THE INFLUENCE OF *ACACIA MEARNSII* INVASION ON SOIL PROPERTIES IN THE KOUGA MOUNTAINS, EASTERN CAPE, SOUTH AFRICA

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By

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Abstract

The invasion of *Acacia mearnsii* in the Kouga catchment, Eastern Cape, South Africa, has various negative impacts on the ecosystem. These impacts include: reduced species richness, increased water use, increased nutrients and increased N cycling rates.

The native shrubby fynbos vegetation has adapted to the acidic nutrient poor soils and Mediterranean climate of the Kouga Mountains. Fynbos, however, is currently being out competed by the much taller *Acacia mearnsii* trees, due to their competitive nature and ability to fix nitrogen, thereby enriching the soil. The invaded sections of the valley bottoms and lower hill slopes are characterised by an almost complete monoculture of *Acacia mearnsii*, with very few fynbos species still present.

The Department of Water and Environmental Affairs sponsored Working for Water programme started clearing *Acacia mearnsii* in 1996 in the Kouga Mountains. Cleared sites have remained bare for long periods, indicating that soil properties are not favourable for indigenous propagule re-establishment. The aim of this research was to assess how *A. mearnsii* invasion and clearing affect fynbos recovery through its impact on soils. This was done by characterising vegetation and soil properties on fynbos, infested and cleared slopes. Vegetation cover for various growth forms was determined and a species list was compiled for each plot. The slope angle, surface hardness, litter cover, bare ground cover and soil depth were measured in the field, whereas water repellency, particle size and the chemical composition were measured in the laboratory.

Furthermore, the plant establishment capacity of soils from fynbos, infested and cleared slopes was calculated. This was done by germinating fynbos seeds and growing fynbos plants in soils from the various slopes. The effect that invasion and clearing has on soil erosion was quantified using erosion plots on fynbos, infested and cleared slopes.

The invasion and clearing of *Acacia mearnsii* led to an increase in soil nutrients, especially nitrogen, phosphorus, potassium, carbon and manganese. Furthermore, soils became more acidic, with increased water repellency and reduced surface hardness. The vegetation changed to a tree-dominated structure, replacing the native species. Native plant germination was relatively unaffected by invasion and clearing, with an increase in germination just after clearing. Plant growth of a native grass, *Themeda triandra*, and herb, *Helichrysum umbraculigerum*, has increased on soils from cleared slopes.

This study showed that soil movement increased on slopes which are invaded and cleared of *Acacia mearnsii*, with erosion rates doubling on invaded slopes.

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List of Abbreviations

WfW	-	Working for Water
ECRP	-	Eastern Cape Restoration Programme
DWEA	-	Department of Water and Environmental Affairs
A.	-	*Acacia*
pH	-	Potentiometric Hydrogen ion concentration
N	-	Nitrogen
P	-	Phosphorus
C	-	Carbon
B	-	Boron
CEC	-	Cation Exchange Capacity
K	-	Potassium
Na	-	Sodium
Ca	-	Calcium
Mg	-	Magnesium
Cu	-	Copper
Zn	-	Zink
Mn	-	Manganese
ICP-OES	-	Inductively Coupled Plasma Optical Emission Spectrometry
WDPT	-	Water Drop Penetration Time

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Unless otherwise stated, this thesis presents the authors own work.

Chapter 1: Introduction

1.1 Overview

The invasion of the South African fynbos biome by the Australian Black Wattle tree (*Acacia mearnsii* de Wild) has various negative impacts on the fynbos ecosystem. Reduced species richness (Galatowitsch and Richardson, 2005), increased water use (Dye *et al.*, 2001), increased nutrients (Musil and Midgley, 1990) and increased nitrogen cycling rates (Yelenik *et al.*, 2004) are all negative impacts experienced under similar invasion conditions.

The native fynbos vegetation has adapted to the acidic nutrient poor soils and Mediterranean climate. These sandy soils are leached from nutrients due to the relatively high rainfall (mean annual precipitation of 600 mm (Rebelo *et al.,* 2006)). Fynbos is out competed by the much larger *A. mearnsii* trees (Galatowitsch and Richardson, 2005) which fix nitrogen, thereby enriching the soil (Musil and Midgely, 1990; Yelenik *et al.,* 2004). Invaded sections of valley bottoms and hill slopes in the study area are characterised by an almost complete monoculture of *A. mearnsii*, with the occasional fynbos species still growing.

The Department of Water and Environmental Affairs (DWEA) sponsored the Working for Water (WfW) programme to start clearing invasions in the Kouga Mountains in 1996 (Moore, 2008 pers. comm.). The study area is located in the Kouga Mountains, Eastern Cape, South Africa. The vegetation is similar to the Mediterranean vegetation, characterised by fine leafed shrubs. The geology consists mainly of quartzitic sandstones that result in shallow sandy soils with a very low nutrient content. The study area is discussed in more detail in section 1.4 below. Where these infestations were cleared, sites remained bare for long periods after clearing, indicating that soil properties are not favourable for propagule re-establishment of indigenous vegetation. In order to reduce these post-clearing impacts on the ecosystem, rehabilitation is necessary to restore functional slope and river ecosystems.

WfW has initiated a natural resource restoration programme, the Eastern Cape Restoration Programme (ECRP), which aims to restore both social and financial capital of rural communities by restoring natural resources (Rhodes Restoration Research Group, 2008). ECRP supports two major programmes: the Riparian Fynbos Restoration Project, that this study forms part of, and the Subtropical Thicket Restoration Project. The Riparian Fynbos

Restoration Project focuses on enhancing the work of WfW by rehabilitating degraded land after clearing non-native invasions (Rhodes Restoration Research Group, 2008).

The Riparian Fynbos Restoration Project aims to establish small scale restoration projects in priority catchments in the Eastern Cape. These projects will contribute towards restoring ecological functionality in degraded systems (Rhodes Restoration Research Group, 2008).

The goals of the programme are as follows (Rhodes Restoration Research Group, 2008 p.1):

1. “To demonstrate successful riparian restoration in pilot projects at an operational scale.”
2. “To develop Best Management Practices at an operational scale for restoration after the clearing of alien invasive vegetation by WfW.”
3. “To develop and implement a defensible monitoring and evaluation protocol for restoration projects.”
4. “To undertake a cost-benefit analysis of restoration activities.”
5. “To communicate and build support for the outcomes of the pilot projects.”
6. “To develop restoration protocols to be applied and tested in other South African river eco-regions.”
7. “To build capacity in the field of natural resource restoration.”

This project should meet some of the needs that Richardson and Van Wilgen (2004) expressed regarding information on rehabilitation of invaded river systems and adjacent hill slopes. In order to rehabilitate these cleared areas successfully, one needs to understand what changes have occurred in the soils due to both invasion and subsequent clearing. Very little information is available on the effect of *A. mearnsii* on soil properties in South Africa. The programme has identified this knowledge gap: the effect that invasion and clearing has on soils and the consequence for restoration (Rhodes Restoration Research Group, 2008).

It is hypothesised in this study that the invasion of *A. mearnsii* leads to altered physical and chemical soil properties that influence fynbos establishment. The study focuses mainly on the change in chemical and physical properties of the soil and relates these to the potential or suitability for seedling establishment of fynbos. Furthermore, erosion measurements made it possible to asses the influence of altered vegetation cover on soil erosion.

1.2 Problem statement

A. mearnsii has invaded much of the riparian and lower slope fynbos vegetation in the head waters of the Kouga Mountains. The Working for Water programme has been clearing some of these stands since 1996 (Moore, 2008, pers. comm.). Many of the sites that have been cleared are still bare and fynbos recovery on these sites is considered very slow. The greatest effect of the *Acacia* species on soil seems to be the altered chemical status (Musil and Midgley, 1990). Yelenik *et al.* (2004) found that *Acacia* stands increase nitrogen cycling rates within the low-nutrient environment of fynbos. These *Acacias* are nitrogen fixers, increasing soil nutrients (Musil and Midgley, 1990), especially nitrogen (Yelenik *et al.*, 2004). Chou *et al.* (1998) found that *Acacia* species had an allelopathic effect on other plants, resulting in reduced growth rates. Both the changed soil chemistry and the allelopathic effect can influence fynbos re-establishment, and therefore explain these bare slopes. Such bare slopes increase the potential for erosion (Richardson and Van Wilgen, 2004), and lead to a loss of valuable topsoil.

1.3 Aims, key research questions and objectives

1.3.1 Aim

The aim of this research was to assess how *A. mearnsii* invasion and clearing affect fynbos recovery through its impact on soils.

1.3.2 Research questions

- Does *A. mearnsii* invasion influence physical and chemical soil properties and vegetation structure on hill slopes adjacent to headwater streams in the Kouga Mountains?
- Does the process of clearing *A. mearnsii* influence physical and chemical soil properties and vegetation structure in these areas?
- Do altered soil properties affect native seedling establishment?
- Does altered vegetation cover influence soil erosion on these slopes?

1.3.3 Objectives

- **Characterise vegetation and soil properties for fynbos, infested and cleared sites.** Field sites were surveyed and sampled in order to produce data that can be used in the following objectives.
- **Analyse the soil–vegetation interaction.** The data obtained from field surveys were used to assess what influence *A. mearnsii* and the clearing thereof has on physical and

chemical soil properties and vegetation structure. The data could also indicate relations between soil change and vegetation structure change.

- **Relate germination and plant growth to different soil conditions.** The different soil conditions might have different effects on seedling establishment. In order to verify this, germination and plant growth experiments were carried out. These data could be used to determine whether there are links between soil condition and seedling establishment.
- **Quantify the effect of wattle infestation/clearing on soil erosion.** Soil erosion measurements on the different cleared sites produced data that were used to indicate the effect of wattle infestation/clearing on soil erosion. Erosion is a problem in disturbed areas, and these measurements informed us about the extent of the erosion in this specific case. The effect of different soil conditions on seedling establishment might not explain the poor re-vegetation of cleared sites. As an alternative this might be due to a loss of topsoil containing the indigenous seed bank.

1.4 Study Area

1.4.1 Introduction

The study area (Figure 1.1) is located in the Eastern Kouga Mountains, Eastern Cape, South Africa. It lies approximately 100 km west of Port Elizabeth and 20km north-west of the small town Kareedouw, and forms part of the Cape Fold Mountains.

The topography is characterised by long rounded mountain chains, with moderately steep slopes (20˚) exposing some of the geology (Rebelo *et al.*, 2006). The Kouga River drains most of the Kouga Mountains in an easterly direction and flows into the Baviaans Kloof where it joins the Groot River coming from the Karoo, forming the Gamtoos River. The Gamtoos River flows into the Indian Ocean. The four rivers (Figure 1.2) in the study area drain in a south westerly direction, resulting in north and south facing slopes.

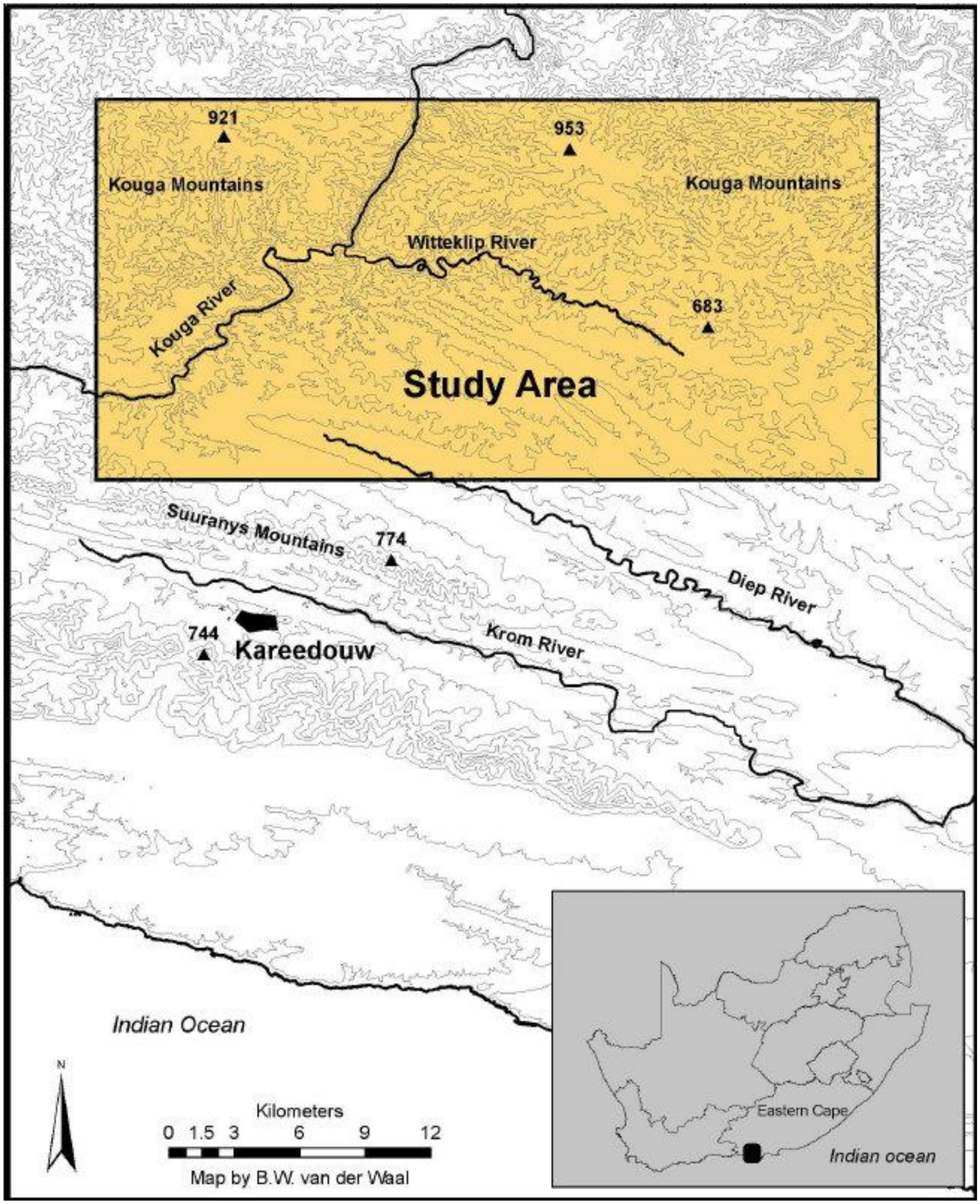

Figure 1.1 Location of the study area in South Africa.

The study area falls within the non-seasonal rainfall area (Cowling and Holmes, 1992), and receives between 300 – 600 mm per year (Boshoff *et al*., 2000), with a bimodal peak in autumn and spring. The average rainfall for Kareedouw, the small town 20 km south west of the study site, is 731 mm per year for the period of 1929 to 2006 (Pietersen, 2009).

Southerly winds in autumn and spring bring orographic rain, associated with frontal systems, on south facing coastal mountains (Rebelo *et al.*, 2006). The summer months are the driest period. Precipitation ranges from drizzle to rainstorms. Frost can occur for 10-40 days per year (Rebelo *et al.*, 2006). Temperature ranges from an average minimum in July of 2.9˚C to an average maximum in February of 27.3˚C (Rebelo *et al.*, 2006).

1.4.2 Geology

The Kouga catchment forms part of the Cape Folded belt, consisting of alternating layers of sandstone and shale, known as the Cape Super Group, with recent deposits of the Grahamstown Formation (Toerien and Hill, 1989; Boshoff *et al.*, 2000). The dominant rock exposures are sandstone members. The dominance of sandstone affect soil properties, leading to a tendency towards acidity (Rebelo *et al.*, 2006). The main geological formations in the study area are described according to Toerien and Hill (1989) as:

1. Grahamstown Formation – this scattered high-level terrace deposits border mountain ranges. The formation consists of sandstone and quartzite blocks that are cemented by siliceous material. This layer is normally 6 meters thick.
2. Cederberg Shale Formation – this is carbonaceous thinly bedded silty formation, which disappears at places because of folding.
3. Goudini Sandstone Formation – a medium grained quartzitic sandstone with a brownish colour. Cross bedding with layers of shale of less than a meter thick occurs. This formation is 260 meters thick.
4. Peninsula Sandstone Formation – the formation is most prominent and consists of coarse to medium grained super mature sandstone, which can become quartzitic in places. Rounded quartzitic pebbles are scattered in the sandstone and lenticular shale layers of a meter thick can occur. The formation can occur up to 3 000 meters thick.

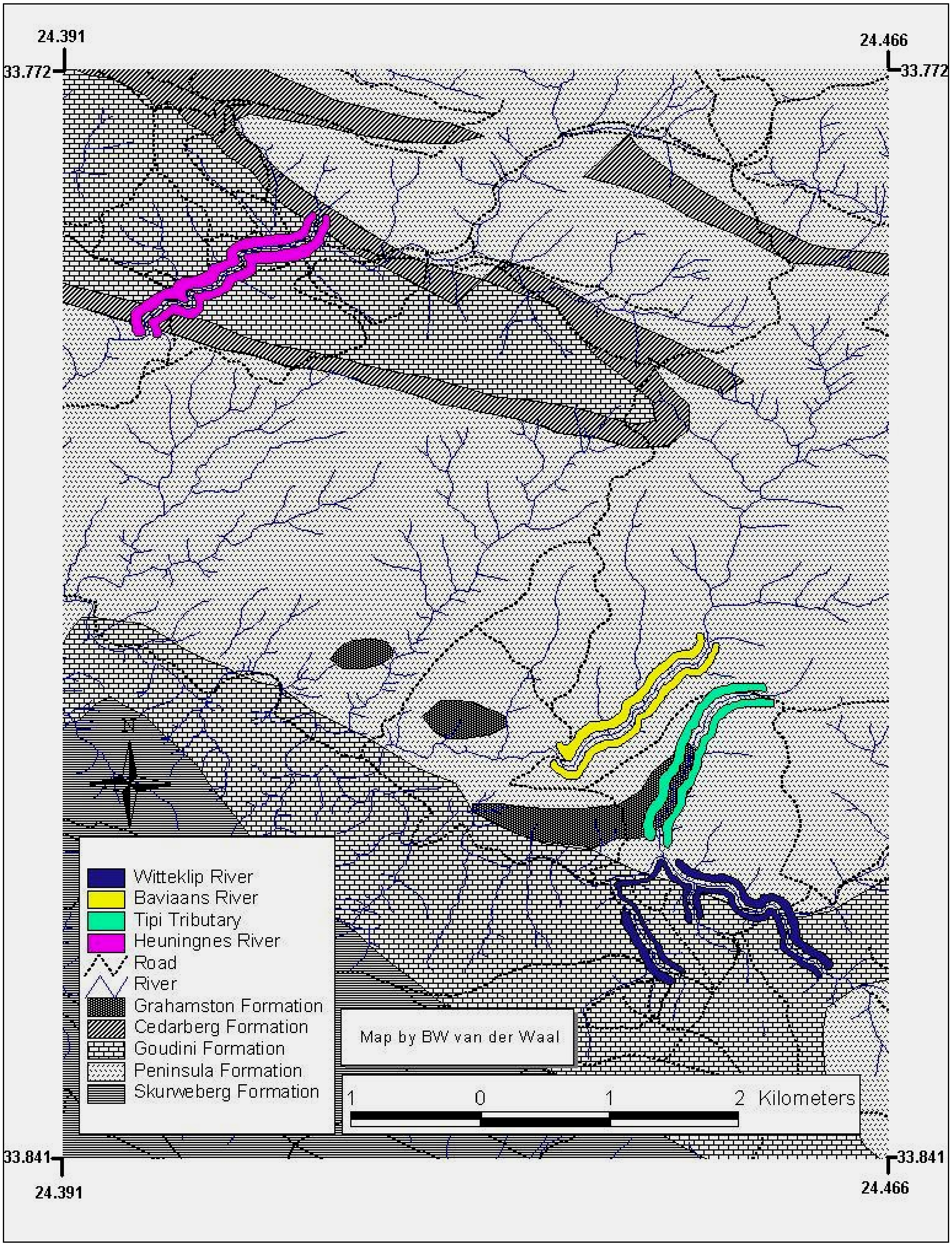

Figure 1.2 The location of sampling sites and underlying geology. The coloured lines refer to the areas that were sampled along each of the rivers.

5. Skurweberg Sandstone Formation – This formation consists of white, resistant, medium to coarse-grained quartzose sandstone with a high quartzitic content. This layer can be up to 420 meters thick.

1.4.3 Soils

The soils of this area have been described by Rebelo *et al.* (2006). They are mainly shallow acidic lithosol soils that are derived from quartzitic sandstones of the Witteberg Group (Nardouw Subgroup) and sandstones of the Table Mountain Group (Rebelo *et al.*, 2006). The tallus slope can have slightly deeper soils as material accumulate in these areas from up slope areas (Rebelo *et al.*, 2006). The soils with a higher quartzitic concentration are stony and consist mainly of sand, with very low free iron oxides and a clay content of less than 5 % (Rebelo *et al.*, 2006). The soils are pale in colour and relate to the Mispah and Glenrosa forms (Rebelo *et al.*, 2006). Mispah soils consist of an orthic A-horizon on rock, and Glenroza of an orthic A-horizon on a lithocutanic B-horizon on Saprolite (Rebelo *et al.*, 2006). The soils with a lower quartzitic content have a higher iron oxide and clay content, and appear yellowish or reddish (Rebelo *et al.*, 2006). Where there are shale bands running through an area the soils are deeper, heavy textured and of the duplex form (Rebelo *et al.*, 2006).

1.4.4 Vegetation

The flora is typically of the Cape Floristic Region. The area has a great diversity of vegetation types, with the most outstanding in the study area being Grassy Fynbos (Rebelo *et al.*, 2006). The most prominent families are *Ericaceae, Proteaceae, Iridaceae, Rutaceae* and *Aizoaceae*. A number of the typical fynbos genera - *Agathosma, Aspalanthus, Crassula, Erica* and *Pelargonium* - are found in the area (Boshoff *et al.*, 2000). *A. mearnsii* is a major problem in the valley bottoms as this water loving, highly competitive, non-native readily invades the native flora, resulting in a monoculture (Fourie, 2008, pers. comm.).

1.4.5 Site history and invasion over the past century

The invasion of slopes in the four rivers (Figure 1.2) started off in the same way, but was cleared at different times (Table 1.1). *A. mearnsii* propagules were planted in the early 1900s near livestock kraals and other areas along the river courses where wood and shelter were needed (Du Plesis, 2008, pers. comm.). These large individuals are still visible in the Heuningnes River as are cut stumps in the Witteklip and Baviaans Rivers. The Tipi tributary has smaller diameter cut stumps at a lower density. This low density is also evident in aerial photographs dating back to 1954 and 1969.

Table 1.1 Summary of the clearing histories of the four study sites. High density refers to an almost complete monoculture of *A. mearnsii* (cover of 50 – 100 percent) and low density to a partial (cover of 10 - 50 percent) invasion.

River	Condition	Density before clearing	No. of follow-ups
Heuningnes	Invaded	High	0
Baviaans	Cleared - 2008	High	3
Tipi	Cleared - 1997	Low	6+
Witteklip	Cleared - 1997	High	11+

Interpretation of aerial photographs dating back to 1954, 1969 and 2003 and interviews with second generation farmers led to the conclusion that the area was mainly used for cattle and sheep farming, with some crop farming on flat land next to rivers. *A. mearnsii* infestations were limited to small areas within the riparian sections of rivers in 1954. The infestations spread down river and to the bottom of slopes in the period up to 1969. In 2003 it is visible that these infestations spread upriver and further up onto the slopes. This is visible in Figure 1.3 for the Heuningnes River. The scale for the original photos used in Figure 1.3 and 1.4 was 1:36 000, resulting in lower resolution for the set in 1954 and 1969.

Some of the areas that were infested in 1969 photos were cleared in more recent photos of 2003. This was the case in Figure 1.4 for the areas in the Witteklip River, Baviaans River and Tipi tributary.

The Heuningnes River has never been touched by any of the clearing teams, resulting in an intact invaded study site. The Baviaans River was cleared in September 1997, received another two follow up clearing treatments and was handed over to the landowner in 2001 (Moore, 2008, pers. comm.). The Baviaans River was completely reinvaded by 2001, with a second clearing that took place in August 2007. Active restoration has been applied to the Baviaans River since the clearing in 2007. The Tipi tributary had a low (10 %) invasion density and was cleared in March 1997 (Moore, 2008, pers. comm.), and received three follow up clearing treatments according to the Water Information Management System (WIMS) dataset. The last follow-up was in 2005 according Bateman (2008). The Witteklip River was cleared in March 1997 (Moore, 2008, pers. comm.), and received six follow up treatments since the clearing according to the WIMS dataset. The WIMS data set only has clearing data for the period 2002 until 2006, thus the number of follow-up treatments might be higher for the Tipi tributary and the Witteklip River. According to Moore (2008, pers. comm.) the Tipi and Witteklip Rivers received two follow-ups per year before 2000, resulting in an additional 3 and 5 follow-ups respectively.

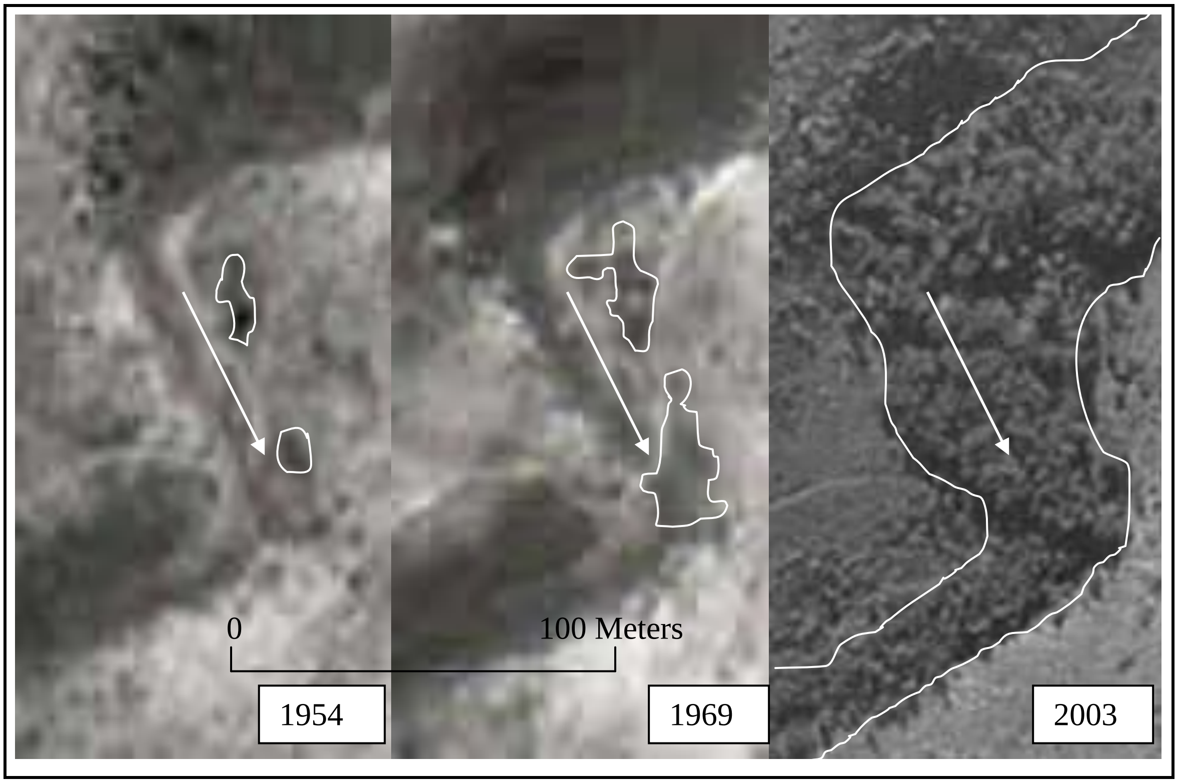

Figure 1.3 Aerial images for 1954, 1969 and 2003 for the same area in the Heuningnes River. *A. mearnsii* invasions are encircled, and the arrow indicates the flow direction of the river.

Garlon 4, a broadleaf herbicide with tryclopyr as the active ingredient, was applied on cut stumps and as foliar blanket spray (Moore, 2008, pers. comm.). Soil micro-organisms (fungi and bacteria) and sunlight break Garlon 4 down, and its half life is reached after 30-45 days (Dow AgroSciences, 1989). The resultant products are carbon dioxide, water and other organic materials. The breakdown is influenced by rainfall, soil temperature and micro-organism activity (Dow AgroSciences, 1989).

According to Bateman (2008, pers. comm.) a major veld fire went through the whole study area in 2002, resulting in burnt stumps that are visible in all of the cleared areas.

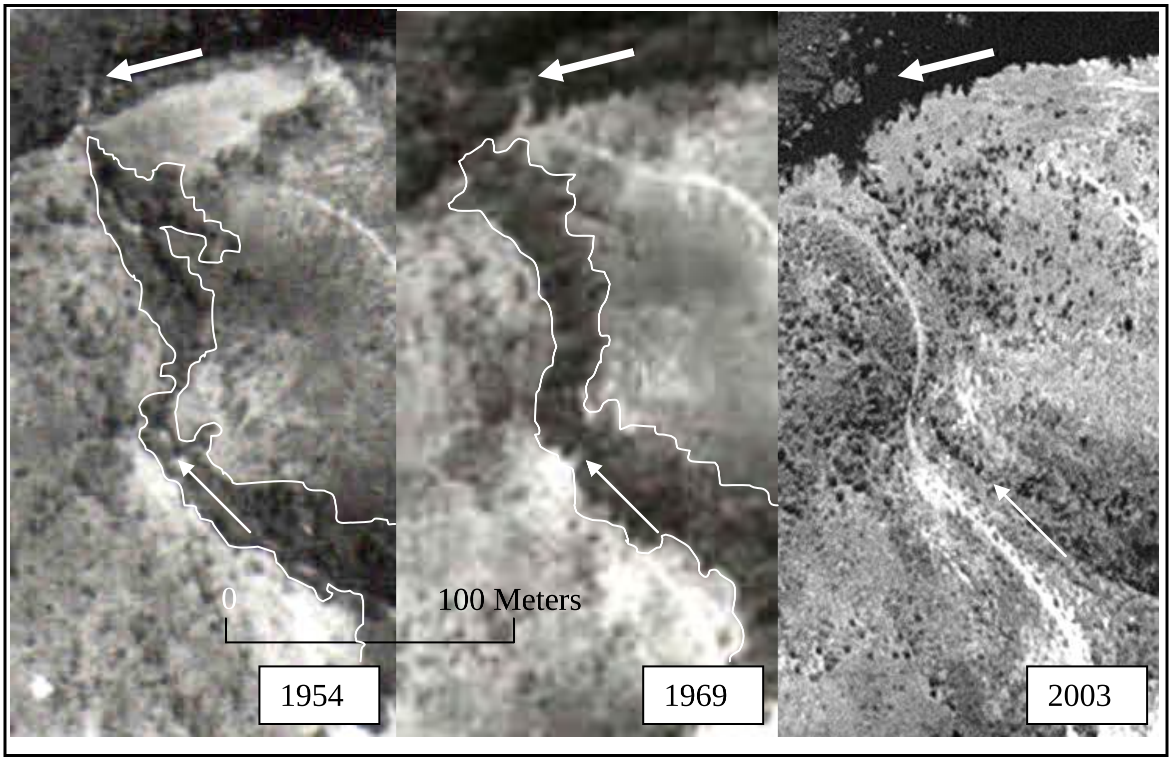

Figure 1.4 Aerial images for the same area along the Witteklip River, indicating *A. mearnsii* (encircled vegetation along river course) invasion for 1954, dense invasion for 1969 and the invaded area cleared in 2003. Arrows indicate flow direction, with the thick arrow indicating the Witteklip River.

1.4.6 Conclusion

These study sites were suitable for this research as they were on private land that formed part of a conservancy. The site history could be traced back through aerial photographs, a database and interviews with landowners and managers. Access to the land was readily available, and the land-owner was more than willing to cooperate. Interaction with other researchers and clearing teams were beneficial for the project.

A lot of clearing has been done on the property, with different retrievable clearing histories. This allowed for investigation in the influence of different treatment effects. However, natural variability of soils and vegetation could potentially influence the clarity of the data analysis.

The study sites were accessible by road, although most of the higher areas are only accessible by jeep track. Most of these tracks are steep and rocky, thus a 4x4 vehicle was essential. The jeep track would cross a river, from where one had to walk to the sampling plots as most of the area was only accessible on foot or by helicopter. The large volumes of soil that were needed for plant establishment studies were transported in large back packs during the cooler hours of the day. The thick shrub and tree stands were challenging to navigate through, but game trails made hiking manageable. On the hikes snakes were a common hazard. During fieldwork I came across many Puffadders (*Bitis arietans*), an Ornate Green Snake (*Philothamnus ornatus*) and a Harlequin Snake (*Homoroselaps lacteus*).

A good site history, willing landowner and helpful colleagues made this study in a relatively remote and harsh location possible. Field work was very physical, but highly enjoyable.

1.5 Thesis outline

Chapter 1 served as a basic introduction to the problem and focus of the study and study area. Chapter 2 reviews literature regarding degradation, rehabilitation, fynbos ecology, effects of non-native invasive plants on soil composition, plant structure and soil erosion. Chapter 3 unpacks the methods used for this study, and Chapter 4 gives the results. Chapter 5 discusses the results and limitations, and links them to the literature. Chapter 6 draws conclusions.

Chapter 2: Literature review

2.1 Introduction

The Fynbos Biome is a member of the Mediterranean Biome, whose geographical spread has been described by Rebelo *et al.* (2006). According to these authors the Mediterranean Biome is located on the western shores of continents, at latitudes north or south of the arid belt associated with the horse latitudes around the Tropic of Cancer and the Tropic of Capricorn. Other members of this biome are present in northern Chile, south-eastern and south-western Australia, south-eastern Europe, the Iberian Peninsula, North Africa, the Middle East, Iran, Mediterranean and Canary Islands and south-western USA. All of these regions have typical Mediterranean climates - dry summers and cool wet winters. Australia, South Africa and some Mediterranean regions are special cases as they have nutrient-poor systems. For this reason, this review will focus only on the Fynbos Biome as it is so unique in its soil characteristics amongst the other Mediterranean Biomes.

Fynbos is situated in the southern tip of Africa and has been recognised as one of the most diverse vegetation types in the world (Manning, 2007). It occupies a relatively small area of 90 000 km2 and is home to almost 9 000 flowering species, with two thirds endemic to the fynbos biome (Manning, 2007). The dominant vegetation ranges between one and three meters in height and consists of evergreen sclerophyllous shrubs (Rebelo *et al.,* 2006) and evergreen grass-like perennials (Manning, 2007). These sclerophyllous shrubs are characterised by small leathery, ridged and heavily cutinised leaves (Jeffre *et al.*, 2007), brown-green to greyish in colour (Manning, 2007). According to Stock *et al.* (1992) this is a result of nutrient poor soils, summer droughts and low growth rates. Fire is an important driver that is responsible for the high plant community diversity (Southey *et al.,* 2009). These shrubs are prone to fire, are actually fire loving, with life strategies specifically adapted to a fire regime (Deacon *et al.*, 1992).

Fynbos is highly susceptible to the invasion of woody species, especially trees (Macdonald *et al.*, 1988). These invasive trees are much larger than fynbos, with high biomass resulting in much higher water use than the surrounding fynbos (Le Maitre *et al.,* 2002). In 1995 the Working for Water programme was initiated with the purpose to restore and maintain natural resources and create jobs by clearing invasive non-native plants (Van Wilgen *et al.,* 1998; Marais and Wannenburg, 2008). The main aim of the project was to conserve water. Much of the work in South Africa was focussed on the effect of invasive trees on water resources. This study focuses on the effect of these invasive *A. mearnsii* trees on soil properties, plant structure and plant biodiversity.

In order to undertake research regarding this highly specialised fynbos ecosystem, it is necessary to explore the literature regarding the ecology of fynbos, the interaction between plants and soils, possible ecosystem disturbances and the effects thereof. This would allow for investigation into the potential changes in the ecosystem due to *A. mearnsii* invasion. For this project, it is also important to look at the definitions of degradation, invasion and rehabilitation, how *Acacia* species influence soil properties and vegetation structure, and the effect of vegetation change on erosion processes. These topics address some of the key points related to this research project and are described in detail below.

2.2 Ecology and environmental conditions in the Fynbos Biome

2.2.1 Distribution in South Africa

Fynbos is mainly found in the south-western Cape, being the dominant vegetation type extending from Vanrhynsdorp in the northwest to Grahamstown in the east (Manning, 2007). Mountain fynbos is the dominant fynbos type (Rebelo *et al.*, 2006), with a pocket of Sand-plain Fynos along the west coast (Manning, 2007), Laterite Fynbos in the south on the Elim Flats (Rebelo *et al.*, 2006), Limestone Fynbos in the south between Walker Bay and Mossel Bay (Rebelo *et al.*, 2006) and Grassy Fynbos east of the Kouga Mountains (Rebelo *et al.*, 2006). Manning (2007) states that fynbos is mainly associated with the quartzitic Cape Fold Mountains and neighbouring coastal shelves. Other fynbos communities are found on sandstone in southern KwaZulu-Natal, the Drakensberg and the eastern Escarpment (Manning, 2007).

2.2.2 Climate in the Fynbos Biome

The Fynbos Biome has a Mediterranean climate, with cold wet winters and dry hot summers (Cowling, 2009). The following paragraph on climate is based on the work of Rebelo *et al.* (2006). The Mean Annual Precipitation (MAP) for the Fynbos Biome is 480 mm, with a minimum in the north-west of 100 mm and a maximum of 3190 mm at Jonkershoek (south-west Cape). Cloud cover is frequent in summer on the high mountains in the south-western parts of the biome. These clouds can produce an additional 500 mm of precipitation on south facing slopes, which does not form part of the measured MAP as it only occurs on relatively small areas on high peaks. Snow-fall (up to a meter deep) occurs on mountains higher than 1000 m above sea level in the Western Cape during July. Snow will last for more or less 2 weeks. Humidity is high (>70 %) along the coast in summer and on higher ground in winter. Frost only occurs on the higher-lying regions. The Mean Annual Temperature for the biome is 12°C on higher-lying areas and 19°C on lower-lying areas. Extremes of -12.2°C and 46.1°C were measured within the biome. Mean Annual Potential Evaporation is relatively high and varies from 1400 mm to 2400 mm. Hail and lightning frequency increases to the east, where lightning peaks with 2 flashes per km2 per year.

The rainfall in the Cape is characterised by a strong Mediterranean winter rainfall regime in the west, decreasing in winter precipitation predictability as one moves east (Cowling, 2009). The current influences on rainfall for the Fynbos Biome is summarised by Deacon *et al.* (1992) as steep consecutive coastal mountains running parallel to the coast and a ridge of high pressure (associated with the Hadley Cell) that extends from the South Atlantic anticyclone to the Cape coast. It migrates south in summer providing dry winds, and more northerly in winter, providing orographic rainfall on west-facing slopes. Deacon *et al.* (1992) also note that the warm Agulhas Current located more towards the east, provides moisture to the costal areas. Persistent westerly winds cause daily variability in temperature and moisture levels, due to cut-off lows, coastal lows and anticyclones (Deacon *et al.*1992).

2.2.3 Geology and soils in the Fynbos Biome

The geology within the Fynbos Biome varies between sandstone, quartzite, granite, gneiss, shale and limestone (Rebelo *et al.* 2006). The prominent mountain ranges dominating the Fynbos Biome, consists mainly of quartzite, which is extremely nutrient poor (Rebelo *et al.* 2006).

Soils in the Fynbos Biome are described by Rebelo *et al.* (2006). They ascribe the high variability of these soils to a range of environmental conditions such as: present and past rainfall, parent material, age of landscapes and terrain type. Of these, Rebelo *et al.* (2006) state parent material to be the main factor influencing the physical and chemical nature of the different soils. There is an east-west change in underlying substrata that gives rise to a soil fertility gradient that increases towards the east (Cowling, 2009). Rebelo *et al.* (2006) classify the soils in the biome into five main groups based on parent material. These groups are: heavy-textured soils, coastal plain soils, soils associated with silcrete and ferricrete, sandy soils of quartzitic fold ranges and other soils. These are described below.

- The heavy-textured soils were derived from shales and slates. These soils are characterised by a proportionately large silt and fine sand fraction, with a high nutrient status, rich in potassium.
- Coastal plain soils formed mainly from dunes, young dunes with varying iron content and fossilised dunes, which are rich in calcium and potassium. These deep sandy soils are mostly leached and have a low pH.
- The soils associated with silcrete and ferricrete are formed from iron and silica cemented unconsolidated sediments or hardpans (duricrusts). These soils are highly leached, low in clay content with a low pH.
- Sandy soils of quartzitic fold ranges can be grouped into mountain slope soils, and pediment and valley floor. The mountain slope soils are formed from resistant quartzites and sandstones that give rise to stony, sandy soils with very low clay and iron content. This soil's pH is low, and has the tendency to become more acidic in areas with increased rainfall. The pediment and valley floor soil are very sandy with a slightly higher clay and carbon content. This soil has a very low pH (<4.0).
- 'Other soils' mainly consist of alluvium coming from quartzitic origin. These sandy soils are normally acidic with a low base status.

The present study focuses on mountain soils. Deacon *et al.* (1992) describe these lithosoils associated with the mountains as shallow, weakly developed soils associated with mountains where the main soil forming process is podzolization. This process includes mobilisation of aluminium, iron and organic matter, which gives rise to podzolic B horizons and leaching, with a loss of soluble salts and clays from the overlaying A horizon. This leaching results in a soil that is poor in plant nutrients, especially in nitrogen and phosphates and bases (Witkowski and Mitchell, 1987; Deacon *et al.,* 1992). Cracks and fissures in the rocks lead to small pockets of hydromorphic soils with micro- climates that are being colonised by plants (Deacon *et al.*, 1992). The combination of climate, relief and soils gave rise to the unique ecological balance in the fynbos biome (Deacon *et al.,* 1992).

2.2.4 Plant-water-soil interactions in fynbos

Low nutrient conditions prove to be the main factor responsible for the shrubby growth form of fynbos (Stock *et al.,* 1992). The low nitrogen content in Mediterranean-type vegetation does not allow for high photosynthetic rates (Mooney *et al*. 1983). An increase in soil nitrogen through fertiliser additions result in increased photosynthetic rates of sclerophyllous plants (Mooney *et* *al*., 1983). Witkowski *et al.* (1990) recorded an increased plant dry weight and shoot extension for two fynbos species, *Leucospermum parile* (Proteoid) and *Phylica cephalantha* (shrub), but only for the first year after nitrogen addition in Sand Fynbos. Phosphorus had a reduced shoot extension effect in some cases (Witkowski *et al.*, 1990). After several years, overall plant mass of fynbos shrubs were unaffected by added nitrogen and phosphorus, but increased for herbaceous plants (Witkowski and Mitchell, 1989). Thus fynbos has a relatively low growth response to added nutrients in comparison to annual herbs. Witkowski (1989) found contradicting results for potted *Protea repens* (Proteoid). The results showed an unexpected decrease in plant biomass for an increase in nitrogen. Furthermore, plant mass increased for an addition of phosphorus, and remained stable for an increase of both nitrogen and phosphorus (Witkowski, 1989).

According to Cowling and Holmes (1992), soil nutrients influence community structure in fynbos. They accredit the higher number of taxa in the south-western regions of the fynbos biome to the strongly developed Mediterranean climate. Further, they argue that the higher frequency of summer rain in the south eastern fynbos biome and more fertile soils favours tropical species (C4 grasses and thicket species) that out compete the fynbos species (Cowling and Holmes, 1992).

Furthermore, rainfall and soils play an important role in community boundaries along aspect and altitude gradients in the mountains (Cowling and Holmes, 1992). At the base of south facing mountains, deep and relatively fertile colluvial soils support proteoid fynbos. As one moves upslope to high altitudes, ericaceous fynbos replaces proteoid fynbos. These higher slopes receive more rain, thus soils are wet more frequently, finer grained and are richer in organic carbon (Cowling and Holmes, 1992). At this altitude on the north facing slopes, rainfall is sparser, resulting in drier conditions. The soils are shallower, and support restioid fynbos (Cowling and Holmes, 1992). As one moves down the drier north facing slope, asteraceous fynbos becomes dominant as such vegetation can tap into deep ground water (Cowling and Holmes, 1992). According to Cowling and Holmes (1992) both restioid and asteraceous fynbos occur on shallow soils.

Stock *et al.* (1992 p.227) researched water as a limiting factor for fynbos vegetation in the Western Cape, stating that "…arguments suggest that resources such as water, light and nutrients are maximally used and this leads to specific plant forms becoming common in the vegetation." In a comparison between fynbos (nutrient poor) and other Mediterranean-type ecosystems (nutrient rich), fynbos had far less water stress during dry summer periods (Stock *et al.*, 1992).

This indicates that water is not as limiting a factor in fynbos as in other Mediterranean-type ecosystems. Campbell and Werger (1988) reason that this is due to the morphological features of fynbos. Manning (2007) relates the thick cuticles and internal struts of carbon rich (lignin) structure to mechanisms that prevent leaves from collapsing under water stress. The leaves are adapted to low nutrient conditions, resulting in high nutrient use efficiency (Specht and Rundel, 1990), that enable leaves to photosynthesise for longer periods but at lower rates than normal leaves (Stock *et al.*, 1992).

Although well adapted, there were still significant differences in water use between the fynbos growth forms. According to Stock *et al.* (1992) shallow rooted restioid and ericoid shrubs showed greater water stress than the deeper-rooted proteoid shrubs. The water potential in proteoid shrubs remains fairly constant during the year because of relatively constant water sources (Van der Heyden and Lewis, 1988). The shallow rooted shrubs maintain their high stomatal conductances throughout the year, resulting in water stress during dry periods (Miller, 1985). These shallow rooted plants are seen as water savers, similar to succulents. Succulents occur mainly on the dry nutrient rich soils in the westerly regions. This indicates that water only acts as a selective pressure for fynbos on nutrient rich soils (Stock *et al.*, 1992).

Photosynthetic rates for shallow rooted plants (restioids and ericoids) were relatively low, compared to the higher rates of deep-rooted plants (proteoids). The higher photosynthetic rates relate to water availability for respiration (Stock *et al.*, 1992). The shallow rooted growth forms are thus influenced by the lack of a constant water supply. These higher rates were interpreted as an adaptive strategy for proteoid species to complete life cycles during later successional stages following fire (Van der Heyden and Lewis, 1989).

Where soils are equally nutrient poor, water seems to be the important driver for community structure. A study by Richards *et al. (*1995) highlighted the importance of soil properties in determining species distributions in the Mountain Ericoid Fynbos community of the Soetanysberg hills, Western Cape. This is an area with a typical Mediterranean-type climate, underlain by Table Mountain Group sandstone and quartzite, with locally occurrence of Bredasdorp limestone. Richards *et al.* (1995) found that vegetation structure could be predicted from various environmental factors, especially at community level. It was also found that on the sandstone slopes, the physical factors relating to soil moisture (rock cover, soil depth and texture) were the main environmental factors affecting the Mountain Ericoid Fynbos community. Soil nutrients (pH and fertility index) played less of a role because they were fairly uniform across the substratum (Richards *et al.,* 1995).

Adaptations of fynbos are noticed that allow for survival in low nutrient environments with high fire risk. The high availability of carbon is utilised by fynbos to grow fire resistant structures, and to build up underground carbohydrate reserves (Pate *et al.*, 1990). These reserves can be used to regenerate plant material after fires, giving the plant a competitive advantage (Pate *et al.*, 1990). Some species use the available carbon to produce cones to protect seeds from fires and predation (Bond, 1985). Other species allocate high quantities of carbon to secondary metabolites, creating toxic substances that prevent herbivory (Stock *et al.*, 1992 and Manning, 2007). The sclerophyllous nature (high lignin content) of the leaves makes them unpalatable to herbivores. Furthermore pollination is influenced by the high carbon and low nutrient availability (Rebelo, 1987). Nectar with high sugar concentration attracts insects and birds, which in turn increases pollination efficiency (Mostert *et al.*, 1980).

From the literature nutrient poor soils can be deduced as the main factor affecting fynbos structure and function. Water is given as the second factor affecting plant structure, function and species distribution. Fynbos can therefore be regarded as highly adapted to the niche environment it grows in. Changes to the physical environment might reduce fynbos's competitive ability, therefore impacting on the potential for restoration after clearing.

2.2.5 Fire

Fynbos is a fire prone vegetation type (Pierce and Moll, 1994) that can accumulate large quantities of fuel more or less 75 cm above ground (Kruger and Bigalke, 1984). Fine wax covered leaves, with high aromatic oil content, on multi branched shoots add to the high flammability of fynbos (Kruger and Bigalke 1984; Edwards 1984). Furthermore hot dry summers with frequent south easterly winds increase the risk of fire (Edwards, 1984). Most fires are human induced, but in remote locations fires are natural, caused by lightning or rock slides (Kruger and Bigalke, 1984).

The large variability in fire occurrence and frequency in fynbos is one of the main forces driving the large plant community diversity (Southey *et al.,* 2009). Fynbos regeneration after fire is divided into obligate sprouters, facultative sprouters and seed-regenerators (Le Maitre, 1992). Obligate sprouters regenerate mainly from woody rootstocks after fire (Manning, 2007), or bird dispersed seeds between fires (Le Maitre, 1992). Sprouters can tolerate fire regimes that vary largely as they are long lived and relatively unaffected by fire (Van Wilgen and Forsyth, 1992). Facultative sprouters can regenerate from dormant buds or seeds (Le Maitre, 1992). Alternatively, non-sprouters are dependant on frequent fires for regeneration as they have seeds that remain dormant until they receive fire related cues (Pierce and Moll, 1994). Long inter-fire periods may deplete seed banks as this period may exceed the lifespan of the seeds (Van Wilgen and Forsyth, 1992).

Fire removes the vegetation cover, changing microclimate and wind profiles (Kruger and Bigalke, 1984). Pierce and Moll (1994) found that the removal of the insulating vegetation by fire exposed the seeds to greater diurnal temperature changes. This proved to be an important cue for fynbos germination (Kruger and Bigalke, 1984; Pierce and Moll, 1994). Burnt vegetation will allow wind dispersed seeds to be distributed over longer distances as there are fewer obstructions.

Fire in Mediterranean vegetation increases the availability of plant nutrients, such as nitrogen and phosphorus in soil, which play an important role in early life stages of fire adapted plants (Rundel, 1983). Rundel (1983) further states that dense, fast growing herbaceous plants play a key role in tying-up post fire available nutrients, as shrubs are slow growing and have less capacity to store nutrients. This helps to retain nutrients in the system, preventing loss through surface runoff. Rundel (1983) further argues that low nutrient levels in these fire regulated nutrient cycling ecosystems might drive the selection process of fire adapted plants, with high potential flammability, as a means to recycle limiting nutrients for the next generation. These fynbos plants are fire loving with life strategies that make them fit for the fire regime (Rundel, 1992; Deacon *et al.*, 1992).

2.3 Degradation and rehabilitation

Degradation is the loss of vegetation structure, productivity and biodiversity (Lamb and Gilmour, 2003). A degraded area will still contain some of the native species, but it will have lost its pre-disturbance ecological integrity. This may take place as a result of over exploitation and repeated disturbances (Lamb and Gilmour, 2003). This leads to an altered vegetation cover and goes hand in hand with non-native species that colonise degraded sites. This can result in erosion, forming gullies and landslips, altered fire regimes, water pollution and siltation of water bodies. Collectively, these cause the degraded site to loose its productivity and many of the ecological goods and services that were formerly provided (Lamb and Gilmour, 2003).

This is the case in the Kouga Mountains were the riparian areas and adjacent lower parts of the slopes are completely invaded by *A. mearnsii.* A dense monoculture developed, suffocating the native plants and resulted in biodiversity loss. Once the non-natives were removed, land lay bare for long periods, making it vulnerable to erosion. This increased erosion potential can have negative impacts on the geomorphological processes downstream.

It is very difficult to measure degradation, as it is a very subjective process (Lamb and Gilmour, 2003). Lamb and Gilmour (2003) question how one can determine the benchmark of where ecosystem function is lost, what references are used in determining degradation and how does one know that a site has crossed the degradation threshold. Our limited understanding of ecosystem processes binds us to measures such as vegetation composition change, water reduction and species loss (Lamb and Gilmour, 2003). Reasons for degradation after a disturbance might be that there are too few pre-disturbance individuals left or a change in the biophysical environment or consecutive disturbances disrupting necessary succession (Lamb and Gilmour, 2003). Answers to these questions are debatable at best, and studies on degraded sites will help us unpack some of the unknown workings of these systems. Some systems can recover naturally, others not (Lamb and Gilmour, 2003). What we do know is that we need to act in order to revive some of the ecosystem functioning, which leads us to the question: what can we do about the degradation?

Human intervention is needed to alter the system so that rehabilitation can take place. Rehabilitation can be defined as the intervention that will repair or restore systems or parts of systems that have lost a degree of the ecosystem function, structure, biodiversity and ecosystem goods and services (Grenfell *et al*., 2007). This intervention aims to restore the system to a state similar to the pre-disturbance state. Grenfell *et al.* (2007) state two main objectives of rehabilitation:

- Reinstalling natural ecological driving forces within the degraded area
- Reinstalling a habitat that will allow the recovery of previous ecosystem function, structure, biodiversity and ecosystem services.

The measure of the success of rehabilitation remains difficult as rehabilitation is a lengthy process and natural systems are unpredictable and corrective intervention is often needed (Lamb and Gilmour, 2003). The following measures of success are given by Lamb and Gilmour (2003):

- Ecosystem stability – disturbances brought to a halt, vigorous plant growth, adequate plant cover and community structure (herbs, forbs, grasses etc.), good native species representation, decline in non-native species, regeneration and reproduction of wanted species, stable soil surfaces and adequate water quality coming from these sites.
- Efficiency – decreasing need of intervention such as fertilising, watering and weeding.
- Flexibility – increasing land use possibilities.

It is also important that local people are involved in the rehabilitation process and desire its success. Consequently the human community will gain ecological awareness and the rehabilitated site will start producing economic benefits.

2.4 Plant invasions

Fynbos is highly susceptible to invasion of non native plants, especially woody plants, and is a common form of degradation (Macdonald *et al.*, 1988; Roura-Pascual *et al.*, 2009). Macdonald *et al.* (1988) found that any disturbance in Mediterranean vegetation was followed by the invasion of herbaceous plants, such as annual grasses. These disturbances mainly occur due to the removal of invasive trees.

The following section will address general mechanisms and drivers of plant invasions, whilst the next section will focus on the effects on the soils and vegetation in the Fynbos Biome specifically.

2.4.1 General

Invasive plants can best be described as non-native (from some other geographic location) plants that have a definite ecological and economic impact (Lockwood *et al.*, 2007). It is estimated that, next to habitat destruction, non-native invasive species are the second largest cause for extinction and endangerment of native species (Ehrenfeld and Scott, 2001). Invasive species can have major effects on resource availability and can reduce the relative abundance of native species (Didham *et al.*, 2005).

Invasive non-native plants affect most ecological relationships (Lockwood *et al.*, 2007). Invasive species can affect the native genetic integrity. This happens by forming hybrids that can be sterile, reducing reproductive ability or have increased vigour, out competing native plants and reducing their reproduction (Lockwood *et al.*, 2007). On the individual level, invasive species alter the fitness or traits of natives. Furthermore, this has an effect on the population level as certain species are reduced, altering the population structure. If this is the case for more than one species, the community structure can be changed. In a worst case scenario many native species can go extinct from a community (Lockwood *et al.*, 2007). This can in turn affect the way material flows through an ecosystem. All non-native invaders have the competitive ability over native species to capture resources (Lockwood *et al.*, 2007). Some invaders use more resources, in more effective ways, and excrete by-products that can improve their competitive ability. The altered ecosystem can impact at the landscape level, potentially contributing to regional and finally global ecological change (Lockwood *et al.*, 2007).

2.4.1.1Drivers of invasion

All invasions begin with the successful transport of some propagules from its native location to a new location (Lockwood *et al.*, 2007). Invaders have the advantage of escaping their natural predators when they establish themselves in a new area (Mooney, 2005). According to Lockwood *et al.* (2007) these invasions can be natural or human assisted. The human assisted invasions can be intentional or unintentional, with intentional invasions being associated with good records of when and how it took place. The reason for intentional movement of propagules is mainly for food and environmental enhancement (Lockwood *et al.*, 2007).

Disturbance, whether natural or human induced, triggers the invasion process. In the work of Didham *et al.* (2005) the main driver for invasion is the growing human population, which requires development of industry and agriculture to support the population. This development alters land use, land cover, biogeochemical cycles and increases CO_2. All of the abovementioned play a role in global climate change (Didham *et al.*, 2005).

These changes or disturbances affect native organisms negatively, but create opportunities for non-native species to establish (Lockwood *et al.*, 2007). This can occur if there is a supply of non-native propagules that can tolerate the conditions after a disturbance. Native plants are capable of co-existing or out competing non-native plants given the right growth conditions (Daeler, 2003). The problem occurs where conditions are altered, as non-native plants have a greater phenotypic plasticity, resulting in more competitive growth and fecundity (Daeler, 2003). Altered habitat and competitive abilities can cause invaders to become dominant (Didham *et al.*, 2005).

Increased CO_2 levels cause C3 plants (most plants use this photosynthetic pathway) to grow more vigorously, not favouring natives or non-natives (Dukes, 2000). The increased CO_2 levels consequently increase temperatures (Lockwood *et al.*, 2007). This in itself can act as a disturbance that can be the driver for non-natives to become dominant in a community. This disturbance driven passenger model is depicted by Didham *et al.* (2005) where native plants are disturbed by habitat change, and non-natives colonise the open space. The model predicts that the result of clearing non-natives in this case will not lead to an increase of native species dominance.

Some non-natives are very aggressive, invading native vegetation that results in a disturbance that creates a breeding ground for themselves as well as other non-natives (Lockwood *et al.*, 2007). These species have been regarded as ecological engineers (Jones *et al.*, 1994). Effects of these engineers on ecosystem functioning can pose a threat to biodiversity (D'Antonio and Vitousek, 1992). This is a result of invaders that suppress and replace the native vegetation (Richardson *et al.*, 1997). Didham *et al.* (2005) depicts this invasion in a driver model where non-native plants invade undisturbed native vegetation, which leads to a decrease of native dominance. The model predicts that removal of these invasive species will have no effect on native species dominance.

Didham *et al.* (2005) sketches another scenario, the interacting driver model, a combination of the driver and passenger model. Here exists a synergistic effect of habitat destruction and aggressive invasive species creating a positive feed back loop, fuelling invasion and habitat destruction. The model predicts that the removal of invasive non-native plants will result in an increased dominance of certain native species, but a decrease of other natives.

2.4.1.2 Influences of invasive non-native plants on soil properties

Invasions of non-native plants may have obscure effects on soils, but these alterations might change the future invasibility (the level to which an existing community is susceptible to new species) and invasiveness (ability to invade) of species (Ehrenfeld and Scott, 2001). Ehrenfeld (2003) suggests that non-native invasive plants, as in the driver model, increase their invasiveness by altering nutrient availability in the affected ecosystem. It would also appear that increased nutrient availability will promote invasions of aggressive species (Ehrenfeld, 2003). Nutrient cycling may also be altered as a result of the changes in physical soil properties due to changed vegetation (Kelly, 1998). Invasive plants can affect soil nutrients and carbon dynamics through various mechanisms such as N-fixation, different photosynthetic pathways and plant structure (Ehrenfeld, 2003). These mechanisms affect how plants interact with the physical and biological components of soil.

The effect invasive species have on natural nutrient dynamics will depend on how different their traits are from the native community's traits (Ehrenfeld, 2003). In a review of the impacts of 56 non-native plant species, Ehrenfeld (2003) concluded that invasive non-native species tend to have a higher standing biomass, net primary production, growth rates and shoot:root ratio (greater aboveground C allocation) than the species they replace. Furthermore, invasive non-native species tend to produce litter that decays faster than native litter, increase exchangeable inorganic nitrogen, nitrogen mineralization and nitrification (Ehrenfeld, 2003). Non-natives also disrupt the timing of nutrient fluxes and spatial distribution of soil pools (Ehrenfeld, 2003).

2.4.2 Fynbos invasions

2.4.2.1 Influences of Acacia invasion on soil properties

Nitrogen fixing invaders such as *Acacia* species can fundamentally change resource availability and the competition balance between native and non-native plants, resulting in altered community structure and ecosystem functioning (Lockwood *et al.*, 2007). Yelenik *et al.* (2004) studied the effect of a nitrogen-fixing *Acacia* species, *Acacia saligna*, on Sand Plain Fynbos. The *Acacia* stands increased nitrogen cycling rates within the low-nutrient environment of fynbos. They found that although the soil temperatures at 5cm depth were warmer for cleared than invaded stands, this did not affect the rate of mineralization between intact and cleared *Acacia* stands. Soils from *Acacia* infestations had a higher nitrogen concentration, with no concentration difference between infested and cleared *Acacia* sites. Litter fall was four times greater in the *Acacia* stands than in natural fynbos on similar soils. *Acacia* litter had double the nitrogen concentration, resulting in a 10 times greater nitrogen return to the soil. Phosphorus, calcium, magnesium and potassium were found at higher concentrations in soil under *Acacia* than soil under fynbos. Soil pH decreased in soils that were affected by *Acacia* trees (Montgomery, 2001).

Musil and Midgely (1990) found similar results on Sand Plain Fynbos. *Acacia saligna* infestation doubled the average concentration of calcium, magnesium, potassium, total nitrogen, manganese, boron and zinc. Sodium, iron, copper and available phosphorus showed higher concentrations in the infested vegetation, but were not statistically significantly different. Burnt plots showed a significant increase for soil pH, calcium, manganese and available phosphorus. These increased concentrations lasted for 3-6 months after the fire. Soil pH, potassium, sodium, iron, zinc and available phosphorus varied significantly during the year. Lately Jovanovic *et al.* (2009) also found an increase in nitrogen where *Acacia saligna* invaded Sand Plain Fynbos.

Musil and Midgley (1990) ascribe the change in soil chemical status to the great amount of relatively nutrient-rich *Acacia* leaves decomposing at a faster rate than the leaves coming from indigenous sclerophyllous vegetation. Musil and Midgely (1990) state that burning these invasive species standing, with their leaves, would eliminate the effect of these decomposing leaves created by other methods such as mechanical and chemical control.

2.4.2.2 Effects of Acacia invasion on fynbos vegetation structure

Fynbos is especially susceptible to invasion of *Acacia*, *Hackea* and *Pinus* species as they have the capacity to spread into natural systems independently of humans (Roura-Pascual, 2009). Non-natives take advantage of window periods after fire through their highly efficient seed dispersal and high fire resilience (Richardson and Cowling, 1992). Invasion of fynbos by tree species will affect fuel properties, which will influence the fire regime and, as fynbos is fire dependent, will affect community composition (Van Wilgen and Van Hensbergen, 1992).

Richardson and Cowling (1992) found a decrease in native plant richness at a local and community level where non-natives have invaded Mountain Fynbos. An increase of non-native tree canopy led to a decrease of native plant cover (Beater *et al.,* 2008; Fourie, 2008). Galatowitsch and Richardson (2005) found that *Acacia* invasion and clearing led to a reduction in fynbos species richness. Furthermore, Beater *et al.* (2008) found that clearing large woody invasive species for a decade led to a decrease of large non-native trees (>5m), but an increase in small trees (<2m). This was accompanied by a decrease of grass and herbaceous cover, and an increase in bare soil and leaf litter under the trees.

Fynbos invasion by non-native trees decrease the potential for self rehabilitation from seed bank (Holmes, 2002; Vosse *et al.,* 2008). Holmes (2002) and Vosse *et al.* (2008) ascribe this to a decrease in native species represented in the soil stored seed bank after long periods (40 years) of invasion. The remaining native seed bank mainly consists of herbs, with a decrease in shrub representation (Holmes, 2002). Fisher *et al*. (2009) also found an increase in non-native seeds where there is a decrease in native seeds. Musil (1993) found that all indigenous plant forms regenerated less successfully in *Acacia* infested areas compared to neighbouring fynbos after fire. Musil and Midgely (1990) ascribed this to the reduced seed production by parent plants as an effect of increased soil nutrients. Further, Richards *et al.* (1997) found that soil nutrients (nitrogen and phosphorus) mainly determined fynbos species distribution. This might eliminate species that were previously colonising the area by not replacing the non-natives as there is a lack of suitable native propagules that can grow under the altered soil conditions.

The increase of nutrients after *Acacia* invasion may favour ephemeral species as they can utilise the increased nutrients (Witkowski and Mitchell, 1989). Similarly, Yelenik *et al.* (2004) found that increased nitrogen availability after *Acacia* invasion and clearing led to higher growth rates of weedy grass species. This might influence restoration of native vegetation negatively as fynbos is shade intolerant, reducing the success of fynbos seedling establishment (Yelenik *et al.*, 2004). Musil (1993) found that the seedling:parent ratio of native species decreased in *Acacia* infested areas, which can lead to the extinction of native species as parents are not being replaced. Proteoid growth forms were the most sensitive to *Acacia* invasion, followed by ericoid and restiod growth forms (Musil, 1993). Musil also found that shoot and root dry weight of native seedlings were unaffected by the nutrient increase under *Acacia* trees. Furthermore the shoot:root ratio of native plants increased, which was attributed to a reduction in light within the *Acacia* stand (Musil, 1993).

Non-native seeds have a competitive advantage as they germinate faster than native seeds, giving them dominance from early life stages (Fisher *et al*., 2009). Bernhard-Reversat (1999) found an allelopathic effect (a plant produced biochemical that negatively influence the growth and development of other organisms) of litter leachates of *Acacia auriculiformis* in the root growth of rice seedlings. The leachates of litter had a shortening and thickening effect on the roots. No effect was observed on above ground growth forms (Chou *et al.,* 1998). This shortening of the roots might make a seedling less fit to establish and lower its competitive ability.

Acacia species prove to have various influences on chemical soil properties and vegetation structure. The greatest effect of *Acacia* infestations on Sand Plain Fynbos soils were shown to be increased nutrients and increased nitrogen cycling rates (Musil and Midgely, 1990; Yelenik *et al.*, 2004). This is due to the higher biomass and increased leaf litter decomposition. *Acacia* trees are highly competitive and reduce available resources. This invasion by *Acacia* species replaces native shrubs and restios with tall non-native trees. The enriched soil conditions can favour growth of herbaceous plants, and allelochemicals from *Acacia* trees can influence the re-growth growth of native plants (Witkowski and Mitchell, 1989; Bernhard-Reversat, 1999). The native seed bank decrease on invaded slopes, limiting the potential for self rehabilitation after clearing.

2.5 Soil erosion processes in Mediterranean regions

Soil erosion is a dynamic process that is affected by various factors. The invasion of fynbos by *A. mearnsii* leads to a change from a shrubby to a tree dominated vegetation structure. As these trees use large quantities of water, they were removed by the Working for Water programme. Where these trees were removed, soil remains bare for several years. This change in vegetation structure and cover can increase the risk of soil erosion. Soil erosion leads to a loss of valuable topsoil and sedimentation of downstream impoundments. It is therefore important to understand what effect invasion and clearing have on soil erosion. Some of the main soil erosion factors and their effects are reviewed below.

2.5.1 Vegetation

Vegetation structure is one of the key factors affecting hydrology and sediment production, and it is very dynamic, changing from season to season (Mitchell, 1990). Vegetation cover correlates well with plant form and biomass, with trees having the highest cover followed by shrubs and then the degraded sites (Francis and Thornes, 1990). The sparse vegetation cover on degraded sites explains their high erosion rates. Sala and Cavlo (1990) found a similar trend where shrubland had a higher erosion rate than woodland. Small shrubs, however, proved to be the best retaining agents for sliding debris (Sala and Cavlo, 1990). However, Vacca *et al.* (2000) found an increase in erosion under eucalypt trees, compared to shrub, and relates it to the reduced ground cover under the trees.

Vegetation cover is another key factor influencing soil erosion. Sala and Cavlo (1990) found that runoff was well correlated with rainfall intensity, but there was some spatial variation due to differences in surface material and vegetation cover. Thinning of vegetation led to an increase in runoff (Lesch and Scott, 1997). This is due to a reduction in leaf area that can intercept rainfall. Another result of a reduction in vegetation cover is increased soil crusting (Maestre *et al.*, 2002; Mills and Fey, 2004). Crusting reduces infiltration and increase runoff (Maestre *et al.*, 2002). Erosion rates were lowest in areas with undisturbed vegetation (López-Bermúdez *et al.*, 1998). Where plant cover was decreased, soils had less fine particles as a result of the loss of finer particles through erosion (Xu *et al.*, 2009). Bryan and Campbell (1986) found that most sediment in a river system originates from degraded areas, areas where vegetation cover had been dramatically reduced.

Vegetation density plays an important protective role of reducing effective rainfall intensity, and this effect increases as vegetation becomes denser (Faulkner, 1990). An increase in plant density increases the irregularities that reduces runoff and erosion, and reduces the interconnectedness of the sediment transport system (Sala and Cavlo, 1990). López-Bermúdez *et al.* (1998) found an inverse relationship between litter cover and soil erosion. Litter acts as storage for rain, reducing runoff and erosion (Sala and Cavlo, 1990; Xu *et al.*, 2009).

2.5.2 Rainfall volume and intensity

High rainfall volumes cause soils to become saturated. Saturated soils will increase runoff, and result in increased erosion (Vacca *et al.*, 2000). Francis and Thornes (1990) found that the relationship between vegetation, runoff and erosion is intensity dependent. High intensity rainfall reduces the time water has to infiltrate and results in increased runoff. The link between rainfall intensity and erosion is clear in the work of Chirino *et al.* (2006) and Xu *et al.* (2009), with rainfall volume also playing a key role in erosion. Furthermore, Faulkner (1990) and Gallart *et al.* (2002) concluded that bare sites would generate runoff during average rainfall events, whereas none of the vegetated sites produced runoff during average events, only during high intensity storms. This was similar for the study done by Francis and Thornes (1990), which had very low erosion rates for low intensity rain for all vegetation cover scenarios, including degraded sites. High intensity rainfall had an enormous erosion effect on bare slopes, decreasing exponentially as cover increased (Francis and Thornes, 1990). Annual sediment yields are highly variable, with the majority of sediment exports occurring during the high intensity rain events (Scott *et al.*, 1998).

2.5.3 Water repellency

Soils that are water repellent decrease infiltration and increase runoff (DeBano, 1971). Soils become more water repellent as hydrophobic organic substances cover particles and reduce the attraction between soil and water (DeBano *et al*., 1967). According to DeBano (1981), organic material that is in contact with soil for long periods, especially large soil particles such as sand grains, will increase water repellency. Furthermore, fire increases soil water repellency, an effect that can last for years. Scott and Van Wyk (1990) found a positive relationship between fire intensity and the degree of repellency. Fires oxidises organic material in surface soils, breaks down soil aggregates, which in turn changes physical characteristics of soil, increasing erosion potential as particles are more mobile and plant cover is reduced (Kruger and Bigalke, 1984; Scott and Van Wyk, 1990). Extreme surface soil temperatures (>300˚C) during fires will reduce the organic content of the upper most layer of soil and reduce the water repellency (DeBano, 1981). This leads to a thin wettable layer overlaying a highly water repellent layer (DeBano, 1981; Scott and Van Wyk, 1990; Scott and Van Wyk, 1992). During a rain event this top wettable layer can become water saturated, and slide on the repellent layer (DeBano, 1981; Scott and Van Wyk, 1990; Scott and Van Wyk, 1992). This can lead to major increases in erosion on slopes following hot fires.

Strong water repellency can develop in soils without the heating of fires (Scott, 2000). Scott (2000) found a strong increase in water repellency in soils under *A. mearnsii* plantations, compared to soils under fynbos. Scott (2000) argue that canopy cover and plant litter in forests might disguise repellency, but once removed will lead to severe increases in soil erosion during rain storms. Scott (1993) found a slight increase in water repellency in soils under fynbos after fire, but a strong increase in repellency under non-native invasive trees. Increased erosion after fires leads to a drastic increase in the export of nitrogen and phosphorus from surface soils in afforested areas (Thomas *et al.*, 1999).

2.5.4 Runoff

Scott *et al.* (1998) found that sediment yields increased exponentially with an increase in runoff. This is due to the increase in erosive energy with increased runoff (Scott and Van Wyk, 1992). Runoff and sediment yields increased for a short period (less than a year) after a mild fire in Swartboskloof and Bosboukloof, a fynbos mountain catchment and pine afforested fynbos mountain catchment respectively (Scott and Van Wyk, 1992). Furthermore, steeper slopes resulted in increased runoff and sediment yields. Scott and Van Wyk (1992) found an increased sediment yield and runoff after fire on a local scale. However, at a catchment scale, influences were minimal due to sediment retention by good vegetation cover in the riparian zones. Thus, soil movement takes place on the steeper slopes, with deposition on the more gentle slopes.

2.5.5 Aspect

Aspect plays a major role on vegetation response and infiltration patterns (Faulkner, 1990). Kirkby *et al.* (1990) found that south facing slopes had significantly less biomass than north facing slopes in the northern hemisphere. This is due to the shaded effect on the north facing slopes, which reduces surface temperatures and increases moisture availability for plants (Kirkby *et al.*, 1990). Plant biomass increases with the increase in moisture availability (Kirkby *et al.*, 1990). Cerda *et al.* (1995) and Andreu *et al.* (2001) found increased soil aggregate stability, lower soil compaction and soil erosion rates on north facing slopes. In the southern hemisphere the inverse is noticed, north facing slopes are warmer, drier, have less biomass, higher soil aggregate stability and compaction, and be more susceptible to soil erosion than south facing slopes. An increase in biomass on the soil surface will increase infiltration, making the south facing slopes even more plant friendly.

2.5.6 Erosion rates

Average erosion rates for undisturbed mountain catchments in the south-western Cape are very low, less than 1 tonne/ha/year (Scott *et al.,* 1998). Scott (1993) recorded a rate of 0.01 tonne/ha/year for an unburned fynbos slope. Kosmas *et al.* (1997) recorded an average erosion rate of 0.067 tonne/ha/year for 95 locations on intact Mediterranean vegetation. Fire in fynbos resulted in increased rates that ranged from 0.1 to 7.4 tonne/ha/year (Scott *et al.,* 1998). Euston-Brown, (2000) found that Table Mountain Sandstones proved to be highly susceptible to erosion after vegetation changes.

In afforested fynbos catchments Scott *et al.* (1998) recorded rates ranging between 0.016 and 0.156 tonne/ha/year on slopes. Kosmas *et al.* (1997) recorded average erosion rates of 0.238 tonne/ha/year on afforested slopes in the Mediterranean region. Logging in these afforested fynbos catchments increased the rates to between 0.14 and 1.6 tonne/ha/year (Scott *et al.,* 1998). Large slash piles resulted in hot fires that increased the erosion rates tenfold ranging from 10 to 26 tonne/ha/year (Scott and Van Wyk, 1990).

The erosion rates mentioned above were measured using large runoff plots ranging in size from 8 x 2 meters to 20 x 8.33 meters (Scott and Van Wyk, 1990; Scott, 1993; Kosmas *et al.,* 1997; Scott *et al.,* 1998). The erosion rate for a plot was determined by dividing the weight of sediment produced by the plot area. This rate was then upscaled to tonne per hectare to give a catchment sediment yield. The sediment yields on large plots are relatively low compared to smaller plots, as sediment storage plays a bigger role on large plots. De Luis *et al.* (2001) suggest that erosion rates from small sized erosion plots should be used as an indication of soil mobilisation instead of overall down slope erosion rates.

2.6 Relevance of this study

This theoretical background provides an important perspective to this study, which aimed to determine the influence of the *A. mearnsii* invasion and clearing on the physical and chemical soil properties, vegetation cover and structure in the Kouga Mountains. The research provided a baseline indication of the current degradation of the ecosystem. Research also addressed the difference between invaded and cleared sites and the surrounding relatively natural sites. Additional research considered the response of native plants on altered soil conditions by means of germination tests and seedling establishment trials. Erosion was also considered in this study, as slopes remained bare for long periods after clearing of non-native vegetation. It is a concern that this can lead to increased soil erosion.

The majority of the work on fynbos from the literature was done in the Western Cape during the 1980s up till recently by Witkowski, Cowling, Le Maitre, Holmes and Rebelo. The focus of the work on fynbos shifted from ecology in the early research to invasion and rehabilitation in the latest work. Much of the work during that period in the Western Cape was done by Scott and Van Wyk. The majority of the erosion literature was drawn from the Mediterranean, covering a time frame from the 1980s up till recently.

The key points highlighted in the literature are: fynbos is highly specialised and highly susceptible to invasion and degradation, the invasion of fynbos will change the vegetation structure and decrease the ability for self-rehabilitation, invasive trees will change the physical and chemical soil character and soil mobilisation will increase on disturbed slopes. These key trends from the literature can be used to interpret the results obtained in this study. Results from this study give us insight into how the Kouga ecosystem is affected by *A. mearnsii*, and what effects can be expected for future rehabilitation.

Chapter 3: Research methods

3.1 Introduction

The aim of this study (section 1.3.1, page 3) is to assess how *A. mearnsii* invasion and the clearing of this species affect fynbos recovery through its impact on soils. This was to be achieved through the following objectives: characterise vegetation and soil properties for natural, infested and cleared sites; analyse the soil–vegetation interaction; relate germination and plant growth to different soil conditions and quantify the effect of wattle infestation/clearing on soil erosion. The research hypotheses that were tested were as follows:

- *A. mearnsii* invasion and clearing thereof changes: soil surface hardness, water repellency, % litter cover, % bare ground, pH, N, P, K, C, Na, Ca, Mg, Zn, Mn, Cu, B, CEC, the number of native species or native plant cover.
- *A. mearnsii* invasion, and the clearing thereof, decrease germination and increase plant growth.
- *A. mearnsii* invasion and clearing increase erosion.

Various methods were used to test these hypotheses. The methods used to locate sampling plots, to sample and analyse soil and vegetation, measure plant establishment performance and monitor soil erosion are described below.

3.2 Site selection and sampling design

This study focussed on four sites (river reaches), each with a different clearing history (refer to Figure 1.2 for the location of the rivers). The ergodic hypothesis, where time is substituted for space, was used (Thornes and Brunsden, 1977; Phillips, 2006). Slopes that were cleared at different times were assumed to be exposed to similar environmental conditions and therefore will follow the same evolutionary changes over time. Samples from slopes with different times since clearing represent the effect of time on soil properties. The rivers were similar in their valley form, with the following clearing histories:

- Heuningnes River - a densely invaded river.
- Baviaans River - a river with slopes that were densely invaded before the first clearing in 1997. It was completely reinvaded and received a

second clearing in 2007. Active restoration has taken place since then.

- Tipi Tributary - a river with slopes that were not densely invaded. The river was cleared in 1997 and was followed up since then.
- Witteklip River - a river with slopes that were densely invaded before it was cleared in 1997. The clearing has been frequently followed up since 1997.

One of the main challenges of this study was what sampling strategy to use, so that fynbos and disturbed sites could be compared, in this highly variable landscape. After much consultation with experts (Anthony Mills, Richard Cowling, Saskia Fourie, Kate Rowntree and James Gambiza) it was decided to use a paired sampling design. The pairing was chosen in order to minimise the variability between the two plots that were being compared. A natural fynbos plot (reference) was paired with an adjacent downslope invaded or cleared plot (Figure 3.1) having the same slope gradient and aspect. None of the plots were in the riparian vegetation. It could therefore be assumed that there were no inherent differences between the two plots before the invasion. Having the reference plot upslope from the invaded or cleared plot would minimise reference plot contamination.

A number of (90) potential paired plots were selected based on oblique aerial photographs taken from a helicopter in 2008. A Garmin Etrex GPS was used to record the position of each photo at the time of the flight. The photos and 1:50 000 topographic maps were carefully inspected in order to locate the areas that had the highest potential for this study. The GPS coordinates were used to navigate to the potential areas on the ground. On-the-ground assessments were then used to select the best positioning for a paired plot. The criteria for choosing a paired plot were:

- A clear, visible boundary line between natural and invaded or cleared vegetation.
- The slope configuration (angle and curvature) and rockiness were similar for the paired plots. Care was taken not to select a paired plot on a drainage line, as geomorphologic processes typical of these land forms might influence soil characteristics.

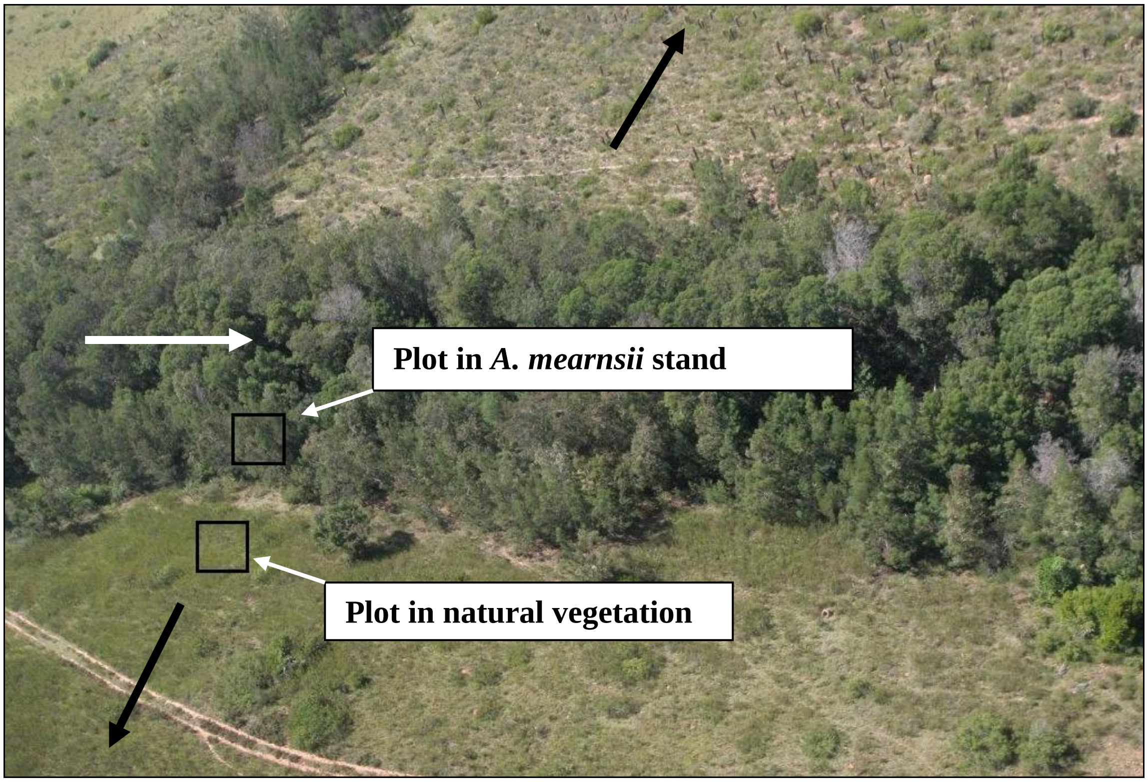

Figure 3.1 An oblique image of the invaded Heuningnes River indicating a paired plot. The black arrows indicate the slope direction and the white arrow the flow direction.

The distance between the two plots in a pair depended on the vegetation transition. In some cases the boundary between fynbos and degraded vegetation was sharp (a few meters) to cases where the transition was about 25 meters wide.

A total of 20 paired plots were selected per river, on alternating banks, ten on the left hand slope and ten on the right hand slope. This resulted in 40 plots per river for four rivers, thus 160 plots in total. Care was taken in the layout of the paired plots to prevent pseudo replication. Each pair was hydrologically independent, with a different aspect and slope. The paired plots were located more or less 100 meters apart, covering a reach of 2000 meters per river. A section of river indicates the design in Figure 3.2.

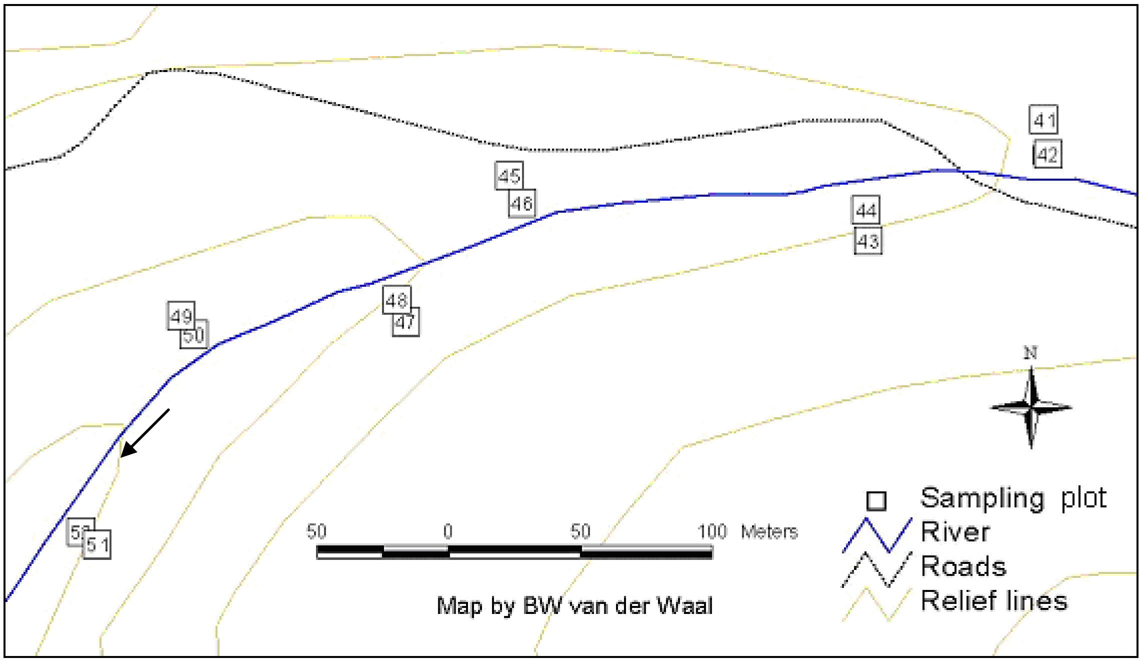

Figure 3.2 Map indicating 6 paired plots in the Tipi tributary. The arrow indicates flow direction.

A 5 x 5 m plot was laid out on the selected area, marked at its centre with a numbered tag. The location of a plot was recorded using a GPS. All the soil and vegetation variables were measured in the 5 x 5 m plots.

3.3 Soil characterisation

3.3.1 Field methods

Field measurements were done during a dry period, with a maximum rainfall event of 1.5 mm in a day, in September 2008.

3.3.1.1 Plot description

For each of the plots, slope, aspect, soil surface hardness and soil depth were measured. The slope of each plot was measured using an Abney level. The aspect was measured using a compass. Soil surface hardness was measured using a penetrometer; an average was calculated using nine readings per plot. Soils were dry during all measurements. These nine readings were spread out over the plot in an x shape, one measurement in the centre, followed by 2 measurements 1.2 m apart towards each of the corners. Soil depth was measured by hammering a pointed steel rod into the soil, with a diameter of 12 mm, until something solid stopped any further penetration (Richards *et al.* 1995). The penetration depth was measured on the rod with a tape measure.

Readings were taken in a similar layout as for surface hardness. Surface stoniness was estimated by comparing the stone cover within a plot to a diagram indicating a range of cover percentages (Hodgson, 1976).

3.3.1.2 Soil collection

A total of 160 bulked soil samples were taken using a 75 mm diameter bucket auger. A bulked sample was collected from each plot. This bulked sample consisted of five bucket loads that were excavated up to a 50 mm depth where possible. The soils were very shallow on the slopes, to maintain consistency all samples were taken to a 50mm depth. The five samples were collected in a similar x layout, one in the middle of the plot, and the other four 1.2 m from each of the corners. Each bulked sample was thoroughly mixed with a steel spatula, and then spread out on a large sheet of paper and left to air dry for 48 hours at room temperature.

Bioassays required a large volume of soil. For this reason a bulked sample of about 40 kg was taken at three of the paired plots, or six plots per river. This resulted in 24 large bulked samples with a total weight of about a tonne. These bulked samples were selected from north facing slopes as they potentially had enough soil to satisfy that quantity. Three sites were selected so that one would fall in the upper, middle and lower part of each selected river reach. These samples were taken in the 5 x 5 m plots, auguring approximately every 25 cm, in a grid pattern, where the surface allowed. The soil was put through a 3 mm garden sieve to dispose of all rocks and stones. Sieving was done in the field in order to reduce the weight of the soil that had to be carried out of the valley in backpacks.

3.3.2 Laboratory methods

3.3.2.1 Soil chemical analysis

The dried samples were sieved using a 2mm sieve, and a 200g sub-sample was sent to BemLab, Somerset West, for chemical analysis. BemLab used standard methods as described in Appendix 1 to analyse the soil for: total nitrogen (N) using a nitrogen analyser, available phosphorus (P) using the Bray II extraction method and inductively coupled plasma optical emission spectrometry (ICP-OES), organic carbon (C) using the Walkley-Black method, boron (B) using ICP-OES for a hot water extraction and cation exchange capacity (CEC) using an ammonium acetate extraction. Potassium (K), sodium (Na), calcium (Ca) and magnesium (Mg) were analysed by ICP-OES using ammonium acetate extractions from the soil samples. Hydrochloric acid extractions and ICP-OES were used for copper (Cu), zinc (Zn) and manganese (Mn) analysis. The pH was determined using a potassium chloride suspension and a pH meter.

3.3.2.2 Soil physical analysis

Particle-size and water repellency were measured in the Rhodes University soils laboratory. Particle-size analysis was done using a combination of sieving and hydrometer analysis. The soil was pre-treated with hydrogen peroxide as described by Gee and Bauder (1986). The hydrometer method was used to determine the clay and silt fractions, and sand (fine, medium and coarse) fractions by sieving as described in Appendix 2.

Repellency was measured by using the water drop penetration time (WDPT) method (Wessel, 1988). This method was used by Doerr *et al.* (2006) in New South Wales on quartsitic sandstone, which is similar to the soil of the Kouga Mountains. Three droplets of deionised water were placed on the soil sample surface (Doerr *et al.,* 2006). The time it took for each of the droplets to be completely absorbed was recorded. An average was calculated per sample.

3.4 Soil erosion on fynbos, invaded and cleared slopes

Erosion pins were utilised to measure erosion. The short nature of the project only allowed a 19 month monitoring period. The erosion was measured in a paired fashion, with erosion plots within the natural and infested or cleared vegetation. Paired plots were situated on similar slope angles. Paired plots were situated on both sides of the river, at three different locations along a river. The fynbos-invaded erosion plots were situated along the Heuningnes River (28 plots, 14 fynbos and 14 invaded), and fynbos–cleared along the Witteklip River (24 plots, 14 fynbos and 14 cleared). A third set of erosion plots was set up along the recently (Spring 2007) cleared Baviaans River (35 plots, 17 fynbos and 18 cleared). The plots along the Baviaans River were established 20 October 2007 and the plots along the Witteklip and Heuningnes Rivers 12 March 2008.

The erosion pins were made using 3.2 mm arc welding rods, with a length of 340 mm. These rods were painted with white enamel paint. An isolation tape flag was stuck near the end of the rod to increase their visibility. These transformed welding rods were renamed as erosion pins. Nine pins were placed in a square (3 x 3), 50 cm apart. The soil is very rocky and pins had to be moved further or closer than the desired 50 cm in order to find a spot where the pin would go into the soil. The pins were hammered in perpendicular to the soil surface until standing firm. The distance from the top of the pin to the soil surface was measured. Organic material (litter) was moved in order to reach the soil surface to do the measurement, but was replaced afterwards. All measurements were done on the down slope side of the pin. The orientation of the plot was determined by facing down river with the plot in front of the observer. The pins in a plot were measured from the top left moving along to the top right, moving to the middle left, continuing this pattern to the bottom right pin. The pins were measured every 18 weeks.

The former height measurement was subtracted from the latest measurement so as to indicate the height of the soil lost or gained for a specific pin for that period. The average soil loss depth for the erosion plot was calculated. This soil loss depth was converted to sediment yield volume per hectare, by multiplying the depth by the area of a hectare. The volume per hectare was converted to tonne per hectare by multiplying the volume by 1.6 (bulk density of sandy soil). The tonne per hectare value was converted to a rate by multiplying by 365 over the number of days that the measurement was done over. This gave a yield unit that was comparable to units from the literature.

The pins were installed on the 14th of January 2008 in the Baviaans River and on the 11th of March 2008 in the Witteklip and Heuningnes River. Measurements were done on the 18th of June and 27th of October in 2008 and on the 6th of May and 26th of October in 2009.

Daily rainfall data for the time period of erosion measurement were collected from Onverwacht Farm, five kilometres south of the Baviaans, Tipi and Witteklip River, and ten kilometres south of Heuningnes River. The farm is located in the same main valley as the study sites and is the closest weather station with daily records.

3.5 Vegetation analysis

A floristic study of vegetation was completed, using visual cover percentage as a subjective measure. A cover percentage was assigned to each of the following growth and soil cover forms: trees, shrubs, woody herbs, forbs, geophytes, restios, sedges, grasses, bare soil, rock and plant litter. The cover for different growth forms overlapped, resulting in a total vegetation cover of more than 100 percent for well vegetated plots. Furthermore species were identified for each of the growth forms. Species that had a cover of less than 1% were only recorded, and not identified, as time was a limiting factor. A species list for each plot was used as an indication of species richness.

Vegetation sampling was done after soil collection, using the same plots as for soil collection. A single 2 x 2 meter plot (Vlok, 2008 pers. com.) positioned in the centre of the 5 x 5 meter plot was used for all the growth forms except trees (plants higher than 2.5 meters), where the full plot was used.

3.6 Plant establishment performance on different soils

Bioassays were used to determine the ability of the soil to facilitate seedling establishment. Bioassays were described by Emmens (1969) as the use of a living organism as an experimental medium, in order to observe plant growth response. Seedling establishment was defined as the ability of a seed to germinate and grow into a plant. Seedling establishment was split up into germination and plant growth on soils from different invasion backgrounds. A dicotyledon *Helichrysum umbraculigerum* (*Everlastings*) and a monocotyledon *Themeda triandra* (Red Grass) were used for growth assessments. *Lactuca sativa* (*lettuce*) and *H. umbraculigerum* seeds were used for the germination assessments. *T. triandra* seeds were excluded from the germination assessment as seed viability and quantity was very low. *L. sativa* was used in the germination experiment, as this species is very sensitive to allelochemical agents and therefore acts as an effective indicator (Chou *et al.*, 1998).

3.6.1 Germination on soils from invaded and cleared slopes

Germination on soils from fynbos, invaded and cleared slopes was investigated. Germination percentages gave an indication of the influence of *A. mearnsii* invasion and clearing on the soil's ability to facilitate seed germination. Seeds were germinated in petri dishes as described by Souto *et al.* (2001). Sub-samples from the large 40 kg bulked samples were used. Germination was done using 20.65 g of soil per petri dish. The soil was brought to field capacity by adding 10 ml of water. Whatmann 3MM blotting paper was placed on the soil in order to absorb some of the soil solution and to act as a growth surface.

20 seeds per species were placed on the blotting paper. Each of the 24 large bulked soil samples had ten replicates per species, resulting in 240 germination dishes per species. The dishes were placed in a Conviron CMP 3246 controlled environment system, with a temperature of 25 ˚C and humidity of 60 %. The number of germinated seeds for *L. sativa* was counted after 36, 48, 60, 72 and 96 hours. *H. umbraculigerum* took longer to germinate and was counted after 8, 10, 12 and 14 days. While counting, visual observations relating to root length were made. 14 days was the maximum time as soil borne mould and fungi became a dense mat and made it difficult to count. Germination percentages were calculated for each time interval.

3.6.2 Relative growth rate

The 24 large bulked soil samples were moved to a poly tunnel at the Botany Department of Rhodes University. Each 40 kg bag was thoroughly mixed and split into twenty 1.25 litre plastic plant bags. *H. umbraculigerum* was planted in ten of the bags and *T. triandra* in the remaining ten for each of the 24 soil samples. These 240 plant bags per species were divided into three blocks, two of the blocks had three replicates and one block had four replicates, to make the bags more accessible (Figure 3.3). This resulted in a total of six one meter squared blocks. A sprinkler with an irrigation radius of one meter was installed in the middle of each block. The irrigation was controlled by a Gardena C 1060 profi water computer. The sprinklers were operational every night at 20h00 for 16 minutes, providing sufficient water to keep the soil moisture above wilting capacity. All of the bags were brought to field capacity before planting any seeds or seedlings. This was done by adding 150ml of water to each bag.

H. umbraculigerum seeds are particularly small and seedlings would be difficult to recognise amongst other seedlings germinating from the soil stored seed bank. To overcome this problem, small seedlings from seedling trays were transplanted into each bag. The seedlings were left to establish and where mortality was high new seedlings were transplanted. High temperatures made the transplanting less successful.

Leaf length measurements for *H. umbraculigerum* began once a seedling had seven leaves. The length (mm) of each leaf was measured with a standard ruler. Leaves were measured every 12 days as plants grew very rapidly. Leaf shapes varied for plants grown in soils with different vegetation scenarios (fynbos, *A. mearnsii* infested or cleared). Over a 100 leaves, spanning a wide range of sizes, were harvested at the end of the growth assessment from plants grown in each of the soils with different vegetation scenarios. Specific leaf area was calculated using a fixed camera and WinDias 2.0 software. In order to convert leaf length to leaf area formulas were used. Formulas were derived by fitting a polynomial through the leaf area-leaf length scatter plot. The formula $y = 0.0022x2 + 0.0162x$ was used for plants grown in soils that were collected from plots with fynbos vegetation, the formula $y = 0.003x2 - 0.0105x$ for *A. mearnsii* invaded plots and $y = 0.002x2 + 0.0327x$ for cleared plots. A total leaf area was then calculated per plant. Plants that died during the measuring period were excluded from the calculations.

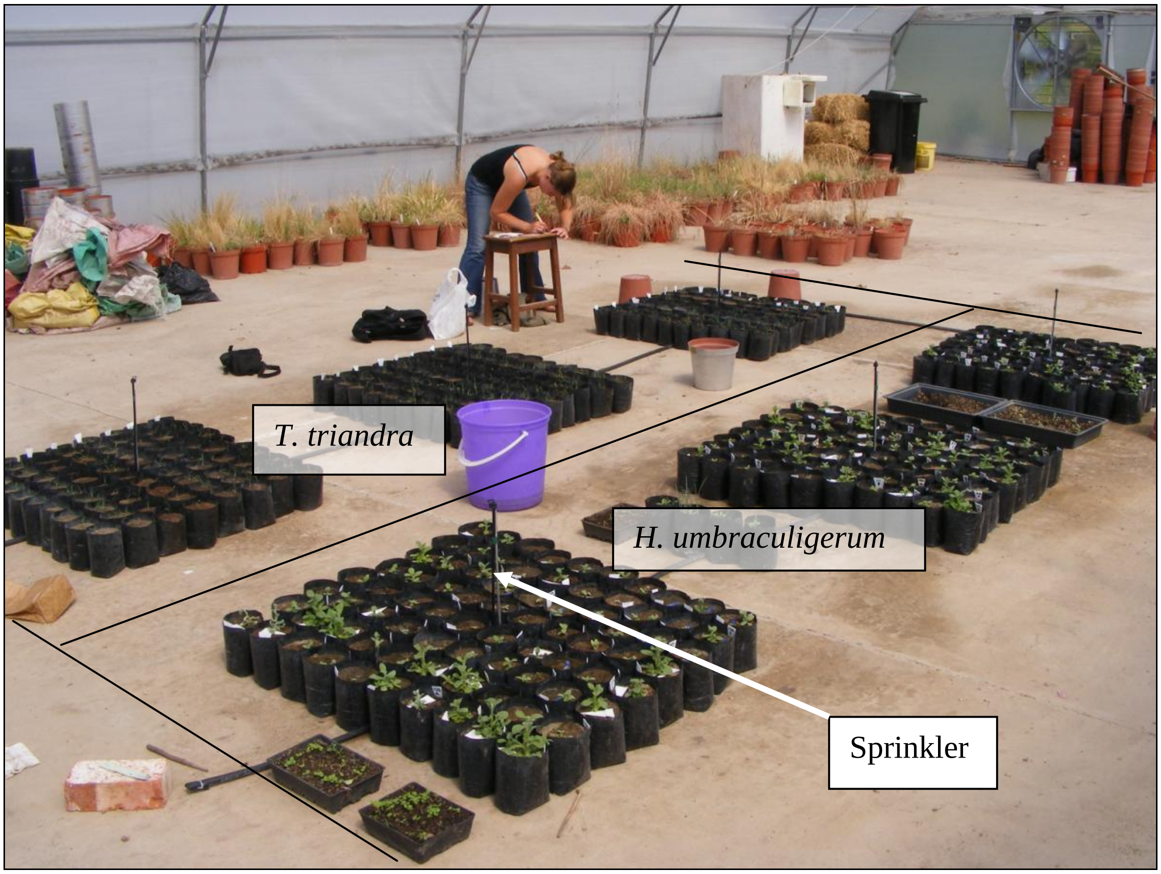

Figure 3.3 Lay out of the 240 bags for *T. triandra* and 240 for *H. umbraculigerum* in the poly tunnel.

T. triandra seeds were planted one cm deep, five per bag and seeds germinated after three days. The smallest and biggest seedlings were eliminated, resulting in similar sized seedlings in all the bags. Leaf length measurements were done every 14 days. Leaf area was determined by measuring each leaf, and using the formula y = 0.0002x2 + 0.0112x to calculate area. The formula was derived in the same way as for the *H. umbraculigerum*. A total leaf area was calculated per plant.

Caterpillars and grasshoppers were present on some plants, thus the pesticide, Blue Death, was used to reduce their effect. Efekto Virikop, a wettable powder fungicide and bactericide, was also used to protect the seedlings against soil borne disease. Plant bags were randomly shuffled in a block once a week. This was done to minimise the micro climate effect that might exist within a block of plants.

H. umbraculigerum was harvested after 48 days and *T. triandra* after 56 days. Plants were washed, dried at 60 °C for 48 hours (Richards *et al.*, 1995) and weighed.

Table 3.1 Summary of the different data sets, indicating the number of sites, number of samples/plots per site and total number of samples/plots.

Variable	No. of sites	Samples/plots per site in fynbos	Samples/plots per site in invaded/cleared vegetation	Total
Soil character (physical and chemical)	4	20	20	160
Vegetation survey	4	20	20	160
Germination	4	3 samples with 10 replicates per sample	3 samples with 10 replicates per sample	240
Growth response	4	3 samples with 10 replicates per sample	3 samples with 10 replicates per sample	240
Erosion - invaded	1	14	14	28
- Recently cleared	1	18	18	36
- Cleared 11 years ago	1	14	14	28

3.7 Statistical Analysis

Statistica 8 was used for the statistical analysis in this project. The natural log transformation was used to stabilise the variances of the physical, chemical and vegetation data. All the transformed variables passed the Saphiro-Wilks test for normality. The data were tested for differences between paired (fynbos and invaded or cleared) plots, using a paired t-test. The null hypotheses that *A. mearnsii* invasion and clearing thereof does not affect: soil surface hardness, water repellency, % litter cover, % bare ground, pH, N, P, K, C, Na, Ca, Mg, Zn, Mn, Cu, B, CEC, the number of native species or native plant cover, were tested using paired t-tests.

The null hypotheses that *A. mearnsii* invasion, and the clearing thereof, decrease germination and increase plant growth were tested using a dependent t-test. The data were stabilised using an arcsin(sqrt(Y/100)) transformation. All the transformed variables passed the Saphiro-Wilks test for normality. A correlation analysis was done for N, P, K and relative leaf area growth rate, for each of the species.

The hypothesis that *A. mearnsii* invasion and clearing increase erosion was tested using independent t-tests. The plot variables were used in a forward stepwise regression so as to tease out which variables explained the soil erosion rates the best. This analysis indicated patterns of different erosion rates for natural, invaded and cleared sites and the key variables that influence erosion.

Chapter 4: Results

4.1 Introduction

This study compared a multitude of physical and chemical properties of soils, vegetation structure, species richness and erosion rates for fynbos slopes and neighbouring *Acacia mearnsii* infested and cleared slopes. The aim was to see if invasion and clearing affect the soil properties and potential for rehabilitation. This was achieved by comparing results for fynbos plots to those for adjacent plots in the following conditions: high density *Acacia mearnsii* infested slopes (Heuningnes), slopes cleared one year ago (Baviaans) of high density invasion and slopes that were cleared eleven years ago with either a low density (Tipi River) or high density (Witteklip River) invasion before clearing. We assumed that time since clearing represent the effect of time on soil properties. Erosion rates are placed within the context of contemporary rainfall intensity data.

A paired t-test was used to compare the fynbos with the invaded or cleared sites (section 3.6).

4.2 External site variables

The paired sampling design was chosen, as it would minimise the effect of spatial variation. Variables that could be expected not to be significantly impacted by invasion and clearing include soil texture, soil depth, rock cover and slope. These variables were therefore analysed to test the null hypothesis that there was no difference between the fynbos plots and invaded or cleared plots.

A comparison of particle size distribution (Figure 4.1) shows that this varies between sites but is similar for the fynbos and invaded or cleared slopes in a specific site. There were no differences in particle size distribution between fynbos and invaded or cleared slopes, except for the reduction ($p<0.05$) in coarse sand on the recently cleared slopes (Table 4.1 -4.4). This is a one percent reduction, and is related to the catena effect, where particles become finer as one moves down a slope.

There were no differences in slope angle or rock cover between fynbos and invaded or cleared plots in each of the rivers (Table 4.1 – 4.4). Soil depths were also similar on different plots, except in the recently cleared river (Baviaans), where soils were deeper on the cleared slopes.

This is what one would expect for normal catena processes, an increase in soil depth as one move down a slope.

These results indicate that there was variation in external variables between sites, but no differences within sites. This low variation within sites indicates that the paired sampling design was an effective way of dealing with the high variability in the study area.

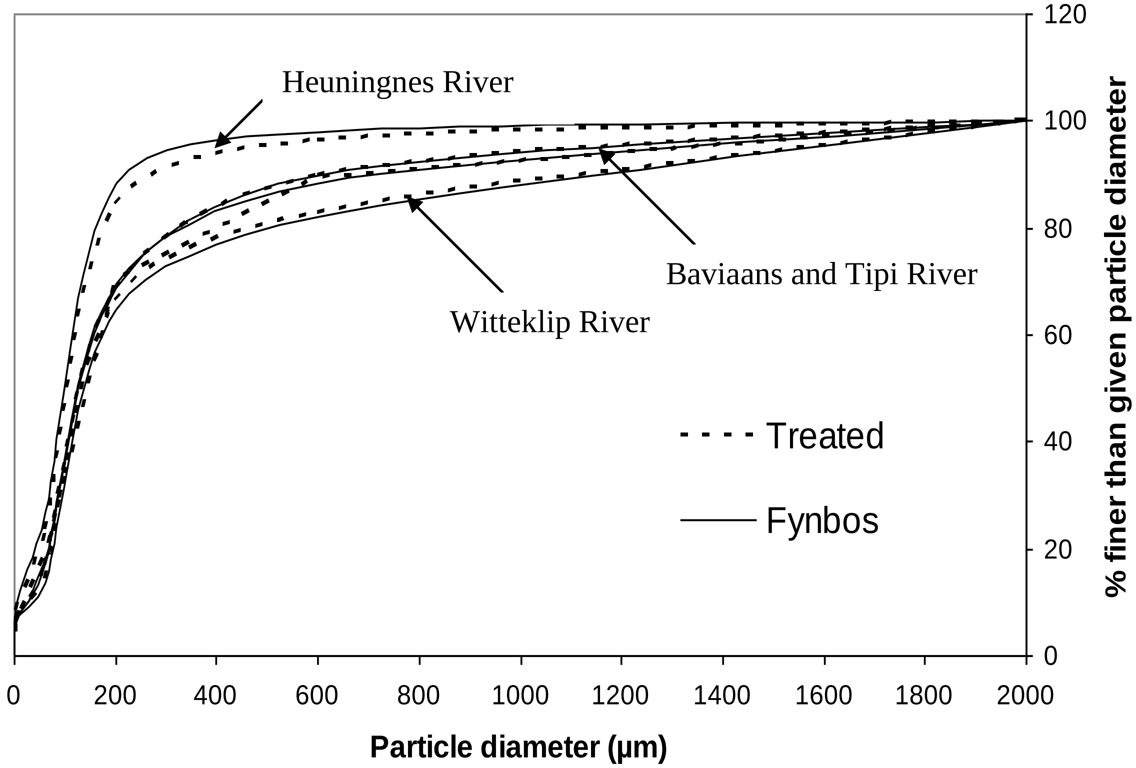

Figure 4.1 Soil particle diameter for fynbos and *A. mearnsii* invaded and cleared slopes in the Heuningnes, Baviaans, Tipi and Witteklip Rivers. The particle size distribution line for plots on treated and fynbos slopes for each river follow each other closely.

Table 4.1 Paired t-tests for external site variables on slopes in the Heuningnes River.

	Fynbos		Invaded					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Slope angle (˚)	16.26	1.80	15.60	1.64	20	0.57	19	0.572
Soil depth (cm)	19.96	9.22	14.45	2.34	12	0.11	11	0.916
% Clay	9.84	0.62	8.40	0.48	20	1.91	19	0.071
% Silt	17.42	1.01	15.30	0.96	20	1.52	19	0.145
% Fine Sand	60.89	2.10	60.73	1.54	20	0.11	19	0.913
% Medium Sand	9.89	1.43	11.99	1.38	20	-1.34	19	0.195
% Coarse Sand	1.96	1.09	3.58	0.33	20	-1.43	19	0.169
% Rock	15.37	4.65	11.95	4.26	20	-1.29	19	0.213

Table 4.2 Paired t-tests for external site variables on slopes in the Baviaans River. Significant changes ($p<0.05$) highlighted with bold print.

Variable	Fynbos Mean	Fynbos Std.Err.	Cleared Mean	Cleared Std.Err.	n	t-value	df	p
Slope angle (˚)	20.47	1.52	19.42	1.48	20	1.38	19	0.185
Soil depth (cm)	**12.96**	**1.37**	**17.13**	**2.05**	**20**	**-4.49**	**19**	**0.000**
% Clay	6.26	0.57	6.53	0.39	20	-0.29	19	0.778
% Silt	12.37	1.10	12.53	0.93	20	-1.09	19	0.288
% Fine Sand	51.67	1.78	51.89	1.57	20	-0.40	19	0.695
% Medium Sand	18.41	1.15	18.91	0.91	20	-0.76	19	0.459
% Coarse Sand	**11.28**	**1.38**	**10.15**	**1.16**	**20**	**2.26**	**19**	**0.036**
% Rock	25.63	6.95	20.84	5.90	20	0.90	19	0.381

Table 4.3 Paired t-tests for external site variables on slopes in the Tipi River.

Variable	Fynbos Mean	Fynbos Std.Err.	Cleared Mean	Cleared Std.Err.	n	t-value	df	p
Slope angle (˚)	23.70	1.40	23.42	1.66	20	1.01	19	0.323
Soil depth (cm)	8.22	0.68	8.97	0.57	20	-1.79	19	0.090
% Clay	6.65	0.43	6.79	0.53	20	0.06	19	0.951
% Silt	10.65	0.91	10.47	0.84	20	0.52	19	0.607
% Fine Sand	51.39	1.48	51.72	1.46	20	0.00	19	0.996
% Medium Sand	20.94	1.38	21.11	1.30	20	-0.84	19	0.414
% Coarse Sand	10.36	0.90	9.91	0.84	20	0.36	19	0.723
% Rock	48.00	4.65	48.68	5.11	20	-1.84	19	0.081

Table 4.4 Paired t-tests for external site variables on slopes in the Witteklip River.

Variable	Fynbos Mean	Fynbos Std.Err.	Cleared Mean	Cleared Std.Err.	n	t-value	df	p
Slope angle (˚)	21.60	1.04	21.80	1.18	20	-0.26	19	0.801
Soil depth (cm)	8.44	0.58	8.63	0.38	20	-1.04	19	0.310
% Clay	6.90	0.67	6.75	0.68	20	0.29	19	0.772
% Silt	6.55	0.55	7.60	0.90	20	-0.24	19	0.816
% Fine Sand	51.20	2.54	51.90	2.36	20	-0.67	19	0.510
% Medium Sand	17.57	1.21	16.52	1.13	20	1.66	19	0.113
% Coarse Sand	17.78	2.13	17.24	2.19	20	0.86	19	0.399
% Rock	56.00	5.77	56.00	6.47	20	0.98	19	0.341

4.3 Change in soil properties

4.3.1 Physical soil properties

4.3.1.1 Change in physical soil properties on Acacia mearnsii invaded slopes

The null hypotheses, which states that *A. mearnsii* does not change physical soil properties, was tested for the variables: surface hardness, water repellency (WDPT), litter cover and bare ground cover. The null hypothesis was rejected for surface hardness, water repellency and plant litter, and accepted for bare ground. It can therefore be stated that invasion did affect penetration, water repellency and plant litter. Slopes invaded by *A. mearnsii* decreased ($p<0.001$) in surface hardness by 38 percent (Table 4.5). Water repellency increased ($p<0.001$) four fold and plant litter cover by 68 percent on invaded slopes (Table 4.5).

Table 4.5 Paired t-tests for physical soil properties in the Heuningnes River. (Water drop penetration time (WDPT)). Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Invaded					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Penetration (kg/cm^{-2})	**8.79**	**0.67**	**5.47**	**0.41**	**20**	**6.10**	**19**	**0.000**
WDPT (s)	**1.82**	**0.39**	**8.80**	**2.26**	**20**	**-4.85**	**19**	**0.000**
% Plant litter	**48.37**	**4.51**	**81.20**	**4.15**	**20**	**-4.98**	**19**	**0.000**
% Bare ground	8.21	2.17	7.90	1.99	20	0.25	19	0.809

4.3.1.2 Change in physical soil properties over time since clearing

The soils on slopes cleared one year ago of high density invasion, had a lower ($p<0.005$) surface hardness and increased bare ground cover if compared to neighbouring fynbos slopes (Table 4.6 - 4.8). The soils on slopes cleared of low density invasion eleven years ago, had a lower surface hardness ($p<0.005$) and litter cover ($p<0.05$) and higher water repellency ($p<0.001$) and bare ground ($p<0.005$) cover than neighbouring fynbos slopes (Table 4.6 - 4.8). The soils on slopes cleared of high density invasion eleven years ago had lower surface hardness ($p<0.005$) and higher water repellency ($p<0.001$) and bare ground cover ($p<0.05$) than neighbouring fynbos slopes (Table 4.6 - 4.8).

The time since clearing for the different sites was used as a surrogate for change over time. The null hypothesis that states that the effect of *A. mearnsii* invasion on physical soil properties decreases over time after clearing was tested for: surface hardness, water repellency, plant litter and bare ground cover. The null hypothesis was accepted for surface hardness, water repellency and bare ground cover, and rejected for plant litter. The difference in surface hardness decreased over time from 30 percent to 24 percent (Table 4.9). The difference in repellency decreased for sites that were cleared a long time ago, if compared to the repellency of the invaded site. The repellency was four and one times higher on the cleared slopes for the Tipi and Witteklip Rivers respectively (Table 4.9). The difference in plant litter cover decreased with time since clearing on cleared slopes, except for the Tipi River, with slopes cleared eleven years ago of low density invasion, where plant litter was lower than on fynbos slopes. The bare ground cover was lowest on slopes that were cleared for the longest time. The difference in bare ground cover decreased from three fold to 38 percent over time since clearing (Table 4.9).

Table 4.6 Paired t-tests for physical soil properties on fynbos and cleared slopes in the Baviaans River. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Penetration (kg*cm^{-2})	**2.09**	**0.13**	**1.46**	**0.09**	**20**	**3.93**	**19**	**0.001**
WDPT (s)	2.56	0.76	2.61	0.51	20	-1.03	19	0.315
% Plant litter	44.47	3.35	52.26	5.25	20	-0.36	19	0.723
% Bare ground	**5.24**	**1.30**	**20.21**	**3.59**	**20**	**-4.09**	**19**	**0.001**

Table 4.7 Paired t-tests for physical soil properties on fynbos and cleared slopes in the Tipi River. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Penetration (kg*cm^{-2})	**3.80**	**0.29**	**3.16**	**0.27**	**20**	**3.44**	**19**	**0.003**
WDPT (s)	**5.68**	**2.46**	**29.00**	**8.89**	**20**	**-12.52**	**19**	**0.000**
% Plant litter	**33.55**	**3.36**	**20.05**	**3.14**	**20**	**2.47**	**19**	**0.023**
% Bare ground	**10.18**	**1.86**	**29.63**	**4.32**	**20**	**-4.10**	**19**	**0.001**

Table 4.8 Paired t-tests for physical soil properties on fynbos and cleared slopes in the Witteklip River. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Penetration (kg*cm^{-2})	**5.86**	**0.78**	**4.43**	**0.59**	**20**	**3.56**	**19**	**0.002**
WDPT (s)	**20.65**	**6.86**	**40.46**	**10.02**	**20**	**-4.35**	**19**	**0.000**
% Plant litter	22.55	3.08	16.40	2.85	20	2.02	19	0.058
% Bare ground	**10.95**	**2.30**	**15.10**	**2.86**	**20**	**-2.33**	**19**	**0.031**

Table 4.9 A summary of the physical soil variables for the cleared slopes. Percentage change is relative to the neighbouring fynbos vegetation. Significant changes ($p<0.05$) highlighted with bold print.

Variable	Baviaans (high density, cleared for 1 year)		Tipi (low density, cleared for 11 years)		Witteklip (high density, cleared for 11 years)	
	Sig. change	% change	Sig. change	% change	Sig. change	% change
Penetration ($kg*cm^{-2}$)	**yes**	**-30**	**yes**	**-17**	**yes**	**-24**
WDPT (s)	no	2	**yes**	**411**	**yes**	**96**
% Plant litter	no	18	**yes**	**-40**	no	-27
% Bare ground	**yes**	**286**	**yes**	**191**	**yes**	**38**

4.3.2 Chemical soil properties

4.3.2.1 The effect of Acacia mearnsii invasion on soil chemical properties

The null hypothesis that *A. mearnsii* invasion does not change the chemical soil composition was tested for the variables: pH, N, P, K, C, Cu, Zn, Mn, B, CEC, Na and Ca. The null hypothesis was rejected for pH, N, K, Mn and Na, and accepted for P, C, Cu, Zn, B, CEC and Ca. The values are given in Table 4.10. N increased by 23 percent, K by 21 percent, Mn by 50 percent and Na by 90 percent.

Table 4.10 Paired t-tests for chemical soil properties on fynbos and invaded slopes in the Heuningnes River. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Invaded					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
pH	**4.90**	**0.07**	**4.52**	**0.06**	**20**	**6.56**	**19**	**0.000**
N (mg/kg)	**1588.95**	**72.74**	**1952.50**	**89.05**	**20**	**-3.70**	**19**	**0.002**
P (mg/kg)	4.37	0.47	5.45	0.86	19	-1.63	18	0.121
K (mg/kg)	**141.37**	**11.84**	**170.20**	**12.60**	**20**	**-2.68**	**19**	**0.015**
C (mg/kg)	23063.16	1415.19	24510.00	1557.26	20	-1.17	19	0.258
Cu (mg/kg)	0.18	0.02	0.19	0.02	20	-0.61	19	0.550
Zn (mg/kg)	0.93	0.18	1.07	0.25	20	-0.28	19	0.785
Mn (mg/kg)	**28.24**	**4.45**	**41.50**	**7.68**	**20**	**-2.96**	**19**	**0.008**
B (mg/kg)	0.43	0.06	0.58	0.07	20	-1.05	19	0.306
CEC (cmol(+)/kg)	5.06	0.52	6.00	0.51	20	-1.90	19	0.073
Na (cmol(+)/kg)	**0.11**	**0.01**	**0.21**	**0.03**	**20**	**-3.82**	**19**	**0.001**
Ca (cmol(+)/kg)	4.70	0.39	4.21	0.30	20	1.50	19	0.151
Mg (cmol(+)/kg)	2.22	0.21	2.15	0.16	20	0.01	19	0.996

4.3.2.2 Change in soil chemical properties over time since clearing

The soils on slopes cleared of high density invasion a year ago had higher acidity ($p<0.005$), N ($p<0.01$), P ($p<0.005$), K ($p<0.05$), C ($p<0.01$) and Mn ($p<0.001$) concentrations than soils on neighbouring fynbos slopes (Table 4.11 – 4.14). The soils on slopes cleared of low density invasion eleven years ago had higher acidity ($p<0.001$) and Mn ($p<0.005$) concentrations, but lower Cu ($p<0.05$), Zn ($p<0.005$), CEC ($p<0.05$), Ca ($p<0.01$) and Mg ($p<0.05$) concentrations than soils on neighbouring fynbos slopes (Table 4.11 – 4.14). Soils on slopes cleared of high density invasion eleven years ago had higher N ($p<0.005$), P ($p<0.001$), C ($p<0.001$), Zn ($p<0.05$), Mn ($p<0.005$) and Ca ($p<0.05$) concentrations than soils on neighbouring fynbos slopes.

Time since clearing was used as a surrogate for the effect of time on chemical properties. The following null hypothesis was tested: the chemical concentration difference in soils on fynbos slopes and slopes cleared of *A. mearnsii* stay the same with time since clearing. This hypothesis was tested for the variables: pH, N, P, K, C, Cu, Zn, Mn, B, CEC, Na and Ca. The hypothesis was rejected for pH, N, P, K, C, Cu, Zn, Mn, B, CEC, Na and Ca, where the difference in concentration changed with time since clearing. There were no differences in concentration for Na, CEC and Mg on slopes cleared of high density.

The difference in pH was lowest for sites that were cleared the longest time ago, except on the cleared slopes in the Tipi River, where the difference in pH was larger than any of the other rivers (Table 4.11 – 4.14). The decrease in difference in the Witteklip River might be due to the high acidity of the fynbos slopes. The difference in acidity decreased from 0.38 units, for recently cleared slopes, to no difference for slopes that have been cleared for the longest time.

The difference in N concentration was higher for sites that have been cleared for the longest time (Table 4.11 – 4.14). The difference in N concentration increased from 26 to 40 percent for slopes that have been cleared for the longest time (Table 4.14). The increase in the N concentration difference can be related to the N concentration of the fynbos slopes in the Witteklip River. The N concentration is lower on these fynbos slopes, compared to the other fynbos slopes, and results in an increased concentration difference for slopes that have been cleared for the longest time period.

The difference in K concentration decreased from 20 percent to no difference for slopes that have been cleared for the longest time (Table 4.11 – 4.14). The difference in Mn concentration decreased from 89 to 39 percent for slopes that have been cleared for the longest time (Table 4.14).

There was a difference ($p<0.005$) of 60 percent in P concentration on slopes cleared of high density *A. mearnsii* infestations, with narrowing difference of seven percent for slopes that have been cleared for eleven years (Table 4.14). Zn and Ca concentrations were 47 and 27 percent higher on cleared slopes in the Witteklip River, with no difference in concentration in any of the other rivers (Table 4.14).

The C concentration was higher on slopes that were cleared of high density invasion (Table 4.11 and 4.13). The C concentration difference was higher for sites that were cleared for the longest time, and ranged from 16 percent for recently cleared to 20 percent for sites cleared eleven years ago.

The drastic increase in acidity on cleared slopes in the Tipi River may have led to nutrient leaching. The nutrient concentration for Cu, Zn, Ca, Mg and CEC were lower on the cleared slopes than on neighbouring fynbos slopes (Table 4.12). Furthermore, the leaching may have reduced the expected increase in N, P, C and Mn so that there were no differences between fynbos and cleared slopes (Table 4.12).

Table 4.11 Paired t-tests for chemical soil properties on fynbos and cleared slopes in the Baviaans River. Significant changes (p<0.05) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
pH	**4.82**	**0.05**	**4.44**	**0.09**	**20**	**3.94**	**19**	**0.001**
N (mg/kg)	**1431.58**	**95.54**	**1810.00**	**140.82**	**20**	**-2.87**	**19**	**0.010**
P (mg/kg)	**4.26**	**0.28**	**7.11**	**0.86**	**20**	**-3.71**	**19**	**0.001**
K (mg/kg)	**128.89**	**8.71**	**155.16**	**16.73**	**20**	**-2.19**	**19**	**0.041**
C (mg/kg)	**20831.58**	**801.13**	**24157.89**	**1149.79**	**20**	**-3.12**	**19**	**0.006**
Cu (mg/kg)	0.12	0.01	0.12	0.01	20	0.28	19	0.779
Zn (mg/kg)	1.03	0.42	0.96	0.25	20	-0.88	19	0.389
Mn (mg/kg)	**10.95**	**0.93**	**20.64**	**1.89**	**20**	**-5.80**	**19**	**0.000**
B (mg/kg)	0.48	0.03	0.51	0.04	20	-0.19	19	0.848
CEC (cmol(+)/kg)	5.80	0.32	6.37	0.37	20	-0.97	19	0.347
Na (cmol(+)/kg)	0.10	0.02	0.11	0.02	20	-0.11	19	0.917
Ca (cmol(+)/kg)	4.00	0.18	4.28	0.36	20	-0.11	19	0.914
Mg (cmol(+)/kg)	1.37	0.09	1.28	0.09	20	1.21	19	0.242

Table 4.12 Paired t-tests for chemical soil properties on fynbos and cleared slopes in the Tipi River. Significant changes (p<0.05) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
pH	**4.58**	**0.07**	**4.07**	**0.04**	**20**	**7.54**	**19**	**0.000**
N (mg/kg)	1526.50	81.44	1792.11	139.76	20	-2.09	19	0.050
P (mg/kg)	4.65	0.37	5.37	0.63	20	-1.74	19	0.098
K (mg/kg)	118.45	6.71	109.79	7.53	20	0.77	19	0.448
C (mg/kg)	21530.00	1156.99	21926.32	1767.77	20	-0.48	19	0.638
Cu (mg/kg)	**0.14**	**0.01**	**0.11**	**0.01**	**20**	**2.44**	**19**	**0.024**
Zn (mg/kg)	**0.64**	**0.13**	**0.46**	**0.10**	**20**	**3.79**	**19**	**0.001**
Mn (mg/kg)	**8.49**	**0.81**	**11.20**	**0.93**	**20**	**-3.26**	**19**	**0.004**
B (mg/kg)	0.42	0.02	0.41	0.02	20	0.25	19	0.803
CEC (cmol(+)/kg)	**4.89**	**0.27**	**3.93**	**0.37**	**20**	**2.33**	**19**	**0.031**
Na (cmol(+)/kg)	0.09	0.01	0.08	0.01	20	0.46	19	0.654
Ca (cmol(+)/kg)	**3.20**	**0.20**	**2.45**	**0.20**	**20**	**2.92**	**19**	**0.009**
Mg (cmol(+)/kg)	**1.26**	**0.08**	**1.03**	**0.08**	**20**	**2.11**	**19**	**0.049**

Table 4.13 Paired t-tests for chemical soil properties on fynbos and cleared slopes in the Witteklip River. Significant changes ($p<0.05$) highlighted with bold print.

Variable	Fynbos Mean	Fynbos Std.Err.	Cleared Mean	Cleared Std.Err.	n	t-value	df	p
pH	4.40	0.11	4.36	0.12	20	0.47	19	0.641
N (mg/kg)	**1285.50**	**97.31**	**1798.00**	**153.02**	**20**	**-3.71**	**19**	**0.001**
P (mg/kg)	**6.70**	**1.01**	**10.70**	**1.62**	**20**	**-7.35**	**19**	**0.000**
K (mg/kg)	109.15	11.39	119.05	10.06	20	-1.27	19	0.220
C (mg/kg)	**19990.00**	**1272.62**	**24005.00**	**1650.75**	**20**	**-4.64**	**19**	**0.000**
Cu (mg/kg)	0.13	0.01	0.12	0.01	20	1.31	19	0.205
Zn (mg/kg)	**0.92**	**0.23**	**1.35**	**0.26**	**20**	**-2.28**	**19**	**0.035**
Mn (mg/kg)	**9.77**	**2.28**	**13.57**	**2.63**	**20**	**-3.88**	**19**	**0.001**
B (mg/kg)	0.33	0.04	0.38	0.04	20	-1.14	19	0.269
CEC (cmol(+)/kg)	3.72	0.48	4.50	0.57	19	-0.78	18	0.444
Na (cmol(+)/kg)	0.09	0.02	0.09	0.01	20	-0.38	19	0.711
Ca (cmol(+)/kg)	**2.73**	**0.28**	**3.46**	**0.36**	**20**	**-2.44**	**19**	**0.025**
Mg (cmol(+)/kg)	1.06	0.11	1.23	0.14	20	-1.38	19	0.184

Table 4.14 A summary of the chemical variables for the cleared slopes. The percentage change for pH was not given as a percentage but in terms of change in units. Significant changes ($p<0.05$) highlighted with bold print.

Variable	Baviaans (high density, cleared for 1 year)		Tipi (low density, cleared for 11 years)		Witteklip (high density, cleared for 11 years)	
	Sig. change	% change	Sig. change	% change	Sig. change	% change
pH	**yes**	**-0.38 units**	**yes**	**-0.58 units**	no	-0.04 units
N (mg/kg)	**yes**	**26**	no	17	**yes**	**40**
P (mg/kg)	**yes**	**67**	no	15	**yes**	**60**
K (mg/kg)	**yes**	**20**	no	-7	no	9
C (mg/kg)	**yes**	**16**	no	2	**yes**	**20**
Cu (mg/kg)	no	1	**yes**	**-21**	no	-9
Zn (mg/kg)	no	-6	**yes**	**-28**	**yes**	**47**
Mn (mg/kg)	**yes**	**89**	**yes**	**32**	**yes**	**39**
B (mg/kg)	no	6	no	-3	no	17
CEC (cmol(+)/kg)	no	10	**yes**	**-20**	no	21
Na (cmol(+)/kg)	no	8	no	-6	no	-1
Ca (cmol(+)/kg)	no	7	**yes**	**-24**	**yes**	**27**
Mg (cmol(+)/kg)	no	-7	**yes**	**-18**	no	16

4.4 The change in vegetation structure

4.4.1 The effect of Acacia mearnsii invasion on vegetation structure

The null hypothesis, that *A. mearnsii* invasion does not decrease the number of species and the cover of fynbos growth forms, was tested. The hypothesis was rejected for number of species.

Furthermore the hypothesis for plant cover was accepted for restios and geophytes but rejected for trees, shrubs, woody herbs, forbs, grasses and sedges. The number of species per plot decreased ($p<0.001$) by 54 percent due to *A. mearnsii* invasion. Furthermore, the tree cover increased ($p<0.001$) 11 fold. This increase in tree cover reduced the shrub cover ($p<0.001$) by 86 percent, woody herb cover ($p<0.001$) by 94 percent, forb cover ($p<0.005$) by 85 percent, grass cover ($p<0.001$) by 90 percent and sedge cover ($p<0.05$) by 94 percent (Table 4.15).

Table 4.15 Paired t-tests for the number of plant species and growth form cover percentages per plot for fynbos and invaded slopes in the Heuningnes River. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Invaded					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Species richness per plot	**17.68**	**1.05**	**8.10**	**0.80**	**20**	**8.35**	**19**	**0.000**
% Trees	**7.87**	**3.08**	**94.75**	**2.32**	**20**	**-10.65**	**19**	**0.000**
% Shrubs	**45.21**	**5.27**	**6.53**	**2.70**	**20**	**6.67**	**19**	**0.000**
% Woody herbs	**14.05**	**3.13**	**0.83**	**0.24**	**20**	**7.57**	**19**	**0.000**
% Forbs	**2.29**	**0.64**	**0.35**	**0.08**	**20**	**3.48**	**19**	**0.003**
% Grasses	**47.63**	**6.08**	**5.23**	**2.06**	**20**	**9.20**	**19**	**0.000**
% Restios	0.26	0.15	0.03	0.03	20	1.00	19	0.330
% Sedges	**1.76**	**1.04**	**0.10**	**0.06**	**20**	**2.77**	**19**	**0.012**
% Geophytes	0.32	0.10	0.08	0.04	20	-0.81	19	0.428

4.4.2 The effect of Acacia mearnsii clearing on vegetation structure

The slopes cleared one year ago showed lower ($p<0.001$) species richness, shrub, woody herb, grass and restio cover than neighbouring fynbos slopes (Table 4.16 – 4.19). The slopes cleared of low density had higher tree cover ($p<0.001$) and lower species richness, shrub ($p<0.005$), woody herb ($p<0.001$), forb ($p<0.05$), grass ($p<0.001$), restio ($p<0.001$) and sedge ($p<0.05$) cover than neighbouring fynbos slopes (Table 4.16 – 4.19). The slopes that were cleared of high density invasion eleven years ago had lower species richness ($p<0.001$), shrub ($p<0.001$), woody herb ($p<0.005$), restio ($p<0.005$) and sedge ($p<0.005$) cover than neighbouring fynbos slopes (Table 4.16 – 4.19).

Time since clearing was used as a surrogate for the effect of time on species richness and vegetation cover. The null hypothesis that species richness and the cover of different fynbos growth forms will remain constant with time since clearing was tested. The null hypothesis was rejected for species richness, as the difference in species richness was smaller for sites that were cleared for the longest time. The difference in species richness decreased from 65 percent for recently cleared sites to 40 percent for sites cleared for the longest time (Table 4.16 – 4.19).

The null hypothesis was accepted for restios, and rejected for trees, shrubs, woody herbs, forbs, grasses and sedges. The difference in tree cover on fynbos and cleared slopes was eliminated on slopes cleared of high density *A. mearnsii* invasions (Table 4.16 and 4.19). The tree cover on the slopes cleared of low density *A. mearnsii* invasion was 24 fold higher (Table 4.19). The difference in shrub cover decreased from 85 percent on recently cleared slopes to 80 percent for sites that were cleared for the longest time period, woody herb from 96 to 46 percent and grasses from 75 percent to no difference (Table 4.16 – 4.19). There was no cover difference for sedges and forbs on the recently cleared slopes, but the difference was lower for sites cleared eleven years ago, if compared to the difference in cover on invaded slopes. The difference for forb cover decreased from 85 percent to no difference and for sedge cover from 94 to 84 percent (Table 4.15 and 4.19).

The possible leaching effect in the Tipi River resulted in a very slow recovery of species richness, woody herb cover, forb cover and grass cover (Table 4.15 – 4.19).

Table 4.16 Paired t-tests for the number of plant species and growth form cover percentages for fynbos and cleared slopes in the Baviaans River. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Species richness per plot	**22.16**	**1.07**	**7.84**	**0.98**	**20**	**9.58**	**19**	**0.000**
% Trees	3.71	1.85	9.45	5.19	20	0.95	19	0.353
% Shrubs	**45.16**	**5.21**	**6.79**	**2.84**	**20**	**6.49**	**19**	**0.000**
% Woody herbs	**24.32**	**2.34**	**1.00**	**0.34**	**20**	**17.22**	**19**	**0.000**
% Forbs	0.92	0.13	0.66	0.13	20	1.90	19	0.072
% Grasses	**42.11**	**5.33**	**10.55**	**3.09**	**20**	**4.83**	**19**	**0.000**
% Restios	**17.53**	**3.38**	**0.58**	**0.33**	**20**	**6.10**	**19**	**0.000**
% Sedges	1.55	0.30	1.95	1.02	20	0.31	19	0.761
% Geophytes	3.50	2.08	0.24	0.08	20	2.03	19	0.057

Table 4.17 Paired t-tests for the number of plant species and growth form cover percentages for fynbos and cleared slopes in the Tipi River. Significant changes (p<0.05) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Species richness per plot	**21.90**	**1.42**	**7.05**	**0.70**	**20**	**9.27**	**19**	**0.000**
% Trees	**0.73**	**0.26**	**17.87**	**3.60**	**20**	**-5.25**	**19**	**0.000**
% Shrubs	**39.00**	**5.60**	**5.50**	**1.86**	**20**	**3.75**	**19**	**0.001**
% Woody herbs	**17.60**	**3.56**	**2.71**	**0.95**	**20**	**4.69**	**19**	**0.000**
% Forbs	**0.90**	**0.23**	**0.11**	**0.05**	**20**	**2.24**	**19**	**0.038**
% Grasses	**32.00**	**5.12**	**7.37**	**1.51**	**20**	**5.25**	**19**	**0.000**
% Restios	**20.40**	**3.09**	**0.11**	**0.11**	**20**	**10.63**	**19**	**0.000**
% Sedges	**6.15**	**3.18**	**0.53**	**0.08**	**20**	**2.44**	**19**	**0.025**
% Geophytes	1.15	0.50	0.06	0.04	20	0.62	19	0.545

Table 4.18 Paired t-tests for the number of plant species and growth form cover percentages for fynbos and cleared slopes in the Witteklip River. Significant changes (p<0.05) highlighted with bold print.

	Fynbos		Cleared					
Variable	Mean	Std.Err.	Mean	Std.Err.	n	t-value	df	p
Species richness per plot	**19.35**	**2.10**	**11.70**	**1.32**	**20**	**6.42**	**19**	**0.000**
% Trees	2.38	1.54	1.18	0.33	20	0.85	19	0.407
% Shrubs	**38.80**	**3.12**	**7.83**	**2.32**	**20**	**6.76**	**19**	**0.000**
% Woody herbs	**17.03**	**3.02**	**9.25**	**2.88**	**20**	**3.50**	**19**	**0.002**
% Forbs	2.05	1.01	3.00	1.97	20	0.37	19	0.718
% Grasses	24.85	4.32	33.88	5.94	20	-0.65	19	0.521
% Restios	**12.15**	**4.42**	**0.20**	**0.11**	**20**	**3.75**	**19**	**0.001**
% Sedges	**3.33**	**0.75**	**0.53**	**0.14**	**20**	**3.65**	**19**	**0.002**
% Geophytes	3.68	1.31	0.43	0.16	20	1.93	19	0.069

Table 4.19 A summary of the number of species and vegetation cover changes that took place on cleared sites. The percentage change is relative to the neighbouring fynbos in that specific river. Significant changes (p<0.05) highlighted with bold print.

Variable	Baviaans (high density, cleared for 1 year)		Tipi (low density, cleared for 11 years)		Witteklip (high density, cleared for 11 years)	
	Sig. change	% change	Sig. change	% change	Sig. change	% change
No. of plant species	**yes**	**-65**	**yes**	**-68**	**yes**	**-40**
% Tree cover	no	155	**yes**	**2365**	no	-51
% Shrub cover	**yes**	**-85**	**yes**	**-86**	**yes**	**-80**
% Woody herb cover	**yes**	**-96**	**yes**	**-85**	**yes**	**-46**
% Forb cover	no	-29	**yes**	**-88**	no	46
% Grass cover	**yes**	**-75**	**yes**	**-77**	no	36
% Restio cover	**yes**	**-97**	**yes**	**-99**	**yes**	**-98**
% Sedge cover	no	25	**yes**	**-91**	**yes**	**-84**
% Geophyte cover	no	-93	no	-95	no	-88

4.5 Seedling establishment on soils from different sites

4.5.1 Germination performance

4.5.1.1 Lactuca sativa germination on soils from different sites

The null hypothesis that soils from *A. mearnsii* infestations and cleared slopes do not affect *L. sativa* germination was tested. There were no significant differences in germination percentages of *L. sativa* seeds after 96 hours on soils from *A. mearnsii* infestations and cleared slopes (Table 4.20 and Figure 4.2). Although the null hypothesis of no affect was accepted, Figure 4.2 indicates a difference for the Baviaans site and seeds germinated on the soils from cleared slopes in the Baviaans River had shorter and thicker roots, and conform to the decrease in germination on these slopes (Figure 4.3). The reduction in germination on cleared slopes in the Baviaans River was not statistically significant.

Table 4.20 Paired t-tests for the transformed germination percentages of *L. sativa* after 96 hours on soils from the different sites.

	Fynbos		Invaded/Cleared					
	Mean	Std. Err.	Mean	Std. Err.	N	t	df	p
Heuningnes River	70.17	3.95	68.17	4.82	30	0.28	29	0.781
Baviaans River	74.33	4.18	59.83	5.96	30	1.97	29	0.059
Tipi River	79.33	2.86	79.50	3.44	30	-0.25	29	0.808
Witteklip River	63.50	5.70	69.67	5.41	30	-1.54	29	0.135

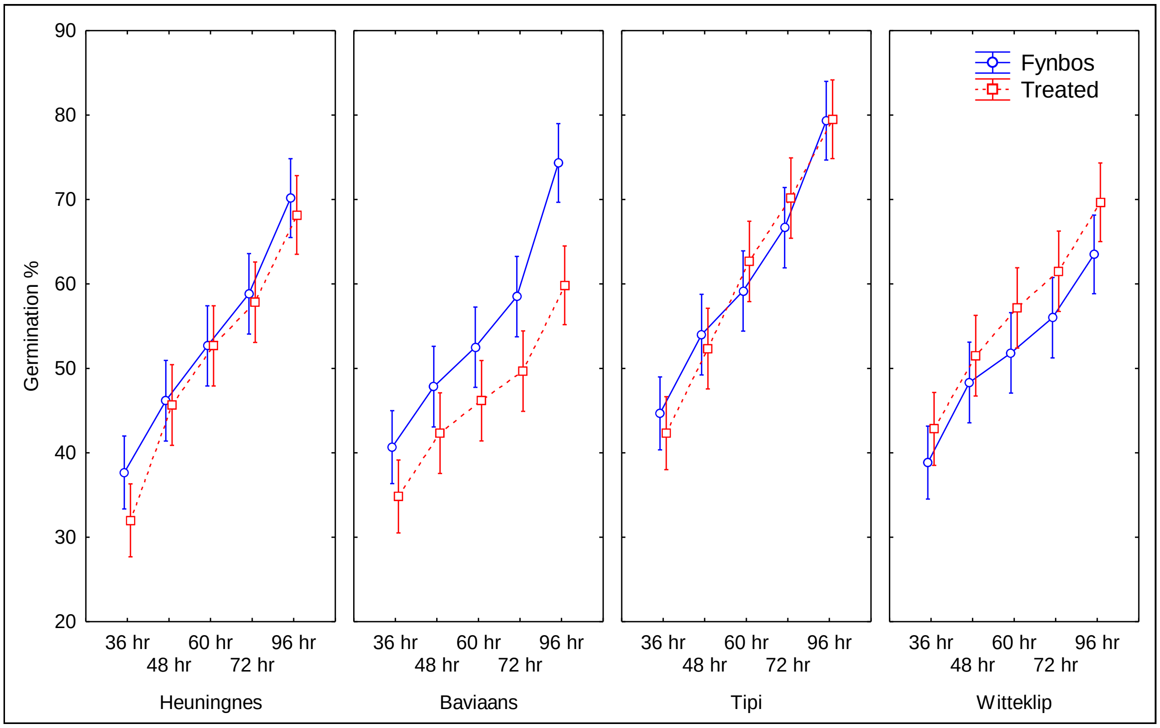

Figure 4.2 Non-transformed data for *L. sativa* seed germination for the time period 36 hours to 96 hours. The soils were taken from fynbos, invaded and cleared slopes in the Heuningnes, Baviaans, Tipi and Witteklip Rivers. Points indicate mean values and bars standard error.

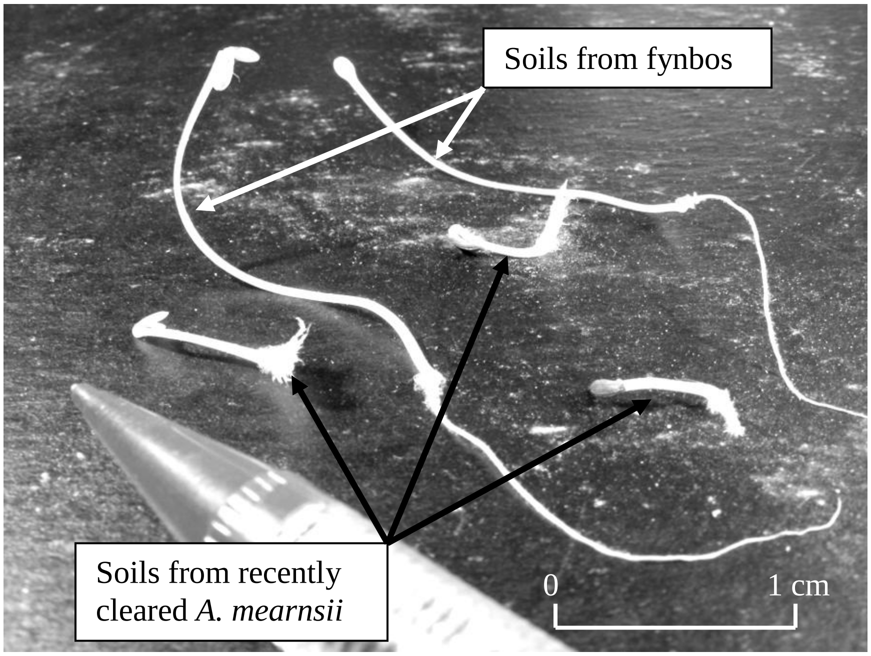

Figure 4.3 A macro photo of *L. sativa* seeds 96 hours after incubation on soils coming from a paired plot in the Baviaans River. Note the difference in root length.

4.5.1.2 Helichrysum umbraculigerum germination on soils from different sites

The null hypothesis that *H. umbraculigerum* germination is not affected by soils from *A. mearnsii* stands and areas that were cleared of *A. mearnsii*, was tested. The hypothesis was accepted for soils from invaded slopes and slopes cleared eleven years ago, and rejected for recently cleared slopes. There was a 76 percent increase ($p<0.01$) in germination for seeds incubated on soils from the recently cleared slopes in the Baviaans River (Table 4.21 and Figure 4.4). Visual differences in root length were observed while counting. Although more seeds germinated, roots were shorter for seeds germinated on soils from the recently cleared Baviaans River (Figure 4.5).

Table 4.21 Paired t-tests for the transformed germination percentages of *H. umbraculigerum* seeds after 14 days. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Invaded/Cleared					
	Mean	Std. Err.	Mean	Std. Err.	N	t	df	p
Heuningnes River	13.00	1.49	9.33	1.06	30	1.39	29	0.176
Baviaans River	**11.17**	**1.21**	**19.67**	**2.38**	**30**	**-2.81**	**29**	**0.009**
Tipi River	11.50	1.27	10.17	1.14	30	0.59	29	0.557
Witteklip River	11.50	1.29	13.17	1.32	30	-0.77	29	0.446

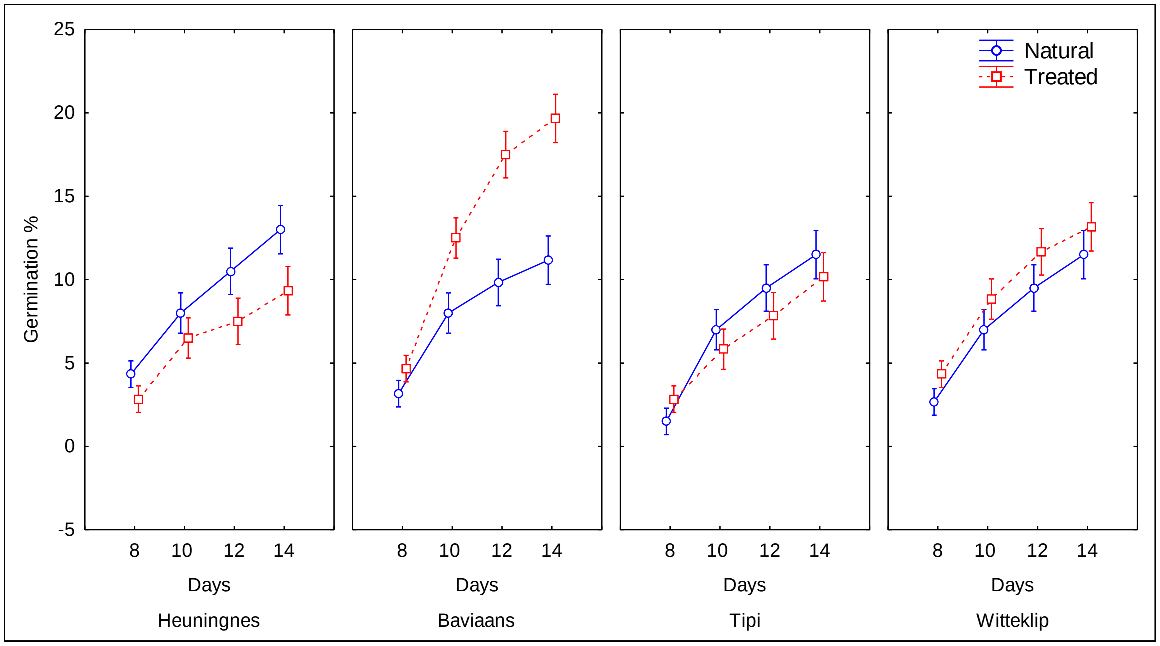

Figure 4.4 Non-transformed data for *H. umbraculigerum* germination 8 -14 days after incubation. The soils were taken from fynbos and A. mearnsii invaded or cleared slopes in the Heuningnes, Baviaans, Tipi and Witteklip Rivers. Points indicate mean values and bars standard error.

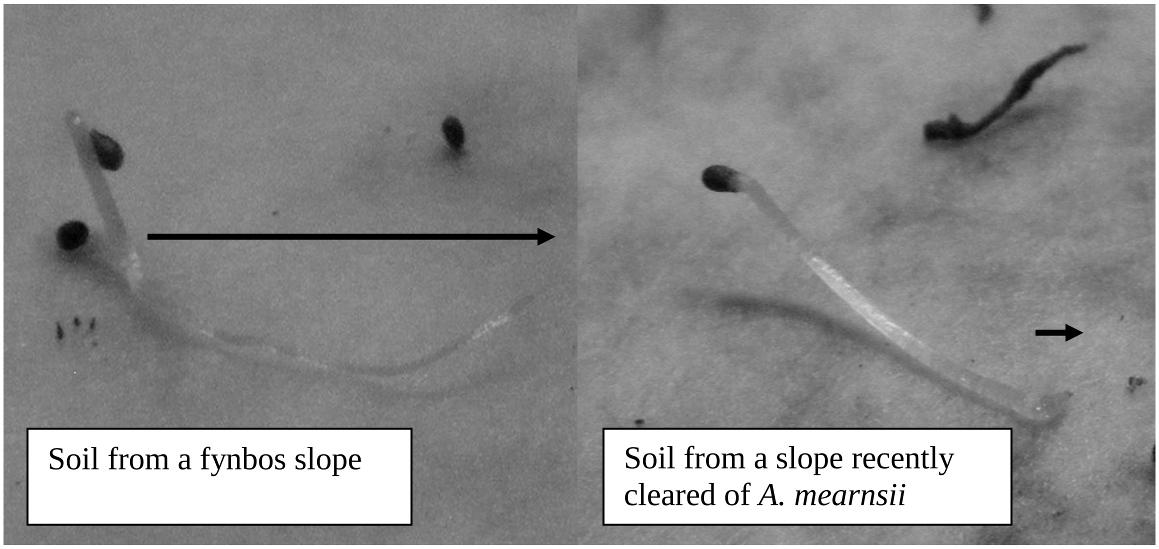

Figure 4.5 A macro photo of *H. umbraculigerum* seeds 14 days after incubation on soils from a paired plot in the Baviaans River. Note the difference in root length indicated by the arrows.

4.5.2 Relative growth rate of plants in soils from the different sites

4.5.2.1 Helichrysum umbraculigerum growth performance

The null hypothesis that relative leaf area growth rate for *H. umbraculigerum* is not affected by soils from *A. mearnsii* invaded slopes and cleared slopes was tested. The hypothesis was accepted for the Heuningnes, Baviaans and Tipi Rivers, and rejected for the Witteklip River. The increase in leaf area was 17 percent higher (p<0.005) for plants grown on cleared than on fynbos slopes in the Witteklip River (Table 4.22). A correlation analysis for relative leaf area growth rate, using 182 samples, found a positive correlation (p<0.01) of 0.209 for nitrogen concentration.

Table 4.22 Paired t-test for relative growth rate (cm2/day) for *H. umbraculigerum*. Significant changes (p<0.05) highlighted with bold print.

	Fynbos		Treated				
	Mean (n)	Std. Err.	Mean (n)	Std. Err.	t	df	p
Heuningnes River	0.136 (16)	0.006	0.126 (22)	0.009	-0.10	36	0.921
Baviaans River	0.121 (24)	0.007	0.126 (19)	0.006	-0.53	41	0.600
Tipi River	0.116 (28)	0.005	0.124 (24)	0.009	-0.91	50	0.369
Witteklip River	**0.125 (27)**	**0.005**	**0.146 (22)**	**0.004**	**-3.26**	**47**	**0.002**

4.5.2.2 Themeda triandra growth performance

The null hypothesis, that the relative leaf area growth rate of *T. triandra* is not affected by soils from *A. mearnsii* invaded and cleared slopes, was tested. The hypothesis was accepted for *A. mearnsii* invaded slopes and slopes cleared eleven years ago, but was rejected for recently cleared slopes. *T. triandra* leaf area growth rate increased ($p<0.01$) by 26 percent on soils cleared from *A. mearnsii* in the Baviaans River (Table 4.23) The accumulated effect of the increase in leaf area growth rate can be seen in Figure 4.6. This increase in leaf growth rate increased the plant dry weight by five fold over a 48 day growth period. A correlation analysis for relative leaf area growth rate, using 225 samples, found a positive correlation ($p<0.001$) of 0.358 and 0.331 for nitrogen and phosphorus concentration respectively.

Table 4.23 Paired t-tests for the relative leaf area growth rate (cm2/day) of *T. triandra* grown on soils from fynbos and *A. mearnsii* infested and cleared slopes. Significant changes ($p<0.05$) highlighted with bold print.

	Fynbos		Treated				
	Mean (n)	Std. Err.	Mean (n)	Std. Err.	t	df	p
Heuningnes River	0.109 (20)	0.004	0.097 (30)	0.005	0.36	48	0.717
Baviaans River	**0.086 (30)**	**0.006**	**0.109 (30)**	**0.004**	**-3.11**	**58**	**0.003**
Tipi River	0.082 (29)	0.004	0.087 (29)	0.004	-0.84	56	0.403
Witteklip River	0.112 (29)	0.003	0.121 (28)	0.004	-1.68	55	0.099

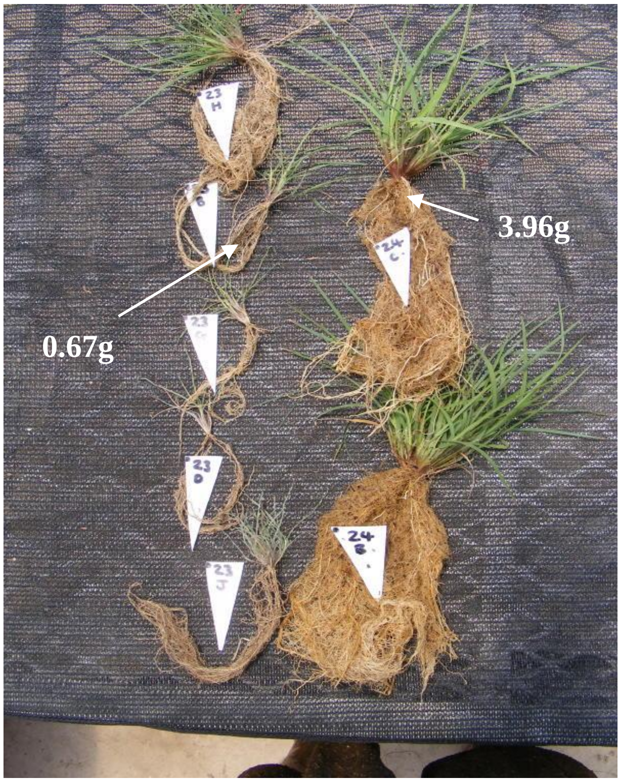

Figure 4.6 The effect of a 26 percent increase in relative leaf area growth rate (cm2/day) for *T. triandra* planted in soils from the Baviaans River. Plants on the right were planted in soils from slopes cleared of *A. mearnsii* and plants on the left on soils from neighbouring fynbos slopes. Note the difference in biomass accumulated over 48 days. The average weight (n = 10) is given.

4.6 Soil erosion

4.6.1 Daily rainfall records for Onverwacht Farm during 2008 and 2009

The highest rainfall event, of 78 mm in a day, occurred during time period two (T2) (Table 4.24 and Figure 4.7). This event was almost double the intensity of the peak event of time period four and almost four times greater than that of time period one and three (Table 4.24). These values can be compared to long term rainfall erosivity. The peak event for the past five years was 150 mm in a day and occurred in August 2006. This event was double that which was experienced during the soil erosion monitoring period.

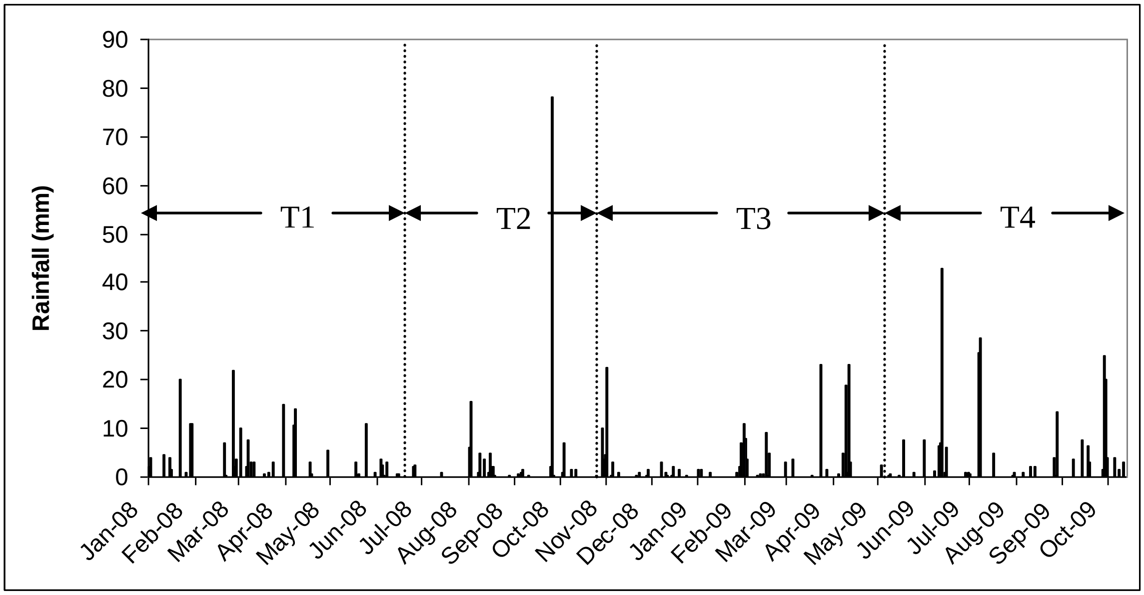

Figure 4.7 Daily rainfall records measured at Onverwacht Farm for the period of 14 January 2008 to 26 October 2009. Time period one to four are indicated on the graph.

Table 4.24 Total and peak rainfall for the four time periods over which erosion measurements took place.

Time period	Total (mm)	Peak (mm)
T1 (14 Jan. ‘08 – 18 Jun. ‘08)	194.1	22
T2 (18 Jun. ’08 – 27 Oct. ’08)	145.7	78
T3 (27 Oct. ’08 – 6 May ’09)	191.8	23
T4 (6 May ’09 – 26 Oct ’09)	249.2	43

4.6.2. Soil erosion rates for different vegetation conditions over the total monitoring period

Average soil erosion rates were similar on slopes in fynbos and slopes cleared of *A. mearnsii*, but rates were almost twice as high on *A. mearnsii* infested slopes (Figure 4.8). Although rates were twice as high for invaded slopes, there were no significant differences between the erosion rates of fynbos, invaded and cleared slopes. This might be due to the high variance that was experienced on invaded and cleared slopes. The high variability over time is explained in section 4.6.2. When considering all sites, slope angle and vegetation cover were significant ($p<0.05$) variables related to soil erosion (Table 4.25), with slope angle being the most significant, followed by vegetation cover, stone cover, treatment and aspect. The invaded sites had similar slope angles and vegetation cover percentages as fynbos sites. The only difference was the vegetation structure (see section 4.4.1).

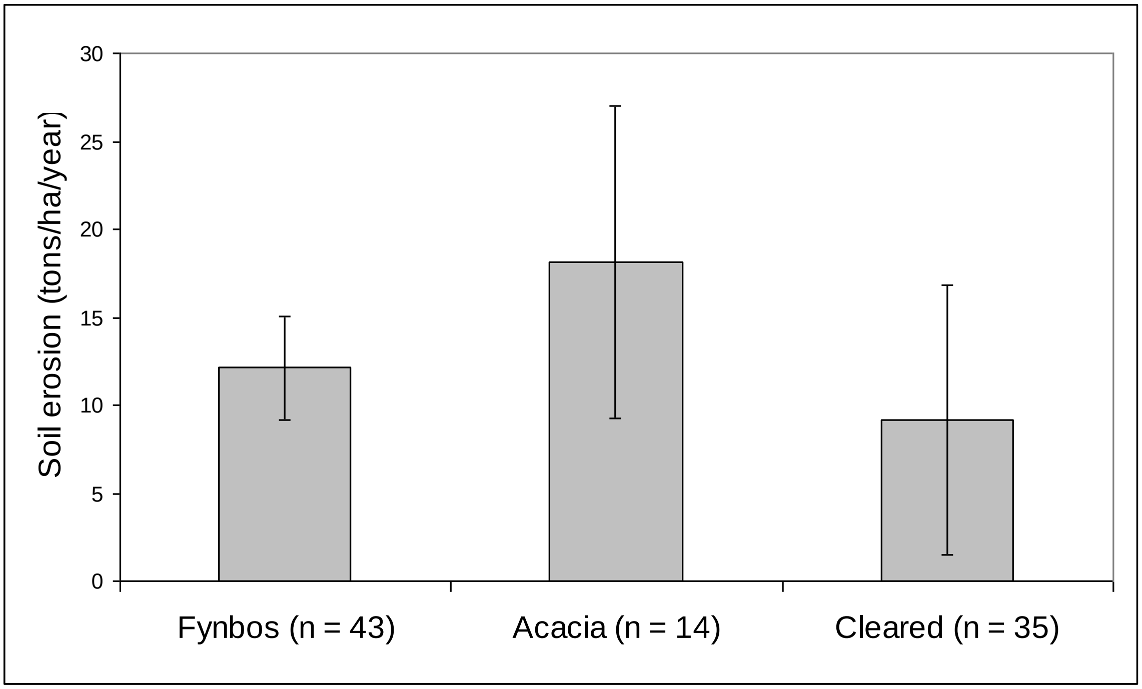

Figure 4.8 Erosion rates on slopes with fynbos, *A. mearnsii* trees and slopes cleared of A. mearnsii. Whiskers indicate standard error.

Table 4.25 The interaction between key variables and erosion rates during the full monitoring period. A stepwise regression was used including all sites. Significant changes (p<0.05) highlighted with bold print.

	B value	Std.Err.	t value	p-level
Slope	**-0.01**	**0.01**	**-2.30**	**0.024**
Veg cover (%)	**-0.01**	**0.00**	**-2.32**	**0.023**
Stone Cover (%)	0.00	0.00	-1.92	0.058
Treatment	-0.17	0.10	-1.67	0.099

4.6.3 Soil erosion rates for individual sites over the different time periods

The null hypothesis that altered vegetation cover does not increase soil erosion was tested (section 3.6). The null hypothesis was accepted for all vegetation cover types, except for slopes that were cleared for the longest time period during high intensity rainfall. The soil erosion rate in the Witteklip River increased ($p<0.05$) by 90 percent during time period two (Figure 4.9 – 4.11). The hypothesis was therefore accepted for invaded and recently cleared slopes, but can be rejected for slopes cleared eleven years ago. High rates were linked to high intensity rainfall events. Erosion took place during high intensity rainfall events, followed by deposition during low intensity events. What can also be seen in figure 4.10 and 4.11 is that erosion and deposition events are higher on cleared than on fynbos slopes.

The average erosion rates given in Section 4.6.2 are much lower than for the shorter time periods. This is as result of calculating the average erosion rate for the highly variable deposition and erosion events over the four time periods. Calculating an average is misleading as it hides the smaller soil movement events that occur over time.

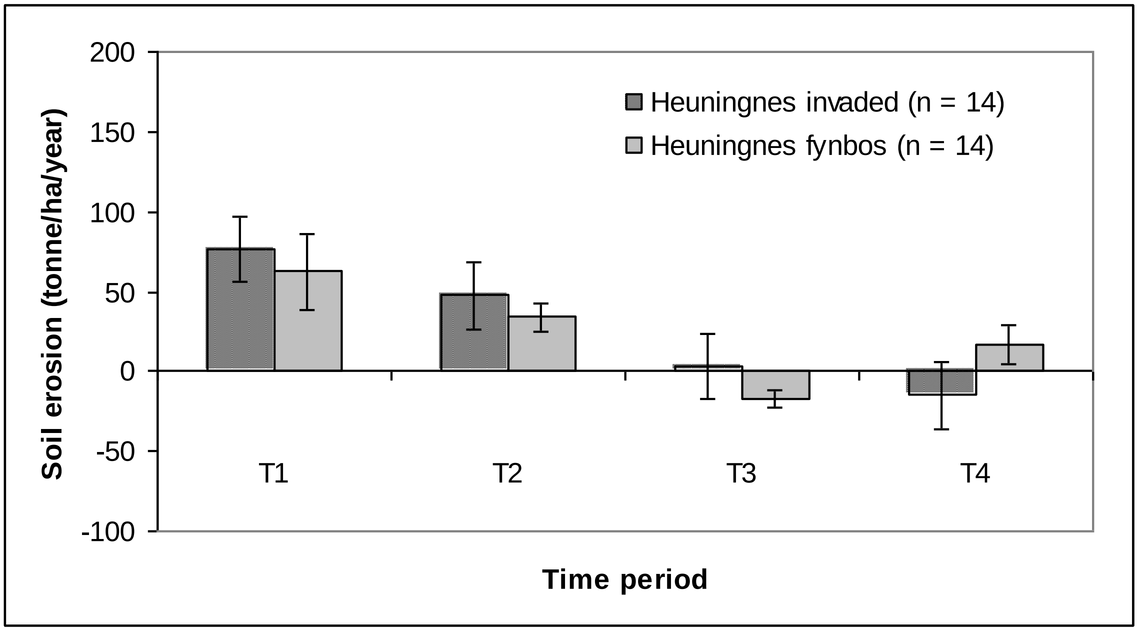

Figure 4.9 Erosion rates for fynbos and *A. mearnsii* infested slopes in the Heuningnes River. Time periods are given in Table 4.21 and whiskers indicate standard error.

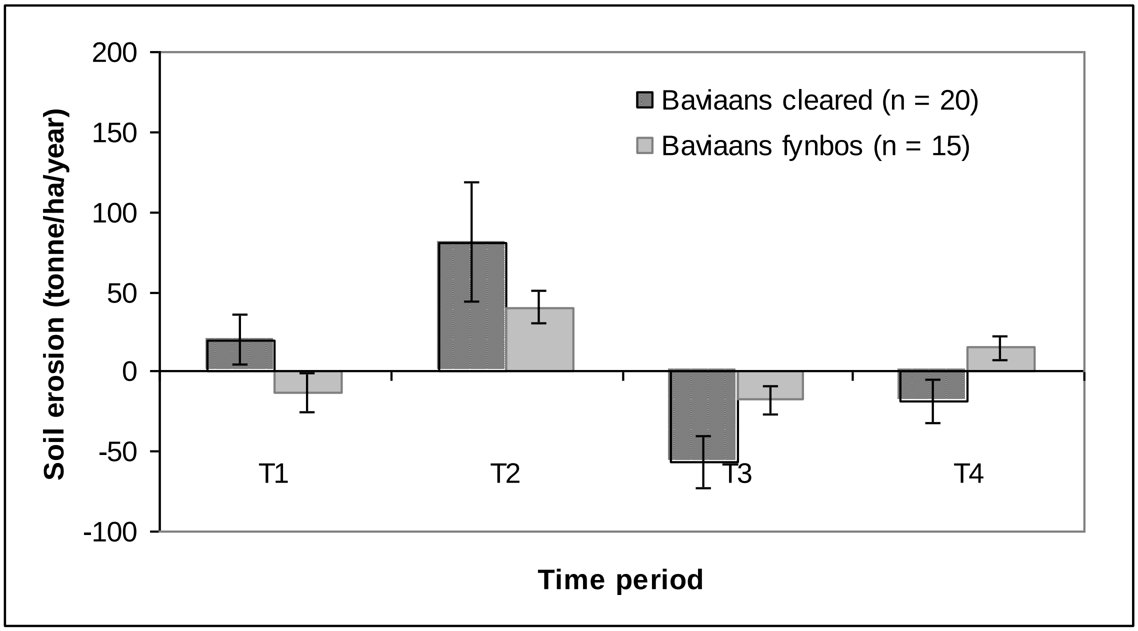

Figure 4.10 Erosion rates on slopes in the Baviaans River for fynbos slopes and slopes cleared of *A. mearnsii*. Time periods are given in Table 4.21 and whiskers indicate standard error.

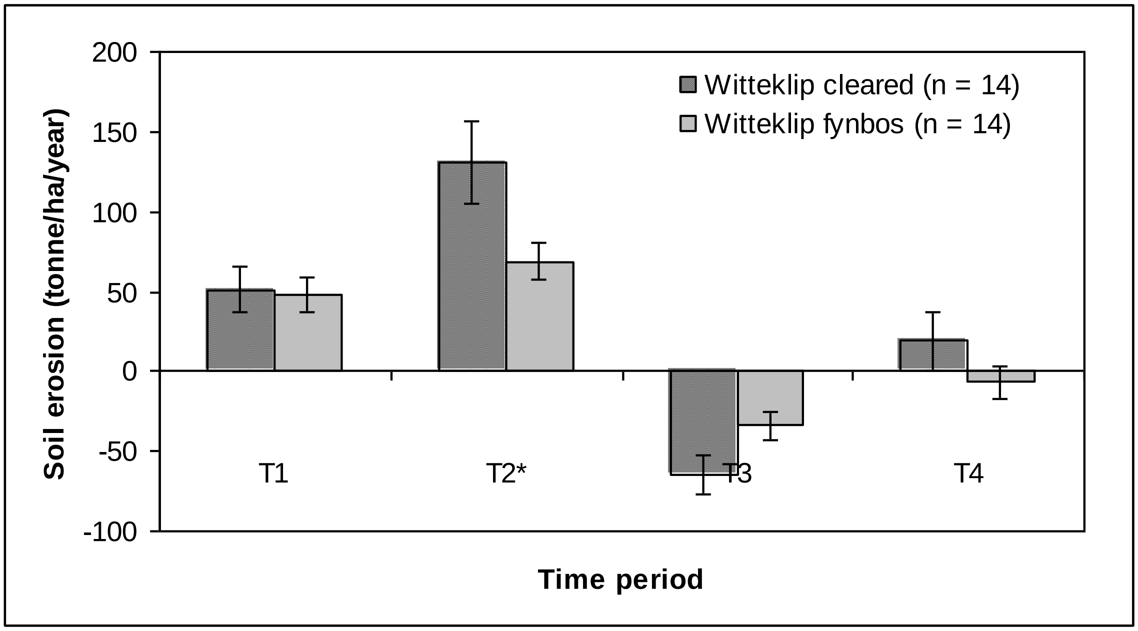

Figure 4.11 Soil erosion rates for fynbos slopes and slopes cleared of *A. mearnsii* in the Witteklip River (* indicates $p<0.05$). Time periods are given in Table 4.21 and whiskers indicate standard error.

Chapter 5: Discussion

5.1 Introduction

This chapter discusses and gives explanations for the findings presented in the previous results chapter. It attempts to explain what changes *A. mearnsii* invasion brings in terms of physical and chemical soil properties and vegetation structure. Furthermore, it assesses the influence of different clearing histories on these soil properties and vegetation structure, and how seed germination and seedling growth were affected. The change in erosion rate for invaded and cleared slopes is explained.

5.2 Soil characteristics for different sites

The physical character of the soils varied for the different sites. This was a result of the different parent materials (Figure 1.2) and the difference in average slope angle between the sites (Table 4.1 – 4.4). This difference between sites is also reflected in the chemical and physical variables for fynbos slopes in the different sites (Table 4.5 – 4.14). This highlights the high variability in this study area. The use of paired plots minimised this variability and enabled us to examine the changes that *A. mearnsii* brought in soil character.

5.3Physical soil properties

5.3.1 Change in physical soil properties on slopes invaded by Acacia mearnsii

A. mearnsii invasion changed the physical nature of the soils in terms of surface hardness, water repellency and plant litter cover. The proportion of litter cover was higher on invaded slopes, and is a result of the higher biomass of trees compared to shrubs. This increase of litter input resulted in an increased litter cover and is similar to the findings of Musil and Midgley (1990) and Yelenik *et al.* (2004).

Soil surface hardness was lower on slopes that were invaded by *A. mearnsii* trees. This could be attributed to the increase in organic matter, due to higher litter fall, on invaded slopes. Ehrenfeld (2003) found that the leaves of invasive species decompose at a higher rate than the leaves of native plants. The increased biomass and higher decomposition rates increased the organic content of the top layer of the soil.

Soil water repellency increased on *A. mearnsii* invaded slopes and was related to the increase in organic matter as a result of increased litter input during invasion. DeBano *et al*. (1967) and Scott (2000) found similar results, where increased organic content increased water repellency.

5.3.2 Change in physical soil properties due to clearing

The proportion of bare soil cover decreased with time after clearing on slopes cleared of *A. mearnsii*. This is due to the increase in fynbos cover as plants re-established (section 4.4.2). This contradicts what Beater *et al.* (2008) found in savannah after clearing. They found an increase in bare ground cover over time after clearing.

Soil surface hardness was lower on slopes that were cleared of *A. mearnsii.* This contradicts the findings of Mills and Fey (2004) and Maestre *et al.* (2002). They found that soils had a high tendency to crust where vegetation cover was low. The tendency to crust was related to a reduction in soil carbon. This indicates that increased organic matter prevents the bare soil from crusting. The difference in soil surface hardness decreased over time after clearing. This can be related to the decrease in organic matter on slopes that were cleared for the longest time.

Soil water repellency decreased over time after clearing. This relates to the decrease in soil organic matter in the form of leaf litter as found by DeBano *et al*. (1967) and Scott (2000).

5.4 Chemical soil properties

5.4.1 The effect of Acacia mearnsii invasion on chemical soil properties

The *A. mearnsii* invasion altered the chemical composition of the soil. The pH, nitrogen, potassium, manganese and sodium concentrations differed among invaded and fynbos slopes.

Soils invaded by *A. mearnsii* were more acidic. This finding supports that of Montgomery (2001) but contradicts those of Musil and Midgely (1990) on *Acacia saligna* invaded Sand Plain Fynbos.

Furthermore, soils invaded by *A. mearnsii* had higher concentrations of nitrogen, potassium and manganese as reported by other workers (Musil and Midgely, 1990; Yelenik *et al.*, 2004). The increase in sodium contradicts findings of Musil and Midgely (1990) and Yelenik *et al.* (2004), where the sodium concentration was not affected by invasion. Musil and Midgely (1990) relate this increase in nutrients to the high input of the nutrient rich leaves of *A. mearnsii* that decompose faster than fynbos leaves.

5.2.2.2 The effect of Acacia mearnsii clearing on soil chemical properties

The pH gap between fynbos and cleared slopes narrowed with time since clearing. Similarly, the difference in concentration of potassium and manganese decreased with time since clearing. Contradicting responses were found for nitrogen, phosphorus and carbon, where concentrations increased with time since clearing. The increase in zinc and calcium on *A. mearnsii* cleared slopes in the Witteklip River confirms the findings of Musil and Midgely (1990) on invaded slopes. The reason for this increase being found only for sites that were cleared for the longest time period is unknown.

The slopes that were cleared of low density invasion eleven years ago, had a different trend. The difference in pH was the largest for this site. This might have resulted in a leaching effect, with copper, zinc, cation exchange capacity, calcium and magnesium concentrations lower on slopes cleared of *A. mearnsii* than on fynbos slopes. These losses contradict what Musil and Midgely (1990) found and could be a localised effect. This different response for the Tipi River demonstrates the high variability and unpredictability of natural systems.

5.5 The change in vegetation structure

5.5.1 The effect of Acacia mearnsii invasion on vegetation structure

A. mearnsii invasion led to a tree dominated vegetation structure. This change in vegetation structure is what Lockwood *et al.* (2007) predicted would happen during invasion. Furthermore, the invasion of fynbos by *A. mearnsii* fitted Didham *et al.*'s (2005) interacting driver model, where a synergistic effect between habitat destruction and an aggressive invader caused a positive feedback loop that fuelled further invasion and habitat destruction.

The invasion significantly decreased the cover of all other fynbos growth forms such as shrubs, woody herbs, forbs, grasses, restios, sedges and geophytes. Beater *et al.* (2008) and Fourie (2008) have reported similar findings. This reduction in fynbos abundance is a result of the competitive ability of the invasive *A. mearnsii* trees (Lockwood *et al.,* 2007). *A. mearnsii* trees form a dense canopy that reduces light, and use large amounts of water. This places more stress on fynbos, a smaller growth form, and leads to a decrease in fynbos species and abundance (Didham *et al.*, 2005). Plant species richness decreased on all slopes invaded by *A. mearnsii* and is a result of the highly competitive advantage of the trees that led to a degraded native vegetation composition. This reduction in species richness is supported by the work of Richardson and Cowling (1992) and Galatowitsch and Richardson (2005).

From the measures taken, including vegetation structure and species diversity, it is possible to conclude that the system is degraded. The understanding of just how degraded the ecosystem is, remains limited, as discussed by Lamb and Gilmour (2003).

5.5.2 The effect of clearing Acacia mearnsii on vegetation structure

The removal of the dominant tree canopy exposed the remaining sparse fynbos cover on the slopes. Cleared plots were poorly vegetated (Table 4.16 – 4.19). The difference in shrub, woody herb, restio and sedge cover between fynbos and cleared slopes decreased for plots that were cleared for the longest time. The recovery of shrub and restio cover was very slow for sites that were cleared for the longest time period. The forb and grass representation recovered for sites that were cleared for the longest period as reported by other workers (Witkowski and Mitchell, 1989). This is a result of opportunistic plants with short life cycles.

The recovery of the native vegetation structure was very slow, with the exception of forbs and grasses. The interacting driver model of Didham *et al.* (2005) predicted that the removal of the invasive plant would lead to the dominance of certain native species and a reduction of other native species. This can be seen in the quick recovery of grass and forb cover, and slow recovery of shrub and restio cover.

The reason for the slow recovery of shrubs and restios might be related to seed availability and altered soil nutrients. Vosse *et al.* (2008) found that native species are poorly represented in the soil seed bank in non-native infestations. This is a result of reduced seed production by fynbos plants when soils increase in nutrient concentration (Musil and Midgely, 1990). Furthermore, soils became more fertile, resulting in seeds that are present to be less successful as they are adapted to poorer soil conditions. This reduction in native seeds and altered chemical conditions explain the poor recovery of the native vegetation cover.

The species richness gap between fynbos and cleared slopes narrowed very slowly for sites that were cleared for the longest time. Galatowitsch and Richardson (2004) found similar results in the Western Cape for native tree species, and relate it to the fact that native species do not respond well to disturbance. Native seeds will only germinate on stable substrates, compared to non-native species that are disturbance triggered. Results by Beater *et al.* (2008) in savannah contradict this positive rehabilitation trend. They found a decrease in native cover over time after clearing.

A. mearnsii stumps coppice successfully after clearing if not properly poisoned. Furthermore, a lack of vegetation cover triggers *A. mearnsii* seeds to germinate, resulting in a quick re-growth of *A. mearnsii* saplings. This is reflected in the high tree cover on cleared slopes (Table 4.14 – 4.16). Galatowitsch and Richardson (2004) and Beater *et al.* (2008) found a similar trend where clearing favoured non-native reinvasion. Working for Water teams treat this reinvasion by the use of a broadleaved herbicide, as a blanket spray (Moore, 2008). This is applied up to two times per year (Moore, 2008). This herbicide kills all *A. mearnsii* saplings, but also native broadleaved plants (Dow AgroSciences, 1989). This will lead to a reduction in the native broadleaved adults and seedlings and ultimately seed bank, reducing the chances of fynbos re-vegetation on cleared slopes. Grasses have the advantage of not being affected by broadleaved herbicides.

The slopes that were cleared of a low density *A. mearnsii* invasion in the Tipi River had a very slow recovery response compared to the Witteklip River. This might be related to the leaching effect of soil chemicals as postulated in section 5.2.2.2.

Although all the vegetation on cleared slopes recovered to some extent, the slopes remained in a degraded state. This indicates that the recovery process is slow in these ecosystems and that intervention is needed.

5.6 Seedling establishment on soils from different sites

5.6.1 Germination performance

5.6.1.1 Lactuca sativa germination on soils from Acacia mearnsii invaded and cleared slopes

The germination response of *L. sativa* was unaffected by the influence of *A. mearnsii* on soils. This contradicts what Souto *et al.* (2001) found for *L. sativa* seeds germinated on soil from under *Acacia melanoxylon* stands. They found a decrease in germination on soils that are invaded by *A. melanoxylon* compared to soils from native Oakwood stands. They relate the decrease to allelochemical leachates from the leaves of *A. melanoxylon.*

Seeds germinated on soils from the recently cleared slopes had a visible shortening and thickening effect on the roots (Figure 4.3). This supports what Bernhard-Reversat (1999) found for rice seedlings germinated in *Acacia auriculuformis* leachates. This could be due to the injuries caused by allelochemicals (Bernhard-Reversat, 1999) or the herbicide Garlon 4 (Nissen, 2009, pers. comm.) that was used as a blanket spray during follow up treatments.

5.6.1.2 Helichrysum umbraculigerum germination on soils from Acacia mearnsii invaded and cleared slopes

H. umbraculigerum germination was stimulated in soils from the recently cleared slopes. This increase in germination contradicts what Souto *et al.* (2001) found for *L. sativa* seeds germinated on soil from under *Acacia melanoxylon* stands. This indicates that *H. umbraculigerum* seeds are not affected by allelochemicals, but stimulated.

The roots showed a shortening effect (Figure 4.5), similar to those of *L. sativa* (section 4.6.1.1), in soils from recently cleared slopes. This supports the work of Bernhard-Reversat (1999).

5.6.2 Relative growth rate of plants in soils from the different sites

5.6.2.1. Helichrysum umbraculigerum growth performance on soils from Acacia mearnsii invaded and cleared slopes

H. umbraculigerum growth increased on soils from slopes cleared of *A. mearnsii* in the Witteklip River and is related to the high nitrogen concentration. Witkowski *et al.* (1990) found similar results where fynbos grew more vigorously after nutrient additions. Rundel (1983) states that this ability of herbaceous plants, to make use of nutrient fluxes, is related to their adaptation to fire regimes, where herbaceous plants tie up the available nutrients after a fire. This growth stimulation on soils from cleared slopes was reflected in the increase of herbaceous cover on slopes cleared for the longest time (section 5.5.2).

5.6.2.2 Themeda triandra growth performance in soils from Acacia mearnsii invaded and cleared slopes

T. triandra growth rate was higher in soils from recently cleared slopes. Yelenik *et al.* (2004) found similar results and stated that grass species grew vigorously on slopes cleared of *A. mearnsii*. This can be seen in the occurrence of the grass *Cynodon dactylon* on half of the number of plots cleared of *A. mearnsii*, in relation to a 10 percent occurrence in fynbos. This increased growth of grass on soils from cleared slopes was reflected in the recovery of grass cover on slopes that were cleared for the longest time (section 5.5.2).

The increased growth rate was positively correlated with nitrogen and phosphorus concentrations (Section 4.5.2.2). This supports the results of Yelenik *et al.* (2004).

5.7 Soil erosion rates for different vegetation conditions

The average erosion rates of fynbos and cleared slopes were similar, with rates double as high on invaded slopes. The increase in erosion rate on invaded slopes may be related to the increase in soil water repellency (Table 4.5). DeBano (1971) found that increased soil water repellency decreased infiltration and increase runoff. Work by Scott *et al.* (1998) proved that increased runoff would increase soil erosion. The higher erosion rate for invaded slopes is thus a result of increased runoff which increases soil erosion.

The erosion rate variability was a lot less on the fynbos slopes than on the *A. mearnsii* invaded and cleared slopes (Figure 4.8). López-Bermúdez *et al.* (1998) found similar responses, where slopes with undisturbed native vegetation were more stable than disturbed sites.

The low average erosion rate on the cleared slopes was a result of averaging erosion and deposition events for the full monitoring period (Figure 4.10 and 4.11). High erosion events were followed by high deposition events which reduced the average erosion rate.

The erosion and deposition events on cleared slopes were approximately double as high as on fynbos slopes (Figure 4.10 and 4.11). The increase in soil movement on the cleared slopes was a result of reduced vegetation cover (Table 4.22). This supported the work of Sala and Cavlo (1990) where vegetation cover was negatively related to soil movement.

The positive relationship between slope angle and erosion rate (Table 4.22) is similar to the results found by Scott and Van Wyk (1992).

The high variability in erosion rate on disturbed slopes is explained by the differences in rainfall intensity over the four measurement periods (Table 4.21). The first time period was characterised by low intensity rainfall, which resulted in low erosion rates (Figure 4.10 and 4.11). The increase in rain intensity in the next time period led to an increased erosion rate on cleared, but also fynbos slopes. The third time period is characterised by a long wet period with low intensity rainfall.

This resulted in deposition of sediment from upslope that were dislodged during the previous erosion period. The last measuring period experienced some erosion as rainfall events were of higher intensity. There is thus a strong relation ship between rainfall intensity and erosion rate. This relationship has been demonstrated by the work of Francis and Thornes (1990), Vacca *et al.* (2000), Chirino *et al.* (2006) and Xu *et al.* (2009).

The erosion–rainfall response in the invaded river differed from that of the rivers that were cleared. This might be as a result of differences in rainfall at this site. The invaded river is double the distance from the weather station. Rainfall events could be highly localised, with a different rainfall pattern for this time period.

The erosion rates measured for this study in fynbos were three orders of magnitude higher that those measured in fynbos by Scott (1993) and in Mediterranean vegetation measured by Kosmas *et al.* (1997). Erosion rates were hundred fold higher for invaded slopes than those measured by Scott *et al.* (1998) for afforested slopes. The measured erosion rates after clearing were tenfold higher than that recorded by Scott *et al.* (1998) for deforested slopes, and more similar to the rates recorded after a hot fires on deforested slopes. De Luis *et al.* (2001) found that small erosion plots do not give a true reflection of down slope erosion rates, but rather a reflection of soil movement. These rates are an indication of highly localised soil movement events, and cannot be extrapolated to slope or catchment scale. The difference in erosion rates for this study and the work of Scott and Kosmas *et al.* might be due to the size difference of erosion plots. We used one m2 erosion pin plots, where the other workers used large (16 - 176 m2) runoff plots.

The changes that take place due to *A. mearnsii* invasion and clearing are presented in a conceptual map in Figure 5.1. This map gives information on soil and vegetation changes that take place due to invasion, clearing and time since clearing. The map predicts that rehabilitation is needed in order to restore fully functional fynbos on the cleared slopes. No intervention may lead to further degradation and possibly re-invasion.

5.8 Study limitations and future research

Time and financial constraints limited the investigation of the effects of *A. mearnsii* on soils, vegetation and erosion. The number of plant species that were used in bioassays was limited as growth response measurements were very time consuming. Future studies should look at nutrient availability for plants, and water availability for seedling recruitment on slopes cleared of *A. mearnsii*. The effects of herbicide on existing fynbos should be quantified, and key fynbos species that are herbicide tolerant should be targeted for active rehabilitation of slopes cleared of *A. mearnsii*. Cost-benefit analysis should be done on the hand pulling of *A. mearnsii* saplings, and compared to the current follow up practices. Hand pulling will reduce the effect of herbicide damage of fynbos plants. The hypotheses that hand pulling will enhance rehabilitation of slopes cleared of *A. mearnsii* should be tested. Soil erosion should be carried out over a longer period, with more detailed rainfall intensity monitoring on-site. Large soil erosion plots should be established next to small plots to see if results are different.

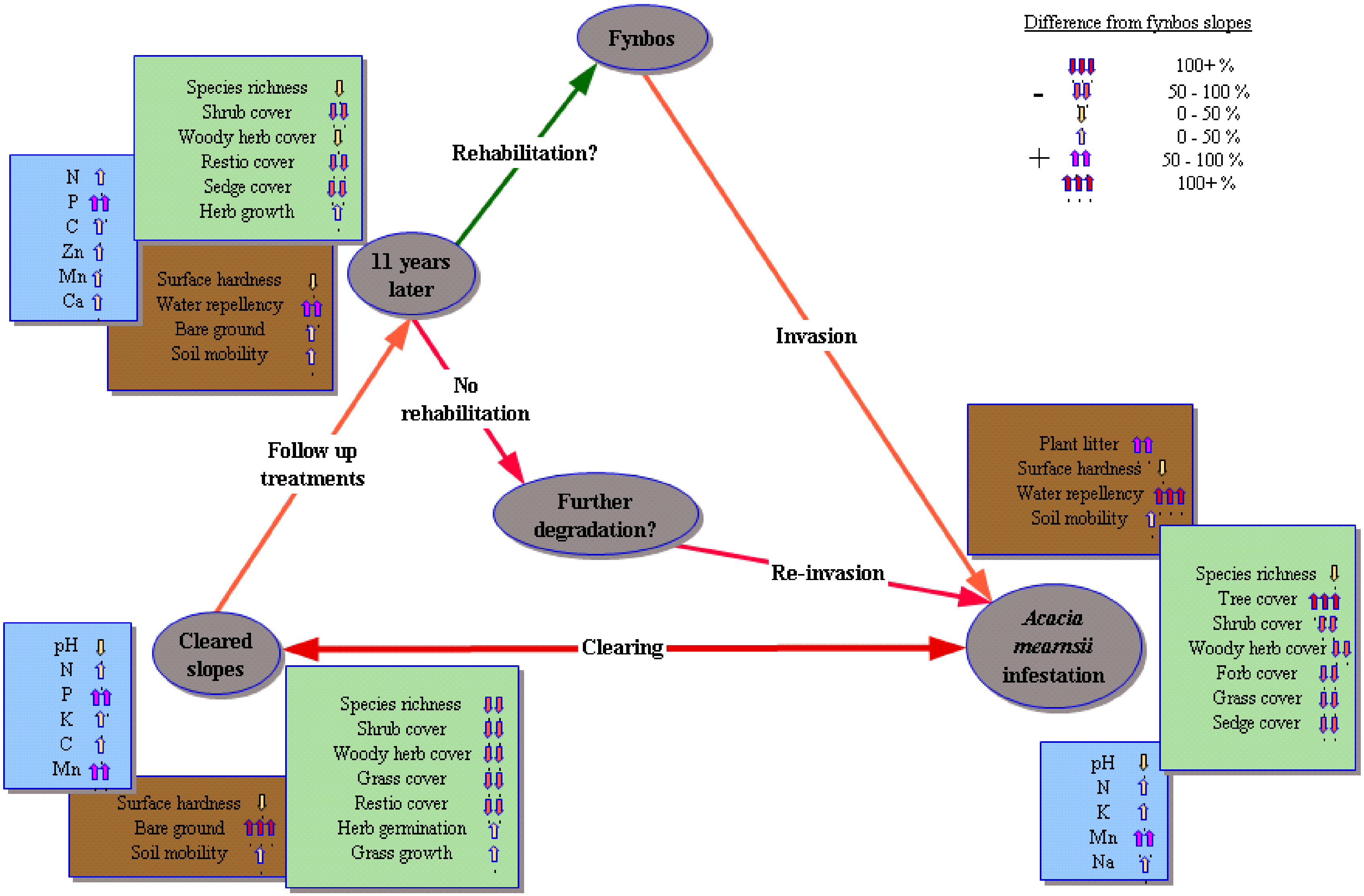

Figure 5.1 Conceptual map of changes in soil and vegetation characteristics that take place on invaded and cleared slopes in the Kouga Mountains

Chapter 6: Conclusions

This project aimed at unravelling the change that takes place in soil due to the invasion and clearing of *Acacia mearnsii* on slopes next to headwater streams in the Kouga Mountains. A number of physical and chemical soil variables were analysed on fynbos, invaded and cleared slopes. Vegetation cover and species richness were also compared, and the influence invasion and clearing has on soil erosion was analysed. This chapter gives a summary of the conclusions of the results and the probable influence clearing will have on restoration.

A. mearnsii invasion led to an increase of litter input into the soil, which decreases soil surface hardness and increases water repellency of the soil. The removal of the *A. mearnsii* trees decreased the litter input and resulted in a decrease in litter cover and an increase in bare soil surface over time after clearing. The difference in surface hardness and soil water repellency between fynbos and treated plots decreased with time since clearing.

The invasion of *A. mearnsii* increased the acidity and concentration of nitrogen, potassium, sodium and manganese in soil. Phosphorus and carbon concentrations were higher on cleared slopes than on fynbos slopes. The difference in concentration narrowed with time since clearing for most chemicals, except nitrogen and carbon, which increased.

The vegetation structure has been degraded on *A. mearnsii* invaded and cleared slopes. Didham *et al.*'s (2005) interacting driver model fits the response of invasion and clearing well. The model predicts that the removal of the invasive species will lead to an increase of certain native species and a reduction of other native species. The removal of *A. mearnsii* led to the improvement of herbaceous plant cover with time since clearing. Shrub and restio cover improved unsuccessfully over the eleven years after clearing.

Seed germination was relatively unaffected by *A. mearnsii* invasion and clearing. *Lactuca sativa* germination, a very sensitive test species, was unaffected by soil from invaded and cleared slopes, but *Helichrysum umbraculigerum*, a native herb's, germination was stimulated on recently cleared slopes. The roots of both species germinated on soils from recently cleared slopes had injuries, in the form of shortened roots.

H. umbraculigerum's growth rate increased on slopes that were cleared eleven years ago and *Themeda triandra's* growth rate on soils from the recently cleared slopes. The chemical and physical change in soil after clearing favour herbaceous seedling establishment, under conditions where there are sufficient plant water availability. The effect of increased herbaceous plant growth on soils from cleared slopes is reflected in the recovery of this growth form representation on cleared slopes. The effect that root injuries will have on the seedling's establishment is still unknown. Further research on the effect and lifespan of allelochemicals and herbicides should be performed on a wide variety of native species.

Soil erosion rates on invaded slopes doubled compared to that of fynbos slopes. This is due to an increase in soil water repellency and the low plant cover near the soil surface. Average soil erosion rates on slopes cleared of *A. mearnsii* was similar to that of undisturbed fynbos slopes. This is due to the high variation in erosion and deposition on the cleared slopes over time, and indicates that soils are far less stable on cleared slopes. Low intensity rain results in deposition on the lower slopes, with erosion during high intensity rainfall. The soil erosion monitoring should be continued so as to further investigate the soil movement in the Kouga Mountains.

This research highlighted the high variability in this Grassy Fynbos ecosystem. The sites did not respond in a linear fashion over time as assumed in the space for time analog. However, trends could still be drawn for certain variables.

A. mearnsii invasion and clearing changes physical and chemical soil properties, but the change stimulates herbaceous plant establishment. The slow natural restoration on the cleared slopes can be related to the reduction in the seed bank, reduced plant water availability and the effect of herbicides and increased soil nutrients. Future research should address the influence of these factors on a range of native species, from all growth forms, in order to find native plants that would be suitable for restoration work on these cleared slopes.

This research has contributed to the understanding of soil characteristics on the fynbos, invaded and cleared slopes of the Kouga Mountains. The results (Figure 5.1) form part of monitoring of the system and can be used as a management tool and platform from which further research may progress.

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