EFFECTS OF REMOVING *ACACIA MEARNSII* ON THE WATER TABLE, SOIL AND VEGETATION PROPERTIES IN THE TSOMO VALLEY OF THE EASTERN CAPE PROVINCE, SOUTH AFRICA

By

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I declare that The effects of the removal of invasive Acacia mearnsii on the water table; soil and vegetation properties in the Tsomo valley of the Eastern Cape, South Africa is my own work, that it has not been submitted for any degree or examination in any other university, and that all the resources I have used or quoted have been indicated and acknowledged by complete reference.

Hloniphani P.M. Moyo   January 2010

ABSTRACT

Invasion of rangelands by trees such as *Acacia mearnsii* threaten native vegetation and their habitats because invasive trees compete with native vegetation for resources such as water, nutrients and the sun. The non-indigenous trees have higher evapotranspiration rates that eventually lead to reduction in stream and river flows. *Acacia mearnsii* also have inhibitory effects on growth of underneath vegetation through exudates from either their roots or leaves that prevent either the germination or the full establishment of native vegetation. The purpose of this study was to investigate (1) the effects of *Acacia mearnsii* on water table; and (2) the effect of *Acacia mearnsii* on soil and vegetation properties in the Eastern Cape Province, South Africa.

Water table was measured using HOBO data loggers in plots where *A.mearnsii* was cleared and where it was not. There was significantly higher ($p < 0.05$) recharge in plots where trees were cleared (-8.7m) compared to where they were not (-9m). Soil samples were collected from inside and outside *Acacia mearnsii* canopies and analyzed for pH, soil organic carbon, soil aggregate stability, electrical conductivity and nitrogen content. Soil from under *A.mearnsii* canopies was more acidic (3.05) compared to that from outside (5.47). Nitrogen content was significantly higher in soils from under the canopies (0.52 mg kg^{-1}) compared to those from outside (0.21 mg kg^{-1}). Soil aggregate stability was not significantly different ($p > 0.05$) between soils from inside and those from outside canopies.

Total herbaceous production was measured using the dry-weight method. Total herbaceous production was significantly higher ($p < 0.05$) in the plots that were cleared (290 kg ha^{-1}) than in those that were not (22 kg ha^{-1}).

Soil samples were collected from inside and outside canopies, incubated in the greenhouse to determine the seed bank botanical composition and the seed bank size. Soils from outside canopies had higher seed bank size ($p < 0.05$) (59% of total seedling emergence) than soils from under the canopies (41%). *Acacia mearnsii* contributed 5% of total seedling emergence in trays while forbs contributed 50%, showing that in the absence of *A.mearnsii,* germination increases. It is concluded that *Acacia mearnsii* significantly reduces groundwater recharge, soil pH, biomass production and native vegetation seed bank whilst increasing soil nitrogen and soil organic carbon under its canopies.

ACKNOWLEDGEMENTS

DEDICATION

This work is dedicated to my late mother, Miss Winnie.

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LIST OF ABBREVIATIONS

ANOVA	Analysis Of Variance
C/N	Carbon and Nitrogen ratio
DWAF	Department of Water Affairs
GLM	Generalized linear model
N	Nitrogen
Na	Sodium
Mg	Magnessium
Mn	Manganese
K	Potassium
B	Boron
Ca	Calcium
Fe	Iron
Fwet	soil fast wetting
Swet	soil slow wetting
MechD	Soil mechanical disintegration
BC	Basal cover
SAS	Statistical Analysis Software
WfW	Working for Water

1. INTRODUCTION

South Africa has one of the largest problems of invasive alien plants in the world with the forest sector being one of the country's major sources of infestation (Richardson, 1998). A large proportion (38 %) of the area invaded by woody alien plants in South Africa is occupied by species used in commercial forestry, in particular, *Pinus* and *Acacia* species (Rouget *et al*., 2002). *Acacia mearnsii* is considered a major invasive tree species in South Africa having invaded about 2.5 million hectares of land (Galatowitsch and Richardson, 2004).

A. mearnsii threatens native habitats by outcompeting indigenous vegetation for water, nitrogen and organic materials; replacing grass communities, reducing biodiversity, and increasing water loss from the riparian zones (Nyoka, 2003). Non-indigenous species, like *A. mearnsii*, with evapotranspiration rates higher than those of the native flora, significantly alter hydrological regimes and lower water tables (Le Maitre *et al.*, 2000).

The displacement of fynbos vegetation by populations of invasive Australian *Acacias* has been attributed mainly to habitat modification by the *Acacias* themselves, through mineral enrichment associated with increased litter fall and decomposition (Musil, 1993; Campbell, 2000). Afforestation and tree invasion or encroachment affects many types of grasslands globally; however, the effects on ground water and salt fluxes remain poorly understood in rangelands (Jobbagy and Jackson, 2003). Soils under *A. mearnsii* become desiccated more quickly than under grasses affecting germination and growth (Galatowitsch and Richardson, 2004). The dry mass of litter weight of *A. mearnsii* under invaded *Acacia* stands is greater than that of

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uninvaded native vegetation (Witkowski, 1991), leading to assumptions that the dense layer inhibits establishment of indigenous seedlings (Richardson, 1998).

Data on water use by the different invading species are limited and research to address these gaps should have a high priority (Le maitre *et al.*, 2000). Quantifying the amount of water table change can lead to recommendations that take into consideration threshold levels of *A. mearnsii* that would affect rangeland production and vegetation diversity.

Objectives

The general objective of the study was to determine the effects of *A. mearnsii* removal on water table recharge, vegetation and soil properties in the Tsomo Valley of the Eastern Cape.

The specific objectives of the study were to determine the effects of *A. mearnsii* on;

- water table fluctuation

- vegetation regrowth

- soil chemical and physical properties

- soil seed bank

Hypotheses

1. Clearing of *A. mearnsii* has a significant effect of water table recharge because the water table level will rise after clearing trees

2. The removal of *A. mearnsii* has an effect on vegetation regrowth since *A. mearnsii* canopies support low vegetation diversity

3. Since *A. mearnsii* affects vegetation regrowth, soils with properties that negatively affect vegetation growth such as low pH are expected

4. *A. mearnsii* affects seed bank for vegetation growth through preventing establishment of vegetation underneath the canopies

2. LITERATURE REVIEW

Introduction

Invasive woody species are plants that are able to survive, reproduce and spread unaided, sometimes at alarming rates, across the landscape. They have detrimental effects on commercial trees and give management problems (Nyoka, 2003). Species that invade natural ecosystems, whether indigenous or exotic, are referred to as environmental weeds (Campbell, 1998). Invasive trees pose serious threats to nature conservation in the world (Scholes and Archer, 1997). The invasion of rangelands by exotic species represents a change in plant community composition whereby the substitution of an indigenous species by one or more exotic species has important consequences for conservation, ecosystem structure and function of the invaded ecosystem (Ehrenfeld, 2003).

Invasive species are a symptom of native community habitat degradation. They often prevent the re-establishment and generation of native species in invaded areas (Ogden and Rejmánek, 2005). When split into the major biomes in South Africa, the fynbos (a Mediterranean-type shrub land) is the most invaded biome. It is invaded largely by *Pinus*, *Acacia* and *Hakea* species in the mountains, lowlands and along some major river systems in South Africa (DWAF, 1997).

About 750 tree species and around 8 000 shrubby, succulent and herbaceous species are recorded as having been introduced into South Africa in the 19th century (van Wilgen *et al*., 2001).

Of the 8 750 species, 161 are now regarded as invasive, 68 % of these invasive species are classified as woody (Nyoka, 2003).

The total area invaded by alien trees in South Africa is over 100 000 km^2, which is over 8 percent of the country's total area (Ward, 2004). These invasions are mostly concentrated in wetter regions of the country and along river systems (Nyoka, 2003) (Figure 2.1). The area of commercial woody plantations in South Africa is estimated at 1.5 million ha (57% pine, 35% eucalypts and 8% wattle), covering just 1.2% of South Africa (Scott *et al*., 1998).
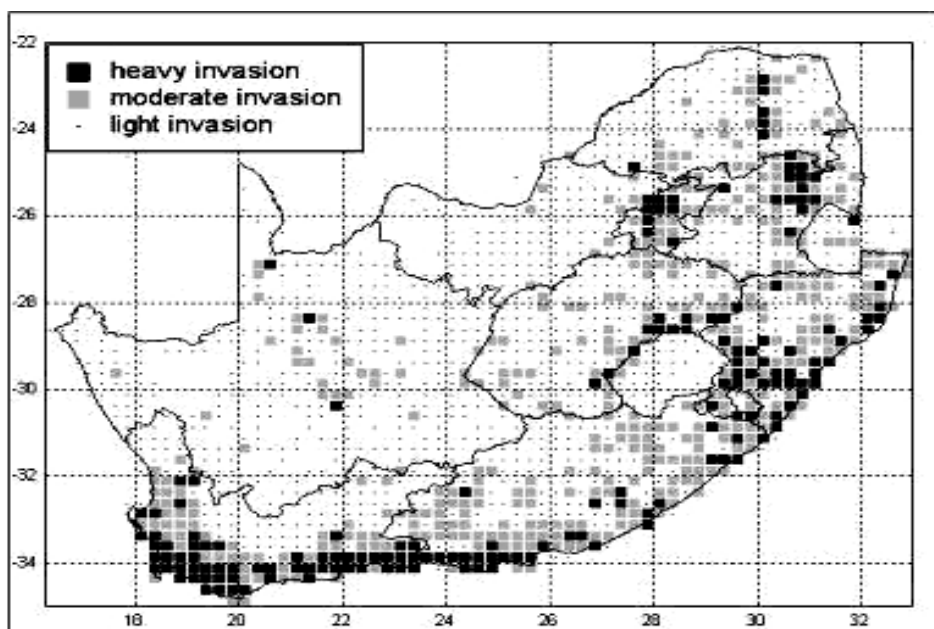
Figure 1. Extent of invasions in South Africa (Source ARC, 2007)

Effects of invasive trees on soil moisture properties

An invasive plant species alters water balance if it gains access to water sources previously unavailable to it, or lightly utilized by the indigenous vegetation (Luken and Thieret, 1997). Alien invaders are estimated to use approximately 6.7% of the mean annual runoff over the entire area of South Africa, translating to over 3 billion m^3 of water per year. In the Western Cape Province invasive trees use as high as 15.8% of the mean annual run-off (Enright, 1999). Commercial forest plantations are estimated to reduce surface runoff by about 1.4 billion m^3 or 3.2% at a national scale. These reductions are important because South Africa has a mean annual rainfall of only 490 mm and less than 10% of this becomes surface runoff (Macdonald and van Wilgen, 2002). The water ending up in rivers and streams decreases due to reduced surface run-off by the plantations.

Witkowski's (1991) work on *Acacia* stands in South Africa has shown that these species affect the stream flow linearly. An increase in intensity of the *Acacia saligna* and *Acacia cyclops* decreased the stream flow (Witkowski, 1991). An increase in the height and biomass through *A. mearnsii* infestations increases rainfall interception and transpiration, causing a decrease in stream flow (Le Maitre *et al*., 2000). Tree plantations have high evaporative potential (about 65%) realized through increased groundwater use due to higher biomass production (Jobbagy and Jackson, 2003).

In South Africa Jobbagy and Jackson (2003) showed that despite heavy rains during their study, a forest Castelli plantation was able to reverse vertical groundwater fluxes, maintaining low

levels of soil moisture and interrupting recharge almost completely. Forest harvesting changes

the rate of transpiration, water-table dynamics. As leaf area is reduced after harvesting, the water

table rises due to less utilization of groundwater (Rejmánek, 2000). The water table has also been

shown to increase significantly after clear cutting or selective harvesting of trees in coastal plain

systems in KwaZulu Natal, South Africa (Smit, 2005).

Dye and Jarmain (2004) conducted a study in 2004 in the Western Cape's Wellington and Groot

Drankennstein area and KwaZulu- Natal (Seven Oaks) on water use by *Acacia mearnsii.* They

found that *Acacia mearnsii* used 7mm of rainfall per day and they estimated additional loss of

185 mm of rainfall used annually by *A. mearnsii*, showing the level of water consumption for *A*

.mearnsii.

Invading alien plants reduced surface runoff in the Keurbooms catchment by about 20%, four

times more than plantations in four South African catchment areas (Dye and Jarmain, 2004)

(Table 1). The figures in brackets represent the water use of the plantations and invasive trees

from the total run-off (Table 1).

Table 1: Estimated water use and % total use of run-off by invasive plants and commercial plantations (Adapted from Le Maitre *et al*., 2000):

Catchment area	Water-use (millions of m^3 &% of total runoff)	
	Commercial plantations	Invasive trees (current)
Sonderend	1.4 (0.3)	33.3 (7.2)
Keurbooms	11.5(5.7)	47.4(22.1)
Upper Wilge	0.5(0.1)	26.9(6.0)
Sabie-Sand	138.2(18.9)	69.3(9.4)

The least water consumer per year was *Acacia saligna* (Table 2). In South Africa *Acacia mearnsii* is the highest in terms of water consumption per year, followed by *Acacia cyclops* (Table 2).

Table 2: Water Consumption of Different Invasive Trees in South Africa (Adapted from Le Maitre *et al*., 2000):

Species name	Estimated Mean Annual Water Use (m^3)
Acacia mearnsii	577
Acacia cyclops	488
Acacia dealbata	248
Acacia mixed spp.	243
Eucalyptus spp.	214
Prosopis spp.	192
Acacia saligna	171

Effects of controlling invasive trees on vegetation structure

The removal of dominant native trees facilitates invasive plant establishment in recovering landscapes. As a result invasive plants that are able to respond favorable to fluctuating post disturbance conditions are more likely to establish and spread (Brown *et al*., 2006) limiting growth of native plants or slowing the rate of change in species composition (Chornesky and Randall, 2003).

Native vegetation growth improves after removal of invasive trees (Davis *et al*., 2000). In Davis *et al*., (2000)'s study native species richness did not, however, increase with time after the removal of the invasive plant, but fluctuated due to rainfall. The removal of the invasive *Impatiens glandulifera* in South Africa resulted in an increase in total species richness of grass species (Hejda and Pyšek, 2006). Invasion by *Impatiens glandulifera* had negligible effect on invaded riparian vegetation and did not affect the establishment and species richness of the vegetation.

Dry matter yield of grasses increased in consecutive rainy seasons for areas where *Acacia nilotica* trees were removed compared to the areas with *Acacia nilotica* trees in South Africa (MacDonald, 2004). As *Acacia nilotica* increased in biomass production and basal area both the perennial and annual grasses decreased in terms of total yield (kg ha^{-1}). As the tree canopy cover increased perennial grass productions were decreased from a high value of 1800kg ha^{-1} to a low value of 500kg ha^{-1} (MacDonald, 2004).

Smit (2005)'s work showed that grass dry matter yield increases in consecutive rainy seasons as tree thinning is intensified until total removal of the trees in a woodland, therefore, high tree density reduced grass dry matter production. Cleared farms were shown by Surridge (2006) to have a higher dry matter yield from about 2 tonnes compared to a low figure of 1.19 tonnes of dry matter on bushy and encroached farms. In the Molopo area, grazing capacity was reduced by at least 50% as a result of *Acacia mellifera* (black thorn) encroachment and native vegetation displacement (Brown and Archer, 1999). *Acacia* tree species like *Acacia mearnsii* can displace native species by hybridization and altering the behavior and abundance of native plant pollinators (Chornesky and Randall, 2003).

Holmes and Cowling (1996), working on the control of *Acacia saligna* in South Africa, concluded that recruitment, species richness and diversity was high in areas where trees were totally removed. Total stand species richness was nearly as high as in recently invaded stands compared to uninvaded areas (Holmes and Cowling, 1996). McCarthy (1997) demonstrated that removal of garlic mustard from a deciduous forest understory resulted in increased richness and abundance of understory species, especially annuals and woody perennials.

A study on the effects of *Acacia sophorae* on plant composition of coastal grasslands by Costello *et al* (2000) in South Eastern Australia showed that uninvaded $2m^2$ quadrants recorded more species diversity (22 plants) compared to the invaded quadrants that recorded a species diversity of 15 plants. Analyses of the quadrant data showed that species richness in invaded areas decreased with an increase in *Acacia sophorae* variables like number of branches in the

quadrants, percentage cover of the *Acacia* trees and tree diameter. An increase in the tree branch number had a direct negative effect on plant species diversity (Costello *et al*., 2000).

Effects of invasive trees on soil properties

Research has shown that individual plant species, many of which are nonnative and invasive, alter soil organic matter quality and quantity, nitrogen status through nitrogen fixation, nutrient stocks, nutrient cycling rates, soil moisture content, pH and cation distribution of soils in invaded ecosystems (Corbin and D'antonio, 2004). Nutrient dynamics are altered as a result of changes in physical properties of the soil caused by the introduction of new species (Ehrenfeld *et al*., 2001). Other introduced species release compounds that inhibit their own growth, as well as that of competitors (Musil, 1993).

Plants impact the diversity of soil microbial communities by variations among their root exudates, chemical quality and the overall quantity of aboveground litter (Kourtev *et al*., 2002). The changes in the soil microbial community are related to differences in the quantities and qualities of inputs to the soil by different species of plants (Ehrenfeld, 2003). Changes also result from alterations in the patterns of species dominance within the plant community since the effects of a given species on ecosystem processes are regulated by its relative abundance within the community (Ehrenfeld *et al*., 2001).

Potential impacts on microbial communities from invasive species often occur because non-native species differ in plant morphology, phenology, and leaf litter chemical composition

compared to co-occurring native plants (Ehrenfeld, 2003). Phytotoxic chemicals that leach from the leaves and litter of eucalypts during rainfall and fog drip events directly inhibit germination and retard seedling growth of grasses, as well as of the eucalypts themselves (Dukes and Mooney, 2004).

Montgomery (2001) in South Africa reported that soils under *Acacia mearnsii* have lower pH_{KCl} values (4.4) than adjacent soils under grassland (5.3). Brown *et al*., (2006) in Puerto Rico investigated the effects of the invasive *Syzygium jambos* on soil properties and found that areas under a high density were higher in minerals phosphorous, calcium, and potassium compared to areas under low *Syzygium jambos*. In the case of *B. thunbergii*, a woody invader in New Jersey, USA, Ehrenfeld *et al*., (2001) showed that litter was high in N, that the soil under these shrubs had higher pH, and higher nitrification rates than adjacent soil under *Vaccinium pallidium*, a native shrub.

Heneghan *et al*., (1994) compared the effects of the invasive buckthorn (*Rhamnus cathartica*, L.) on the soil chemistry. They showed that areas that are invaded by buckthorn had twice the % soil Nitrogen (0.54%) compared to those of uninvaded areas (0.27%). Soils in buckthorn dominated areas had 80% more carbon (6.83%) than open areas (3.81%). Potassium Chloride extractable Nitrogen did not differ between buckthorn areas and open areas across the three sites. Soil pH was significantly higher in buckthorn areas than open areas, indicating the low acid effects of the invasive tree.

Neal *et al*., (2001) showed that soil under *Hieraciurn* invasion had total organic carbon and total nitrogen concentrations that were significantly higher under *Hieraciurn* than under areas without *Hieraciurn* due to higher dry-matter production and carbon inputs. *Hieraciurn* presence also influenced soil pH, base saturation, and mineral-nitrogen concentrations (Neal *et al*., 2001). These data support the idea that *Hieraciurn* depletes available soil nitrogen pools, potentially increasing *Hieraciurn's* competitive advantage over other grassland species (Neal *et al*., 2001).

In South Africa, Musil (1993) found that soils under *Acacia saligna* stands had higher concentrations of N, Ca, Mg, K, Mn and B compared to soils without *A.saligna*. Soils under *Acacia saligna* stands were lower in Fe concentration compared to the soils outside the stands. Comparing the soil properties of soils under greenhouse conditions in South Africa, Ehrenfeld *et al*., (2001) found that soils from under the invasive *Microstegium vimineum* shrub and *Berberis thunbergii* grass had higher pH compared to the soils under native *Vaccinium pallidum* shrub.

Extractable ammonium concentrations were higher under the exotics than under the native *V.pallidum*, while the nitrate concentrations were similar. Ehrenfeld *et al*., (2001) also showed that net ammonification rates were similar for both the exotics and the native species. Higher overall soil respiration for the exotics was partly due to larger root biomass compared to the native. They further showed that nitrification rates were higher in soils beneath both exotic species, compared to soils beneath the native *V.pallidum*.

Effects of invasive trees on grass seed bank

Plants propagate themselves by vegetative or sexual reproduction, with many species using a combination of these strategies (Schelling, 2006). Seed banks are very vital for restoration of plant communities through nature management authorities as they provide vegetative reproductive means (Oke *et al*., 2006). There are difficulties when considering restoration in recently forested areas through the use of buried seeds because some of the shade tolerant species do not always form a persistent seed bank (Bossuyt and Hermy, 2001). The soil seed bank serves as a propagule storage system and is very important to species establishment (Levin, 1990). The soil seed bank consists of viable and ungerminated seeds.

The composition of soil seed bank is influenced by the existing above ground vegetation (Wilson *et al*., 1993). The soil seed bank is, therefore, an important component of the forest, which impacts several aspects of ecological functioning, including genetic, population and community level dynamics (Martins and Engel, 2007). The soil seed bank influences the genetic variation of a plant population by acting as a buffer to changes in a population's genetic composition resulting from major fluctuations in population size. Seed banks play an important role in population dynamics because seeds enter the soil seed bank as they are produced by local plants and fall to the ground or disperse into an area (Schelling, 2006).

A species' survival in a challenging environment is, at least, partially dependent on it having a persistent seed bank, which can survive in the soil for many generations (Wilson *et al*., 1993). The soil seed bank impacts community composition by serving as a reservoir for species not currently present in the above ground layer (Levin, 1990).

In a study by Bossuyt *et al* (2005) soils from under a calcareous grassland, seed bank were compared to those from the scrub vegetation. There was no decline in seed density or species richness under the scrub vegetation and the species composition of the seed bank was similar to that of the calcareous grassland. This means that calcareous grassland species or their seeds remain persistent under the developing scrub vegetation for at least 15 years whilst less light-demanding or competitive species only gradually establish and replenish the soil seed bank.

Seed dynamics play an important role in structuring and maintaining plant communities (Leckie *et al*,. 1999). Seeds sustain populations during temporarily unfavorable conditions; allow establishment in new areas, and can introduce novel genotypes to populations (Martins and Engel, 2007). Seed banks are particularly important in maintaining species and genetic diversity in communities and in allowing species to persist through disturbance or adverse conditions (Leckie *et al*,. 1999). One of the factors influencing the recovery of natural vegetation is the nature of the indigenous soil seed bank, and its persistence may play a very important role in facilitating the recovery of ecosystems after the clearance of invasive alien vegetation (Fourie, 2008).

Causes of invasion

Plants invade new environments in response to environmental changes, movement of animal vectors, evolution of new biotypes, and activities of humans (Young and Longland, 1996). Invasions by exotic species can change ecosystem functioning and threaten ecosystem biodiversity (Drake *et al*., 1989). Changes in stream flows, the frequency of wildfires, and

firefighting promote invasion by non-natives whilst suppressing the growth and development of native vegetation (Matthews and Brandt, 2004). The most critical factor causing woody plant encroachment is chronic high levels of herbivory by domestic animals, causing a reduction of the above ground grass biomass and access to the light through overgrazing (Van Auken, 1999).

Invasion of rangelands by invasive species can be caused by introduction of such species for commercial wood production (Campbell, 2000). In some coastal grasslands in south-eastern Australia, removal of grazing and burning appears to trigger invasion by indigenous shrubs, especially *Acacia sophorae* (Costello *et al*., 2000). The actual invasion of an environment by new species is influenced by an additional three factors: the number of propagules entering the new environment (propagule pressure), the characteristics of the new species, and the susceptibility of the environment to invasion by new species (invasibility) (Collins, 2005).

Invasions can also be caused by seed dispersal of exotic seeds to other sites leading to persistence and proliferation of the exotic plants on the site (Germino *et al*., 1996). Disturbing native vegetation, through cultivation and ploughing, also causes invasion as this often prepares a seed bed for invader weeds (DWAF, 1997).

Native fynbos tree species lack mycorrhyzal associates and are, therefore, less efficient at nutrient uptake compared to alien invasive trees (Smita, 1998). The development of water management programs which usually affect natural water flows such as the construction of reservoirs, dams, river diversions, flow regulations, and irrigation projects often create

conditions unsuitable for the colonization and regeneration of native riparian species (DWAF, 1997).

Environments prone to invasions

Characteristics of invaded environments often include: geographical and historical isolation, low diversity of native species, high levels of natural disturbance or human activities, and absence of co-adapted enemies, including competitors, predators, herbivores, parasites and diseases (Davis *et al*. 2000). Areas where the vegetation and soil have been disturbed by humans or domestic animals are more susceptible to invasion (Sangha, 2003). Rivers are naturally prone to soil disturbance through flooding (DWAF, 1997).

Riparian ecosystems are highly prone to invasion by alien plants, largely because of their dynamic hydrology, nutrient levels, and ability to disperse propagules (Galatowitsch and Richardson, 2004). The severity of weed infestations in riparian vegetation increases as the stream or river flows downstream, particularly as a river flows through the highly developed alluvial soil flats (Witkowski and Garner, 2008).

A community is prone to invasion when the composition and diversity of the resident vegetation is low. The higher the composition and diversity, the more the resources are utilized fully and the less prone the environment will be to invasion (Collins, 2005). People from some cultures are more likely to introduce plants from their homelands when they migrate to new regions (Lemmens and Wulijarni-Spetjiptoed, 1991). Seeds are also transported by flowing water

(Ogden and Rejmánek, 2005). Riparian zones are unique and dynamic ecosystems with complex disturbance regimes and river bank communities are generally considered to be prone to plant invasions (Hejda and Pyšek, 2006).

The ecology of *Acacia mearnsii*

A. mearnsii belongs to *Acacia* section Botrycephalae, a group of 44 mostly arborescent species characterized by having bipinnate adult foliage and flower heads normally arranged in elongated racemes (DWAF, 1997). *A.mearnsii* is native to Australia (Ramachandran, 1993). *A. mearnsii* is a fast-growing pioneer species which reaches its maximum growth rate 3–5 years after planting (Campbell, 2000). It is the most important invader of forest and plantation margins, riparian zones and savannas, woodlands and roadsides (Musil, 1993). *A. mearnsii* is a fast growing leguminous tree and it is often used as a commercial source of tannins and fire wood for local communities (DWAF, 1997).

Unlike the African *Acacias* that are pioneering succession from grasslands to savannah or from savanna to drier-type forests, *A. mearnsii* occupies a higher position in that, it is present as a normal component of the climax high forest in its native habitat (Nyoka, 2003). *A. mearnsii* trees provide bark 5–10 years after seeding and one tonne of *A. mearnsii* bark is sufficient to tan 3 hides, best adapted for sole leather and other heavy goods; the leather is as durable as that tanned with oak bark (DWAF, 1997). *A. mearnsii* is an evergreen tree, 6 to 20 m high and 10 to 60 cm in diameter (Musil, 1993).

The stems for the *A. mearnsii* do not have spines or prickles, the leaves are bipinnate, on petioles 1.5–2.5 cm long, with a gland above and rachis 4–12 cm long with numerous raised glands all along its upper side (Ramachandran, 1993). The pinnae occur in 8–30 pairs, pinnules in 16–70 pairs (Cenci *et al*., 2006). It has dark brown pods that are finely hairy (Henderson, 1998). Apart from producing copious numbers of seeds it generates numerous suckers resulting in monotypic thickets (Robles and Chapin, 1995). It grows in disturbed mesic habitats (Le Maitre *et al*., 1996). Soils with lateritic pan close to the surface are most unsuitable for *A. mearnsii* growth and production (Costermans, 1994).

It is fast growing but short lived, moderately frost tolerant and vigorous at high elevations (DWAF, 1997). It thrives on poor dry soils but favors deeper, moister and more fertile soils (Ward, 2004). The invasiveness of this species is partly due to its ability to produce large numbers of seeds which lie dormant and may be triggered to germinate following bush fires and the development of a large crown that shades other vegetation (Nyoka, 2003). It is a light-demanding species, which is sensitive to fire when young (< 3 years) (Bromilow, 1995). Its lower altitudinal range is decided by the fact that trees cannot stand high summer temperatures, and the upper altitudinal limit is based on the fact that the tree does not tolerate temperatures below 0 ^{0}C (NFTA, 1985). Winter frosts and cold winds during the early part of the rainy season affect growth and survival rate of *A. mearnsii*, but older trees can withstand mild frost (Lemmens and Wulijarni-Spetjiptoed, 1991).

Areas experiencing severe hailstorms and snowfall are unsuitable for the growth of the invasive tree (Campbell, 1993). Adequate soil moisture is a prerequisite for satisfactory growth of the

water demanding tree (Dye and Jarmain, 2004). *A. mearnsii* trees cannot withstand drought because of their superficial root system and high rate of transpiration (Dell'Porto *et al*., 2006). *A. mearnsii* is a hermaphrodite and flowers profusely in the winter (Campbell, 2000). Trees begin to yield fertile seed from the age of 5 years, giving good annual crops (Albecht, 1993).

The minute, fragrant flowers are self-fertile, but cross-pollination occurs and bees are the main pollinators (Albecht, 1993). Pods mature in 14 months, and gravity or propulsion from drying dehiscent pods initiates seed dispersal (Costermans, 1994). In southern Africa, *A. mearnsii* flowers from late July to October (Nyoka, 2003). Plants commence flowering when about two years old (Le Maitre *et al*., 1996). A pod normally contains 1 to 14 seeds, but the average is 7 seeds (Binggeli, 1996). There are between 48 600 to 70 200 seeds per kilogram of soil (Nyoka, 2003).

From the fifth or sixth year onwards, annual seed crops are normally copious (Le Maitre *et al*., 1996). By contrast, in its native range, good seed years are relatively infrequent, and often no seed at all may be produced in some years, mainly because of the action of insect pests (DWAF, 1997). The species does not seem to produce viable seeds at lower altitudes, which explains why most of the other plantings in Zimbabwe never became invasive (Nyoka, 2003).

The species has all the three attributes of a successful invader; short juvenile phase; good annual seed crop and small seed size (Rejmanek, 1995). *Acacia mearnsii* is regarded as an out crossing species with partial self-compatibility (Sherry, 1971). It can be crossed with *Acacia decurrens*, hybrids show more sterility than parents (Nyoka, 2003). Meiosis is regular for *A. mearnsii* with

no gross cytological abnormalities, and sterility may be due to gene differentiation between species (Henderson, 1998). Estimates of out crossing rates are variable, ranging from 48 to 100% (Nyoka, 2003).

The natural range of distribution of *A. mearnsii* is bound by latitudes 34° S and 44° S and altitudes from sea level to 885 masl (Ramachandran, 1993). The tree coppices poorly, it is a noxious weed in South Africa, it has been called "green cancer" because it spreads vigorously (Richardson, 1998). The seed is spread by running water. Heat from fire improves seed germination (Campbell, 2000).

The species was declared an invader species in South Africa in 1984, after years of protracted debate on proposals for the species to be declared a weed (DWAF, 1997). The species is rated as number eleven in the invaders list in the fynbos biome, fifth in the KwaZulu Natal Province and sixth in Gauteng Province (DWAF, 1997). The species is also considered to be the major invader in the mist belt of the eastern highlands of Zimbabwe (Nyoka, 2003). *A. saligna* out competes local plants by overtopping them, growing to a height of 3 to 10 meters (Holmes and Cowling, 1997). Since native plants are shade-intolerant, they typically do not fare well in *Acacia*-infested habitats (Campbell, 2000).

With their large canopy and high transpiration rates, *Acacias* also reduce soil water availability (Costello *et al*., 2000). *Acacia mearnsii* is susceptible to various pathogens, some of which are responsible for economic losses in plantations physiological disorder known as 'gummosis', in which gum is exuded in the absence of any obvious injury (Grice, 2005). According to Binggeli

(1996) gummosis is a serious problem in commercial plantations outside of the natural climatic range of *A. mearnsii* because it reduces bark quality and hinders its stripping. In South Africa this term has been applied to a complex of diseases associated with *A. mearnsii* (Henderson, 1998).

In Australia, the total number of insects associated with *A. mearnsii* is high (Costermans, 1994). The leaf-eating fireblight feetle, *Pyrgoides orphana* (Coleoptera), is a serious pest (Duke, 1983). In South Africa some 200 species of insects have been noted damaging trees of *A. mearnsii*, with about 30 of these being of economic significance (DWAF, 1997). According to Musil (1993) many insects feed on *A. mearnsii* and some cause serious, sporadic damage that affects its survival or growth and form. The fireblight beetle and Gall Rust fungus are particular problems for *A.mearnsii*.

Acacia mearnsii in South Africa

Acacia mearnsii reached southern Africa in about 1864, when it was introduced into South Africa (Nyoka, 2003). The purpose of introduction of this species was for shade, windbreaks, fuel wood and later for its tannin bark (Nyoka, 2003). One hundred years after first introduction, there were close to 324 000 ha of *A. mearnsii* in South Africa (Musil, 1993). The area then progressively declined until it reached the current official level of about 107 000 ha (DWAF, 1997). *Acacia mearnsii* is now considered a major invasive tree species in South Africa having invaded about 2.5 million hectares of land (Galatowitsch and Richardson, 2004).

Uses of *Acacia mearnsii*

The timber from *A. mearnsii* is used for building materials, and the pulp and wood chips are used to produce paper (National Academy of Sciences, 1980). In rural communities in South Africa the trees are important as a source of building material and fuel (Campbell, 2000). *A. mearnsii* has some known medical applications, such as its use as a styptic or astringent (Rouget *et al*., 2002). The planting of *A. mearnsii* has also been used as a soil stabilizer to decrease the rate of erosion (DWAF, 1997). The plantation industry promotes the use of *A. mearnsii* (among other similar species) as a potential soil improver (Rejmanek, 1995). The leaves have high protein content, about 15% (Lemmens and Wulijarni-Spetjiptoed, 1991).

In Hawaii, *A. mearnsii* has been fed to cattle during drought periods (Campbell, 2000). The extra floral nectaries of *A. mearnsii* containing about 20% pollen protein and 40% sugar and its late flowering makes the tree a suitable bee forage (DWAF, 1997). Originally distributed as a source of tannin, *A. mearnsii* is now recognized as a valuable fuel wood (Rejmanek, 1995). Wood is moderately dense with specific gravity about 0.75, splits easily and burns well with a calorific value of 3500-4600 kcal kg^{-1} (Grice, 2005). The charcoal from *A. mearnsii* is extensively used in Brazil and Kenya. In Indonesia the tree is extensively used as a domestic fuel and for curing tobacco (Albrecht, 1993).

The pulp productivity of *A. mearnsii* is about 320 kg/m^{-3} (DWAF, 1997). *Acacia mearnsii* is also used for rayon (Campbell, 2000). The wood from *A. mearnsii* is used for house poles, mine props, tool handles, cabinet work, joinery, flooring, construction timber and matchwood

(DWAF, 1997). *A. mearnsii* bark is the most widely used tannin material in the world (Grice, 2005). The species has been planted as a shelterbelt, a fire belt and as a shade tree in plantations (DWAF, 1997). *A. mearnsii* is an effective nitrogen fixer and has an annual yield of wet leaves of 21-25 t ha^{-1}, containing 240-285 kg of nitrogen (Smita, 1998), it thus can restore and regenerate soils (Duke, 1983).

A. mearnsii is an attractive tree, flowering profusely in late winter (Grice, 2005). In central Java and in Kenya, foliage is used as green manure to improve agricultural yield (Nyoka, 2003). Sawdust of *A. mearnsii* has been found to be an excellent medium for growing edible mushrooms in China. Poles with bark intact are used to support oyster racks in New South Wales (Lemmens *et al*., 1991). The species' attributes of a successful invader, such as a short juvenile period and a short interval between large seed crops, imply early and consistent reproduction (Rejmanek, 1995). The ability to attain positive population growth rates under strenuous conditions allows the invading species to increase in abundance while in the presence of numerically superior, well established residents (Seabloom *et al*., 2003).

A typical early successful invader like *A.mearnsii* is able to monopolize recently disturbed ground due to its rapid dispersal to the site (Grice, 2005). The invader should be able to have rapid growth and reproduction in the absence of significant competition for space and nutrients (Richardson, 1998). For a successful invader to alter community and ecosystem properties it must have an impact on energy, nutrient, or water flow, on the disturbance regime, or on the community response to the disturbance regime (Rouget *et al*., 1997).

To be successful, an invasive species must both survive and attain a positive rate of increase while living on the resources left unconsumed by the resident species (Seabloom *et al*., 2003). *Acacias* have been known to nodulate promiscuously with local bacteria and have mostly benefited from the resident biota (Smita, 1998). Invasive trees produce small seeds and have a short period between large seed crops, and all these are important factors for successful invasion (Luken and Thieret, 1997).

Positive effects of invasion

While the invasive plants have detrimental effects on the environment, economy and the ecosystem, the presence of the plants brings benefits as well (van Wilgen *et al*., 2001). The South African government has used the opportunities offered by the need to control and prevent the spread of *A.mearnsii* for labor-intensive clearing programs to generate a range of benefits (DWAF, 1997). This involves engaging unemployed people in labor-intensive clearing, follow-ups and rehabilitation projects aimed at bringing invasions of alien plants under control (DWAF, 1997).

The Working for Water Programme was started in October 1995, with the aim of protecting water resources and creating employment opportunities (van Wilgen *et al*., 2002). The program employs the poorest members of the communities settled closest to the alien infested areas, targeting women especially single mothers (DWAF, 1997). In 2001, South Africa exported 1.2 million tons of *A.mearnsii* wood product worth around R360 million (US$31.5 million) from 130,000 ha of managed plantations centered in the provinces of Mpumalanga and KwaZulu-

Natal in northeast South Africa, and from black wattle control programmes (van Wilgen *et al*., 2002).

Detrimental effects of invasive trees

When discussing the ecological impact of invasive plants, there are several known parameters quantifying the impacts of invasive tress and these include: range, abundance and effect per individual tree species (Scott, 2005). The effect of an invasive species on ecosystem processes is expected to depend on whether it differs functionally from native species or its requirements are lower than the native species (Vitousek *et al*., 1997). When a new species is introduced, its effects on nutrient cycling will depend on how different it is from the constellation of traits present within the existing plant community (Ehrenfeld, 2003). Invasive alien tree species cause both environmental and economic losses in South Africa (Bromilow, 1995).

Invasive alien plants affect the capacity of ecosystems to deliver goods and services (Guthrie, 2007), such as water purification, soil generation, decomposition and nutrient cycling (Vitousek 1986). The dynamics of invasive species may depend on their abilities to compete for resources and exploit disturbances relative to the abilities of native species (Seabloom *et al*., 2003). Invaded ecosystems often have altered species composition and community structure (Vitousek *et al*., 1997). The alien trees reduce groundwater recharge and alter sediment dynamics in stream and river courses, contributing to the silting of dams and converting perennial streams to seasonal ones (Enright, 1999).

If introduced or spread into habitats with closely related species, invasive species could interbreed with native species resulting in changes to the genetic makeup of either species through hybridization with local related genera (e.g. *Rubus* sp.), thus exchanging genes (Galatowitsch and Richardson, 2004). Exotic plant invasions affect the fluxes of a variety of other elements, including P cations, and metal pollutants (Ehrenfeld, 2003). The Tamarisk, a non native invader plant in California, has a very high salt content that affects the soil and native plants around it, showing the potential negative effects of invasive trees around native trees (Chornesky and Randall, 2003).

Thickets of alien plants species increase the fire hazard, therefore, increasing the costs of fire protection and the degree of damage caused by fires (Smita, 1998). Invasive plants have negative effects on all components of biological diversity, from genes to whole ecosystems (Henderson, 1998). In South Africa, invasive alien tree species and shrubs have negative effects such as reduction in stream flow and available water, loss of potentially productive land and loss of grazing potential in rangelands (Nyoka, 2003).

Invasive species also poison humans and livestock (e.g. *Melia azedarach* and *Lantana camara*), increase soil erosion following fires in heavily invaded areas, cause siltation of dams and change habitat suitability for native animal species (Binggeli *et al*., 1998). For instance in the Mitchell grasslands of northwest Queensland (Australia) *Acacia nilotica* reduces pasture production by 50% at 25-30% tree canopy cover or $2m^2$ basal area per hectare (Musil,1993). In the United States, the Office of Technology Assessment found that between 1906 and 1991, 79 non indigenous species caused documented losses of $97 billion (Binggeli *et al*., 1998).

The Office also estimated annual costs at several billion dollars, more in years when large-scale control programs were underway (Pimentel *et al*., 2000). Work on *Acacia melanoxylon* indicated that residues from the *Acacia* trees produce a strong inhibitory effect that is toxic and affects germination at the initial phases and the greatest effects are on the growth of a plant (Gordon, 1998). The tree releases phenolic compounds into the soil that inhibit growth and development of vegetation. The soils sampled from the *Acacia* tree showed strong toxic effects on radicle growth of *L. sativa* and on germination of *D. glomerata* (Gordon, 1998).

Most vascular plants form mycorrhizal associations with arbuscular mycorrhizal fungi (AMF), and many plants are highly dependent on this association for their growth and survival, particularly native woody perennials and others found in late-successional communities (Stinson *et al*., 2006). The invasive Garlic mustard targets arbuscular mycorrhizal fungi (AMF), which form mutually beneficial relationships with many forest trees (Levine *et al*., 2002). Exotic plants have been found to be poorer hosts and depend less on native AMF than native plants (Hicks, 2004).

The invasive trees release phytochemicals into soils as root exudates, as a result of damaged root tissue, or in the form of leaf litter (Callaway *et al*., 2003). *A. mearnsii* stabilizes sediments by colonizing deposits (for example banks) (Dye and Jarmain, 2004) but its shallow root system is easily washed out during floods, releasing sediments, blocking bridge arches and storm drainage systems (Surridge, 2006) as shown in Plate 1.

(a)   (b)

Plate 1: (a) *Acacia mearnsii* stands and (b) woody stubs of fallen trees near water path
(Photographer H.P.M. Moyo, 28 September 2007)

Invasion figures for Zimbabwe cannot be ascertained as yet, but potentially over 200 000 ha is affected, most of it in the mist belt of the eastern highlands (Nyoka, 2003). In South Africa and Zimbabwe, the species is the most important invader of riparian habitats, roadside and veldt, grasslands, forest edges and woodlands (Nyoka, 2003).

Invasive plant control and management

South Africa has one of the largest problems with invasive alien plants in the world (Richardson *et al*. 2005). In order to be successful, a control program against an invasive plant must take into account a wide range of extensive life history information about that plant (Campbell, 2000). An integrated approach involving the combined use of different control methods is necessary to manage invasive alien plants effectively (van Wilgen *et al*., 2000).

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Successful management of communities and ecosystems containing invasive species involves assessing whether the invaders have significantly altered the ecosystem from its pre-invasion condition (Campbell, 2000). It also involves recognizing and measuring specific community and ecosystem properties potentially being altered by the invader and developing strategies that return communities and the associated ecosystem processes to the pre-invasion state (Ward, 2004).

The control of *Acacia mearnsii*

The most common means for removing invasive *Acacias* from the fynbos is by mechanical clearing (Mugasi *et al*., 2000). This is typically done by individuals pulling up *A. mearnsii* seedlings by the roots and by chopping down the larger trees through the use of chain-saws (Plate 2). Mechanical clearing is the primary method of restoration used by the Working for Water project (DWAF, 1997). If alien plants are cut and left at the site, the large amount of dead biomass may result in more intense fires killing indigenous plant seed banks (Holmes and Cowling 1997).

(a)   (b)

Plate 2: (a) A Working for Water employee cutting *A. mearnsii* trees and (b) invasion by
    A. mearnsii (b) (Photographer H.P.M. Moyo, 28 September 2007)

Fire is used in the control of invasive trees (Campbell, 2000). This technique, however, is problematic because fire plays a vital role in the natural regeneration of both *A. mearnsii* and native fynbos species (DWAF, 1997). Fire is also the main disturbance factor that creates an "invasion window" that allows alien invasive trees to establish in the fynbos (Richardson, 1998). This is because fires may break the dormancy of the seeds and hence cause germination (van Wilgen *et al.*, 2001).

When using chemicals to control *A.mearnsii*, it is important that selective herbicides are used where grasses are present, and that diesel-based herbicides are not used along watercourses, so as to avoid contaminating the water (DWAF, 1997). Integrated control measures can be used in the control of invasive trees. Some examples of integrated control used on invasive alien species in South Africa include the control of *Hakea* species and *Opuntia stricta* (Guthrie, 2007).

Trees are felled and the cut stumps treated with a chemical herbicide or a mycoherbicide, a formulation of fungal spores that can be sprayed onto a plant causing fungal diseases) (DWAF, 1997). Biocontrol agents are then released on the regrowth, or on an adjacent biocontrol refuge area, from where they can colonize the cleared area when the invaders regrow or when the seeds germinate (DWAF, 1997). *Acacia mearnsii* is a difficult target for biological control (Campbell, 2000). The biological control of *Acacia mearnsii* species was initiated in 1973 (Campbell, 2000).

The programme targeted seed eating organisms that do not damage the vegetative parts of the *Acacia mearnsii* tree. Control programmes incorporate mechanical methods (felling and fire) and biological control (Guthrie, 2007). The plants are felled using chainsaws, and left for 12 to

18 months, then burnt (DWAF, 1997). Integrated control is used at times and it involves the use of herbicides on scattered populations, and biological control agents are released on larger infestations (Guthrie, 1997).

3. STUDY SITE

The field experiments were conducted in the Tsomo Valley near Elliot (31˚ 28′ South; 28˚ 3' 0'' E') at an altitude ranging between 1350 and 2000m above sea level. The dominant grass species is *Cymbopogon themeda*, occurring in summer. Other most prominent grass species include *Themeda triandra*, *Cymbopogon plurinodis* and *Eragrostis curvula.* The area is summer rainfall grassland which is invaded mostly by *Acacia mearnsii* and in some instances *Acacia karoo* (DWAF, 1997).

Located in the sourveld, growth is more rapid and regular than in sweetveld in this area due to annual rainfall which is about 750 mm and the rainy season usually extends from November to April (DWAF, 1997). This vegetation type provides good spring grazing (Danckwerts *et al.*, 1989), since livestock is the main form of farming taking place in this area. The larger part of the study area can be considered invaded as *A. mearnsii* stands are distributed densely in the area. The soils in the area are mainly sandy soils, although clayey loam soils derived from shale mudstone and sand stone can occur (DWAF, 1997). These soils extend to the entire surface of nearby catchments and to surrounding areas.

The Tsomo valley was selected because of the abundance of the *Acacia mearnsii* trees adjoining water bodies. The mountainous area is mainly dominated by *A. mearnsii* trees and these stretch further away from water bodies into the veld, with an average canopy height of 4 m. Temperatures vary between extremes during seasons with extremes of -8^0C recorded during the

winter season and 40^{0}C during the summer season, with an average of 17^{0}C during day time (DWAF, 1997). The grass layer mostly remains intact, except where severely overgrazed.

Although not densely distributed, the area is partly invaded by the Eucalyptus species that in some places are found integrated with *Acacia mearnsii*. Small thickets of *Acacia karoo* trees although not widely distributed, can be found in this area.

Figure 2. Map of the Eastern Cape showing the location of the study area, Elliot

4. EFFECT OF *ACACIA MEARNSII* REMOVAL ON THE WATER TABLE

Introduction

Invasive trees alter and utilize more water compared to indigenous trees because of their generally higher transpiration rates per unit leaf area (Enright, 1999). Recent climate-soil-vegetation modeling suggests that, given the same soil type between forested and bare soil conditions, forested soils have higher moisture losses (about 30% more) from evapotranspiration than bare soils (Zhang and Schilling, 2006). This results in forested soils producing less groundwater recharge than bare soils.

Introduced vegetation changes the surface characteristics of habitats through altering plant to plant interactions (Dye and Jarmain, 2004). They significantly influence soil water balance as they increase in dominance (Le Maitre *et al*., 2000); they further alter soil water balance through shifts in phonological schedules (Luken *et al*., 1997).

In semi-arid savanna ecosystems, the suppressive effect of an increase in woody plant density on herbaceous plants, mainly grasses, is largely through competition for soil water (Smit and Rethman, 1999). If a plant is introduced in an ecosystem, it will have seasonal pattern of canopy formation and physiological activity differing from the native species in the community (Enright, 1999). Such differences lead to degradation of ecosystem resources. Invasives are very competitive as shown by *Melaleuca quinquenervia* which is a very prolific rooter regardless of competing vegetation in the Netherlands (Lopez-Zamora *et al*., 2004). *M. quinquenervia*

develops root densities greater than many mature native species at an early age and in the soil surface during soil drying periods, even while competitive grasses are dying out (Lopez-Zamora *et al*., 2004).

Salt cedar (*Tamarix ramosissima*) is a great consumer of water in Russia; a single large plant can absorb 100 liters of water a day (Friedmanm, 2000). This results in the lowering of the ground water, drying up of springs and marshy areas, as well as reduction in water yield of riparian areas. Experiments aimed at assessing the effects of clearing on groundwater have not been adequately integrated with other components of the hydrological cycle in modeling of groundwater dynamics (DWAF, 1997).

The reasons for increased water use and whether such increases should be expected from all species of invading alien trees under all environmental conditions are not well understood (Calder and Dye, 2001). The few South African catchments and evaporation studies that have yielded water use data so far are too few to provide an adequate foundation for the countrywide estimation of evaporation in invaded regions (Calder and Dye, 2001).

The objective of the study was to quantify the water table fluctuations due to presence of *Acacia mearnsii*.

Experimental layout and data collection

The field experiment was a complete randomized design with two replicates. Two treatments were tested: (a) presence of *Acacia mearnsii* trees (b) absence of *Acacia mearnsii* trees. There were a total of four experimental units (plots) for the experiment. Four plots, measuring 20m x 10m each, were selected based on the presence of natural wells. The plots were largely dominated by clayey-loam soils. Clearing of *Acacia mearnsii* trees was done using chain-saws in September 2007 so as to start the recordings in November, when the rains start falling. In doing so a cyclic comparison in fluctuations would be obtained to compare between the wet and dry period. Infiltration rate, as affecting water seepage and water table level, was measured using the Double ring infiltrometer. Weather variables such as rainfall were monitored using a local weather.

In each of the four plots, a Data Logger (HOBO Pro U20-001-03, 250-Foot Data Logger, Onset , 2005) machine was installed at a depth of 90cm to estimate the changes in the level of water table. The Data loggers were measuring daily temperatures, atmospheric pressure and the change in water storage from the natural wells in ^{0}C, KPa and meters respectively, within a radius of 20 meters. The data loggers were set up to record the above parameters every 15 minutes and the data was retrieved from the machines at two week intervals and then averaged over a month.

Data analysis

All data sets were subjected to normality test to ascertain compatibility with assumptions of analysis of variance. The F test was conducted as appropriate for randomized complete design with the generalized linear model of SAS (SAS 1999). Correlation analysis was conducted to establish the level and direction of relationship between the weather variables with change in water storage using Pearson's Correlation coefficient. The data were split to wet and dry season. The equation adapted from Loheide *et al*., (2005), Dunne and Leopold (1978), and Dr. Georgianne, pers.comm'2008[1];

dS = (dW x **specific yield – R**) x **A**

Where change of water elevation (dW) represents the rise and fall in the level of the water table depth

dS= Change in storage

R=Rainfall

A=Watershed area

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[1] Dr. Georgianne W. Moore (Asst. Professor of Ecohydrology Ecosystem Science & Management, Texas A&M University)

Specific yield is the volume of water released from storage per unit land surface area per unit drop in the water table. The specific yield can be estimated based on sediment texture of the soil when the depth to the water table is above 1 m (Loheide *et al*., 2005). These values are based on the assumptions that the water table is deeper than 1 m, and that the readily available specific yield is essentially independent of the magnitude of the diurnal fluctuations and antecedent moisture conditions. The watershed area for Elliot was estimated from the digitalized elevation model.

Results

Rainfall during data collection constantly increased from November (2007) to February (2009), the highest amount received in February 2008 (Figure 3). Rainfall received decreased gradually from February 2008 to June 2008 (Figure 3).
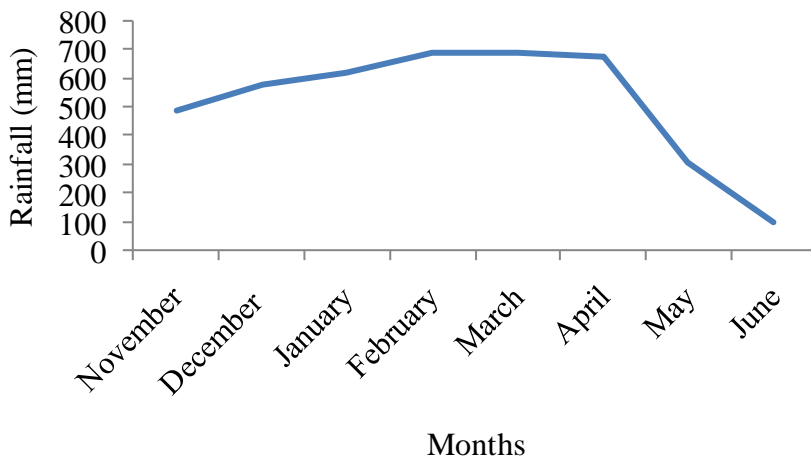
Figure 3. Rainfall amount received during data collection period (November 2007-August 2008)

In this study, the fluctuation of the change in water table storage from November (2007) to May (2009) did not vary significantly ($p > 0.05$) (Figure 3). The wet and the dry period recorded a significantly different ($p < 0.05$) change in water storage (-9.35 m and -8.8m respectively) (Figure 3) due to higher rainfall received during that period. The change in storage for the wet period showed a constant increase as the months progressed while a slight change in storage is seen between the first and third months only and, thereafter, a constant decrease occurred to the last month of the dry period (Figure 4).

The data showed that cleared plots averaged higher (-9 m) than not cleared plots (-8.7 m) during

the recording period (Figure 5). The fluctuations from the data loggers are clearer during the wet

period of data collection (Figure 4). A and D represent data loggers with fluctuations measured

in cleared plots while B and C represent data loggers with fluctuations measured in non cleared

plots.
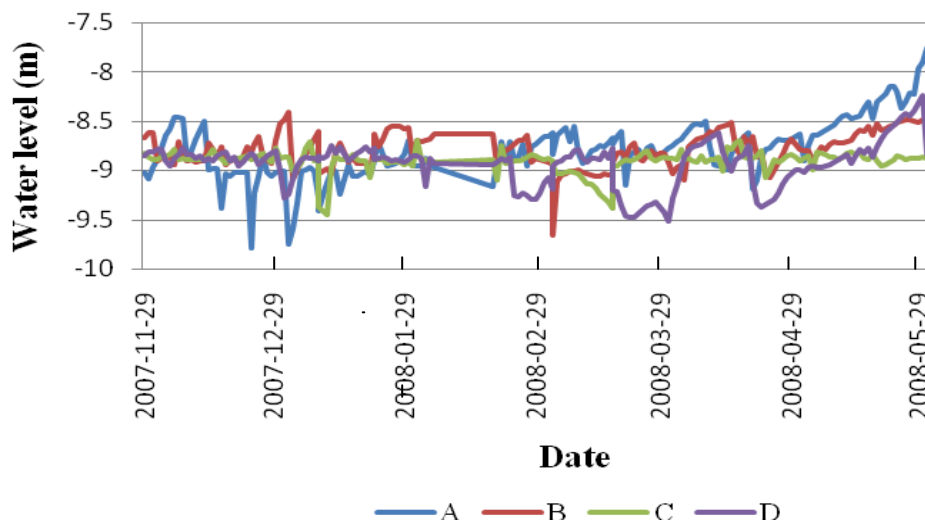
Figure 4. Water table fluctuations from the four data loggers

Water table fluctuations did not show significant differences ($p > 0.05$) when comparing the four

data loggers (Figure 5). Water table fluctuations are less defined during the initial months of the

wet period but the difference between the fluctuations is however clearly defined and visible as

the wet season approaches its end (February and March) (Figure 5). Water table recharge

increases in the cleared plots as the water fluctuates towards positive levels compared to the not

cleared plots (Figure 5). Differences between fluctuations are minimal during the initial stages of

the wet period when comparing the cleared and not cleared plots but become clearer as the

months progress, becoming clearer and, therefore, higher during the dry period (Figure 5).
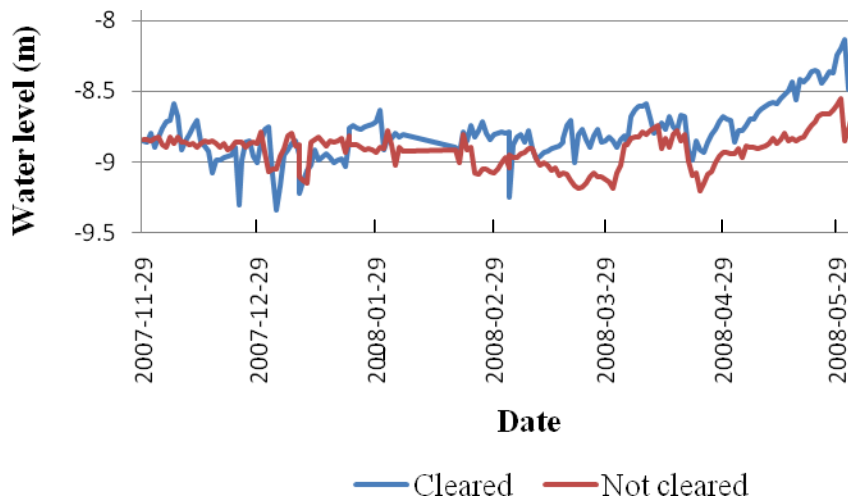
Figure 5. Effects of clearing *A. mearnsii* on water level fluctuations

The infiltration rate was not correlated to the change in water table storage. Infiltration rate was significantly and positively correlated to the change in water storage (r=0.56, p < 0.05) for the cleared plots while infiltration rate was insignificantly correlated (r=-0.93, p > 0.05) to the change in water storage in the not cleared plots.

Discussion

Acacia mearnsii caused a reduction in water table in plots not cleared as it has high biomass production that in turn leads to increased maintenance requirements for water and nutrients. Water table fluctuations in cleared plots increased water table recharge because clearing trees reduced water uptake by vegetation. These results are similar to Prinsloo and Scott (2002)'s in South Africa (Western Cape) where there was $12m^{-3}day^{-1}ha^{-1}$, $10.4 \ m^{-3}day^{-1}ha^{-1}$ and $8.8 \ m^{-3}day^{-1}ha^{-1}$ increase in stream flow after clearing *A. mearnsii* and *Acacia longifolia* in Knorhoek, Oaklands and Du Toitskloof in the Western Cape .

The invasive *A. mearnsii* reduced groundwater recharge through altering interception, infiltration, surface runoff, transpiration through the use of their deep rooting system. This also allowed *A. mearnsii* to utilize that water that would have ended up in rivers or streams instead. The presence of *A. mearnsii* affected water recharge by directly extracting groundwater from saturated strata and reducing the proportion of rainfall that eventually recharged by interfering with the passage of precipitation from the atmosphere to the water table in the soil.

A. mearnsii decreased water table recharge because it was extracting soil water in the unsaturated zone through its roots, to feed transpiration, thereby decreasing amount of percolating water that reached the saturated zone. Water tables can fluctuate considerably due to seasonal and annual changes in inflows of water and as expected the wet season had higher mean moisture water table recharge compared to the dry season due to the higher rainfall amounts received during the

wet season. Plant water use varies within and between days, as well as within and between seasons. Therefore such a variation between seasons was expected for this study.

Tree harvesting, like the plots that had *A. mearnsii* cleared, changes the rate of transpiration as leaf area is reduced causing reduced water use by the tree. Therefore clearing *A. mearnsii* reduced total leaf area thus increasing water storage in the soil. *A. mearnsii* reduced water storage where there were trees and this may have been due to that *A. mearnsii* increased the bulk density of the soil, thereby reducing soil porosity. The less the soil was porous, as expected, the low the water quantity that would percolate.

The water received through rainfall was higher in the wet season than that consumed by the trees leading to an increased recharge of the water table. The results of this study are similar to those found out by Khanzada *et al.*, (1998) when the water consumption by *Acacia nilotica* and *Acacia ampliceps* increased during the dry period of their study and was not significant in the wet season due to rainfall received. For most rainfall events there is a rise in water table level. Water infiltrates directly from the surface to the water table and rises in the water level, therefore, occur very quickly. Rainfall thus plays a critical part in evaluating the groundwater levels. In the summer when most of the rainfall events occur, the trees are in a growth period and the temperature is high the outflow will be positive. In winter months the contribution of these elements are minimal and it would be expected that with no, or very little, rainfall no outflow would occur but the fractured gneiss in the catchment slowly releases water into the system.

Infiltration rate was positively correlated to the change in water storage in the soil, most likely because a higher infiltration rate results in increased water percolation, thereby increasing the water table level. Infiltration rates from the cleared plots were higher than those from the not cleared plots due to the differences in soil porosity. The soil from the cleared plots was more porous as observed from its infiltration rate as compared to the not cleared plots. *Acacia mearnsii* bound the soil particles together in the plots not cleared thereby reducing the porosity of the soil.

A rise in water table increased the total biomass production in the plots that were cleared of *A. mearnsii* because clearing the plots reduced the amount of water loss through evapotranspiration and this water was instead utilized by the vegetation growing where *A. mearnsii* had previously grown. The basal cover also increased with an increase in water table recharge due to a reduction in water loss through evapotranspiration by trees, a similar observation to the total biomass observation. These results are similar to what Licata *et al*., (2008) found. Although working on a different species, they found the invasive Ponderosa pine (*Pinus ponderosa* Doug. ex. Laws) to have higher evapotranspiration rates than the native cordilleran cypress (*Austrocedrus chilensis* (D. Don) Pic. Serm. et Bizzarri).

Conclusion

A. mearnsii had a negative effect on water table recharge because the change in water storage was lower for the plots that had trees compared to the plots that did not have trees. The wet season recorded minimal water loss from the water table due to the rainfall received compared to the dry season. Invasion of rangelands by *A. mearnsii* therefore causes serious threats to vegetation development because it converts water meant for utilization by native vegetation to its own use. The depression of water tables through consumptive use of water by invasive trees like *A. mearnsii* can lead not only to a reduction in available water, but also to a corresponding decrease in species diversity.

5. EFFECT OF *ACACIA MEARNSII* ON VEGETATION GROWTH

Introduction

Alien vegetation affects the structure of plant communities (Orr *et al*., 2006). Resident native species displacement, a phenomenon where plant invaders reduce local diversity by displacing established species, occurs through competition for resources, allelopathic effects, or development of antagonistic soil microbial feedbacks (Yurkonis *et al*., 2005). Introduced vegetation impacts N availability through changing litter quantity and quality, rates of N_2-fixation, and N loss (Evans *et al*., 2001). Invasions decrease native vegetation growth, development and ecosystem stability by altering resource supply, changing trophic structure and relationships, or facilitating disturbances like fire through increased biomass production (Evans *et al*., 2001).

Invaders tend to outcompete plants in communities that they invade for resources such as moisture and nutrients (Drake *et al*., 1989). This competition is removed physically or chemically by humans, allowing human valued plants a much greater chance of surviving the changes in their environments. Effects of alien trees, like *Acacia mearnsii*, include changes in the diversity and relative abundance of native species, and alteration of conditions that would enhance succession and regeneration of communities over time (Orr *et al*., 2005). Invasive trees reduce colonization success through site saturation by invader propagules, which take up available germination sites, reducing establishment rates of other species (Binggeli, 1996).

Plant species introduced as a result of human activity for beneficial purposes are recognized as components of ecosystems that interrupt resource partitioning because of their invasive nature (D'Antonio *et al*., 1998). Alien vegetation, including *Acacia mearnsii,* form dense stands, maintain a high green leaf area throughout the year, replace seasonally dormant grasslands, permitting high rates of evapotranspiration (Dye and Jarmain, 2004).

Communal grazing areas are being converted to bushland by the encroaching trees like *A. mearnsii* which tends to exclude grasses and herbs (de Neergaard *et al*., 2005). The effects of invasive trees on community structure are difficult to measure, in part, because invasions often occur simultaneously with habitat change (D'Antonio *et al*., 1998). The extent of the impact of invasives on vegetation is not yet fully understood (Macdonald and van Wilgen, 2002). Policy makers, resource managers and researchers need information about how tree species impact vegetation structure and diversity (Macdonald and van Wilgen, 2002).

Despite the recent recognition of the invasion of rangelands by exotic species as a problem, there are many areas in the world where information on the impact of instability caused by invasion on ecosystem function is limited (Drake *et al*., 1989). Few field studies have addressed the impact and extent to which plants interfere with each other (DWAF, 1997).

The objective of this study was to investigate the effects of *A. mearnsii* on herbaceous biomass production, basal and aerial cover, and vegetation species composition in the Tsomo Valley, Eastern Cape, South Africa.

Experimental layout and data collection

The effect of *A. mearnsii* on vegetation growth was monitored on four plots, measuring 20m x 10m. Two of the plots were cleared of *A. mearnsii* and two of the plots were not cleared of *A. mearnsii*. In each plot four 1m^2 quadrats were located randomly. In each quadrat, percentage aerial cover was visually assessed and categorized into green coverage, dead plants, litter, rocks and bare cover. Herbaceous species were identified, clipped and classified into categories as a dominant class, sub-dominant classes, dicotyledons, legumes and species that were either unidentified or insignificant in terms of diversity, classified as other species. Composite samples of each category were separated for each plot, weighed fresh and oven dried for 48 hours at 60°C.

The dry mass of the dominant and subdominant classes of the grasses were weighed, calculated and converted to dry matter weight per hectare (kgDM-ha^{-1}). The basal cover was measured using the step point technique (Etchberger & Krausman, 1997). Fifty points were taken at an interval of half a meter. The presence of plants was recorded as the strikes on the base of the plants and the absence of plants recorded as bare and rock hits. Relative abundance of plants out of the total hits was used as percentage basal cover. Grass species were identified according to van Oudtshoon (1992). The data were collected in both the wet and dry seasons.

Data analysis

All data sets were subjected to normality test to ascertain compatibility with assumptions of analysis of variance. Descriptive statistics were used to obtain percentage species composition for herbaceous and woody vegetation. Since the proportion of the aerial cover (green cover, dead material and bare ground) was recorded as percentages, the data on proportions was also analyzed as percentages. The analysis of variance (ANOVA) with GLM procedure of SAS system (SAS Institute, 1999) was used to analyze the effect of treatment on herbaceous biomass, basal and aerial cover.

Where differences occurred, LSD mean separation was used to determine which means differed. The Pearson correlation coefficients were estimated to generate correlation coefficients for any relationship within the combinations of the vegetation variables, and for the correlation coefficient between the herbaceous biomass production and change in water table fluctuation.

The model for the effect of *A. mearnsii* on the vegetation growth was;

Herbaceous biomass production = $\mu + a_i + b_j + a_i b_j + e_{ij}$

Where μ = overall mean
a_i = treatment effect
b_j = season effect
$a_i b_j$ = season X treatment effect
e_{ij} = error term

Results

Herbaceous biomass production

The clearing of *A. mearnsii* had a significant effect ($p < 0.05$) on total herbaceous biomass production (Figure 6).
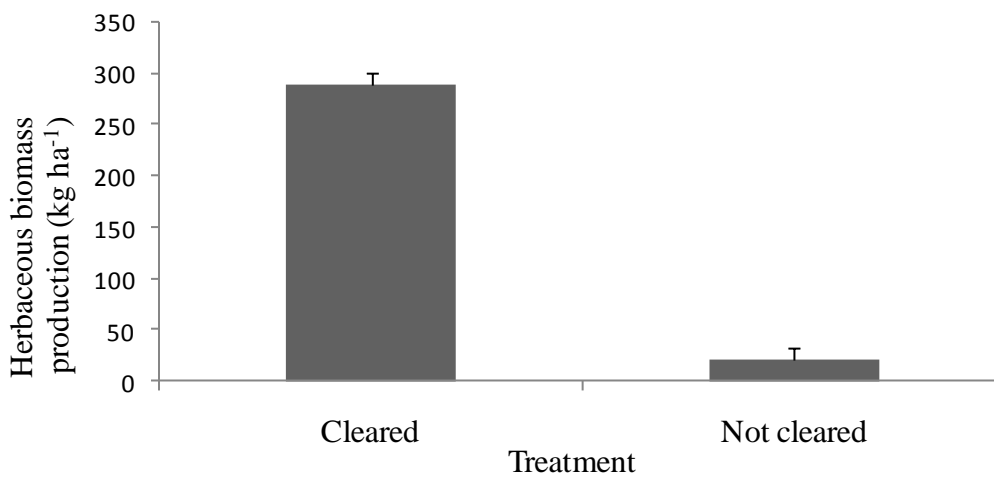
Figure 6. Effect of *A. mearnsii* clearing on total biomass production with standard error bars

The production in plots with no trees recorded significantly higher ($p < 0.05$) herbage biomass production (290 kg ha^{-1}) compared to the plots that had trees (22 kg ha^{-1}) for this study (Figure 6). The wet season had a higher total biomass production (240kg ha^{-1}) compared to the dry season (75kg ha^{-1}) (Figure 7).
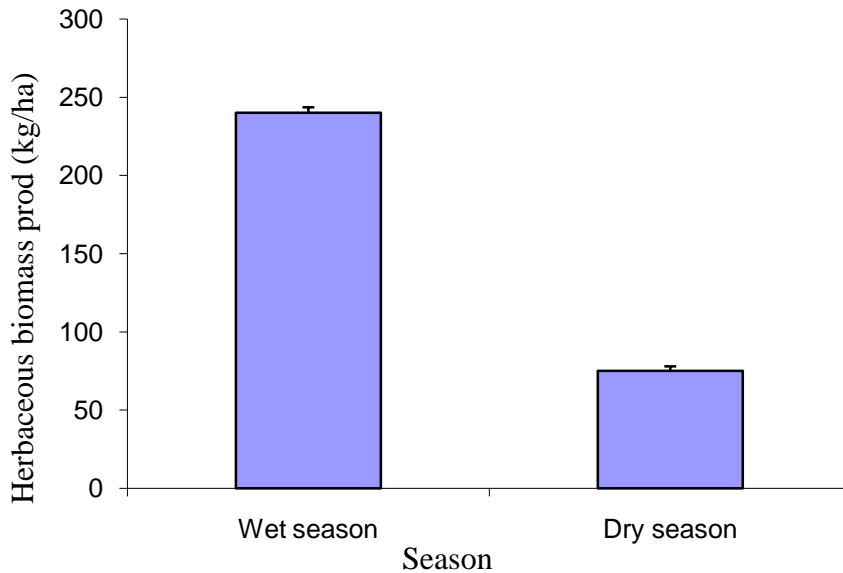
Figure 7. Herbaceous biomass production for the dry and wet season with standard error bars

Basal and aerial cover

The season significantly affected ($p < 0.05$) the basal cover. The wet season had a higher BC (10.1 m² ha^{-1}) compared to the dry season (4 m² ha^{-1}).The plots that did not have trees also had a BC higher (14 m² ha^{-1}) than that of the plots that had trees (1.5 m² ha^{-1}).

The difference in basal cover in the cleared plots for the wet (22.94 m² ha^{-1}) and the dry season (14.93 m² ha^{-1}) was significantly higher ($p < 0.05$) while the difference in the plots that had trees for both the wet and dry season was not significantly different.

The proportion of dead plant material, bare ground and litter material were significantly different for both the plot and the season ($p < 0.05$). The amount of green vegetation was not significantly

different (p > 0.05) when compared in the wet and dry periods, and also in the cleared and not cleared plots. Cleared plots had 48% dead whereas plots not cleared had 52% dead. The wet season had (70%) green coverage compared to 30% for the dry season.

The dominant grass species (*Cynodon dactylon*) contributed 50% of the species composition whereas *Eulelia velosa* contributed 25% and *Panicum equiniva* 17% from all the plots. Herbaceous species composition did not change significantly in the plots during data collection as grass species dominance was maintained in all the plots.

Discussion

Invading species can affect the trajectory of invaded vegetation by altering natural disturbance regimes and reducing native species diversity (Yurkonis *et al*., 2005). Cleared plots had higher basal cover than plots that had trees in the wet and dry season for this study. This implies that the not cleared plots had higher plant-inter-space, which indicates a level of degradation due to *A. mearnsii.* The lower basal cover was due to the fact that *A. mearnsii* blocked and, therefore, reduced the amount of sun reaching the floor; negatively affecting the growth and development of understory vegetation. Invasive trees affect plant communities by altering resource dynamics such as sunlight and interspecific interactions. Although not significantly different, green coverage was higher where *A. mearnsii* was cleared. Green coverage was also higher for the wet season compared to the dry season.

The dead plant material varied due to clearing and not clearing *A. mearnsii* with the wet season recording higher percentages. This could be due to the higher total biomass production. Herbaceous biomass production was significantly higher in cleared plots because non-clearing of *A. mearnsii* reduced herbaceous biomass production. *A. mearnsii* grows to a height of approximately 10m blocking and reducing light availability to growing understory vegetation through its shade effect. Reduced light therefore led to slow and reduced growth of herbaceous vegetation. High woody plant abundance, as was the case for the current study and also found out by Abdallah *et al*., (2007), leads to the suppression of herbaceous vegetation cover.

Working with *Acacia tortilis* Abdallah *et al*., (2007) found similar results; *Acacia tortilis* reduced total grass cover and grass yields compared to outside the canopy. Competition for light is often cited as the primary mechanism for change following tree invasion, resulting in shading of native plants by invasive shrubs. *A. mearnsii* either intercepted rainfall, therefore reducing infiltration, or absorbed more water thereby reducing available moisture for use by herbaceous vegetation.

A rise in water table increased the total biomass production in the plots that were cleared of *A. mearnsii.* Clearing plots reduced the amount of water loss through evapotranspiration making more water available for vegetation growth

Conclusion

Clearing *A. mearnsii* increased herbaceous biomass production. For rangeland managers, *A. mearnsii* is not desirable because it reduces grazing capacity of the area available to animals and measures need to be taken to reduce its impact. Where ecosystem properties have been altered by non-indigenous species like *A. mearnsii*, the control of the species and restoration of ecosystem functions are necessary if the extent of invasion impacts are to be reduced.

6. EFFECTS OF *ACACIA MEARNSII* ON SOIL PHYSICAL AND CHEMICAL PROPERTIES

Introduction

Plant species strongly influence the rate at which nutrients cycle within an ecosystem through litter-quality feedbacks that differ from native vegetation (Dukes *et al*., 2004; Ehrenfeld, 2003). Compared to the adjacent open areas, soils under tree canopies often have higher concentrations of organic matter, available nitrogen and exchangeable cations and also have a better physical structure and improved water infiltration (Abule *et al*., 2004).

Alien vegetation affects soil properties by producing organic acids in soil solution derived from the decomposition of litter and organic matter from leaf leaching and washing (Dijkstra *et al*., 2001). Invasive plants suppress growth of other plants through allelochemicals released as exudates from leaves, stems or roots (El Khawas and Shehata, 2005). Shifts in plant community composition resulting from exotic invasions are as likely to be associated with changes in soil properties as those associated with natural successional processes (Ehrenfeld *et al*., 2001).

Introduced plants affect soil nutrient dynamics by differing from native species in tissue chemistry, plant morphology and phenology (Ehrenfeld, 2003). Soil properties such as pH and the amount of base cations available in the soil differ greatly under trees of different species and when invading trees are compared with native vegetation (Dijkstra *et al*., 2001). These differences, whilst reflecting variations in soil parent material, can also be caused by differences

among tree species and their deposits. Invasions decrease ecosystem stability by altering

resource supply, changing trophic structure and relationships, and suppressing native vegetation

growth (Vitousek, 1986). Soil nitrogen is one of the most important factors that determine

ecosystem stability. The introduction of a single species can have large effects on rates of

nitrogen transformations by changing litter quality and quantity (Scott, 2005).

The structure and functional diversity of microbial communities in the soil is closely related to

plant species composition above-ground. This relationship, therefore, provides an important link

between above and below-ground processes in terrestrial ecosystems (Kourtev *et al*., 2003). In

order to demonstrate that the exotics cause changes in soil, rather than preferentially occupying

microsites with different soil qualities, it is necessary to demonstrate that soil properties change

with presence of invasive plant (Ehrenfeld *et al*., 2001). Mechanisms have been identified by

which plants can alter the physical, chemical, and biological properties of soils. These involve

changes in the quantity, quality, and/or timing of inputs of plant-derived substrates.

Changes in properties of the soil also occur due to variations in microclimate associated with

changes in the density and height of the vegetation (Ehrenfeld *et al*., 2001). Variations in nutrient

cycling processes that arise due to changes in vegetation reflect alterations in the soil microbial

community (Kourtev *et al*., 2003). Although invasions by exotic plant species represent a major

threat to biodiversity and ecosystem stability, little attention has been paid to the potential

impacts of these invasions on nutrient cycling processes in the soil (Ehrenfeld, 2003).

The objective of this study was to investigate effects of *A. mearnsii* on soil chemical (nitrogen, electrical conductivity, soil organic carbon and pH) and physical properties (soil fast and slow wetting, and mechanical disintegration).

Experimental layout and data collection

Soil sampling was done from invaded and uninvaded areas. Five *A. mearnsii* invaded areas measuring approximately 5mX10m were chosen and paired with five uninvaded areas with the same area. Invaded areas consisted of *A. mearnsii* trees only and these were heavily invaded areas for at least the past five years, based on available farm records. The uninvaded sites were mostly dominated by grass species in the Tsomo Valley. In each of the areas chosen, soil samples were collected using a soil auger in the 0-15cm horizon, mixed to obtain a composite sample and a sub-sample taken for laboratory analyses. A total of 10 soil samples were taken from same soil type, clayey loam. Sampling was done in flat areas so as to remove the slope effect. The samples were transported in a rigid box to avoid breakage of aggregates.

Samples were oven dried at 60°C for 48 hours to unify the sample moisture condition. Thereafter subsamples were subjected to (a) fast wetting, by immersion in water, (b) mechanical disaggregation, by shaking and (c) slow wetting, by capillary action (Le Bissonnais, 1996). The aggregates were then oven dried after ethanol treatment and sieved through six mesh sizes, that is; 2000 µm, 1000 µm, 500 µm, 200 µm, 100 µm and 50 µm. The mean weight diameter was calculated for each sampled site and interpreted based on classification table by Le Bissonnais

(1996). Soil pH was measured on 2:5:1 water to soil ratio suspension as described by Okalebo *et al*., (2002) on the electrode pH-meter for a saturated soil paste using distilled water.

Total nitrogen was determined by digestion as described by Okalebo *et al*., (2002).The electrical conductivity (EC) was measured in the saturated paste extract to determine the level of salinity (Okalebo *et al*., 2002). After resting it for one hour, soil particle density was determined using the hydrometer method. Soil organic carbon content was determined by the sulphuric acid and aqueous potassium dichromate ($K_2 Cr_2 O_7$) mixture (Anderson and Ingram, 1993). After complete oxidation from the heat of solution and external heating, the residual $K_2 Cr_2 O_7$ was titrated against ferrous ammonium sulphate (Anderson and Ingram, 1993).

Data Analyses

For the test of differences in the soil between the invaded and uninvaded area, a randomized complete design (RCD) was implemented and the data was analyzed using T-test of SAS (10) (SAS, 1999). The Pearson's correlation was used to estimate all possible paired combinations of the variables to obtain correlation coefficients for their relationship. Least significant differences (LSDs) were also calculated from the appropriate standard errors (SE).

The model for the effect of *A.mearnsii* on soil physical and chemical properties;

$$Y = \mu + a_i + e_{ij}$$

Where μ= overall mean
    a_i= treatment effect
    e_i= error term

Results

A. mearnsii had no significant effect ($p > 0.05$) on soil electrical conductivity (EC), however, the soils from under the canopies had higher EC (5.48 uScm^{-1}) compared to those from outside canopy stands EC (5.18 uScm^{-1}) (Table 3). Soil pH and carbon were significantly different ($p < 0.05$) in the soils collected from under and outside canopies (Table 3). Soil pH was acidic (3.1) on soils collected from under canopies while soil from outside tree canopies was less acidic (5.5) (Table 3). Soil nitrogen content was significantly different ($p < 0.05$) in soils from under tree canopies than from outside, with soils from under the canopies higher (0.52) in nitrogen content compared to soils from outside the canopies (0.2) (Table 3).

Table 3; Soil chemical properties between the invaded and the non-invaded sites

Site	Invaded	Open land
EC (uScm^{-1})	5.18[a] ±0.084	5.48[a] ±0.094
pH (H$_2$O)	3.05[a] ±0.58	5.47[b] ±0.65
Soil organic carbon (%)	1.9[a] ±0.1	1.48[b] ±0.1
N (mg kg^{-1})	0.52[a] ±0.02	0.21[b] ±0.01

[a, b] means with different superscript within the same row (basal area) are significantly different ($p < 0.05$).

Soil aggregate stability was not significantly different ($p > 0.05$) for soils collected from under the canopies and from outside the canopies.
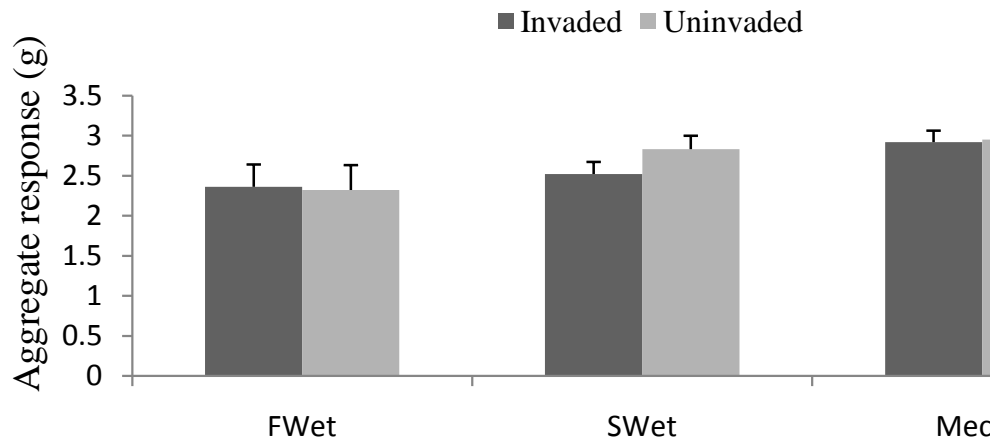
Figure 8. Effect of invasion on soil physical properties (soil aggregate stability) with standard error bars

There was negative correlation between EC and N (r= -0.72, p=0.0283), negative correlation between pH and N (r= -0.718, p=0.0295) and negative correlation between organic carbon and pH (r= -0.786, p=0.0121).

Discussion

This research was designed to examine the effects of *A. mearnsii* on the soil chemical and physical properties because in arid and semi-arid zones, invading trees have negative or positive effects on their immediate environment (Abdallah *et al*., 2007). The results show that soil collected from inside and outside tree canopies had no effect on electrical conductivity. This implies that the trees did not alter the salinity concentration and distribution in the sites where sampling was done. Plant invasions do not always result in consistent and at times significant changes in soil properties. As was the case with the results from this study, Abdallah *et al*., (2007) also found that the EC was not different from under the canopy and outside the canopy.

Electrical conductivity did not differ because the salt content was low for both areas and no fertilizers were applied. These results are different from what Abule *et al*., (2005) found on soils under *Acacia tortilis*. Abule *et al*., 2005 found that electrical conductivity was significantly higher under tree canopies compared to open grasslands.

Aggregate stability is one of the main factors controlling top soil hydrology and erodibility as the aggregates and pores between them affect the movement, aeration and storage of water. In this study, *A. mearnsii* invasion had no significant effect on soil aggregate stability with soils from outside and inside canopies showing similar responses. *A. mearnsii* invasion, therefore, neither improves soil aeration nor increases occurrences of soil erosion. The soil aggregate stability was not significantly different in invaded and non-invaded areas because there was no form of

disturbance in the different areas in the form of cultivation or ploughing hence the expected outcome.

Organic carbon was higher in soils from under the canopies than from outside the canopies of *A. mearnsii*. Such an observation can be attributed to *A. mearnsii* litter because it is a key factor influencing nutrient cycling rates as it affects the rate of its disintegration. The poor the litter quality the slower the disintegration and the lower the organic carbon content of the soil. Increases in OC under trees compared to adjacent bare or grassy ground is common and since soils from under *A. mearnsii* had higher OC, such an increase could be attributed to the higher organic matter deposits resulting from higher vegetative production from *A.mearnsii*.

Working with *Acacia tortilis,* Abdallah *et al*., (2007) found similar results where soils from under *Acacia tortilis* canopies had higher organic matter deposits compared to soils collected from outside the canopies. Merchante *et al*., (2008) also found higher organic carbon under *A.longifolia* canopies. Differences between invasive and native trees in soil organic carbon levels may result from invasive trees that have higher biomass quantity compared to native trees, leading to higher organic matter deposition. As a leguminous tree, therefore, a nitrogen fixer, *A. mearnsii* had an effect on nitrogen content of the soil with soils from under tree canopies having higher N values than soil from outside canopies.

A. mearnsii produces copious amounts of litter with higher N content, which leads to higher nutrient and C content as found by Merchante *et al*., (2008), working with, *A.longifolia*. The

nitrogen inputs can possibly be through nitrogen fixation or nitrification through decomposition of *A. mearnsii* litter.

Unlike most of the low pH values (3.7) under this study, Abdallah *et al*., (2007) found that the pH under the tree canopy was higher (7.7) than that from inside the canopy (2.9). The mechanisms for decreases in pH in response to exotic invasion are due to increased nitrification, high rates of uptake of NH^+_4, and/or changes in litter quality. Hagos and Smit (2004) found similar results. In their study *Acacia mellifera* was shown to decrease pH as distance towards the tree canopy was reduced.

Based on the positive association between increases in exchangeable cations and soil pH (higher base saturation) the lower pH under canopies of *A. mearnsii* relates to the lower content of exchangeable cations in this sub-habitat. Witkowski (1991) found pH values not statistically different when soil samples from under *Acacia cyclops* and *A. saligna* canopies were tested and compared with soils from outside their canopies. Musil (1993) also found no significant differences in pH when soil from under *Acacia saligna* was compared with soil from out of stands. The presence of *A. mearnsii* had a significant effect on soil organic carbon. Exotic plants most commonly affect net primary productivity, carbon acquisition, by differing from native species in overall size, morphology, phenology, or growth rate.

Soil changes may generate a positive feedback that favours the invasive species and complicate restoration strategies. This affects vegetation development and could be detrimental to grazing animals on the farm as the grazing capacity is reduced.

Conclusion

A. mearnsii had an effect on soil chemical properties; pH, soil organic carbon and nitrogen. It can therefore be concluded that *A. mearnsii* reduces pH whilst increasing soil organic carbon and nitrogen. If *A. mearnsii* reduces grass growth and development through depositing exudates, control measures to reduce tree population have to be taken and these can include mechanical removal and chemical application.

7. COMPOSITION OF VEGETATION SEED BANK UNDER *ACACIA MEARSNII* STANDS

Introduction

A soil seed bank is defined as the mature viable seed stock existing in the soil surface, or buried in the soil, duff or litter, in a determinate moment and place (Martins and Engel, 2007). Knowledge about soil seed banks and remaining vegetation following removal of forest ecosystems is crucial in guiding restoration efforts (Zhan *et al*., 2006; Eycott *et al*., 2006). Seed banks sustain populations during temporarily unfavorable conditions and allow establishment in new areas (Martins and Engel, 2007). A persistent seed bank remains in the soil for more than one season, and is a common strategy in areas where probability of adult survival and reproduction may vary temporally. A general decline with increasing age in seed bank density has been recorded for invaded areas (Holmes and Cowling, 1997).

Plant invaders directly or indirectly affect ecosystem resilience through their impact on soil seed banks (Mason *et al*., 2007). Soil seed banks are important in savanna ecosystems where grasses are a large part of the vegetation; it can reduce the probability of population extinction. The seed bank is a major source in establishing aboveground plant communities following environmental and ecosystem changes such as alien plant invasions (Solomon *et al*., 2005).

Since a soil seed bank has the potential to strongly influence community composition, especially during times of environmental fluctuation or disturbance, in forests, it is an important component

to consider when rehabilitating a previously invaded area (Lemenih and Teketay, 2006; Bossuyt and Hermy, 2001).

Species richness and abundance in seed banks provide information on the potential of a community to regenerate as it reflects the richness of species present in the local vegetation and in the immediate vicinity (Oke *et al.,* 2007). Fewer studies have looked at the effect of invasive species on the soil seed banks in grassy fynbos communities of the Eastern Cape (Fourie, 2008).

The objectives of this study were to investigate the extent to which the soil seed bank is affected by the presence of *A. mearnsii* and to investigate the potential contribution of the buried seed bank in determining species richness of the ground flora after removal of *Acacia mearnsii*.

Experimental layout and data collection

A soil seed bank is defined as seeds at or beneath the soil surface that are capable of germination (Solomon *et al*., 2005). A total of 40 soil samples were collected for the study making two treatments; 20 soil samples were randomly collected using an 8 cm diameter soil auger at 5 cm depth from directly under *A. mearnsii* canopies and another 20 samples from outside *A. mearnsii* canopies. The 5cm depth was used because seed abundance and diversity is highest in this layer; this layer represents the greatest soil depth from which many species can successfully emerge (Dalling and Denslow, 1998).

Sampling was done once at the end of September 2007. This sampling period represents the start of the growing period after seed production event. This can serve as indication of viable seeds not germinated in the field over the season. Sand for use in the study was sterilized through soaking in HCl for 24 hrs and rinsing in distilled water. Before use the sterile sand was checked for possible seed contamination. This was done by keeping the sterile sand moist over a clean flat floor and examining it regularly for germinating seeds over a 3-week period in the greenhouse. Labeled plastic pots with a depth of 5 cm and a diameter of 20 mm were filled with sterile sand to a depth of approximately 20 mm for the greenhouse study.

Soil samples, weighing 850 g were spread over the sand in each plastic pot to a depth of 20 mm. The pots were placed in the greenhouse based on soil sampled from canopy and outside canopies. The temperature in the greenhouse was kept between 19–22 ^{0}C during the day and 10–12 ^{0}C during the night throughout the experimental period so as to provide and maintain conditions that stimulate germination.

Seedling emergence from the pots was scored weekly and species identification according to van Oudtshoon (1992), where possible, done immediately. Any plant that could not be identified at the seedling stage was allowed to grow until identification was possible. Every week, the soil samples were stirred to stimulate seed germination. Color-coded and labeled sticks marked new emergent seedlings in each tray.

Grass species were identified according to (Danckwerts and Teague, 1989). Each pot was hand-watered twice daily, in the morning and the afternoon. To monitor contamination from seeds

dispersed within the greenhouse, four control trays containing only seed free soil were placed in each replicate. Seed densities were estimated from the number of seedlings germinated from the soil over a six-week period. 'Dominant species' were defined as those whose populations account for at least 5% of total seed bank community. Data collection lasted for a period of 6 weeks.

Data analyses

The data was collected weekly and analyzed using T-test for SAS version 10 (SAS, 1999). Total germination for the whole duration of the study was obtained for the two sites.

The model for the effect of *A. mearnsii* on composition of vegetation seed bank was;

$$Y = \mu + a_i + e_{ij}$$

Where Y= composition of seed bank

μ= overall mean

a_i = treatment effect

e_{ij}= error term

Results

No seedling emergence occurred in the control trays indicating that there were no airborne seed contaminants. Unlike in soils from uninvaded sites there was no seedling emergence in the first week of the study for soils from invaded site. Seedling emergence started only in the second week of the study.

Soils collected from outside *A. mearnsii* canopies had significantly higher (p < 0.05) seedling emergence (50) compared to soils from inside canopies (18) from the 1ˢᵗ to the 3ʳᵈ week. Soils collected from inside canopies also had lower emergence (34) compared to soils from outside canopies (68).

Overall *A. mearnsii* had a significant effect (p<0.05) on vegetation emergence as trays with soil from outside *A. mearnsii* canopies recorded higher seedling emergence (113) compared to 80 for soils from under tree canopies. The mean emergence for soils from outside tree stands was higher (5.6) compared to the mean emergence for soils from inside tree canopies (2.7). There was no correlation between invasion and the total emergence for this study. Emergence increased from week 1 to 6, with soils from outside tree canopies having higher plant emergence compared to the invaded site (Figure 8).
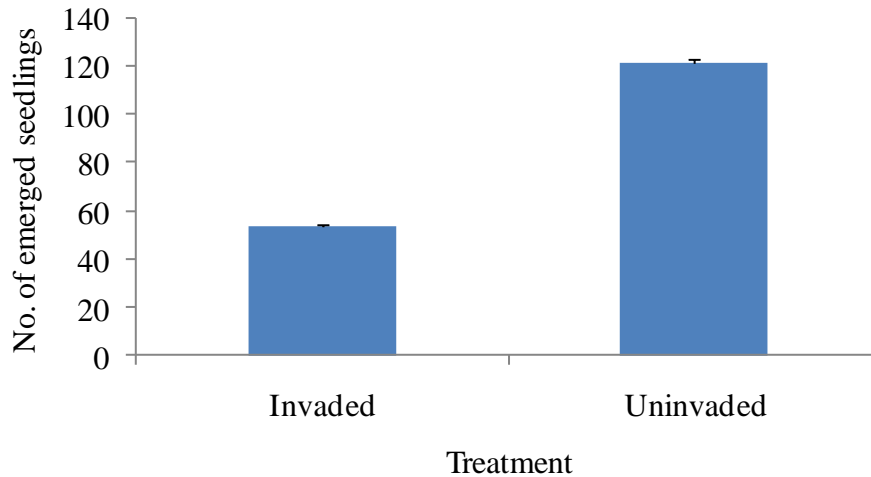
Figure 9. Seedling emergence for the whole study period (Week 1 to 6)

Seed bank botanical composition

A total of 193 seedlings emerged in the greenhouse, with a total of 9 species representing both grass and non-grass plant species and one seedling identified under the *Aristida* family (Table 4). Soils collected from outside tree canopies had significantly higher emergence ($p < 0.05$) (113 seedlings) that emerged while trays with soils from inside canopies had 80 seedlings that emerged in total for study (Table 4). Forbs dominated the flora, contributing 50% of the total seedling emergence compared to the woody and grass vegetation (Table 4). From the total seedlings that emerged, *Acacia mearnsii* contributed 5% of the total seedling emergence (Table 4). Species composition from the invaded site was dominated by *A. mearnsii* and forbs, representing the woody and the weedy plants. *A. mearnsii* was the only woody plant that emerged from all the trays (Table 4).

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Table 7: Botanical composition of the soil seed bank under invaded and uninvaded sites

Species	Uninvaded	Invaded
Cynodon dactylon	11	11
A.mearnsii	0	17
Forbs	31	28
Sporobolus africunus	16	9
Chloris vigata	16	0
Eragrostis chloromelus	5	0
Eragrostis plana	7	6
Cyprus spp	3	9
Aristida Spp	3	0
Aristida congesta	12	0
Panicum equiniva	9	0
Total	113	80

Discussion

In this study pots with soils from the invaded canopies had lower diversity and density compared to the trays from the uninvaded stands. This could be because under invasive tree stands, like under *A. mearnsii*, tree tops form a barrier that block the sun. This causes a reduction in the incidence of wavelength radiation of the red light beam which has a strong inhibition effect on seed germination of certain vegetation species. Seeds then remain dormant in the soil for a long period, and as soon as they are stimulated by light and/or high temperature, they germinate. Pots that contained soils from under 'uninvaded' stands had no *A. mearnsii* emergence. This can be attributed to the fact that seed dispersal of dry tropical woody species are limited to very short distances from forest edges when their sizes are large enough to limit dispersability. This causes seed inputs from the woody *A. mearnsii* to areas further from the stands to be minimal.

The herbaceous vegetation dominated the seed bank (>50%) in the two locations. Under the canopied stands such an observation could be due to the openings in the forest canopy where sampling was done which enhances the dispersal of seeds of these herbaceous species to plots. This result agrees with that by Oke *et al*., (2006) where herbaceous vegetation contributed 96% of the total seed densities of the seed bank in a secondary lowland rainforest in Nigeria.

Lack of emergence of any other woody tree species other than *A. mearnsii* from all the trays is due to low seed production or the lack of definitive dormancy mechanism in most woody species. The lack of emergence is further due to the inability of the trees to co-exist and compete with *A. mearnsii* for soil nutrients and water. A specific set of micro-environmental conditions must also be specifically satisfied for the successful seed germination and establishment of the seed bank of native vegetation.

The relatively low abundance of the grasses in soils from under the invaded stands is attributed to their inability to form persistent seed banks and to their low turnover of seed. For a species with low persistence, availability of seeds in the seed bank depends on seed production by the established population in the preceding year. The inability to form persistent seed banks may render grass species, such as the dominant *Sporobolus africunus*, vulnerable due to the low level of seedling recruitment.

The seedling emergence method has been used in many studies because of easy identification of emerging seedlings and presumes that number of seedlings detected equals the number of viable seeds. This method, however, can underestimate the true seed bank, as vital germination cues,

e.g. a heat pulse and fluctuation of soil temperatures may be absent (Oke *et al*., 2006). Favourable weather has been shown to be a critical factor in the germination of seeds stored in the seed bank (Oladipo and Oke, 2007).

Conclusion

A. mearnsii had a significant effect on soil seed bank composition. The seed bank was dominated by forbs for both soils from invaded and uninvaded sites whereas *A. mearnsii* dominated the invaded sites. Although viable persistent seed banks can be found in invaded sites, alien invasion results in a decrease in species richness, diversity and abundance. It is suggested that *A. mearnsii* blocks the sun hence preventing the establishment of vegetation. The management response to the suppression of light is to cut trees, with an expectation that the light distribution may thus be restored, allowing a re-establishment of the competitive status in the woodland understory.

8. GENERAL DISCUSSION

Invasive alien plants have significant impacts on native ecosystems as some of the species alter the composition and structure of vegetation: those species that replace indigenous vegetation over substantial areas often alter ecosystem processes (Richardson *et al.,* 2000). Invasive trees can increase soil erosion if the natural ground cover is excluded by invading trees and invasive trees have increased transpiration rates which lead to increased water use and to a reduction in water flows in streams and rivers (Le Maitre *et al*., 1996, Enright, 1999). Some alien species alter litter quantity and quality (Witkowski, 1991). To design sustainable management strategies for *A*. *mearnsii* in rangelands and to make the necessary recommendations, it is crucial to understand the levels of impacts of the species and to what extent it alters ecosystem function.

In Chapter 4, the effect of *A. mearnsii* removal on water table fluctuations was tested. The wet and the dry period recorded a significantly different change in water storage. Water table fluctuations did not show significant differences when comparing the four data loggers. Cleared plots had higher water table fluctuations than the not cleared plots. Water table recharge increases in the cleared plots as the water fluctuates towards positive levels compared to the not cleared plots. The invasive *A. mearnsii* reduced groundwater recharge through altering interception, infiltration, surface runoff, transpiration through the use of their deep rooting system. This also allowed *A. mearnsii* to utilize that water that would have ended up in rivers or streams instead. The presence of *A. mearnsii* affected water recharge by directly extracting groundwater from saturated strata and reducing the proportion of rainfall that eventually

recharged by interfering with the passage of precipitation from the atmosphere to the water table in the soil.

In Chapter 5, the effect of *A. mearnsii* removal on vegetation growth was tested through monitoring herbaceous biomass production. The clearing of *A. mearnsii* had a significant effect ($p < 0.05$) on total herbaceous biomass production. The production in plots with no trees recorded significantly higher ($p < 0.05$) herbage biomass production (290 kg ha^{-1}) compared to the plots that had trees (22 kg ha^{-1}) for this study. *A. mearnsii* grows to a height of approximately 10m blocking and reducing light availability to growing understory vegetation through its shade effect. Reduced light therefore led to slow and reduced growth of herbaceous vegetation.

To further ascertain whether reduced growth of herbaceous vegetation was caused by *A. mearnsii* or due to exudates from *A. mearnsii*, a follow up study was done on the soil properties collected from under this species' canopy and from outside *A. mearnsii* canopies. *A. mearnsii* had no significant effect on soil electrical conductivity. Soil pH and carbon were significantly different in the soils collected from under and outside canopies. Soil pH was acidic on soils collected from under canopies while soil from outside tree canopies was basic. Soil nitrogen content was significant different in soils from under tree canopies than from outside, with soils from under the canopies higher in nitrogen content compared to soils from outside the canopies. Soil aggregate stability was not significantly different for soils collected from under the canopies and from outside the canopies. Electrical conductivity did not differ because the salt content was low for both areas and no fertilizers were applied. The soil aggregate stability was not significantly different in invaded and non-invaded areas because there was no form of disturbance in the

different areas in the form of cultivation or ploughing. Soils from under *A. mearnsii* had higher organic carbon; such an increase could be attributed to the higher organic matter deposits resulting from higher vegetative production from *A. mearnsii*.

In Chapter 7 the effect of *A. mearnsii* on soil seed bank composition was monitored through incubating soil samples in the greenhouse. *A. mearnsii* had a significant effect on vegetation emergence as trays with soil from outside *A. mearnsii* canopies recorded higher seedling emergence compared to soils from under tree canopies. The mean emergence for soils from outside tree stands was higher compared to the mean emergence for soils from inside tree canopies. There was no correlation between invasion and the total emergence for this study. Forbs dominated the flora compared to the woody and grass vegetation. From the total seedlings that emerged, *A. mearnsii* contributed 5% of the total seedling emergence. Species composition from the invaded site was dominated by *A. mearnsii* and forbs indicating that *A. mearnsii* had higher seed production compared to any other woody plants. *A. mearnsii* was the only woody plant that emerged from all the trays.

CONCLUSIONS

Water table fluctuations were lower in cleared plots due to higher water consumption compared to non-cleared plots. The findings from this study indicate that trees reduce the level of the water table. That the herbaceous biomass production was lower in not cleared plots indicates that *A. mearnsii* reduced vegetation growth. The findings from this study also indicate that in order for herbaceous vegetation to grow, *A. mearnsii* trees can be reduced so that pH levels increase. Soils

from under *A. mearnsii* canopies had lower species diversity indicating that this species affects vegetation diversity; a strong case for its removal.

LIMITATIONS

The equipment for the water table study was sourced overseas, arriving late. This led to delays in data collection and a reduction in the study period. The machines that we used for measuring the water table fluctuations were expensive, only a few could be purchased. The researcher acknowledges the possibility that results might not necessarily represent direct changes in water table due to consumption by *A. mearnsii* trees; seepage from other water sources such as streams or rivers could have confounded the relationship. The increased herbaceous biomass yield could also be attributed not only to the increase in water table levels but also due to increased exposure to sunlight, as an example.

RECOMMENDATIONS

It is recommended that *A. mearnsii* be reduced to sustainable levels that do not impact negatively on rangeland production levels; a small area can be set aside for shade or firewood.

A longer study period of at least 3 years is suggested for the assessment of changes in water table fluctuations due to removal of *A. mearnsii*. A study on mineral changes due to *A. mearnsii* presence and microbial activity should be done. More replicates are suggested in further research.

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