woolley	1
		Segmented Leaf	2
		Maple Carrot	14
		Carrot Top	28
		Large Clover	3
		Lippia	3
		Small Clover	51
		Red/Green Tree	3
		Smells Like Rosemary	7
		Rambling Oak	4
		Rose Weed	19
		Grasses	3
		Heavily Segmented Fern	39
		Purple Carrot Top	6
		Thorny Strawberry	1
		Thick Dark Green Carrot Top	40
		Milk Weed	4
		Thorny Weed	3
		Thick Carrot Top	28
	3	Carrot Top	6
		Rose Weed	3
		Heavily Segmented Fern	3
		Lippia	1
		Grasses	1
		Sage	1
		Milk Weed	1
		Maple Carrot	2
		Small Clover	1
		Rambling Oak	2
		Thorny Milk Weed	1
	4	Thick Dark Green Carrot Top	1
	5	Unidentified Dicot	1
		Grasses	2
		Carrot Top	3
		Large Clover	2
		Heavily Segmented Fern	1
	6	Unidentified Dicot	2
		Carrot Top	1
		Grasses	2
	7	Heavily Segmented Fern	1
		Milk Weed	2
		Thorny Milk Weed	1
		Segmented Leaf	1
		Unidentified Dicot	1
	8	Unidentified Dicot	1
		Total Germinants	**305**
		Total Species Count	**25**
UP1	**1+2**	Lippia	3
		Segmented Leaf	3
		Thin Woolley with Flowers	1
		Thick Woolley	4
		Grasses	2
Plot	**Week**	**Plant**	**Number**
		Milk Weed	3
		Woolley Weed	27
		Carrot Top	22

		Small Clover	2
		Frilly Weed	1
	3	Smells Like Rosemary	2
		Upright Woolley	1
		Carrot Top	2
		Woolley Weed	3
	4	Grasses	1
		Lippia	1
		Carrot Top	1
	5	Grasses	1
		Thick Woolley	1
	6	Grasses	3
		Thick Woolley	1
		Total Germinants	**85**
		Total Species Count	**12**
UP2	**1+2**	Segmented Leaf	3
		Stumpy Bright Green Weed	1
		Smells Like Rosemary	3
		Small Clover	1
		Lippia	7
		Thin Woolley with Flowers	2
		Thick Woolley	2
		Sage	11
		Weird Succulent	8
		Unidentified Dicot	1
		Carrot Top	9
		Grasses	4
		Thick Dark Green Carrot Top	5
	3	Unidentified Dicot	1
		Sage	1
		Weird Succulent	2
		Carrot Top	1
		Small Clover	1
		Upright Woolley with Flower	1
		Red/Green Tree	1
		Dark Green Carrot Top	1
	5	Thin Woolley	1
		Carrot Top	1
		Small Clover	1
		Upright Woolley with Flower	1
		Red/Green Tree	1
	6	Lippia	1
		Unidentified Dicot	2
	10	Dark Green Carrot Top	1
	11	Unidentified Dicot	1
	13	Grasses	1
	14	Grasses	1
		Total Germinants	**78**
		Total Species Count	**19**
UP3	**1+2**	Grasses	5
		Weird Succulent	166
		Fluffy Carrot Top with Weird Flower	1
		Smells Like Rosemary	5
Plot	**Week**	**Plant**	**Number**
		Thin Woolley	1
		Red/Green Succulent	1
		Thick Dark Green Carrot Top	14

		Unidentified Dicot	3
	3	Weird Succulent	17
		Smells Like Rosemary	1
		Unidentified Dicot	2
	4	Grasses	1
		Weird Succulent	3
		Thick Dark Green Carrot Top	1
		Unidentified Dicot	2
	5	Grasses	4
		Unidentified Dicot	1
	7	Weird Succulent	1
		Unidentified Dicot	1
Plot	**Week**	**Plant**	**Number**
	12	Grasses	4
		Unidentified Dicot	2
		Total Germinants	**236**
		Total Species Count	**8**
UP4	**1+2**	Lippia	2
		Segmented Leaf	1
		Smells Like Rosemary	3
		Upright Woolley with Flower	10
	3	Unidentified Dicot	1
		Upright Woolley with Flower	2
	4	Unidentified Dicot	2
		Small Clover	1
		Carrot Top	1
	5	Carrot Top	2
		Milk Weed	1
		Total Germinants	**26**
		Total Species Count	**8**
UP5	**1+2**	Unidentified Dicot	2
		Sage	2
		Upright Woolley with Flower	1
		Smells Like Rosemary	6
		Segmented Leaf	2
		Thin Woolley with Flower	2
		Lippia	5
		Weird Succulent	4
		Purple Single Leaf	3
		Little Bright Green	2
	3	Little Bright Green	9
		Segmented Leaf	1
		Lippia	2
		Woolley (dead)	1
		Unidentified Dicot	1
		Carrot Top	1
	4	Thick Dark Green Carrot Top	1
		Purple Single Leaf	1
		Little Bright Green	1
		Sage	1
		Carrot Top	1
		Lippia	2
		Woolley Tree	1
Plot	**Week**	**Plant**	**Number**
	5	Unidentified Dicot	2
		Lippia	1
		Grasses	2

6	Carrot Top	1
7	Grasses	1
	Little Bright Green	1
8	Unidentified Dicot	1
	Lippia	1
	Total Germinants	**62**
	Total Species Count	**16**

Appendix K: Statistics for germination data

		Levene's Test for Equality of Variances		**T-test for Equality of Means**						
		F	**Sig.**	**t**	**df**	**Sig. (2-tailed)**	**Mean Difference**	**Std. Error Difference**	**95% Confidence Interval of the Difference**	
									Lower	**Upper**
T=3										
Lippia – *E. camaldulensis*	Equal variances assumed	.174	.687	-1.416	8	.195	-.22499	.15892	-.59145	.14147
Lippia – *E. populnea*	Equal variances assumed	.054	.822	-.964	8	.363	-.22743	.23582	-.77124	.31638
Total minus lippia – *E. camaldulensis*	Equal variances assumed	.013	.912	-.521	8	.617	-.12565	.24130	-.68209	.43079
Total minus lippia – *E. populnea*	Equal variances assumed	.044	.840	-1.243	8	.249	-.20678	.16633	-.59033	.17678
Total – *E. camaldulensis*	Equal variances assumed	.016	.903	-.544	8	.601	-.13081	.24029	-.68491	.42330
Total – *E. populnea*	Equal variances assumed	.013	.913	-1.617	8	.145	-.25015	.15474	-.60698	.10667
T=14										
Lippia – *E. camaldulensis*	Equal variances assumed	.298	.600	-1.000	8	.347	-.21339	.21337	-.70541	.27863
Lippia – *E. populnea*	Equal variances assumed	.202	.665	-.899	8	.395	-.22046	.24520	-.78589	.34497

Total minus lippia – *E. camaldulensis*	Equal variances assumed	.008	.930	-.619	8	.553	-.14158	.22867	-.66890	.38573
Total minus lippia – *E. populnea*	Equal variances assumed	.318	.588	-.729	8	.487	-.12547	.17217	-.52250	.27156
Total – *E. camaldulensis*	Equal variances assumed	.007	.935	-.636	8	.542	-.13634	.21428	-.63046	.35778
Total – *E. populnea*	Equal variances assumed	.040	.847	-1.086	8	.309	-.16954	.15607	-.52945	.19036

Appendix L: Statistics for pseudo-species data

		Levene's Test for Equality of Variances		**T-test for Equality of Means**						
									95% Confidence Interval of the Difference	
		F	**Sig.**	**t**	**df**	**Sig. (2-tailed)**	**Mean Difference**	**Std. Error Difference**	**Lower**	**Upper**
Species Richness – *E. camaldulensis*	Equal variances assumed	.106	.754	-1.957	7	.091	-.17710	.09050	-.39109	.03688
Species Richness – *E. populnea*	Equal variances assumed	1.228	.300	1.114	8	.297	.09386	.08422	-.10034	.28806

Appendix M: Average lippia cover

	Lippia (subjective)
BC1	79.625

BC2	28.7625
BC3	43.75
BC4	34.25
BC5	95.625
UC1	81
UC2	31.8875
UC3	22
UC4	72.875
UC5	51.4375
BP1	32.325
BP2	22.9375
BP3	45.2625
BP4	44.625
BP5	54.25
UP1	22.7625
UP2	51.25
UP3	21.375
UP4	7.5625
UP5	39