South African Grassland Ecology and its Restoration

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Declaration

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Nicholas Paul Zaloumis

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Introduction:

Grasslands are characterised by open vegetation cover, made up of predominantly a continuous grassy layer. Well-known examples of grassland include the Prairies of North America (Knapp *et al.* 2004), the Campos of South America (Overbeck *et al.* 2007) and the temperate grasslands of Australia (Morgan 1999). Grasslands also occur throughout Africa and Madagascar (White 1983, Bond *et al.* 2008). Despite covering around 11% of the earth's vegetated land surface (Ramankutty & Foley 1999) grassland systems are still largely under-appreciated and under-conserved even with the increasing realisation that these systems are some of the most threatened systems in the world (Lubke *et al.* 1996, Overbeck *et al.* 2007, Bond & Parr 2010). Grassy biomes are heavily utilised by human activities and face increasing anthropogenic pressure as human populations increase and with it the need for the resources that grasslands provide (Myers *et al.* 2000, Reyers *et al.* 2001, Hoekstra *et al.* 2005). Agriculture, afforestation, urban expansion and mining are the main drivers of grassland loss.

Grasslands misunderstood

There is the unfortunate public misconception that, while forests are 'pristine' and 'ancient' landscapes, grasslands are derived from anthropogenic clearing and burning of forest and thus are secondary in nature (Bond & Parr 2010). One of the reasons why natural grasslands were thought to be secondary systems is because mesic grasslands do not exist in equilibrium with their climate and they could return to a forest state if fire was excluded from the system (Acocks 1953). As evidence arose, that South African grasslands could actually be natural systems, Ellery & Mentis (1992) and Meadows & Linder (1993) challenged the idea that South African grasslands were anthropogenic systems. If South African grassland were truly secondary in nature we would expect to find these 'novel' open environments with traits ill adapted to frequent fire disturbances and comprised of little biodiversity and few endemics. On the contrary, much evidence supports the idea that tropical and sub-tropical grasslands may be ancient, derived before humans even played a major role in influencing the world's landscape (Overbeck *et al.* 2007, Bond & Parr 2010).

Fire is a natural disturbance that plays an influential role in C4 grasslands where the majority of grass species have the C4 photosynthetic pathway. C4 dominated biomes support the highest fire frequencies in the world (Mouillet & Field 2005, Chuvieco *et al.* 2008, Archibald *et al.* 2013). Grasslands in South Africa and South America support diverse and distinct faunal and floral communities. Upland grasslands can contain 82 plant species within 1000m^2 making it the second most diverse vegetation in South Africa (van Wyk 1998 ,Oudtshoorn *et al.* 2011). Forbs, herbaceous plant species, are one notable grassland plant growth form that makes up a considerable proportion of South African grassland biodiversity (Figure 1.1). They are able to resprout after fire from a multitude of woody or tuberous root systems (Bews 1925, Hilliard & Burtt 1987,Williams *et al.* 2000) known as underground storage organs (USOs) which are protected from the fire itself (Uys 2006, Overbeck & Pfadenhauer 2007). Grassland forbs are thought to be able to persist in the landscape for very long periods without the need to successfully reproduce (Zaloumis & Bond 2011). However without fire they disappear in as little as a decade (Fynn *et al.* 2004, Uys *et al.* 2004, Uys 2006). Another little known but remarkable grassland growth form is the geoxylic suffrutex, otherwise known as an 'underground tree', which also resprouts directly after fire (White 1976). These are woody plants that support extensive underground stems and a number of reduced vegetative and fertile aerial parts. The forb USO and geoxylic suffrutex habits are two specific fire-adapted traits that could not have been derived in the last 2000 years or since the last glacial maximum and distinguish our grasslands from northern hemisphere grassland equivalents (Bond & Parr 2010).

The importance of grasslands

Globally grasslands contribute significantly to environmental, economic and cultural values in the provision of several important ecosystem services (Reyers & Tosh 2003, Overbeck *et al.* 2007, Blignaut *et al.* 2008, Hardy 2008, O'Connor & Kuyler 2009). Globally the importance of their role in climate regulation and carbon sequestration still needs to be investigated; such as their role in the global carbon and energy cycles. Some ecosystem services are well documented and can have significant consequences in a local context. These include the production of high quality grazing, water moderation and erosion control (Reyers & Tosh 2003, Overbeck *et al.* 2007, Blignaut *et al.* 2008, Hardy 2008). Within South

Africa intact natural grasslands play an important role in the major summer-rainfall catchment areas where they stabilise soils, promote rainfall infiltration, and support more streamflow than pine plantations grown under the same conditions (Bosch & Hewlett 1982, Le Maitre *et al.* 2002, Driver *et al.* 2004, Hardy 2008, Blignaut *et al.* 2010). Catchments with high levels of erosion and therefore high siltation rates have negative economic impacts for the management of our major water reservoirs by increasing the need for further civil engineering construction works (Driver *et al.* 2004, Blignaut *et al.* 2010). Grasslands are solely responsible for providing water to the entire Gauteng province (Reyers & Tosh 2003). Natural grasslands also provide high quality grazing for our beef industry (Hardy 2008).

Grasslands in South Africa

In this study I focus on the grasslands of South Africa (Figure 1.2 and 1.2). The grassland biome is the second largest biome in South Africa mostly situated on the high central plateau commonly known as the Highveld (Figure 1.3a, Mucina & Rutherford 2006). The coastal grasslands make up one component of the grassland-forest mosaic that is the Indian Ocean Coastal Belt biome (Figure 1.3b, Mucina & Rutherford 2006). The characteristics of the grassland vegetation types sampled in the study are elaborated upon further within Chapter 2. South Africa's mesic grasslands are seasonal with an annual rainfall above 500mm to 700mm that predominantly falls in the summer months (Ellery *et al.* 1995). My study focuses on grasslands below about 2000m which are dominated by C4 grass species (Mucina & Rutherford 2006).

South Africa boasts a long history of ecological work focused in the grassland biome. Much of this work has explored the impacts of land-use practices such as grazing and fire management driving changes in the environment and the influence of climatic and soil variables that play an important role in grassland ecology (Tainton 1981, O' Connor & Bredenkamp 1997, Mucina & Rutherford 2006). The investigation of plant diversity patterns within South African grasslands has also featured prominently in the grassland literature (Acocks 1953, Cowling *et al.* 1989, van Wyk 1998).

As conservation biologists and grassland scientists began to realise the importance of our grassland systems we saw the establishment of several protected areas and implementation of several conservation related programs to try and encourage their protection. 1999 saw the establishment of South Africa's first world heritage site, iSimangaliso Wetland Park, which encompasses one of the largest remaining areas of coastal grassland (Taylor 2004). Soon afterward the uKhahlamba Drakensberg Park, situated in the higher grassland regions of KwaZulu-Natal, followed suit also being declared a world heritage site in 2000. There are several well-known parks within most of the South African provinces that are included in the biome boundary. The only national park is the Golden Gate National Park in the Free State. Despite this, however, only just 2.2% of the biome is under any formal conservation protection (Hardy 2008). In an effort to advance grassland conservation several conservations programs have been developed to promote conservation ideals with landowners. SANBI's Grassland Program, Bird Life South Africa and WWF have been actively involved in supporting projects like the Enkangala Grasslands Project which encourages landowners to engage in conservation.

Grassland conservation and restoration

South African grasslands are facing increased habitat loss and fragmentation and have become one of the most threatened vegetation types in South Africa making them a priority for conservation efforts (Matsika 2007). Restoration can be used both as a tool in the conservation of grasslands and to improve our understanding of this system (Young 2000). There has been considerable active restoration work done on our mined coastal dune forest systems in KwaZulu-Natal (Mentis & Ellery 1994, Avis & Lubke 1996, Lubke & Avis 1999, Wassenaar *et al.* 2005). However, South African grassland restoration literature is sparse (Lubke *et al.* 1996) and often only compares post-disturbance grasslands to natural grassland. There are however some studies that have focused on the re-seeding of common grass species and these have reported some success in re-establishing vegetative cover and some grazing value (Mentis 1999, Oudtshoorn *et al.* 2011). Otherwise there is a lot of active unpublished restoration work, which usually focuses on establishment of fast growing grasses. These efforts are generally not monitored or are only noted in the grey literature. The lack of monitoring of a restoration project after the active restoration has finished has most likely slowed our understanding of grassland restoration problems. Monitoring of restoration should be encouraged to develop the feedback and learning process (Prober &

Thiele 2005). Secondary South African grasslands have been shown not to follow natural succession when left to recover. They may establish a grassy layer, but the plant community does not represent that of the natural vegetation (Oudtshoorn *et al.* 2011, Zaloumis & Bond 2011). In these circumstances the key grassland elements have failed to re-colonise the degraded environment and we need to determine if this is because of physical limitations, due to changes in soil attributes, or biological limitations such as plant competition or plant reproductive constraints (Kardol *et al.* 2008). In the case of our mesic grasslands it could be the latter that is the more limiting factor (Zaloumis & Bond 2011).

Structure of this Thesis

The aim of this study is to investigate how human related disturbances affect mesic grasslands. I identified what was lost from the system after a disturbance and what biological constraints ecologists and managers will face when approaching their restoration. I then investigated biological limitations to grassland restoration by attempting species re-introduction into secondary grasslands and exploring the interaction between grasses and forbs. The chapters of the thesis represent different parts of the study and have been written independently to facilitate publication. This inevitably means there is some replication of text and references in the three data chapters.

Chapter 2: From a distance secondary grasslands can look very similar to natural grasslands. However, there is some evidence that secondary coastal grasslands do not recover towards their pristine state through natural successional processes and diverge to very different community composition (Zaloumis & Bond 2011). To explore this further, I tested the hypothesis that mesic grasslands throughout South Africa were negatively altered by human related disturbances. Coastal and upland grasslands that had been subjected to afforestation or ploughing and then allowed to recover were sampled during the growing season. These secondary grasslands included different recovery ages that were then compared to natural grassland reference states to identify how these grassland states differ and how the differences may influence grassland restoration efforts.

Chapter 3: Biological barriers may prevent grassland forb species from re-colonising secondary grasslands (Bond & Parr 2010, Zaloumis & Bond 2011). I attempted to transplant natural grassland mixed species sods into two montane and one coastal grassland to re-introduce natural grassland species into secondary grasslands. I investigated sod species establishment and survival over a period of 12 months to test the hypothesis that natural grassland sods could be successfully transplanted into secondary grasslands. The transplant experiment included grass removal and control treatments with observations on the effects of pre- and post-transplant burning on transplant success. The project was set up to allow repeated monitoring in the future with the intention of monitoring sod species survival and dispersal success.

Chapter 4: Secondary grasslands were dominated by competitive mono-specific grass swards (Chapter 2). Little is known about how South African grass and forb species interact in a natural grassland context and how this changes with secondary grassland grass species. I tested the hypothesis that grassland forbs were limited by grass competition by investigating the effect of several common natural and secondary grass species on grassland forb establishment. The experiment was conducted in pots in a glasshouse on the Pietermaritzburg campus of UKZN.

The thesis concludes with a chapter summarising the results with a discussion of the implications of this study for future work on restoration and on South African grasslands more generally.

Figure 1.1: Examples of the diverse range of flowering forbs found in South African grassland.

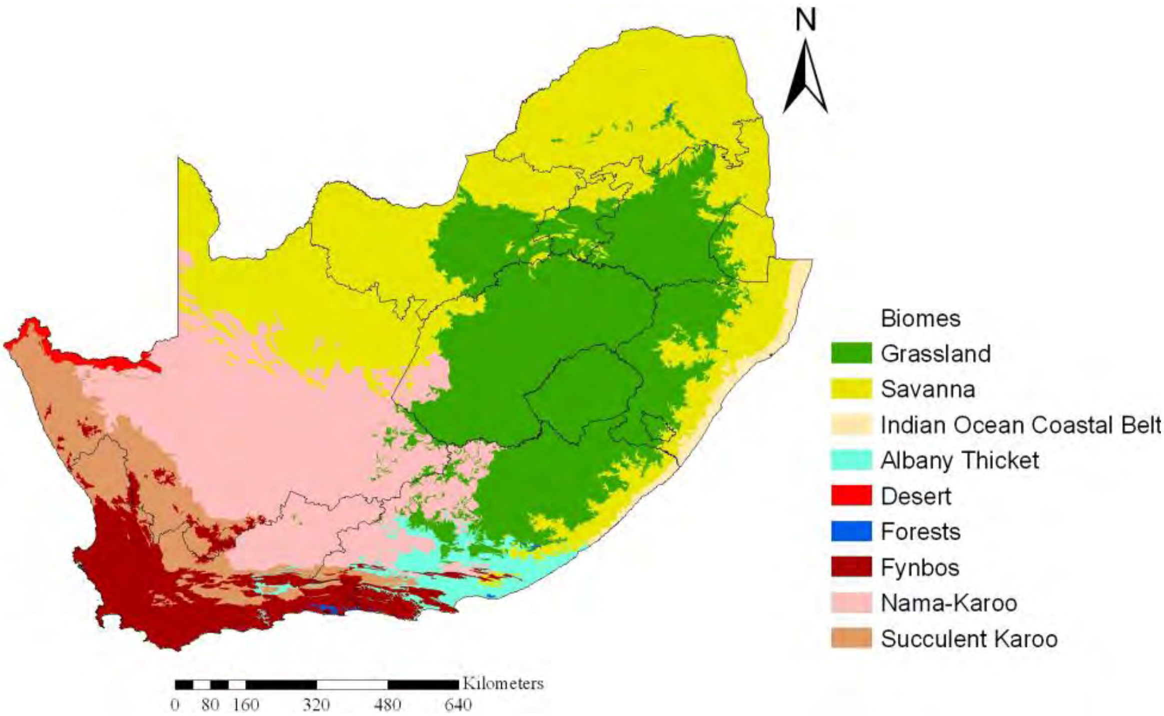

Figure 1.2: Map of South Africa with the Grassland biome in Green and the Indian Coastal Belt Biome in beige (taken from Mucina & Rutherford 2006).

a. Upland grasslands of Buffelskloof Nature Reserve

b. Coastal dune grassland-forest mosaic of the Eastern Shore, iSimangaliso Wetland Park

Figure 1.3: Examples of the Upland grassland biome (a), showing pine plantations in the background, and the Coastal Indian Ocean coastal belt biome (b), showing a natural coastal grassland-forest mosaic.

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Chapter 2: Mesic C4 grasslands of South Africa and their fragility

Abstract:

Grasslands are considered to be the most threatened biome in South Africa and yet they are still generally misunderstood. Often ignored as part of the conservation estate, grasslands are viewed as secondary vegetation derived from the felling of forests and therefore suitable targets for afforestation. However, the mesic grasslands of southern Africa are now thought to be ancient and much more extensive in the last glacial. They are very rich in perennial forb species, many of which have large underground storage organs and seldom recruit from seed. We studied the diversity of natural grasslands within Mpumalanga and KwaZulu-Natal and compared them with secondary grasslands after different forms of disturbance, including afforestation and ploughing. We found that natural succession in secondary grasslands failed to restore the diversity of forbs, with very poor recovery, even decades after removal of plantation forests. Secondary grasslands were completely missing the resprouting forb component typical of natural grasslands and they responded differently to natural disturbances such as fire. Thus primary grasslands appear to be at least as fragile as primary forests and may take decades to centuries to recover from major disturbance. We discuss the biological and physical constraints that our grasslands plant species face in a restoration context. It is important to promote the value of our grasslands and, until we can develop practical methods for successful active grassland restoration. We need to strive to conserve what we have left.

Introduction:

As the pressure of human activities increases on the natural environment, further conservation efforts are needed to sustain the increasingly fragmented natural landscape. Included in these efforts are; a. the development of better land use management practices that mitigate degradation and promote diversity, b. the placing of more land under formal protection and c. the restoring of land that has been already degraded. Ideally the conservation of pristine environments is the most important goal to achieve. In contrast, restoration tends to be viewed as a last, but increasingly necessary resort (Young 2000, Prober & Thiele 2005, Hobbs 2007). With growing demands on natural resources and land itself there will be a greater reliance on restoration to maintain biodiversity and return some level of functionality to already degraded land. For restoration to become an effective conservation tool there is a need to develop a better understanding of the original plant communities and functions in the target system that is due to undergo restoration (Soule & Kohm 1989, Soc. Eco. Rest. 2004, Prober & Thiele 2005, Le Stradic 2012). To do this, one needs to determine how human disturbances have had an effect on the original system by comparing a suitable reference of the natural system with a degraded system. The reference system then helps us identify targets and set goals when undergoing restoration.

To identify the reference system, one needs to know the makeup of the natural landscape that came before the disturbance occurred. Unfortunately this is not always possible. Often one must settle with the next best thing; a pristine system within close proximity to the degraded area (Bakker & Berendse 1999, Soc. Eco. Rest. 2004, Zaloumis & Bond 2011, Le Stradic 2012). Sometimes finding these reference landscapes can be difficult as surrounding land uses may not be conducive to preserving the pristine state and natural land may have been subjected to some form of historical land use degradation (Pers. observations, Harris *et al.* 2006).

One natural landscape that poses a challenge to restoration ecology is the grasslands of Southern Africa (Lubke *et al*. 1996, Mentis 2006, Zaloumis & Bond 2011). These grassy systems are extensive, with the grassland biome being the second largest biome in South

Africa after the Savanna. The grassland biome supports a rich and distinct diversity of plant and animal life (Cowling *et al.* 1989, Rebelo *et al.* 1997). In the upland grasslands from the Eastern Cape to Mpumalanga, studies have shown that plots of $100m^2$ can contain up to 40 species. Within a $1000m^2$, a natural grassland state can support an average of 82 different species making grassland the second most diverse vegetation in South Africa after renosterveld (van Wyk 1998 , van Oudtshoorn *et al.* 2011). Grassland landscapes support a plant community rich in forb species (herbaceous non-grass plants) often with a woody or tuberous root system (Bews 1925, Hilliard & Burtt 1987, Williams *et al.* 2000, Zaloumis & Bond 2011). Forbs largely account for grassland species richness with grass species accounting for only a small proportion (Uys *et al.* 2004, Zaloumis & Bond 2011). A woody plant element called a geoxylic suffrutex (commonly known as an 'underground tree') also occurs frequently throughout the biome (White 1976). This growth form is particularly rich in species within the coastal grasslands of KwaZulu-Natal.

These forbs, grasses and geoxylic suffrutices are all resprouting species that respond positively to the presence of fire. When fire is removed, grassy systems can be converted to woodlands and forests over time, particularly in mesic systems (>750 mm mean annual precipitation in South Africa) (Bond *et al.* 2003, Uys *et al.* 2004). A grassland system without fire would see a shift from shade intolerant forb and grass species to shade tolerant woody and herbaceous species (Fynn *et al.* 2004) and ultimately a biome shift to closed forest (Parr *et al*. 2012).

Despite grasslands being globally widespread there is a lack of understanding of their ecological functioning (Lubke *et al.* 1996, Bond &Parr 2010). Part of this is because their diversity and general value have largely been neglected by the public and conservation biologists until relatively recently (Lunt 1994, O'Connor 2005). Most South African literature has focused on the use of grasslands for livestock farming and therefore is biased towards a rangeland perspective (Tainton 1981, Post & Kwon 2000, Conant *et al.* 2001, to mention a few). Little has been recorded on alternative grassland functions and diversity (Uys *et al.* 2004, Carbutt & Edwards 2004, 2006, Fynn *et al.* 2004, Zaloumis & Bond 2011). However, South African grasslands have been well served by popular books on species identification (Hilliard & Burtt 1987, Pooley 1998, Pooley 2003).

Like other grassy biomes around the world, South Africa's grasslands have been subject to much human alteration, mostly in the form of conversion to crop land and forestry but also as urban expansion and mining (Rebelo 1997, van Wyk 1998, Mucina & Rutherford 2006). The grassland biome is the most transformed biome in South Africa, with 30% of the transformation being irreversible (van Oudtshoorn *et al.* 2011). Another 30% is only partially degraded by agriculture and bad management practices or is encroached upon by woody species. The remaining 40% remains relatively pristine. However, fragmentation of grassland patches increased drastically in six years between 1994 and 2000 from 4017 patches to 13503 while average patch size decreased substantially from 44.5km^2 to only 13.75km^2 (Matsika 2007). Transformation of this magnitude has made grasslands the most threatened biome in South Africa (van Wyk 1998).

Misunderstanding grasslands has not helped us in identifying how they react to human related disturbance. Two previous studies have shown that South African grassland systems are fragile and difficult to restore after being transformed. The first studied secondary grassland that was recovering after coal mining (Mentis 2006) and the second studied grassland recovery after being excised of pine plantations (Zaloumis & Bond 2011). There have also been several theses on the topic (Olivier 2008, Kruger 2012). To improve our knowledge in this regard we would have to increase the number of post recovery and restoration monitoring projects (Prober and Thiele 2005). Grassland restoration is also complicated by the fact that grassland systems are often not at equilibrium with their local climate and soils (Lubke *et al*. 1996, Bond *et al.* 2003). In short, grassland systems have been difficult to rehabilitate successfully, let alone restore, far more than was previously thought.

Since grasslands are under threat of being degraded it is important to identify what this means in terms of species richness, composition and function. Once we have a better understanding of these characteristics we can identify what is missing from disturbed and degraded landscapes and determine methods to restore them.

The aim of this study is to characterise pristine grassland communities and compare them to secondary grasslands which have been subjected to human related disturbances. We asked the following questions:

1. What is the composition of South African grasslands? What characteristics or elements define the biome and are there any key functional traits that are common?
2. Can we develop an identification toolkit that incorporates more than grass species? Can we use this to help practitioners to easily distinguish natural grassland from secondary grassland?
3. How do different land-uses affect natural grassland characteristics and do secondary grasslands recover once the land use is removed?
4. Which factors, biological and physical, are most limiting in South African grassland restoration efforts?

This chapter aims to explore the community make up of natural grasslands. A better understanding of our grassland biome can help us promote its value in conservation efforts and develop policy guidelines to inform better land-use and management decisions. Grasslands, besides their intrinsic value, have much to offer both scientifically and economically. Studies like this are one way we can put them into the public eye.

Methods:

Study areas

Our study areas occurred within the eastern side of South Africa in the mesic part of the Grassland biome. We were able to find areas suitable for the study in the high elevation parts of the biome in Mpumalanga province and mid elevation and coastal grassy biomes of KwaZulu-Natal province. Study areas had to both have secondary grassland sites as well as comparable reference natural grasslands sites within close proximity to each other. This made finding suitable sites difficult and we were lucky to find several options (Figure 2.1).

Figure2.1: The four main study areas situated within Mpumalanga and KwaZulu-Natal

We managed to find three Highveld grassland areas, classified as thee Mesic Highveld Grassland Bioregion in The Vegetation of South Africa, Lesotho and Swaziland (Mucina &

Rutherford 2006). Two occurred near the town of Lydenburg, on Lydenburg Montane Grassland (Gm18 – Mucina & Rutherford 2006), Makobulaan Nature Reserve (-25° 12' 30.24" South, 30° 33' 40.104" East), a privately run pine plantation and Buffelskloof Nature Reserve (-25° 18' 15.156" South, 30° 31' 11.928" East). Both occur within mountainous terrain which has largely been afforested with pine plantations with fragmented islands of grassland retained. Mean annual precipitation (MAP) for this area is 858mm with a rainfall range between 660mm and 1180 mm. Frequent mists occur during most months of the year and the region experiences frost. The mean annual temperature (MAT) is 14.1˚C. The grasslands occur on high elevation plateaus with undulating plains, peaks, slopes, hills and deep valleys all set within the Northern Escarpment region. The soil of this grassland type is mostly derived from shale and quartzite with occasional dolerite intrusions.

One area close to the town of Wakkerstroom (-27° 14' 25.3674" South, 29° 59' 9.6714" East) occurred within the Wakkerstroom Montane Grasslands (Gm 14 – Mucina & Rutherford 2006). The MAP for the area is 902mm with a yearly range between 800mm and 1250mm. The MAT is 14.1 ˚C with these grasslands experiencing very cold winters and mild summers. The grasslands occur on low mountains and undulating plains. Soils are derived from mudstone, sandstones and shale of the Karoo Supergroup.

One area occurring within a mid-altitudinal grassland area is classified as Sub-escarpment Grassland Bioregion, within the Drakensberg Foothill Moist Grassland type (Gs 10 – Mucina & Rutherford 2006). This was on the Blue Crane Farm near Nottingham Road (-29° 19' 10.8114" South, 30° 4' 13.62" East). Within this grassland type the MAP is 890mm with a MAT of 14.6 ˚C. The region experience frost days. These grasslands occur on a moderately rolling and mountainous landscape which is often carved out by river gorges. Soils are derived from mudstones and sandstones. The soils of the sedimentary parent material are well drained with a depth of 800mm and a clay content between 15- 55%.

The final study area was located in the Indian Ocean Coastal Belt including both Maputaland Coastal Belt(CB 1) and Maputaland Wooded Grassland (CB2) vegetation types on the Eastern Shores of the iSimangaliso Wetland Park (-28° 11' 25.8354" South, 32° 28' 45.084" East). Here the MAP can be up to 1200mm, with about 1000mm on Lake St. Lucia. Most days have high humidity and temperatures and no frost. MAT is 21.1 ˚C. The vegetation is a mosaic of coastal dune forest and grassland occurring on flat coastal plains bordering wetlands and undulating old vegetated dunes. The geology is mostly up to 18000 years old quaternary sediments of marine origin. The sandy soils are highly leached and very nutrient poor.

Plot selection

Plots were spread out throughout the sample areas and included all available secondary and natural areas. Plots were selected to represent natural grasslands or secondary grasslands (recovering after plantation trees have been excised or after having been cultivated) referred to as ‘vegetation treatments’. Plots in the two vegetation treatments were selected to be similar in geologies, fire regimes (recently burnt or not) and were often within close proximity of one another. Where possible, age (time since recovery) treatments were included. Plots were randomly selected within a treatment using a spun stick.

Makobulaan Nature Reserve (MNR) had 32 plots, 16 for each vegetation treatment. All plots were burnt annually. Plot ages ranged from 10 to 40 years since clear-felling. Recovery was from afforestation. Plots ranged between 2000 and 2080m above sea level.

Buffelskloof Nature Reserve (BKNR) had 24 plots, 12 for each vegetation treatment. All plots were burnt every second year. Secondary grassland plots were 20 years old. Recovery was from afforestation. Plots ranged from 1600 and 1750m above sea level.

Wakkerstroom (WS) had 8 plots, 4 for each treatment. All plots were burnt annually. The age of secondary grasslands plots were 18 years since land was withdrawn from cultivation. Recovery was from ploughing. Plots ranged between 1780 and 1810m above sea level.

Blue Crane Farm (BCF) had 16 plots, 8 for each vegetation treatment. Fire treatments differed within the study area. Secondary plots were 20 years old. Recovery was from ploughing. Plots ranged between 1475 and 1520m above sea level.

Eastern Shores, iSimangaliso (ESI) had 64 plots, 32 per vegetation treatment. Fire treatments differed within the study area. The time since deforestation ranged from 4 – 18 years. Recovery was from afforestation. Plots ranged between 30 and 90m above sea level.

Sampling method

Survey data sampling occurred during the summer growing season during the course of the study from late September until late February. Plots were sampled using nested one by one meter, one by two meters and two by two meters quadrats all nested within a circular quadrat of five meters radius (Total plot area = 78.5m: Figure 2.2). Afforested plots and recently clear-felled plots (harvested pine stands between pine rotations, so still being used for a plantation) were sampled with 78 x 1 meter long transects to determine which grassland species survived in plantation understoreys

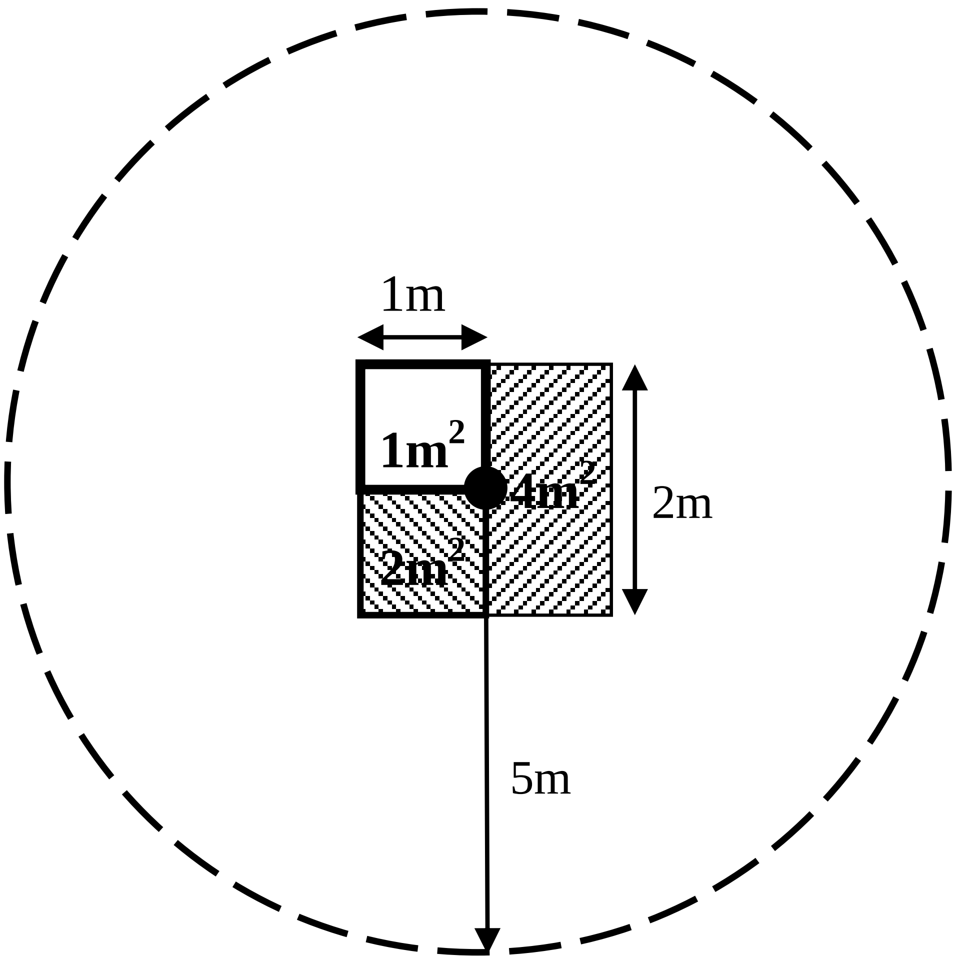


Figure2.2: Nested quadrat method of sampling at each plot

Plant survey

Plant species were recorded in the initial one by one meter plot. Additional species were added with successive quadrat sizes giving a cumulative total. The first individual of each plant species was excavated for further species identification and documentation of plant functional traits, specifically below ground root structure. Within the initial one meter squared plot the total plant ground cover was assessed and the top five most abundant species were identified and their individual percentage cover estimated. The Eastern Shores cover estimates were not completed as it was not part of the initial study plan when it was first surveyed.

Plant species were identified with the use of local field guides (Pooley 1998, van Oudtshoorn 1999 – Refer to Appendix 1 and 2 for examples of grassland forbs and geoxylic suffrutices). Herbarium samples and knowledgeable local experts aided in the identification of hard-to-identify species where possible. The plant characteristics scored for each species included medicinal properties or traditional uses and preference for disturbance when noted in the guide books. Species were noted as resprouters if they had any underground storage organs (USOs – Appendix 3) in the form of thickened root stocks as mentioned in Uys (2006) and Overbeck & Pfadenhauer (2007).

Underground biomass survey

At Makobulaan and Eastern Shores, we estimated underground forb biomass. Within the plots an area of 50x50cm was excavated up to a depth of 25cm. Below ground biomass of non-grassy plants was removed. Root structures were separated from soil and grass roots using a sieve. The wet weight was measured immediately after excavation. The bags were then put in a drying oven at 70^{o}C for 96 hours to ensure that the large USO's were completely dry. Once dry, the dry weight for each sample was measured.

Soil Sampling

Soils were sampled at all the mid and high altitudinal grassland study areas. iSimangaliso's Eastern Shores have had previous soil studies comparing soil characteristics between the secondary and natural treatments. Soils were not sampled here but our plot sampling design was based around their sampling sites (James 1998). Soil samples were taken between 5cm and 15cm below the ground. The soil was dried at 70°C for 48 hours and sieved (2 mm mesh). Dried soil samples were further sieved (0.5 mm mesh) to measure

Total Carbon and Nitrogen. Around 40 mg of soil was weighed into a tin capsule then combusted in a Thermo Flash EA 1112 series elemental analyser and the gasses fed into a Delta Plus XP isotope ratio mass spectrometer (Thermo Electron Corporation, Milan, Italy). Two in-house, and one IAEA, standards were used to calibrate the results. Soil samples were sent to Bemlab (www.bemlab.co.za) and Elsenburg (Institute for Plant Production) for further soil property analysis. Bemlab measured the Total P and Aluminium levels in the soil for the Makobulaan Nature Reserve soil samples. Elsenburg labs were used to measure the Sum of Bases, pH (KCl) and available Phosphorus (P (citric acid)) for Makobulaan and Buffelskloof Nature Reserve and Blue Crane Farm. Zinc and Manganese levels were also measured for Buffelskloof. Soil pH was determined by shaking 2g soil in 20 mL 1 M KCl at 180 r.p.m. for 60 min, centrifuging at $10000g_0$ for 10 min and measuring the supernatant pH. Soil was prepared for Bray II P analysis by extracting 2g soil in Bray II solution (Bray & Kurtz 1945) before filtering it though Machrey-Nagel MN 616 filter paper. The filtrate was then analysed colorimetrically using the malachite green method (Motomizu *et al.* 1983). For total P soil was digested in an Agua regia solution (2 parts HCl to 1 part HNO3) in a sand bath for two hours, filtered through Whatman no.1 filter paper. The filtrate was analysed against ICP-AES standards (Varian Vista MPX, Melbourne, Australia). Soil samples were filtered through Whatman no. 2, made up to 200 mL and the sum of bases, K, Na, Ca and Mg, was measured using ICP-AES analysis.

Data analysis

General Linear Models (GLM) were used to determine differences and interactions within and between treatments. Elevation and study area, when used, were selected as random factors. Vegetation and fire treatments were fixed factors. Species richness was used as one of the main diversity metrics to determine recovery success in this study. However, species richness is not the best measure when used by itself as it does not indicate species composition of a secondary landscape reacting to restorative attempts (Zaloumis and Bond 2011). We included several other measures to test restoration success. Species composition within treatments helps us determine the common plant elements that make up a community while species turnover within plots indicates heterogeneity from plot to plot for a study area within each treatment. Similarity measured between treatments can also be useful and is often used in restoration projects to test how secondary grassland areas compare to a benchmark natural state (van Aarde *et al. 1996a,b,* Wassenaar *et al. 2005*). However, similarity measures typically emphasise common elements and will often fail to reveal rare but important elements or functional traits that are missing from secondary grassland areas (Zaloumis and Bond 2011). In this study we use several methods to explore the difference between secondary and natural grasslands from across the east side of the country.

Species Richness

GLM's were used to investigate the contribution of study area, elevation, vegetation and fire treatment differences and interactions between plots for all plants and for forb species in particular. Elevation was broken into three classes: Study areas above 1700 meters, 1400 - 1700 meters and below 200 meters. Vegetation treatments consisted of three land uses; natural grassland, grasslands recovering after afforestation and then excised, and grasslands recovering from ploughing and then land abandonment. Fire treatments included burnt and unburnt (Refer to Appendix 4 – 6 for examples of burnt and un-burnt treatments).

Species rarefaction curves for each study area were created in R developmental software to compare natural and secondary vegetation. These curves were produced by randomly re-sampling plots in 1000 permutations within the vegetation treatments. The average accumulated number of species was then plotted with increasing number of plots within a vegetation treatment (Gotelli and Colwell 2001).

A rarefaction plot with a high starting point and a steep starting gradient ending in a high number of total species would show high point diversity and a high level of heterogeneity within a study areas treatment compared to the alternative. Curves that start with lower point diversity and have a shallow gradient asymptoting earlier in the species accumulation curve suggest low levels of heterogeneity in the environment.

Species turnover was compared between vegetation treatments using a similar method to *Schwilk et al.* (1997), measuring intra-plot richness and accumulation by regressing the average cumulative plant and forb richness against quadrat size for each nested plot.

Species richness and abundance was investigated for afforested and recently clear-felled pine stands. Species richness was then compared to natural and secondary treatments for the study areas where these pine stands were sampled.

Plant ground and dominant species cover

GLMs were again used in the analysis of plant cover. Here vegetation treatments included natural, afforested and ploughed. Fire treatments were also used in the analysis. Total plant ground cover of plots and the total cover of the five species with highest cover per plot were used to determine any differences between vegetative treatments.

We also then compared the difference between each of the top 5 dominant plants species for each vegetation treatment.

Community composition

The distribution of plant and forb species for natural and secondary treatments for each sample area was broken down into plants and forbs that only occurred in natural plots, secondary plots or that were common to both. Species were separated into plants and forbs that occurred only within one treatment or the other or that were common to both. Plant species were then broken down into four growth forms: forbs (Appendix 1), graminoids, woody and geoxylic suffrutex (Appendix 2) for each study area. Woody elements included trees and shrubs. This was determined for both natural and secondary treatments.

We used a detrended correspondence analysis (Decorana) to ordinate sample plots for each study area and establish patterns of community composition for all species.

Family richness and abundance distribution tables for natural and secondary treatments were also drawn up for each study area. We then identified the families with the most species and families that had highest occurrences in plots.

When investigating the effects of fire treatments (burnt/un-burnt) no differences in pattern were observed so they were excluded for this analysis.

Biomass

Dried below ground non-graminoid root biomass was compared using a T-test between natural and secondary treatments for both MNR and ESI study sites separately. Average plot biomass was calculated as dry mass in kg/m^2.

Succession

Both the Eastern Shores and Makobulaan Nature Reserve gave us the rare opportunity to investigate grassland succession during recovery. The data from Eastern Shores has already been partially reported in Zaloumis and Bond (2011). In this study we added more data points, updated and reanalysed patterns in succession. Eastern Shores had four different ages of time since recovery of up to 17 years and Makobulaan Nature Reserve also had four different ages of time since recovery but starting 40 years before this study. With these data sets, we were able to determine whether or not recovering afforested areas showed a successional trajectory for plant and forb species richness. Mean plant and forb species were regressed against age since clearance.

Soil

Soil properties were compared within study area vegetation treatments with the use of GLMs. As Makobulaan Nature Reserve had four separate age groups, soil recovery succession was also assessed between each age and the natural average. Makobulaan and Buffelskloof Nature Reserves' pine stand plots were also compared to the secondary and natural treatments.

Statistical analyses

Statistical analyses were performed using the STATISTICA 11 software package (StatSoft: Tulsa, OK, USA) with some analyses being done in Rstudio. Ordination and similarity analysis was performed in Community Analysis Package 2.13 (Henderson & Seaby 2002). QGIS V1.7.4 was used for all GIS work (GNU 2012).

Results:

Sample Areas

The four main sample areas included MNR – Makobulaan Nature Reserve, BKNR – Buffelskloof Nature Reserve, BCF – Blue Crane Farm and ESI – Eastern Shores, iSimangaliso. WS - Wakkerstroom had too few plots to include in some analyses. The largest study area sampled was the Eastern shores with 64 plots in total and a total of 294 species counted. Makobulaan and Buffelskloof had 32 and 24 plots with 164 and 174 species respectively. The two ploughed areas, Blue Crane farm and Wakkerstroom had only 16 and 8 plots, but still had a large count of total species, 124 and 97 respectively.

Species richness

In all study areas, natural vegetation treatment plots had far greater average plot species richness compared to secondary plots (Table 2.1: Plants - Table 2 and Forbs - Figure 2.3). Within the natural treatment, all study areas had similar plant and forb average species richness except for the lowland study area, ESI, which had a significantly lower average species count for all plants and forbs (Table 2.2). Within the secondary treatment all sample areas shared similar average plant species numbers. For forb species, all secondary plot averages were similar in species number (Figure 2.3).

Table 2.1: GLM for all study areas compared between sites and vegetation treatments using natural and secondary grassland total plant and forb species richness averages separately.

	d.f.	F-value	P
Total Plants			
Study areas	4	8.4	$p < 0.0001$
Grassland Treatment	1	430.5	$p < 0.0001$
Interaction	4	19.1	$p < 0.0001$
Forbs Only			
Study areas	3	17.9	$p < 0.0001$
Grassland Treatment	1	459.5	$p < 0.0001$
Interaction	3	18.3	$p < 0.0001$

Table 2.2: Average total plant species richness per nested plot (78.5m^2). # Plots = number of plots per treatment per study area, N = natural, S = secondary.

		Mean per Plot	
Study Area	# Plots	N	S
MNR	16	43.44	8.75
BKNR	12	41.58	11.75
WS	4	40.75	15.00
BCF	8	37.13	13.63
ESI	32	29.09	14.09

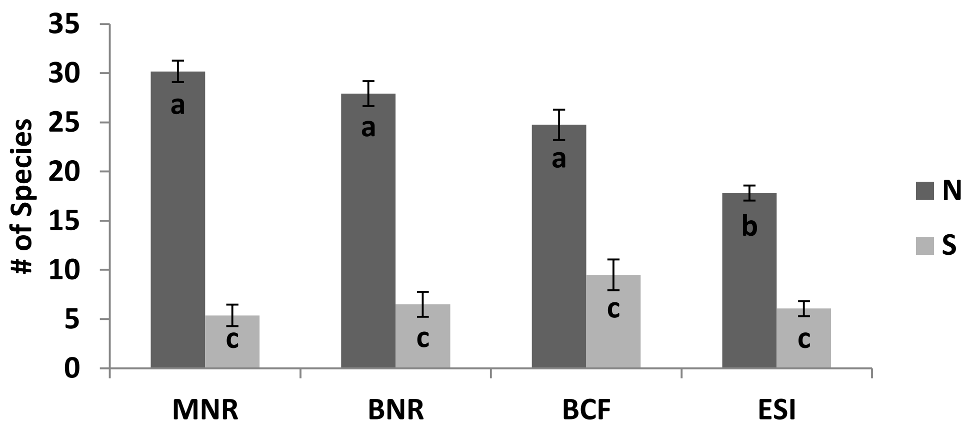


Figure 2.3: Average species richness of forbs per nested plot (78.5m^2) for natural (N) and secondary (S) treatment plots for each study area. MNR, BNR - Mesic Highveld Grassland Bioregion. BCF - Sub-escarpment Grassland Bioregion. ESI - Indian Ocean Coastal Belt. MNR, BNR and ESI - S are pine treatments. BCF - S are ploughed treatments.

Species richness patterns for fire, elevation and vegetation treatments

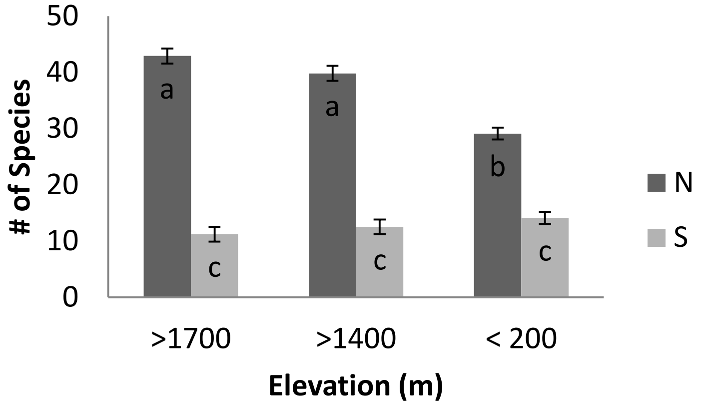
The effects of fire on plot species richness showed no significant interaction with elevation (Table 2.3, p > 0.55), a proxy for study area, when comparing total plant species. The same pattern was seen for forb species richness (Table 2.3: p > 0.35).

Plant species richness showed a significant interaction between elevation and vegetation, with the state of vegetation, natural or secondary, being the more important factor (Figure 2.4a, P<0.0001). There was no difference in richness between secondary grasslands of different elevations. The inland natural grasslands study areas (> 1700 and > 1400m) had a greater species richness when compared to the natural coastal grassland study area, < 200m. Forb species richness showed a similar pattern with elevation even when looking at plot point diversity (Figure 2.4b, P<0.0005).

Table 2.3: GLMs comparing total plant and forb species richness averages between elevation and fire treatments for all study areas.

	d.f.	F-value	P
Total Plants			
Burnt vs. Un-burnt	1	1.8	p = 0.31
Elevation	2	3.3	p = 0.23
Interaction: Fire and elevation	2	0.6	p = 0.55
Forbs Only			
Burnt vs. Un-burnt	1	0.3	p = 0.67
Elevation	1	9.0	p = 0.21
Interaction: Fire and elevation	1	0.8	p = 0.38

a.



b.

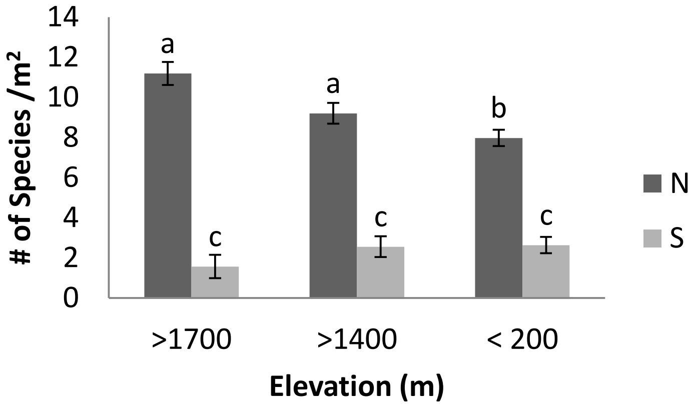


Figure 2.4a and b: a. Average species richness per nested plot for both total number of species for natural (N) and secondary (S) treatment plots at each elevation range across the study. F-value: 27.88, P<0.0001. b. Average forb species per meter squared for N and S treatments at each elevation range across the study. F-value: 9.23, P<0.0005.

When comparing plant species richness for burnt and un-burnt plots between all three vegetation treatments, natural, afforested (pine) and ploughed, there was a significant interaction with vegetation having a stronger influence (Figure 2.5, Table 2.4: $p < 0.001$). Fire only had an effect on natural burnt and un-burnt treatment plots. Natural plots supported significantly more plant species than both secondary treatments. Ploughed and afforested plots had similar plant species numbers. The same pattern was seen for forbs (Table 2.4, $p < 0.005$).

Burnt treatments showed contrasting patterns when coastal and inland grassland vegetation treatment plots were analysed separately. Coastal natural grassland vegetation burnt plots were significantly richer than un-burnt natural grassland plots (T-value = -2.2, $p < 0.05$). Inland natural burnt grassland plots were not significantly different to un-burnt natural plots (T-value = 0.88, $p = 0.38$). This same pattern was seen for forb species in the natural treatment plots. When secondary grassland treatments were separated into coastal and inland vegetation, inland secondary un-burnt plots had significantly more species than burnt plots (T-value = -2.1, $p < 0.05$). In secondary coastal grasslands un-burnt plots and burnt plots were not significantly different (T-value = 1.6, $p > 0.11$). For forbs species in secondary grasslands there was no effect observed from fire, with both burnt and un-burnt secondary plots showing similar forb numbers (All – T-value = 0.3, $p > 0.77$, Coastal – T-value = -0.2, $p > 0.82$, Inland – T-value = -0.2, $p > 0.86$).

Table 2.4: GLMs comparing total plant and forb species richness averages for vegetation treatments (Natural vs. pine and ploughed) with fire treatments for all study areas.

	d.f.	F-value	P
Total Plants			
Vegetation	2	204.2	**$p < 0.0001$**
Burnt vs. Un-burnt	1	0.1	$p = 0.77$
Interaction: Fire and vegetation	2	8.0	**$p < 0.001$**
Forbs Only			
Vegetation	2	171.7	**$p < 0.0001$**
Burnt vs. Un-burnt	1	2.1	$p = 0.15$
Interaction: Fire and vegetation	2	5.8	**$p < 0.005$**

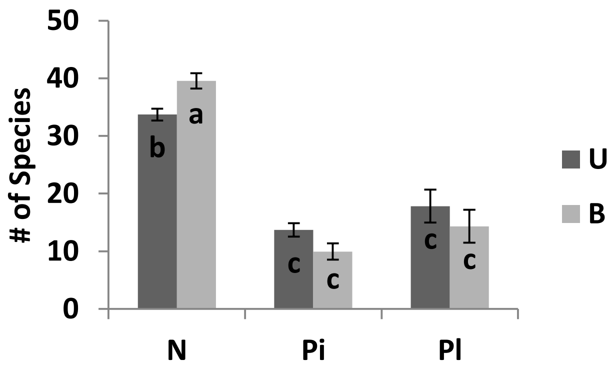


Figure 2.5: Average total plant species richness per nested plot (78.5m^2) for burnt and unburnt plots comparing vegetation treatments. N – Natural, Pi – Afforested, Pl – Ploughed. F-value: 7.94, $p < 0.001$.

Study area heterogeneity

Species accumulation graphs showed the same pattern for both total plant species (Figure 2.6a, c, e and g) and forb species (Figure 2.6b, d, f and h). Natural treatment accumulation curves started higher than secondary curves and had a steeper starting gradient than secondary curves. In some cases the gradient for secondary curves was almost flat (Figure 2.6d and f). The secondary curves started to asymptote much earlier than the natural curves. The best examples of the general pattern were for MNR and ESI which had more plots to use in the analysis (Figure2.6a and b and g and h).

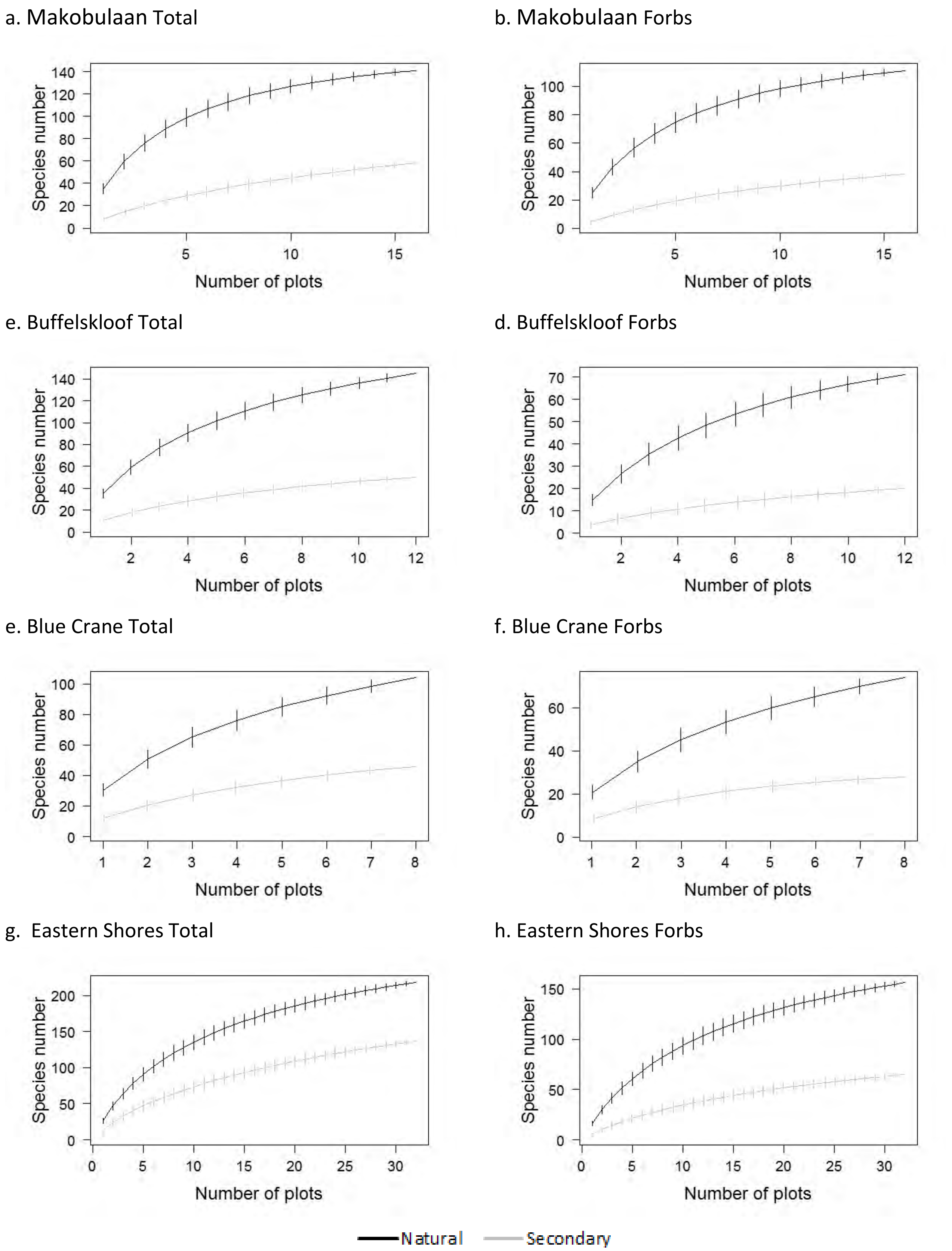


Figure 2.6 a - h: Average species rarefaction curves showing increasing species with increasing plots for both natural and secondary treatments. The left column shows total species and the right column shows forbs species only.

Species Turnover

Intra-plot species accumulation rates for natural vegetation treatments were far greater for both average total plant species and forb species for all study areas (Figure 2.7).

a. b.

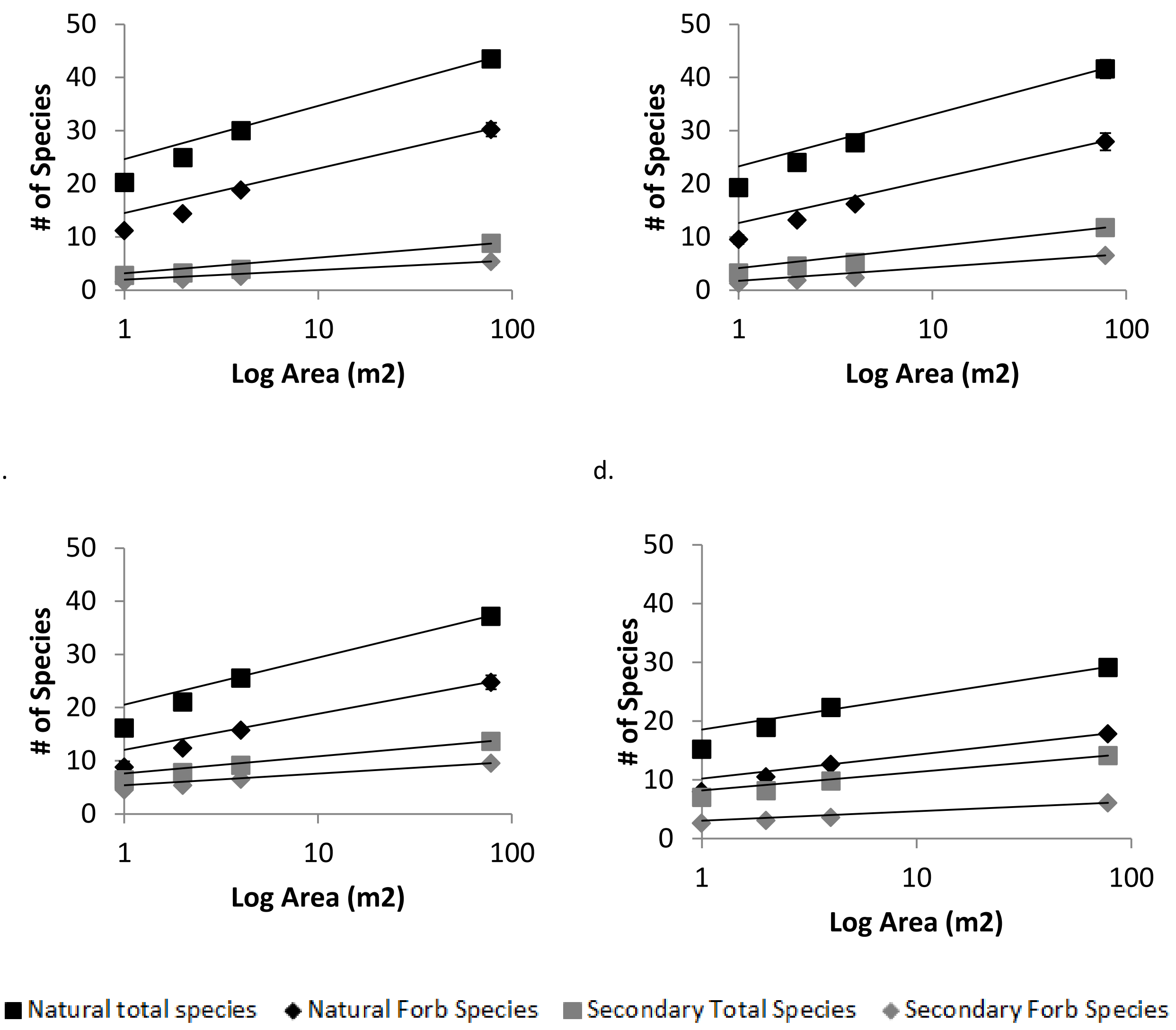


Figure 2.7 a - d: Species richness turnover for increasing nested plot area from 1m^2 to 78m^2 for main sample areas a. MNR (Natural total (NT): R^2 = 0.87, p = 0.07, Natural Forb (NF): R^2 = 0.88, p = 0.06, Secondary Total (ST): R^2 = 0.88, p < 0.05, Secondary Forb (SF): R^2 = 0.96, p < 0.05) b. BKNR (NT: R^2 = 0.89, p = 0.06, NF: R^2 = 0.90, p < 0.05, ST: R^2 = 0.97, p < 0.05, SF: R^2 = 0.98, p < 0.05), c. BCF (NT: R^2 = 0.84, p = 0.08, NF: R^2 = 0.90, p = 0.08, ST: R^2 = 0.89, p = 0.06, SF: R^2 = 0.88, p = 0.06), d. ESI (NT: R^2 = 0.79, p = 0.11, NF: R^2 = 0.80, p = 0.09, ST: R^2 = 0.89, p = 0.06, SF: R^2 = 0.95, p < 0.05). Squares indicate total species and diamonds indicate forb species only. Black indicates natural sites and grey indicates secondary treatments.

Understorey in Pine stands

Species richness in pine plantation stands (PS) was not significantly different from recently harvested plantation stands (CF). Both had similar plant species richness to secondary grassland treatment plots (8.3±2.2 vs. 5.3±1.4, T-value: -0.96, p = 0.84). Natural vegetation plots had significantly greater richness than all other vegetation treatments (Figure 2.8 F-value, p < 0.0001). All pine treatments showed similar species richness averages to secondary grasslands. No species co-occurred between the CF or PS treatments and the natural grasslands sampled. Very few individual plants were found within PS and CF transects with the average number of individual plants observed per transect in recently harvested plots at only 9±2.6 with even less within intact plantation plots at 5.4±2.4.

a.

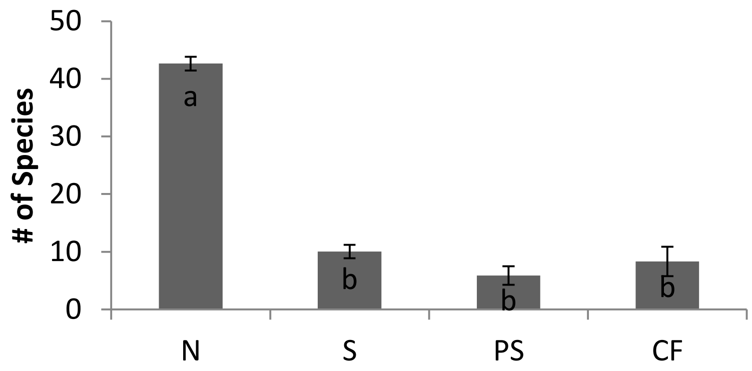


Figure 2.8: Average number of species found under afforested (PS) and recently clear-felled and burnt (CF) plots compared to natural (N) and secondary (S) treatment plots for the MNR and BKNR study areas.
F-value: 178.11, P<0.0001

Plant ground and dominant species cover

No significant interaction was observed between fire and vegetation treatments for total plant ground cover (Figure 2.9a, Table 2.5, $p > 0.40$). Burnt plots had significantly less plant cover than un-burnt plots (Table 5, $p < 0.0001$). Fire affected natural plot cover for the top five species only making the interaction between fire and vegetation treatments significant (Figure 2.9b, Table 2.5, $p < 0.0001$). For both secondary vegetation types there was no difference found between burnt and un-burnt plots. Natural vegetation plots had significantly less top five species cover than both secondary vegetation treatment plots, which did not differ from each other (Table 2.5, $p < 0.0001$).

Table 2.5: GLMs comparing the average total plant ground cover and the average total cover of the top five species for vegetation treatments (Natural vs. pine and ploughed) with fire treatments for all study areas.

	d.f.	F-value	P
Total plants ground cover			
Vegetation	2	15.2	**$p < 0.0001$**
Burnt vs. Un-burnt	1	21.0	**$p < 0.001$**
Interaction: Fire and vegetation	2	0.9	$p = 0.40$
Total cover for top five species			
Vegetation	2	246.1	**$p < 0.0001$**
Burnt vs. Un-burnt	1	11.15	**$p < 0.005$**
Interaction: Fire and vegetation	2	17.6	**$p < 0.0001$**

a. b.

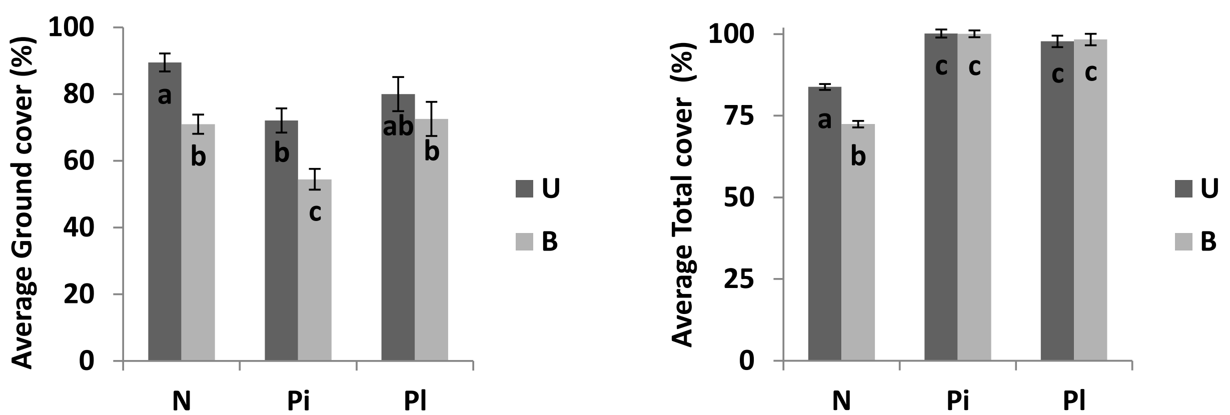


Figure 2.9: Total plant ground cover (a) and total cover of the top 5 most dominant species (b) for each vegetative treatment, N – Natural, Pi – Pine, Pl – Ploughed, for both burnt (B) and un-burnt (U) treatments.

Cover from the top 5 dominant species found in plots was significantly different between vegetative treatments (Figure 2.10, $p < 0.0001$). The 1st most dominant species for the natural treatment cover was significantly less than those of both secondary treatments. The 2nd most dominant species for natural treatment cover was more than that of secondary excised plots, but not different to the old-ploughed treatment. The 3rd to 5th most dominant species for natural vegetation plots had a higher average than both secondary vegetation treatments. The last 3 species for secondary treatment plots did not differ significantly from each other making up very small percentages of the total cover.

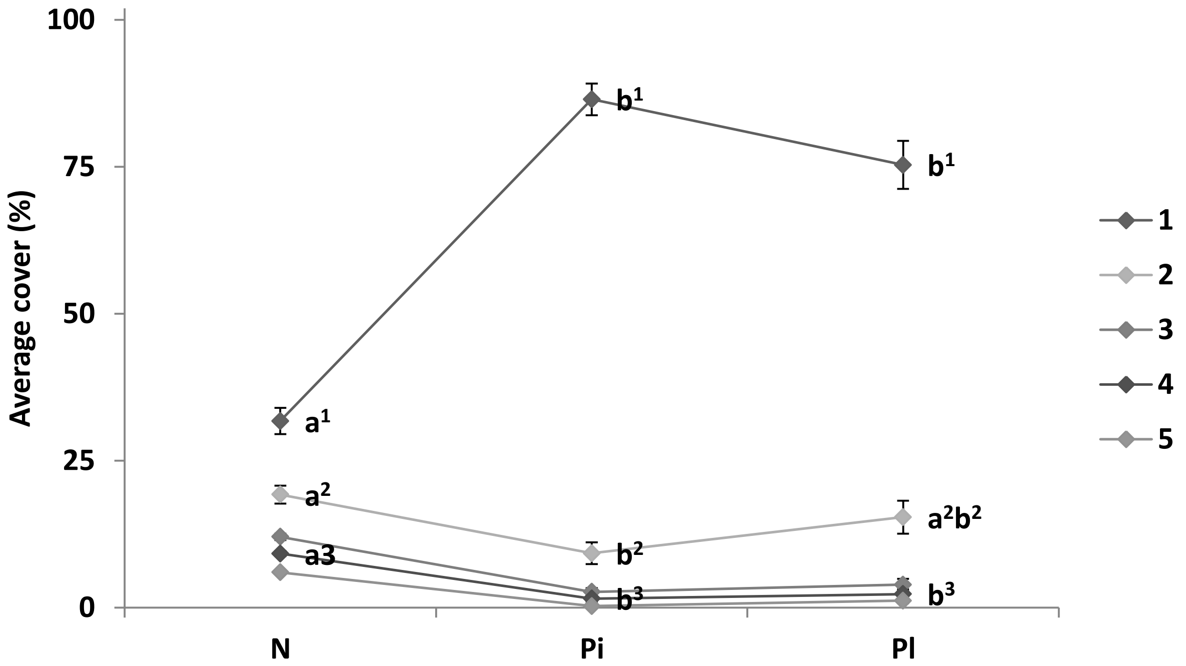


Figure 2.10: Average total percentage cover for each of the top 5 most dominant species per plot for each vegetative treatment. The number on the data label relates to the species prevalence. 1st most dominant species = 1, etc. F-value: 140, $p < 0.0001$.

Community composition

Most plant species and forb species occurred within natural treatment plots for each study area (Figure 2.11a and b). The majority of plant species were found only in natural treatments.

a. b.

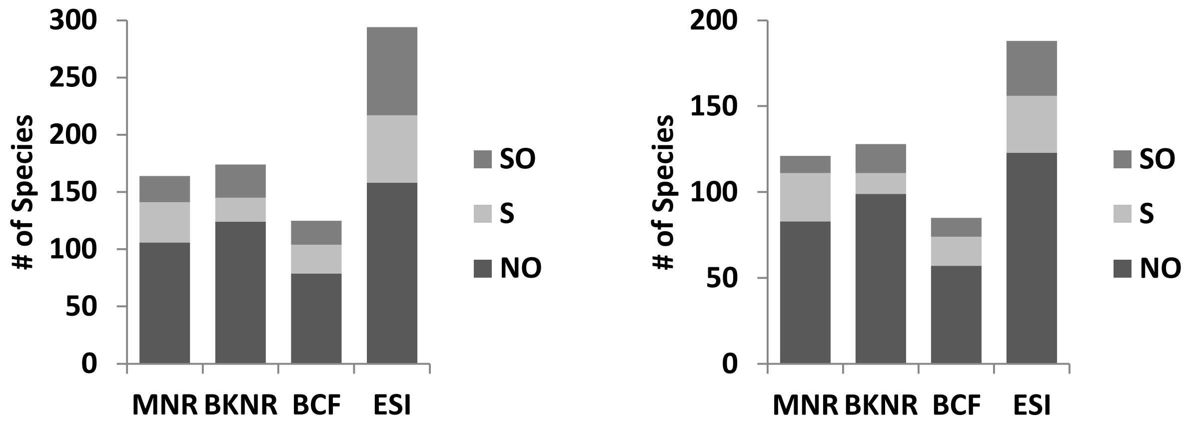


Figure 2.11: Breakdown of occurrence for total species (a) and forb species (b) for natural and secondary vegetation treatments. SO – secondary only, S – Shared by both treatments, NO – Natural Only.

Plant growth form composition

Overall there were far more forbs and geoxylic suffrutices in natural plots than secondary grasslands. Graminoid species were slightly more common in natural plots and woody species were far more prevalent in secondary plots.

In the natural treatment plots forbs made up the majority of study area plant species, between 75 – 80% (Figure 2.12a). Graminoid species were the second most abundant plant growth form making up around 20% of all species. Geoxylic suffrutices were rare, only occurring in three of the four study areas. They made up about 7% of total species for ESI, where they were very common. Woody species were very rare in natural treatments and did not occur in some areas. In the secondary treatment plots proportions changed substantially with forbs still making up the majority of the species but reduced to only 45 – 60% (Figure 2.12b). Graminoid species made up between 20 – 30%. Woody species made up between 10 – 20%. Geoxylic suffrutices occurred very rarely in secondary plots.

a.   b.

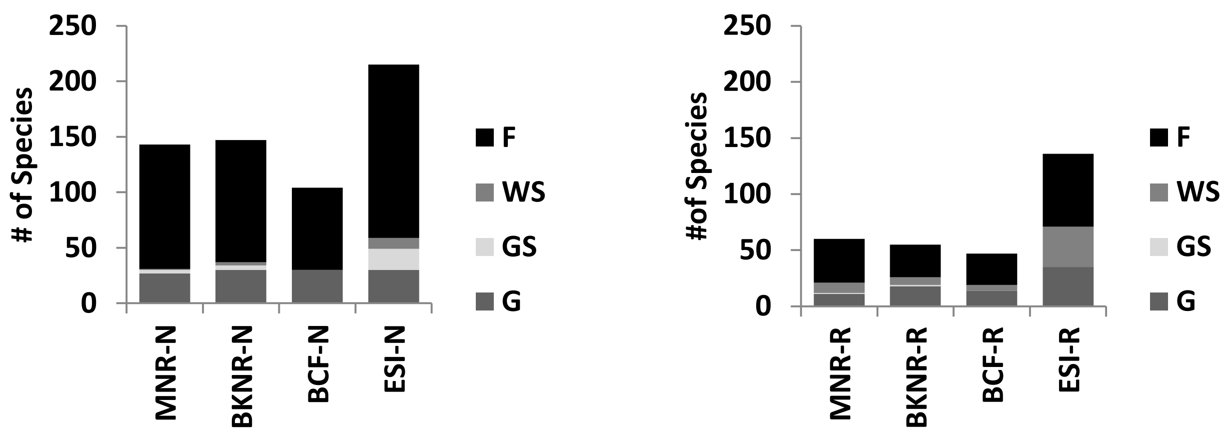


Figure 2.12: Breakdown on plant elements of species in natural (a) and secondary (b) treatments for each main study area. F – Forbs, WS – Woody Species, GS – Geoxylic Suffrutice, G – Grasses.

Ordinations

The ordinations separated natural and secondary grassland plots into two distinct plant communities for each study area (Figure 2.13).

MNR, BKNR (Figure 2.13a and b) showed a strongly clustered group of natural plots separated from a greater spread set of secondary plots. Four grasses, *Themeda triandra, Rendlia altera, Panicum natalense* and *Koeleria capensis* and one forb species *Eriosema kraussianum* made up the five most common plant species found in the MNR natural treatments. At BKNR the five most common natural treatment species included three grasses, *T. triandra*, *R. altera* and *Eragrostis racemosa*, one forb, *E. kraussianum* and one geoxylic suffrutex, *Aeschynomene rehmannii var. leptobotrya*. The secondary grassland treatment of MNR was dominated by 1 grass, *Eragrostis curvula*, three forbs species, *Rumex crispus* (an alien species) and two *Helichrysum* species as well as one shrub, *Helichrysum splendidum*. Common secondary grassland species at BKNR included three grasses *E. curvula, Cymbopogon* spp. and *Trichopteryx* spp one forb species, *Constomium natalense* and one fern species, *Ophioglossum reticulatum*. BCF (Figure 2.13c) natural plots were more spread out along y axis but had a similar pattern of secondary plots to MNR and BKNR. Common natural treatment species included three grass species, *T. triandra, Heteropogon contortus and Tristachya leucothrix* and two forb species, *Acalypha peduncularis and Helichrysum pallidum.* The top five common species for BCF secondary treatments included two grass species, *E. curvula* and a *Panicum* species and three alien forb species *R. crispus,* a *Oxalis* species and a *Bidens* species, . ESI (Figure 2.13d) had natural plots that were more spread out compared to other study areas. The secondary plots were also spread out but still distinguishable from the natural plots. Common species in the ESI natural treatments included two grass species, *T. triandra* and *Aristida junciformis,* two forb species, *Vernonia oligocephala* and *Pentanisia prunelloides* and one geoxylic suffrutex, *Parinari curatellifolia.* The five most common species in ESI secondary treatments included two grass species, *Dactyloctenium aegyptium* and *Eragrostis ciliaris,* one sedge, *Mariscus albomarginatus* and two forb species, an Asteraceae weed and *Cassytha filiformis* (an alien). Helichrysum krausii, a woody shrub, was the most common plant species found in the plots off the ESI study area, occurring frequently in both natural and secondary grassland.

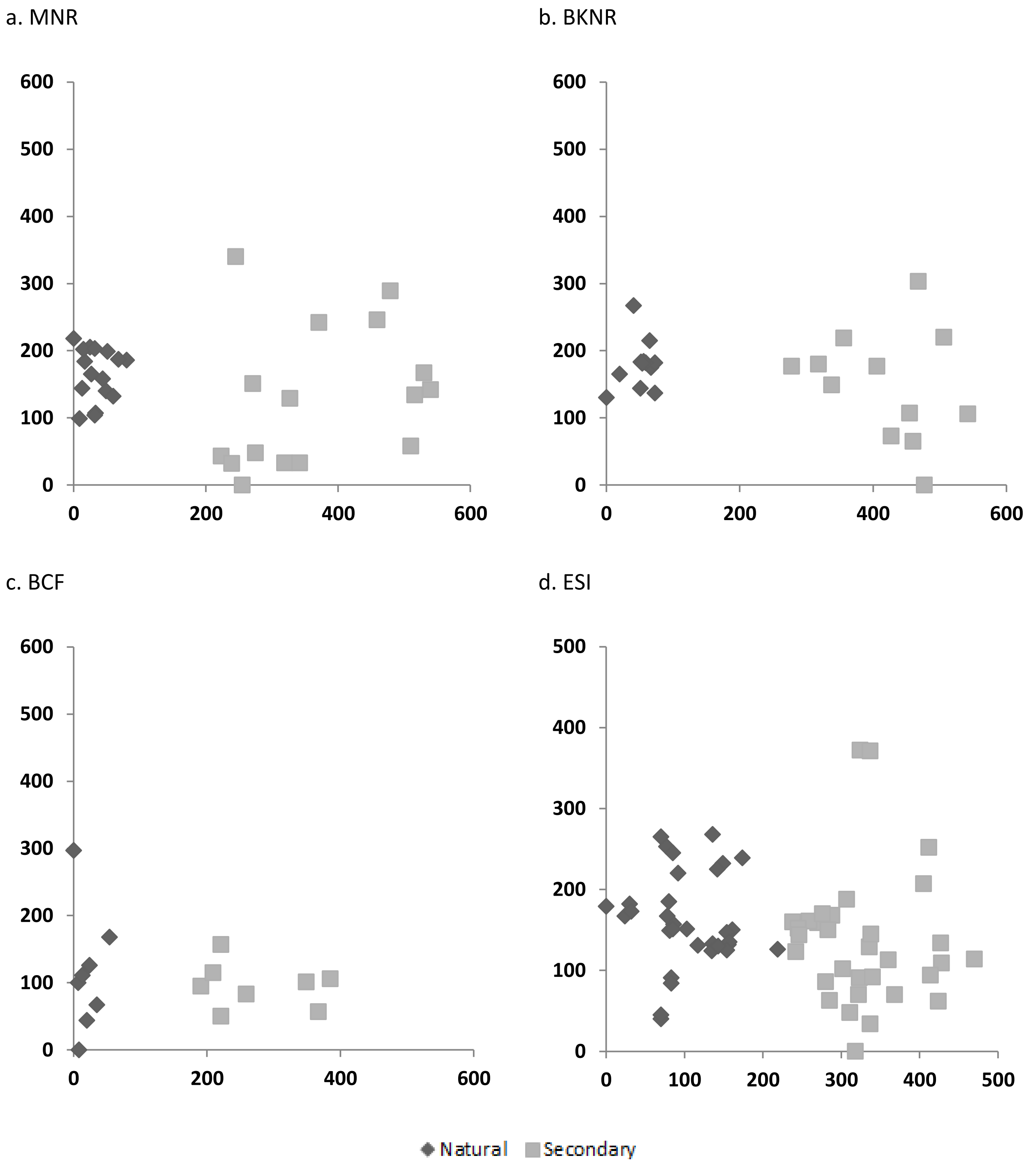


Figure 2.13: Detrended correspondence analysis (Decorana) plots showing relative similarity in composition of total plant species for plots sampled in natural and secondary vegetation for each study area. Rare species were not downweighted. MNR (a) axis 1, eigenvalue = 0.6268 and axis 2, eigenvalue = 0.4221, BKNR (b), axis 1, eigenvalue = 0. 7607 and axis 2, eigenvalue = 0. 3352, BCF (c) axis 1, eigenvalue = 0.6460 and axis 2, eigenvalue = 0.3445, ESI (d) axis 1, eigenvalue = 0.7189 and axis 2, eigenvalue = 0.3189.

Plant family richness and abundance

Natural plots supported greater family diversity than secondary plots (Appendix 7 – 10).

Within MNR, vegetation treatments shared 10 dicot and five monocot families. Natural treatments included an additional 20 dicot and 10 monocot families while secondary treatments included an additional 10 dicot and 3 monocot families.

BKNR vegetation treatments shared only two dicot and four monocot families. Natural treatments included an additional 24 dicot and eight monocot families while secondary treatments included an additional nine dicot and two monocot families.

BCF treatments shared six dicot and five monocot families. Natural treatments included an additional 15 dicot and seven monocot families while secondary treatments included an additional nine dicot and one monocot families.

ESI vegetation treatments shared 17 dicot and 4 monocot families. Natural treatments included an additional 22 dicot families and 12 monocot families while secondary treatments included an additional 10 dicot and five monocot families.

Members of the Asteraceae were found in all study areas in the natural vegetation treatments and typically were among the top two richest families: MNR had 37 species, BKNR had 35 species, BCF had 21 and ESI had 23. Poaceae and Fabaceae were the only other two families found within all the study areas. As regards Poaceae, MNR had 22 species, BKNR had 24, BCF had 25 and ESI had 22. For Fabaceae, MNR had 10, BKNR had 18, BCF had 11 and ESI had 35 species. Iridaceae and Euphorbiaceae were common families for three of the study areas. Cyperaceae was rich in species at BCF and ESI. ESI also had an additional two species-rich families, Scrophulariaceae and Commelinaceae.

For secondary treatments, only Poaceae was rich in species for all the study areas. MNR had 9 species, BKNR had 15, BCF had 12 and ESI had 25. Species of Asteraceae had high species richness at all study areas except BKNR. Fabaceae had rich species numbers for BKNR and ESI. Cyperaceae was rich at ESI.

Family abundance data showed that most families found in both grassland vegetation treatments were biased to the natural treatment and were rare in occurrence within secondary treatments (Appendix 7 – 10). These excluded Asteraceae, Poaceae and Cyperaceae which were usually evenly abundant in both.

Below ground biomass

Natural treatment plots held greater non-graminoid root biomass than secondary treatment plots (Table 2.6, Figure 2.14). The natural plot average for ESI was 5.33kg.m^{-2} of dry mass, significantly greater than the secondary plot dry mass average of 0.41kg.m^{-2} (Figure 2.14a, $p > 0.005$). Natural plots in MNR had an average dry mass of 1.41kg.m^{-2}, significantly greater than the secondary plots dry mass average of 0.26kg.m^{-2} (Figure 2.14a, $p > 0.05$). Bracken fern (*Pteridium aquilinum*) rhizomes were present in some of the secondary plots at MNR and greatly increased the amount of root biomass found in one site by 60 times, from 0.01kg.m^{-2} to 1.54kg.m^{-2}.

Forb root biomass showed a similar pattern. The natural plot average for forb root dry mass in ESI was 3.17kg.m^{-2}, significantly greater than the secondary plot average of 0.12kg.m^{-2} dry mass (Figure 2.14b, $p < 0.05$). MNR natural plot forb root dry mass average of 1.30kg.m^{-2} was significantly greater than the secondary plots forbs root dry mass average of 0.01kg.m^{-2} (Figure 2.14b, $p < 0.05$). Excluding graminoids, the MNR natural treatment plant root biomass was made up entirely of forbs while ESI natural treatment forbs only made up 60% of the biomass and the rest was made up of geoxylic suffrutices. For the MNR secondary treatment forbs only made up 3% of the root biomass, the rest was bracken fern, while for ESI forbs made up 30% and the rest was made up mostly of the roots of *Helichrysum kraussii* and also alien woody species such as guava (*Psidium guajava*) and *Chromolaena odorata*.

Table 2.6: Independent t-test by variables to compare the average below ground biomass for non-graminoid plant species and forb species between vegetation treatments (Natural vs. secondary) for Makobalaan (MNR) and Eastern Shores (ESI).

	d.f.	T - value	P
Plant biomass			
MNR	9	-2.39	**$p < 0.05$**
ESI	14	-3.54	**$p < 0.005$**
Forb biomass			
MNR	9	-2.75	**$p < 0.05$**
ESI	14	-2.57	**$p < 0.05$**

a. b.

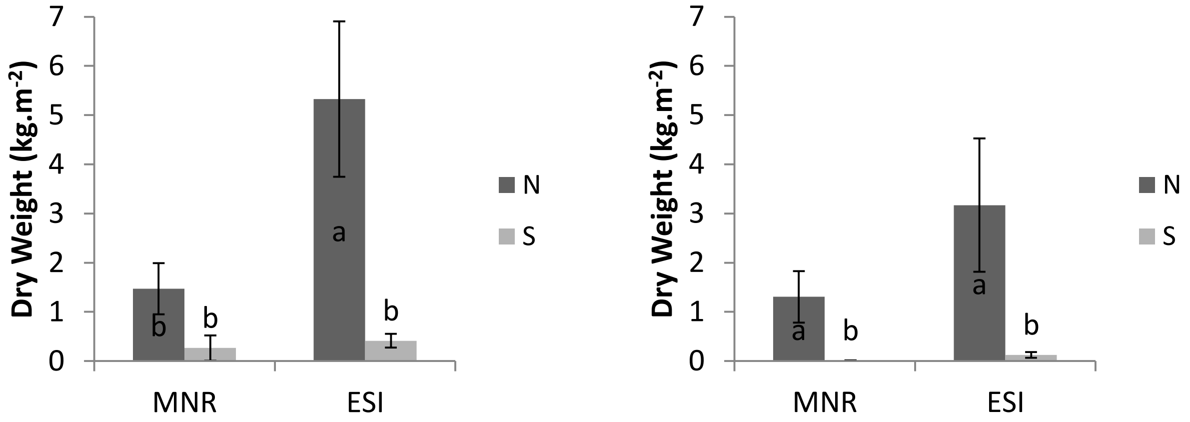


Figure 2.14: Average below ground biomass of non-graminoid plant species (dry mass in kgm^{-2}) for two study areas, Makobulaan Nature Reserve and the Eastern Shores, iSimangaliso. a. Non-graminoid root biomass (includes woody species) and b. forb only root biomass for natural (N) and secondary (S) treatments.

Succession

There was no successional trend of increasing plant and forb richness for either MNR or ESI (Figure 2.15a-d, Tukey post-hoc tests). No turnover in species richness was seen between younger and older secondary MNR plots, with the same species being dominant in both ages. Some turnover in species richness was seen in the older secondary ESI plots but these were only additional pioneer or weedy species not found in natural plots.

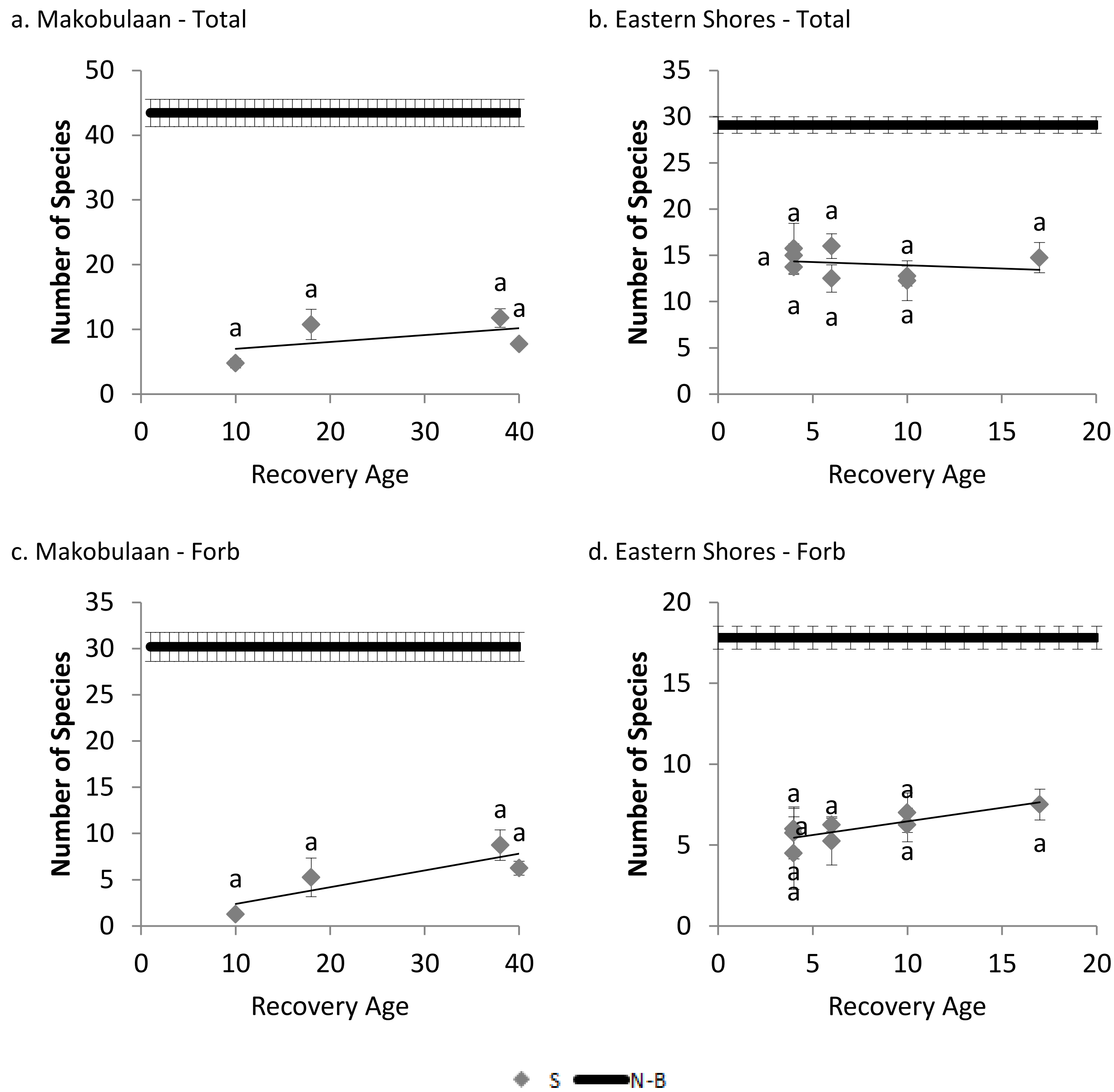


Figure 2.15: Change in species richness with increasing time since deforestation for Makobulaan (a, c – 40 years) and Eastern Shores study areas (b, d – 20 years). The top two figures show total species richness with time and the bottom two figures show forb species richness with time (a. R^2 = 0.25, p = 0.50, b. R^2 = 0.05, p = 0.62, c. R^2 = 0.75, p = 0.14, d. R^2 = 0.65, p < 0.05). The solid line (N-B) represents the average number of species for natural treatments and the grey diamonds (S) represents secondary plot age species averages.

Soils

Soil properties at each of the study areas measured showed a mix of results (Table 2.7a and b). Makobulaan (Table 2.7a - MNR) secondary treatment plots differed significantly from natural treatment plots for only three of the tested nutrients, Ca, Mg and K, all of which were lower in secondary treatments ($p < 0.0005$). Pine stands were also found to be significantly lower for these three nutrients when compared to natural plots ($p < 0.0005$). Pine stands had significantly more C% ($p < 0.005$) and N% ($p < 0.0005$) than natural plots. Although pine stands and secondary plots had far greater mean Al concentrations than natural grasslands, there variances were high and therefore not significantly different from natural plots ($p = 0.19$). Secondary plots were mostly similar in soil properties to natural plots, but there was a significant difference found between some younger and older secondary plots for N%, C% and P – Total ($p < 0.0005$).

For N%, C%, Ca, Mg and K, younger plots had similar mean values to the natural plots. Older plots, however, had significantly lower means than those of the natural plots ($p < 0.0005$). Al averages found in 10 year old secondary plots were significantly greater than natural plot averages ($p < 0.0005$), while older secondary plots had similar averages to natural plots. Available P increased with increasing plot age but this was not statistically significant.

Buffelskloof (Table 2.7b - BKNR) secondary plots were significantly different from natural plots for all tested nutrients ($p < 0.05$) except for pH and Mn. Means for secondary plots were lower than those of natural plots. Pine stands had significantly lower nutrient concentrations than natural plots for pH, Ca and K ($p < 0.05$).

Blue Crane Farm (Table2.7b – BCF) secondary ploughed plots did not differ significantly from natural plots for any nutrient value.

Table 2.7a and b: Soil properties comparison for vegetation treatments (Natural - N vs. secondary - S and pine stand – PS or Ploughed – Pl) for Makobalaan (MNR - a), Buffelskloof (BKNR - b) and Blue Crane (BCF - b). MNR includes two separate age treatments, 10 and 40 years since deforestation. Significant differences are shown for natural treatment values versus all other treatments, ns: p >0.05, ***: p < 0.0005, **: p < 0.005, *: p < 0.05. **Bold** denotes significant differences between the two age treatments. n = number of replicates.

a. MNR	N n = 11	S n = 16		PS n = 10		10 years n = 4		40 years n = 4	
N%	0.25	0.20	ns	0.31	**	**0.27**	ns	**0.11**	***
C%	4.10	3.44	ns	6.02	***	**5.07**	ns	**1.83**	***
pH	3.83	3.81	ns	3.5	ns	3.7	ns	3.85	ns
	n = 8			n = 6					
P - total (mg/kg)	319.00	287.00	ns	444.00	ns	**386.86**	ns	**206.06**	ns
P - avail. (mg/kg)	28.38	16.13	ns	38.50	ns	13	ns	22.25	ns
Ca (cmol/kg)	0.95	0.45	***	0.24	***	0.35	***	0.25	***
Mg (cmol/kg)	0.52	0.21	***	0.18	***	0.18	***	0.13	***
Na (mg/kg)	13.88	10.65	ns	18.33	ns	13.00	ns	7.25	ns
K (mg/kg)	91.38	39.50	***	38.33	***	40	***	33	***
Al (mg/kg)	212.31	394.21	ns	646.60	ns	**1106.50**	***	**188.02**	ns

b.	BKNR					BCF		
	N n = 8	S n = 7		PS n = 4		N n = 4	Pl n = 3	
N%	0.34	0.16	***	0.27	ns	0.47	0.37	ns
C%	5.67	2.90	**	5.84	ns	7.64	5.80	ns
PH	3.99	3.91	ns	3.7	*	4.00	4.07	ns
P - avail. (mg/kg)	28.13	14.29	*	20	ns	36.00	40.00	ns
Ca (cmol/kg)	0.57	0.27	**	0.18	**	1.65	1.44	ns
Mg (cmol/kg)	0.37	0.14	*	0.17	ns	1.01	0.81	ns
Na (mg/kg)	18.50	9.71	*	19.25	ns	22.50	18.67	ns
K (mg/kg)	74.00	30.71	**	28.25	**	154.25	129.00	ns
Mn (mg/kg)	15.38	2.94	ns	1.8	ns			
Zn (mg/kg)	0.66	0.30	**	0.52	ns			

DISCUSSION:

Composition and characteristics of the South African grassland biome

Natural grassland systems throughout South Africa support diverse plant communities. In the present study, resprouting forbs dominated the grasslands in terms of number of species, being the most diverse plant life form. The plant alpha diversity was far greater at higher elevations overall, while the coastal mosaic grassland supported far more woody individuals including many geoxylic suffrutices which are far more abundant and diverse in this area. Although all of the natural grasslands were heterogeneous, there were far more locally widespread species sampled at higher elevations with the Eastern Shores area supporting a lot more rare plant species that gave the Eastern shores a higher beta diversity. Forb richness in natural grasslands was seemingly increased in burnt areas, as a result of fire-stimulated growth seen in natural grassland forb species (Uys *et al*. 2004). Fire decreased ground cover in the higher natural grasslands and particularly the cover of the dominant species compared to un-burnt veld. Forbs are able to take advantage of this post-fire reduction in above ground grass competition.

This research indicated several possible criteria that could be used to identify natural grassland and distinguish it from secondary grassland (Appendix 11). Focusing on natural grassland characteristics one could identify a natural grassland by the diversity in number and cover of its plant species. More specifically this would include a. an evenly distributed grass species vegetation cover combined with b. a high alpha diversity in forb species.
c. High resprouting forb USO biomass would be the third criteria to test for. These criteria are more distinct in natural grassland if they have been recently burnt.

Are secondary grasslands recovering from afforestation and ploughing?

Both afforestation and ploughing had negative effects on secondary grassland systems. Each land use resulted in fundamentally different plant communities compared to natural grassland vegetation and although differing slightly with different elevations this pattern is similar for all study areas. As in the secondary coastal grassland sampled by Zaloumis and

Bond (2011), higher elevation secondary grasslands supported far more homogenous communities that are species poor with little intra-plot turnover. Secondary grasslands were dominated by fewer widespread invasive species and communities were very monospecific. Burnt secondary grasslands had a barren appearance. The plants they supported had no fire stimulated flowering response. Unlike natural grasslands, there was no change in proportional plant cover, or a seeming increase in forb richness, as there were no forbs to take advantage of the reduced grass competition. Grasses therefore remain the visually dominant plant life form.

Secondary grasslands were not only missing a whole suite of natural grassland species but families and plant functional types had been lost. Most notable was the lack of resprouting forbs. Natural grasslands contained up to 31000kg of below ground forb root mass per hectare compared to less than 5% of this biomass in secondary grasslands. Resprouting forbs had been replaced by weedy or woody elements, members of the Asteraceae and Fabaceae being the most frequent. This catastrophic loss of forbs in secondary grasslands raises a number of questions: How important are forbs in the context of grassland ecology? What are the implications of removing resprouting forbs from grassland systems? At a local scale forbs and their roots are food for animals (Laden & Wrangham 2005). Natural grasslands support far higher grazing potential than secondary grasslands (Mentis 1999). Resprouting forbs probably play an important role in nutrient cycling within the soil because of lower C:N ratios than C4 grasses. However this would need further investigation. Along with an intact grassy layer they help prevent top soil loss, responding appropriately to and promoting resilience against natural disturbances such as fire and high intensity erosive rainfall. Bare burnt secondary grassland are prone to soil erosion as they have little in the way of established plants to hold the soil together when faced with high levels of water run-off. The large quantity of underground forb biomass has to play an enormous role in the carbon cycle, trapping carbon in the soil within perennial underground storage organs. When these forbs are removed all this carbon is released back into the atmosphere while secondary grasslands lack the forbs with large USOs and may therefore be slow to sequester carbon. This only incorporates the non-graminoid plant species sampled during this study, however, grass roots biomass, which we suspect to be far lower in secondary grassland, could also contribute to carbon sequestration in natural grasslands.

The grasslands within the Eastern Shores study area occurred on sandy, naturally nutrient-poor soils (Taylor 2004). For the measured soil parameters conifer plantations have had little effect on these soils and post-afforestation soil attributes were found to be similar in most respects to natural grasslands (James 1998) so that soil differences are unlikely to account for divergent grassland composition in this particular instance.

Blue Crane Farm was situated in grassland that has well drained, nutrient poor sedimentary material with a low to medium proportion of clay content (Mucina & Rutherford 2006). Our investigation into soil attributes showed that for this study area nutrient level did not differ, yet there was an increased level of weedy elements rather than natural plants that had established in the secondary grassland. It is possible that fertilizer was added when this area was cultivated and it has managed to maintain similar nutrient levels to natural grassland. It would appear that, like with some ex-arable grassland in Australia (Scott & Morgan 2012), changes in soil properties are not the main driver of recovery after cropland abandonment.

The higher elevation grasslands of Makobulaan and Buffelskloof were both situated in grasslands derived namely from shale and quartzite which are also nutrient poor (Mucina & Rutherford 2006). Buffelskloof and Makobulaan showed contradicting results with Buffelskloofs secondary grasslands being lower than natural grasslands in most soil attributes tested while Makobulaan mostly matched secondary and natural soils with similar nutrient elements. Soils of older secondary grasslands at Makobulaan had *lower* nutrient values than their younger counterparts. This suggests a loss of nutrients after pine plantations are cleared perhaps because secondary grasslands are unable to maintain their initial nutrient levels due to the loss of a functioning nutrient cycle or, as these two areas are mountainous, through the erosion of the remaining top soil. These hypotheses could be explored in further studies. If excised and clear-felled pine plantations share similar nutrient levels to the surrounding natural grassland one should aim management efforts to prevent or limit the nutrients lost with increasing recovery age. Further restoration studies could explore such management or restoration interventions.

Potential physical and biological restoration constraints for South African grassland

Restoration success in the secondary grasslands of our study has been poor. There are two main obstacles that determine and drive secondary succession, namely physical and biological limitations (Pywell 2002, Soc. Eco Rest. 2004, Suding *et al.* 2004, Kardol *et al.* 2008). The focus of this study was on the latter. However, in most restoration studies, it is often the physical environment that takes the spotlight (Mentis 1999, Young 2000, Mentis 2006). Human related disturbances can alter soil characteristics and these impede plant re-establishment, whether it is through an increase in 'toxic' elements or the loss of important nutrients (Bradshaw 2000, Shu *et al*. 2005, Mentis 2006, Le Stradic 2012). The addition of specific nutrients can also be detrimental: for example, adding nitrogen to the soil can promote the spread of weedy and alien species (Cole and Lunt 2005, Prober & Thiele 2005, Norman *et al.* 2006, Kardol *et al.* 2008, Scott & Morgan 2012). These instances are more relevant in the case of mined grasslands or similar severe disturbances that have resulted in the loss, mixing or contamination of top soil (Mentis 1999, Mentis 2006) or cropland that has been fertilised, or depleted of nutrients (Scott & Morgan 2008, van Oudtshoorn *et al.* 2011). Our soil evidence in excised plantations and abandoned ploughed land suggests that it cannot be the only factor playing a role; at least it is not the biggest limiting factor. Plant communities have managed to establish in both coastal and higher elevation secondary grasslands but are not even close to representing their reference natural grasslands, even after 40 years.

Biological constraints seem to be the prime limitation on the successful recolonisation of secondary grasslands by natural grassland elements. Firstly, grasses are competitive life forms (Scholes & Archer 1997, Bond 2008, Dickson & Busby 2008), and have been shown to affect forb seedling establishment (Chapter 3). Mono-specific stands of invasive grass species within secondary grasslands are hostile environments for seedling recruitment. Secondly, grassland forbs have the ability persist in the grassy fire adapted land scape (Bond & Parr 2010, Zaloumis & Bond 2011). Traits that promote persistence come at the cost of those that promote recruitment (Bond & Midgley 2001). This trade-off makes grassland forbs poor colonisers and probably accounts for the absences of any successional trend towards a natural grassland state. Seed and dispersal limitations as key limiting restoration factors have been noted in many other instances of restoration, particularly in grassland systems (Lunt 1994, Pywell 2002, Buisson & Dutoit 2004, Cole & Lunt 2005, Prober & Thiele 2005, *Buisson et al.* 2006, Dickson & Busby 2008, Kardol *et al.* 2008, Scott & Morgan 2012). In Western Australia restored shrublands after bauxite mining lacked most native resprouting species found in un-mined vegetation (Norman *et al*. 2006) and, as in this study, Norman *et al.* (2006) found no evidence of succession towards increasing numbers of resprouters even with applied active restoration methods.

This study has allowed us to understand the biological consequences of human disturbances on our ancient grassland systems. We see a community shift from a diverse persistent forb and grass assemblage to one with a few annual forbs and mono-specific dominant grasses. The implications of this have largely been ignored in the South African context as more and more pristine grasslands are planted up and/or ploughed.

Our priority is to promote grassland conservation. However, there are large areas of afforested, mined and ploughed land that need restoration to the original grassland state meriting the development of methods that will help promote restoration. Restoration priorities should include the immediate management of secondary grassland soil to ensure no further top soil and nutrients are lost, however, a second phase is needed that re-introduces native plant propagules. The aim here is to re-introduce some native species to help prevent invasive species from creating entire mono-specific stands (Pywell 2002) and to re-establish several important grassland functions. Returning grazing value and grass cover is much more attainable (Mentis 1999) than completely restoring diversity, below ground biomass and ensuring top soil stability. However, a multiphase plan can at least encourage a more natural succession. Without a focus on the biological limitations, any restoration effort of secondary grasslands is unlikely to achieve true restoration and, as our soil evidence suggests, could even lead to further degradation. Without a natural species composition secondary grasslands respond differently to natural disturbances, such as fire and grazing, compared to natural grasslands. The incorrect application of frequent burning of secondary grassland may expose more bare soil to erosion.

Our grasslands differ fundamentally from those of Europe and North America (Bond & Parr 2010) and this would need to be considered when approaching conventional methods of grassland restoration. South African grasslands need frequent fire otherwise we see a loss of shade-intolerant forbs and grasses and the community shifts to more shade loving species (Fynn *et al*. 2004, Uys *et al*. 2004), forbs also have a fire stimulated response (Zaloumis & Bond 2011). North American prairies, in contrast, lose forb diversity with a high fire frequency. In the prairies, grazing reduces grassy biomass and promotes forb diversity (Collins 1992, Leach & Givnish 1996, Collins *et al*. 1998, Olff & Ritchie 1998, Knapp *et al*. 2004). This being the case our secondary grasslands may function more like these northern hemisphere systems.

In comparison, Australian temperate and South American grasslands show more similarities to our grasslands. They are both fire driven systems with fire stimulated flowering (Lunt 1994, 1995, Lunt & Morgan 1999, Morgan 1999, Overbeck *et al.* 2007, Overbeck & Pfadenhauer 2007). Australian grasslands, after fire suppression, have some grass species that self-shade and few native species that survive longer than 10 years in the absence of burning (Lunt 1995, Cole & Lunt 2005). Brazil's tropical grassy biomes share several characteristic families and genera with our grasslands, support a large diversity of resprouting forb and geoxylic plant species and lose these species without frequent fire events (Overbeck *et al.* 2007). Like South African grasslands, Brazil's grasslands are ancient systems and are rapidly being transformed for agriculture (Overbeck & Pfadenhauer 2007).

Conclusion:

This study has shown that secondary grasslands are fundamentally different from natural grasslands and when resprouting forbs are removed from the grassland system they are very unlikely to return. This work has helped us to develop criteria to better identify natural grasslands. We need to determine how to overcome the identified biological barriers that are restricting grassland succession. We cannot continue to ignore grassland systems in a conservation context because without any easy way of restoring grasslands we risk completely losing more and more of this valuable system to human related activities. Grasslands in South Africa are fragile systems, at least as fragile as primary forests and may take decades to centuries to recover from a major disturbance.

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Chapter 3: Transplanting mixed species sods as a method for re-introducing natural resprouting species into secondary grasslands in South Africa.

Abstract:

The conservation of threatened ecosystems is not always possible. When this is the case ecological restoration can be used as a conservation tool. South African grasslands are currently severely threatened and fragmented systems, their value and diversity only just beginning to be understood. We have much secondary grassland within South Africa that has failed to restore through the process of natural succession. Practical restoration methods are needed that take into account the biological limitations that grasslands face which prevent natural succession after their diversity has been lost. We identified secondary grassland vegetation plots to transplant grassland mixed species sods collected within natural mesic grassland areas situated in both Mpumalanga and KwaZulu-Natal. Sods were placed within two different grass treatments, cleared and un-cleared, to help identify whether grass competition played a role in restricting sod species survival. Using species percentage cover and richness we monitored the success of these recovery plots over a period of 12 months. Although sods lost species percentage cover and richness over this time period it was within an acceptable range. The biggest loss was of the forb and geoxylic suffrutex plant growth forms. However, forbs on average only lost at most a third of their richness. Grass competition did not appear to play a role within this short term study. Although sod transplants may be an impractical method of restoration on a larger scale, we were able to determine that it is possible to increase species diversity within secondary grasslands by this method. We can build on this study to develop better ways of re-introducing natural grassland species into secondary grassland.

Introduction:

Restoration ecology is a useful tool in conservation and can be used to help towards preserving threatened ecosystems when formal conservation protection is not possible (Hobbs & Norton 1996, Young 2000, Pywell *et al*. 2002, Prober & Thiele 2005, Dickson & Busby 2008, Kardol *et al*. 2008). Conservation is also not the only reason why one may consider a restoration project. Rehabilitating or restoring land to a point where it is partially productive and regains some level of ecological function can be useful in commercial, private and communal lands. One example is to return livestock grazing potential of rangeland after it has been mined, controlling weed invasion, securing soil to prevent erosion and restoring hydrological function (Mentis 1999, Driver *et al.* 2004, Suding *et al.* 2004, Cole & Lunt 2005). In the process these goals can also add conservation value to the land and help contribute to other restoration targets.

The goal of restoration is to assist with the recovery of degraded or damaged ecosystems. Restoration involves the re-establishment of native ecological processes, functions and the productivity of the ecosystem facilitating the return of the original plant community structure and diversity over time (Soc. Eco Rest. 2004). This necessitates the use of historical and existing references within the landscapes and commonly includes sampling of undisturbed natural vegetation within close proximity to the recovering area.

Grassland systems are under increasing pressure around the world, have suffered some extreme levels of habitat loss and are being identified more and more as conservation priorities (Lunt 1995, Myers *et al.* 2000, Reyers *et al.* 2001, Bonn & Gaston 2005, Hoekstra *et al.* 2005, *Buisson et al.* 2006a, Overbeck *et al.* 2007, van Oudtshoorn *et al.* 2011). As a result grassland restoration has become an important issue especially in the context of what grasslands provide for humans. Intact grassland systems provide several notably important ecosystem services; grazing production, water moderation and erosion control are some examples (Reyers & Tosh 2003, Overbeck *et al.* 2007, Blignaut *et al.* 2008, Hardy 2008, O'Connor & Kuyler 2009). South Africa's grasslands provide considerable high quality grazing land to our livestock industry, while being the source of water for the entire Gauteng

Province as well as a considerable portion of South Africa's population. Intact grasslands ensure that this water does not all rush directly into our river systems after a rainfall event (Driver *et al.* 2004, Hardy 2008, Blignaut *et al.* 2010). This same reduction of surface flow ensures that valuable top soil is not eroded away and lost to the river and ocean systems, silting up dams on the way. Restoration that reduces siltation rates has large economic benefits by extending the life of major water reservoirs thereby reducing the need for constructing new major civil engineering works (Driver *et al.* 2004, Blignaut *et al.* 2010).

Even though grasslands are more and more considered highly valuable, there is still a lack of literature available on restoration efforts, especially in South Africa. Grassland restoration projects have been increasing particularly within Europe, North America and Australia (Schramm 1978, Lunt 1994, Leach & Givnish 1996, Pärtel *et al.* 1998, Pywell *et al.* 2002, Cole & Lunt 2005, Prober & Thiele 2005, Buisson *et al.* 2006b, Dickson & Busby 2008, Kardol *et al.* 2008, van Oudtshoorn *et al.* 2011, Piqueray *et al.* 2011). However, successful grassland restoration is still very difficult with most projects showing varied or limited success. Some return vegetation that only vaguely resembles the natural system, with more successful projects also returning several useful or common species. This restored vegetation does not often respond to natural drivers such as fire in the same way as the original vegetation (Grant & Loneragan 2001, Smith *et al.* 2004, Norman *et al.* 2006). Full restoration is generally still out of reach.

Most available grassland restoration literature is from the Northern Hemisphere where grassland systems may differ fundamentally from our southern equivalents (Bond & Parr 2010, Zaloumis & Bond 2011). The application of restoration methods developed for northern grasslands may therefore be ineffective unless these differences in the ecology of grasslands are recognized and considered in each local context. There are a growing number of reports on grassland restoration from the southern hemisphere, including work in southern Africa, Australia, and South America (Lunt 1994, Lunt & Morgan 1999, Mentis 1999, Fynn *et al.* 2004, Uys *et al.* 2004, Cole & Lunt 2005, Mentis 2006, Overbeck *et al.* 2007, van Oudtshoorn *et al.* 2011, Zaloumis & Bond 2011, Le Stradic 2012, Scott & Morgan 2012).

These can be used as guidelines to the most practical methods on how best to improve the accomplishment of restoration goals.

There are two key limitations for grassland restoration (Ash et al. 1994, Bradshaw 1997, Bakker & Berendse 1999, Cole & Lunt 2005, Prober & Theile 2005, Mentis 2006, Shu et al. 2005, Kiehl 2010). Physical factors can slow or prevent restoration. These include fragmentation of the vegetation producing barriers between sites that prevent propagule dispersal. Dispersal failures can be through the loss of dispersers or increased distances between sites. Physical problems also include alteration in soil physical and chemical properties which can influence plant growth and competition. Soil disturbances can promote weedy species through the presence of added fertilizers, or reduce the pool of plants that can easily establish because of residual toxic elements or the hardening or erosion of top soil.

There are also biological reasons that may make it difficult for plants to re-colonise recovering areas (Bakker et al. 1996, Hoffman 1998, Norman et al. 2006, Overbeck & Pfadenhauer 2007, Bond & Parr 2010, Zaloumis & Bond 2011, Chapter 2). South African and South American grasslands are populated by resprouting species that are adapted to fire and grazing. These plants may have traded off sexual reproduction for the ability to persist in frequently burnt grasslands (Bond & Parr 2010, Zaloumis & Bond 2011). Successful seedling establishment may be very rare while vegetative reproduction by the slow process clonal spread is relatively common.

Re-seeding disturbed habitats is a common method used for reintroducing native species in grasslands (Pywell *et al.* 2002, Buisson *et al.* 2006b, Dickson & Busby 2007, Prober & Theile 2005, Kardol *et al.* 2008). Even if seed is available it can often be too expensive. A key limiting factor for restoration projects in Australian and South African grasslands has been access to sufficient seeds and propagules for the majority of the species (Cole & Lunt 2005, Lunt, Zaloumis & Bond 2011). In South Africa this could be particularly limiting since the grasslands are rich in species and a large proportion of the original species composition has been lost (Chapter 2). One study in Australian temperate grassland concluded that, even for relatively small areas, enormous numbers of plant seeds would be needed to successfully re-establish just a few species in their natural densities (Lunt 1994). This may be impractical or even impossible as sufficient seed or seedlings of plants required for restoring the South African grasslands are probably not available or easy to attain. Even if seed banks are available for forb species in temperate grasslands they are poor in species and seed abundance and tend to be short-lived (Lunt 1995). Physical and biological limitations must therefore be identified before we can determine the best approach for a restoration project.

Restoring cover and basic common pasture species will reduce erosion, help prevent weed invasions and allow for restoration of fire regimes (Cole & Lunt 2005, Prober & Theile 2005). However many of these recovered grasslands tend to be mono-specific stands, not high value grazing land and tend to discourage any further colonising of native species (Mentis 1999, Chapter 2, Chapter 4). Most grass species that were easy to use in previous restoration projects or easily invaded secondary grasslands on their own are usually highly competitive (Fynn *et al.* 2009). A study in the USA showed that forb seedling establishment can be negatively affected by high densities of dominant grass species (Dickson & Busby 2007). However, if done correctly, grass restoration is also the initial step in the restoration process preparing a suitable environment to promote grassland restoration efforts.

Collecting sods or turfs from a reference habitat is one way of re-introducing plant species into secondary environments (Pärtel *et al.* 1998, Cole & Lunt 2005). Ideally this recreates a source of seeds and living plant species within a microenvironment derived from the primary source. In the South African context seed banks are not present and species are suspected of reproducing vegetatively (Uys 2006). Establishing adult plants of forbs and grasses through sod transfer could allow species to gain a foothold in the secondary environment. It is also possible that combining a method like sod transfer with large scale seeding, spreading of hay and/or plug restoration projects could improve biodiversity in the restored system.

To determine whether a sod transfer method has been a success requires both short term and long term considerations. One way is to compare species cover and richness over time. However for a short term study species cover may be less useful as a plant species may lose most of its cover during and after the transplant process but remain alive and dormant within the sod. If this is the case we would expect a drop in percentage ground cover for a follow up survey at the end of the growing season and some gain in ground cover again once the sod is established during the next growing season. Fire may also complicate assessments when measuring cover. This could be due to the vegetation cover reduction of sods sourced from burnt grassland, or, when restored plots that have been burnt are re-surveyed. In diverse grasslands, species composition may be too heterogeneous for evaluating restoration success. Another option is to consider the different plant growth forms found in the grasslands. By measuring these growth forms separately we can determine if any functional groups suffer a higher proportion of species loss than any other.

The focus of this chapter is to assess the success of sod transplants as a tool for restoration. Since grass is known to be highly competitive and limiting to the establishment of non-grass seedlings, it is important to understand how mono-specific stands in recovering grasslands affect restoration efforts. We planted mix species sods in areas that we had cleared of plants and in areas where no previous clearing had been done. We measured species percentage cover and species richness over 12 months after transplanting to evaluate success. We were then able to identify potentially useful criteria for evaluating plant establishment success in future grassland restoration efforts. We had no experimental control over the burning of natural and secondary grasslands used in the study. Sods were collected from available natural grassland, both burnt and un-burnt. Some secondary grassland that contained restoration plots where burnt during the winter between the initial and final evaluations of transplanted sods. However this allowed us to investigate whether selecting sods from burnt or un-burnt natural grassland had an effect on the plant establishment. We could also measure if burnt and un-burnt restoration plots showed different comparative results in success. The role of fire on sod transplanting and establishment would seem to be important in this fire adapted landscape. It could affect the initial establishment success by either promoting or hindering the way natural grassland species cope with transplanting. Harvesting recently burnt sods may add extra stress to the plant species when being transplanted because these plants are already responding to a fire disturbance.

The study also aimed to establish a small scale long term monitoring project to determine, once established, how transplanted sods cope in the secondary environment and if they manage to successfully expand beyond the original transplanted sod. True restoration success would be for species found in the sods to increase and eventually to successfully reproduce in restored grassland.

We asked the following questions:

1. Do grassland plant species survive sod transplanting and if they do, how well?
2. Which plant growth forms struggle to establish?
3. Can we establish criteria to determine successful transplanting of mix species sods?
4. How does grass competition affect transplanting success?
5. Can fire influence sod and plot transplant success?

Restoration requires time so that projects that have strong baselines as well as the potential to be monitored indefinitely are key to helping derive and refine restorative techniques in the future. Though I report results after only the first year of restoration, the project was designed to allow continued future monitoring to determine longer term success in increasing diversity in restored grasslands.

Methods:

Study area selection

Three study areas were used in this study. Found in mesic grassland systems, Buffelskloof Nature Reserve (BKNR) in Mpumalanga, Blue Crane Farm (BCF) and Eastern Shores, iSimagaliso (ESI) in KwaZulu-Natal. Each had multiple examples of secondary and primary grassland within close proximity. BKNR and ESI were both areas with land recovering from afforestation while BCF had land recovering from ploughing. These study areas had also undergone survey comparisons between recovering and natural grassland states (Chapter 2, Zaloumis & Bond, 2011). The land owners and managers of these areas were interested in trying restoration methods for their secondary grasslands and to support the long term monitoring of the project after the initial restoration stage was completed.

BKNR occurs within the Lydenburg Montane Grassland system (GM18 – Mucina & Rutherford 2006). This area's mean annual precipitation (MAP) is 858mm with a rainfall range between 660mm and 1180mm during the year. Frequent mists occur during most months of the year and the region experiences frost. The mean annual temperature (MAT) is 14.1˚C. The grasslands occur on high altitude plateaus with undulating plains, peaks, slopes, hills and deep valleys, all set within the Northern Escarpment region. The soil of this grassland type is mostly derived from shale and quartzite. -25° 18' 15.156" South, 30° 31' 11.928" East.

BCF occurred near Nottingham Road, within a mid-altitudinal grassland system called the Drakensberg Foothill Moist Grassland type (Gs 10 – Mucina & Rutherford 2006). Within this grassland type the MAP is 890mm with a MAT of 14.6 ˚C. The region experiences frost. These grasslands occur on a moderately rolling and mountainous landscape which is often carved out by river gorges. The soil geology is dominated by mudstones and sandstones. The soils of the sedimentary parent material are well drained with a depth of 800mm and a clay content between 15- 55%. -29° 19' 10.8114" South, 30° 4' 13.62" East.

ESI occurs within the Indian Ocean Coastal Belt including both Maputaland Coastal Belt (CB 1) and Maputaland Wooded Grassland (CB 2) vegetation (Mucina & Rutherford 2006). ESI makes up part of the iSimangaliso Wetland Park. Here the MAP can be up to 1200mm, with about 1000mm on Lake St. Lucia. Most days have high humidity and high temperatures with no frost. MAT is 21.1 ˚C. The mosaic of coastal dune forest and grassland occurs on flat coastal plains bordering wetlands and undulating old vegetated dunes. The soils are derived from young material that is mostly less than 18000 years old consisting of quaternary sediments of marine origin. The soil consists of mainly yellowish sands, which are highly leached and nutrient poor-28° 11' 25.8354" South, 32° 28' 45.084" East.

Plot selection and preparation

Plots were selected in sites that were sampled during the survey work (Chapter 2). They were randomly selected within a recovering area by the using a spun stick method. We prepared 7 plots at both the BKNR and BCF and 12 plots at ESI. At BKNR plots ranged from 1600m and 1750m above sea level. At BCF plots ranged between 1475m and 1520m above sea level. At ESI plots ranged between 30m and 90m above sea level.

Plots were prepared and planted during the growing season of 2011/12 from November until January. We started with BKNR in November, moved to BCF in December and then ESI in January. Staves and rope were used to set up a five x five meter squared plot. Twenty-five $1m^2$ quadrats were then marked out (Figure 3.1). Using a random number generator, 14 of these quadrats were then selected and prepared to receive sods directly from natural grassland areas. Seven of these quadrats were left alone to be used as grass competition controls labelled as un-cleared (U) treatments. The remaining seven were prepared as cleared (C) treatments. These quadrats were marked out and we removed all the plant biomass both above and below-ground from the entire meter squared. After this was completed we dug 30x 30cm holes within the centre of all treatment quadrats to receive the prepared sods.

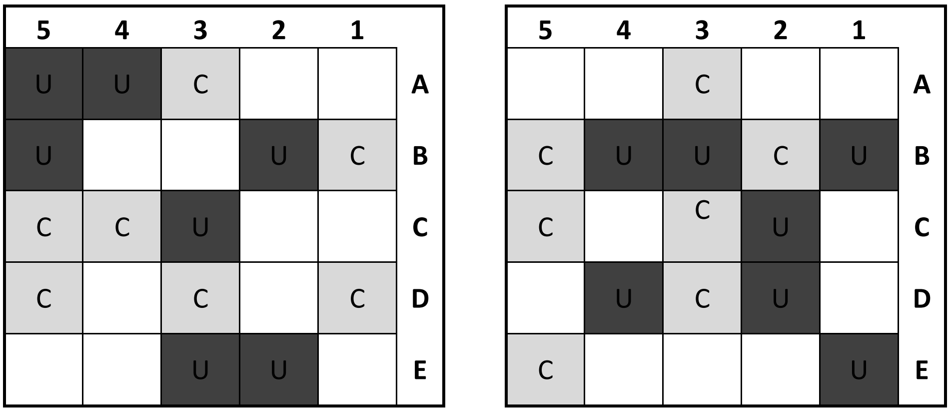

Figure 3.1: Two examples of how restoration plots were set up. Seven un-cleared (U) and seven cleared (C) quadrats. 1 – 5 and A – E were used to map the plots. Refer to Appendix 12.

Mixed species sods were collected at natural grassland sites within close proximity, (either within walking distance of the plot or within access of a road). Sods weighed up to about 30kg and could not be carried far easily (Appendix 12). Fourteen grassland sods of around 25cm x 25cm were selected for each plot and then dug up. A wandering transect method was used to locate common grassland grass and forb species that would be ideal to target for the experimental sods. At ESI, sods had to be contained in beer case boxes to prevent the sandy soil from spilling from a sod. The inland areas, BKNR and BCF, had more clay-rich soil and sods could be transported without losing soil. Once dug up, sods were transported to the plots and placed in their allocated prepared holes. Sods sourced from recently burnt and un-burnt sites were noted. This allowed for a comparison between restoration plots that only received burnt sods and those that only received un-burnt.

Sods were then planted into their allocated quadrats, tightly packing in soil around the sod, and then watered well. Each plot received around 60 litres of water. A 20l knapsack and 5 – 10l water bottles were used to water the sod to drench their roots as much as possible. Stakes were positioned on one or two corners of the plot to mark out its position and orientation for follow up surveys.

Surveys

Three separate surveys were conducted during the course of the project. Before the sods were planted they were surveyed for species richness and species specific percentage ground cover (survey T1). In March, near the end of the growing season, we returned to re-survey each plot in all three study areas (survey T2). This survey was used primarily to observe how the sods coped with the transplant shock. In November 2012 in the middle of the next growing season we returned for the final survey at the BCF and ESI study areas (survey T3). During the follow up surveys, T2 and T3, all species, within the original sod extent were included in the survey. It was noted if restoration plots had been burnt during 2012 since receiving sods. Unfortunately time constraints prevented us from surveying BKNR for a third time. For this reason BKNR has been excluded from most of the analyses for this study. Analyses of the change in the number of species from the T1 to T2 survey were used to assess how successfully plants had established in their new environment after having faced one winter season. Comparison of T3 to the T2 survey in March could also help determine the best time for transplanting sods.

Data analysis

General Linear Models (GLM) where used to compare species richness and percentage cover for time and grass competition treatments. Study areas were analysed for these separately. Net loss of species richness and percentage cover was compared between study area and grass competition treatments.

Sod species richness and percentage cover

Species richness and percentage ground cover was summed up for each quadrat and the average of all the quadrats was compared between the three surveys over the 12 month period of the study. Sod survival was accessed by using the T1 data as a baseline and comparing it to the later survey data.

Plant growth forms

To understand how different grassland plant growth forms respond to transplanting we calculated the average species richness and percentage cover for all surveys of the following growth forms: graminoids – grasses and sedges, forbs and geoxylic suffrutices (see White 1979). The change in values between surveys was then assessed. These changes in time were also used to assess if certain growth forms needed longer recovery time to stabilise within a new environment.

Study areas

Average net change in plot (T2-T1) data were assessed between all the study areas to look for different initial responses towards transplanting. Average T3-T1 was compared between BCF and ESI as they were the only two study areas with a final survey. This was an attempt to establish if there is a general pattern between the study areas.

Burnt and un-burnt plots

Restored plots that received burnt sods were compared to plots that received un-burnt sods. The comparison was only done at T1 and T3 to establish an effect of fire on percentage cover and richness numbers at T1 and species survival by T3. Plots that had been burnt in the period between T2 and T3 were also compared to un-burnt plots to determine if fire had an effect on percentage cover and richness for the final survey.

Statistical Analysis

GLM statistical analysis was performed using the STATISTICA 11 software package (StatSoft; Tulsa, OK, USA).

Results:

Initial sod plant cover and richness

Across the study each sod supported, on average, between 6 and 9 species and between 80 and 110% cover (Table 3.1). BKNR supported the highest total average richness and BCF supported the highest total average plant percentage cover. BCF was the only study area that initially showed a significant difference between grass treatments with sods in cleared quadrats supporting a higher species richness and cover than sods in un-cleared quadrats (Table 2.1: Richness: F-Value: 5.4, $p < 0.05$ and Cover: F – Value: 7.7, $P < 0.01$). No significant difference for species richness and percentage cover was found between the grass treatments of both BKNR and ESI.

Final sod plant cover and richness

By survey T3 each study area had shown a loss in the number of species and the total cover percentage of species overall (Table 3.1, 'Tfinal'). Cleared and un-cleared quadrats showed similar averages for both richness and percentage cover for all but one study area. ESI plant percentage cover showed a significant difference between grass treatments (Table 3.1: F-value: 9.1, $p < 0.005$).

Table 3.1: Total average plant percentage cover and species richness of quadrats for both treatments in each study area. Significant difference between grass treatments (Cleared (C) and Un-cleared (U)) = *.

Study Area	N	Grass Treatment	**Species Percentage Cover (%)**		**Species Richness**	
			T1	**Tfinal**	**T1**	**Tfinal**
BKNR	49	C	82.4	73.1	8.6	6.9
BKNR	49	U	81.2	68.0	8.0	6.4
BCF	49	C	106.9*	75.7	7.3*	6.1
BCF	49	U	99.3*	76.3	6.5*	6.0
ESI	70	C	85.7	87.6*	7.6	6.8
ESI	70	U	90.3	73.1*	8.0	6.9

Change of sod diversity for BCF and ESI over 12 months

Initially both BCF and ESI showed similar patterns of species and percentage cover loss (Figure 3.2). BCF lost significant plant cover by the T2 survey dropping about 30% and remaining the same by the T3 survey (Figure 3.2a, Table 3.2, $p < 0.0001$). With the same pattern BCF average species richness dropped significantly by 1.4 species per plot, recovering by 0.6, but not significantly so (Figure 3.2b, Table 3.2, $p < 0.0001$).

ESI also significantly dropped 30% by the T2 survey and recovered significantly by 20% for the T3 survey (Figure 3.2c, Table 3.2, $p < 0.0001$). Average species richness for ESI dropped significantly by 2 by T2 and then recovered significantly by 1 species per quadrat by T3 (Figure 3.2d, Table 3.2, $p < 0.0001$).

Table 3.2: GLM of Blue Crane Farm and Eastern Shores, iSimangaliso restoration plot averages for three surveys over 12 months assessing the changes in plant percentage cover and species richness for transplanted sods.

Treatment	**d.f.**	**F-value**	**P**
Blue Crane Farm survey comparison (n = 98) **Eastern Shores**			
Cover	2	42.8	p < 0.0001
Richness	2	49.8	p < 0.0001
iSimangaliso survey comparison (n = 140)			
Cover	2	19.6	p < 0.0001
Richness	2	21.3	p < 0.0001

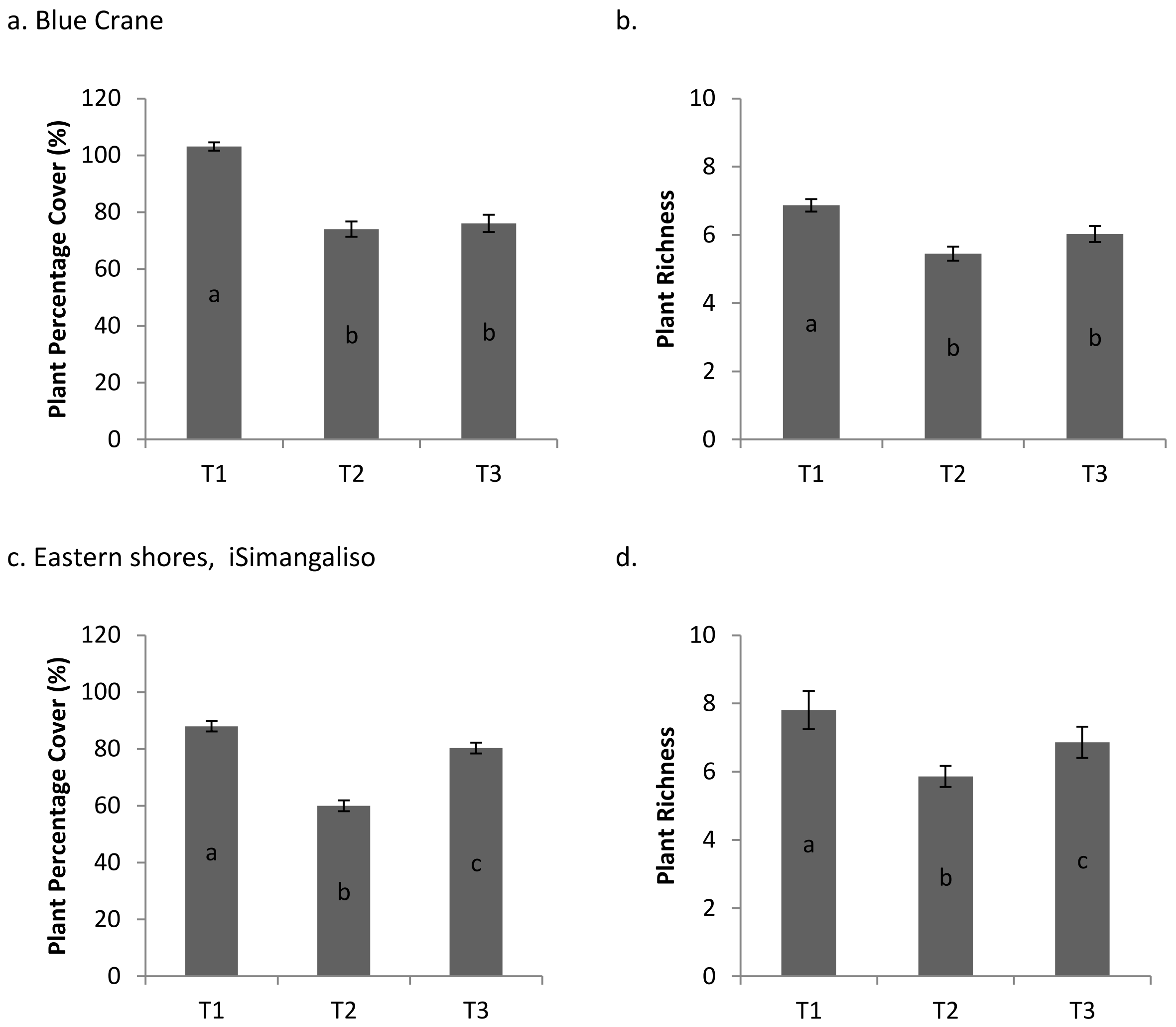

Figure 3.2: Average quadrat plant percentage cover and species richness of restored quadrats for three surveys over a period of 12 months for both Blue Crane Farm (BCF, a and b) and Eastern Shores, iSimangaliso (ESI, c and d).

Plant growth form composition

Grass species dominated the BCF transplanted sods percentage cover in the initial survey, T1 (Figure 3.3a). Forbs then contributed the second highest followed by sedges with the least. Grasses and forbs showed a loss of percentage cover by survey T2 and observed similar values at survey T3. The loss of grass and forb percentage cover between T1 and T2 was significant (Table 3.3, $p < 0.0001$).

Grass and forb plant species made up the majority of the species richness for BCF transplanted sods at T1 (Figure 3.3b). Grass and sedge richness stayed the same throughout the study while the average number of forb species per sod dropped significantly between T1 and T2 and did not change again by T3(Table 3.3, P < 0.0001).

ESI featured four plant elements including Geoxylic Suffrutices (Figure 3.3c, Appendix 2). Grass species dominated the initial average percentage cover, followed by forbs. Sedges and geoxylic suffrutices both made up small amounts. Sedge percentage cover increased slightly between all the sample periods. Otherwise all other plant elements experienced a loss of cover with geoxylic suffrutices experiencing the most and almost disappearing. Grasses dropped by 20% cover per sod at T2 and then recovered by 10% at T3. Forbs had lost half their percentage cover by T2 but managed to gain some back by T3. Other than sedges, all other plant elements experienced significant decreases in percentage cover between T1 and T2 and a significant recovery was seen only for Forbs and Grasses between T2 and T3 (Table 3.3, P < 0.0001)

Forbs made up the largest average species richness per sod at ESI, followed by grasses, then geoxylic suffrutices and sedges (Figure 3.3d). By T2 forbs dropped by 0.8 and remained at a similar average by T3. Grasses and sedges remained the same for all surveys. Geoxylic suffrutices dropped significantly almost disappearing by T2 and did not change again by T3. The drop in species richness averages for forbs and geoxylic suffrutices was significant between T1 and T2 surveys, geoxylic suffrutices losing the highest proportion of species (Table 3.3, P < 0.0001).

Table 3.3: GLM of Blue Crane Farm and Eastern Shores, iSimangaliso restoration plot averages for three surveys over 12 months assessing the changes in plant growth form percentage cover and species richness for transplanted sods with time. F – Forbs, G – Grasses, S – Sedges, GS – Geoxylic suffrutices

Treatment	d.f.	F-value	P
Blue Crane Farm survey comparison (n = 98)			
Cover – T1	4	5.3	p < 0.0001
Richness – T1	4	6.1	p < 0.0001
Eastern Shores, iSimangaliso survey comparison (n = 140)			
Cover – T1	6	8.5	p < 0.0001
Richness – T1	6	9.7	p < 0.0001

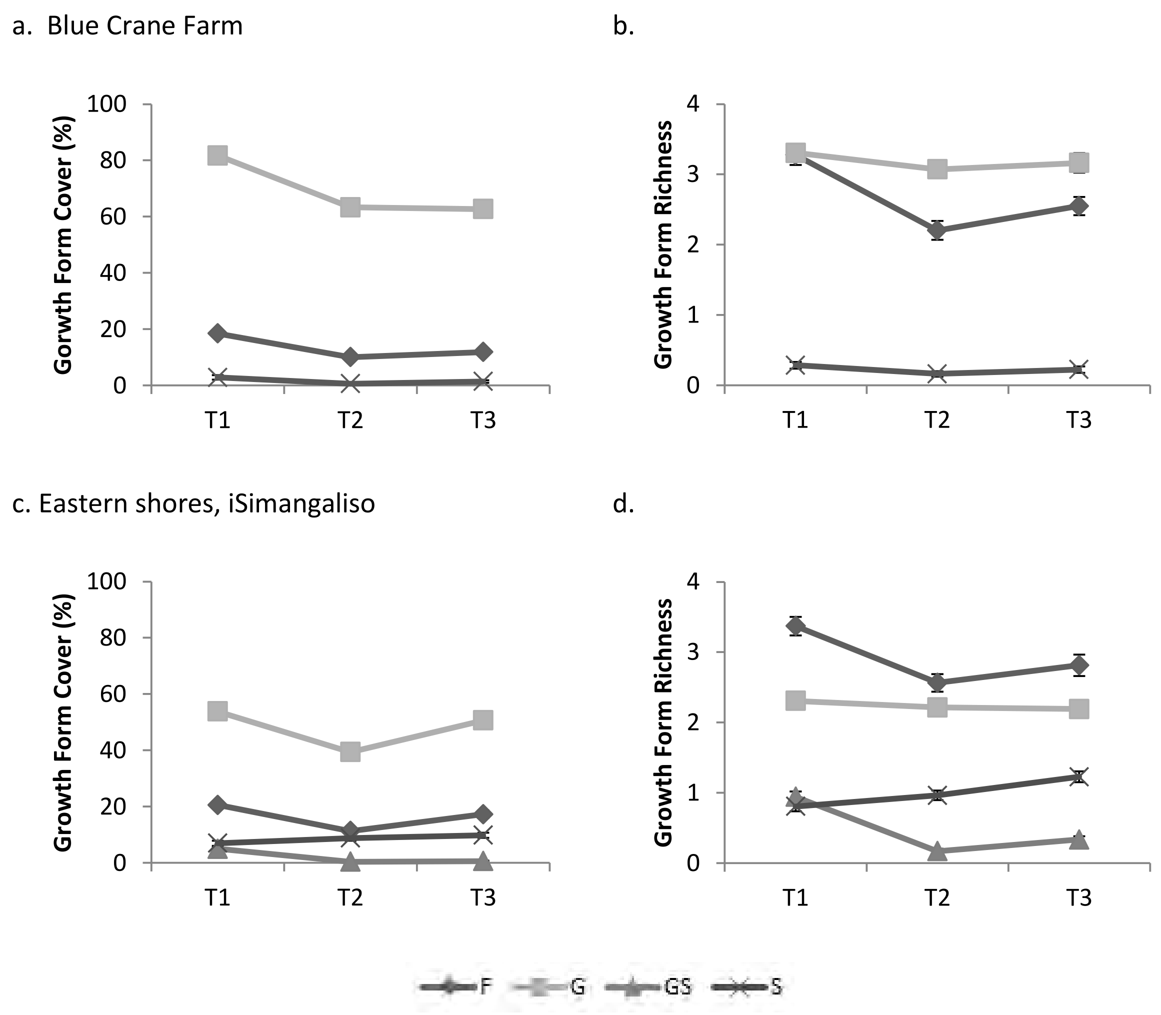

Figure 3.3: Average plant growth form percentage cover and species richness make up per quadrat for each survey time at both the Blue Crane Farm (a and b) and the Eastern Shores, iSimangaliso (c and d). F – forbs, G – Grass, GS – Geoxylic suffrutices, S – Sedges.

Comparison of plant cover and richness after four months

All restoration study areas observed a loss in average percentage cover between surveys T1 and T2 (Figure 3.4a). BKNR lost the least, followed by ESI, while BCF lost the most. In general sods in cleared quadrats lost less cover than sods in un-cleared quadrats but this was not significant (Table 3.4, p = 0.17). BKNR and BCF both lost similar cover for their comparative grass treatments while ESI lost less cover in the cleared sods compared to the un-cleared sods, but this was not quite statistically significant (Table 3.4, Figure 3.4a, p = 0.17).

Net change in species richness was negative for all the study areas and they all lost similar numbers of species per sod (Figure 3.4b). There was no significant grass treatment affect recorded within or between sites (Table 3.4, p = 0.98).

Table 3.4: GLM comparing all three study area restoration plot averages for the three surveys over 12 months. Net changes between surveys were assessed in plant percentage cover and species richness for transplanted sods with time and between grass treatments.

Treatment	d.f.	F-value	P
Percentage Cover			
BKNR vs. BCF vs. ESI	2	15.3	p = 0.06
Cleared vs. un-cleared	1	4.3	p = 0.17
Study Area vs. Grass	2	1.8	p = 0.17
Richness			
BKNR vs. BCF vs. ESI	2	1.6	p = 0.39
Cleared vs. un-cleared	1	0.0	p = 0.98
Study Area vs. Grass	2	1.0	p = 0.37

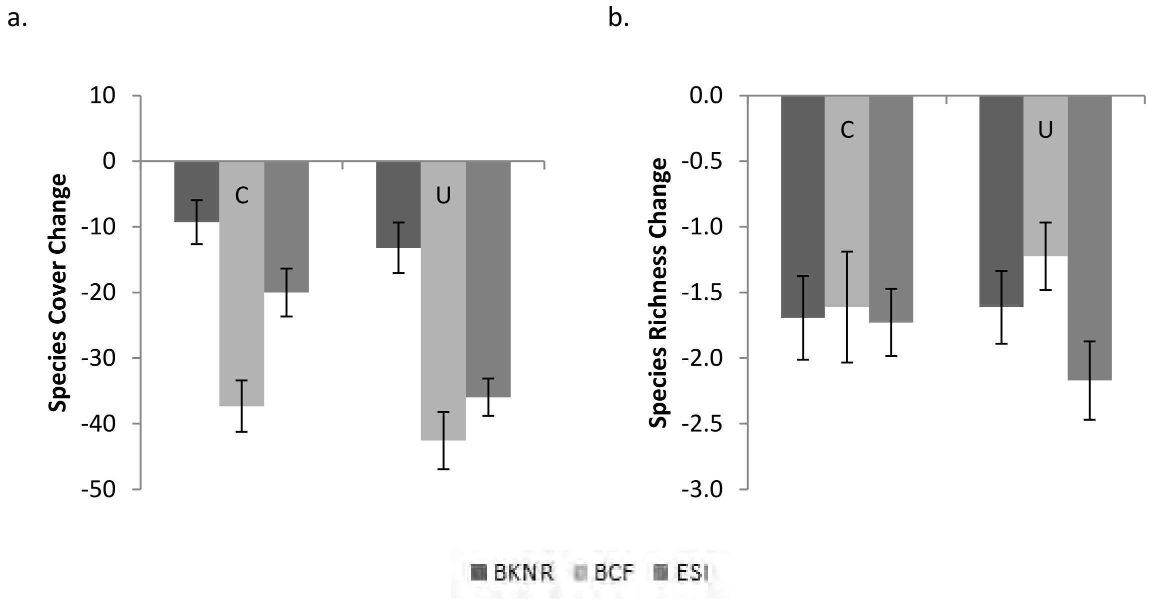

Figure 3.4: Average net change in sod plant species richness and percentage cover of restored quadrats at the T2 survey in both grass treatments for all study areas. C – Cleared, U – Un-cleared.

Comparison of plant cover and richness after 12 months

No significant difference in net cover loss was found between BCF and ESI for T3-T1 (Table 3.5, Figure 3.5a, p = 0.40). There was no significant difference between grass treatments overall (Table 3.5, p = 0.76). There was a significant interaction between study area and grass treatment with ESI cleared plots losing no cover while all the other treatments lost similar amounts of cover (Table 3.5, p < 0.002). For species richness no significant effect was found between study areas or grass treatment and for the interaction between the areas and grass treatments (Table 3.5, Figure 3.5b).

Table 3.5: GLM comparing Blue Crane Farm and Eastern Shores, iSimangaliso restoration plot averages for the three surveys over 12 months. Net changes between surveys were assessed in plant percentage cover and species richness for transplanted sods with time and between grass treatments.

Treatment	d.f.	F-value	P
Percentage Cover			
BCF vs. ESI	1	2.0	p = 0.40
Cleared vs. un-cleared	1	0.2	p = 0.76
Study Area vs. Grass	1	10.0	p < 0.002
Richness			
BCF vs. ESI	1	0.1	p = 0.87
Cleared vs. un-cleared	1	0.3	p = 0.68
Study Area vs. Grass	1	2.0	p = 0.16

a. b.

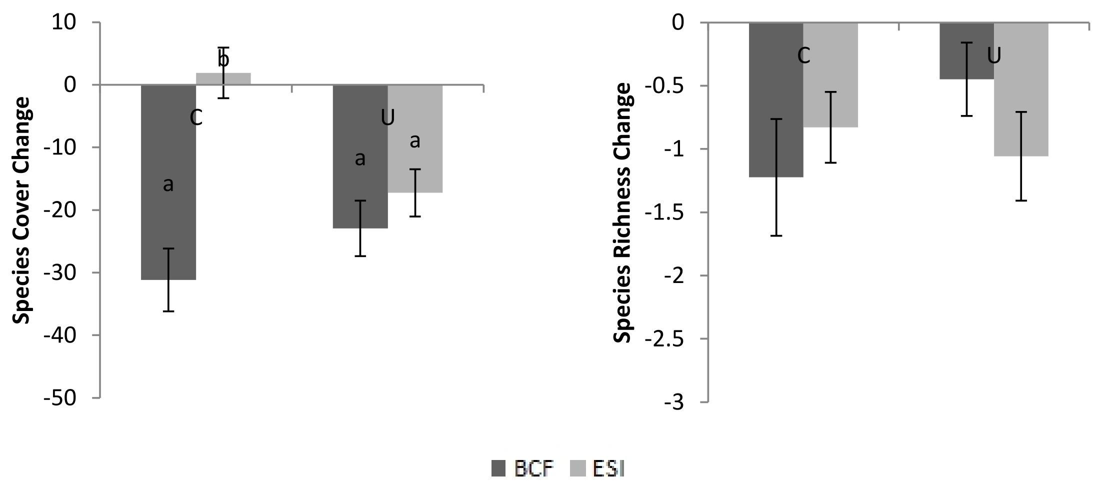

Figure 3.5: Average net change in sod plant species richness and percentage cover of restored quadrats at the T3 survey in both grass treatments for BCF and ESI. C – Cleared, U – Un-cleared.

Using burnt and un-burnt sods for transplanting

During transplanting, BCF cover for burnt sods was significantly lower than un-burnt sods, 97.1% compared to 105.5% (F-value: 7.5, $p < 0.01$). At ESI burnt sods also had less cover compared to un-burnt sods, 84.0% compared to 90.2%, but the difference was not statistically significant (F-value: 2.9, $p = 0.09$).

For plant richness BCF showed similar averages for species numbers in both burnt and un-burnt sods (F-value: 0.1, $p = 0.74$). Average plant numbers for ESI were higher in burnt sods compared to unburnt sods, 8.5 versus 7.3 (F-value: 9.3, $p < 0.005$).

BCF losses at T2 were not significantly different between treatments (Cover – F-value: 2.8, $p = 0.10$, Richness – F-value: 0.2, $p = 0.66$). ESI showed a higher initial loss for T1 burnt sods in percentage cover and species richness by survey T2. Loss in percentage cover was calculated as -36.5 for burnt and -22.3 for un-burnt (F-value: 8.8, $p < 0.005$). Loss in plant richness came to -2.8 for burnt sods and -1.4 for un-burnt sods (F-value: 12.5, $p < 0.001$).

By the final survey there was still no significant difference in species richness between fire treatments for either study area. Originally un-burnt sods recorded a net loss of -0.7 versus a net loss of -1.2 in originally burnt sods (F-value: 0.76, $p = 0.39$).

Effect of fire at survey T3

Survey T3 showed no differences between sod averages of both percentage cover and richness in burnt and un-burnt restored areas (Table 3.6).

Table 3.6: GLM significance values for survey 3 plant percentage cover and species richness in BCF and ESI comparing un-burnt and burnt restoration plots.

Treatment	d.f.	F-value	P
Percentage Cover			
BCF	1	2.1	p = 0.16
ESI	1	0.9	p = 0.35
Richness			
BCF	1	0.1	p = 0.77
ESI	1	1.8	p = 0.18

Discussion:

Transplanting established plants creates possible sources of propagules for natural species within secondary habitat. This could facilitate the restoration process in vegetation where resprouters are dominant and thus do not easily re-colonise recovering land (Chapter 2, Norman *et al.* 2006, Zaloumis and Bond 2010). In this study we managed to successfully transplant the majority of 364 sods from natural grassland into secondary grassland and measure their success in re-establishing in a recovering environment.

This study was able to increase the species diversity of each restoration plot on a small scale despite the losses of some species and cover from transplanted sods. These losses were proportionally small compared to the numbers recorded on the sods introduced and resulting in a number of common grassland species being successfully re-introduced into the secondary grassland.

There was no clear effect of grass competition. The only case where there was an effect was from the coastal grasslands. Here cleared quadrats lost less cover than the un-cleared quadrats. This is what you could expect for sods surrounded by competitive monospecific grass stands without a cleared boundary. However, plant richness between treatments did not show the same pattern. It is possible that the apparent absence of a grass competition effect was because this was only a short study. The effect of grass competition may yet emerge in the longer term. It is also likely to depend on the management practices of the area. Monospecific grass stands can both create competition for below ground resources and for sunlight in the secondary areas (Chapter 3). If these secondary grass swards were allowed to shade out the sods in the future, we should expect to see a greater loss the of shade intolerant species in the sods (Uys *et al.* 2004). If competition for resources was more severe then we would also expect the more competitive grasses to eventually replace the re-introduced natural species over time (Fynn *et al.* 2010).

Species richness measurements for plant growth forms can help to determine which functional groups are more successful at establishing after a transplant. Graminoid elements, grasses and sedges, did far better in the transplanting compared to forbs and geoxylic suffrutices which lost the highest proportion of their percentage cover and richness. This is because the below ground habit of forbs and geoxylic suffrutices make them far more difficult to practically and safely remove from the ground. Forbs come in all sizes (Uys 2006) but some are very large (Zaloumis & Bond 2011) and you cannot avoid chopping off bits of their underground storage organs (USO) thereby possibly compromising transplanting success. USOs may also increase the length of dormancy a forb undergoes before resprouting after a transplant. A year may not be enough to record a forb recovery from this treatment. Also, it was usually a few of the hardier more common forbs that were regularly recorded in the final survey. It would be beneficial to identify why this is so. The USOs of some forbs are very hardy and difficult to damage while others are very easy to break or cut (pers. observation). Geoxylic suffrutices are even more difficult to transplant as they form clones with very large subterranean structures (White 1979). When removing these you are basically removing a small segment of their structure and the survival of this element is more likely dependant on how much of the structure you were able to take with the sod. As forbs and geoxylic suffrutices are important components of grasslands making up the majority of the grasslands biodiversity it is important to ensure minimum loss of these elements during the transplanting process. Our results in this study indicate that this is possible for forbs as three quarters of their species survived the process. Geoxylic suffrutices suffered the most, almost completely disappearing from the majority of sods and therefore failed to successfully establish once transplanted.

Change in plant cover within the original sod area was far more variable between and within the study areas than species richness. This potentially makes richness a better measure of success for sod establishment in this study. Cover may also be far more reliant on post transplanting rain events. It is possible that this is the reason for the smaller initial loss of plant cover for BKNR, which saw the best post-transplanting rain. It did however initially lose similar amounts of plant species to the other two sites. Unfortunately there is not a final survey of BKNR to provide a proper comparison. However we can predict that there would also have been a recovery seen in plant species and potentially an increase in percentage cover. Hopefully another survey after a few years will be able to include all three sites for a complete comparison.

Fire did not seem to have an effect on the outcome of this study. Harvested sods that had been burnt did not show any lower diversity values than the sods harvested from un-burnt areas. This may be because grassland forbs, supported by their USO's, are likely to be able to respond quickly to multiple forms of defoliation disturbances during a single growing season. Fire also had no measurable influence on recorded cover and richness data from restored plots that had been burnt before the final survey was completed.

Practicalities of mixed species sod transplants

This study indicated several serious limitations that one would face using sod transplants for restoration on a larger scale. It also suggests some potential opportunities for further investigation in grassland restoration.

Graminoids were by far the easiest plant element to transplant and re-establish. Unlike forbs, grasses were supported by a thick layer of roots below ground which were much easier to excavate without having to worry about killing them. Although forb transplants were partly successful, it may be worth finding a more practical way of re-introducing larger numbers of them as adults or established seedlings to increase the proportion that are successfully transplanted.

The timing of this study was limited by the constraints of a masters thesis. The transplanting occurred in the middle of the growing season. Discussion amongst some stake holders involved in this project along with field observations suggest that the ideal time to do the transplants would have been during the dormant season especially towards the end of winter. During this time plants would be inactive and daytime temperatures during the transplanting would not be those experienced in summer thus reducing the transpiration stress that a forb would experience. The only limitations with this would be the inability to identify plant species that are not flowering or even present above ground at this time. One could try marking out sods to sample during the growing season to use in restoration plots during the dormant season. Should one want to do another study during the summer period, it may be worth trimming the above ground biomass of forb species to reduce the transpiration stress as well as using plants that had not been burnt.

Criteria for measuring transplant success

The aim of this study was to determine if sods can survive a transplant and re-establish in recovering areas. They did. The best metric of sod re-establishment success was species richness. Cover was far more variable and although some plants, especially forbs, had considerable cover to start with, they were often then reduced to just a single resprouting stem. Monitoring in the future should still include both cover and richness because they can both be used to determine how sod plant species, after establishing, have started to increase their distribution by dispersing successfully within the recovering area.

In this study the transplanting of natural grassland sods was marginally successful. Unfortunately this is not the most practical method to use in grassland restoration efforts. There are few intact natural grassland available that can be used for such a project (pers. observation), especially on a larger scale, unless one was to transplant from an entire area that was being developed. Remaining areas of natural grasslands can also be difficult to access (one of the reasons why they are still intact). Sods are impractical to move over larger distances because of the considerable range of forb USO sizes. If forbs are delicate to transplant, one would end up losing several of the larger USO species. Also, on a larger scale, a project like this could really damage natural grasslands that are sourced for the sod harvesting and one would have to consider the pros and cons of such an endeavour.

We need to develop a better understanding of grass and forb reproduction, both sexual and vegetative. At the same time we can build on the work of this project as we know that forbs and grasses can be successfully transplanted. One suggestion would be to establish a project where plant propagules, both plant specimens and seeds, are collected for an area intended for restoration. From these an attempt could be made to create artificial sods with the local soil, or at least to grow a sufficient number of the plants individually for large scale re-introduction into recovering areas.

Conclusion:

Mix species sod transplanting as a possible method for re-introducing natural resprouting species has some potential. We established seemingly viable populations and increased the diversity of natural grassland species in the study areas. However, the longevity of this success can only be measured through continual monitoring. We now have an established project that we can build on and continue to monitor in the future.

We found that transplanting worked, but the method of using sods, by itself, may just not be practical for large scale projects. It is important to build on such a project to help establish alternative and more practical methods for grassland restoration. This could include the collecting of seed and other plant propagules and growing these plants in specialised nurseries specifically managed for restoration projects.

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Chapter 4: Grass competition as a constraint for grassland forb seedling establishment

Abstract:

A great diversity of forb and grass species co-exist within the primary grasslands of southern Africa, yet little is known of their interaction when competing for similar resources. The conditions that determine forb seedling establishment in the natural environment are still unknown. In the context of grassland restoration the implications of grass-forb competition on discouraging forb seedling establishment becomes far more relevant from a management context.

The aim of this study was to explore the effect of grass competition on forb establishment in a glasshouse pot experiment. Propagules of five forb species were planted into pots containing tillers of five different grass species with different levels of competitiveness. Forb leaf and root traits were then measured and compared between competition treatments. Forb growth was found to be limited by grass competition, with some grass species having more of an affect than others. Forb species that were able to respond to the high levels of grass competition by changing their growth habit coped better. Forb seedlings require a spatial and temporal 'gap' to successfully establish under medium and high levels of grass competition. The dimensions of a gap, and how to simulate their formation, have important implications for grassland management and the restoration of secondary grasslands. A field project simulating potential gap conditions under natural and secondary grassland conditions would be the next step in furthering our understanding of grassland seedling gaps.

Introduction:

Whereas the effects of grass on tree growth and recruitment in savannas has been a topic of a number of studies (Scholes & Archer 1997, Bond 2008, Tedder *et al.* 2012), very little is known about forbs and how they establish and compete in a grass sward. Forbs, a largely ignored and unexplored grassy biome component, make up a significant proportion of grassland biodiversity in South Africa (Uys *et al.* 2004, Zaloumis & Bond 2011) and in similar tropical and sub-tropical montane grasslands elsewhere (Overbeck *et al.* 2007, Overbeck & Pfadenhauer 2007, Bond & Parr 2010). Grassland ecology and forb-grass interactions have been explored to some extent in the prairies of North America and the Temperate Grasslands of eastern Australia (Collins 1992, Lunt 1994 and 1995, Leach & Givnish 1996, Collins *et al.* 1998, Olff & Ritchie 1998, Lunt & Morgan 1999, Morgan *et al.* 1999a and 1999b, Knapp *et al.* 2004, Dickson & Busby 2008) including an investigation of the effect of grass competition on annual forbs in North America (Gillespie & Allen 2004). In South America and South Africa, the study of grassland forbs has been neglected until very recently, especially in comparison with work on the grasses (Uys *et al.* 2004, Zaloumis & Bond 2010, Overbeck *et al.* 2007, Overbeck & Pfadenhauer 2007). Two specific studies have focused on the direct interaction of native grass and forbs in Kansas, North America (Dwyer 1958) and how competitive grasses can have a negative effect on forb establishment in prairie restoration (Dickson & Busby 2008).

The growth form of grasses makes them very competitive. Grasses are effective competitors for light and are able to respond quickly to resource pulses (Bond 2008). Grasses have shallow, but dense and fibrous root systems and are also intense competitors for nutrients and water within their rooting zone (Harris 1977, Scholes & Archer 1997, Cramer *et al.* 2010). Trees, being large and long lived, have a growth form with the ability to access resources above and below ground that grasses cannot (Smith & Shackleton 1988, Bond 2008). How do forbs fare in this competitive milieu?

Like trees in a savanna, resprouting grassland forbs are a plant growth form that co-exist with grasses. They make up a considerable proportion of the diversity of grasslands and contribute to its biomass and cover. In the case of southern Africa and South America, they are persistent growth forms that could survive for decades or maybe even centuries (Uys *et al.* 2004, Overbeck *et al.* 2007, Overbeck & Pfadenhauer 2007, Bond & Parr 2010). Along with the South African grass species they even have adaptations to frequent defoliation events like fire and herbivory because of characteristic woody, tuberous or bulbous storage organs below the ground (Underground Storage Organs – USOs). Unlike trees, however, forbs have negligible above-ground woody biomass. They are short and often over-topped by surrounding grass species. Established forbs are able to respond to fire very quickly and can flower within weeks of a burn, with many species flowering before the dominant grass species begin to dominate again (Chapter 2, Uys *et al.* 2004, Zaloumis & Bond 2011). Forbs and grasses also appear to occupy the same surface soil layers (Chapter 2).

Thus grasses and forbs have extensive niche overlap on many habitat axes. Both compete for sunlight and exist mostly in the same height zone, although some dominant grasses can grow much taller than most forbs. However, many native grass species are short and tend to keep the grass layer heterogonous in height (Fynn *et al.* 2010, Fynn *et al.* 2011, Observation). Height is also controlled by natural disturbances such as grazing or fire. Both grass and forbs recover quickly after fire (Zaloumis & Bond 2011). Forbs and grasses mostly share the same soil depth. Grass roots in the South African grasslands occur between 0 – 20 cm, being extremely dense for the top 7 – 13 cm (especially in the high veld) and then thinning out at 15 - 20cm with coastal grasslands having significant grass rooting as deep as 25cm (Chapter 2, pers. obs.). Many forbs also exist in this same soil plane with some forbs that can produce large storage organs being able to occur as deep as 40 cm or more.

Dwyers (1958) work in North America showed that forbs could occupy two soil levels under the ground. Dicot forbs, which he refers to having ‘rhizomatous’ roots, compete in the same soil level as grass roots and can actually decrease grass biomass. Dicot forbs that he refers to as having taproots are able to go deeper than the grass layer and therefore don’t compete with grasses, with minor impacts on grassy biomass. This suggests that 1) forbs could be important competitors in a grassland system and 2) there are two tiers of soil that forb USOs can occupy.

However, all forb seedlings, like those of trees, have to compete with grass. Reproductive opportunities are likely to be very rare and forb seedlings are very rarely seen in South Africa's natural veld, especially the more productive grassland vegetation types with high rooting biomass. As for trees, forbs will probably need a gap of some sort to establish and persist (Bond 2008, Cramer *et al.* 2010). In trees this gap is often related to disturbance regime such as a drought, grazing or change in fire frequency. Is there a forb gap and what are its spatial and temporal dimensions? When there is a gap, how successful are seeds at germinating and establishing? How fast do they grow especially when not limited by competition with grasses?

The aim of this study is to identify how forbs are affected by grass species competition. We expect that forbs will struggle to establish when under fierce competitive pressure. Grasses are thought to vary in their competitiveness. Do some grass species have a greater effect on forb seedling growth than others? The species effects are relevant because often the more competitive grasses make up mono-specific or species poor grass stands in recovering areas or are used in restoration projects. This information can then be used to consider what mix of grass species to utilise for restoration work (Chapter 2). If stands with more diverse grass species facilitate forb establishment, then introducing more grass species into secondary grasslands could help restoration efforts.

We asked the following questions:

1. How well do grassland forbs establish when facing grass competition?
2. Do different grass species affect forb establishment and growth differently?
3. Does mixing grass species help facilitate forb establishment and growth better when compared to a single species of grass?

As for savannas, it is possible that the interactions between forbs and grasses are dynamic and affected by many factors. All these questions have important implications for the management and restoration of grassland systems.

Methods:

Forb and grass species selection

This study was carried out at the UKZN Life Sciences Campus in Pietermaritzburg, within the Grasslands Science department greenhouse. Naturally occurring grassland forb seeds and seedlings were difficult to find and even more difficult to successfully germinate. I acquired the forb species propagules from the Silverglen Nursery, situated South of Durban. I managed to obtain multiple seedlings, cuttings and seeds for five widespread grassland forb species, four of which were monocot, *Aloe maculata, Aristea woodii, Eucomis autumnalis and Watsonia densiflora* and one a dicot, *Senecio speciosa*. Individual grass tussocks were collected from a local grassland patch close to the greenhouse. These were used to create hundreds of individual tillers. Five widespread, co-occurring native grassland C4 grass species were selected.

Eragrostis curvula is a tall strongly tufted species which invades grassland fertilised with nitrogen (Fynn *et al.* 2005). It is a grass that is often used in rehabilitation projects and also dominates the cover of many recovering excised afforested stands and old fields (Chapters 2). It is known to be a highly competitive species (Fynn *et al.* 2010).

Themeda triandra is a relatively short, narrow leaved species in the grasslands dominating well drained infertile soil (Fynn *et al.* 2005). It is famous for its stands of red in winter indicating healthy grazing veld. It is also often used in rehabilitation projects. It is a highly competitive grass in the right conditions (Fynn *et al.* 2010). It is not often found in recovering excised afforested and/or old field grasslands.

Aristida junciformis is a medium sized species with very narrow needle like leaves, common on infertile soils and when veld is under a high level of selective grazing pressure (Fynn *et al.* 2005). It is not a very competitive species (Fynn *et al.* 2010). *Aristida* species can be found in recovering excised afforested and old field grasslands.

Heteropogon contortus and Tristachya leucothrix are both relatively short species, with the latter species not being greatly competitive (Tainton *et al.* 1990, Fynn *et al.* 2010). They are not found in recovering excised afforested and old field grasslands. These two species were used with *Themeda triandra* to make the mixed grass species treatment.

Pot preparation

140 pots with a diameter of 20cm and depth of 27cm were filled to the rim with potting soil. The pots were divided into groups of seven to replicate each treatment.

The forb seedling were then planted individually in the middle of each pot after which four grass tillers were added to each grass species treatment pot. They were planted on the edge of the pot equidistant apart. For mixed grass treatments, two of *Themeda triandra* tillers were used with one each of *Heteropogon contortus and Tristachya leucothrix.* Control pots were not planted with grass tillers. The planting of tillers at the same time as the forbs delayed the competitive effect of the grass species on the forb seedlings until the grass tillers had effectively established in the pots.

Each forb species was planted into a control (no grass) treatment. Each forb species then had two or more grass species treatments. The treatments for each forb species are listed in table 4.1. In the experiment, *Aloe maculata* (Aloaceae)*, Eucomis autumnalis* (Hyacinthaceae) and *Senecio speciosa* (Asteraceae) each had 28 pots with four grass treatments, *Watsonia densiflora* (Iridaceae) 35 pots with five grass treatments and *Aristea woodii* (Iridaceae) had 21 pots with three grass treatments (Table 4.1). Forbs were allocated grass treatments based on the number of available forb seedlings and to ensure that each grass treatment had at least three forb species comparisons.

Table 4.1: Number of pots per grass treatment (rows) per species (columns).

Species Grass Treatments	***Aloe maculata***	***Aristea woodii***	***Eucomis autumnalis***	***Watsonia densiflora***	***Senecio speciosa***
Control	7	7	7	7	7
Eragrostis curvula	7		7	7	7
Themeda triandra	7	7		7	7
Aristida junciformis		7	7	7	
Mixed	7		7	7	7
Total Pots	**28**	**21**	**28**	**35**	**28**
Alive at end	**25**	**20**	**28**	**10**	**0**

Measurement of response variables

When the forbs were being planted several measurements were taken of the seedlings and cuttings. Leaf length, root length and basal width were measured along with the number of leaves per plant individual. All of the measurements done above ground were then repeated after 1.2 months, 4months and 11 months. Grass height was recorded at 11 months.

After their final above ground measurements at 11 months the forbs were harvested and maximum root length was measured. The above ground and below ground parts of the plant were then collected separately, weighed individually and placed in a drying oven set at 70˚C for 48 hours. *Aloe maculata* required another 48 hours in the oven to lose its moisture content completely. They were weighed again when dried. The same was also done for grass leaf biomass, where total wet weight of above-ground grass biomass was weighed then a sub-sample of the grass was taken and weighed again to be placed into the drying oven. The dry weight of the grass portion was used to calculate the total grass dry weight.

Although I had numerous *Watsonia densiflora* seedlings they did not do well in the transplanting into pots and, except for the control, none of the treatments survived to the 11 month stage. A few made it to the 4^{th} month measurement. *Senecio speciosa* was planted as seed but never germinated.

Data analysis

General Linear Model statistical analysis was performed using the STATISTICA 11 software package (StatSoft;Tulsa, OK, USA). Post-hoc Tukey tests were then used to distinguish significant difference within a GLM among several treatments.

Forb biomass

The average leaf dry weight and root dry weight for each forb species was compared among each grass species treatment. Forb dry root weight was compared to pot grass blade biomass to determine the effect of grass competition on forb USO development.

Plant leaf number and basal width

Forb dry leaf weight was compared to the number of leaves on the forb in a regression analysis. Forb dry root weight was compared to the basal width of the forb in a similar analysis. This helped determine if number of leaves and basal width can be used as a biomass proxy for the rest of the analysis.

Forb biomass growth over 11 months

The average number of leaves and their average basal width was compared for each forb species among each grass treatment for the sampled 11 months. The use of these two proxies was to determine how each forb species responded to grass competition over the year.

Forb leaf and root length

Individual forb leaf and root lengths were compared to pot grass leaf biomass for each grass treatment to look at any trait responses to grass competition.

Forb root wet weight vs. dry weight

The root weight loss for each forb species was compared among grass treatments to determine how the below-ground (USO) parts of each forb species respond to grass competition.

Results:

Forb biomass

Leaf dry weight of each forb species was far greater in the control (no grass) treatments when compared to all other grass species treatments (Figure 4.1). *Aloe maculata* (*Aloe*), *Eucomis autumnalis* (*Eucomis*) and *Aristea woodii* (*Aristea*) showed a significant difference between the control dry leaf biomass and that of the other grass treatments (Table 4.2: For all three species, p < 0.0001). *Aloe*showed the largest proportional difference between the control and the other grass species treatments and there was no difference between the dry leaf biomass of its remaining grass species treatments (Figure 4.1a). *Eucomis* grew significantly more leaf biomass in the mixed grass (Mixed) treatment compared to *A. junciformis* (*Aristida*) and *E.* curvula (*Eragrostis*) grass treatments (Figure 4.1b). The *Aristida* grass treatment did produce more dry leaf biomass on average than *Eragrostis Aristea* produced more dry leaf biomass in the *Aristida* treatment compared to the *T. triandra* (*Themeda*) treatment but the differences were not significant (Figure 4.1c).

Table 4.2: GLM of forb species comparing leaf and root dry weight averages for each grassland treatment.

Species	**d.f.**	**F-value**	**P**
Leaf dry weight (g) (n = 7 per treatment)			
Aloe maculata	3	57.2	p < 0.0001
Eucomis autumnalis	3	82.7	p < 0.0001
Aristea woodii	2	99.6	p < 0.0001
Root dry weight (g) (n = 7 per treatment)			
Aloe maculata	3	40.0	p < 0.0001
Eucomis autumnalis	3	108.1	p < 0.0001
Aristea woodii	2	15.3	p < 0.0005

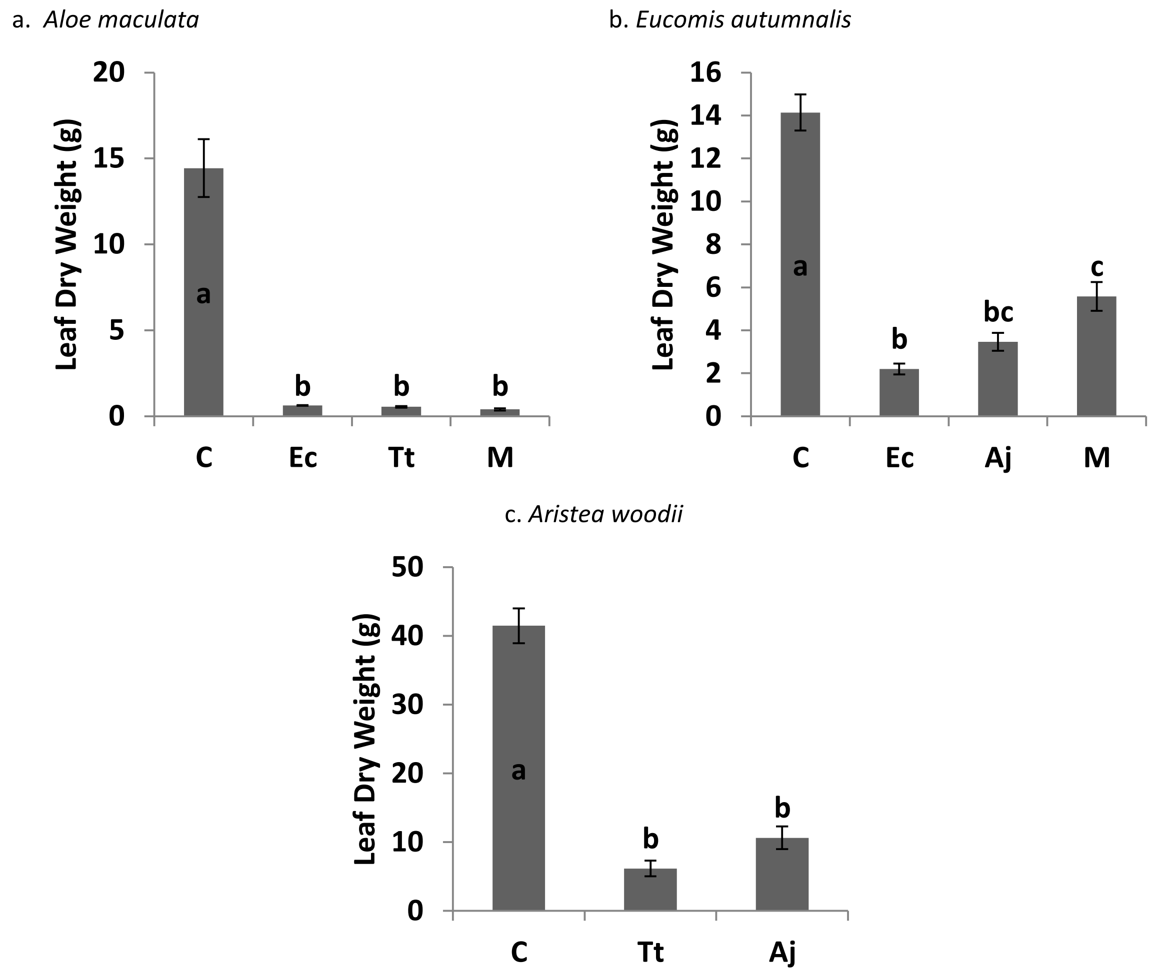

Figure 4.1: Species leaf dry weight for each grass treatment. a. *Aloe maculata*, b. *Eucomis autumnalis* and c. *Aristea woodii*. C – Control, Ec – *E. curvula*, Tt – *T. triandra*, Aj – *A. junciformis*, M – Mixed.

The response of root dry weight was similar to leaf weight patterns with the control treatments for all forb species having dry root biomass that was significantly higher than all the other grass treatments (Figure 4.2, Table 4.2, *Aloe*: $p < 0.0001$, *Eucomis*: $p < 0.0001$, *Aristea*: $p < 0.0005$). There was no difference for *Aloe* dry root biomass among the three remaining grass treatments (Figure 4.2a and b). The *Aristida* grass treatment produced a higher dry root biomass for *Eucomis* when compared with the other two grass treatments although the differences were not significant (Figure2c and d). *Aristea* showed no difference between the two grass treatments (Figure 4.2e and f). Figure 4.2 (b, d and f) shows the relationship between above-ground grass biomass for the different grass treatments and dry weight of forb roots. The presence of any grass strongly suppressed root dry weight relative to the control pots.

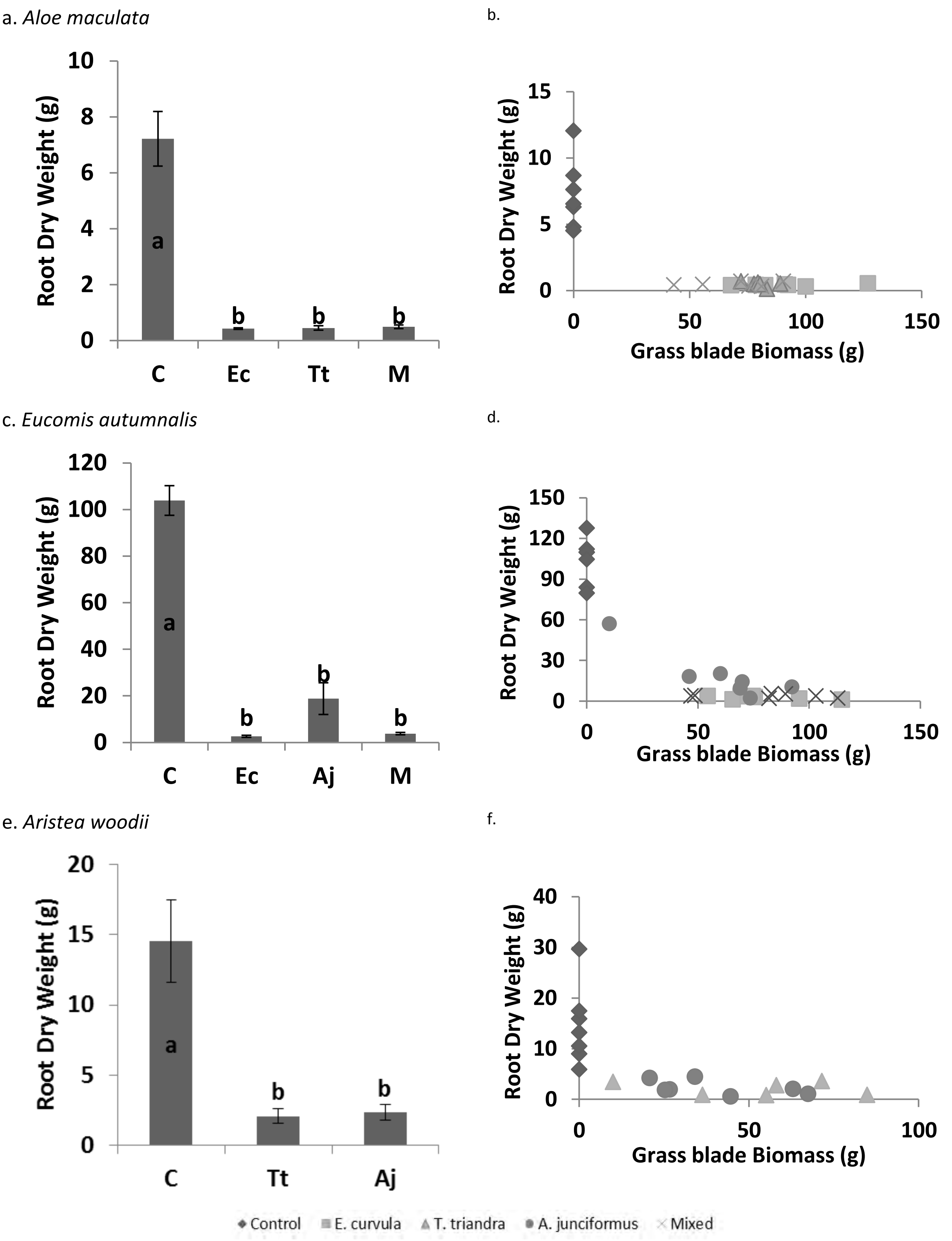

Figure 4.2: A, c and e are species root dry weight for each grass treatment. a and b. *Aloe maculata*, c and d. *Eucomis autumnalis* and e and f. *Aristea woodii*. B, d and f show root dry weight data for individual samples against grass dry leaf biomass. C – Control, Ec – *E. curvula*, Tt – *T. triandra*, Aj – *A. junciformis*, M – Mixed. B, d and f compare grass dry biomass with species root dry weight.

Plant leaf number and basal width

The number of leaves at harvest was positively correlated with leaf dry mass for all forb species (Figure 4.3). Thus leaf number can be used as a proxy to compare growth rate between treatments

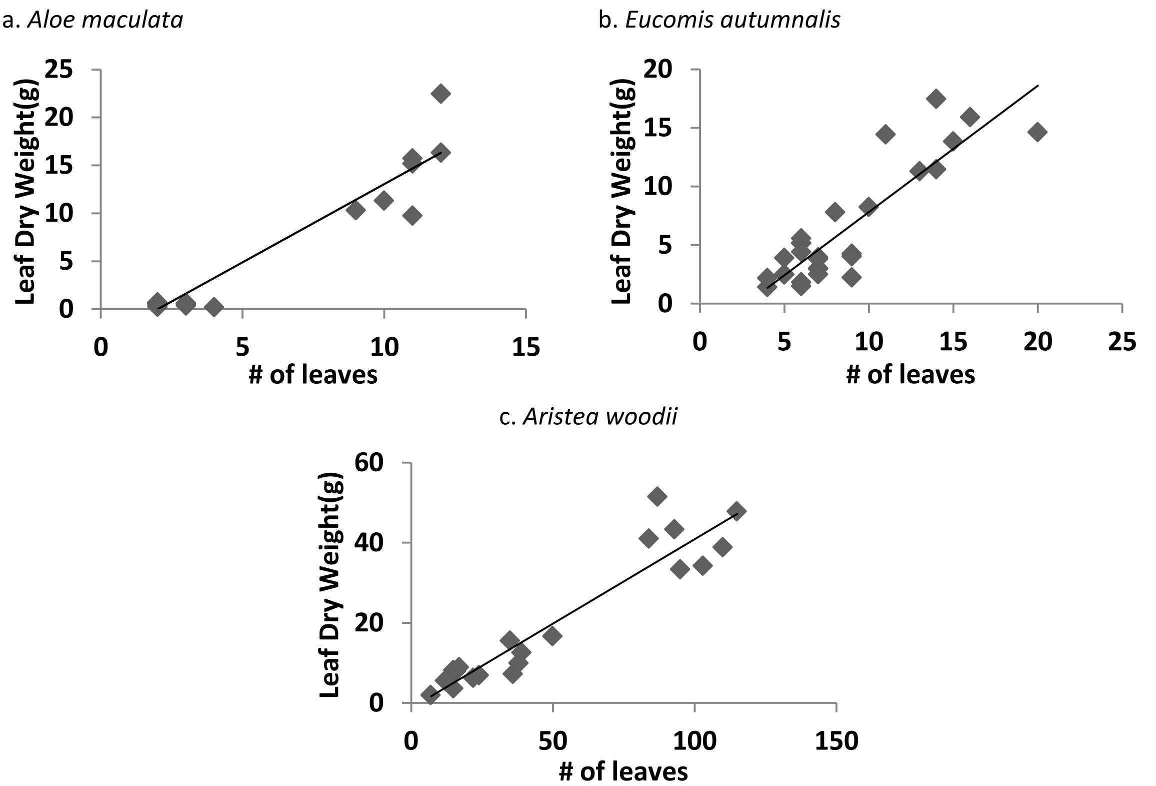

Figure 4.3: Species relationship between leaf # and leaf dry weight. a. *Aloe maculata*: R^2=0.92, p < 0.0001, b. *Eucomis autumnalis*: R^2=0.78, p < 0.0001 and c. *Aristea woodii*: R^2=0.89, p < 0.0001.

The basal width was positively correlated with root dry weight (Figure 4.4). The relationship was weaker for *Aristea* (Figure 4.4c) but still significant. Basal width is thus usable as a proxy for root dry weight.

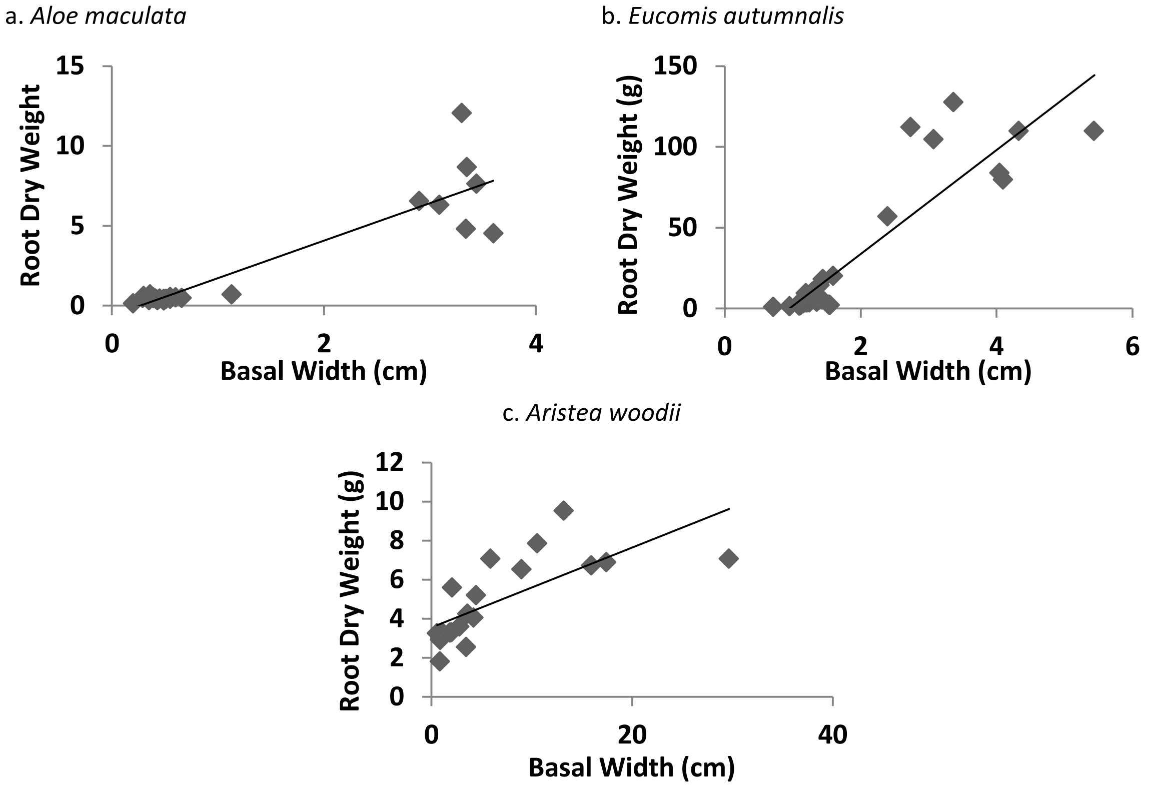

Figure 4.4: Species relationship between plant basal width and USO dry weight. a. *Aloe maculata*: R^2=0.83, p < 0.0001, b. *Eucomis autumnalis*: R^2=0.81, p < 0.0001 and c. *Aristea woodii*: R^2=0.67, p < 0.0001*. * - regression analysis excludes outlier point

Forb biomass growth over 11 months

The leaf number for all forb species increased over the 11 months for the control grass treatment (Figure 4.5). *Aloe* control leaves more than doubled in the period between 4 and 11 months while not significantly changing in any of the grass species treatments (Figure 4.5a). In *Themeda* and Mixed the number of leaves even dropped slightly between 4 and 11 months. *Eucomis*'s number of leaves in the control treatment increased significantly between 1.2 and 4 months and to a lesser extent between 4 and 11 months (Figure 4.5b). All other grass treatments for *Eucomis* showed increasing number of leaves with time, but not nearly as large an increase as the control and not significantly so. *Aristea*'s leaf numbers increased strongly for the control treatment between 1.2 and 4 months and doubled between 4 and 11 months (Figure 4.5c). The grass treatments also showed increasing number of leaves with time, although far less than the control, with the *Aristida* treatments showing significantly more in the final harvest than the *Themeda* treatment.

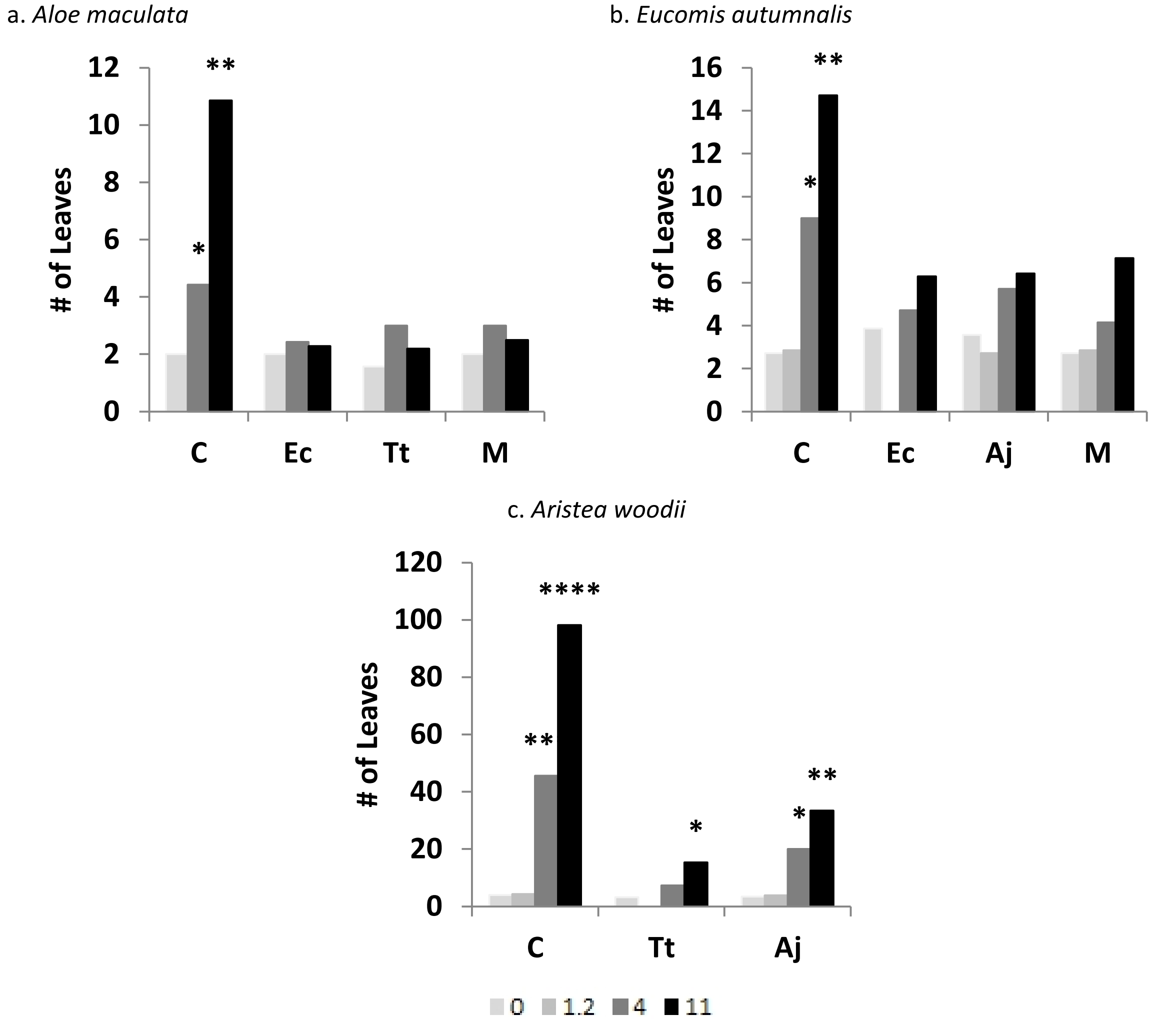

Figure 4.5: Forb species above ground biomass growth at 0, 1.2, 4 and 11 months for different grass treatments. a. *Aloe maculata* (F-value: 82.8, p < 0.0001), b. *Eucomis autumnalis* (F-value: 16.2, p < 0.0001) and c. *Aristea woodii* (F-value: 69.3, p < 0.0001). C – Control, Ec – *E. curvula*, Tt – *T. triandra*, Aj – *A. junciformis*, M – Mixed.
* Indicates significant difference from starting measurement. Each additional * indicates a significant difference from one another.

Basal width increased over time in all control treatments (Figure 4.6). *Aloe* basal width more than doubled in the control treatment between 4 and 11 months (Figure 4.6a). Basal width increased initially in the grass treatments but showed negligible change between 4 and 11 months. There were no significant differences in width among the remaining grass treatments. *Eucomis*'s basal width grew steadily in the control treatment over 11 months

(Figure 4.6b). The remaining grass treatments showed little growth in width with the *Eragrostis* treatment showing the least observable difference between start and end. *Aristida* and Mixed grass treatments showed an observable but small increase over the same growth period.

Aristea's basal width grew substantially in the first 4 months for all treatments, with the control showing the largest increase (Figure 4.6c). Growth in width in the non-control grass treatments did not change much from 4 to 11 months. The *Aristida grass* treatment was significantly higher compared to *Themeda*.

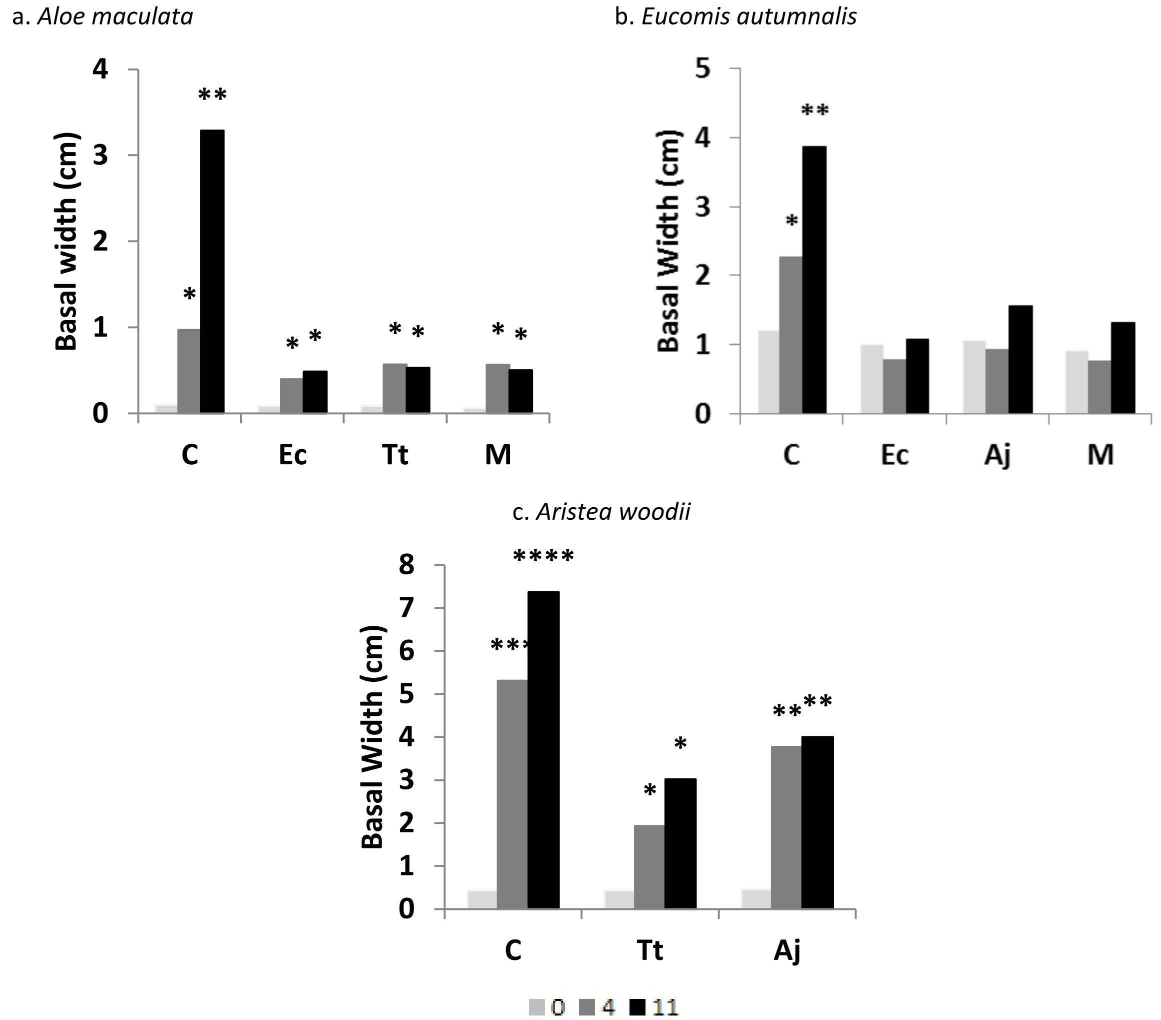

Figure 4.6: Species below ground biomass growth, as indexed by basal width of each forb, at 0, 4 and 11 months for different grass treatments. a. *Aloe maculata*, b. *Eucomis autumnalis* and c. *Aristea woodii*. C – Control, Ec – *E. curvula*, Tt – *T. triandra*, Aj – *A. junciformis*, M – Mixed.
* Indicates significant difference from '0'. Each additional * indicates a significant difference from one another.

Forb leaf and root length

The difference between starting and ending leaf length in response to grass biomass differed for each forb species (Figure 4.7). *Aloe* leaf length difference was significantly higher for control pots than for any of the other grass treatment pots (Figure 4.7a). *Eucomis* and *Aristea* leaf length for the grass treatment pots was equal to that of the control pots (Figure 4.7b and c), while many *Eucomis* plants had longer leaf length within the other grass treatments than compared to the control pots.

Root length response also differed among species. *Aloe* and *Eucomis* root length differences showed the same patterns as their leaf length responses to grass treatments. *Eucomis* roots in grass treatment pots were, on average, half the length of roots in the control treatment.

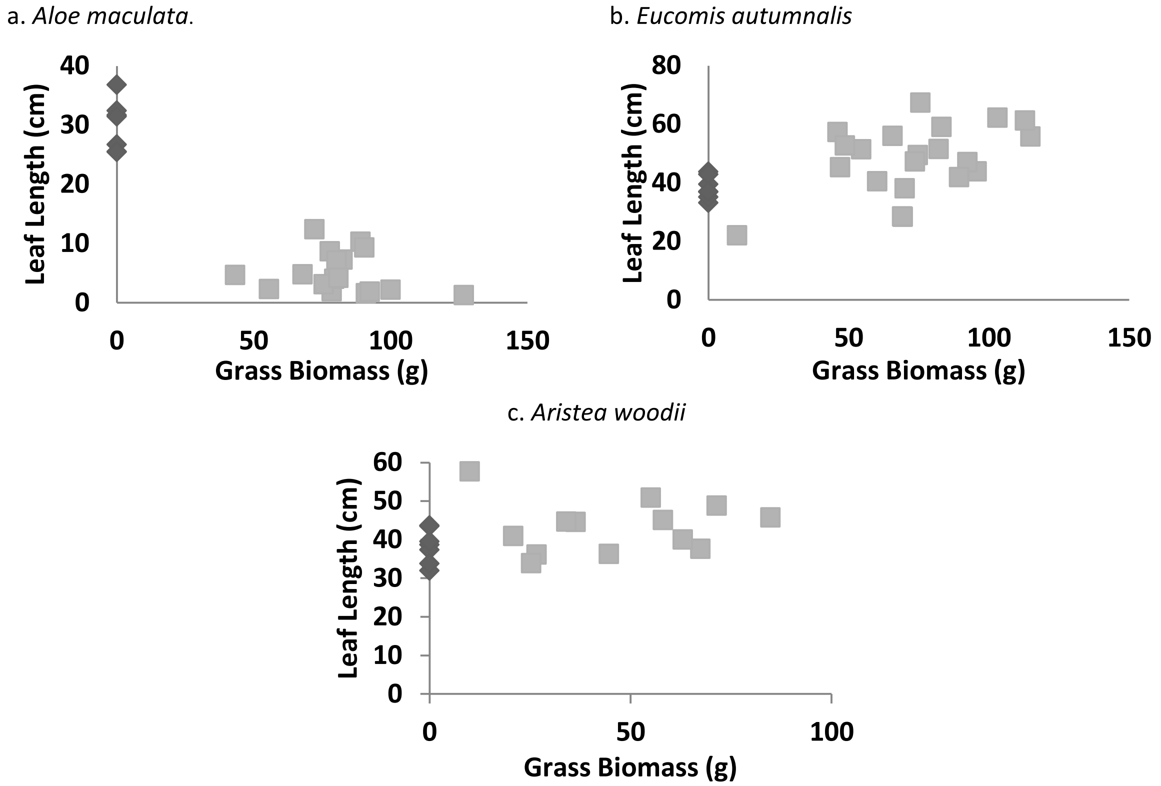

Figure 4.7: Leaf length for each forb species in relation to grass leaf biomass per pot. a. *Aloe maculata*, b. *Eucomis autumnalis* and c. *Aristea woodii*. Diamonds = Control and Squares = Grass treatments.

Forb root wet vs. dry weight

All forb species lost a large amount of root weight in the control pots after harvesting (Table 4.3). Grass species treatments had forbs that weighed less and lost far less weight on average compared to grass control treatments. *Aloe* roots in control pots lost 79 times more weight in grams than the grass treatments. *Eucomis* control roots lost 9.5 times more weight than the *E. curvula* and mixed grass treatments and only four times more weight than the *T triandra* grass treatment. *Aristea* control pot roots lost four times more weight than the remaining grass treatments. A post-hoc Tukey test showed forb species controls treatment average losses to be significantly higher than the grass treatments. The *T triandra* grass treatment for *Eucomis* was significantly higher than the other grass treatments for *Eucomis*, however it was still significantly lower than the control.

Table 4.3: Comparison of wet and dry root weights for each forb species showing the average calculated weight loss and percentage loss among grass treatments. * indicates Tukey post-hoc test significant difference between the control and grass treatments. ** is significant difference from both the control and the other grass treatments.

	Grass treatments				
Forb species	Control	*E. curvula*	*T. triandra*	*A. junciformis*	Mixed
A. maculata	47.2g*/86%*	0.6g/57%	0.6g/55%		0.7g/57%
E. autumnalis	189.1g*/65%*	19.3g/88%		50.32g**/77%	21.5g/84%
A. woodii	35.2g*/71%		8.1/76%	8.70/79%	

Discussion:

This study brings us one step closer to understanding forb-grass dynamics. Investigating how these two growth forms interact in a South African grassland context can help us better determine how forbs establish themselves in such a competitive environment. These findings have important implications for future secondary grassland restoration projects.

This project was completed in a controlled environment giving forb seedlings a ‘head start’ to establish themselves in the pots before the grass competition became an effective limiting factor. Despite this we were able to see some considerable competitive effects. Left alone, the three forbs species had limited ability to compete with the grass stands once the grass species had become established and over shadowed the forbs. All three forbs had different responses to grass competition.

One of the forbs, *E. autumnalis,* managed to respond by manipulating its leaf length, increasing its length in a way which helped it cope better under the high level of grass competition. Another species, *A. woodii* was able to use the way it grows to its own benefit, building on a stable base habit to grow taller and stay at the same level as the grass. These two species, having being able to take advantage of what light they could, both managed to establish viable root structures quickly enough to potentially be able to resprout after a fire. Thus they would be able to take advantage of the reduced grass competition. In contrast *A. maculata* was unable to respond to the grass competition and remained the same size throughout the experiment. In a grassland context, had this project continued for another year, *A. maculata* would need a defoliation event, such as fire, for its continued survival. However it is unlikely that these plants would have survived a fire after being restricted so heavily by grass competition.

Although the effect of grass competition is evident for all grass treatments, we found some evidence that indicates forbs can respond to different level of grass species competitiveness. Grasses like *A. junciformis,* or even a mix of species, could be better for facilitating the forb species recovery.

The experiment was not designed to identify if shading or root competition plays more of a role in limiting forb seedling growth. We do know that grassland forbs are shade intolerant and can be lost from the system if shaded out for too long (Uys *et al.* 2004). In my study, many forb seedlings managed to establish some root stock. It is possible that below ground root competition with grasses would play a more important restricting role in the initial germination of seed and their establishment into seedlings rather than once forbs have matured.

Forbs in control plots were able to take advantage of the lack of grass competition by being able to partition more growth to their root structure while still having a substantial above ground biomass. Some controls even showed evidence of vegetative preproduction with off -shoots stemming from the original roots which produced stems. Not only could they take up more water, but they managed to accumulate far more carbon in the root stock. This process was hampered by the presence of grass species, although forb species were affected differently.

Above ground adult grassland forbs are not like full grown adult savanna trees, they are more similar to resprouting adult trees still below the fire trap. However, unlike trees, they do not need to escape. Both growth forms require seedlings to establish for continued survival. Unlike savanna trees, grassland forbs do not seem to put much effort into recruitment and appear to rely largely on persisting within the system utilising their ability to store carbon beneath the ground (Zaloumis & Bond 2011). Grasslands produce a thick mesh of grass roots under the soil and forbs have to survive frequent natural defoliation events. Poor forb recruitment and long term persistence could be a response to these environmental pressures as seedling recruitment becomes very difficult even under optimal conditions.

Thus restoration by simply throwing forb seeds into secondary grassland is not going to encourage successful forb species establishment. This project was also undertaken in favourable conditions, with potting soil, sufficient water and within a greenhouse to all simulate perfect growing conditions. How the results would change within a field experiment needs to be investigated. One factor would be the already well-established grassy layer within secondary grasslands. To provide regeneration opportunities for forbs a competition gap may be required to create the chance for seedlings to establish in the natural context. Other issues include the fact that the forbs used in this study were easily available at a nursery because they are good at establishing and the seed is relatively easy to find in the primary grassland. This is probably not the case for the majority of grassland forbs.

When recruitment does occur, how does it occur? It would seem that forb seedlings would need to wait for a recruitment gap. A gap may require a period of time or an episodic process that results in the reduction of the high density of grass roots in the soil layer and the above ground grass biomass. The forb seedling would then also require sufficient time to establish adequate resources below the ground to ensure that it may survive the next fire or grazing event. This is possibly why some grassland forb species are restricted to rocky outcrops as their preferred habitat (Pooley, 1998). These outcrops may allow for more of these 'gaps' to form as rocks can be barriers to fire spread (Arno & Gruell 1983, Geldenhuys 1994, Clarke 2002) and could reduce the competitive level of grass by limiting their root occupancy of soils.

Grass competition poses a serious limitation on restoration efforts in grassland systems. Further work needs to be done to determine how to create these gaps in a restoration context. To extrapolate from these results one could set up a project that includes field studies using a greater variety of species, including some dicot forbs. Different grass treatments could be used to estimate the potential size of regeneration gaps and if these sizes differ in secondary and primary grasslands. One would also need to test the effect of different soil properties on forb establishment between the two.

Conclusion:

This project has helped confirm that grass competition plays a major role in limiting forb establishment. Evidence suggests that forbs species can have different responses and adaptations for coping with grass shading and root competition. If competition is high even under these favourable glasshouse conditions, we can assume that it would be far more difficult for forb seedlings to establish in the field. How forbs establish in the field and identifying the conditions that are favourable to create a gap for them to do so still needs to be investigated. How this differs between secondary and primary grasslands would also be important. In a restoration context it would also help to determine which of either soil properties or grass competition can be more constraining for forbs. Further investigation is needed into the effect of different native grass species on forb establishment and whether multi-species grass stands are better for facilitating grassland forb propagation.

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Conclusion:

Grasslands have a history of being ignored in a conservation context and as a result have suffered high levels of transformation and degradation (Matsika 2007, Oudtshoorn *et al.* 2011). Recent studies have suggested that, far from being recent and resilient, our grasslands are possibly ancient and fragile systems (Meadows & Linder 1993, Bond & Parr 2010). The first objective of this study was to broaden our knowledge of South African mesic grassland ecology with a focus on identifying how secondary grasslands differ from primary grasslands. From a distance, natural and secondary grasslands can easily be confused. It was hoped that this thesis could improve our understanding of grassland ecology and to identify the differences between natural and disturbed grassland systems. The second objective was to apply some of this knowledge to grassland restoration. Grassland restoration is difficult and often not successful (Lubke *et al.* 1996, Prober & Thiele 2005). With a better knowledge of South African grassland ecology we could increase the success of grassland restoration projects.

What are the key differences between natural and secondary grasslands?

What is a reference grassland?

It is important to identify and sample the reference community that you are trying to restore (Soc. Eco. Rest. 2004). From this you can determine how disturbance may have changed the grassland ecology and what elements have been lost. The reference community can also help you recognise any physical and biological constraints that you are likely to face when trying to restore secondary grassland (Cole & Lunt 2005, Prober & Theile 2005).

Natural grasslands in South Africa are characterised by low lying, continuous grassy vegetation cover. Both coastal and upland grasslands support a high diversity of non-grassy plant growth forms called forbs. In coastal grasslands grass cover can give way to small patches that are dominated by both grass and geoxylic suffrutice vegetative cover. This is not as common in upland natural grasslands. The grass layer is usually kept heterogeneous in height because of the diversity of grass species (Fynn *et al.* 2010).

Secondary grasslands are dominated by a more or less continuous and low lying grassy layer (Chapter 2). Upland areas in this study had a more homogeneous grass layer than natural grasslands as they are dominated by mono-specific swards of *Eragrostis curvula*. When coastal secondary grasslands were dominated by grass species, they included *Dactyloctenium australis,* a lawn grass species, and less commonly *Digitaria* sp., a bunch grass species. Often the woody shrub *Helichrysum kraussii* would also be prevalent in the landscape. This gave coastal secondary grasslands a less homogenous structure than the upland secondary grasslands when perceived from a distance. These observations suggest that it can still be difficult to distinguish secondary from natural vegetation without taking a closer look.

What are the key differences between natural and secondary grasslands?

Secondary grasslands differed from natural grasslands in several fundamental features (Chapter 2). Key amongst these were depleted diversity within secondary grasslands with much lower species richness and a different species composition compared to natural grasslands. Secondary grasslands were missing resprouting species, a fact that was made clear when they did not respond to fire disturbance in the same way as natural grasslands. Secondary grasslands have much lower below ground root biomass, especially of forb species, contrasting with the strikingly high below-ground biomass of natural grasslands. Secondary grasslands resultant from afforestation and ploughing disturbances are thus severely altered both above and below ground.

Do secondary grasslands recover?

There are many factors that can drive succession (Pywell 2002, Soc. Eco. Rest. 2004). It is obvious that natural succession has failed to restore the composition of the secondary grasslands in our study, even after 40 years (Chapter 2). In fact, there is little or no vegetative difference between younger secondary grassland that have been recovering for up to ten years and older secondary grassland that have been recovering between 15 and 40 years. It would seem that physical limitations related to soil attributes may not be the main constraint on vegetation recovery. Coastal secondary grassland soil attributes were no different from natural grasslands, even a year after the pine forests had been excised (James

1998). Soil may be more important in upland secondary excised grasslands where a higher risk of erosion results in the removal of top soil. Ex-arable land however, showed no difference in the soil attributes tested. The upland soil differences were only briefly explored in this thesis and soil changes and their effects warrant further study.

Biological constraints, in two respects, may be more important than physico-chemical constraints in a grassland restoration context. First, grassland forbs are limited by their reproductive capabilities which do not promote re-colonisation (Overbeck *et al.* 2007, Zaloumis & Bond 2011). Grassland forbs live for long periods of time using their underground storage organs to survive frequent defoliation events and are able to resprout quickly after a fire and take advantage of the reduced grass biomass. This ability to persist could have trade-offs with colonising ability because of different patterns of resource allocation in persisters vs. recruiters (Bond & Midgley 2001). An additional cost to long persistence by resprouting might be accumulation of genetic load from somatic mutations leading to reduced viability and seed production (Lamont & Wiens 2003). The poor colonising ability of highly persistent resprouting species in recovering vegetation has also been noted before in mine restoration sites in Western Australia (Norman *et al*. 2006). A second important biological constraint is that grasses are highly competitive plant growth forms and can inhibit forb seedling establishment success (Chapter 3).

Can sods be used as a method to re-introduce native grasslands species into secondary grasslands?

Can we restore South African grasslands?

What are the major constraints towards helping restore our grassland? It is obvious from Chapter 2 that we need to take a more applied approach to grassland restoration if we want to achieve any form of restoration at all. Chapter 2 has helped us better define what natural grassland are and how it can be negatively affected by disturbance. We need to use this knowledge to develop practical methods and take into account the restoration limitations that South African grasslands face.

After the disturbance has been removed, restoration aims should include ensuring no further degradation occurs and attempting to get the system to respond to natural disturbances again (Soc. Eco. Rest. 2004). Topsoil erosion and the encroachment of alien species are just two examples that restoration efforts must look to avoid. This means that the first phase of grassland restoration should aim to establish a vegetative layer as quickly as possible. Unfortunately, to do so may require non-native species or a few fast growing native species. The second phase is to restore community function and composition by attempting to re-introduce as many native species, of all plant growth forms, into the secondary grassland by overcoming their reproductive limitations (Chapter 2).

What restoration methods could we use in South African grasslands?

There are four main methods available to us; re-seeding, hay transfer, individual plant transfer or sod/turf transfers (Cole & Lunt 2005, Buisson *et al.* 2006, Oudtshoorn *et al.* 2011, Le Stradic 2012). Re-seeding and hay transfer rely on suitable seed availability in the vegetation when seed or hay is harvested. This may be problematic where forb and grass species flower throughout the growing season. One would have to decide on which species to target for such an endeavour. Individual plant transfer could be done, but would require some intensive man power and there is a transplant risk of damaging the plants. Both methods may fail where the secondary environment is unfavourable for receiving the plants or their propagules (Le Stradic 2012). Sod transfer has the benefit of bringing in multiple species as well as a micro soil environment that is similar to that of the source grassland. Sod transfer is also a destructive method and would result in the degradation of sod source sites. All these methods rely on source natural grasslands which may not always be so easy to locate (Ash *et al.* 1994, Bradshaw 1997, Le Stradic 2012). The sod transfer was what I chose to test within the context of this study.

Was the method of introducing mixed species sods into secondary grasslands successful?

Forb and geoxylic suffrutex species were generally difficult to transplant due to their below ground habits. Some forbs were very large and could not even fit in the sod dimensions, while even smaller forbs were not always safe from being damaged in the transfers. Graminoid species transferred successfully. Overall sod transfer was successful with many species, even forbs, establishing and creating small source pockets of native species within the secondary grasslands. Monitoring these species in the future will inform us if they continue to survive and spread.

How are grassland forb species affected by grass competition?

How important is grass competition?

In Chapter 2 we found that the majority of secondary grasslands are dominated by one or two competitive grass species. How these grasses interact with potential native forb species is important for determining how restricting they can be for introduced seedlings. Grasses are important competitors for light in un-burnt natural grassland and can result in the shading out of adult forbs (Fynn *et al.* 2004, Uys 2006). Grass competition is also an important factor for limiting tree seedling establishment (Harris 1977, Scholes & Archer 1997, Bond 2008, Cramer *et al.* 2010) and one can assume that grasses could have the same limiting interactions with forb seedlings. Forb seedlings, like tree seedlings, may require a 'gap' where they are free from grass competition to establish successfully.

Do grassland forb seedlings compete successfully with grass species?

Chapter 3 shows that forb seedlings are severely restricted by grass competition. Faced with different grass species of differing competiveness, all the forb seedlings were negatively affected. The forbs that were tested showed that there may be different growth responses or restrictions to high levels of competition. This includes an inability to grow at all or adjusting their growth habit to attempt to increase access to resources, such as light.

Future research:

Soil attributes in upland grassland systems

The brief exploration of secondary grassland soil characteristics produced some interesting but mixed results. It would be worth doing a thorough study on the soil differences between natural and secondary upland grassland and take a closer look at soil succession when comparing older and younger recovering areas. Soil attributes can play an influential role in driving vegetative succession (Pywell 2002, Soc. Eco. Rest. 2004). Soil and disturbance management for secondary grasslands needs to be better understood to optimise for a natural succession of the environment. If we can confirm Chapter 2's results that mountainous secondary grassland soils start relatively unaltered, compared to natural soils, but then begin to degrade after being left to recover for over 10 years then restoration methods have to include applied mitigation measures to prevent this.

Lack of natural grassland propagule sources for re-colonisation

South African grasslands show evidence of lacking the typical necessary plant propagule sources, such as available viable seed, which are generally not common in resprouter dominated communities (Bond & Midgley 2001, Norman *et al.* 2006, Chapter 2). This type of biological limitation can play an important role in limiting restoration activities (Kardol *et al*. 2008). For South African grasslands reproductive constraints may be the most important driver influencing grassland restoration success. Resprouting grassland plants offer limited opportunities for collecting sufficient viable seed for restoration. Further study into forb reproductive biology could help us determine the most efficient method to collect plant propagules and how best to use them in restoration projects. As it is entirely possible that grassland forbs reproduce vegetatively through the expansion of underground storage organs, we could identify methods that could utilise the cuttings from sampled grassland species and successfully propagate native species back into secondary grasslands.

Increasing the application scale and practical feasibility of mixed species sod transfer methods

Developing tools to create species pools in secondary grasslands could help us kick start natural succession. One potential method is to develop artificial sods that include plants grown in nursery conditions close to secondary grassland sites. Propagules of common forb species could then be collected from natural grasslands and used as sources to develop these sods. This, in conjunction with multi-grass species seeding methods, could be effective in re-establishing native grassland species in recovering areas.

Fire and grazing as management for restoration

Secondary grasslands do not respond to natural disturbances in the same way that intact natural grassland do. Although the recovered grass layer does resprout relatively quickly, the vegetation remains sparse for longer periods of time and is more susceptible to soil erosion. If burning secondary grasslands can promote erosion, does it depend on the time of year of the burn, or can other methods be used to control invasive grass cover while trying to establish other native species?

Defining the grassland forb seedling gap

Further experiments into grassland competition and gap creation need to be done in a field study context. Do gaps in natural and secondary grasslands differ in size? How do soil attributes and disturbances help or hinder the ability of a forb seedling to utilise a gap? How do gaps develop in natural grassland and how can one manage for their creation so that they can be used to help restore secondary grasslands.

Can secondary grasslands be restored?

This study has highlighted some key difficulties that restoration ecologists and managers would face in practically restoring grasslands in South Africa. With a lot of money and effort you could return the system to some partial state of natural grassland or at least initiate a more natural succession for the future. If grasslands can recover, they will take decades, if not centuries, to return to their natural state via succession. There are several methods we could use to try and establish native species within secondary grasslands to encourage higher species diversity and the replacement of lost traits. We at least know that grassland species are transferable. The first phase is to re-establish an effective vegetative layer. Then we should aim to create an environment that encourages natural forb re-colonisation and a natural response to natural disturbances. Establishing the vegetative layer is important and achievable, but returning any diversity and function may be far more difficult than previously thought.

South African and Brazilian grassy biomes share distinct plant growth forms, geoxylic suffrutices and persistent forbs with large underground storage organs, that are not described in the North American Prairies or Australian and the European grasslands (White 1976, Overbeck *et al.* 2007 , Bond & Parr 2010, Simon & Pennington 2012). In the restoration of South African grasslands these two growth forms do not easily re-colonise secondary grasslands and are proving problematic (Zaloumis & Bond 2011, Chapter 2 & 3). We could expect Brazilian grasslands to then face the same grassland restoration limitations as South Africa. Is this context unique to South Africa and Brazil when compared to the global grassland restoration literature? Within a different vegetation type similar constraints to what Zaloumis and Bond (2010) and Chapter 2 describe in grassland recovery have been observed during active restoration of resprouting forest species in recovering Australian Bauxite mines sites (Norman *et al.* 2006). As restoration ecology becomes more popular there is ample literature and case studies one can explore before planning a restoration project. There is even sufficient literature available from Australia and European Grasslands and North American Prairies (Leach & Givnish 1996, Pärtel *et al* 1998, Lunt & Morgan 1999, Prober & Thiele 2005, Buisson *et al.* 2006, Dickson & Busby 2008, Kardol *et al.* 2008, Piqueray *et al.* 2011 to mention a few).

When attempted, grassland restoration techniques focus on re-introducing single or several plant species through re-seeding, hay transfer, individual plant transplanting and turf transplanting methods. How universal are these methods when they are applied to South African grasslands? The re-seeding and hay transfer methods are the most common and have been used in our upland grasslands with some success in establishing more preferable grazing vegetation (Mentis 1999, Oudtshoorn *et al.* 2011). Currently, however, they hold little practical opportunities in restoring diverse South African grassland. As far as we know our South African mesic grasslands have no persistent seed bank, feature very little available viable seed and there have no annuals. Transplanting individual plants, whether seeded or from natural veld, and transplanting tufts or sods, so far also prove to be problematic in a South African context.

South African grasslands are fascinating and diverse systems with much work still to be done in understanding their ecology. Our understanding of South African grassland plant biological traits is still limited and there is a lot of potential for further exploration. At the same time however, work is needed to investigate practical grassland restoration methods in South African grasslands. Although grassland restoration application may be limited in South Africa for now, we can still apply our current knowledge into increasing the success of basic grassland rehabilitation to prevent further landscape degradation. Until we can successfully perform large scale restoration projects that return a large proportion of grassland diversity and function we can still promote the value of the natural grassland systems that we have left. This makes the conservation of our remaining and fragmented grasslands a priority.

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Appendix 1: Example of resprouting grassland forb species

a. *Raphionacme sp.*

b. *Ledebouria mokobulaanensis*

c. *Clerodendrum hirsutum*

d. *Pentanisia prunelloides*

Appendix 2: Examples of grassland Geoxylic suffrutices

a. *Ozoroa sp.* – geoxylic suffrutex habit in coastal grasslands

d. Geoxylic suffrutex sp. with fruit

c. *Pygmaeothamnus chamaedendrum*

Appendix 3: Examples of underground storage organs found in South African grasslands

a. *Raphionacme lucens*

b. *Gnidia kraussiana*

c. *Crinum macowanii*

d. *Hypoxis sp.*

Appendix 4: Examples of burnt highland grassland.

a. Natural grassland

b. Secondary grassland

Appendix 5: Examples of burnt coastal grassland.

a. Natural grassland

b. Secondary grassland

Appendix 6: Examples of un-burnt highland grassland

a. Natural grassland

b. Secondary grassland

Un-burnt Comparison - Highveld

Appendix 7: Family richness and abundance for Makobulaan Nature Reserve.

	Richness				Abundance		
Family	**B**	**N**	**R**	**Total**	**N**	**R**	**Total**
Dicots	**25**	**65**	**16**	**106**	**402**	**95**	**497**
Aizoaceae	1			1	1	1	2
Anacardiaceae			2	2	0	2	2
Asclepiadaceae	1	2		3	7	2	9
Asteraceae	14	23	6	43	166	51	217
Budliaceae			1	1	0	3	3
Campanulaceae		1	1	2	4	4	8
Convolvulaceae		1		1	1	0	1
Crassulaceae			1	1	0	1	1
Dipsacaceae		1		1	4	0	4
Euphorbiaceae	1	5		6	30	2	32
Fabaceae	3	7		10	52	7	59
Geraniaceae		1	1	2	3	1	4
Lamiaceae		1		1	4	0	4
Lobeliaceae		1		1	1	0	1
Malvaceae	1			1	5	4	9
Mesembryanthemaceae		1		1	4	0	4
Oxalidaceae		1		1	1	0	1
Periolocaceae	1			1	3	1	4
Polygalacea		3		3	19	0	19
Polygonaceae		1	1	2	2	7	9
Rosaceae			1	1	0	1	1
Rubiaceae	1	4	1	6	33	2	35
Santalacea	1	2		3	17	1	18
Scrophulariaceae	1	3		4	19	3	22
Thymelaeaceae		2		2	16	0	16
Unknown		4	1	5	7	2	9
Verbenaceae		1		1	3	0	3
Pteridophyta			1	1	0	3	3
Monocots	**10**	**41**	**6**	**57**	**293**	**42**	**335**
Anthericaceae		2		2	12	0	12
Colchicaceae		1		1	2	0	2
Commelinaceae	1	2		3	8	1	9
Cyperaceae	1	4	1	6	25	4	29
Eriospermaceae		1		1	7	0	7
Hyacinthaceae		3		3	7	0	7
Hypoxidaceae	2	4		6	29	2	31
Iridaceae	1	6	1	8	25	3	28
Poaceae	5	17	4	26	177	32	209
Unknown		1		1	1	0	1
Grand Total	**35**	**106**	**23**	**164**	**695**	**140**	**835**

Appendix 8: Family richness and abundance for Buffelskloof Nature Reserve.

	Richness				Abundance		
Family	**B**	**N**	**R**	**Total**	**N**	**R**	**Total**
Dicots	**11**	**89**	**17**	**117**	**300**	**75**	**375**
Acanthaceae		1		1	4	0	4
Aizoaceae		1		1	3	0	3
Asclepiadaceae		2		2	3	0	3
Asteraceae	2	33	3	38	109	20	129
Budliaceae			1	1	0	2	2
Campanulaceae		1		1	1	0	1
Caryophyllaceae		1		1	4	0	4
Chrysobalanaceae		1		1	4	0	4
Crassulaceae		1		1	2	0	2
Euphorbiaceae		7	1	8	22	1	23
Fabaceae	9	9	2	20	79	25	104
Geraniaceae		1		1	2	0	2
Lamiaceae		2		2	4	0	4
Lobeliaceae		1		1	2	0	2
Myricaceae		1		1	1	0	1
Oxalidaceae		1		1	2	0	2
Periolocaceae		1		1	2	0	2
Polygalacea		2		2	8	0	8
Polygonaceae		1		1	5	0	5
Proteaceae		1		1	1	0	1
Pteriphyta			1	1	0	6	6
Rosaceae			1	1	0	1	1
Rubiaceae		5	3	8	9	12	21
Santalacea		3		3	5	0	5
Sterculiaceae			1	1	0	3	3
Thymeleaceae		2		2	6	0	6
Unknown		9	4	13	14	5	19
Verbenaceae		2		2	8	0	8
Pteridophyta			1	1	0	1	1
Monocots	**11**	**35**	**10**	**56**	**201**	**66**	**267**
Anthericaceae	1			1	4	1	5
Commelinaceae	1	1		2	10	4	14
Cyperaceae	2	4	2	8	37	6	43
Eriospermaceae		1		1	2	0	2
Hyacinthaceaea		3		3	8	0	8
Hypoxidaceae		2		2	4	0	4
Iridaceae		6		6	16	0	16
Poaceae	7	17	8	32	119	55	174
Unknown		1		1	1	0	1
Total	**22**	**124**	**28**	**174**	**501**	**142**	**643**

Appendix 9: Family richness and abundance for Blue Crane Farm.

	Richness				Abundance		
Family	**B**	**N**	**R**	**Total**	**N**	**R**	**Total**
Dicots	**11**	**47**	**15**	**73**	**153**	**61**	**214**
Anacardiaceae			1	1	0	1	1
Anthericaceae		1		1	2	0	2
Asclepiadaceae		3		3	5	0	5
Asteraceae	4	17	7	28	50	30	80
Bignoniaceae			1	1	0	1	1
Budliaceae			1	1	0	1	1
Campanulaceae		1		1	5	0	5
Euphorbiaceae		2		2	9	0	9
Fabaceae	2	9		11	37	3	40
Geraniaceae		2		2	5	0	5
Hypericaceae	1	1		2	7	3	10
Lamiaceae		2		2	2	0	2
Lobeliaceae	1		1	2	3	3	6
Malvaceae		1		1	1	0	1
Oxalidaceae	1		1	2	7	5	12
Plantaginaceae			1	1	0	2	2
Polygalacea		1		1	1	0	1
Polygonaceae			1	1	0	5	5
Rosaceae			1	1	0	2	2
Rubiaceae	2	2		4	11	5	16
Santalacea		1		1	1	0	1
Sterculiaceae		1		1	2	0	2
Unknown		3		3	5	0	5
Pteridophyta	2			2	3	2	5
Monocots	**12**	**33**	**6**	**51**	**141**	**46**	**187**
Commelinaceae	1	1		2	11	7	18
Cyperaceae	2	4		6	19	4	23
Hyacinthaceae		2		2	2	0	2
Hypoxidaceae	2	1		3	15	6	21
Iridaceae	1	5		6	12	5	17
Poaceae	6	19	6	31	80	24	104
Unknown		1		1	2	0	2
Total	**25**	**80**	**21**	**126**	**297**	**109**	**406**

Appendix 10: Family richness and abundance for Eastern Shores, iSimangaliso Wetland Park.

	Richness				Abundance		
Family	**B**	**N**	**R**	**Total**	**N**	**R**	**Total**
Dicots	**11**	**47**	**15**	**73**	**635**	**256**	**891**
Acanthaceae	2	1		3	9	5	14
Anacardiaceae			1	1	0	1	1
Apiaceae	1			1	6	3	9
Apocynaceae		1	1	2	2	1	3
Asclepiadaceae	1	1	1	3	3	2	5
Asparagaceae	2			2	18	11	29
Asteraceae	7	16	7	30	99	63	162
Boraginaceae		1		1	1	0	1
Campanulaceae	1			1	2	2	4
Celastraceae		1		1	4	0	4
Chrysobalanaceae		1		1	16	0	16
Convolvulaceae	1	1		2	4	2	6
Dipsacaceae		1		1	12	0	12
Euphorbiaceae	1	7	1	9	25	6	31
Fabaceae	5	30	8	43	128	42	170
Geraniaceae		1		1	2	0	2
Lamiaceae		2		2	3	0	3
Lauraceae	1			1	4	10	14
Lobeliaceae	1			1	18	18	36
Malvaceae		2		2	0	0	0
Myrtaceae		2	1	3	6	6	12
Ochnaceae	1			1	2	1	3
Periplocaceae		2		2	6	0	6
Polygalaceae	1			1	11	2	13
Polygonaceae		2	1	3	15	1	16
Rubiaceae		2		2	19	0	19
Scrophulariaceae	2	6		8	31	6	37
Smilacaceae	1			1	18	10	28
Thymelaeaceae		3		3	7	0	7
Turneraceae	1			1	5	2	7
Unknown	8	45	33	86	158	62	220
Vitacaceae		1		1	1	0	1
Pteridophyta			1	1	0	1	1
Monocots	**22**	**35**	**22**	**79**	**296**	**194**	**490**
Amaryllidaceae		4		4	7	0	7
Anthericaceae		2		2	15	0	15
Araceae		1		1	3	0	3
Asphodelaceae		2		2	2	0	2
Commelinaceae	4	2		6	29	12	41
Cyperaceae	6	3	5	14	78	67	145
Eriospermaceae		1		1	2	0	2
Hyacinthaceae		3	1	4	8	1	9
Hypoxidaceae		2	1	3	6	1	7
Iridaceae	1	2		3	12	4	16
Poaceae	11	11	14	36	131	108	239
Unknown		2	1	3	3	1	4
Total	**59**	**164**	**77**	**300**	**931**	**451**	**1382**

Appendix 11: Preliminary simple, four step method to aid in the identification of natural grassland and distinguish it from secondary grassland (Refer to Figure 1 below). This has been developed from the outcomes of our study for the use of conservation managers, EIA practitioners or anyone with some form of natural science background.

Step 1: Burnt or Un-burnt (refer to Appendix 4 and 5)

Secondary grasslands do not feature many resprouting grass and forb species. A few weeks after a burn natural grasslands exhibit an increase in forb flowering as well as quick vegetation regeneration growth from both grasses and forb species, with a thicker vegetative ground cover. Secondary grasslands show far more bare soil even months after a fire, they also feature few resprouting grass species and very few if any forb species flowering.

After a burn:

Natural grassland: Should be easy to clearly identify a highly diverse range of resprouting grass and forb species. An average of 8 - 12 forb species per meter squared.

Secondary grassland: Lots of obvious bare ground, few species to identify. An average of 1 - 3 weedy forbs species per meter squared.

If grasslands are un-burnt it cannot always be obvious to the untrained eye what may be a natural or a secondary grassland (Appendix 6). Especially if a natural grassland has experienced minor degradation in the past through processes such as intensive grazing. This is usually indicated by some species being more prevalent, promoted through the disturbance.

Step 2: Grass species heterogeneity

Natural grassland has a high diversity of grass species and features a more evenly distributed dominant grass species vegetation cover. In contrast, secondary grasslands are dominated in cover by one or two species and feature few other additional grass species.

Natural grassland: Around five grass species dominate the vegetation cover with possibly several more grass species present. The top five grass species together take up between 60 – 80% of the vegetation cover in a meter squared.

Secondary grassland: One, or sometimes two, grass species dominate the vegetation cover with few other species present. Top grass species often takes up between 90 – 100% of cover in a meter squared.

Step 3: Forb alpha diversity

Natural grasslands are rich in persistent forb species. Recording richness of forb species in several plots should be enough to identify a natural grassland. Also note if the forbs are weedy/annual, or have a solid or woody base leading to a woody root stock or a bulb.

Natural grassland: Would have an average of 7 - 12 forb species per meter squared. Forb species should feature solid bases and support woody root stocks or bulbs.

Secondary grassland: Would feature an average of 2 -3 forbs species per meter squared. Forbs are usually weedy, have little or no root stock at all. Secondary grasslands also sometimes feature a high density of woody shrubs compared to natural grasslands, especially along the coast.

Step 4: Presence of resprouting organs

The forbs in natural grasslands are supported by large underground storage organs (USO's) from which they are able to resprout after a fire disturbance. These USO's make up a considerable amount of below ground biomass in natural grassland systems. USO's as a functional trait are almost completely missing in secondary grasslands. One should be able to dig up any forb within natural grassland to find that it is attached to some form of USO. Digging up even a 30 x 30 cm plot will easily demonstrate if a grassland natural or not.

Natural grassland: Coastal - Holds an average of 2.5 – 5 kg/m^2 of forb species USO biomass. Highland - Holds an average of 0.7 – 1.5 kg/m^2 of forb species USO biomass. This excludes geoxylic suffrutices root biomass which can add up to 7 more kg/m^2 to coastal plots.

Secondary grassland: Holds an average of 0.0 – 0.1 kg/m^2 of forbs species USO biomass. As most secondary grassland forbs are weedy or annual species and they will have no or little root biomass below ground. If a grassland if suspected of being secondary, this is the most obvious of tests to be sure. This excludes woody and/or fern root biomass, which can be substantial in secondary grassland.

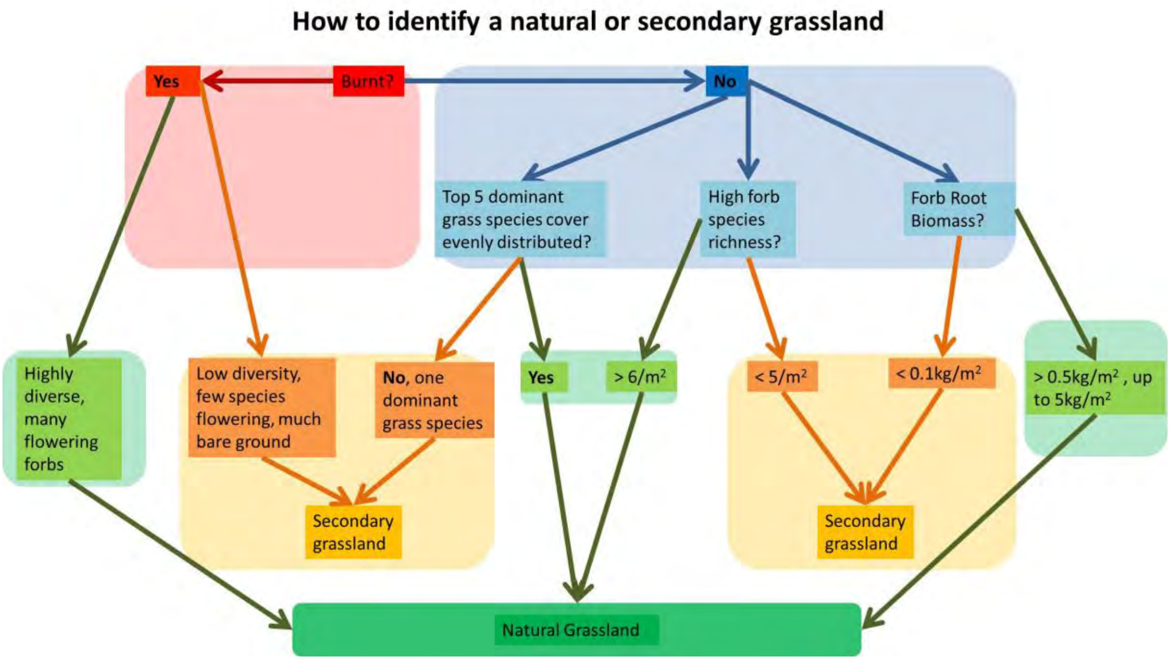

Figure 1: Preliminary four step method to aid in distinguishing natural grassland from secondary.

Appendix 12: Mixed sod restoration plot preparation.

a. Restoration plot preparation

b. Coastal grassland sod with USO sticking out the bottom

c. Prepared plot with sods placed in cleared quadrats

d. Planted sod

e. Sods being transported to plots