

Old field restoration: vegetation response to soil changes and restoration efforts in Western Cape lowlands

by

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Declaration

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Abstract

In the Mediterranean climate regions of the world, agricultural practices have caused considerable landscape transformation and lead to introduction of alien species that now dominate secondary succession on abandoned agricultural fields. Various restoration attempts have been made to reduce alien plant species cover, and to enhance the re-establishment and cover of native plant species. However, results and successes were mostly short-term due to re-growth and persistence of the weedy alien species, which has been suggested to be caused by land use history, especially the nutrient enrichment of soil, and particularly phosphorus and nitrogen.

This study investigated different soil properties (pH, electrical conductivity (EC), soil moisture, as well as available phosphorus (P) and total nitrogen (N)) on 10 and 20 year old abandoned fields, as a function of depth in three habitats (ridge (old cultivated area), ditch (old drainage line) and slope (intermediate zone between ridge and ditch)) on the old fields. The relationship between these soil properties and the vegetation occurring on the two old fields was established. At the same time, restoration treatments (autumn burn, combination of autumn burn and herbicide, herbicide application alone, as well as spring burn) were conducted to reduce the cover and abundance of non-native plant species and *Cynodon dactylon*, and to enhance cover of native species.

Results from the study show that levels of all investigated soil properties were higher on the younger field. The highest difference was observed in EC and pH. Seasonal differences in both soil properties could also be observed. A principal component analysis indicated that the dynamic of all soil properties shaped the vegetation type on old fields, with the main soil properties being dependent on land-use history and time since abandonment. This study suggests that EC and pH could be part of parameters that drive the persistence of undesirable species persistence on old fields and inhibit native plant species instead.

Best treatments to reduce undesirable species number and cover were a combination of autumn burning and herbicide application, as well as herbicide application alone. These two treatments, however, as well as an autumn burn, negatively affected species richness and cover of native species. The autumn burn did not affect alien plant species occurring, whereas a spring burn reduced undesirable grasses, but it could enhance the cover of *C. dactylon*, which is the dominant undesirable plant, and the cover of few native plant species while inhibiting other indigenous plants. Therefore, a spring burn should be used in conjunction with other treatments to reduce alien plant. In addition, seeds of native plant species should be

broadcast on the fields. This study indicated that the effectiveness of these treatments did not exceed one year.

We suggest that old agricultural field restoration should begin with a preliminary study to determine the soil properties that are most changed by land-use in comparison with natural soil, as well as the alien plant species present. Based on this, appropriate restoration treatments must be developed, and monitoring of the site conducted on a regular basis to ensure the success of the treatments. Successful restoration attempts should include both above- and below-ground treatments, while at the same time, considering plant-soil feedback effects to achieve the re-establishment of native plant species.

Opsomming

Mediterreense streke wêreldwyd het dramatiese veranderinge en versteuring ondergaan vanweë landboubedrywighe. Die versteuringe het daar daartoe gelei dat sekondêre suksessie op verlate landbouggrond deur uitheemse indringer plante oorheers word. Verskeie maatreëls en tegnieke was al mee geëksperimenteer om die suksessie van uitheemse plante te inhibeer en inheemse en plaaslike flora spesies se vestiging te bevorder. Vanweë die geskiedenis van die landbouggrond, spesifieke te make met die gebruik van kunsmatige voedingstof veryking (fosfor en stikstof toevoeging), en is die sukses van inheemse spesies egter van korte duur, met uitheemse spesies wat beter by die kondisies aanpas.

Hierdie studie het die eienskappe van verlate landbouggrond ondersoek, met spesiale aandag op grond pH, konduktiwiteit, grond voginhoud, bekombare fosfor en totale stikstof inhoud. Die studie is in twee segmente verdeel, elk met drie onderverdelings. Eerstens is daar op landbouggrond wat 10 en 20 jaar verlate is gefokus. Dié is verder verdeel in drie 'habitate': i) 'n rif (verlate landbou grond), ii) 'n drif (destyds as dreineringskanaal gebruik), en iii) 'n helling (die oorgang sone tussen die twee laasgenoemdes). Die verwantskappe tussen die drie sones was bepaal. Tesame met die voorafgaande verdelings in dié eksperiment is verskeie restourerings praktyke tot die proefgestel. Om die bedekking van, en aantal uitheemse plant spesies, asook *Cynodon dactylon*, te beperk en die hervestiging van inheemse spesies te bevorder, was daar met 'n herfsbrand, 'n kombinasie van herfsbrand en onkruiddoders, onkruiddoders en 'n lenteband praktyk geëksperimenteer.

Resultate het op 'n duidelike onderskeid tussen die 10 jaarou en 20 jaarou grondgetoon. Al die genoemde grondeienskappe was hoër in die 10 jaarou grond, met die grootste verskil in die konduktiwiteit en grond pH. Dié twee eienskappe het ook beduidende seisoenale verskille getoon. Die studie toon verder daarop dat pH en konduktiwiteit kardinaal is in die vestiging en voorbestaan van uitheemse en indringer spesies op verlate landbouggrond.

Van die onderskeie restourerings praktyke waarmee geëksperimenteer was het twee praktyke die beste gevaar, naamlik 'n kombinasie van herfsbrand en onkruiddoders en die toevoeging van onkruiddoders alleenlik. Die nadeel van die twee praktyke, asook die gebruik van herfsbrande, is dat dit die spesiesrykheid en bedekking van inheemse spesies óók negatief affekteer. Die gebruik van 'n lenteband het *C. dactylon* bedekking bevorder, 'n ongunstige resultaat. Verder is die vestiging van inheemse spesies ook deur dié praktyk benadeel. Lenteband gekombineer met ander behandelings moet gebruik word om uitheemse plant spesies te beheer. Die saad van inheemse spesies moet gesaai word oor die lande. Die effektiwiteit van die behandelings duur net vir een jaar.

Restourering van verlate landbougrond is nie 'n eenvoudige proses nie. Die resultate van dié studie toon daarop dat die grondeienskappe van verlate landbougrond eerstens bepaal moet word en met die eienskappe van natuurlikgrond vergelyk moet word. Verder sal 'n opname van gevestigde uitheemse indringer spesies op verlate landbougrond ook voordelig wees voor restourering maatreëls aangewend word. Die uitkomst van die bogenoemde bepalinge moet dus die mees aanvaarbare restourerings praktyk rig. Terselfdertyd is dit belangrik om restourerings vordering op 'n gereelde basis te monitor om sukses te behaal. Laastens toon dié studie daarop dat effektiewe restourering op bo- en ondergrondse behandelings moet fokus en nie net op een van dié nie en dat die saadbank van inheemse plantspesies 'n groot invloed op die sukses van restourasie praktyke kan hê.

This thesis is dedicated to my mother Véronique Meloughe Memiaghe

And the memory of my best friend and father

Jean Prosper Ndong Nze

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In Fang, we said that one finger cannot wash the face, but it is with the hand that the face can be washed through the participation of all fingers. Thereby I would like to thank all people who contributed to the completion of this study.

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Chapter 1: General introduction

1.1. Current status and threat of Cape Lowlands

Due to its high number of endemic flora and fauna, the Cape Floristic Region has been recognised to be one of 34 hotspots in the world (Myers *et al.*, 2000; Mittermeier *et al.*, 2004). The lowland vegetation of the CFR is composed of strandveld, renosterveld and fynbos (Boucher, 1983; Moll & Bossi, 1984; Rebelo *et al.*, 2006). In the lowland areas of the Western Cape, the fynbos biome is mostly dominated by fynbos and renosterveld. West Coast renosterveld has recently been reclassified into alluvium fynbos and shale renosterveld, which are characterized by four and nineteen vegetation types respectively (Rebelo *et al.*, 2006). The lowlands, where these vegetation types occur in a mosaic, have been almost transformed by human activities. Examples for this transformation are the Swartland Alluvium Fynbos and Swartland Shale Renosterveld (Figure 1.1).

Swartland Alluvium Fynbos is mainly characterised by tall shrubs (e.g. *Diospyros glabra*, *Olea europaea*), low shrubs (e.g. *Cliffortia ferruginea*, *Elytropappus rhinocerotis*); Graminoids (e.g. *Calopsis paniculata*, *Cynodon dactylon*), and is a unique habitat to some endemic species, mainly geophytes such as *Geissorhiza furva* and *Moraea villosa*. Swartland Shale Renosterveld is mostly characterised by low evergreen shrubs (e.g. *Anthospermum aethiopicum*, *Elytropappus rhinocerotis*) as well as geophytes (e.g. *Cyanella hyacinthoides*, *Melasphaerula ramosa*), and is a habitat for variety of endemic species such as *Leucadendron verticillatum* and *Babiana angustifolia*. The main differences between both vegetation types are mostly due to the soil texture and rainfall regime. Swartland Alluvium Fynbos is present on alluvium gravel and cobble, while Swartland Shale Renosterveld occurs on clay soil. Both occur in the region with precipitation of 270-980 mm, but Swartland Alluvium Fynbos receives more water than Swartland Shale Renosterveld (Rebelo *et al.*, 2006).

West Coast Lowland Vegetation Types

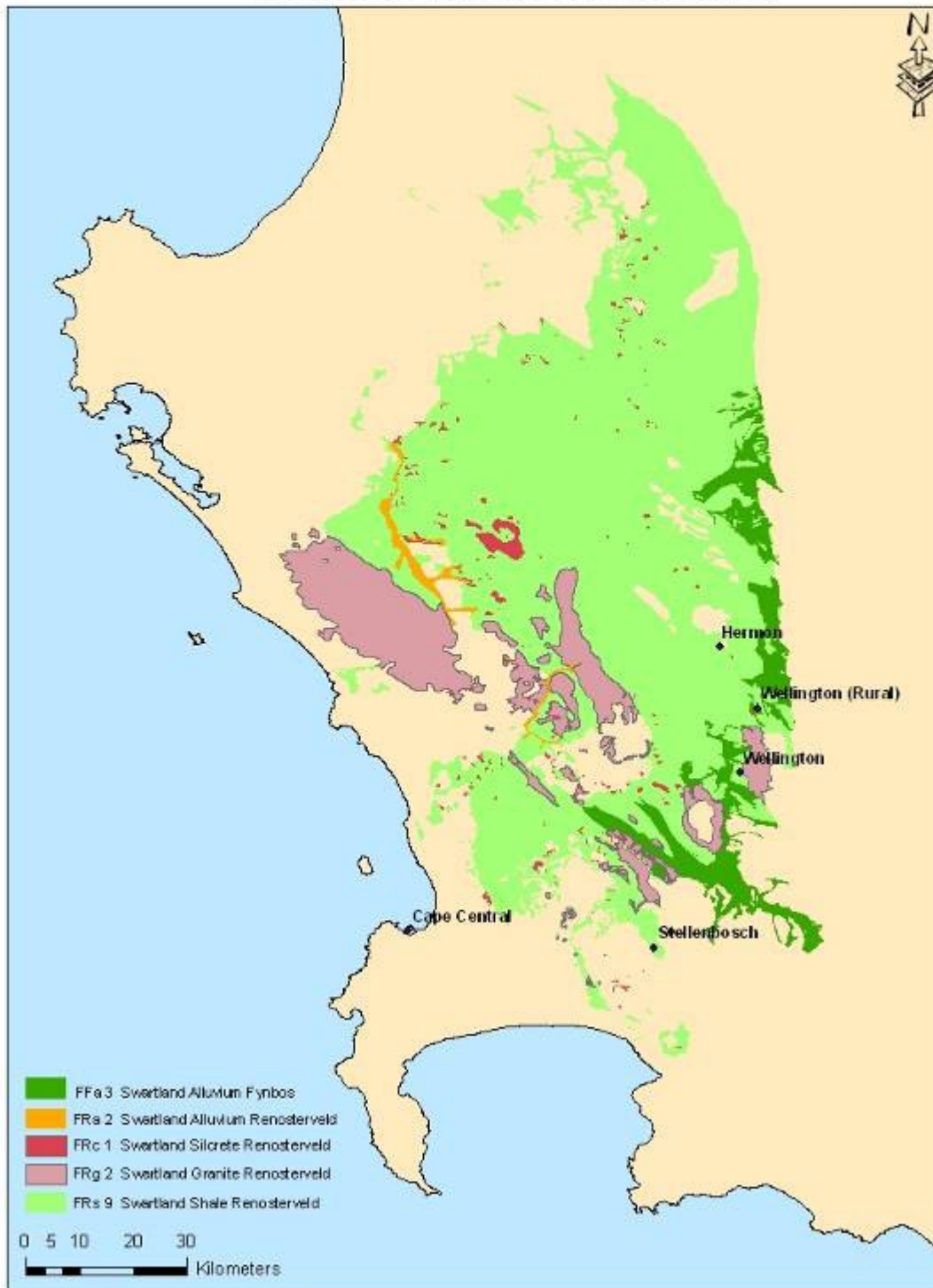


Figure 1.1 Some of the lowland West Coast vegetation types as classified by Mucina & Rutherford (2006). Map constructed using shape files for vegetation type data by Mucina and Rutherford (2006).

The first inhabitants of the CFR are assumed to have come from tropical areas of the African continent approximately half a million years ago (Deacon, 1992). Over 125 000 years ago,

they began to use stone tools and to burn natural vegetation to enhance the growth of geophytes, which were one of their main foods (Klein, 1977; Parkington, 1977; Deacon, 1992). The Stone Age people hunted and gathered in the region for almost 21 000 years (Klein, 1977; Deacon, 1992), but were progressively replaced by Holocene Bushmen, who also used fire as farming practice (Boucher, 1983; Deacon, 1992). Two subgroups of Bushmen, speaking Hottentot and Khoi-khoi, introduced cattle and sheep farming 2 000 years ago (Schweitzer & Scott, 1973; Parkington, 1977; Boucher, 1983; Deacon, 1992; Low & Rebelo, 1996; Hoffman, 1997), influencing the natural ecosystems until about 300 years before present (1700 AD) (Deacon, 1992). However, their impact on the lowland vegetation mosaic was likely negligible, as they were a small, nomadic group of people (Parkington, 1977; Boucher, 1983; Krug *et al.*, 2004a).

Khoi-khoi, with their cattle and sheep farming, attracted the European people travelling to India to stop at the Cape of Good Hope, and to barter livestock and other goods (e.g. water) (Deacon, 1992; Hoffman, 1997). To organise that trade, a Dutch company (Dutch East India Company) established a station in 1652 (Boucher, 1983; Deacon, 1992), causing the increase of cattle and sheep farming in the areas surrounding the station, as European pastoralists and Khoi-khoi needed to be near the trade areas (Boucher, 1983; Deacon, 1992). As the Khoi-khoi left their nomadic life style, they began to frequently burn the lowland vegetation to enhance the abundance of grasses in the areas occupied (Klein, 1977; Boucher, 1981; Low & Rebelo, 1996). In addition, the bartering of goods between Khoi-khoi and Europeans also caused the introduction of non-native plant species, such as shrubs and trees, which were used as ornaments before becoming invasive (Deacon, 1992). These developments (livestock farming and introduction of non-native plants) started the human pressure on the CFR vegetation. Later, around 1700, Europeans decided to settle in the station, provoking its conversion to a colony. After settling, the Europeans integrated Khoi-khoi knowledge into their methods of farming for crop cultivation in the lowland areas, causing further introduction of alien plants species (Deacon, 1992; Hoffman, 1997). This increase of people caused a rise in exploitation of resources provided by CFR ecosystems, due to urbanisation and agricultural activities (Boucher, 1981; Boucher, 1983; Deacon, 1992; Hoffman, 1997).

To deal with the land issue, which was driven by the need for land, cost and resource availability, the Dutch East India Company started to lease the land to be cultivated (Deacon, 1992). This caused further exploitation of the lowland areas. By 1760, the West Coast lowlands were nearly exclusively covered by farms (Deacon, 1992; Hoffman, 1997), with two main activities being conducted on the lowland vegetation mosaic: crop cultivation (e.g.

wheat and grapes), and livestock farming (cattle and sheep) (Boucher, 1981; Deacon, 1992; Hoffman, 1997). The introduction of other crops, such as deciduous fruit farming, caused further expansion of agricultural land, and with the mechanization of agricultural practices, even more areas were transformed (Moll & Bossi, 1984; Deacon, 1992; Hoffman, 1997).

The transformation of most West Coast lowland vegetation types, particularly the Swartland Alluvium Fynbos and Swartland Shale Renosterveld, is the legacy of a land-use history driven mostly by agricultural practices (Parker, 1982; Rebelo, 1992; Heydenrych & Littlewort, 1995; Low & Rebelo, 1996). This has reduced the existence of both vegetation types to fewer than 35% and 10% of their original areas respectively (Rebelo *et al.*, 2006). The remaining extent of Swartland Alluvium Fynbos and Swartland Shale Renosterveld are nearly 18 000 isolated patches (von Hase *et al.*, 2003). Most of those fragments are small patches (von Hase *et al.*, 2003; Rouget *et al.*, 2004), but eight of these have areas of more than 1 000 ha (von Hase *et al.*, 2003), the largest being approximately 7 400 ha in a West Coast lowland vegetation types mosaic (Krug *et al.*, 2004b).

The fragmentation of the remaining natural vegetation has led to severe alteration of ecosystem processes, especially in the smaller fragments. Kemper, Cowling & Richardson (1999) illustrated that the plant species composition, diversity and richness varied more in small patches than in large remnants, and small patches could not contain certain plant species. A difference in fragment size however, did not affect the abundance of some flies and butterflies, but did influence bees and monkey beetles (Donaldson *et al.*, 2002). Fragment size combined with distance from a larger fragment affected seed and fruit set in some plant species (Donaldson *et al.*, 2002), and distance of fragment from natural vegetation affected the vegetation recovery due to limited seed and propagule dispersion (Walton, 2006; Cramer, Standish & Hobbs, 2007; Pueyo & Alados, 2007). Fragmentation of natural vegetation thus can affect the ecosystem processes both within and between fragments.

The remaining natural vegetation of the Swartland Alluvium Fynbos and Swartland Shale Renosterveld is deemed crucial for conservation due to its high species richness of endemic plant species such as geophytes (Walton, 2006). Based on the high proportion of endemic flora found in the remaining fragments and the percentage of original extent remaining, these two vegetation types have been classified as irreplaceable (Ferrier, Pressey & Barrett, 2000; Rouget *et al.*, 2004). Both factors - high species richness and high proportion of endemic flora - have also been used to determine the specific conservation target of each biome, based on the Species-Area Relationship (SAR) (Desmet & Cowling, 2004), which differs from the 10% target fixed by the International Union for the Conservation of Nature (IUCN). Therefore, the

conservation biodiversity targets are 30% for Swartland Alluvium Fynbos and 26% for Swartland Shale Renosterveld (Rouget *et al.*, 2004; Rouget *et al.*, 2006) however, only 25% of Swartland Alluvium Fynbos and 9% of Swartland Shale Renosterveld remain of the original extent. Furthermore, these critical endangered vegetation types (Figure 1.2) are further threatened by agricultural practices, invasion of alien plant species and urbanization (Walton, 2006). Currently, only 1.7% of remaining Swartland Alluvium Fynbos and 0.5% of remaining Swartland Shale Renosterveld vegetation are protected in nature reserves and parks (Rouget *et al.*, 2004; Rouget *et al.*, 2006). Thus, to achieve the conservation biodiversity targets, more natural vegetation fragments need to be included in the reserves and protected areas (Rouget *et al.*, 2004).

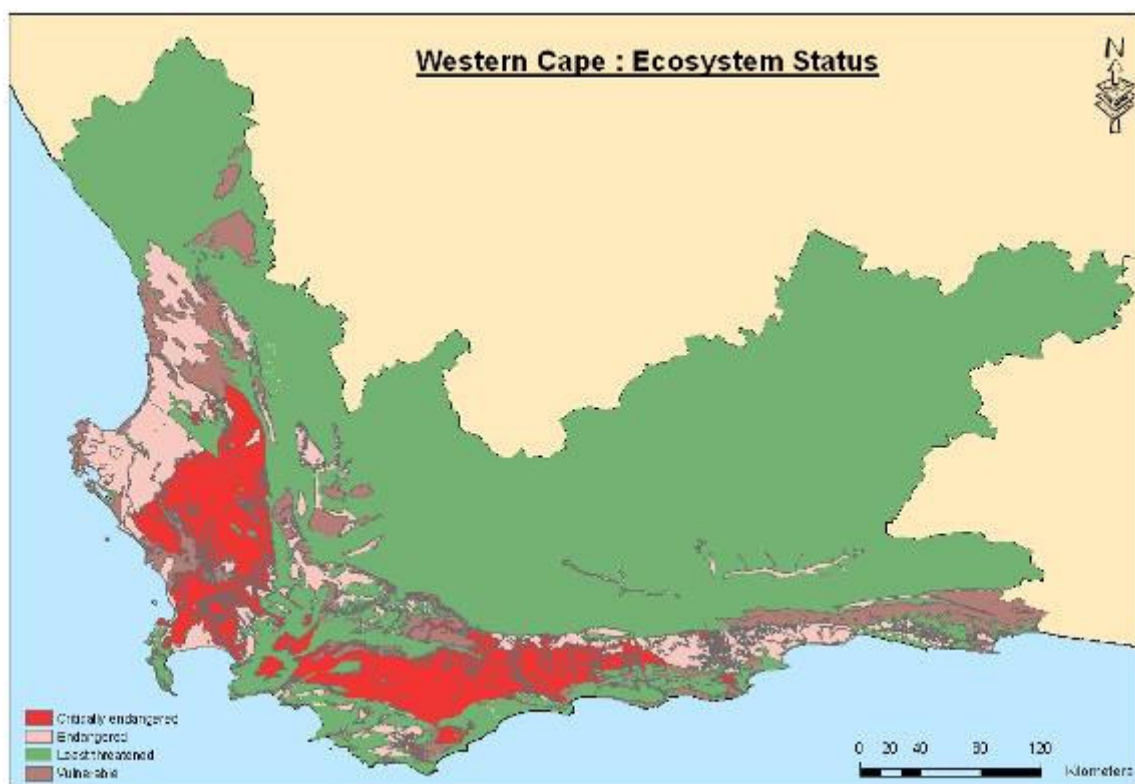


Figure 1.2 Western Cape Ecosystem Status (Rouget *et al.*, 2004).

1.2. Restoration as an important part of conservation

The conservation of the remaining Swartland Alluvium Fynbos and Swartland Shale Renosterveld fragments is compromised by their close proximity to abandoned agricultural fields (McDowell & Moll, 1992; Low & Rebelo, 1996; Krug *et al.*, 2004b). These old fields are the source of alien plant species that are invading the natural vegetation (Kemper *et al.*, 1999; von Hase *et al.*, 2003; Krug *et al.*, 2004b; Rouget *et al.*, 2004; Rebelo *et al.*, 2006). Their introduction is caused by wind and indigenous large herbivores that roam between old fields and natural vegetation (Kemper *et al.*, 1999; Shiponeni, 2003; van Rooyen, 2004;

Shiponeni & Milton, 2006; Mubamu Makady, (in prep.)). This presence of alien plant species could affect the conservation of biodiversity targets. Therefore, restoration of old agricultural fields may reduce their introduction into natural vegetation. In addition, restoration can enhance ecological processes by creating corridors to reconnect natural patches, since such ecological processes are known to be affected by habitat fragmentation (Kemper *et al.*, 1999; Donaldson *et al.*, 2002; Walton, 2006; Cramer, 2007; Cramer *et al.*, 2007; Pueyo & Alados, 2007). The restoration of old agricultural fields could also help to reduce natural vegetation decline in small, isolated patches (Gonzalez, 2000). Furthermore, restoration is important in the understanding of ecosystem dynamics, as it assists in the development of pertinent concepts and theories such as secondary succession, which is affected by land-use history on old agricultural fields (Hobbs & Cramer, 2007). Old field restoration also contributes to the understanding of effects of alien plant species on the re-establishment of natural vegetation, and on changes in soil properties (Hobbs & Walker, 2007).

The restoration study conducted here is part of a range of restoration projects aimed at restoring natural vegetation on old agricultural fields at Elandsberg Private Nature Reserve (EPNR). In particular, the study aims to contribute to an understanding of plant-soil feedback that will help to develop an appropriate restoration method for old cultivated fields in alluvium fynbos / shale renosterveld. In addition, restoration of old fields at EPNR increases habitat available to the endemic geometric tortoise (*Psammobate geometricus*), which is unique to the vegetation types found in that area, and which has the largest remaining population at EPNR (Archer, 1960; Archer, 1967; Greig, 1984; Baard, 1989; 1990; 1993; 1995; Balsamo *et al.*, 2004). The geometric tortoise could therefore be used as a flagship species for restoration and habitat conservation in the region, due to their historical and current tourism value, as is the case for fauna or flora in other areas (Greig & de Villiers, 1982; Simberloff, 1998; Walpole & Leader-Williams, 2002; Sharpley, 2007).

1.3. Thesis Structure

This thesis is composed of five chapters:

Chapter One is a general introduction elaborating on the current status and threat of the Cape lowlands vegetation types, as well as land-use history. Furthermore, this chapter also introduces the importance of restoration of old agricultural fields in the conservation context.

Chapter Two contains a literature review on the effect of soil properties and restoration methods on native plant re-establishment in the Mediterranean climate region particularly in

the Western Cape. This chapter will conclude by synthesising the literature review and presenting the objectives, key questions and hypotheses of the thesis.

Chapter Three investigates the soil changes associated with cultivation (i.e. pH, Electrical conductivity (EC), moisture, available Nitrogen and Phosphorus) and discusses plant-soil feedback.

Chapter Four examines the effectiveness of four treatments, burn (during autumn and spring), herbicide application, as well as a combination of autumn burn and herbicide application, on reducing alien plants species and enhance native species re-establishment.

Chapter Five is comprised of the general discussion, based on the link between soil chemical properties and the vegetation cover. It will also provide the general conclusion and recommendations on the restoration of West Coast lowland vegetations.

The reference style applied for all chapters will follow the African Journal of Ecology.

1.4. References

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Chapter 2:
Effect of soil properties and restoration efforts
on native plant re-establishment
in the Mediterranean climate region

2.1.Introduction

Most of the landscapes of the Mediterranean climate region, which includes Southwest Australia, the Mediterranean Basin, California Floristic Province, Central Chile and the Cape Floristic Region (CFR) (Cowling & Richardson, 1995; Cowling *et al.*, 1996; Myers *et al.*, 2000; Mittermeier *et al.*, 2004), have been transformed, mainly due to agricultural practices, significantly reducing the extent of natural vegetation. Although the Mediterranean climate region represents less than 5% of the Earth's surface, it has about 48 250 known vascular plant species, which represent almost 20% of the world total (Cowling *et al.*, 1996). All Mediterranean climate regions are regarded as biodiversity hotspots due to having highly threatened vegetation (Myers *et al.*, 2000; Mittermeier *et al.*, 2004).

Like the other Mediterranean climate regions, the landscape of the CFR has been highly transformed mostly due to agricultural practices (Parker, 1982; McDowell & Moll, 1992; Heydenrych & Littlewort, 1995; Low & Rebelo, 1996; Hoffman, 1997; Krug, 2004; Rebelo *et al.*, 2006). These activities have introduced non-native plant species that dominate old, abandoned agricultural fields and negatively affect native plant re-establishment in secondary succession thereafter. Although the re-establishment of native species on abandoned cultivated fields has been the focus of a number of studies, satisfying results are still lacking due to slow re-growth of indigenous species, mainly due to the absence of indigenous seeds present in the soil seed bank after cessation of agricultural practices (Hammouda, Heneidy & Elkady, 2003; Reiné, Chocarro & Fillat, 2004). The lack of native plant establishment has also been related to the absence of dispersal of indigenous seeds onto old agricultural fields (Ne'eman & Izhaki, 1996; Hamilton, Holzapfel & Mahall, 1999; Stylinski & Allen, 1999; Seabloom *et al.*, 2003; Shiponeni, 2003). In addition, the presence of non-native species affects negatively on the re-establishment of native species (Hamilton *et al.*, 1999; Stylinski & Allen, 1999; Meiners, Picket & Cadenasso, 2002; Shiponeni, 2003; Bonet, 2004; Gerlach, 2004; Milton, 2004; Shiponeni & Milton, 2006; Coleman & Levine, 2007). Furthermore, grazing by indigenous herbivores might also affect the re-establishment of indigenous species on old agricultural fields (Midoko-Iponga, 2004; Midoko-Iponga, Krug & Milton, 2005).

Soil nutrients may also be responsible for preventing the re-establishment of indigenous plant species, as well as contributing to the persistence of alien grasses (Tilman, 1987; McLendon & Redente, 1991; Zink & Allen, 1998). This literature review attempts to compile a concise report on how soil properties and restoration treatments might play a role in the re-establishment of indigenous plant species on previously cultivated fields.

2.2. Change in vegetation and soil chemistry on agricultural fields after abandonment

The cessation of cultivation is the start of secondary succession, which is driven by a change in species richness and cover of exotic and native plant species. Usually, alien plant species are more abundant than native species after cultivation, and species richness of both indigenous and exotic species increases with time since abandonment. On old fields in a semi-arid Mediterranean region of Spain, the secondary succession was characterised by an increase of shrub species with time since abandonment (Bonet, 2004; Bonet & Pausas, 2004). However, both studies differed in the re-colonisation pattern of exotic plant species and native plant species during secondary succession. The cover of different life form types is dependent to the successional age of the old field, with exotic grasses and shrubs showing contrasting change with time since abandonment (Ne'eman & Izhaki, 1996; Bonet, 2004; Bonet & Pausas, 2004; El-Sheikh, 2005).

In other regions, exotic species have been shown to be abundant up to a few years after abandonment, and are replaced by native species such as shrubs at a later stage (Foster & Tilman, 2000; Gross & Emery, 2007; Prach, Lepš & Rejmánek, 2007). However, based on land-use history, exotic species may persist for a longer period. Their persistence is related to the fact that exotic species replace other exotic species during a secondary succession (Kulmatiski, 2006; Mau-Crimmins, 2007). The secondary succession could also be disturbed by soil chemistry after abandonment.

With time since abandonment, changes in the soil as well as vegetation changes take place. Over the last two decades, a few studies have tried to assess the change in the soil chemistry and their relationship with vegetation change. The modification of nutrients, such as nitrogen and phosphorus, has been the primary focus when investigating secondary succession on old fields. The level of nitrogen influences the vegetation cover during the secondary succession (Tilman, 1987), and a high presence of alien species is related to a high concentration of nitrogen and phosphorus just after the cessation of agriculture practice (McLendon & Redente, 1991).

Similar to vegetation, soil moisture also shows different patterns with time since abandonment, mainly due to the absence of soil management after abandonment of cultivation (Bonet, 2004; Du *et al.*, 2007). The level of soil nutrients also change with time since abandonment. Comparing old fields of different ages, Walton (2006) found that the level of available phosphorus was the highest in a 5-year-old abandoned old field, whereas in 15 and 30 year old fields, levels of total nitrogen were the highest. These changes could be attributed

to a cessation of nutrient enrichment of the soil through fertilisation, as well as the formation of a litter layer. However, the vegetation on the old fields also affects soil chemistry (Ehrenfeld, 2003; Richardson & van Wilgen, 2004; Ehrenfeld, 2006). Non-native species have been recognised to decrease water availability, have very little effect on available phosphorus and nitrogen concentration, and increase the level of organic carbon (Ruecker *et al.*, 1998; Evans *et al.*, 2001; Ehrenfeld, 2003; Mack & D'Antonio, 2003; Hawkes *et al.*, 2005; Domènech *et al.*, 2006). Nevertheless, the observed changes in the soil during the secondary succession has been reported to be mostly land-use dependant (Kulmatiski, Beard, & Stark, 2006; Standish *et al.*, 2006; Kulmatiski & Beard, 2008), and are also dependent upon the number and cover of exotic plant species established during secondary succession. These changes in soil chemistry on old agricultural fields, compared to natural vegetation, might also affect the re-establishment and growth of native species.

2.3. Effect of soil properties on re-establishment of native species

A number of soil properties can influence the re-establishment of native species, as well as contribute to the persistence of exotic species on old fields. Soil properties investigated in the thesis, and discussed here, are pH, electrical conductivity (EC), soil moisture, and selected nutrients, in this case, nitrogen and phosphorus. Pausas & Austin (2001) recognised that pH was a factor related to nutrient and toxin availability. Although Domènech *et al.* (2006) found that soil pH did not affect invasions by *Cortaderia selloana* of abandoned agricultural lands in north-eastern Spain; García *et al.* (2007) showed that pH differed between abandoned agricultural fields, and was the lowest in areas where the percentage cover of grasses was the highest. Coastal renosterveld and alluvium fynbos grow on acidic soils (Cowling & Holmes, 1992; Steyn, 1994; Heydenrych & Littlewort, 1995; Rebelo *et al.*, 2006), with soil pH ranging from 5.1 (under *Elytropsappus rhinocerotis*) to 5.3. (in an open field) (Mills, 2003). The pH was shown to decrease with old-field age, nearing natural levels after 30 years of abandonment (Walton, 2006).

While soil pH provides information on the acidity or alkalinity of the soil, other information is also valuable such as Electrical Conductivity (EC) that is a measure of salt concentration in the soil (van der Watt & van Rooyen, 1995). The level of salinity was not different between invaded and non-invaded areas in Spain, and EC did not have an effect on persistence of alien grasses (Domènech *et al.*, 2006). In West Coast Renosterveld, Mills (2003) found that the electrical conductivity was similar on soils from wheat fields and natural vegetation mostly composed of geophyte species, renosterbos and Cape wire grass. He suggested that the similarity could be caused by the application of fertilizers.

In the wheatbelt of Australia, however, the removal of natural vegetation lead to an increase in salinity levels on cultivated lands (Cramer & Hobbs, 2002; Cramer, Hobbs & Atkins, 2004; Cramer *et al.*, 2007). In other regions, high levels of salinity on old agricultural fields have been related to agricultural practices, such as irrigation, and quality of water that contribute to the increase in salt levels in the soil (Pannell, 2001; Chhabra, 2005; Browning, Bauder & Phelps, 2006; Watt, García-Berthou & Vilar, 2007; Jalali *et al.*, 2008). The high concentration of salt in the soil causes an increase in osmotic pressure, making water uptake difficult for plants (Castellanos *et al.*, 2005; Cixin He, 2005; Mau-Crimmins, 2007). Increased salinity might even affect morphology, reproduction and growth of plants (Cixin He, 2005). As high levels of salinity negatively affect crops (Koyro & Eisa, 2008), an elevated salt concentration in the soil could also have negative effects on both indigenous and exotic plant species. In old fields with high salt concentration in the wheatbelt area of Australia, exotic species were replaced with those that were tolerant to high salinity (Cramer *et al.*, 2004; Cramer *et al.*, 2007). The authors also reported that a high salinity concentration could play a role in preventing the re-establishment and growth of native species. High salinity could be a particular problem in certain lowland soils of the fynbos biome already having potential high levels of natural salts (Rebelo *et al.*, 2006), as agricultural practices in that region might further increase soil salinity levels.

Soil moisture affects plant reproduction and development in semi-arid and arid regions (Maestre *et al.*, 2001). Water stress is one of the main factors influencing growth, reproduction and competitive abilities of plants (Rodríguez-Iturbe & Portorato, 2004). Each plant species or individual has a specific relationship with soil moisture, and the fact that water is continuously, albeit randomly, distributed in the soil, determines the habitat preference of specific species. In a study investigating the effects of soil properties on secondary succession on old fields in Spain, Bonet (2004) found that soil moisture and the vegetation cover were negatively correlated, but he did not specify whether native or exotic plant species were more affected by soil moisture.

Persistence of exotic species, such as alien grasses, negatively influences the re-establishment of native species on old cultivated lands. Eliason & Allen (1997) found that alien grass reduced available soil water that increases competition for native shrubs, and shrub recovery was better in absence of alien grasses in a period where soil moisture was sufficiently high. A decline in available soil water favours the persistence and the growth of exotic plant species such as grasses in southern California (Hamilton *et al.*, 1999), while at the same time, exotic annual grasses affect soil moisture that plays an important role in shrub re-establishment

(Cione, Padgett & Allen, 2002; Gerlach, 2004). In old forest plantations, North *et al.* (2005) reported that herbs reduced soil moisture, which was lowest under shrubs in a Sierra Nevada forest. Grasses and shrubs have also been found to decrease soil moisture (James *et al.*, 2003). Overall, the reduction of available soil water by alien grasses might be an important factor inhibiting the re-establishment of native plants (Eliason & Allen, 1997; Levine *et al.*, 2003).

Nitrogen has been reported to be the main nutrient for plant development (Aulakh & Malhi, 2004; Balasubramanian *et al.*, 2004) and is efficiently used by plants when it is combined with phosphorus (Aulakh & Malhi, 2004). Both nutrients have been related to the dominance of alien grasses over indigenous plants on old agricultural lands (Tilman, 1987; Zink & Allen, 1998; Milton, 2004).

In the western Australian wheatbelt, Hester and Hobbs (1992) found that soil phosphorus and nitrogen levels were highest in those areas with the highest percentage cover of alien plants. Similar results were found in old fields in California (Eliason & Allen, 1997), and for abandoned cultivated lands in Spain (García *et al.*, 2007), cover of alien grasses was the greatest where available phosphorus and total nitrogen were the highest.

Other studies concluded that the persistence of non-native species might only be attributable to one of the two nutrients. In semi-arid sagebrush, persistence of alien grass is mostly determined by nitrogen (McLendon & Redente, 1991). Paschke, McLendon & Redente (2000) showed that the youngest abandoned field had the highest level of available nitrogen, and high levels of nitrogen availability were related to presence of alien grasses. A high concentration of nitrogen contributes to the persistence of exotic grasses and inhibits shrub establishment in a range of ecosystems (Zink & Allen, 1998; Bakker & Berendse, 1999; Cione *et al.*, 2002). However, when investigating the impact of phosphorus and nitrogen on vegetation recovery on abandoned agricultural fields in Spain, Ruecker *et al.* (1998) reported that the soil phosphorus levels were higher in permanently grazed grasslands than in old fields covered by shrubs and alien grasses. Here, available phosphorus seemed to decrease with time since abandonment.

In a study in central Italy, Bonanomi, Caporaso & Allegranza (2006) found that high levels of nitrogen permitted the increase of living biomass, but at the same time favoured very low species diversity. This result could confirm the persistence of alien grass due to high levels of nitrogen on abandoned lands, but recently some studies have indicated that different levels of soil phosphorus and nitrogen do not affect natural shrub cover (Bechtold & Inouye, 2007; Henkin *et al.*, 2006; Holmes, 2008). Nevertheless, other soil properties should be investigated

to understand their impacts on the re-establishment of indigenous species on abandoned fields.

2.4. Restoration of natural vegetation on old cultivated fields

Restoration of old agricultural lands carried out across Mediterranean-type climate regions aims to facilitate the re-establishment of native species by removing exotic species that establish during secondary succession. As cultivation also changes soil properties, new treatments focusing on changing soil chemistry to prevent persistence of exotic species should also be investigated.

A number of different restoration treatments, such as burning, herbicide application or mowing, have been used to reduce cover of alien grasses and to stimulate the re-establishment and re-growth of native plants. In an old-field restoration experiment in a Swartland Alluvium Fynbos and Swartland Shale Renosterveld mosaic, total species richness and diversity (as determined with the Shannon-Wiener Index), were reduced with herbicide, whereas burning was shown to enhance total diversity (Midoko-Iponga, 2004). Furthermore, the combination of autumn burn and pre-emergent herbicide application has been suggested to increase indigenous species diversity (Musil, Milton & Davis, 2005). The application of these two treatments also contributed to an increase in native species cover, particularly forbs and geophytes, while simultaneously reducing alien grass cover (Midoko-Iponga, 2004; Musil *et al.*, 2005). Although the authors worked in the West Coast region of South Africa, difference in treatment costs are most likely related to additional employment costs for labourers in the study by Musil *et al.* (2005). A number of studies in the Cape Floristic Region found that, although herbicide application was the most expensive treatment method, it was the most effective in reducing cover of alien species (Midoko-Iponga, 2004; Musil *et al.*, 2005; Holmes, 2008). However, that decrease of alien plant species did not always facilitate the re-establishment of native species. Where indigenous re-establishment needs assistance, sowing seeds of annual and perennial plants in treated plots could enhance plant density, cover and richness (Midoko-Iponga, 2004; Holmes, 2005; Holmes, 2008). Invasive plants threaten California sagebrush, similar to coastal renosterveld and alluvium fynbos. Cione *et al.* (2002) found that even though the reduction of alien grass cover was possible with herbicide application and hand weeding, another clearing was necessary after six months using other methods to reduce the persistence of alien grasses on old fields. Furthermore, in that region, spring burning has been reported to mostly reduce exotic grasses (e.g. *Bromus diandrus*), while solarisation, which is a warming of the soil with solar radiation through a plastic

solarisation sheet, was recorded to destroy the seed bank, particularly for *Bromus diandrus* (Moyes, Witter & Gamon, 2005).

To restore natural vegetation on an old boreal hayfield, Antonsen & Olsson (2005) found that mowing compared to burning was the best method to enhance the re-growth of native plant species diversity, while the repeated use of glyphosate, an herbicide, has been recorded to remove *Cynodon dactylon* in Arizona (Mau-Crimmins, 2007).

As discussed earlier, the persistence of exotic grasses on old fields has been related to high levels of nitrogen. Some restoration projects have tried to find the best ways to reduce the level of nitrogen in the soil. Cione *et al.* (2002) attempted to reduce the level of nitrogen in the soil where the native Southern Californian coastal sage scrub vegetation was disturbed by frequent fires, high anthropogenic nitrogen deposition and invasive annual weeds. Although nitrogen immobilization by mulch did not reduce nitrogen levels quickly enough to favour native plant re-establishment, they found that the high C: N ratio in available mulch could be a practical answer to decreasing high soil nitrogen levels. Paschke *et al.* (2000) used sucrose as treatment to decrease soil nitrogen levels that significantly reduced available nitrogen. Sucrose is not suitable for large-scale application, however, because of the cost. Other methods such as sawdust application (Baer *et al.*, 2004), or the application of carbon (Baer *et al.*, 2003), are also aimed at reducing nitrogen levels in the soil. To reduce high salinity concentration, Browning *et al.* (2006) proposed using e.g. gypsum and salt leaching as a potential treatment. Gypsum has also been recognised to reduce available phosphorus (Suding, LeJeune & Seastedt., 2004; LeJeune, Suding & Seastedt, 2006). Only recently have old-field restoration attempts begun to include aspects of soil restoration. This may assist in changing the soil properties that have been recognised to prevent and inhibit native species re-establishment. However, few of those restoration projects compared the soil chemistry between old fields and soil under natural vegetation (e.g. Holmes 2005; 2008) before applying their soil treatments; rather they are focusing on certain soil properties based on assumptions. The comparison of soil properties between old agricultural fields and natural vegetation could be used to determine differences in soil chemistry between the sites, and to assist in choosing the most appropriate amelioration treatment via the concentration of substances to be applied to old agricultural fields.

2.5. Conclusion

Secondary succession on old fields does not always follow a straight path with the time since abandonment, mainly due to previous land-use types that affect vegetation change and soil properties differently, as well as the observed plant-soil feedback. Soil properties also

changed with the increasing age of the abandoned old fields. The pH plays an important role in nutrient availability, which is one of the factors driving the competition between alien grasses and indigenous plants. Paschke *et al.* (2000) showed that pH influenced the level of nitrogen. Similarly, electrical conductivity could also play a role in persistence of exotic grasses, as the level of salt in the soil influences pH and water availability for plants. Water is the main factor that promotes native plant re-growth, while persistence of alien grasses reduces soil moisture. In addition, persistence of non-native plants, especially grasses, has been related to high levels of nitrogen and phosphorus in the soil. Although a number of restoration treatments were found to be effective in reducing the cover of alien grasses, they lost their effectiveness after a certain time, making repeat treatments necessary. To prevent this, soil restoration should be used in addition to vegetation restoration to reduce cover of alien grasses more effectively, and to facilitate the re-establishment and re-growth of native plant species.

This study aims to compare the soil properties of old agricultural fields of different ages and their relationship with vegetation cover, and to elaborate an old agricultural fields restoration method in order to reduce alien plant species and to enhance native plant species re-establishment. To achieve these aims, this study will focus on vegetation and soil properties.

Below-ground studies will investigate the changes in soil properties of old agricultural fields, 10 and 20 years after cessation of cultivation, according to their topography. Above-ground studies will focus on the removal of alien grasses through trial treatments; autumn burning; autumn burning in combination with herbicide; herbicide application alone; as well as spring burning, to reduce the cover of undesirable plant species and enhance the cover of indigenous plant species on the 10-year-old agricultural land.

The study aims to answer the following key questions:

1. Are pH, EC, soil moisture, total nitrogen and phosphorus availability different on the three habitats (ditch, slope and ridge) by depth and the old-field age?
2. Which soil properties affect the native plant cover and contribute to the alien grass persistence?
3. Which treatments have the best effect in reducing undesirable plant species and enhancing native plant cover?
4. Is there any different effect in reducing alien plant species cover and enhance native plant cover between autumn and spring burn?
5. How long are the treatments effective after their application?

We predict that:

1. Soil chemistry will differ accordingly to the topography and the age of old agricultural fields since abandonment;
2. Soil salinity and pH will be the main soil properties driving the vegetation changes that will be also influenced by available P and total N;
3. Herbicide alone and combination of autumn burning and herbicide application should have the best effectiveness on reducing cover of alien grass and *Cynodon dactylon*, as well as enhancing some native plant species cover;
4. Autumn and spring burns reduce alien plant species, but spring burn will reduce mostly grasses such as *C. dactylon* that cover old field in summer; and
5. The treatments are effective for at least a year.

2.6. References

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Chapter 3: Soil changes related to transformation

3.1.Introduction

Anthropogenic land-use, such as agriculture, has been one of the main factors driving the transformation of natural vegetation (McDowell & Moll, 1992; Rebelo, 1992; Low & Rebelo, 1996). This led to the degradation of physical soil properties, such as hydraulic soil systems and soil respiration, through compaction practices (Cerdà, 1995; Lipiec & Stepniewski, 1995; McLaughlin & Mineau, 1995; Lipiec & Hatano, 2003), and chemical properties, such as soil organic matter and soil nutrients (García *et al.*, 2007). Agricultural activities such as tillage cause a decrease of soil organic matter and contribute to loss of soil nutrients (Lipiec & Stepniewski, 1995; Naidu *et al.*, 1996; Sibbesen, Skjøth & Rubæk, 2000; Steenwerth *et al.*, 2002; Mills & Fey, 2003; García *et al.*, 2007; Holmes, 2008; Kulmatiski & Beard, 2008). The use of fertilizers, particularly inorganic fertilizers, has been recognised to contribute to the increase of nutrients in the soil, especially nitrogen and phosphorus (McLaughlin & Mineau, 1995; Mills & Fey, 2003). The legacy of agricultural practices contributed to a change in soil properties on old fields after cessation of cultivation activities compared to natural vegetation (McLendon & Redente, 1991; Seabloom *et al.*, 2003; Kulmatiski *et al.*, 2006a).

Agricultural practices have also contributed to the introduction and establishment of exotic plants (Gerlach, 2004; Kulmatiski, 2006; Kulmatiski *et al.*, 2006a). With the cessation of agricultural activities, non-native species are becoming dominant and persistent on old fields due to high levels of nutrients, particularly nitrogen and phosphorus (Tilman, 1987; McLendon & Redente, 1991; Zink & Allen, 1998; Baer *et al.*, 2003; Brandon, Gibson & Middleton, 2004; Milton, 2004; Cramer *et al.*, 2007). Nitrogen is recognised as the main nutrient for plant growth and development (Aulakh & Malhi, 2004; Balasubramanian *et al.*, 2004). High nitrogen levels have been reported to favour an abundance of exotic plant species (McLendon & Redente, 1991; Paschke *et al.*, 2000) and resulting in better grass productivity (Baer *et al.*, 2003). Tilman (1987) also showed that high concentrations of nitrogen increased the abundance and growth rate of annual plant species; inversely, this high level has negative impacts on perennial plant development. Furthermore, nitrogen combined with phosphorus is believed to enhance crop production (Aulakh & Malhi, 2004). The abundance and persistence of exotic plant species have therefore been related to the presence of high levels of nitrogen and phosphorus (Hester & Hobbs, 1992; Eliason & Allen, 1997; Leishman & Thomson, 2005; García *et al.*, 2007). In natural ecosystems, soil nutrients are produced by microorganisms that decompose soil organic matter. However, agricultural practices also affect microbial communities (Mack, D'Antonio & Ley, 2001; Jackson *et al.*, 2003; Kulmatiski & Beard,

2008). Therefore, the predominance of exotic plant species and the alteration of soil conditions inhibit the re-establishment of indigenous species on old fields (Calderón *et al.*, 2000; Hammouda *et al.*, 2003).

The presence of exotic plant species has been documented to alter soil properties such as soil organic matter, nutrients and water (Ehrenfeld, 2003; Richardson & van Wilgen, 2004; Ehrenfeld, 2006). For example, soil nitrogen is altered by the presence of alien species that have high uptake rate, and produce litter that has low net N mineralization (Evans *et al.*, 2001; Ehrenfeld, 2003; Mack & D'Antonio, 2003; Corbin & D'Antonio, 2004; Hawkes *et al.*, 2005; Domènech *et al.*, 2006). Exotic plants species also affect microbial activities by reducing their numbers and inhibiting their mineralization rate (Baer *et al.*, 2003; Kulmatiski & Beard, 2008). Alteration of soil properties by non-native species seems to benefit them in turn, e.g. a decrease in nitrogen level is favourable for the growth of *Bromus tectorum*, but the decline of nitrogen also increased the abundance of native perennial grass (Evans *et al.*, 2001; Paschke *et al.*, 2000).

Changes of soil properties through exotic plant species could be the reason for changes in soil properties with time since abandonment of old fields (Bonet, 2004; Walton, 2006; Du *et al.*, 2007). This could also explain a lack of re-establishment of native species during secondary succession (Kulmatiski, 2006; Mau-Crimmins, 2007). Therefore, the effectiveness of old agricultural land restoration will depend on the condition of both above- and below-ground biotic communities and their interactions.

The aim of this study was to determine the soil properties, particularly pH, electrical conductivity (EC), soil moisture, available phosphorus (P) and total nitrogen (N), on old agricultural fields of different ages and their relationship with vegetation cover. This study will answer the following questions:

1. Do soil properties differ with topography in old fields of different ages?
2. Does vegetation cover differ with topography in old fields of different ages?
3. Which soil properties influence the persistence of undesirable plants and re-establishment of native species?

3.2. Material and methods

3.2.1. Study area

This study was conducted on two abandoned cereal fields of different ages at Elandsberg Private Nature Reserve (EPNR) (Figure 3.1). The first field was abandoned in 1997 (19 ° 02' E and 33 ° 42' S) (Fig 3.2a), the second in 1987 (19 ° 03' E and 33 ° 27' S) (Figure 3.2b). The

old fields serve as buffer zone between agricultural fields and natural vegetation at EPNR. These old agricultural fields are referred to as ‘young field’ and ‘old field’, respectively, throughout the text.

The reserve is part of the farm Elandsberg, which was formed through the fusion of the farms Bartholomeus Klip, Bosplaas, Elandskloof and Langhoogte in 1966 (Nicola Wooding, pers. comm. 2007). Elandsberg is situated 5 km east of Hermon and 25 km north of Wellington in the Tulbagh district of the Western Cape Province. Elandsberg Private Nature Reserve was proclaimed in 1973 to conserve a large population of the endemic geometric tortoise (*Psammobates geometricus*) (Nicola Wooding pers. comm. 2007) found on the Swartland Alluvium Fynbos and Swartland Shale Renosterveld mosaic (Rebelo *et al.*, 2006), previously classified as West Coast Renosterveld by Low & Rebelo (1996). The reserve lies between 33°24’ and 33°30’ South and 19°01’ and 19°05’ East. The reserve is bordered by Swartland Shale Renosterveld and Swartland Alluvium Fynbos surrounding Voëlvlei Dam in the North, cultivated cereal fields in the West, Swartland Alluvium Fynbos and Swartland Shale Renosterveld on the property of Krantzkop ammunition factory (Department of Defence) in the South, and Hawequas Sandstone Fynbos on the Elandskloof Mountain range in the East.

Dutch immigrants were the first farmers to settle in the area in 1705. The farm was used for cattle and sheep grazing until 1936, when the farm was transferred to the current owner. Subsequently, use changed from livestock grazing to game hunting, but in 1960, use was reverted to cattle and sheep grazing (Wooding, 2005). In 1973, a 2 600 ha section of Elandsberg was declared a Private Nature Reserve. The agricultural lands bordering the reserve were abandoned between 1960 and 1997, and serve as a buffer zone between natural vegetation and agricultural lands, increasing the reserve area from 2 600 ha to 3 606 ha. In 1989, Elandsberg Private Nature Reserve was declared a natural heritage site, protecting it from any developments (Nicola Wooding pers. comm. 2007). The recent stewardship agreement between CapeNature and Elandsberg Farms afforded Elandsberg Private Nature Reserve the status of a provincial nature reserve (Nicola Wooding pers. comm. 2008).

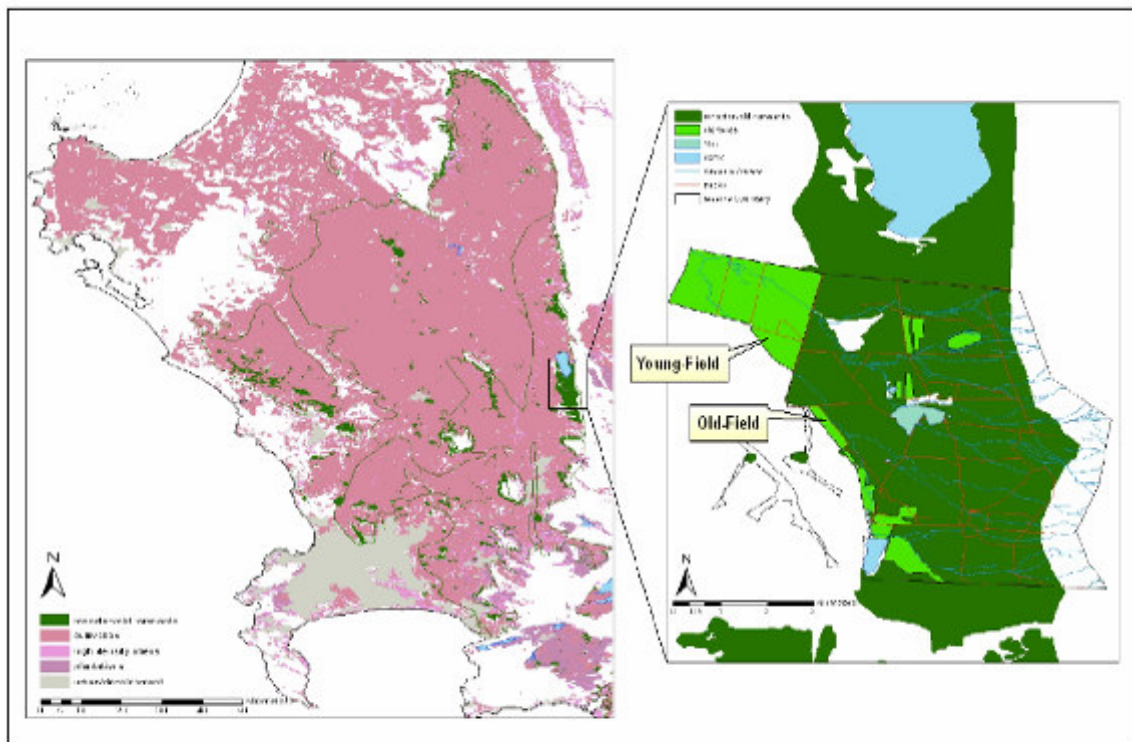


Figure 3.1 Map showing the current extent of transformation of the Western Cape Lowlands (pink: agricultural, grey: urban areas), and remaining natural vegetation remnants (green). Insert shows the location of old fields (bright green) and extent of natural vegetation (Swartland Alluvium Fynbos and Swartland Shale Renosterveld, green) at Elandsberg Private Nature Reserve, and indicates the position of the 10-year-old (young field) and 20-year-old (old field) abandoned cultivated lands. Data for transformed habitats are from Cape Conservation Unit (2003); and geographic information for Elandsberg is from CapeNature (2004). Map obtained from Krug & Krug, (2007).



Figure 3.2 View of the (a) 10 year (young field) and (b) 20 year (old field) old abandoned cultivated lands at Elandsberg Private Nature Reserve.

Vegetation

Three vegetation types occur within the Elandsberg Private Nature Reserve. On the mountains, Hawequas Sandstone Fynbos is dominant, while in the lowlands, previously classified as West Coast Renosterveld (Acocks, 1988; Baard, 1990; Low & Rebelo, 1996), the dominant vegetation types are Swartland Alluvium Fynbos (Rebelo *et al.*, 2006) and Swartland Shale Renosterveld (Rebelo *et al.*, 2006). An ecotone between the mountain and lowland vegetation covers the lower mountain slopes. The old agricultural fields are potentially Swartland Shale Renosterveld or Swartland Alluvium Fynbos (Rebelo *et al.*, 2006). However, the soil texture of these two old fields is characterised by sand, and not clay or loam (Appendix A). Renosterveld does not grow on sandy soil (Rebelo *et al.*, 2006); thereby the natural vegetation on these old fields would likely be Swartland Alluvium Fynbos.

Swartland Alluvium Fynbos is highly threatened, with 75% of its original extent being transformed, mainly due to agricultural practices and urbanization. Currently, 10% of Swartland Alluvium Fynbos is conserved (Rebelo *et al.*, 2006), with EPNR being one of the major locations. Swartland Alluvium Fynbos is characterized by evergreen and both moderately tall and low shrubs. At Elandsberg, the natural vegetation is dominated by low shrubs such as *Elytropappus rhinocerotis* (renosterbos), *Eriocephalus africanus* (wild rosemary / kapokbossie) (Goldblatt & Manning, 2000), and other asteraceous shrubs.

The reserve also harbours a large population of the endemic *Leucadendron corymbosum* (swartveld conebrush / tolbos) and a certain number of native grasses such as *Aristida junciformis* (ngongoni three-awn), *Ehrharta calycina* (polgras / rooisaadgras), *Themeda triandra* (red grass / rooigras), *Sporobolus fimbriatus* (dropseed grass) and *Tribolium uniolae* (hare grass / koringgras) (Goldblatt & Manning, 2000; van Rooyen, 2003). Additionally, a diversity of endemic geophytes is also found in the reserve, the main families are Iridaceae, with plant species such as *Moraea villosa* (peacock flower / blouflappie) *Moraea tripetata* (iris / blou-uintjie), and Oxalidaceae such as *Oxalis hirta* (stamsuring), *Oxalis obtusa* (geeloogsuring), *Oxalis commutata* (sorrel / suring) and *Oxalis versicolor* (candycane sorrel) (Goldblatt & Manning, 2000; Krug, 2004; Wooding, 2005; Walton, 2006).

The abandoned old agricultural fields are almost exclusively covered by *Cynodon dactylon* (couch grass / kweek) an indigenous creeping grass and non-native species such as Mediterranean grasses from Europe. Exotic grasses are mostly represented by *Briza maxima* (big quaking grass / grootbewertjie), *Briza media* (quaking grass), *Vulpia myuros* (rats tail fescue / langbaard swenkgras), *Bromus diandrus* (ripgut brome / langnaaldbromus), and

Bromus pectinatus (Japanese brome / hooigras) (Gibbs Russell *et al.*, 1990; van Wyk, Oudtshoorn & Gericke, 1997; Goldblatt & Manning, 2000; Bromilow, 2001; Shiponeni, 2003; van Rooyen, 2003; Shiponeni & Milton, 2006).

Climate

Elandsberg Private Nature Reserve is located on the West Coast of the Western Cape, which is part of the south-western part of the Southern African sub-continent. This region has a Mediterranean-type climate, which is characterised by hot and dry summers, due to an anticyclone that is a pole-ward migration of a high pressure Hadley Cell and associated with a high frequency of south-easterly winds (Trade-winds), covering a period of three months (December-February) (Baard, 1990; Deacon, Jury & Ellis, 1992; Engelbrecht, 1995; Rebelo *et al.*, 2006). The mean temperature is around 12°C in winter and 23°C in summer (Baard, 1990; Engelbrecht, 1995). Baard (1990) indicated that 30.8°C was the mean daily maximum temperature at Wellington.

The winter months (June to August) are influenced by westerly winds, and characterised by abundant rainfall resulting from cyclonic fronts and associated northwesterly winds arriving with a cold front (Nieman, 1981; Baard, 1990; Deacon *et al.*, 1992; Rebelo *et al.*, 2006).

Geology & soil

Soils at EPNR belong to the Malmesbury Group composed of sandstone formed during the Namibian period (Fry, 1987; Deacon *et al.*, 1992; Cowling & Richardson, 1995; Rebelo *et al.*, 2006; Walton, 2006). In general, alluvium fynbos is found on two types of geological formations: the Malmesbury Group characterised by schists and phyllites, and sandstone (Fry, 1987; Deacon *et al.*, 1992; Cowling & Richardson, 1995; Rebelo *et al.*, 2006). In the south-western Western Cape, the metasedimentary has a Precambrian substratum and granitic material. The Precambrian substratum is characterised by sedimentary rocks that were deformed and metamorphosed during the mountain building in the Pan-African Orogeny, while granite material was formed with the intrusion of large plutons of granitic material during the Saldanian Orogeny (Fry, 1987; Deacon *et al.*, 1992; Gresse & Scheepers, 1993; Cowling & Richardson, 1995; Slabber, 1995; Rebelo *et al.*, 2006).

The Malmesbury shale group was derived from Table Mountain sandstone, which is part of the Table Mountain group (one of the subdivisions of the Cape Supergroup) that eroded over the last 100 million years (Deacon *et al.*, 1992; Cowling & Richardson, 1995; Slabber, 1995; Rebelo *et al.*, 2006). This mountain erosion produced the sandy colluvial material that covered the residually weathered clay. This superposition gave rise to a soil that is very

sandy, with a pale-coloured A/E horizon, and periodically saturated with water on gleyed, prismatic and clay that might be base-saturated and even saline. The sandy soil is usually acid and highly leached. It was formed with the pedimented colluvial and alluvial accumulation products in intra- and intermountain valleys (Rebello *et al.*, 2006).

3.2.2. Experimental design and procedures

We defined three habitats: ditch, slope and ridge, according to the topography of the old fields (Figure 3.3). The ditch is the lower part of the old furrow (drainage line), the ridge the cultivated area and, between these two habitats, the slope is the transitional area (Figure 3.3). In each of the three habitats, two holes of 50 cm by 30 cm deep were dug. Along a vertical transect in the soil pit, samples were taken from 0 cm to 6 cm depth at 2 cm intervals. Below that depth, soil samples were collected at 10 cm, 20 cm, and 30 cm soil depth. Sampling was conducted once every season: winter (August), spring (November), summer (February-March) and autumn (May-June). In total, six soil samples were gathered in each hole, and twelve soil samples per habitat were collected. For each soil sample, soil moisture was determined, then the remaining sample was air dried and sieved to < 2 mm for pH, Electrical conductivity (EC), available Phosphorus (P), and total Nitrogen (N) analyses (García *et al.*, 2007). Additionally three soil samples were randomly collected, with an auger, at 10 cm depth in three sampling points distance by 50 cm to one another, in both natural vegetation and in a cultivated field close to the 20 year old field. The cultivated field was not yet covered by crops in autumn. For these samples, pH, EC, soil moisture and available P were determined.

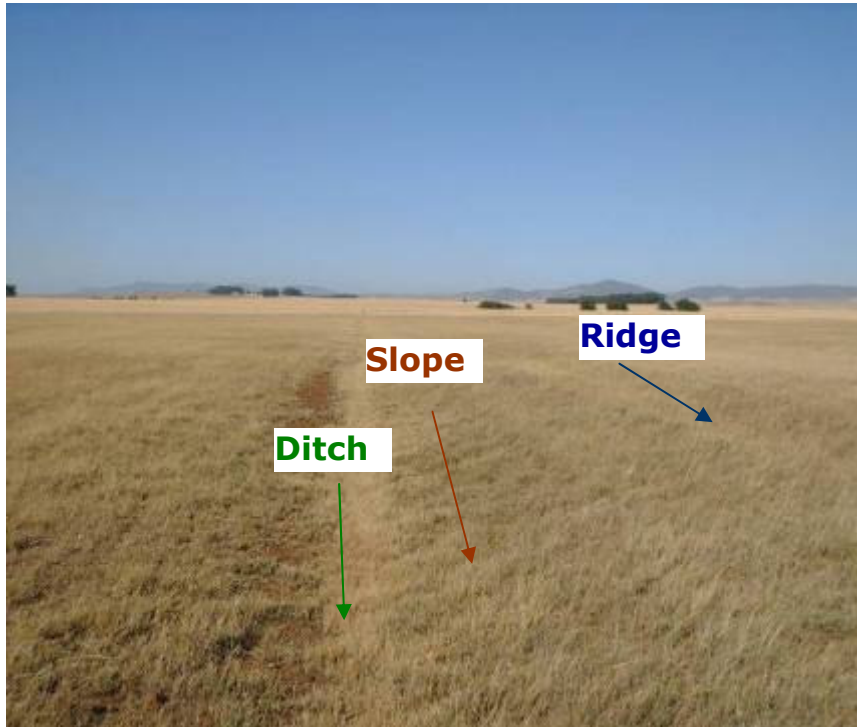


Figure 3.3 Photograph showing the topography with the three habitats (ditch, slope and ridge) investigated on the 10-year-old field.

Soil moisture

To determine soil moisture, 10 g of fresh soil was placed to a tared moisture tin and dried overnight in a drying oven at 105°C. In the morning, the samples were transferred to desiccators to cool and then weighed (Bear, 1964). Soil moisture was determined gravimetrically (from the recorded loss in weight), using equation (1) below (van Reeuwijk, 2002):

$$Moisture(\%) = \left(\frac{A - B}{B} \right) \times 100 \quad (1)$$

where A is 10 g of fresh soil plus tare tin weight

B is dry weight of soil plus tare tin weight

pH

The pH was determined using a soil: KCl suspension, as a KCl solution is less affected by changes in biological and meteorological aspects of the soil, and the pH obtained with this solution mostly closely characterises the true soil pH (Peech, 1965; Bloom, Skyllberg & Summer, 2005).

To determine the pH, 20 g of soil was mixed with 50 ml of 1 M KCl in a beaker. The soil KCl suspension (1:2.5) was stirred for 30 minutes and left to stand for about one hour (Peech, 1965; Page, Miller & Keeny, 1982), after which the pH was measured using a pH meter (Metrohm 744, Swisslab, Germany) calibrated with standard buffer solutions.

Electrical Conductivity (EC)

A water-salt solution is a good conductor of electrical current, and electron transport is proportional to the level of salt present in the solution. Therefore, by determining the electrical conductivity, the total ionized concentration constituent is accurately indicated (Bower & Wilcox, 1965).

Mixtures composed of 10 g of soil and 50 ml of distilled water were shaken for one hour (Page *et al.*, 1982; van Reeuwijk, 2002). Electrical conductivity was determined in the 1:5 soil: water suspension using an EC meter (Jenway 4510, Adcock Ingram, Switzerland).

Available phosphorus

Phosphorus availability was determined using a colorimetric method, which depends on the P concentration in the solution, the concentration of interference elements in the solution to be analysed and the particle acid involved in the procedure (Olsen & Dean, 1965).

A mixture composed of 6.7 g of soil, 50 ml of bray II, 0.03N NH₄ and 0.1 N HCl was made in a 200 ml Erlenmeyer flask. Olsen & Dean (1965) indicated that Bray II extraction is best for P availability assessment in acid soils. The soil: bray II suspension was filtered after the solution was shaken for 40 seconds. Twenty ml of the cleared filtrate were mixed with 10 ml of Boric acid-Molybdae reagent and shaken. Thereafter, 10 ml of A. N. S. A. (Amino Naphol Suphonic Acid), a reducing agent, and 50 ml of distilled water were added, and then left for 10 minutes. This solution activates the coloration of the soil: bray II suspensions. Absorbance was then read at 660 nm with a spectrophotometer (Ultrospec III) (van Reeuwijk, 2002).

Based on this reading, available phosphorus was determined using equation (2):

$$\text{Available P concentration} = \beta \text{ (mg/l)} \times 7.46 \text{ l/kg} \quad (2)$$

β (mg/ l) = value read on the spectrometer and

$7.46 \text{ l/kg} = 50.10^{-3} \text{ l of bray II/ } 6.7.10^{-3} \text{ kg of soil sample}$

Total Nitrogen

The EuroVector instrument was used to determine total nitrogen by combustion. The soil samples were ground and mixed to ensure that the soil sample is an accurate representation of the total soil contents (Toon & Ellis, 1973; Nelson & Sommers, 1996).

After grinding, the sample was transferred into a tin sample cup that was placed into a quartz reactor with a temperature of 1 030° C and a constant flow of He gas. The combination of temperature and O₂ condition oxidised the tin to SnO₂, increasing temperature to between 1 700 and 1 800° C, leading to complete combustion of soil organic matter. This combustion produces CO₂, nitrogen oxides (NO₂,) and H₂O. The helium carries these gas products through a column of chromium dioxide (CrO₂), which catalyses the oxidation of organic fragments, and halogen sulphur oxides are removed by a combination of Co₃O₄ and Ag. The gases produced move through a heated column at 650° C, which contains Cu that removes excess O, and Mg (ClO₄)₂ that removes H₂O. After the removal of these elements, the gases enter the chromatographic column dividing N₂ and CO₂, and the different gasses present are recorded with a thermal conductivity detector (Toon & Ellis, 1973; Nelson & Sommers, 1996)

3.2.3. Vegetation survey

Three control plots (4 m × 20 m), which were set up for an earlier restoration project (Midoko-Iponga, 2004), were used to measure seasonal cover of vegetation on the old field. On the young field, three of the five controls plots (50 m × 50 m) set up for restoration trials (this thesis, chapter 4), were used for the vegetation surveys. The surveys took place in winter (August) and spring (November) 2006, as well as summer (February-March) and autumn (May-June) 2007. The percentage cover of vegetation on each plot was visually estimated for each habitat (ridge, slope and ditch) twice with a randomly placed 1 m × 1 m grid with 100 subsections (1 subsection= 1%) (Milligan, Putwain & Marrs, 2003). Species cover of less than 1% was allocated a percentage cover equal to 0.5 % (McGinley & Tilman, 1993). Recognisable species were identified using previous works (Shiponeni, 2003; Krug, 2004; Walton, 2006), grass identification guides (Grabandt, 1985; Botha, 2001; Bromilow, 2001), and with the assistance of Mr. Raphael Kongor and Prof. Sue Milton (Department of Conservation Ecology and Entomology). The life form and life cycle were ascribed to species according to Germishuizen and Meyer (2003) (Appendix B).

3.2.4. Statistical analysis

Data obtained from soil samples were analysed using the software package STATISTICA 7 (Statsoft Inc, 2006). Data was checked for normality before proceeding with the analyses. Soil moisture data was normally distributed, whereas pH and EC data were not, thus we log₁₀-transformed them to satisfy the assumption of normality, while for graphical interpretation of the results, the inverse transformation was done to display minimum and maximum values in the graphs. A factorial two-way ANOVA was used to test for significant differences in pH

and EC in the three different habitats as function of depth separately for the young and old fields for each season. A two-way ANOVA was used to compare the different levels of pH and EC in the first layer of the profile for each habitat and each season. To compare soil moisture between the two fields, we also conducted a factorial two-way ANOVA but all seasons were combined to observe annual pattern on the three habitats. As the data for available phosphorus and total nitrogen were not normally distributed, a Kruskal-Wallis ANOVA was used to test for significant difference between the three habitats as a function of depth in each abandoned old field. Means and standard deviations of the topsoil samples (0–10 cm) were determined for the young and old fields as well as for natural and cultivated fields.

Species were classified according to their life form. For each 5 m² plot in each habitat, the percentage cover was regarded as absolute for each species identified. Species were allocated to the following life form types: forb, geophyte, grass and shrub (Germishuizen & Meyer, 2003). To determine relative cover (%) for each habitat, absolute cover of each life form was divided by total vegetation cover determined for each habitat. To choose the correct indirect ordination procedure, a Detrended Correspondence Analysis (DCA) was conducted with species data to check the gradient lengths. As we obtained a value of 2.087, we chose the Principal Component Analysis (PCA), according to Lepš & Šmilauer (2003) and Morris & Penderis (2007), to compare the soil properties with vegetation cover of each old field and the combination of both old fields in Canoco for Windows 4.5 (ter Braak & Šmilauer, 2002). Grasses and forbs were further divided into indigenous species and undesirable species such as *C. dactylon* or exotic grasses. Average value of soil properties was calculated per habitat and season.

3.3. Results

3.3.1. Soil properties on two old fields of different ages

pH (KCl)

Pattern in pH value differed between seasons and habitats at 0–2 cm soil depth on the 20 year old field ($F_{16, 12} = 30.673$, $p < 0.0001$) (Figure 3.4). The pH values were significantly different between ridge, slope and ditch as function of depth for all four seasons (winter: $F_{10, 18} = 2.889$, $p < 0.05$, spring: $F_{10, 18} = 17.211$, $p < 0.0001$, summer: $F_{10, 18} = 28.034$, $p < 0.0001$, and autumn: $F_{10, 18} = 21.331$, $p < 0.0001$) (Figure 3.5). In winter, pH of ridge and slope followed a very similar pattern, although the pH values of the ridge were slightly lower than its values in the slope. Ditch soil samples had higher pH values than the other habitats except for 6–10 cm

depth, when pH dropped considerably (Figure 3.5a). In spring, pH values for the three different habitats followed a unique pattern, with ditch and ridge samples having lower pH values than slope samples above 6 cm. Below 6 cm, pH values of slope samples drop considerably (Figure 3.5b). In summer, ditch soil samples showed the lowest pH values above 10 cm soil depth, after which they increased above the pH values of the ridge samples. Both ditch and slope showed a very similar pattern, with a noticeable decrease in pH at 6-10 cm depth (Figure 3.5c). In autumn, pH values of ditch soil samples increased again, and pH values were highest at 0-2 cm and 20-30 cm soil depth. pH values of ridge and slope followed a similar pattern, with slope soil samples having a higher pH than ridge samples (Figure 3.5d).

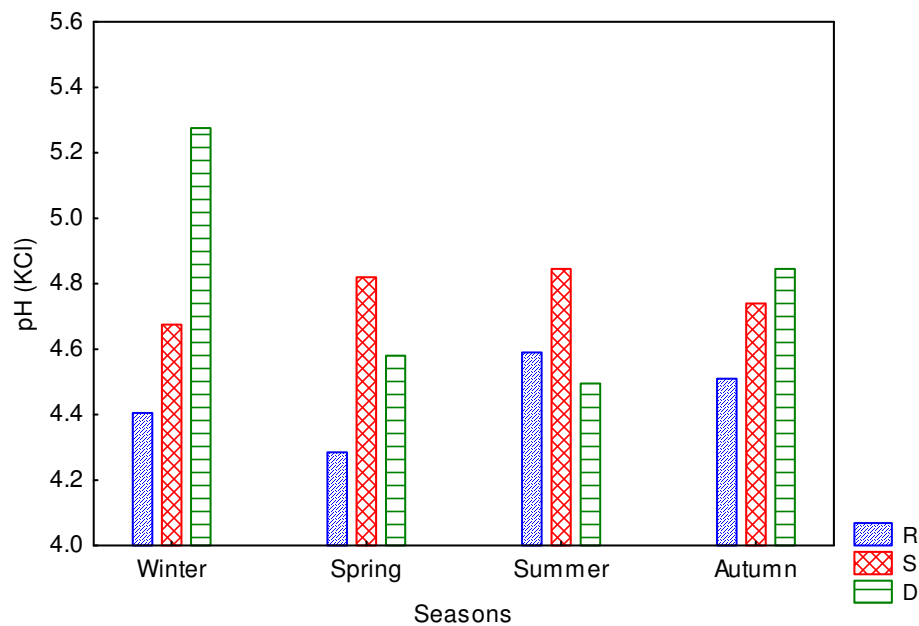


Figure 3.4 Mean pH values for 0–2 cm soil depth on ridge (R, blue), slope (S, red), and ditch (D, green) in the old field for the four seasons sampled.

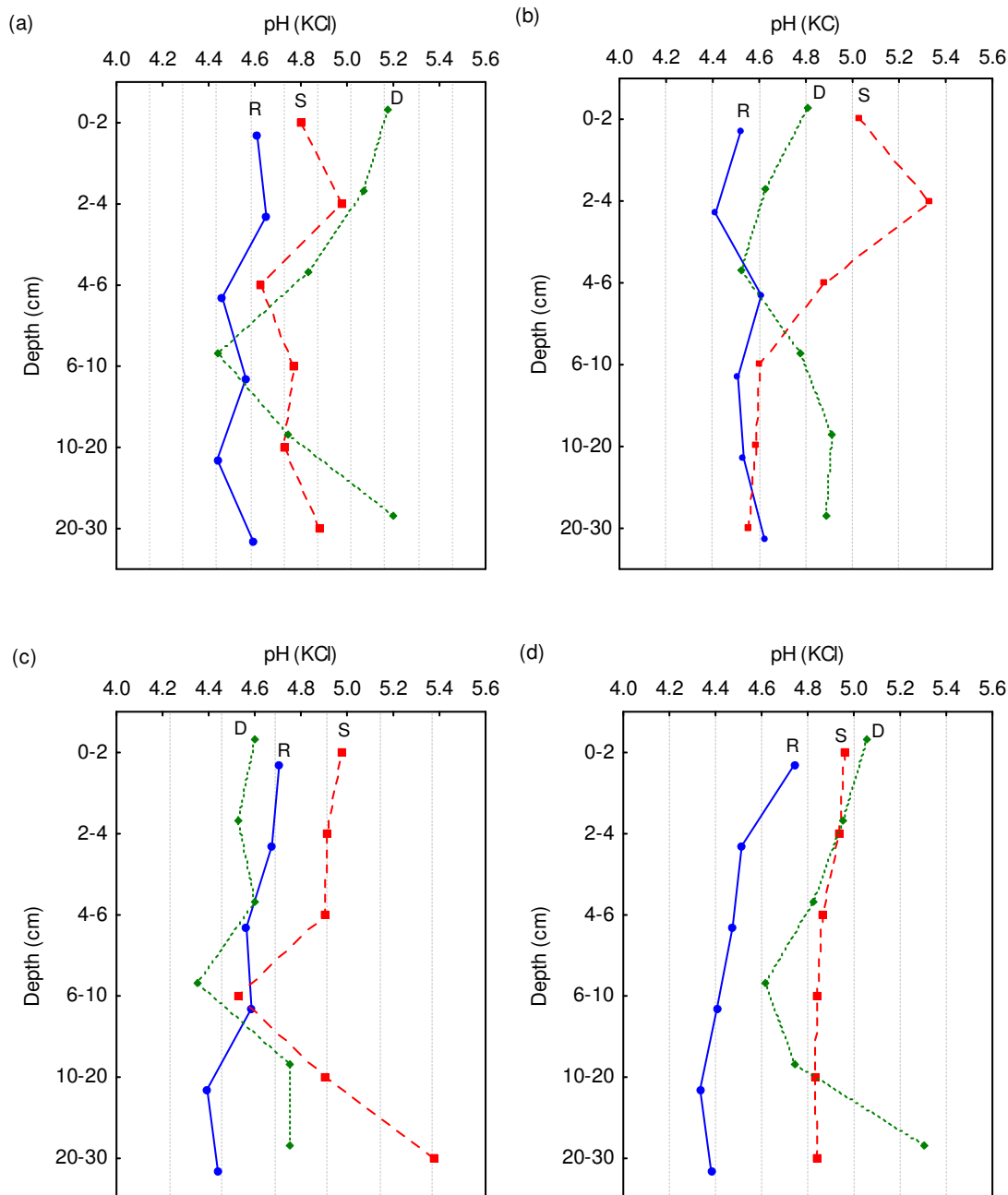


Figure 3.5 Depth profiles of mean pH for ridge (R, blue), slope (S, red) and ditch (D, green) in winter (a), spring (b), summer (c) and autumn (d) in the old field.

In contrast, the pH values on the 10 year old field were not significantly different between seasons and habitats at 0–2 cm soil depth ($F_{16, 12} = 0.228$, $p = 0.959$) (Figure 3.6). The pattern in pH values did not show any difference between habitats as a function of depth in winter ($F_{10, 18} = 0.069$, $p = 0.999$), spring ($F_{10, 18} = 0.201$, $p = 0.993$), summer ($F_{10, 18} = 0.113$, $p = 0.999$) or autumn ($F_{10, 18} = 0.190$, $p = 0.994$) (Figure 3.7). In winter, the pH values of the ridge soil samples were slightly lower than for slope soil samples, but both followed a similar pattern. The ditch soil samples had the highest pH, with change of trend at 4 cm depth (Figure 3.7a). In spring, soil samples pH of the ridge and slope showed a similar pattern, although the ridge

had the lower pH than the slope above 10 cm. Below 10 cm, both habitats had almost the same pH. The ditch soil samples had higher pH values than ridge and slope, but between 4 and 10 cm, the pH value of ditch soil samples was between ridge and slope pH (Figure 3.7b). In summer, the three habitats showed a similar pattern. The ridge soil samples had lower pH values than those of slope and ditch, while soil samples in these habitats had almost the same pH (Figure 3.7c). In autumn, the ridge soil samples showed the lowest pH values, with a different pattern to those in slope and ditch (Figure 3.8d).

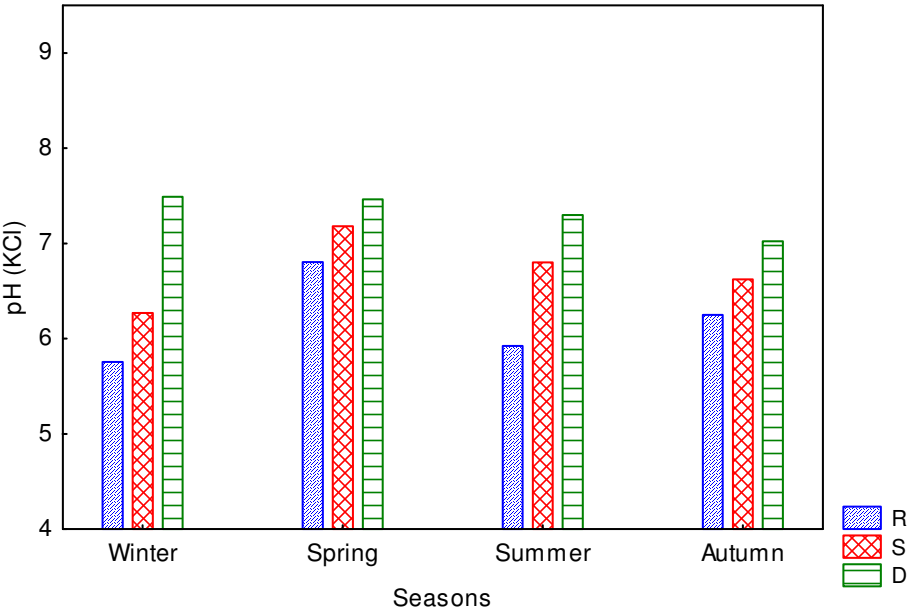


Figure 3.6 Mean pH values for 0–2 cm soil depth on ridge (R, blue), slope (S, red), and ditch (D, green) in the young field for the four seasons sampled.

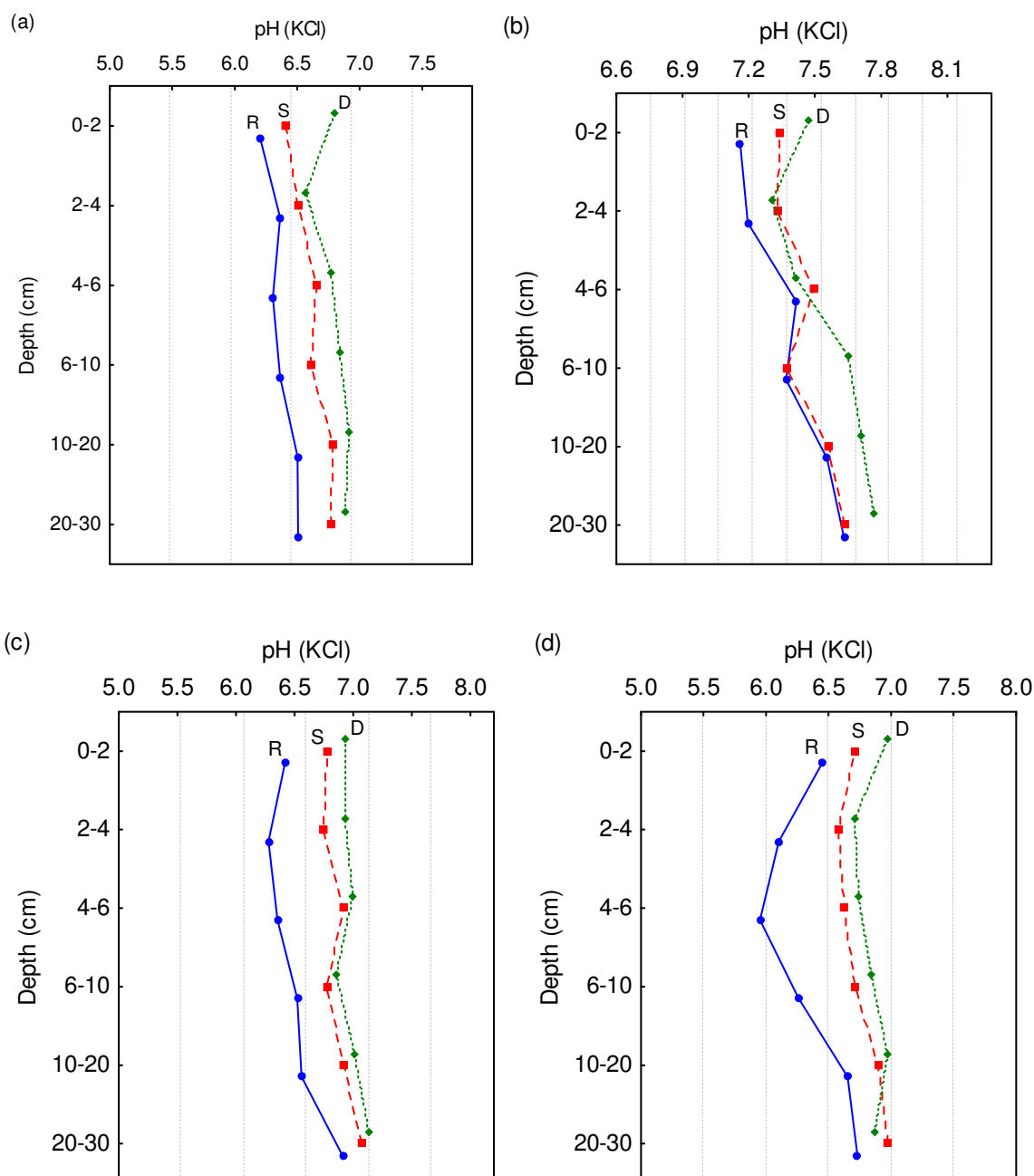


Figure 3.7 Depth profiles of mean pH for ridge (R, blue), slope (S, red) and ditch (D, green) in winter (a), spring (b), summer (c) and autumn (d) in the young field.

Electrical conductivity (EC)

The electrical conductivity was significantly different between seasons and habitats at 0–2 cm soil depth in the old field ($F_{6, 12} = 5.147$, $p < 0.05$) (Figure 3.8). The EC values showed significant differences between the three habitats as function of depth for all four seasons (in winter: $F_{10, 18} = 21.734$, $p < 0.0001$, in spring: $F_{10, 18} = 60.655$, $p < 0.0001$, in summer: $F_{10, 18} = 3.7736$, $p < 0.05$ and in autumn: $F_{10, 18} = 78.741$, $p < 0.0001$) (Figure 3.9). In winter, the EC values were almost constant in the ridge. The slope and ditch soil samples followed similar patterns between the first two soil layers, and the soil samples of the slope had a higher EC

value compared to those of the ditch and ridge. At 4-10 cm, EC of the soil in the ditch and ridge showed similar pattern, with higher EC values than those of slope. Below 10 cm, the EC values increased significantly in the soil samples of the ditch, which had the highest EC values compared to those of ridge and slope (Figure 3.9a). In spring, the EC values decreased in the three habitats between 0-4 cm soil depth, and the ridge soil samples again had the highest EC values when compared to those of the slope and the ditch. Below 4 cm, all three habitats showed an increase in EC, with EC values being highest on the slope (Figure 3.9b). In summer, the three habitats followed a different pattern between 0-10 cm: the ridge and ditch soil samples had almost the same EC values, except in the first layer where EC values were higher on the ridge than the ditch. The slope soil samples had the lowest EC values in that interval. Below 10 cm, the ditch and ridge followed similar pattern and the highest EC values were found in the slope (Figure 3.9c). In autumn, the ridge and ditch followed a similar pattern which was contrary to the pattern of slope soil samples: the slope had the lowest EC values at 0-4 cm depth. Below 4 cm, the slope and ridge followed similar pattern and the EC values were the highest in the ditch (Figure 3.9d).

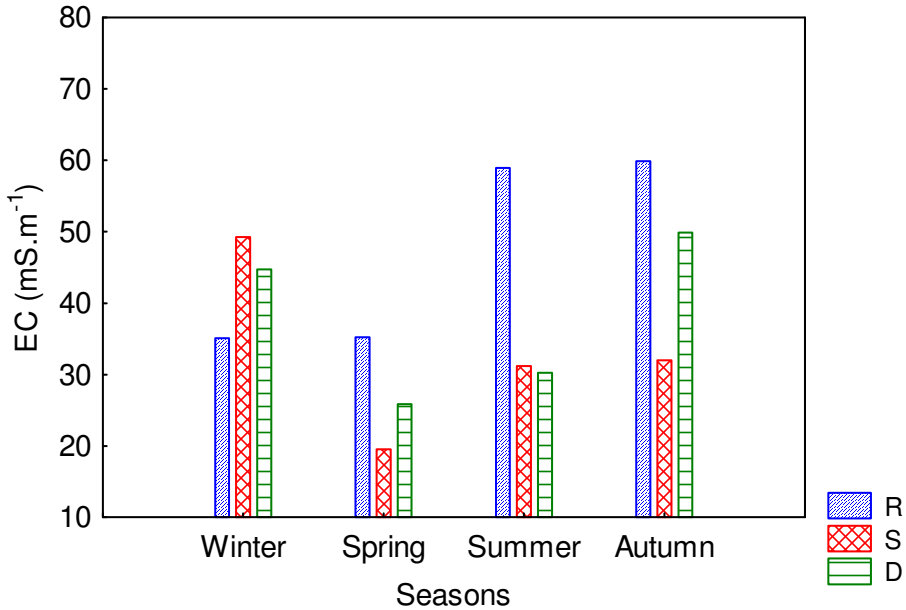


Figure 3.8 Mean EC values for 0–2 cm soil depth on ridge (R, blue), slope (S, red), and ditch (D, green) in the old field for the four seasons sampled.

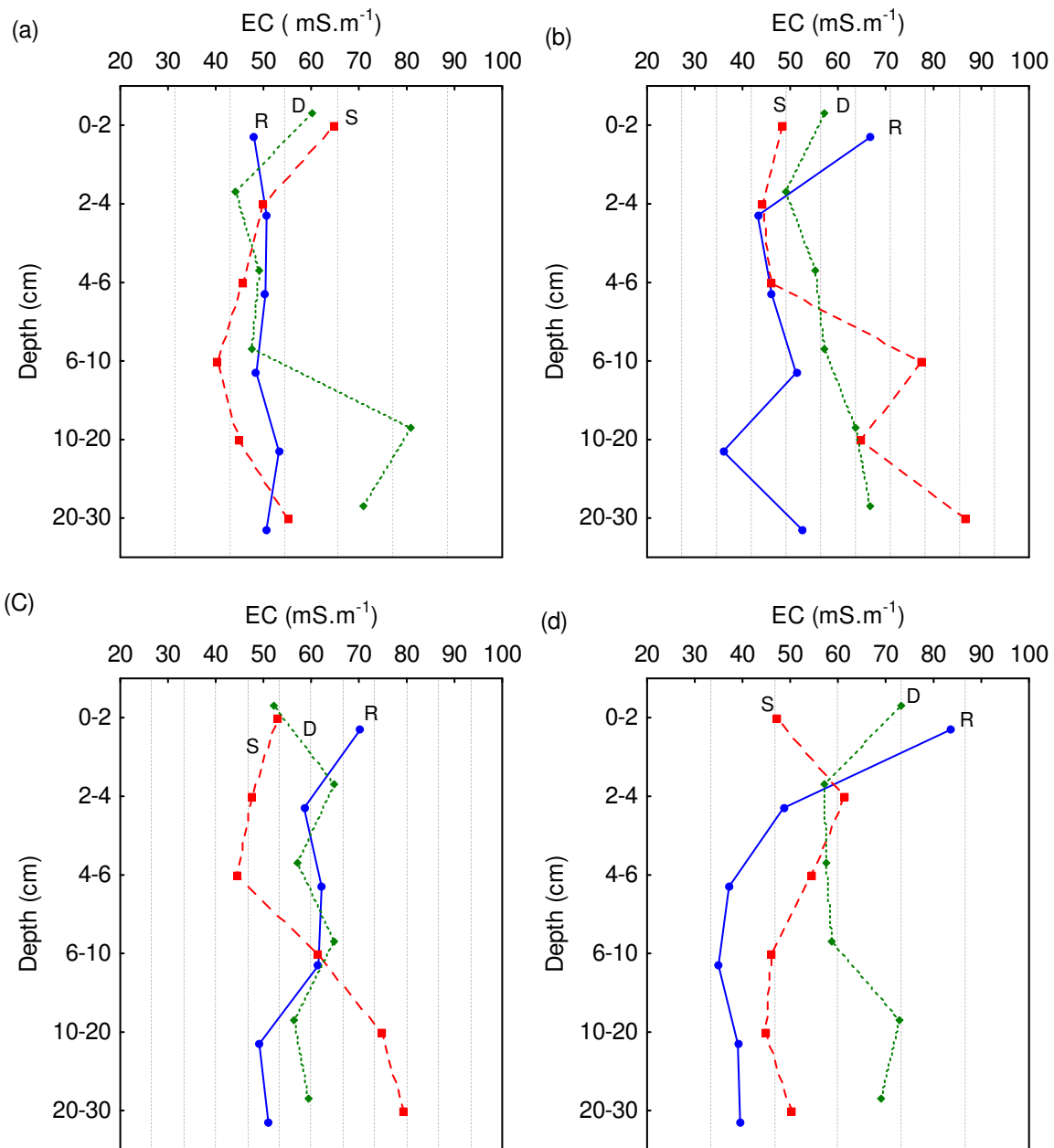


Figure 3.9 Depth profiles of mean Electrical Conductivity (EC) for ridge (R, blue), slope (S, red) and ditch (D, green) in winter (a), spring (b), summer (c) and autumn (d) in the old field.

In the young field, the EC values were significantly different between seasons and habitats ($F_{2, 12} = 3.508$, $p < 0.05$) (Figure 3.10). EC values did not differ between habitats as function of depth in winter: $F_{10, 18} = 0.247$, $p = 0.098$; in summer: $F_{10, 18} = 1.251$, $p = 0.325$; and in autumn: $F_{10, 18} = 0.240$, $p = 0.987$). In spring, EC values showed a significant difference between the three habitats as function of depth (Figure 3.11). In winter, the ridge and ditch followed a similar pattern: EC values were higher in the ditch soil samples compared to those in ridge and slope samples that had the lowest EC values up to 10 cm. Below 10 cm, soil samples of the slope had higher EC than those in the ridge. At 30 cm, soil samples of all three habitats had the same EC (Figure 3.11a). In spring, the three habitats soil samples showed an increase

of EC values throughout the profile. The EC values were the highest in the ditch soil samples, while the ridge and slope had the most similar EC values (Figure 3.11b). In summer, the ridge and ditch followed a similar pattern. EC of the soil samples of the three habitats increased with depth, and EC values were the highest for the ditch soil samples (Figure 3.11c). In autumn, the soil samples of three habitats again followed a similar pattern, with the ditch soil samples having the highest EC values (Figure 3.11d).

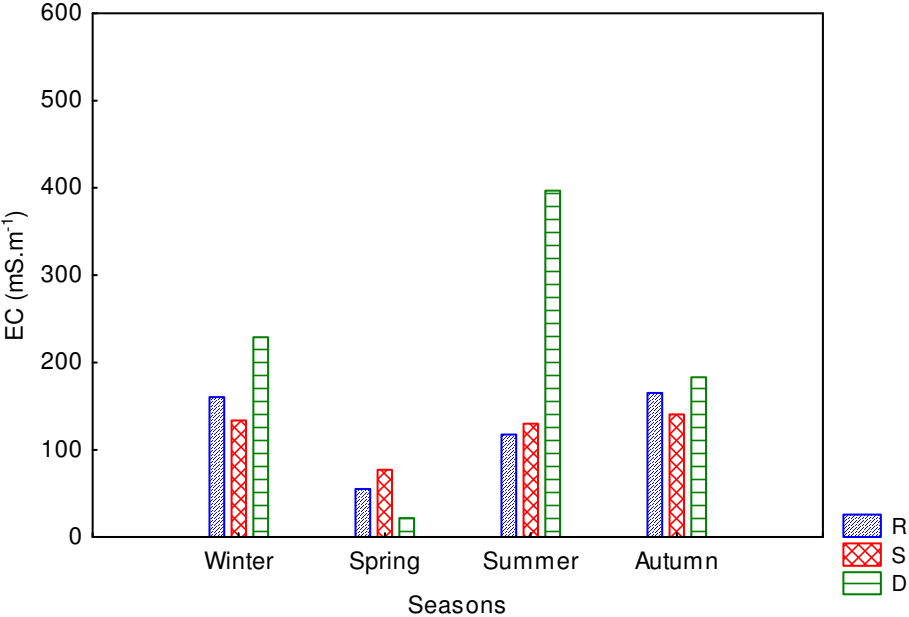


Figure 3.10 Mean EC values for 0–2 cm soil depth on ridge (R, blue), slope (S, red), and ditch (D, green) on the young field for the four seasons sampled.

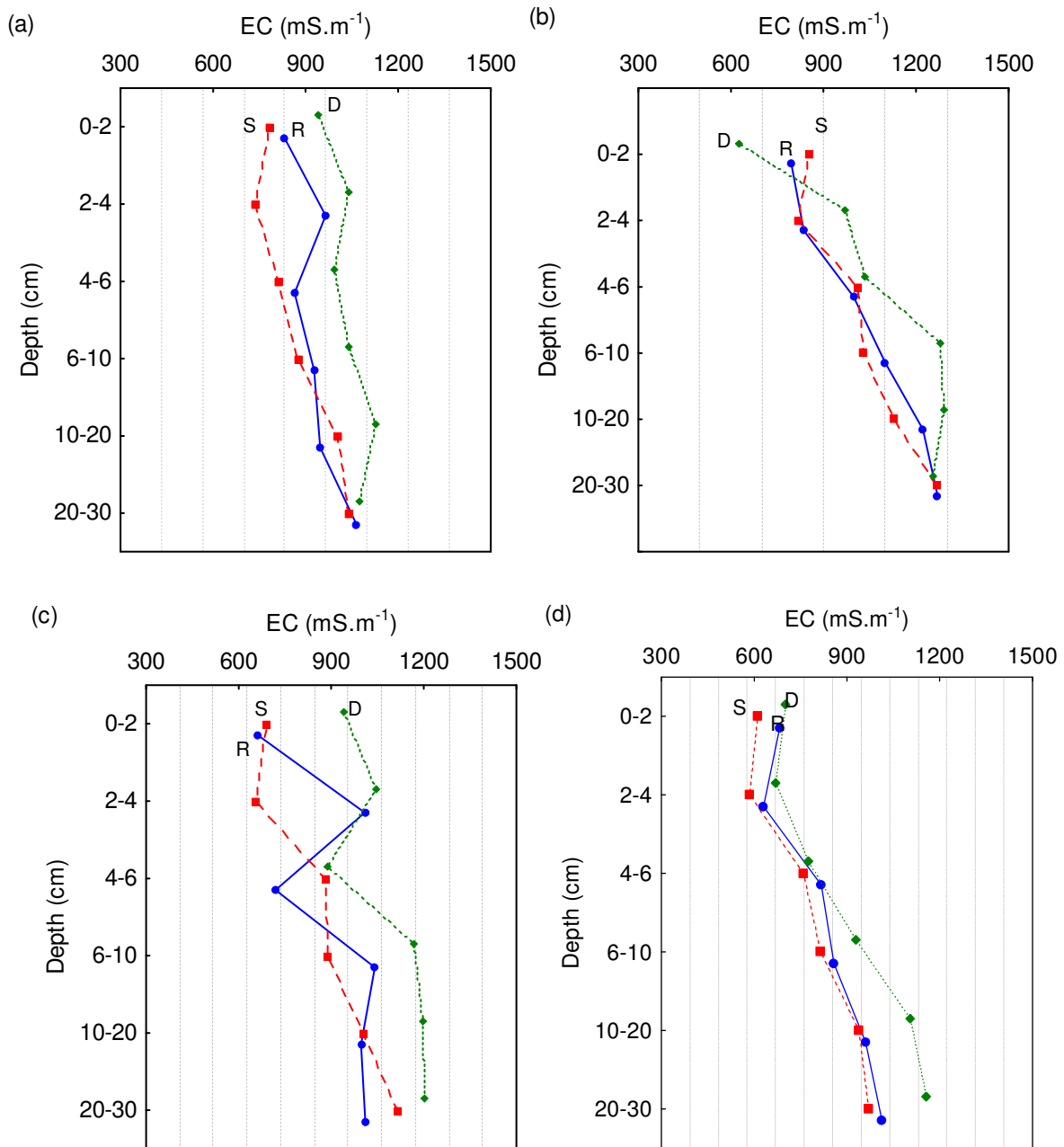


Figure 3.11 Depth profiles of mean EC for ridge (R, blue), slope (S, red) and ditch (D, green) in winter (a), spring (b), summer (c) and autumn (d) in the young field.

Soil moisture

Soil moisture did not differ between the three habitats in the interval 0-2 cm in the old field ($F_{2, 21} = 0.323$, $p = 0.727$). Soil moisture was almost similar between ridge and ditch, its value in both habitats was slightly higher than in the slope (Figure 3.12). Soil moisture did not show any significant differences between ridge, slope and ditch as function of depth on the old field ($F_{2, 126} = 0.721$, $p = 0.703$). The pattern of soil moisture showed an accumulation of water in subsoil rather than in the topsoil. Along the soil profile, the ditch and slope soil samples had higher levels of soil moisture, except around 0-2 cm the ridge had a higher soil moisture level (Figure 3.13).

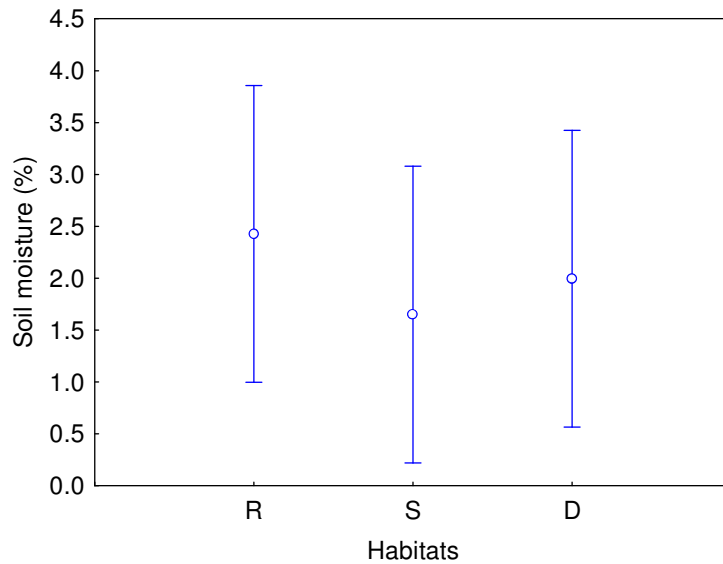


Figure 3.12 Mean annual soil moisture \pm SD for 0–2 cm soil depth for the three habitats (ridge: R, slope: S, ditch: D) in the old field.

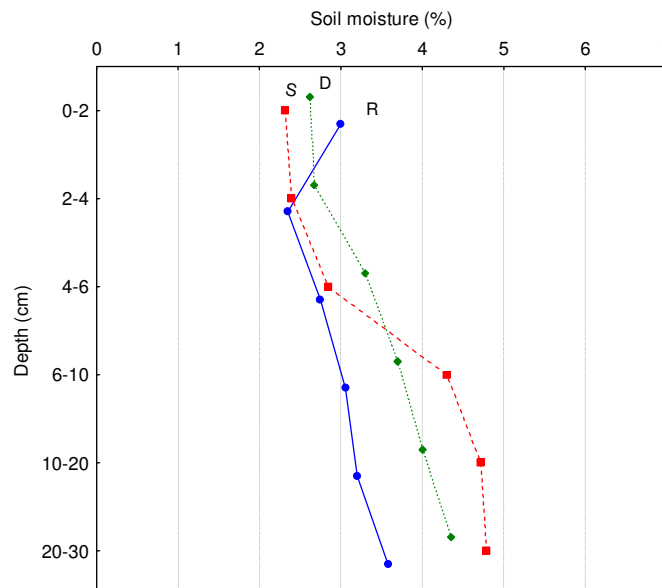


Figure 3.13 Depth profiles of mean annual soil moisture for ridge (R, blue), slope (S, red) and ditch (D, green) for the old field.

In the young field, soil moisture was not significantly different between the three habitats at 0–2 cm ($F_{2, 21} = 2.887$, $p = 0.078$). The soil moisture was slightly higher in the ditch than on the slope and on the ridge (Figure 3.14). The pattern of soil moisture did not differ between the three habitats as function of depth ($F_{10, 126} = 0.780$, $p = 0.647$). Soil moisture of all three habitats showed an increase throughout the profile, and the ditch soil samples had the highest level of soil moisture, except at 30 cm, where the three habitats had almost same level of soil moisture (Figure 3.15).

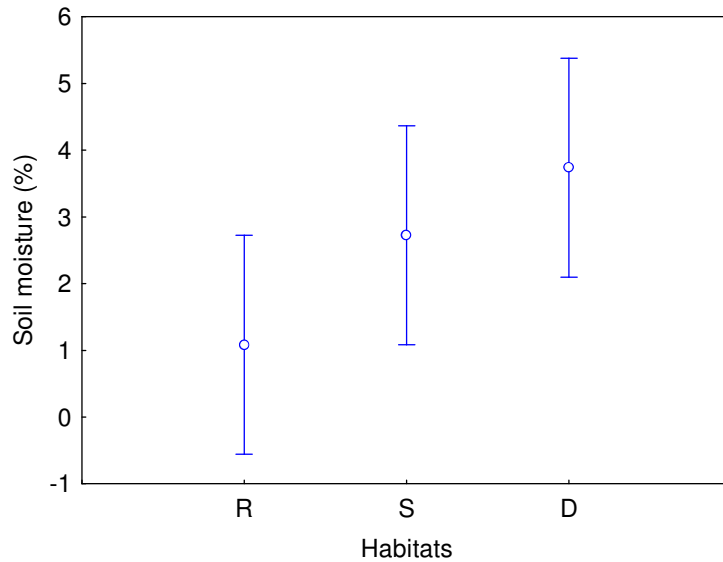


Figure 3.14 Mean annual soil moisture \pm SD for 0–2 cm soil depth for the three habitats (ridge: R, slope: S, ditch: D) in the young field.

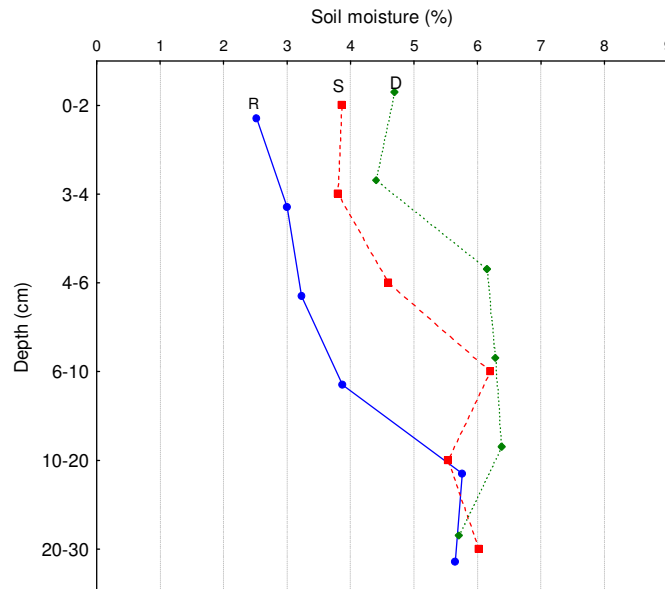


Figure 3.15 Depth profiles of mean annual soil moisture for ridge (R, blue), slope (S, red) and ditch (D, green) for the young field.

Phosphorus and total Nitrogen

In the old field, levels of available phosphorus were only significantly different between ridge, slope and ditch at 0–4 cm soil depth ($H_{2, 24} = 6.667$, $p < 0.05$). Below 4 cm, no significant differences could be observed (4–10 cm: $H_{2, 24} = 5.184$, $p = 0.074$, 10–30 cm: $H_{2, 54} = 3.065$, $p = 0.216$) (Figure 3.16a). P availability was high in all three habitats soil samples above 10 cm. Below 10 cm, the level of available P was very low. The ridge soil samples had the highest level of available P at all soil depths (Figure 3.16a).

Total nitrogen did not differ significantly between ridge, slope and ditch at any of the soil depths (0-4 cm: $H_{2, 54} = 1.316$, $p = 0.517$, 4-10 cm: $H_{2, 54} = 2.494$, $p = 0.287$, 10-30 cm: $H_{2, 54} = 2.619$, $p = 0.269$) (Figure 3.16b). On the ridge, total N was leached to the deepest soil depth sampled, while on the slope, total N was highest at intermediate depths, and in the ditch, total N was highest in the topsoil layer (0-4 cm) (Figure 3.16b).

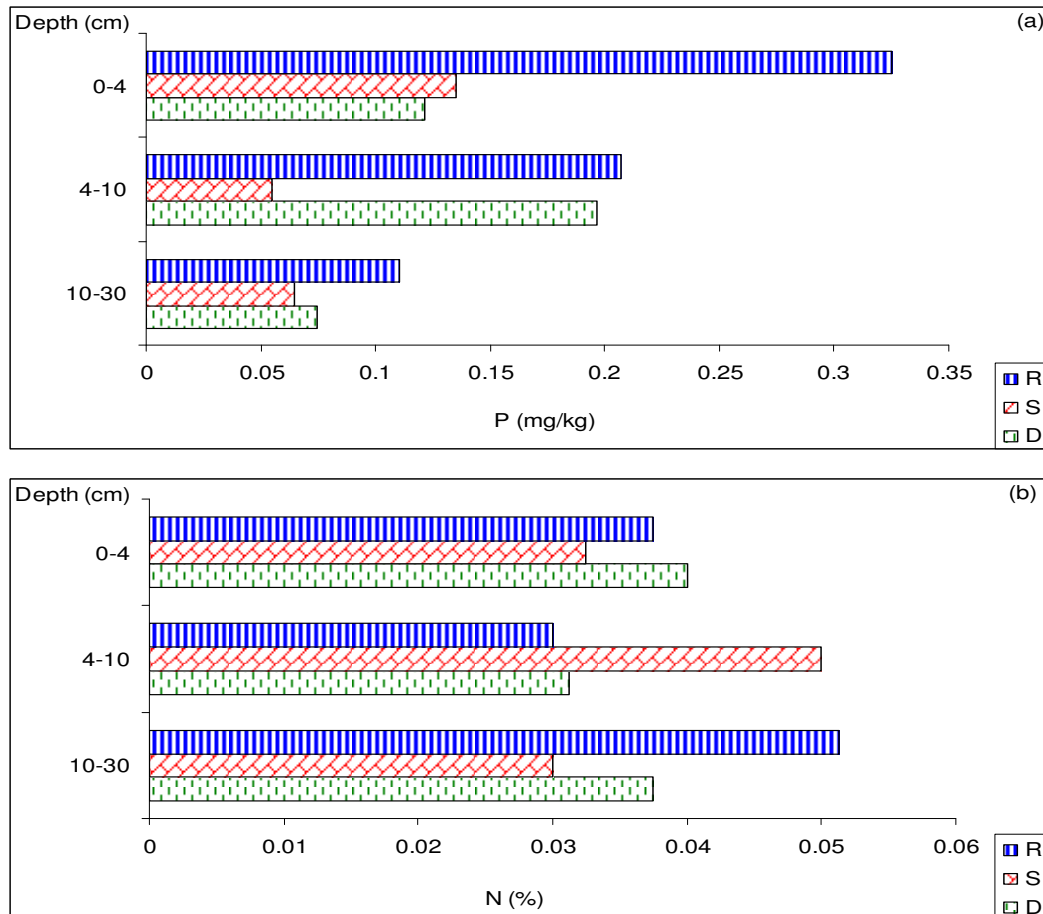


Figure 3.16 Mean available phosphorus (a) and total nitrogen (b) as function of depth in the profiles of Ridge (dark vertical), Slope (diagonal brick) and Ditch (dashed vertical) in the old field.

For the young field, available P showed only significant differences between the three habitats in the interval 4-10 cm ($H_{2, 54} = 7.038$, $p < 0.05$), whereas P availability was not significantly different between ridge, slope and ditch for the other soil depths (0-4 cm: $H_{2, 54} = 3.655$, $p = 0.160$, and 10-30 cm: $H_{2, 54} = 3,497$, $p = 0.174$). However, available P was accumulated in the soil surface in all three habitats (Figure 3.17a).

Total nitrogen was also not significantly different between the three habitats in any of the soil depths sampled (0-4 cm: $H_{2, 54} = 3.982$, $p = 0.136$, 4- 10 cm: $H_{2, 54} = 1.500$, $p = 0.472$, 10-30 cm: $H_{2, 54} = 0.298$, $p = 0.861$). Total nitrogen was leached to below 10 cm on all three habitats profile (Figure 3.17b).

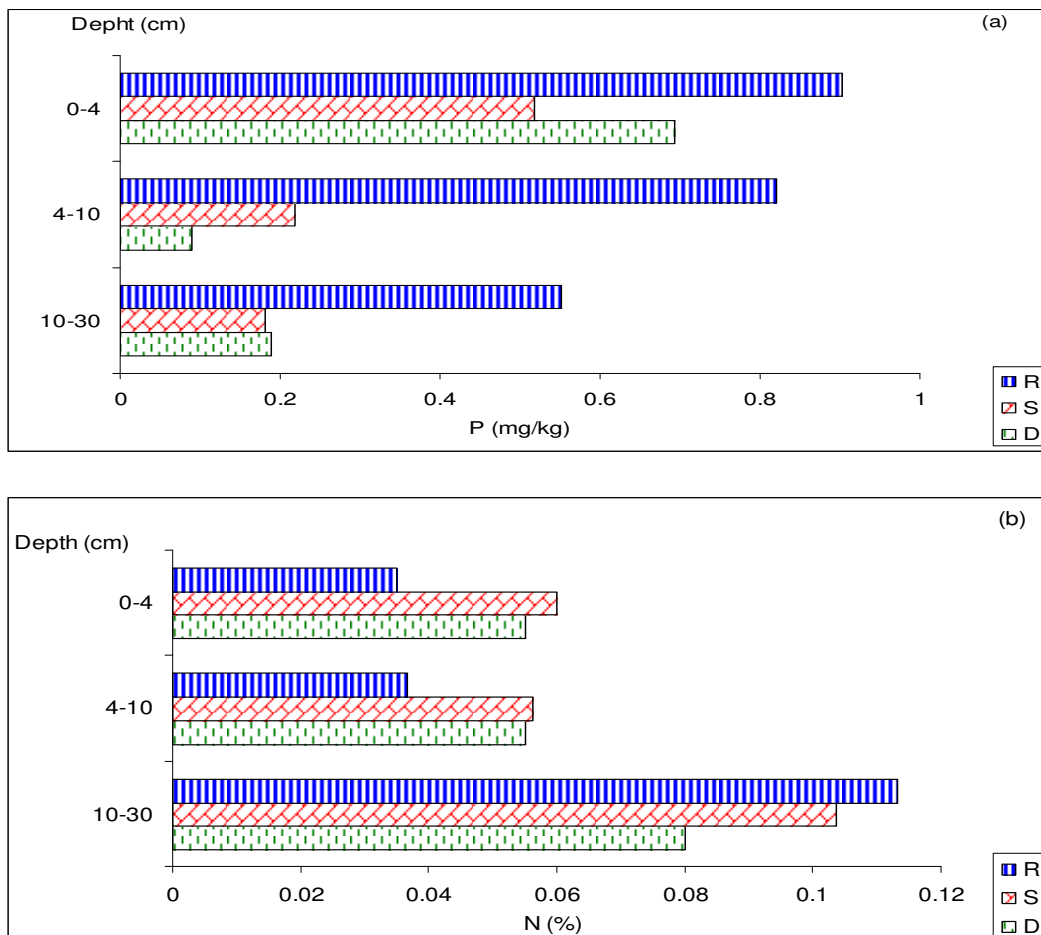


Figure 3.17 Mean available Phosphorus (a) and total Nitrogen (b) as function of depth in the profiles of Ridge (dark vertical), Slope (diagonal brick) and Ditch (dashed vertical) in the young field.

3.3.2. Soil properties of a cultivated field , a natural area, as well as old fields

The young field had the highest level of soil pH, EC and soil moisture, except for available P that was the highest in cultivated field. The cultivated field showed higher soil property levels than natural field and old field. Soil properties were most similar between natural and old field (Table 3.1).

Table 3.1 Mean± SD of pH, EC, soil moisture and P on N (natural), C (cultivated without crops cover), YF (young field), as well as OF (old field) soil at 0-10 cm at end of autumn. (N= 3 soil samples for each soil properties on natural and Cultivated; N= 2 soil samples for each soil properties on young and old fields).

	N	C	YF	OF
pH (KCl)	4.23±0.07	5.63±0.58	6.42±0.57	4.5±0.21
EC (mS.m⁻¹)	22.76±4.46	111.10±11.23	196.60±59.82	37.53±9.42
Soil moisture (%)	2.99±0.06	3.42±0.39	4.11±2.44	2.98±0.89
P(mg/kg)	0.02±0.04	1.24±0.12	0.16±0.27	0.09±0.13

3.3.3. Vegetation on the three habitats (ditch, slope and ridge)

Grasses and forbs covered the young field almost exclusively. Graminoids had the highest cover in the ditch and along the slope, while forb cover was highest on the ridge. On the old field, grasses and forbs were also present, and in addition, geophytes, shrub and hydrophytes had established. Graminoids had the highest cover in all three habitats, but cover decreased from the ridge to the ditch. Shrub and geophyte cover showed the opposite pattern, increasing in cover from the ridge to the ditch (Figure 3.18).

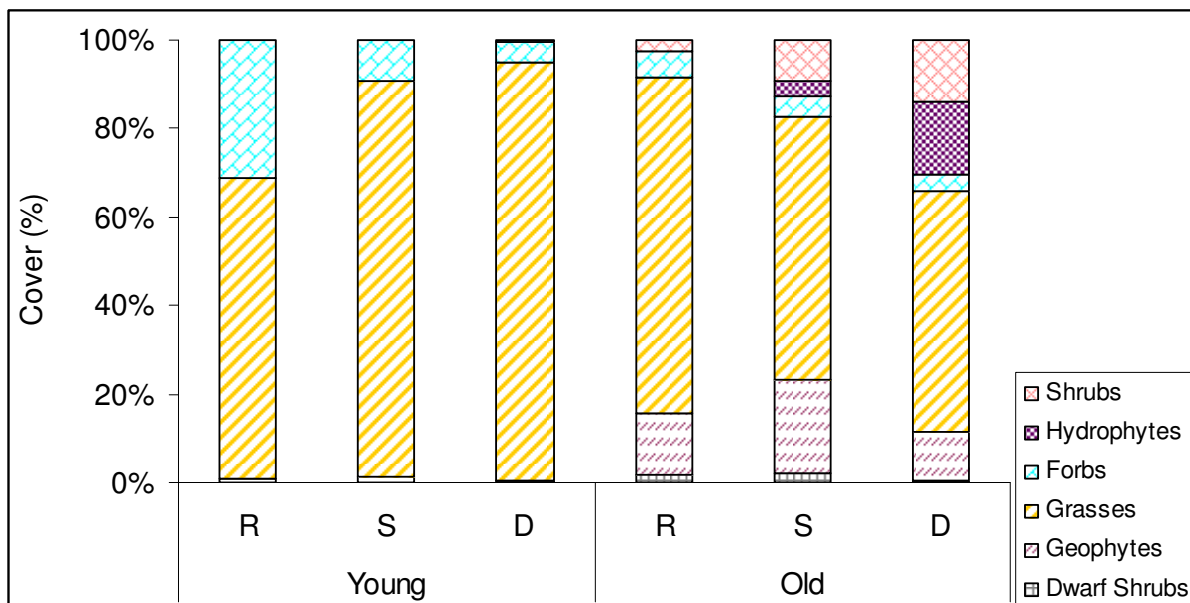


Figure 3.18 Percentage cover of grasses (wide upward diagonal), forbs (diagonal brick), shrubs (outlined diamond), hydrophytes (small checker board), dwarf shrubs (small grid), geophytes (dashed upward diagonal) and shrubs (dashed horizontal) on ridge (R), slope (S) and ditch (D) on two abandoned old fields of different ages.

The principal component analysis, relating vegetation cover to pH, EC and soil moisture, showed that both undesirable and native grasses covers were correlated with pH and EC. Undesirable forbs were negatively related to pH and EC, and slightly positive correlated to soil moisture, which was the main parameter driving geophyte and native forb cover on the young field (Figure 3.19). In contrast, pH in the old field was highly correlated with shrubs and geophytes, and slightly correlated with native grasses. Both grass types were negatively correlated with soil moisture and EC, which was positively related with native and alien forb cover (Figure 3.20). The combined data of both old fields revealed that undesirable grasses and forbs were positively correlated with pH and EC, but exotic grasses as well as *C. dactylon* showed a negative correlation with soil moisture, while exotic forbs showed a positive correlation with soil moisture. Cover of native forbs and geophytes was positively related with soil moisture. Shrubs did not correlate with any of these soil properties. However, indigenous life forms were negatively correlated with pH and EC (Figure 3.21).

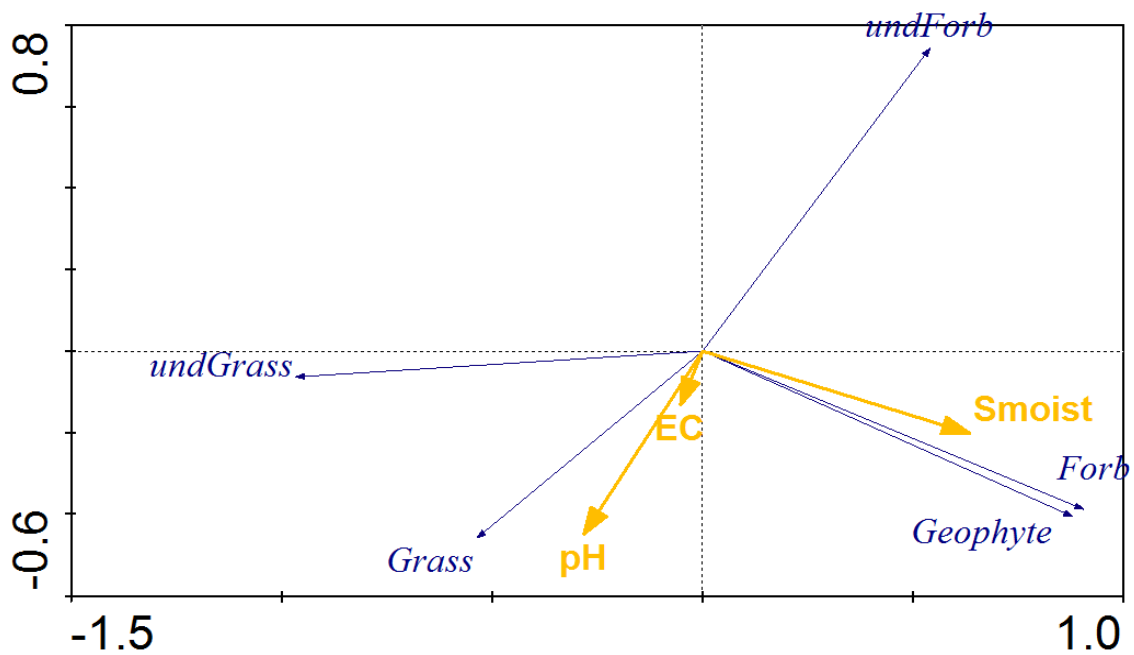


Figure 3.19 Principal component analysis results showing correlations of covers of undGrass (undesirable grasses = exotic grasses + *C. dactylon*), undForb (undesirable = exotic forbs), Grass and Forbs (native grasses and forbs), geophytes and shrubs, with environmental parameters EC, pH and Smoist (soil moisture) on young field. The blue arrows represents the increase of vegetation cover and yellow arrows the increase of soil chemistries on young field.

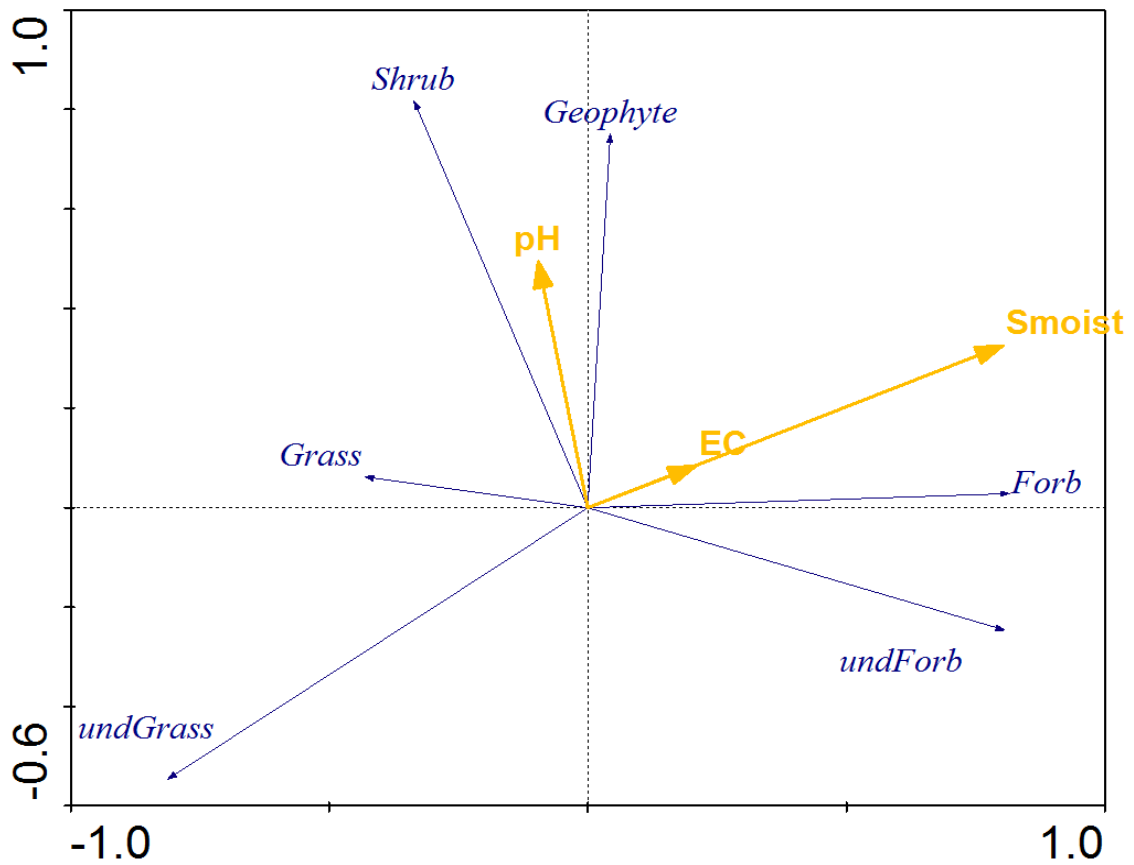


Figure 3.20 Principal component analysis results showing correlations of covers of undGrass (undesirable grasses = exotic grasses + *C. dactylon*), undForb (undesirable = exotic forbs), Grass and Forbs (native grasses and forbs), geophytes and shrubs, with environmental parameters EC, pH and Smoist (soil moisture) on old field. The blue arrows represents the increase of vegetation cover and yellow arrows the increase of soil chemistries on old field.

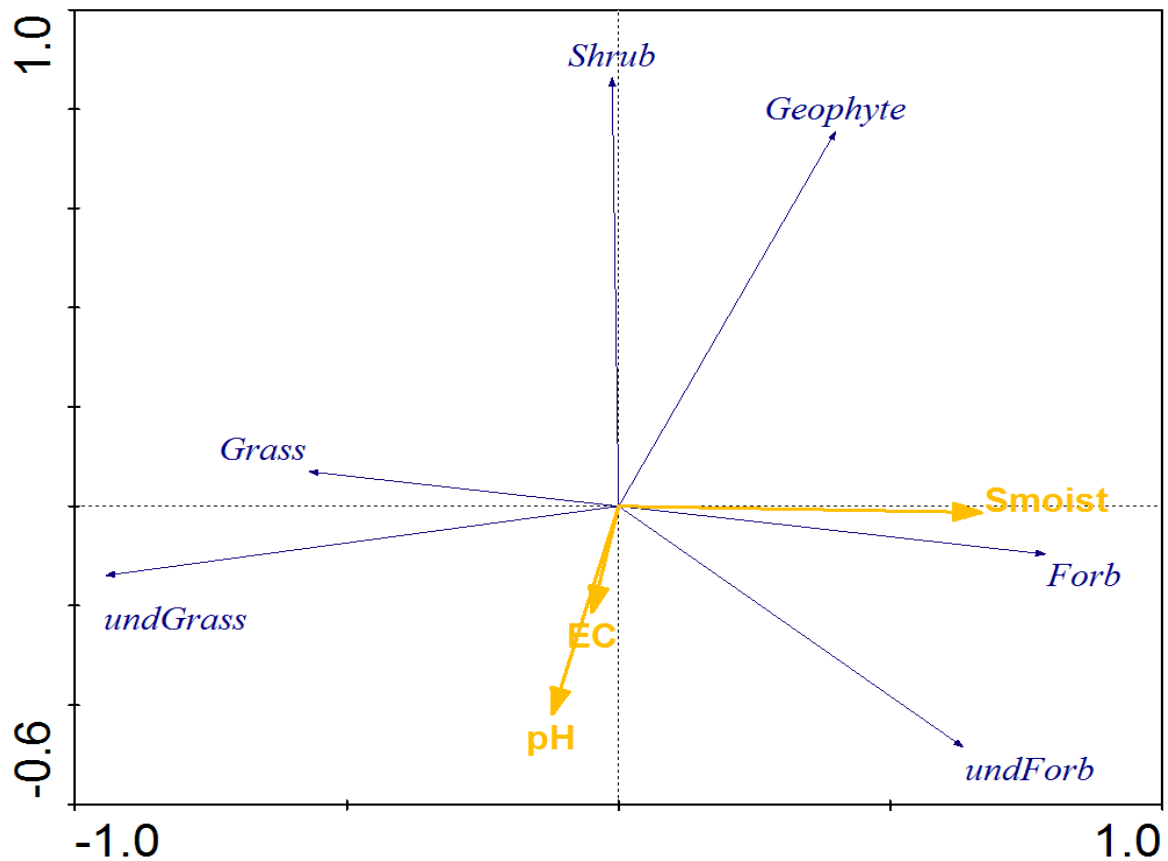


Figure 3.21 Principal component analysis results showing correlations of covers of undGrass (undesirable grasses = exotic grasses + *C. dactylon*), undForb (undesirable = exotic forbs), Grass and Forbs (native grasses and forbs), geophytes and shrubs, with environmental parameters EC, pH and Smoist (soil moisture) with both old fields. The blue arrows represents the increase of vegetation cover and yellow arrows the increase of soil chemistries on old fields.

The principal component analysis, relating vegetation cover to available P and total N, revealed that P and N were highly correlated with native and alien grass cover, and alien forbs. Native forbs and geophytes showed a negative correlation with these two soil properties on the young field (Figure 3.22). On the old field, P was correlated with native and alien grasses, and slightly correlated with alien forbs. Native vegetation cover (shrub, geophytes and forbs) was negatively related to available P on the old field. Total N did not affect any life form on this latter field (Figure 3.23). However, the combination of vegetation and available phosphorus and total nitrogen of both old fields revealed that undesirable grass and forb cover was positively correlated with available phosphorus and total nitrogen. Native grass and forb cover were only slightly related with phosphorus availability and total nitrogen. Shrubs and geophytes cover increased with decreasing values of available P and total N (Figure 3.24).

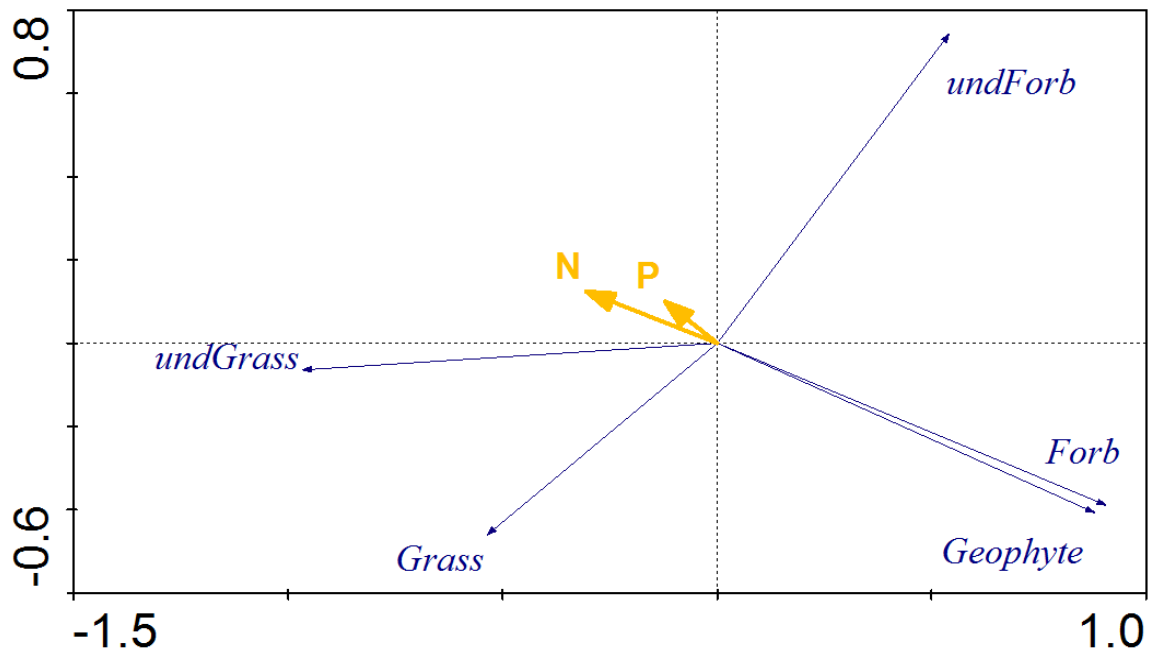


Figure 3.22 Principal component analysis results showing correlation of cover of undGrass (undesirable grasses = exotic grasses + *C. dactylon*), undForb (undesirable= exotic forbs), Grass and Forb (native grasses and forbs), geophytes and shrubs with environmental parameters P (available phosphorus) and N (total nitrogen) on the young field. The blue arrows indicate the increase of vegetation cover and yellow arrows the increase of P and N on the young field.

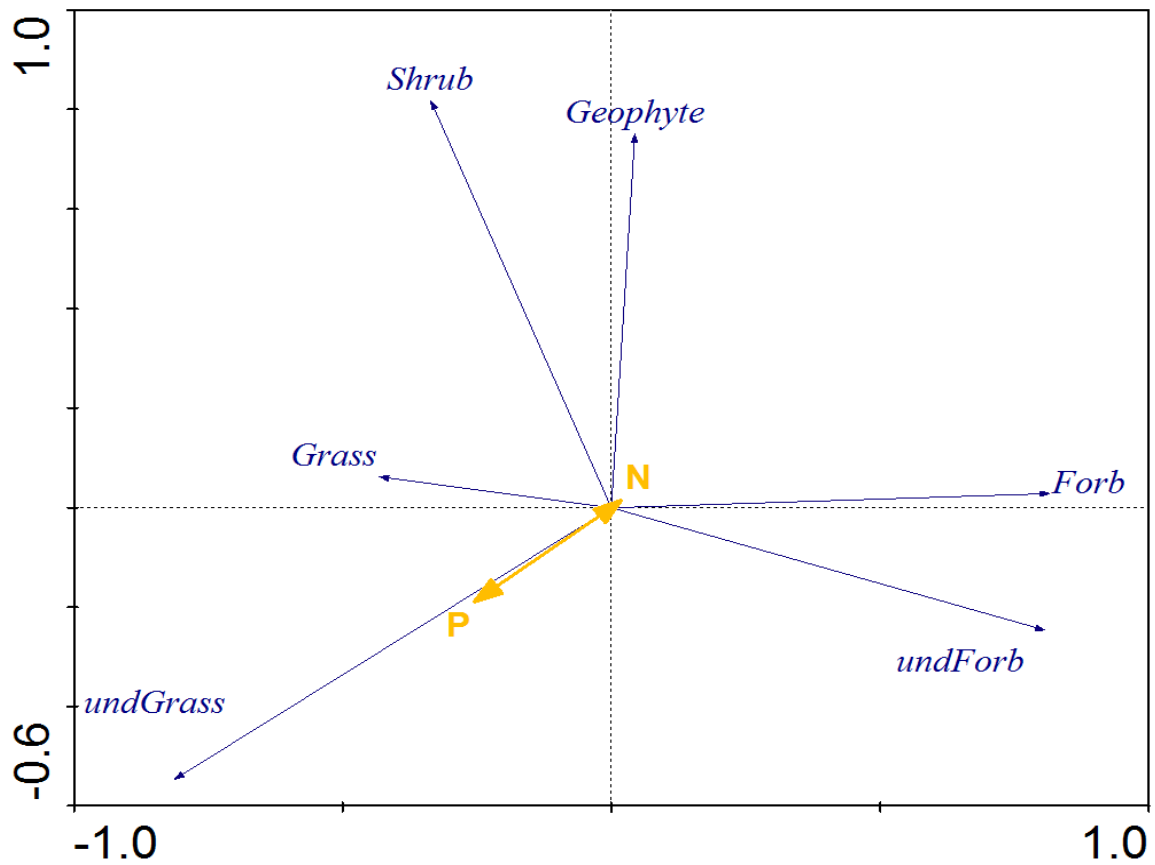


Figure 3.23 Principal component analysis results showing correlation of cover of undGrass (undesirable grasses = exotic grasses + *C. dactylon*), undForb (undesirable= exotic forbs), Grass and Forb (native grasses and forbs), geophytes and shrubs with environmental parameters P (available phosphorus) and N (total nitrogen) on the old field. The blue arrows indicate the increase of vegetation cover and yellow arrows the increase of P and N on the old field.

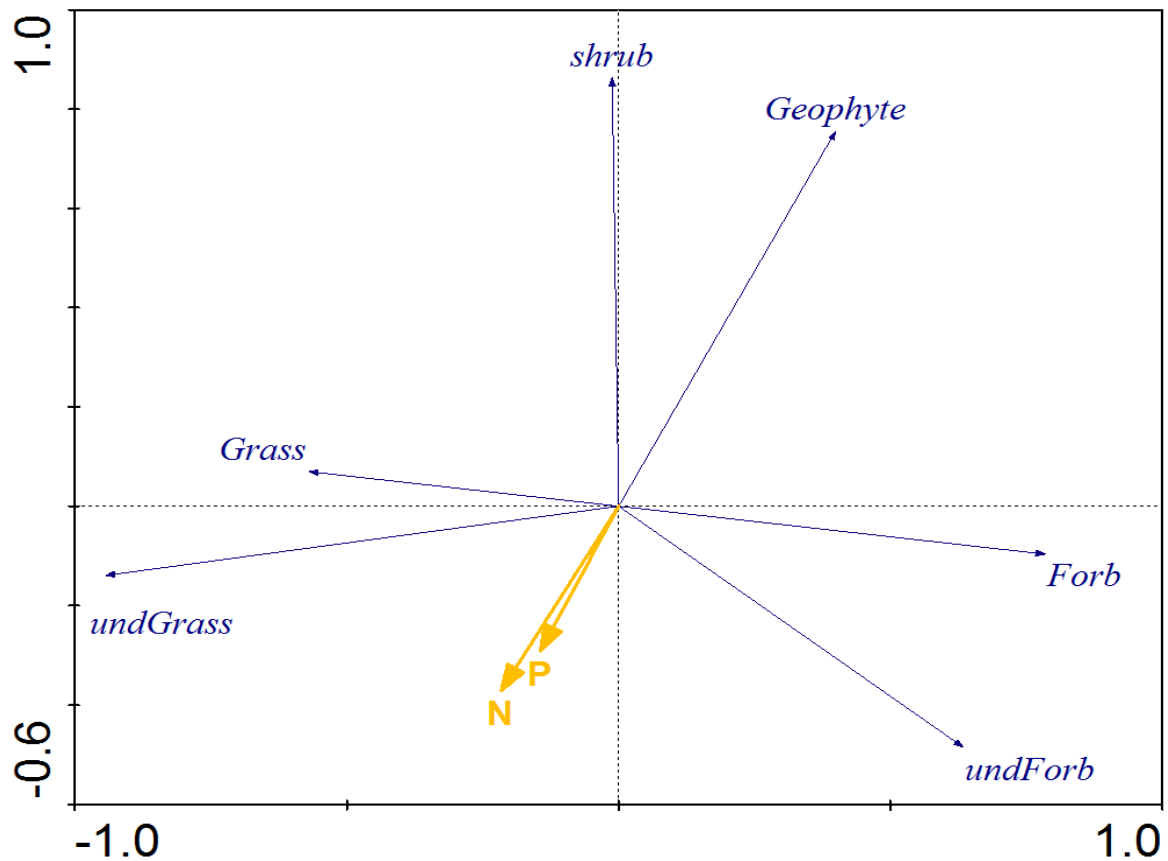


Figure 3.24 Principal component analysis results showing correlation of cover of undGrass (undesirable grasses = exotic grasses + *C. dactylon*), undForb (undesirable= exotic forbs), Grass and Forb (native grasses and forbs), geophytes and shrubs with environmental parameters P (available phosphorus) and N (total nitrogen) on both old fields. The blue arrows indicate the increase of vegetation cover and yellow arrows the increase of P and N on the old fields.

3.4. Discussion

The heritage of agricultural practices on old cultivated lands has led to changes in soil properties, compared to areas still covered by natural vegetation, resulting in changes in vegetation cover. In addition, the establishment of drainage lines has caused a different topography on the old fields, creating three distinct habitats, namely ridge, slope and ditch. The current study investigated how soil properties such as pH, electrical conductivity (EC), soil moisture, available phosphorus and total nitrogen, and their relationship with vegetation cover in each of these three habitats, differed between two abandoned cultivated fields of different ages.

3.4.1. Abandoned cultivated field of different ages

Previous restoration attempts on the older abandoned agricultural field included the reduction and removal of exotic plants, which were shown to inhibit the re-establishment of indigenous species through competition (Midoko-Iponga, 2004; Midoko-Iponga *et al.*, 2005). However,

the continued persistence of alien grass species required investigation of the role of soil properties, especially nutrient accumulation and changes in pH and EC, in preventing the establishment of indigenous species. The first step was to compare selected soil properties on two old fields of different ages. The results indicated that the values of all soil properties examined were higher in the young field than the old field. In the old lands of different ages at Elandsberg, Walton (2006) collected the soil samples on the old field when it was 15 years old and found that available P and total N were lower on that old field than on younger old fields. The decrease of available P with time could be related to consumption of that element during crop cultivation as well as leaching and erosion after abandonment (Bremner, 1965; Sims & Pierzynski, 2005; Du *et al.*, 2007; van der Wal *et al.*, 2007). In contrast, García *et al.* (2007) found that soil properties such as pH, and levels of available P and total N were similar in early (1-12 years) and mid-(12-30 years) succession in abandoned olive groves in the Catalonia region of north-eastern Spain. Previous findings in that region indicate that total N and P availability did not differ between old fields of different ages (Ruecker *et al.*, 1998). Similarly, Bonet (2004) found that soil moisture and total N had the same level in old fields of different ages in an area with high presence of shrubs and grasses in south-eastern Spain.

This difference in soil properties in old fields of different ages could be the consequence of absence of management since abandonment. Lack of management exposes soils to water and wind erosion, leading to leaching of soil nutrients. Standish *et al.* (2006) suggested that the different intensities of soil transformation, based on agricultural needs, could explain the different levels of soil nutrients in old fields of different ages. Exotic plants have been recognised to have the possibility to change the soil properties, such as total N, available P and soil moisture, due to their physiology and reproductive rate (Ehrenfeld, 2003).

Although our results, as well as other studies, have shown that soil properties such as pH, EC, P, N and soil moisture decline with time since abandonment, there is a lack of studies investigating the development of soil properties over longer time spans. Such long term studies could assist in understanding some of the drivers of secondary vegetation succession after abandonment of agricultural lands. In our study, the unfortunate lack of information on land-use history makes it difficult to find the origin of the high levels of salt on the young field, but results from a few soil samples collected in natural areas and cultivated fields indicated that agricultural activities could be the cause of high salinity in abandoned cultivated fields. However, Balasubramanian *et al.* (2004) suggested that the increase of soil salinity could be caused by a “lack of proper drainage”, pressure on soil water and utilisation of “poor-quality water”.

Soil properties of different habitats

Levels of assessed soil properties were higher on the young field and differed between the three habitats. Levels of most soil properties were higher in the old drainage line (ditch) than in the area where wheat was cultivated (ridge). However, the ridge was the area that undergoes the most changes due to ploughing, compaction, sowing of crop seeds and nutrient enrichment by the use of fertilizers. With the abandonment of the old fields, soils on the ridge were eroded and were washed into the ditch, which represented the lowest level of the old field, leading to higher salinity in the ditch compared to the ridge. Cramer *et al.* (2004) and Cramer *et al.* (2007) also defined different habitats according to topography of old fields in order to analyse the different levels of salt concentration. They also found that salinity was elevated in the “low part of the landscape” compared to the “elevated part in the field”. In their opinion, levels of salt were related to the lower quantities of water use by wheat than by native species. The increase in salinity also affects pH (Chhabra, 2005; Suarez, 2005).

In the wheatbelt of south-western Australia, destruction of native species such as eucalyptus increased salt concentration that thus increased soil salinity levels due to the conversion of perennial, deep-rooted vegetation to annual crops (Cramer & Hobbs, 2002; Cramer *et al.*, 2004; Cramer *et al.*, 2007). In other parts of the world, high soil salinity was related mostly to irrigation, as water contained elements that were favourable to the augmentation of natural salt in the soil (Pannell, 2001; Chhabra, 2005; Watt, García-Berthou & Vilar, 2007). Browning *et al.* (2006) revealed that the drainage of natural water could also increase salt load in the soil, but far slower than irrigation, where the water included some elements such as saline-sodic, which contributed to a rapid increase in soil salinity (Jalali *et al.*, 2008). Irrigation could be a contributing factor to the high levels of salt on old fields at Elandsberg, exacerbating the soil of that region which may have an already naturally elevated salt concentration (Rebelo *et al.*, 2006). This study also showed that level of soil salinity changed throughout the season. Electrical Conductivity values were the highest in the topsoil in spring and summer, whereas the subsoil showed an increase in salinity in autumn and winter. The movement of salinity between the surface and the deeper layer may have been caused by water and evaporation, which may have further permitted accumulation of salt on the soil surface (Cramer & Hobbs, 2002; Chhabra, 2005).

The trends shown by soil moisture could be associated with the lower levels of the drainage line compared to the ridge. The ditch has not changed a lot on the young field since abandonment, and the drainage line traps more water than the ridge during the rainfall period. Bonet (2004) showed that moisture levels changed with old fields' age in the Alicante

Province of South-eastern Spain, and Cerdà (1995) reported, further, that the hydrological system in an agricultural field in that region changed after abandonment. In a semi-arid region of China, levels of soil moisture diminished with the age of abandoned cultivated land (Du *et al.*, 2007). These differences in soil moisture levels could be likely associated with an increase in erosion, which increases with age of old agricultural lands.

Levels of available P are inversely patterned to the salt distribution in the three habitats. This could be linked to the discontinuation of phosphorus enrichment through fertilisation, as well as soil textures that caused its augmentation on the ridge, and its leaching contributed to its accumulation to deeper layers (Ruecker *et al.*, 1998; Standish *et al.*, 2006; van der Wal *et al.*, 2007). However, total nitrogen levels were similar in the three habitats, which may result from the absence of a litter layer containing most of the total soil organic matter (SOM). The absence of a litter layer was most likely caused by the agricultural heritage and less vegetation decomposition on old fields compared to natural vegetation (Mills & Fey, 2003; Bonet, 2004; Cerdà, 1995; Kulmatiski *et al.*, 2006a; Cortez *et al.*, 2007).

3.4.2. Development of vegetation cover

This study supports results of previous studies which show that vegetation on old agricultural fields changed with time since abandonment (Bonet & Pausas, 2004; El-Sheikh, 2005; Cortez *et al.*, 2007). Just after abandonment, exotic grasses, forbs and little natural vegetation cover the fields. Bonet (2004) suggested that this trend was related to the presence of weeds during the cultivation period. After the cessation of agricultural activities, the cover of exotic species increases rapidly. A number of studies demonstrated that grass cover decreased, and shrub cover increased with increasing age of the old field (Bonet, 2004; Bonet & Pausas, 2004; El-Sheikh, 2005). However, forb cover remained constant (García *et al.*, 2007; Römermann *et al.*, 2005). Kulmatiski (2006) indicated that the cover of exotic grasses and forbs was still higher although the cover of native species, such as shrubs, increased the cover of indigenous species. The persistence of non-native plant species cover was related to the increase of some despite others decreasing over time. These variations through secondary succession could be related to excess of nutrients, like nitrogen and phosphorus, as well as other soil properties (e.g. pH, electrical conductivity and soil moisture) (Tilman, 1987, Ruecker *et al.*, 1998; Evans *et al.*, 2001; Ehrenfeld, 2003; Mack & D'Antonio, 2003; Bonet, 2004; Hawkes *et al.*, 2005; Domènech *et al.*, 2006; Du *et al.* 2007).

3.4.3. Relationship between vegetation and soil properties

The study showed that the levels of available P and total N were under 1 mg/kg and 1% respectively on both old fields compared to cultivated land. Cramer *et al.* (2007) suggested

that nitrogen and phosphorus from fertilizer could persist for a year, during which they were consumed mostly by crops (Ruecker *et al.*, 1998; Zhao *et al.*, 2005). The fertilizer, farms feeds, agricultural remedies and stock remedies Act of South Africa (1947) defines the way in which fertilizers with nitrogen and phosphorus should be used to avoid a negative effect on the landscape. However, even during agricultural practices, crops use only about 20% to 50% of fertilizers spread in the field because high levels of those fertilizers were lost through off-take by plant leaves, erosion and leaching, and by volatilized gases such as NH₃ (Balasubramanian *et al.*, 2004; Krupnik *et al.*, 2004; Mosier, Syers & Freney, 2004; Peoples *et al.*, 2004). Therefore, these two nutrients do not seem to play the main role in the increase of cover of exotic species and the decrease native of species on old fields.

This result was supported by studies on other old fields in Mediterranean and semi-arid climate regions. Holmes (2008) could not discover any effect of nitrogen on species establishment on old agricultural lands, and Henkin *et al.* (2006) found that available phosphorus at different concentrations did not affect the shrub cover in western Galilee. For sagebrush vegetation, Bechtold & Inouye (2007) showed that shrub cover was similar between control and nitrogen addition treatments. However, Walton's (2006) work suggested that the high cover of exotic grasses and forbs could be due to high levels of available P and total N, and García *et al.* (2007) indicated that cover of grasses was higher in soils where nitrogen content and P availability were higher. The result of our study also showed that the cover of grass increased with an increase in available P and total N levels. Inversely, those nutrients are decreased by grasses (Aranibar *et al.*, 2008). Evans *et al.* (2001) reported that the cover of *Bromus sp.* reduced available nitrogen. This decrease of nitrogen is caused by rapid decomposition nitrogen-poor leaves, which then produces less N in the litter (Evans *et al.*, 2001; Ehrenfeld, 2003; Tjoelker *et al.*, 2005).

Overall, the effect of nitrogen and phosphorus on the cover of exotic and native species is debatable, as these different findings could be related to the fertilizer type and usage intensity. The impact of salinity, acidity and moisture is likely to be more important in shaping vegetation cover on the old fields investigated.

High salt concentration has been related to the slow re-establishment of indigenous plants, such as shrubs, and the high presence of exotic grasses (Cramer *et al.*, 2004; Cramer *et al.*, 2007). Koyro & Eisa (2008) indicated that the elevated salt concentration affected seeds and growth of crops. Koyro & Eisa's 2008 study showed that the high level of soil salinity was favourable for the cover of exotic grasses and forbs, and Cixin He (2005) suggested that elevated salt concentration affected plant development by disturbing the movement of all

necessary elements through water transport (Thorne & Seatz, 1955; Bower & Wilcox, 1965). Furthermore, the increase of soil salinity affects water availability for plants (Castellanos *et al.*, 2005; Cixin He, 2005; Mau-Crimmins, 2007). The high salt concentration makes pH neutral or alkaline, as observed on the 10-year-old field. A low pH favours the augmentation of soil organic matter, which plays important role in re-establishment of indigenous species on old fields (Naidu *et al.*, 1996). A number of studies have indicated that native species of Swartland Alluvium Fynbos grow on acidic soil (Steyn, 1994; Heydenrych & Littlewort, 1995; Mills, 2003; Walton, 2006), the high pH thus restricts establishment of these species.

Soil moisture plays an important role in plant growth, survival and reproduction in semi-arid and arid climate regions (Maestre *et al.*, 2001; Rodríguez-Iturbe & Portorato, 2004). In this study, the result showed an increase in shrub cover at low soil moisture levels. North *et al.* (2005) also found that shrub cover was high where soil moisture was low, but the effect of soil moisture on cover of native forbs in this study was contrary to these results. James *et al.* (2003) also recorded that shrubs were present where soil moisture levels were low. However, the persistence of exotic grasses is related to the level of water availability (Hamilton *et al.*, 1999), and it has been found that exotic grasses reduced soil moisture (Gerlach, 2004; Cione *et al.*, 2002). In this study, the PCA results showed a negative correlation between exotic grasses and soil moisture on both old fields. Our result could suggest that alien grasses used more water and therefore reduced this principal element for re-establishment and persistence of native species (Eliason & Allen, 1997, Levine *et al.*, 2003; Kulmatiski, Beard & Stark, 2006b). It has been reported that shrub cover increased while grass cover decreased in the ditch (old drainage line) (Midoko-Iponga, 2004; Krug, 2008) and a similar result was found in this study.

3.5. Conclusion

This study has shown that all soil properties examined were different between the two old agricultural lands, and differed on the old fields according to their topography. However, soil properties of the three habitats as a function of depth did not show a clear pattern. Life form cover changed, through a decrease in the cover of undesirable plants and a slight increase of cover of native species, with the topography and ages of the old fields.

Our study showed a relationship between vegetation and soil properties based on pH and EC that influence undesirable grasses and forbs. However, the presence of the exotic species themselves could affect these soil properties through a plant-soil feedback loop. Levels of pH and EC affected also the soil moisture that was related mainly to native species, but at the same time, salinity also affects pH and soil moisture. Soil salt concentration could thus be one

of principal soil properties that drive the development and cover of species on the abandoned cultivated lands assessed. Available P and total N also contributed to the persistence of alien species. Therefore, the decrease of soil salinity should be one of the main focal points during a restoration project. Browning *et al.*, (2006) suggested that gypsum application, which needs to be investigated further to determine its impact on soil properties, fynbos germination and growth, and salt leaching decreased the level of salt found in the soil. However, to prevent the increase of salinity in the soil, perennial species such as shrubs, could be maintained on the pathways, or farmers could use salt tolerant crops, such as tall wheat grass (Pannell, 2001).

3.6. References

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Chapter 4:
Restoration Trials: Vegetation response
to burning and herbicide application

4.1.Introduction

The Mediterranean climate regions across the world have been transformed mostly through agricultural practices. Degradation of natural ecosystems has facilitated the introduction of exotic species (D'Antonio & Vitousek, 1992) that self-establish and eventually dominate the vegetation on abandoned cultivated fields. This dominance is mainly due to the success of alien plant species in the competition for space, nutrients and light with native species (Midoko-Iponga, 2004; Vilà & Weiner, 2004; MacDougall & Turkington, 2005). Therefore, secondary succession is often characterised by the high presence of alien species that suppress native species (Hamilton *et al.*, 1999; Styliniski & Allen, 1999; Gerlach, 2004; Milton, 2004; MacDougall & Turkington 2005; Coleman & Levine, 2007). Undesirable species also inhibit re-establishment of natural vegetation on old agricultural lands (Meiners *et al.*, 2002; Bonet, 2004; Domènech *et al.*, 2006; Kulmatiski, 2006; Standish *et al.*, 2007).

The persistence of those undesirable species on old fields is driven by high seed production and long-lived soil seed banks (Shiponeni, 2003; Milton, 2004). For *Cynodon dactylon*, clonal spread and resprouting capacity are important characteristics that enhance its persistence and spread. In addition, this high presence of non-native species on old fields was also due to their predominance in the dung of (indigenous) large herbivores that roamed the old fields (Shiponeni, 2003; Milton, 2004; Shiponeni & Milton, 2006; Mubamu Makady, (in prep.)). Furthermore, the dominance of undesirable species can also be explained by a lower representation of native species due to limited seed dispersal that reduces their establishment in old fields (Ne'eman & Izhaki, 1996; Hamilton *et al.*, 1999; Seabloom *et al.*, 2003; Pugnaire *et al.*, 2006).

Therefore, this restoration study applied various burning and herbicide application regimes, namely seasonal burning, herbicide application and a combination of burning and herbicide application, to suppress the number and cover of undesirable species or reduce their growth. The main objectives of this study were:

1. To investigate the treatment that has the best effect on reducing the number of undesirable species and enhances the species richness of native species.
2. To determine the treatment that best reduces the cover of dominant undesirable species and improves the cover of indigenous species in old agricultural lands.
3. To compare the different effects of autumn and spring burn on reducing undesirable species and enhancing native plant species cover.

4. To determine the effectiveness of the treatments over a one year-period.

4.2. Material and Methods

4.2.1. Study site

The study was carried out on an abandoned agricultural field bordering Elandsberg Private Nature Reserve (EPNR). EPNR is situated 25 km north of Wellington and 5 km east of Hermon (19°03' E, 33°27' S) and are bordered on the east by the Elandskloof Mountains, in the west by cultivated lands (mostly wheat and canola) and in the north by (undisturbed) Swartland Shale Renosterveld and Swartland Alluvium Fynbos surrounding the Voëlvlei water treatment facility. The reserve area is a mixture of Swartland Shale Renosterveld and Swartland Alluvium Fynbos (Rebelo *et al.*, 2006). The old lands are covered mostly by *Cynodon dactylon* and alien invasive grasses *Briza maxima* and *B. minor*; *Bromus diandrus* and *B. pectinatus*; *Lolium sp*; *Poa annua* and *Vulpia myuros* (Shiponeni, 2003). For a more detailed description of EPNR, see chapter 3.

4.2.2. Study design

A completely randomised design was used to delimit 20 plots of 50 m x 50 m in a triangular 10-year-old agricultural field (abandoned in 1997). The 2 500 m² plots were separated by buffer strips of 20 m width to avoid interaction between treatments. The treatments applied, in this restoration study, are those used in previous rehabilitation or restoration experiments in this and similar vegetation types, such as autumn and spring burn (AB and SB), herbicide application (H), a combination of autumn burning and herbicide application (BH), and control (untreated) (C). Each treatment was repeated five times (Figure 4.1).



Figure 4.1 Layout of treatment plots on the old field with view of old drainage lines. Herbicide (red), autumn burn (purple), burn/herbicide combination (dark green), spring burn (light green), and control (blue).

Treatment application

Treatment application took place in two phases. The autumn burn was conducted at the end of March 2006, and thereafter the plots allocated to this treatment were divided into two plots (24 m × 50 m) separated by a 2 m gap. Five plots of 1 200 m² and five 2 500 m² plots were treated with the herbicide, which was applied two weeks after the burn and repeated once after two weeks. The herbicide used was Gallant Super, which is part of the pyridintyl-oxyphenoxy group of herbicides. It contains an acetyl-coenzyme A (acetyl-CoA) carboxylase mode of action and is a selective post-emergent herbicide used to control annual and perennial grasses.

The herbicide (2 l) was mixed with half a spray tank (120 l) of clean water to obtain an application rate of 0.57 l/ha, which is best to control invasive alien grasses like *Avena fatua*, *Bromus diandrus* and *Lolium temulentum* which mostly occur on old agricultural fields. In general, the water should have a pH between 4.5 and 5.5, to ensure the optimal efficiency of Gallant Super. Water with a pH higher or lower than 4.5-5.5 may reduce the effect of the herbicide. In order to keep the pH in the interval 4.5-5.5, we added 10 ml of Curabuff as a buffer (application rate: 100 to 400 ml per 100 ml water). The herbicide was sprayed on a day with low wind to stop the herbicide from blowing away from the plot and to achieve higher infiltration of the herbicide (spray rate 35 l/ha).

The spring burn was conducted in the middle of November 2006, when most native plant species were dying or in a dormant stage. The autumn and spring burns were started using one litre of Eco-fire, which is used to increase the heat of the fire, with 10 l half diesel-petrol mix. The fire was managed with 2 cisterns of 1 500 l and 750 l of water by wetting the ground surrounding the plots that were burned.

4.2.3. Data collection and analyses

To monitor change in vegetation cover in each treatment, we sampled the same three habitats (ditch, slope and ridge) as used for soil collection (see Figure 3.3 in chapter 3). Five permanent quadrates of 1 m × 1 m were set up randomly in each habitat (5 m² per habitats, 15 m² per plot and 75 m² per treatment). To estimate percentage cover, we used a 1 m² grid with 100 subsections equal to 1% each (Milligan *et al.*, 2003). Species with a cover of less than 1% were assigned a percentage cover of 0.5 % (McGingley & Tilman, 1993). Seasonal surveys took place from August 2006 to October 2007, with August (winter) as period 1 (3 months after treatment), November (spring) as period 2 (6 month after treatment), February-March (summer) as period 3 (9 months after treatment) and May-June (autumn) as period 4 (12 months after treatment). The survey of the spring burn plots was conducted from February 2007 to October 2007, with February- March (summer) as period 1, May-June (autumn) (period 2), August (winter) (period 3) and October (spring) as period 4.

Species were identified using previous works (Shiponeni, 2003; Krug, 2004; Walton, 2006) and grass identification guides (Grabandt, 1985; Botha, 2001; Bromilow, 2001), with the help of Mr. Raphael Kongor and Prof. Sue Milton (Department of Conservation Ecology and Entomology). However, not all collected plants could be identified because of very small leaf stage and the absence of flowers. Life form and life cycle were ascribed to species identified according to Germishuizen & Meyer (2003).

Statistical data analyses were conducted in STATISTICA 7 & 8 (StatSoft Inc. 2006; 2007). The individual species were classified into two groups: indigenous (forbs, geophytes and grasses) and undesirable species (exotic forbs and grasses including *C. dactylon*), following Midoko-Iponga (2004). As the same individual plant species did not occur over the four periods, ANOVA analysis was done with indigenous and undesirable plant species. Counting the number of species in each replicated treatment and considering each 1 m² quadrat separately determined the species richness of indigenous and undesirable species. Differences in species richness and cover of native and undesirable species were examined using one-way ANOVA for each period (3, 6, 9 and 12 months) after treatment application, with Bonferroni tests as post-hoc tests. P-values for each period were multiplied by 4 to correct for the four periods analysed (Bonferroni, 1936). For the comparison of the effects of three treatments (autumn burn, herbicide application, combination of autumn burn and herbicide application) and untreated plots on species richness and percentage cover of indigenous and undesirable plants over the four periods, a factorial ANOVA was used, using Bonferroni Post-hoc tests. For all parametric analyses, the non-normality of residuals was corrected for using the bootstrap method, which re-samples the present samples' residuals (Efron & Tibshirani, 1993).

Total cover of indigenous and non-native life forms (forbs, geophytes and grasses), as well *C. dactylon*, was determined for each replicated treatment with five 1 m² quadrats in each habitat. The effects of the four treatments on total cover were analysed using Kruskal-Wallis ANOVA, as the data were not normally distributed. Mann-Whitney U-test was used as post-hoc test. The total percentage cover was determined by calculating the sum of the percentage cover for each replicated treatment. The percentage cover of vegetation was relative at the 1 m² quadrat, but as not all species could be identified, the analysis was done with absolute cover values for individual species. As species cover overlapped, total cover exceeded 100% (Ne'eman & Izhaki, 1996). The average cover of non-native and native species was calculated. The species with the highest means were analysed using Kruskal-Wallis ANOVA to test for significant differences between the four treatments and using the Mann-Whitney U-test post-hoc test.

For comparison between autumn and spring burns, the total cover of native and non-native life forms was determined. As these data were not normally distributed, a Kruskal-Wallis ANOVA was used to analyse the difference between total cover after treatment application. The analysis was done with autumn burn and spring burn separately due to the overlapping of season issues.

4.3.Results

4.3.1. Treatments effects: in indigenous species

Species richness of indigenous plants was not significantly different between treatments three months ($F_{3, 96} = 0.384$, $p = 3.057$), 9 months ($F_{3, 50} = 2.622$, $p = 0.243$) and 12 months ($F_{3, 86} = 1.549$, $p = 0.830$) after the treatments were applied (Figure 4.2). Only 6 months after application, a significant difference between treatments could be observed ($F_{3, 93} = 19.834$, $p < 0.0001$). The species richness of native species was the lowest in the autumn burn plots and then treated with herbicide (BH), while the untreated plots had the highest richness of indigenous species.

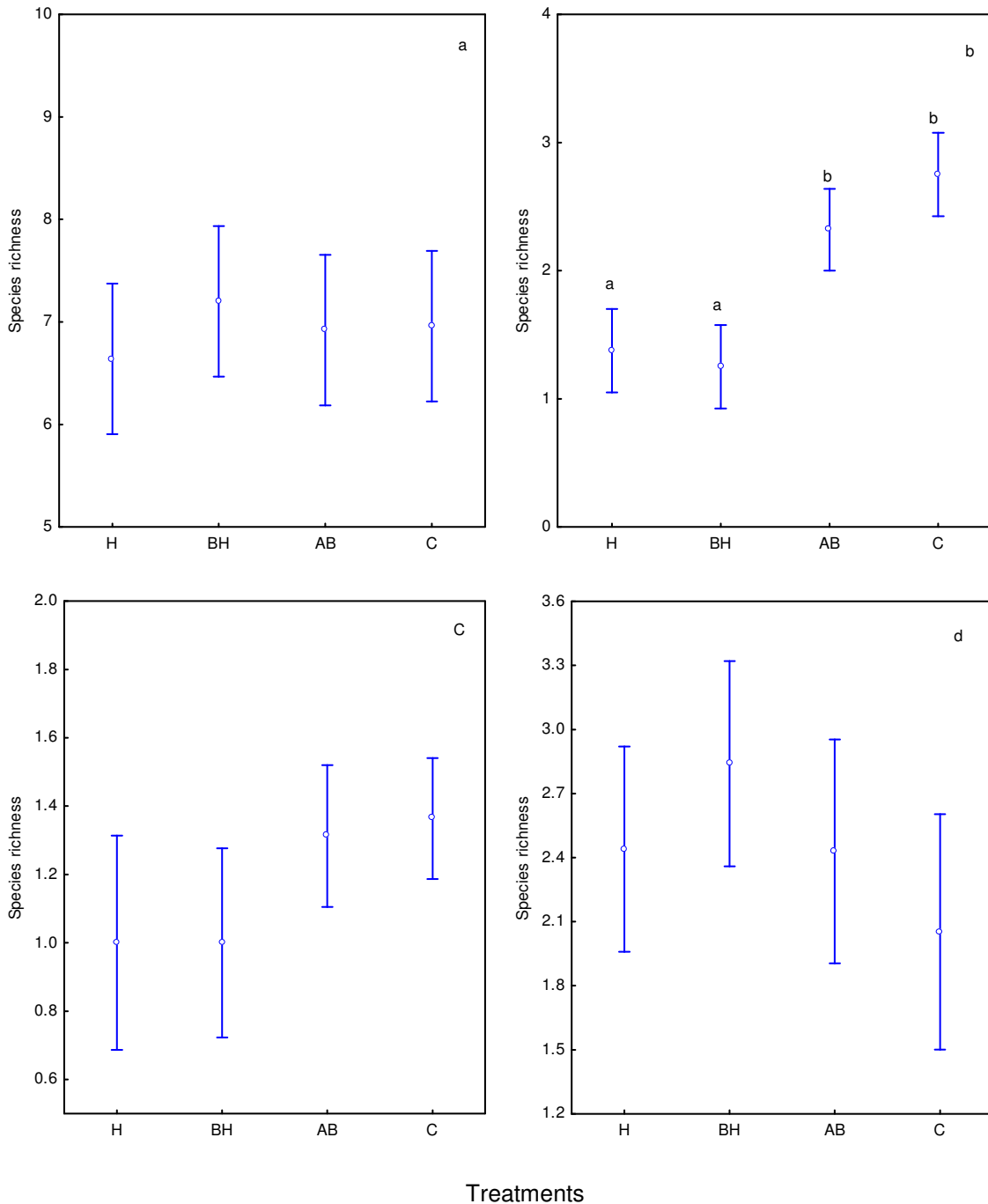


Figure 4.2 Mean species richness \pm SD of native plants 3 months (a), 6 months (b), 9 months (c) and 12 months (d) after herbicide (H), combination burn-herbicide (BH), autumn burn (AB) and control (C) treatments. Lower case letters indicate significant difference between treatments.

Results of the factorial ANOVA indicate a significant effect of the treatments on the number of native species ($F_{9, 325} = 2.761$, $p < 0.05$). Autumn burn, herbicide application and a combination of autumn burning and herbicide application did not affect the number of native species at three months, nine and twelve months after treatment application, but BH and H had a negative effect on native species richness, with the lowest number of native species observed after 6 months (Figure 4.3).

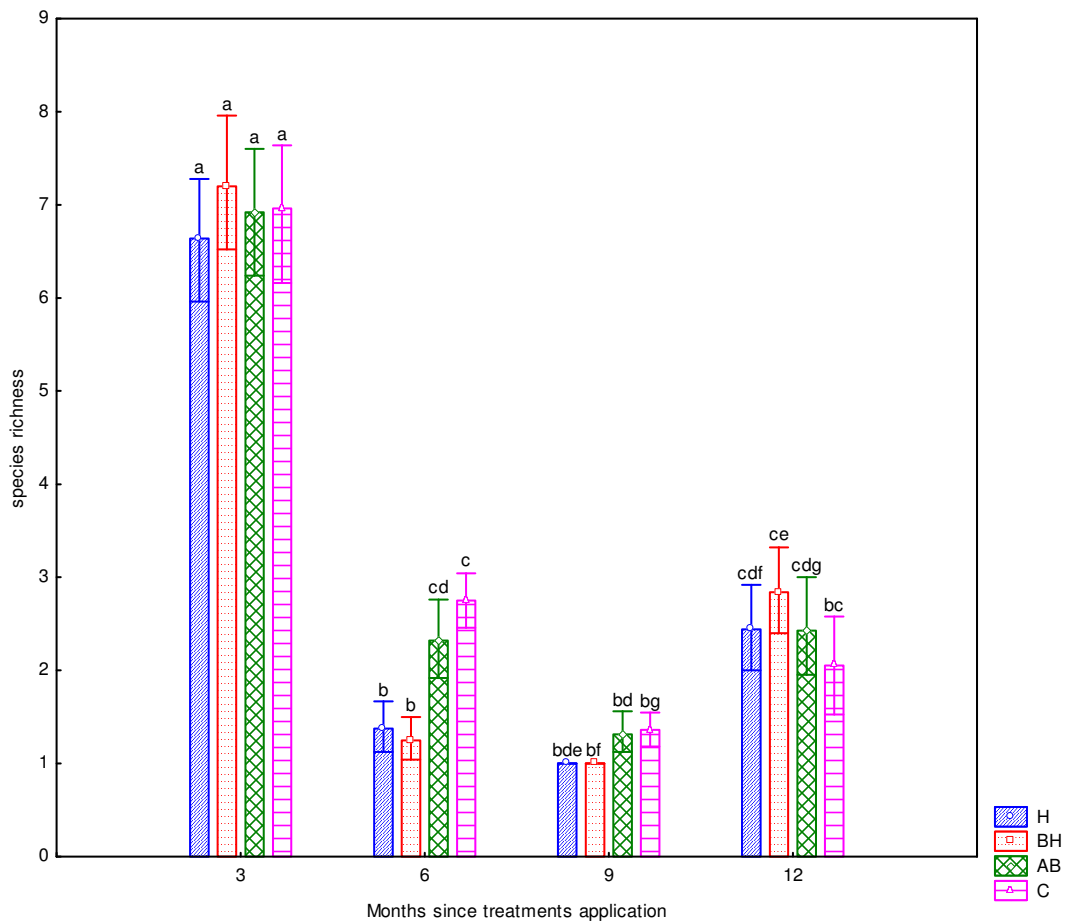


Figure 4.3 Mean species richness \pm SD of indigenous plants at 3, 6, 9 and 12 months after treatment application (autumn burn: AB, burn and herbicide: BH, herbicide: H, control: C. Lowercase letters indicate significant differences (Bonferroni Post-hoc tests) after bootstrap transformation.

Percentage cover of native species was not significantly different between treatments at any stage after application (3 months, $F_{3, 1096} = 2.702$, $p = 0.177$; 6 months, $F_{3, 303} = 1.911$, $p = 0.510$; 9 months $F_{3, 90} = 2.699$, $p = 0.201$ and 12 months, $F_{3, 281} = 2.444$, $p = 0.257$) (Figure 4.4). Based on a factorial ANOVA, the effectiveness of treatments on native species cover was not significantly different between treatments throughout the four periods ($F_{9, 1770} = 1.409$, $p = 0.178$). However, cover of native species was higher on herbicide, burn-herbicide and autumn burn plots than on control plots after 3 and 12 months, whereas native species cover was higher on autumn burn and control plots after 6 months and on herbicide, autumn burn and control plots after 9 months. Overall, the cover of indigenous species was the highest in all four treatments three months after treatment application (Figure 4.5).

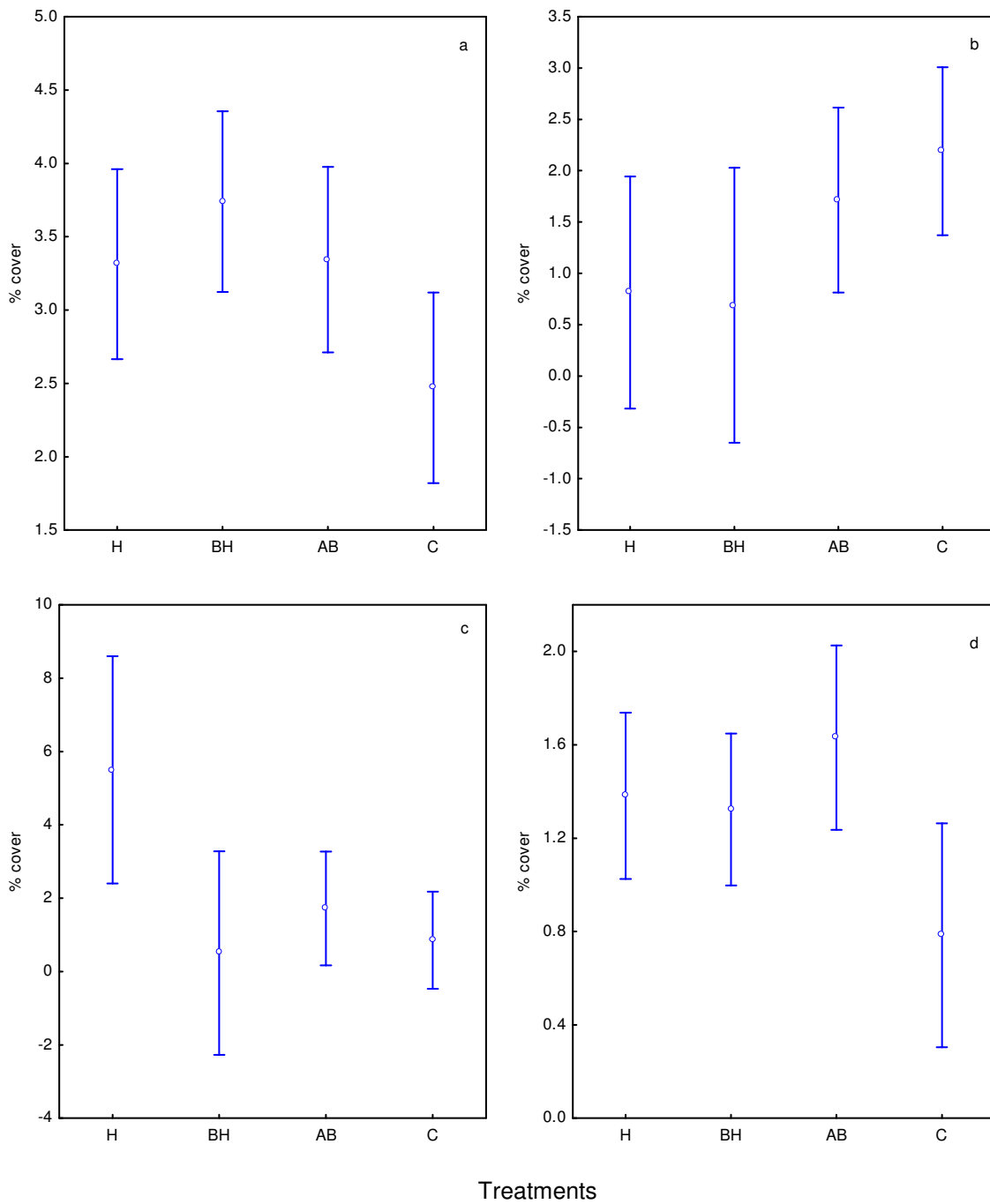


Figure 4.4 Mean percentage cover \pm SD of indigenous species three months (a), 6 months (b), 9 months (c) and 12 months (d) after treatment: herbicide (H), burn and herbicide (BH), autumn burn (AB) as well as control (C).

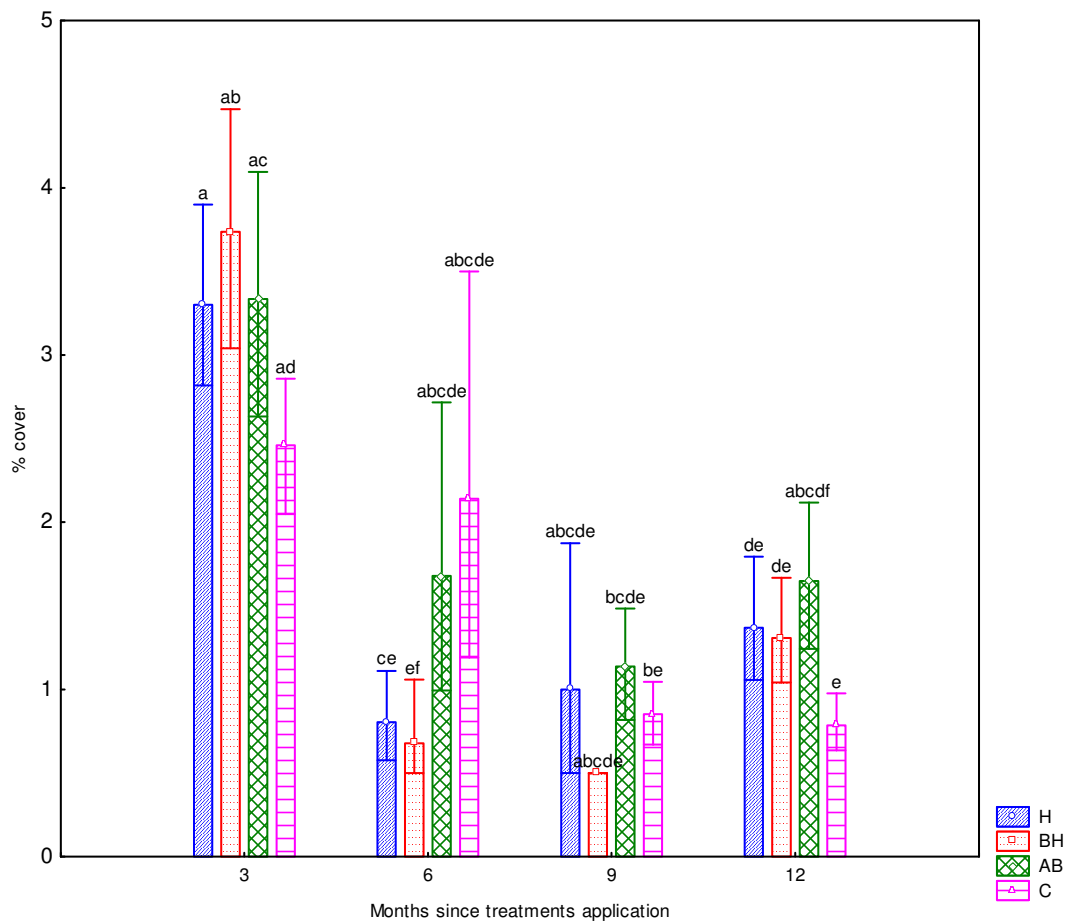


Figure 4.5 Mean percentage plant cover \pm SD of indigenous plants at 3, 6, 9 and 12 months after treatment application (autumn burn: AB, burn and herbicide: BH, herbicide: H, control: C). Lowercase letters indicate significant differences (Bonferroni Post-hoc tests) after bootstrap transformation.

Percentage cover of indigenous grasses and geophytes was significantly different between treatments ($H_{3, 341} = 15.017$, $p < 0.001$, $H_{3, 341} = 11.034$, $p < 0.05$ respectively). The cover of native grasses was the highest in untreated plots, whereas geophyte cover was slightly higher on autumn burn treatments than on burn-herbicide plots (Table 4.1). However, the percentage cover of native forbs was found not to be significantly different between treatments ($H_{3, 341} = 4.155$, $p = 0.245$) at the end of the study (Table 4.1).

Comparing the effects of all four treatments on individual indigenous species with the highest percentage cover at the end of the study, native forbs such as *Drosera trinervia* showed significant differences in cover between treatments ($H_{3, 400} = 11.790$, $p < 0.05$). Cover of grasses, such as *Pentaschistis curvifolia* ($H_{3, 400} = 23.700$, $p < 0.0001$) and *Pentaschistis pallida* ($H_{3, 400} = 59.120$, $p < 0.0001$) was also significantly different between treatments. These three native species had the highest cover in the control plots (Table 4.2). Geophytes such as *Ornithogalum thyrsoides* ($H_{3, 400} = 9.160$, $p < 0.05$) and *Oxalis pes-caprae* ($H_{3, 400} = 9.130$, $p < 0.05$) showed significant differences between treatments, and had the highest cover in autumn

burn treatments (Table 4.2). The percentage cover of *Geissorhiza confusa* ($H_{3, 400} = 11.750$, $p < 0.05$) and *Satyrium erectum* ($H_{3, 400} = 8.000$, $p < 0.05$) was also significantly different between treatments at the end of the study. The percentage cover of those two geophytes was the highest on burn-herbicide plots (Table 4.2).

Table 4.1 Mean percentage cover \pm SD of indigenous life forms on autumn burn (AB), burn-herbicide (BH), herbicide (H) and control (C) plots. Differences were tested for significance with Kruskal-Wallis ANOVA (N= 341, DF= 3, *= $p < 0.05$; **= $p < 0.001$, NS=non significant), using Mann Whitney U-tests as post-hoc test. Lowercase letters indicate significant difference between treatments. Most effective treatments at the end of the study after 12 months are underlined.

Life form	Treatments				H	p-value	sign
	H	BH	AB	C			
Grasses	1.12 \pm 4.58 ^{ab}	0.37 \pm 1.10 ^b	2.44 \pm 6.48 ^{ad}	<u>3.07\pm8.25^d</u>	15.017	0.001	**
Forbs	9.01 \pm 14.90	11.22 \pm 18.24	7.58 \pm 14.28	6.42 \pm 11.53	4.155	0.245	ns
Geophytes	2.75 \pm 4.17 ^a	3.37 \pm 4.88 ^a	<u>3.91\pm9.30^{ad}</u>	1.15 \pm 1.56 ^d	11.034	0.011	*

Table 4.2 Mean percentage cover \pm SD of indigenous species on autumn burn (AB), burn-herbicide (BH), herbicide (H) and control (C) plots. Differences were tested for significance with Kruskal-Wallis ANOVA (N= 400, DF= 3, *= $p < 0.05$; **= $p < 0.001$, ***= $p < 0.0001$, NS=non significant), using Mann Whitney U-tests as post-hoc test. Lowercase letters indicate significant difference between treatments. Most effective treatments at the end of the study after 12 months are underlined.

Life form	Treatments				H	p-value	sign
	H	BH	AB	C			
Grasses							
<i>Imperata cylindrica</i>	0.11 \pm 0.76	0.09 \pm 0.80	0.10 \pm 0.37	0.06 \pm 0.25	7.750	0.055	ns
<i>Pentaschistis curvifolia</i>	0.01 \pm 0.11 ^a	0.01 \pm 0.11 ^a	0.08 \pm 0.24 ^{be}	<u>0.16\pm0.46^{de}</u>	23.700	0.000	***
<i>Pentaschistis pallida</i>	0.01 \pm 0.08 ^a	0.02 \pm 0.09 ^a	1.49 \pm 5.60 ^{be}	<u>2.18\pm7.42^{de}</u>	59.120	0.000	***
<i>Tribolium hispidum</i>	0.77 \pm 4.04	0.19 \pm 0.38	0.45 \pm 2.03	0.36 \pm 0.79	1.690	0.638	ns
Forbs							
<i>Cotula coronopifolia</i>	0.25 \pm 1.62	0.02 \pm 0.20	0.23 \pm 2.00	0.28 \pm 1.32	6.240	0.100	ns
<i>Cotula turbinata</i>	3.22 \pm 7.21	6.65 \pm 14.05	4.11 \pm 9.28	2.52 \pm 5.14	1.900	0.593	ns
<i>Dischisma capitatum</i>	0.20 \pm 1.09	0.32 \pm 1.02	0.16 \pm 0.62	0.18 \pm 0.58	4.210	0.239	ns
<i>Drosera trinervia</i>	0.09 \pm 0.51 ^a	0.04 \pm 0.24 ^{ad}	0.10 \pm 0.67 ^{bd}	<u>0.34\pm1.12^c</u>	11.790	0.008	**
<i>Felicia bergeriana</i>	3.38 \pm 7.27	1.84 \pm 5.75	1.77 \pm 4.44	2.33 \pm 5.16	0.730	0.865	ns
Geophytes							
<i>Cyanella hyacinthoides</i>	0.20 \pm 0.41	0.40 \pm 0.79	0.33 \pm 0.74	0.31 \pm 0.65	1.530	0.674	ns
<i>Geissorhiza confusa</i>	0.12 \pm 0.27 ^{ac}	<u>0.34\pm0.90^{ad}</u>	0.06 \pm 0.20 ^b	0.16 \pm 0.38 ^{cd}	11.750	0.010	**
<i>Ornithogalum thyrsoides</i>	0.08 \pm 0.18 ^a	0.03 \pm 0.11 ^{be}	<u>0.11\pm0.33^{ad}</u>	0.05 \pm 0.24 ^{cde}	9.160	0.027	*
<i>Oxalis commutata</i>	0.59 \pm 3.04	0.61 \pm 3.03	1.61 \pm 8.08	0.12 \pm 0.52	1.070	0.783	ns
<i>Oxalis pes-caprae</i>	0.19 \pm 1.15 ^{ab}	0.04 \pm 0.24 ^{bde}	<u>0.24\pm0.91^{ad}</u>	0.01 \pm 0.10 ^{ce}	9.130	0.027	*
<i>Oxalis purpurea</i>	0.14 \pm 0.93	0.02 \pm 0.12	0.09 \pm 0.57		4.780	0.188	ns
<i>Oxalis versicolor</i>	0.59 \pm 1.59	0.84 \pm 2.23	0.69 \pm 2.28	0.09 \pm 0.40	6.610	0.852	ns
<i>Satyrrium erectum</i>	0.26 \pm 0.56 ^{ac}	<u>0.46\pm0.84^{be}</u>	0.24 \pm 0.39 ^{ade}	0.27 \pm 0.68 ^{cd}	8.000	0.046	*
Dwarf shrub							
<i>Helichrysum asperum</i>	0.36 \pm 1.37	0.14 \pm 0.35	0.37 \pm 1.07	0.06 \pm 0.20	5.550	0.130	ns

4.3.2. Treatments effects: Exotic species and *C. dactylon*

The number of undesirable species (exotic species and *C. dactylon*) was significantly different between treatments after 3 months ($F_{3, 96} = 7.084$, $p < 0.0001$), 6 months ($F_{3, 96} = 28.365$, $p < 0.0001$) and 9 months ($F_{3, 95} = 9.208$, $p < 0.0001$) (Figure 4.6). The lowest number of species was found in the autumn burn and herbicide treatments, and herbicide only treatments, except for the first period, where species richness was lowest in the combined autumn burning and herbicide application. Based on factorial ANOVA results, the impact of those four treatments on species richness of undesirable plants was also significantly different throughout the four periods (after 3 months, 6 months, 9 months and 12 months) ($F_{9, 383} = 9.058$, $p < 0.0001$). The species richness of undesirable species was lowest in the burn/herbicide treatment and high in burn, herbicide and control plots after three months, while the number of species was similar in herbicide and burn/herbicide plots, and lower than in autumn burn and control plots after 6 and 9 months. However, the species richness of undesirable species was similar in all four treatments after 12 months (Figure 4.7).

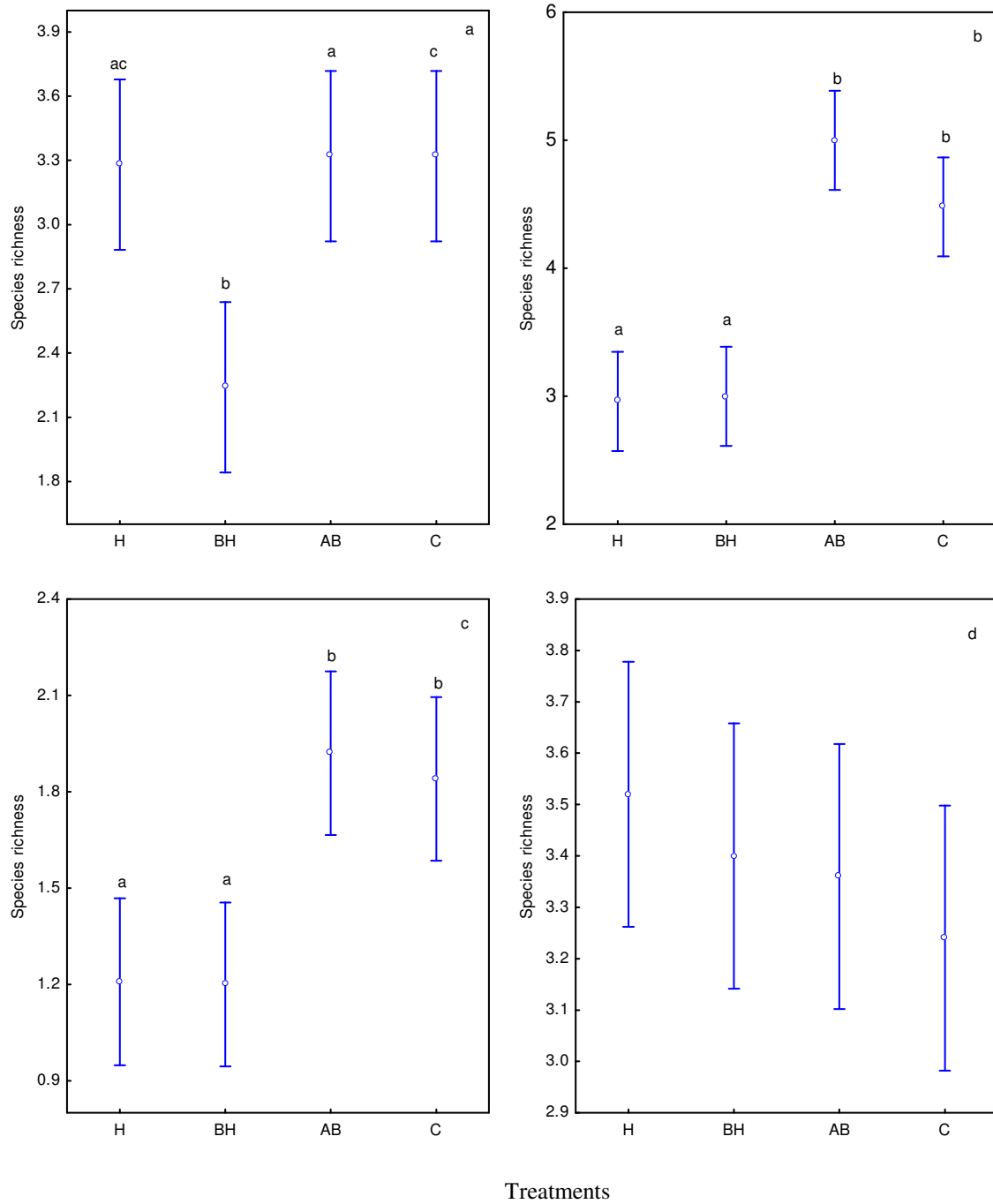


Figure 4.6 Mean species richness \pm SD of undesirable plants 3 months (a), 6 months (b), 9 months (c) and 12 months (d) after herbicide (H), combination autumn burn-herbicide (BH), autumn burn (AB) and control (C) treatments. Lowercase letters indicate significant difference between treatments.

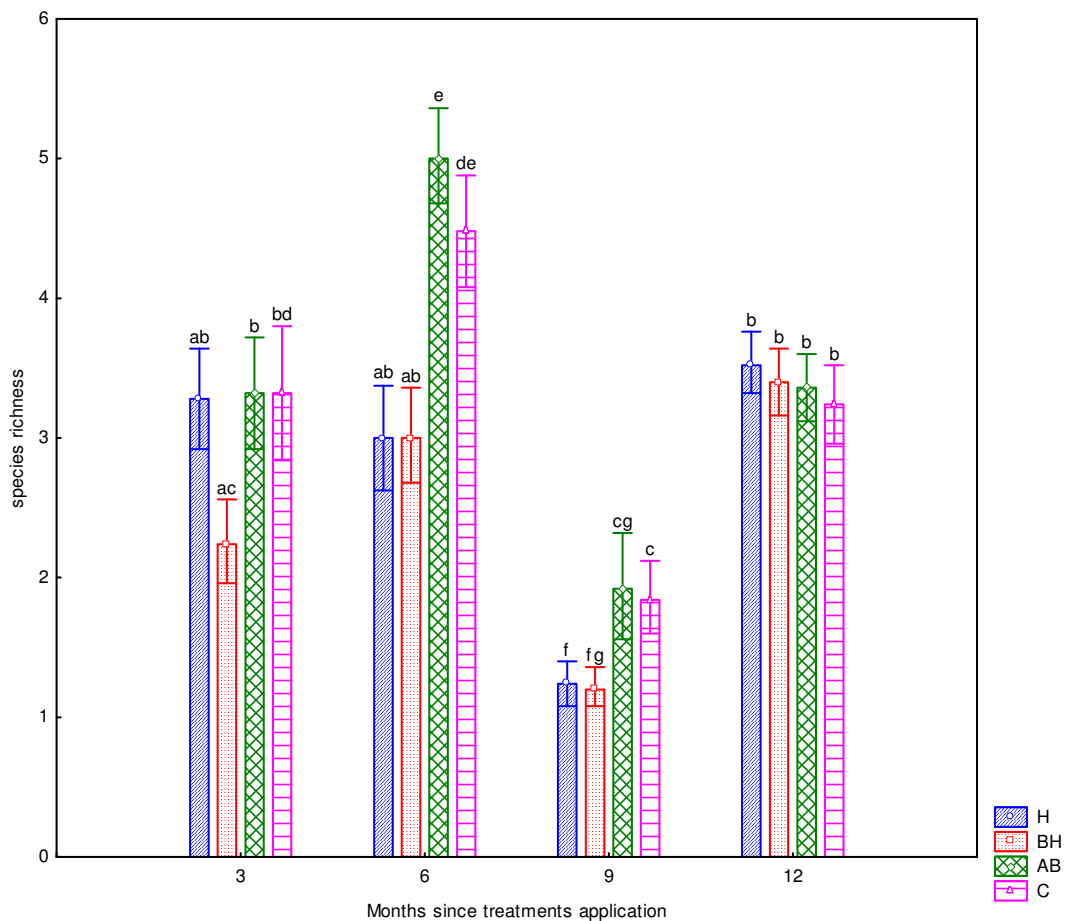


Figure 4.7 Mean species richness \pm SD of undesirable plants at 3, 6, 9 and 12 months after treatment application (autumn burn: AB, burn and herbicide: BH, herbicide: H, control: C). Lowercase letters indicate significant differences (Bonferroni Post-hoc tests) after bootstrap transformation.

Percentage cover of undesirable species was not significantly different between treatments three months after treatments were carried out on the field ($F_{3, 533} = 2.542$, $p = 0.222$) (Figure 4.8a). However, cover was significantly different between treatments after 6 months ($F_{3, 747} = 11.886$, $p < 0.0001$) (Figure 4.8b). Cover of undesirable species was the lowest on BH plots and the highest on untreated plots. The cover of undesirable species also showed significant differences between treatments after 9 months ($F_{3, 352} = 14.495$, $p < 0.0001$) (Figure 4.8c). Here, cover was the lowest in autumn burn plots and then treated with herbicide. The same pattern was observed after 12 months, at the end of the study ($F_{3, 749} = 10.529$, $p < 0.0001$) (Figure 4.8d).

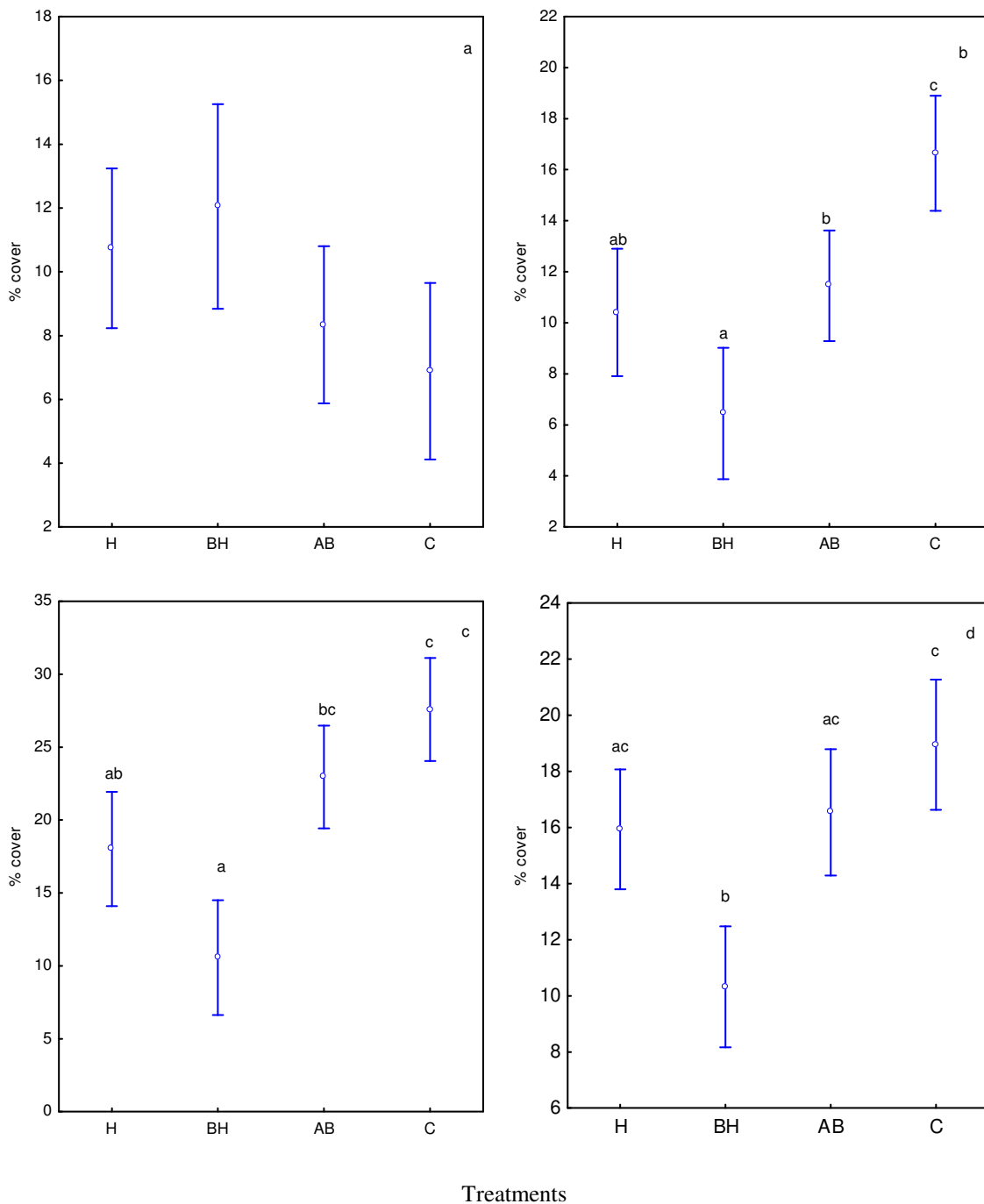


Figure 4.8 Mean percentage cover \pm SD of undesirable plants 3 months (a), 6 months (b), 9 months (c) and 12 months (d) after herbicide (H), combination autumn burn-herbicide (BH), autumn burn (AB) and control (C) treatments. Lowercase letters indicate significant difference between treatments.

Based on comparison with factorial ANOVA, the four treatments influenced alien species cover significantly different over the four periods ($F_{9, 2381} = 6.591$, $p < 0.0001$). Cover of undesirable plants was similar in all four treatments after three months, but slightly higher in burn/herbicide, herbicide and autumn burn plots than in control plots. Thereafter the lowest percentage was found on the burn/herbicide treatment plots, and herbicide and autumn burn treatments had a similar effect on undesirable species cover. The highest cover was found in control plots after 6 months. After 9 months, a combination of autumn burning and herbicide

application yielded the lowest cover of undesirable species. Herbicide application alone and autumn burning had a similar effect as the control treatment, even though the highest cover was reached in control plots. However, after 12 months, a combination of autumn burning and herbicide application had the highest effect in reduction of undesirable species cover, while the highest cover was found on control plots, whereas the cover of alien species was similar in herbicide application and autumn burn plots at the end of the study (Figure 4.9).

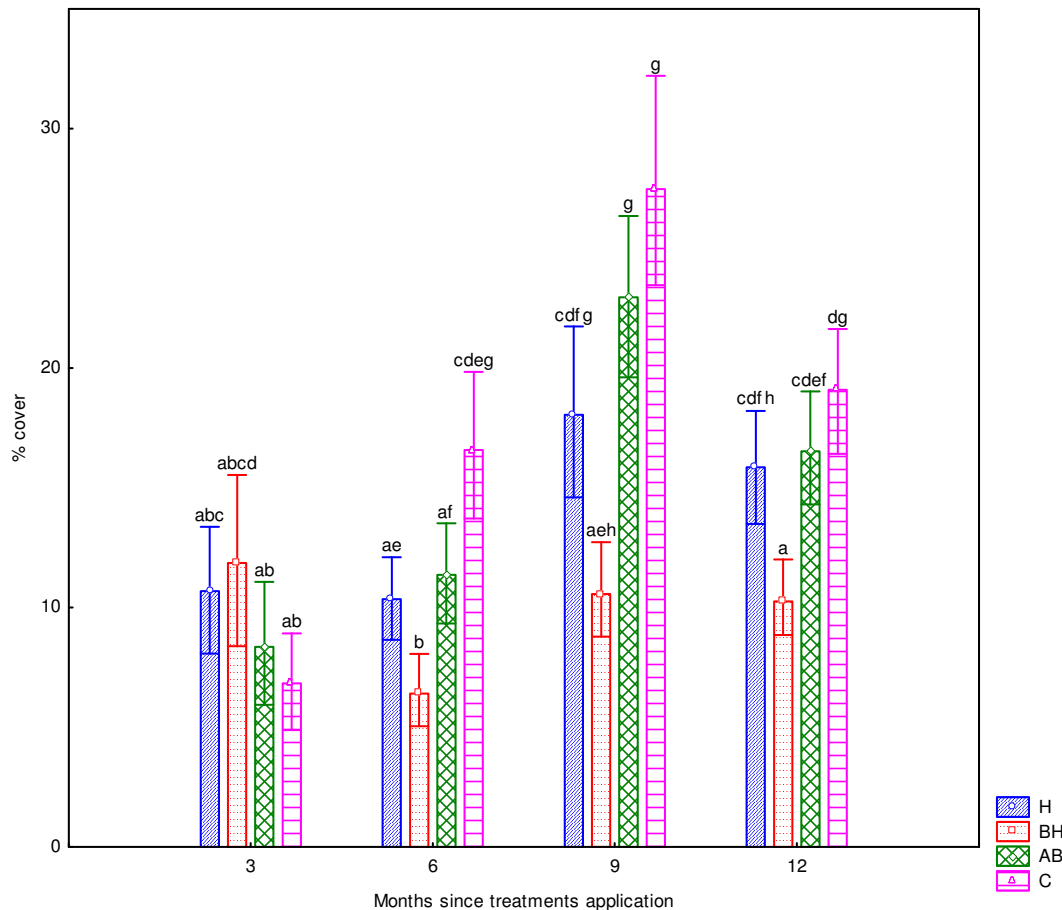


Figure 4.9 Mean percentage cover \pm SD of undesirable plants at 3, 6, 9 and 12 months after treatment application (autumn burn: AB, burn and herbicide: BH, herbicide: H, control: C). Lowercase letters indicate significant differences (Bonferroni Post-hoc tests) after bootstrap transformation.

Cover of *C. dactylon* was significantly different between treatments after 12 months ($H_{3, 400} = 69.618$, $p < 0.0001$). The cover of *C. dactylon* was significantly reduced after the burn/herbicide treatment (Table 4.3). However, the percentage cover of alien forbs ($H_{3, 400} = 2.570$, $p = 0.4626$) and exotic grasses ($H_{3, 400} = 7.610$, $p = 0.054$) was not significantly different between treatments after 12 months at the end of the study (Table 4.3).

Among those alien grasses that had the highest percentage cover *Avena fatua* cover was found to be significantly different between treatments overall ($H_{3, 400} = 11.510$, $p < 0.05$). While the cover of *Briza maxima* ($H_{3, 400} = 39.050$, $p < 0.0001$), *Bromus pectinatus* ($H_{3, 400} = 17.340$,

$p < 0.0001$) and *Lolium perenne* ($H_{3, 400} = 15.270$, $p < 0.001$) was significantly reduced by the burn/herbicide treatment (Table 4.4). Cover of a non-native forbs species, *Erodium botrys*, was significantly different between treatments at the end of the study ($H_{3, 400} = 7.910$, $p < 0.05$). The cover of this species was most reduced by herbicide treatment (Table 4.4). *Hordeum murinum* showed a similar pattern ($H_{3, 400} = 40.260$, $p < 0.0001$), but its cover was found to be lowest on burn/herbicide treatment plots (Table 4.4).

Table 4.3 Mean percentage cover \pm SD of exotic life forms and *C. dactylon* on autumn burn (AB), burn-herbicide (BH), herbicide (H) and control (C) plots. Differences were tested for significance with Kruskal-Wallis ANOVA (N= 341, DF= 3, ***=p< 0.0001, NS=non significant), using Mann Whitney U-tests as post-hoc test. Lowercase letters indicate significant difference between treatments. Most effective treatments at the end of the study after 12 months are underlined.

Life form	Treatments				H	p-value	sign
	H	BH	AB	C			
Grasses	5.11 \pm 12.73	4.33 \pm 12.00	4.34 \pm 7.65	6.42 \pm 12.96	7.610	0.054	ns
Forbs	29.90 \pm 35.81	19.90 \pm 26.98	21.81 \pm 31.43	17.39 \pm 21.64	2.570	0.462	ns
Grass (<i>C. dactylon</i>)	45.60 \pm 34.56 ^a	<u>26.12\pm21.56^c</u>	65.44 \pm 45.07 ^b	78.49 \pm 54.83 ^d	69.618	0.000	***

Table 4.4 Mean percentage cover \pm SD of undesirable species on autumn burn (AB), burn-herbicide (BH), herbicide (H) and control (C) plots. Differences were tested for significance with Kruskal-Wallis ANOVA (N= 400, DF= 3, *=p< 0.05; **=p< 0.001, ***=p< 0.0001, NS=non significant), using Mann Whitney U-tests as post-hoc test. Lowercase letters indicate significant difference between treatments. Most effective treatments at the end of the study after 12 months are underlined.

Life form	Treatments				H	p-value	sign
	H	BH	AB	C			
Grasses							
<i>Avena fatua</i>	-		0.40 \pm 0.18		11.510	0.009	**
<i>Briza maxima</i>	0.04 \pm 0.16 ^a	<u>0.01\pm0.80^c</u>	0.14 \pm 0.29 ^{be}	0.44 \pm 1.00	39.050	0.000	***
<i>Briza minor</i>	0.13 \pm 0.56	0.04 \pm 0.15	0.17 \pm 0.56	0.90 \pm 0.36	5.530	0.136	ns
<i>Bromus catharticus</i>	-			0.01 \pm 0.07	2.000	0.569	ns
<i>Bromus pectinatus</i>	<u>0.01\pm0.07^{ac}</u>	<u>0.01\pm0.07^c</u>	0.24 \pm 1.16 ^{be}	0.13 \pm 0.65 ^{de}	17.340	0.000	***
<i>Lolium perenne</i>	0.02 \pm 0.12 ^a		0.05 \pm 0.15 ^b	0.01 \pm 0.07 ^{de}	15.270	0.001	**
<i>Vulpia myuros</i>	4.26 \pm 12.14	4.18 \pm 11.93	1.43 \pm 3.35	3.19 \pm 9.05	0.930	0.816	ns
Forbs							
<i>Conyza bonariensis</i>	0.30 \pm 1.59	0.25 \pm 1.19	0.02 \pm 0.10	0.04 \pm 0.18	0.360	0.276	ns
<i>Erodium botrys</i>	<u>1.16\pm2.55^{acd}</u>	1.44 \pm 2.78 ^{ae}	4.16 \pm 6.59 ^{bf}	2.87 \pm 5.5 ^{def}	7.910	0.047	*
<i>Hordeum murinum</i>	0.63 \pm 2.39 ^{ac}	<u>0.08\pm0.26^c</u>	2.25 \pm 5.52 ^{be}	2.54 \pm 7.34 ^{de}	40.260	0.000	***
<i>Hypochaeris radicata</i>	26.56 \pm 33.42	17.35 \pm 24.95	15.16 \pm 26.77	12.77 \pm 17.87	4.000	0.259	ns
<i>Trifolium repens</i>	1.85 \pm 4.66	0.85 \pm 2.43	2.47 \pm 7.70	1.69 \pm 5.14	3.890	0.272	ns

4.3.3. Effect of burn season on native and alien plant species

Native plant species

Percentage cover of native grasses, forbs and geophytes was significantly different between seasons after the autumn burn ($H_{3, 58} = 50.280$ $p < 0.0001$, $H_{3, 58} = 47.712$ $p < 0.0001$, $H_{3, 58} = 46.614$ $p < 0.0001$ respectively). Native grasses emerged in spring, six months after treatment, and cover was the highest in summer (nine months after treatment). Native forbs emerged after three months in winter, disappeared during spring and summer and re-emerged after 12 months, in autumn. Geophyte cover was highest in winter, after three months, and decreased in spring (six months after treatment). No geophytes were found in summer, but cover increased in autumn, 12 months after treatment (Table 4.5).

Comparison of the effects of the autumn burn on indigenous species showed that among grasses, the cover of *Pentaschistis pallida* ($H_{3, 58} = 42.281$, $p < 0.0001$) was significantly higher in spring than in summer, while cover of *Tribolium hispidum* was slightly higher in summer ($H_{3, 58} = 35.200$, $p < 0.0001$) (Table 4.6). Of the native forbs, *Cotula turbinata* ($H_{3, 58} = 52.981$, $p < 0.0001$) and *Felicia bergeriana* ($H_{3, 58} = 55.343$, $p < 0.0001$) had the highest percentage cover after three months, in winter (Table 4.6). Among geophytes, *Oxalis cummutata* showed the highest percentage cover in winter ($H_{3, 58} = 25.958$, $p < 0.0001$), while *Oxalis versicolor* had the highest cover in autumn ($H_{3, 58} = 46.390$, $p < 0.0001$) (Table 4.6).

Table 4.5 Mean percentage cover \pm SD of indigenous life forms on autumn burn plots after 3 months (winter), 6 months (spring), 9 months (summer) and 12 months (autumn). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 58, DF= 3, ***= $p < 0.0001$). The highest percentage cover is underlined.

Life form	Months after treatment application				H	p-value	sign
	3	6	9	12			
Grasses		<u>2.03\pm1.04</u>	1.56 \pm 1.46		50.280	0.000	***
Forbs	4.51 \pm 1.73			0.43 \pm 1.55	47.712	0.000	***
Geophytes	<u>3.07\pm3.94</u>	0.07 \pm 0.18		1.37 \pm 0.62	46.614	0.000	***

Table 4.6 Mean percentage cover \pm SD of indigenous species on autumn burn plots after 3 months (winter), 6 months (spring), 9 months (summer) and 12 months (autumn). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 58, DF= 3, *=p< 0.05, ***=p< 0.0001, NS=non significant). The highest percentage cover is underlined.

Life form	Months after treatment application				H	p-value	sign
	3	6	9	12			
Grasses							
<i>Imperata cylindrica</i>		0.26 \pm 0.26	0.38 \pm 0.84		17.244	0.000	***
<i>Pentaschistis curvifolia</i>		0.36 \pm 0.33			29.814	0.000	***
<i>Pentaschistis pallida</i>		<u>4.48\pm5.57</u>	1.19 \pm 0.89		42.281	0.000	***
<i>Tribolium hispidum</i>		0.52 \pm 0.21	<u>1.73\pm5.49</u>		35.200	0.000	***
Forbs							
<i>Arctotheca calendula</i>	0.70 \pm 0.74			0.40 \pm 1.55	35.889	0.000	***
<i>Cotula coronopifolia</i>	1.53 \pm 5.16				5.834	0.120	ns
<i>Cotula turbinata</i>	<u>6.04\pm2.26</u>			0.03 \pm 0.13	52.981	0.000	***
<i>Dischisma capitatum</i>	0.62 \pm 0.81				22.309	0.000	***
<i>Drosera trinervia</i>	0.55 \pm 1.36				12.086	0.007	*
<i>Felicia bergeriana</i>	<u>4.58\pm3.68</u>				55.343	0.000	***
Geophytes							
<i>Cyanella hyacinthoides</i>	0.85 \pm 0.53			0.38 \pm 0.39	38.301	0.000	***
<i>Geissorhiza confusa</i>	0.33 \pm 0.31				29.928	0.000	***
<i>Moraea villosa</i>	0.06 \pm 0.17				5.836	0.119	ns
<i>Ornithogalum thyrsoides</i>	0.33 \pm 0.31			0.40 \pm 0.54	21.732	0.000	***
<i>Oxalis commutata</i>	<u>7.18\pm12.62</u>				25.958	0.000	***
<i>Oxalis pes-caprae</i>	0.53 \pm 0.99			0.86 \pm 1.73	8.474	0.037	*
<i>Oxalis purpurea</i>				0.60 \pm 1.18	12.086	0.007	*
<i>Oxalis versicolor</i>	0.70 \pm 2.57			<u>2.02\pm1.75</u>	46.390	0.000	***
<i>Satyrium erectum</i>	0.38 \pm 0.29	0.06 \pm 0.17		0.18 \pm 0.27	17.867	0.000	***

For the spring burn, percentage cover of native grasses, forbs and geophytes was significantly different between seasons ($H_{3, 19} = 13.415$, $p < 0.05$; $H_{3, 19} = 12.075$, $p < 0.05$; $H_{3, 19} = 9.416$, $p < 0.05$ respectively). Native grasses emerged after three months (in summer), and were observed again after 12 months, in spring. Grass cover was higher in summer than in spring. Native forbs and geophytes were found at the site six months (autumn), nine months (winter) and 12 months (spring) after treatment. Cover of forbs was the highest in winter, while the highest geophyte cover was observed in autumn (Table 4.7).

Comparing individual plant species, *Tribolium hispidum* had the highest cover among grasses in summer ($H_{3, 19} = 7.060$, $p = 0.070$). For the forb species, *Cotula turbinata* showed the highest percentage cover in winter ($H_{3, 19} = 11.381$, $p < 0.05$), while *Dischisma capitatum* cover was the highest in autumn ($H_{3, 19} = 17.474$, $p < 0.0001$). Among geophytes, *Oxalis commutata* ($H_{3, 19} = 6.230$, $p = 0.100$) and *Oxalis pes-caprae* ($H_{3, 19} = 15.176$, $p < 0.05$) had the highest percentage cover in autumn and winter, while *Satyrium erectum* cover was slightly higher in autumn than in the other seasons ($H_{3, 19} = 9.695$, $p < 0.05$). In spring, *O. cummutata* had the highest percentage cover (Table 4.8).

Table 4.7 Mean percentage cover \pm SD of indigenous life forms on spring burn plots after 3 months (summer), 6 months (autumn), 9 months (winter) and 12 months (spring). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 19, DF= 3, $*=p < 0.05$), The highest percentage cover is underlined.

Life form	Months after treatment application				H	p-value	sign
	3	6	9	12			
Grasses	<u>2.87\pm4.75</u>			1.48 \pm 0.53	13.415	0.003	*
Forbs		0.45 \pm 0.76	<u>2.44\pm1.14</u>	1.66 \pm 1.14	12.075	0.007	*
Geophytes		<u>2.58\pm2.49</u>	1.21 \pm 0.30	1.47 \pm 0.52	9.416	0.024	*

Table 4.8 Mean percentage cover \pm SD of indigenous species on spring burn plots after 3 months (summer), 6 months (autumn), 9 months (winter) and 12 (spring). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 19, DF= 3, *= $p < 0.05$; **= $p < 0.001$, ***= $p < 0.0001$, NS=non significant), using Mann Whitney U-tests as post-hoc test. Lowercase letters indicate significant difference between treatments. The highest percentage cover is underlined.

Life form Species	Period after treatment application				H	p-value	sign
	3	6	9	12			
Grasses							
<i>Pentaschistis densifolia</i>				0.30 \pm 0.27	9.450	0.024	*
<i>Pentaschistis pallida</i>	0.37 \pm 0.25			2.20 \pm 4.36	9.107	0.028	*
<i>Tribolium hispidum</i>	<u>2.62\pm4.92</u>			2.30 \pm 4.32	7.060	0.070	ns
Forbs							
<i>Artotheca calendula</i>		0.20 \pm 0.27	0.95 \pm 0.62		14.012	0.003	*
<i>Cotula turbinata</i>		0.70 \pm 1.30	<u>2.82\pm1.43</u>	0.30 \pm 0.27	11.381	0.009	*
<i>Dischisma capitatum</i>				<u>2.35\pm1.35</u>	17.474	0.000	***
<i>Drosera trinervia</i>			2.24 \pm 1.06	<u>1.46\pm1.92</u>	17.474	0.000	***
<i>Ursina anthemoides</i>			0.75 \pm 0.35	0.60 \pm 0.22	14.443	0.002	**
Geophytes							
<i>Cyanella hyacinthoides</i>		0.56 \pm 0.15	0.76 \pm 0.22	0.40 \pm 0.22	13.433	0.004	*
<i>Moraea villosa</i>				0.35 \pm 0.78	2.800	0.423	ns
<i>Ornithogalum thyrsoides</i>		0.10 \pm 0.22	0.50 \pm 0.87	0.10 \pm 0.22	2.275	0.517	ns
<i>Oxalis commutata</i>		<u>7.40\pm9.42</u>	2.28 \pm 1.36	3.05 \pm 1.87	6.230	0.100	ns
<i>Oxalis pes-caprae</i>		<u>3.94\pm2.82</u>	2.60 \pm 2.04		15.176	0.002	*
<i>Oxalis purpurea</i>		0.20 \pm 0.44			2.800	0.423	ns
<i>Oxalis versicolor</i>		1.11 \pm 0.82	0.63 \pm 0.41	0.60 \pm 0.42	8.311	0.040	*
<i>Satyrium erectum</i>		1.07 \pm 1.07	0.96 \pm 0.18	0.65 \pm 0.49	9.695	0.021	*

Alien plant species

A comparison of cover of alien grasses on plots burned in autumn showed a significant difference throughout seasons after one year ($H_{3, 60} = 29.858$, $p < 0.0001$). Grass cover was recorded after three months (winter), six months (spring) and nine months (summer). It seemed to be higher in summer and slightly lower in winter than in spring, but no grasses were recorded in autumn (Table 4.9). Cover of *C. dactylon*, an undesirable indigenous creeping grass, was significantly different between seasons ($H_{3, 60} = 34.322$, $p < 0.0001$). Cover of *Cynodon dactylon* was decreasing from spring to autumn (Table 4.9). The cover of alien forbs was also significantly different between seasons at the end of the study ($H_{3, 60} = 43.174$, $p < 0.0001$). Exotic forbs were present three months, nine months and 12 months after treatment. The cover of alien forbs was the higher in winter than in autumn and summer, when cover was the lowest (Table 4.9).

Among those invasive alien grass species, which were mainly recorded after the autumn burn in spring, *Vulpia myuros* ($H_{3, 60} = 48.653$, $p < 0.0001$) had the highest percentage cover. In summer, *Vulpia sp.* cover was the highest compared to two other species, *Briza maxima* ($H_{3, 60} = 19.116$, $p < 0.0001$) and *Bromus pectinatus* ($H_{3, 60} = 39.336$, $p < 0.0001$) (Table 4.10). Among alien forbs, *Hypochaeris radicata* ($H_{3, 60} = 42.487$, $p < 0.0001$) had the highest cover in winter and autumn, followed by *Trifolium repens* ($H_{3, 60} = 43.875$, $p < 0.0001$) and *Erodium botrys* ($H_{3, 20} = 44.430$, $p < 0.0001$) (Table 4.10). *Hordeum murinum* was recorded in winter, spring and summer, its cover was slightly higher in spring than in the other seasons ($H_{3, 60} = 11.401$, $p < 0.0001$) (Table 4.10).

Table 4.9 Mean percentage cover \pm SD of exotic life forms and *C. dactylon* in autumn burn plots after 3 months (winter), 6 months (spring), 9 months (summer) and 12 months (autumn). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 60, DF= 3, ***= $p < 0.0001$), The highest percentage cover is underlined.

Life form	Months after treatment application					p-value	sign
	3	6	9	12	H		
Grasses	1.90 \pm 2.62	1.93 \pm 0.89	<u>2.49\pm3.47</u>		29.858	0.000	***
Forbs	<u>8.43\pm7.78</u>		0.93 \pm 2.71	7.02 \pm 6.11	43.174	0.000	***
Grass (<i>C. dactylon</i>)		<u>30.26\pm5.45</u>	29.52 \pm 5.87	28.43 \pm 5.45	34.322	0.000	***

Table 4.10 Mean percentage cover \pm SD of undesirable species on autumn burn plots after 3 months (winter), 6 months (spring), 9 months (summer) and 12 months (autumn). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 60, DF= 3, *=p< 0.05, ***=p< 0.0001, NS=non significant), The highest percentage cover is underlined.

Life form	Months after treatment application				H	p-value	sign
	3	6	9	12			
Grasses							
<i>Avena fatua</i>		0.26 \pm 0.26			27.231	0.000	***
<i>Briza maxima</i>	0.13 \pm 0.23	0.36 \pm 0.30	0.06 \pm 0.17		19.116	0.000	***
<i>Briza minor</i>		0.56 \pm 0.58			39.336	0.000	***
<i>Bromus catharticus</i>		0.03 \pm 0.13			3.000	0.392	ns
<i>Bromus pectinatus</i>		0.65 \pm 1.01	0.06 \pm 0.26		22.590	0.000	***
<i>Lolium perenne</i>	0.16 \pm 0.24	0.16 \pm 0.24			11.800	0.008	*
<i>Vulpia myuros</i>		<u>2.56\pm1.28</u>	0.26 \pm 0.77		48.653	0.000	***
Forbs							
<i>Conyza bonariensis</i>	0.10 \pm 0.21			0.06 \pm 0.17	5.793	0.122	ns
<i>Erodium botrys</i>	<u>4.68\pm4.15</u>		0.66 \pm 2.58	2.49 \pm 1.619	44.430	0.000	***
<i>Hordeum murinum</i>	2.01 \pm 2.96	2.29 \pm 3.67	2.17 \pm 3.67		11.401	0.009	*
<i>Hypochaeris radicata</i>	<u>12.77\pm13.48</u>		0.26 \pm 1.03	11.51 \pm 11.44	42.487	0.000	***
<i>Trifolium repens</i>	<u>6.06\pm10.13</u>			5.59 \pm 10.15	43.875	0.000	***

A comparison of cover of alien grasses on the spring burn plots showed a significant difference throughout seasons after one year ($H_{3, 20} = 18.506$, $p < 0.0001$). Grasses were recorded only 12 months after treatment, in the following spring (Table 4.11). Cover of *C. dactylon* was significantly different ($H_{3, 20} = 16.280$, $p < 0.001$). *Cynodon dactylon* was recorded from summer to spring with the highest percentage cover in spring (Table 4.11). The cover of alien forbs was also significantly different between seasons at the end of the study ($H_{3, 20} = 15.832$, $p < 0.001$). Forbs were present from autumn onwards, six months, nine months and 12 months after treatment. The cover of alien forbs was slightly higher in autumn than in winter, with the lowest cover in spring (Table 4.11).

Among those invasive alien grass species, which were only recorded in spring, *Vulpia myuros* ($H_{3, 20} = 18.506$, $p < 0.0001$) and *Briza minor* ($H_{3, 20} = 18.506$, $p < 0.0001$) had the highest percentage cover, while *Briza maxima* ($H_{3, 20} = 18.555$, $p < 0.0001$) and *Lolium perenne* ($H_{3, 20} = 10.059$, $p < 0.05$) had the lowest percentage cover (Table 4.12). Among alien forbs that were recorded in autumn, winter and spring, cover of *Hypochaeris radicata* was highest in autumn ($H_{3, 20} = 17.051$, $p < 0.0001$), while cover of *Trifolium repens* ($H_{3, 20} = 10.173$, $p < 0.05$) and *Erodium botrys* ($H_{3, 20} = 18.555$, $p < 0.0001$) was highest in winter. In spring, all three species had the lowest percentage cover (Table 4.12).

Table 4.11 Mean percentage cover \pm SD of exotic life forms and *C. dactylon* on spring burn plots after 3 months (summer), 6 months (autumn), 9 months (winter) and 12 months (spring). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 20, DF= 3, **= p< 0.001, ***=p< 0.0001). The highest percentage cover is underlined.

Life form	Period after treatment application				H	p-value	sign
	3	6	9	12			
Grasses				3.76 \pm 1.07	18.506	0.000	***
Forbs		<u>11.45\pm1.96</u>	9.56 \pm 3.53	3.94 \pm 1.41	15.832	0.001	**
Grass (<i>C. dactylon</i>)	17.14 \pm 1.04	14.88 \pm 9.8	22.44 \pm 2.57	<u>24.76\pm3.57</u>	16.280	0.001	**

Table 4.12 Mean percentage cover \pm SD of undesirable species on spring burn plots after 3 months (summer), 6 months (autumn), 9 months (winter) and 12 months (spring). Differences were tested for significance with Kruskal-Wallis ANOVA (N= 20, DF= 3, *= $p < 0.05$; **= $p < 0.0001$). The highest percentage cover is underlined.

Life form	Period after treatment application				H	p-value	sign
	3	6	9	12			
Grasses							
<i>Briza maxima</i>				1.20 \pm 1.45	18.555	0.000	***
<i>Briza minor</i>				<u>2.99\pm1.90</u>	18.506	0.000	***
<i>Bromus catharticus</i>				0.30 \pm 0.27	10.059	0.018	*
<i>Bromus pectinatus</i>				1.02 \pm 1.10	18.603	0.000	***
<i>Lolium perenne</i>				0.30 \pm 0.27	10.059	0.018	*
<i>Vulpia myuros</i>				<u>6.76\pm4.70</u>	18.506	0.000	***
Forbs							
<i>Erodium botrys</i>		1.41 \pm 0.59	1.48 \pm 0.49	1.55 \pm 0.45	11.191	0.011	*
<i>Hypochaeris radicata</i>		<u>20.38\pm3.49</u>	15.24 \pm 4.90	1.90 \pm 3.02	17.051	0.000	***
<i>Trifolium repens</i>		6.53 \pm 5.59	<u>10.20\pm5.20</u>	3.20 \pm 7.15	10.173	0.017	*

4.4. Discussion

The persistence of undesirable species on old agricultural fields is thought to inhibit the re-establishment of native species on such old fields, and could pose a further threat to surrounding natural vegetation. Therefore, in this study, conventional treatments were implemented to reduce the number and cover of undesirable species, while at the same time improving indigenous plant species richness and cover on an old field. The efficiency of autumn and spring burn in reducing alien grass cover and enhancing native plant species cover was also examined, as was the effectiveness of these treatments after one year.

4.4.1. Species richness

Autumn burning, herbicide application and a combination of both treatments affected species richness of alien and indigenous species. The species richness of undesirable vegetation was reduced mostly by the combination of autumn burning and herbicide application. Krug *et al.* (2004) found that species richness of exotic grasses decreased most under autumn burn treatments and herbicide application. The herbicide application also reduced the number of undesirable species in this study, but a small decline of undesirable species was observed in the autumn burn treatment compared to plots left untreated. In contrast, Lesica & Martin (2003) stated that burning might be favourable for establishment of non-native species. In our study, the number of native species was also most reduced through the combination of autumn burning and herbicide application. Similarly, Krug & Krug (2007) found that species richness of native plants was reduced three years after burning. However, Midoko-Iponga (2004) found the highest species richness of indigenous species on untreated plots. A similar result was found in this study, with the number of undesirable and native species being the highest in the control plots, followed by autumn burn treatments.

Although Holmes & Newton (2004) observed that fynbos vegetation regenerates from the seed bank after fire, this study showed that the number of native species did not increase after the autumn burning of an old agricultural field. This highlighted that the seed bank of native species in the vegetation type investigated is either depleted or does not even exist in old fields (Shiponeni, 2003; Krug & Krug, 2007), most likely due to the agricultural practices such as ploughing, that were conducted on the old cultivated lands (Hammouda *et al.*, 2003; Reiné *et al.*, 2004). In sand fynbos, seeds of long-lived species had low persistence in the seed bank after removal of aliens (Holmes, 2002). Therefore, the soil seed bank could play an important role for the re-vegetation potential of native species in old cultivated fields (Wills & Read, 2007). In addition, seeds of most native species are only dispersed over a short-distance

from the area where the natural vegetation still exists (Stylinski & Allen, 1999; Shiponeni, 2003; Pugnaire *et al.*, 2006), which further reduces the presence of propagules of indigenous species on old fields.

However, seed dispersal could not be the only barrier to the re-establishment of native species. Standish *et al.* (2007) recognised that indigenous shrub and grasses of the Australian wheatbelt are not seed dispersal limited, due to their wind-dispersal characteristics. In the CFR, seed of indigenous forbs and shrubs such as *Elytropappus rhinocerotis* have been recognised to disperse over long distances (Shiponeni, 2003). Furthermore, Krug (2008), in his modelling of *E. rhinocerotis* seed dispersal, confirmed that this shrub species was dispersed over long-distance into the old fields. Therefore Shiponeni (2003) and Krug (2008) ascribed that seed dispersal could not be a limiting factor for native species re-establishment, especially shrubs, on old fields in Swartland Alluvium Fynbos/Swartland Shale Renosterveld, even though they found that the quantity of seed decreased after certain distances.

Seed of native species also had to overcome environmental barriers, such as physical and chemical soil properties, as defined for invasive species by Richardson *et al.* (2000). The legacy of agricultural practices on soil properties has been recognised to affect re-establishment of indigenous species seed (North *et al.*, 2005; Standish *et al.*, 2006). Likewise, the high level of soil salinity on this old field (Chapter 3 in this thesis) could affect seeds, particularly of shrub species (Cramer *et al.*, 2004; Cramer *et al.*, 2007; Koyro & Eisa, 2008).

Thus, the trial treatments, particularly the combination of autumn burning and herbicide application, reduced the number of undesirable species even though they are dominant in the seed bank on the old field investigated (Shiponeni, 2003). However, this treatment also had a negative impact on the species richness of indigenous species.

4.4.2. Vegetation cover

Aims of the trial treatments were to identify which treatment reduces the cover of undesirable species and enhances the cover of native species, based on the assumption that one of the reasons for the slow re-establishment and growth of indigenous species is competition for space, resources and/or light between both types of plants. It had been found that the re-establishment of indigenous shrubs was successful in areas where undesirable species were completely removed by the use of herbicide (Cione *et al.*, 2002; Midoko-Iponga, 2004; Midoko-Iponga *et al.*, 2005), or hand-weeding (Midoko-Iponga, 2004). In our study, the change in indigenous species cover indicated that the use of fire and herbicide could positively affect the succession of native species.

Despite the sizeable decrease in indigenous grasses cover mostly due to a combination of autumn burning and herbicide application and herbicide application alone, per a previous study (Midoko-Iponga, 2004), particularly the decrease was recorded for *Pentaschistis curvifolia* and *Pentaschistis pallida*. The autumn burn and herbicide treatments were shown to be beneficial to other native grasses such as *Tribolium hispidum* and *Imperata cylindrica*. Additionally, cover of native forbs, such as *Cotula turbinata* and *Felicia bergeriana*, also showed an increase in cover throughout the autumn burn and herbicide treatments. Moreover, the cover of geophytes, such as *Ornithogalum thyrsoides*, *Oxalis pes-caprae*, *Geissorhiza confusa* and *Satyrium erectum* showed an increase after fire and a combination of autumn burning and herbicide application. Midoko-Iponga (2004) found that native forbs and geophytes cover increased throughout the treatments, especially herbicide and fire, and Musil *et al.* (2005) reported that a moderately intense fire could be favourable for the development of geophytes. Holmes (2008) found that herbicide application had a positive effect on native species by reducing the cover of alien grasses and forbs. This increase in cover of small native species could be related to increased open space due to the reduction of exotic species (Goldberg, 1987; Wilson & Tilman, 1993; Foster, 1999; Bakker & Wilson, 2004; Corbin & D'Antonio, 2004; Huston, 2004).

In that regard, this augmentation of native plant cover was related to a significant reduction of undesirable species, achieved mostly through a combination of autumn burning and herbicide application. Cover of the most undesirable grasses (*C. dactylon*) was reduced using the combination treatment as well as herbicide alone. Mau-Crimmins (2007) also found that *C. dactylon* cover was most reduced using an herbicide with glyphosate as the main element on an old field in Arizona. Furthermore, the cover of certain exotic plants such as *Lolium perenne*, *Briza maxima* and *Bromus pectinatus* also showed a decrease mainly due to a combination of autumn burning and herbicide treatments. The investigated annual grasses are adapted to fire (Milton, 2004); therefore, it is essential to apply herbicides on burned plots to inhibit their capacity to regenerate after fire (D'Antonio & Vitousek, 1992; van Rooyen, 2003; Brooks *et al.*, 2004; Milton, 2004). Moreover, the cover of alien forbs such as *Hordeum murinum* was reduced most using the combination treatment, and *Erodium botrys* cover was also affected but rather by the herbicide treatment. This herbicide effect on *E. botrys* could be related to the high salt level (Chapter 3 in this thesis), since *E. botrys* also occurs in old fields in the Australian Wheatbelt, however only in areas that are not affected by high salinity levels (Standish *et al.*, 2006; Cramer *et al.*, 2007). The sensitivity to the herbicide in our case could be because *E. botrys* is a glycophytic species that is not salt tolerant (Parsons & Martini-Lamb, 2003; Cixin He, 2005). We assume, therefore, that the high salt level in the old fields

investigated in this study affect the morphology of *E. botrys* so that it becomes sensitive to herbicide.

It has been recognised that a decrease in cover of certain exotic species enhances other non-native plant growth. The reduction of these exotic species and *C. dactylon* seemed to be favourable mostly to alien forbs such as *Hypochaeris radicata*, an exotic forb with the highest cover throughout the trial treatments in this study. Mau-Crimmins (2007) also observed that the elimination of *C. dactylon* encourages the establishment of other undesirable plants on an old field. This replacement of non-native species among themselves affects indigenous species by reducing their re-establishment and growth in old fields (Kulmatiski, 2006). In addition, differing salt levels between this abandoned agricultural field and an older abandoned agricultural field in the same reserve could have stimulated alien plants and affected the growth of native species (Cramer *et al.*, 2004; Cramer *et al.*, 2007; Cixin He, 2005).

Across all treatments, the combination of autumn burning and herbicide application showed the best result in reducing undesirable species cover and increasing of a few indigenous plants, even though it also had a negative effect on native species cover.

4.4.3. Differences between autumn and spring burn treatment

This study also investigated the effects of two different burn seasons (autumn and spring fires) on native and exotic species. Autumn burn has been the main prescribed seasonal fire used to reduce the impact of non-native plant species in old cultivated lands throughout the Mediterranean climate regions. Cowling, Pierce & Moll (1986) and White, Pendleton & Pendleton (2006) observed that autumn burns enhanced the cover of grasses in natural vegetation. In this study, burning the old field in autumn could inhibit alien grass or enhance the cover of alien forbs such as *Hypochaeris radicata*. Burning in autumn seemed to increase the cover of native grasses as *Pentaschistis pallida*. However, the result of autumn burn could be masked by seasonal effects, as the main growing period in the Western Cape lowlands are winter and spring. The effectiveness of treatment might also be influenced by climatic conditions, e.g. low or high winter rainfall. Furthermore, Midoko-Iponga (2004) and Musil *et al.* (2005) have shown that an autumn burn can have a negative effect on indigenous species. In our study, an autumn burn showed both negative and positive effects on native and non-native plants, mainly depending on the growth period of the species. Musil *et al.* (2005) increased the intensity of the autumn burn, making it comparable to a mid-spring burn, to investigate the impact of heat on exotic and indigenous species cover. They found that intense

fire was successful in reducing alien species, but at the same time negatively affected native species, this was also the case in our study.

In this region, fire intensity is related to season (Govender, Trollope & van Wilgen, 2006), and spring burns are generally hotter. The heat of fire depends on the amount and the level of biomass that represents the fuel (Arianoutsou, 2004). Therefore, in this study, the burning of certain areas on old agricultural field was conducted in the middle of spring. Unfortunately, the heat of fire was not as hot as expected due to a lack of biomass and due to the fact that the biomass was not dry enough (Bernard Wooding pers. comm., 2007). However, for the purposes of this study, the comparison between the effects of autumn and spring burns showed that the autumn burn reduced alien grasses only slightly, whereas the spring burn burned spring-growing exotic species, which seemed to inhibit the cover of native species. Gillespie & Allen (2004) also found that spring burning effectively reduced the cover of invasive alien grasses. In our case, the mid-spring burn significantly affected the cover of individual exotic grasses, such as *Lolium perenne* and *Briza maxima*, but at the same time, cover of other alien grasses, particularly *Vulpia myuros* and *Briza minor*, seemed to increase. Moyes *et al.* (2005) also found that a spring burn affected an alien grass species, *Bromus diandrus*. *Cynodon dactylon* was mostly represented by young and fresh shoots, whereas *A. fatua* and *L. perenne* plants were almost dying and rather dry when the spring burn was carried out. However, spring burn seemed to increase *C. dactylon* cover by maintaining it as the main undesirable plants species by increasing its cover in spring on the old fields, while on autumn burn plots cover of *C. dactylon* is the decreases slightly. The contrasting effects of the spring and autumn burns can also be related to the life cycle of plant species in general. Lesica & Martin (2003) recognised that the impact of both fires on non-native species cover depended on the germination period of those species. A cool fire before germination could encourage the growth of seedlings, while a burn after germination could increase the mortality of adult plants; thereby also reduce the recruitment into the soil seed bank for the next year.

To conclude, the spring burn could also be used as a restoration method to reduce the impact of undesirable species and enhance native plants species cover such as geophytes. However, spring burn seemed to inhibit some native plants and enhance the dominant undesirable plant (*C. dactylon*) thereby its intensity, the time of the application and the stage of life cycle of the species should be considered for the effectiveness of that treatment.

4.4.4. Period of treatments effectiveness

The duration of a restoration project is a problem, and raises the question of how long should a study be conducted? Usually, the monitoring of vegetation cover on restored areas is done

over one to three years after the treatments have been applied (Cione *et al.*, 2002; Holmes & Newton, 2004; Midoko-Iponga, 2004; Antonsen & Olsson, 2005; Holmes, 2005; Musil *et al.*, 2005; Krug & Krug, 2007; Holmes, 2008), which does not necessarily mean that the treatments are still active. To reduce *C. dactylon*, Mau-Crimmins (2007) applied an herbicide each year during monitoring to reduce *C. dactylon*. In this study, we examined the effectiveness of our treatments three, 6, 9 and 12 months after their application. The results showed that the maximum efficacy of those treatments was between three months and 9 months, which could explain why after 12 months all the treatments showed the same pattern. Midoko-Iponga (2004) did not find any difference on the number of species between burned, herbicide application and brush-cutting treatments after one year. Furthermore, Krug & Krug (2007) analysed the number of species on those treatments more than two years later and found that the species richness of indigenous species was the same in all treatments.

However, one year monitoring after treatment application, as per this study, could not be enough to improve the vegetation on old fields in the long-term. The results in this study suggest that further treatment applications are needed due to the persistence of undesirable species, such as *C. dactylon*, with continuous monitoring needed to assess the effect of those treatments on undesirable plant species.

4.5. Conclusion

Fire or herbicide application alone does not always succeed in reducing presence or growth of undesirable species. Depending on the quality of fire, it can even destroy native species or enhance the growth of undesirable species. The effectiveness of herbicides can be reduced when alien plant species become herbicide-resistant because of repetitive use of the same herbicide (Hashem *et al.*, 2001). The Western Cape has been recognised as an area with a high potential of species resistant to herbicide, particularly to acetyl-coenzyme A, a component of ACCase herbicides such as Gallant Super (Bromilow, 2001). In Australia, such a resistance had been identified in most agricultural areas (Hashem *et al.*, 2001). Matarczyk *et al.* (2002) and Pywell, Webb & Putwain (1995) recognised that herbicide application could also threaten indigenous species. Thus, to avoid herbicide-resistant species, different herbicides should be used or combined with other treatments (Hashem *et al.*, 2001; Diggle, Neve & Smith, 2003; Vila-Aiub *et al.*, 2005). Therefore, the herbicide application combines with burning or other treatments such as ploughing could be more effective.

In this study, the combination of autumn burning with herbicide application showed a strong impact on undesirable species by effectively reducing their number and cover of individuals such as *C. dactylon*, *B. maxima*, *B. pectinatus* and *H. murinum*. The reduction of those

undesirable plant species opened space, permitting the germination of the native seed that remained in the soil seed bank and the growth of two geophytes species: *G. confusa* and *S. erectum*. The combination of herbicide application with ploughing showed positive results by significantly reducing undesirable species in old fields located at the ecotone of Fynbos and Karoo (Holmes, 2005). Holmes (2008) also used a combination treatment by combining herbicide application with mulch, and found that the relative success of that combination treatment was only due to the herbicide because the condition was not favourable for the efficacy of the second treatment.

The period of treatment application could also influence their result. For example the post-emergence herbicide Gallant*Super should be applied when target species are fresh and green (Bromilow, 2001; Matarczyk *et al.*, 2002; Dow Agrosiences Southern Africa, 2004). Therefore, the treatment combination should begin with this herbicide and then be followed by the application of another treatment such as ploughing.

The spring burn reduced alien grasses, particularly those that grow between spring and summer. However, it could cause the decrease of native plants and the increase of some undesirable plant such as *C. dactylon*, therefore the intensity of the fire should be managed in order to avoid the aliens' soil seed bank to grow and at the same time, favour the germination of native species. Then the spring burn should be combined with other treatments and the spread of native plant species seeds.

Finally, trial treatments seems to be effective at least 9 months after the treatments were applied, therefore continued treatment application will be imperative to obtain better results in reducing alien species on old agricultural fields. A minimum of 3 years could be necessary for an old agricultural field restoration study; the specific duration will depend on the results of previous monitoring to determine whether or not there has been any reduction of alien plant species.

Recommendations

- To reduce alien plant species cover, a post-emergent herbicide (e.g. Gallant Super) should be applied twice, with a two week-interval between treatments;
- Autumn/ spring burn should be conducted two weeks after the second herbicide application;
- During the first three years, the monitoring should be carried out nine months after the burn, above, followed by the re-application of the combined treatment each year thereafter;

- Continuous monitoring will help to determine if further application will be needed;
- Additionally, sowing of native plant species seed will be necessary.

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Chapter 5 General Discussion and Conclusion

5.1 Old field restoration: the interplay of soil properties and vegetation cover

The dominance of exotic plant species years after cultivation ceased, as well as their persistence after one or repeated clearing attempts, has led restoration ecologists to investigate soil properties and their impact on plant species (Stylinski & Allen, 1999; MacDougall & Turkington, 2005; Kulmatiski, 2006; Kulmatiski *et al.*, 2006; Coleman & Levine, 2007). Suggestions have been made that the persistence of exotic plant species could be related to high levels of nutrients in the soil, particularly nitrogen (N) and phosphorus (P) (Tilman, 1987; McLendon & Redente, 1991; Hester & Hobbs, 1992; Eliason & Allen, 1997; Zink & Allen, 1998; Baer *et al.*, 2003; Brandon *et al.*, 2004; Leishman & Thomson, 2005; García *et al.*, 2007). This study examined several soil properties, namely electrical conductivity (EC), pH, moisture, available P and total N and investigated their impact on the vegetation cover at 10 and 20 years after abandonment of cultivation. Cover of undesirable species was reduced using autumn and spring burning, herbicide application, as well as a combination of autumn burning and herbicide application.

In this study, electrical conductivity, pH and soil moisture were higher in the 10-year-old field than in the 20-year-old field. The elevated salinity level, which had higher concentration on the ditch (old drainage line) on the younger field, could be the cause of the higher pH observed, and can affect soil water availability. Increased soil salinity has been reported to be a major problem on old agricultural fields in other regions of the world (Pannell, 2001; Cramer & Hobbs, 2002; Cramer *et al.*, 2004; Chhabra, 2005; Cramer *et al.*, 2007; Watt *et al.*, 2007). The high salinity level has been recognised to affect water availability due to osmotic pressure (Castellanos *et al.*, 2005; Cixin He, 2005), thereby disturbing plant nutrition systems and development (Cixin He, 2005; Koyro & Eisa, 2008).

In contrast to the suggestion that increased levels of P and N in the soil might be responsible for the persistence of alien species, levels of these two nutrients were rather low in this study, indicating that these two nutrients do not play a major role in the persistence of alien plant species. Rather, the high soil salinity could one of the main parameters that contribute to the persistence of some undesirable plant species during secondary succession, which is characterised by an abundance of alien forbs and grasses, as well as the indigenous creeping grass *Cynodon dactylon*. *Cynodon dactylon* is the most dominant plant species on both old fields investigated (Shiponeni, 2003; Midoko-Iponga, 2004; Walton, 2006) in this study, as

well as on old agricultural fields in other regions such as Arizona (e.g. Mau-Crimmins, 2007). However, further research is required into the impacts of elevated salinity levels on indigenous seed germination and growth in the west coast vegetation types. Other studies also refuted the notion that P and N are the main soil properties shaping secondary vegetation on old agricultural fields (e.g. Bechtold & Inouye, 2007; Henkin *et al.*, 2006; Holmes, 2008).

As shown in this study, the impact of different soil properties or combinations thereof on vegetation cover and secondary succession will depend on the age of the old fields as well as the land-use history (McLendon & Redente, 1991; Seabloom *et al.*, 2003; Kulmatiski *et al.*, 2006). This is likely due to plant-soil feedback mechanisms that drive the change in dominant vegetation and soil properties (Evans *et al.*, 2001; Mack *et al.*, 2001; Ehrenfeld, 2003; Corbin & D'Antonio, 2004; Hawkes *et al.*, 2005; Domènech *et al.*, 2006).

A combination of autumn burning and herbicide application was best in reducing alien plant species, while at the same time facilitating the establishment and growth of native geophyte and forb species. Although the use of herbicide alone produced a better result in reducing alien species, repeated applications could cause the development of herbicide-resistance such as is already a problem in the Western Cape (Bromilow, 2001; Hashem *et al.*, 2001). Combining herbicide application with another treatment, such as burning (this study), or ploughing (Holmes, 2005) is preferable, as a combination of treatments has been reported to reduce the development of herbicide-resistant plants species (Hashem *et al.*, 2001; Diggle *et al.*, 2003; Vila-Aiub *et al.*, 2005). However, a combination of autumn burning and herbicide application has been shown to affect native species negatively (Pywell *et al.*, 1995; Matarczyk *et al.*, 2002), therefore, the use of this treatment combination needs to be carefully considered. Timing of the treatment application also seems to play a role. In our study, burning of young stage succession in mid-spring reduced alien grasses could be satisfactorily, but it could enhance the most dominant undesirable plant *C. dactylon* and few native among forbs and geophytes as well as it could inhibit other native plant species.

5.2. The way forward

Until recently, old-field restoration, particularly in South Africa, has been focussed mostly on the removal of alien plants species (e.g. Midoko-Iponga, 2004; Holmes, 2005; Musil *et al.*, 2005; Mau-Crimmins, 2007). However, the results of this and other studies have related the persistence of alien species as well as poor re-establishment of native plant species on soil properties changes after cultivation (Zink & Allen, 1998; Cione *et al.*, 2002; Cramer *et al.*, 2007; Holmes, 2008). Richardson *et al.* (2000) proposed a model of different barriers that alien species cross in order to become invasive. However, native species also cross the same

barriers in order to re-establish on an agricultural field after cessation of cultivation. These barriers that need to be overcome include dispersal mode and distance, habitat disturbances, and soil condition, as indicated in old field community assembly model (Cramer, 2007), before they can successfully establish on old fields (Seabloom *et al.*, 2003; Cramer *et al.*, 2007). Additionally native species may also require a “recruitment window” (e.g. bare ground following fire or other disturbances) in order to establish. In the case of Swartland Alluvium Fynbos and Swartland Shale Renosterveld, it has been reported that most of the plant species that are found on old fields are dispersed by wind or large herbivores (Shiponeni, 2003); the survival and growth of those seeds will also depend on quantity of seed. For instance, taken collectively, *Elytrophapus rhinocerotis* and *Tribolium hispidum* have the highest density of native plant species by a sizeable margin and thus they also have the highest potential to re-establish on old fields (Shiponeni, 2003; Krug, 2008). Other grasses (e.g. *Ehrharta* sp., *Pentaschistis airoides*, *Ficinia* sp.) and geophytes (e.g. *Oxalis versicolor*, *O. purpurea* and *Ornithoglossum* sp.) dispersed by animals could also re-establish themselves due to their presence in the seed bank.

Therefore to overcome the rest of the barriers to re-establishment, old field restoration needs to remove the undesirable plant species (e.g. *C. dactylon*) that compete with most native life forms (Midoko-Iponga, 2004; this study), and also change certain areas’ soil properties. Favourable soil properties in, for example, a ditch (old drainage line), which forms an effective trap for wind-dispersed seeds, have the potential to create microclimates to enable the growth of native plant species (Krug, 2008). These microhabitats should also be sown with life forms such as forbs and other species of the West Coast lowland vegetation mosaic that are seed-limited (Shiponeni, 2003). Plants establishing in these microhabitats will assist in spreading seed farther into the old fields (Krug, 2008).

In addition, the use of treatments such as burning and herbicide application to remove alien species, which have been shown to inhibit the re-establishment and slow the re-growth of native species (Midoko-Iponga, 2004; Musil *et al.*, 2005; Holmes, 2005; 2008), should also be combined with treatments that ameliorate soil condition. Salt leaching and gypsum application has been recognised to decrease levels of salinity in the soil (Browning *et al.*, 2006), and gypsum has also been recorded to decrease phosphorus (Suding *et al.*, 2004; LeJeune *et al.*, 2006). However, the effect of gypsum on soil properties and on lowland vegetation types in the fynbos biome needs to be investigated to determine its suitability in old field restoration. Furthermore, nitrogen levels have been reported to be reduced by using carbon, sawdust or sucrose (Paschke *et al.*, 2000; Baer *et al.*, 2003; Baer *et al.*, 2004). Thus,

restoration of old fields should also aim to make soil conditions more favourable for the establishment of indigenous species.

To restore old fields successfully, both above- and below-ground conditions of the old field must be considered. Therefore, a combination of treatments should be applied to reduce the cover of alien plant species with ameliorated soil conditions. Such improved soil conditions make them more suitable for reintroduction of native species to old agricultural lands. Figure 5.1 presents a conceptual model that could guide old-field restoration in the Cape Floristic Region, using weed control by trial treatments and soil reclamation through application treatments, as discussed in this study.

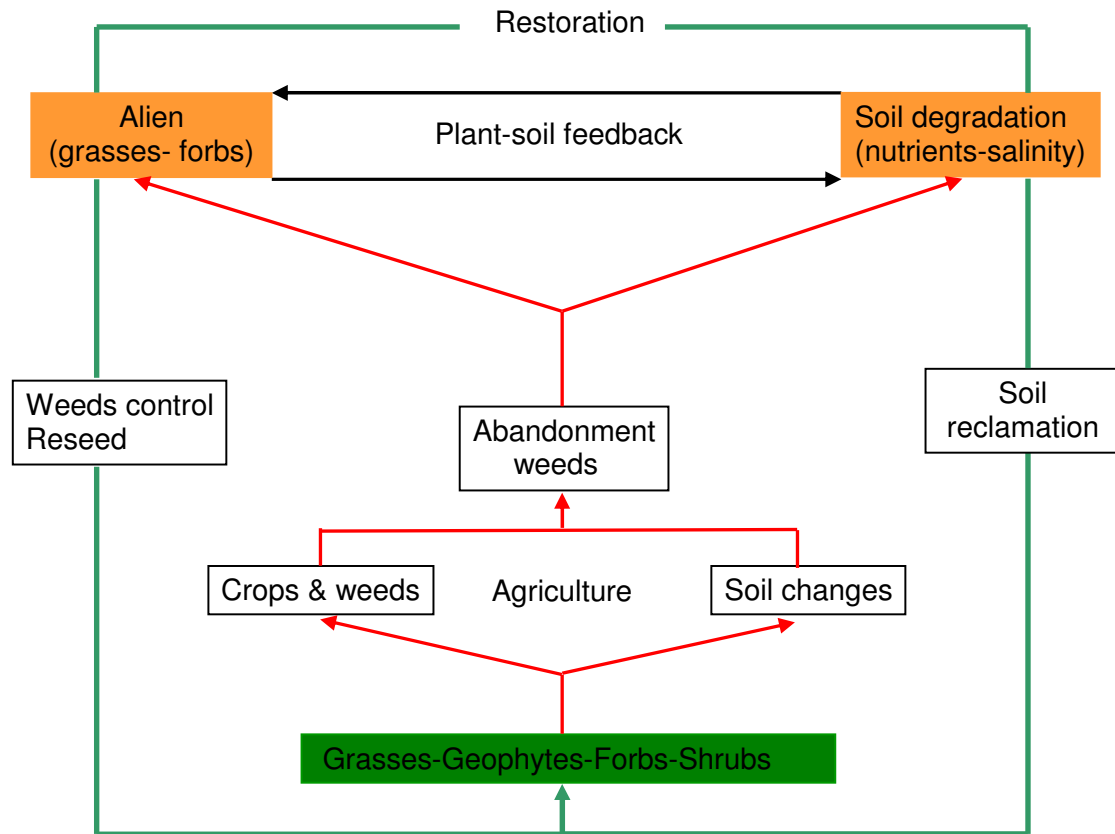


Figure 5.1 Conceptual state and transition model for old agricultural land restoration. The red arrows indicate the degradation of natural vegetation and soil properties, while green arrows indicate the potential effect of above and below ground treatments on re-establishment of native plant species.

5.3 Recommendations for future restoration attempts

For future restoration attempts of old fields (in the CFR), we recommend that:

- Any restoration of an old agricultural field should begin with a preliminary study. This preliminary study should provide an overview of the main alien plant species that occur in a specific area, as well as the soil condition in comparison to natural areas;
- Based on the results of the preliminary study, the treatment most appropriate to removing the alien plants while still facilitating the re-establishment of indigenous species, should be chosen. Potential treatments could be a combination of herbicide application with burning or mowing, and soil amelioration using mulch;
- As the chosen herbicide, a post-emergent herbicide would be preferable to a pre-emergent herbicide, as this also has the potential to contaminate the soil (Bromilow, 2001; Zaady, Levacov & Schachak, 2004; Musil *et al.*, 2005);
- A combination of herbicide application with spring burn could be the most preferable treatment, as it will have less negative impact on indigenous species;

- Restoration treatments of old agricultural fields should be carried out for a period of at least three years, during which the above-ground treatments should be applied the first year, and repeated whenever necessary thereafter. At the same time, a monitoring programme needs to be established to determine the effectiveness of the treatments. A suitable monitoring period could be every 6 months, or monthly during the growing season. Soil amelioration needs to be repeated according to changes in soil properties and the monitoring will depend on each soil property. The soil should be monitored seasonally for salinity problem and once annually for pH and soil nutrients.
- Seeds of indigenous species, especially species with short dispersal distances, should be sown to overcome any re-establishment barriers (Shiponeni, 2003; Midoko-Iponga, 2004; Holmes, 2005; 2008). However, it might be difficult to obtain sufficient seeds of indigenous species. This may be overcome by community or eco-tourist participation to collect seeds from natural vegetation. Nurseries should be set up, also, to grow indigenous species.

5.4. References

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Appendix

Appendix A.

Percentage of sand elements in soil texture of D (ditch), S (slope), and R (ridge) on old fields at Elandsberg Private Nature Reserve.

		Percentage of sand			
Sites	Habitats	Depths (cm)	Coarse	Medium	Fine
Old Field	D	0-10	16.44	36.99	34.93
		10-20	17.80	36.44	32.20
		20-30	19.29	31.43	33.57
	S	0-10	19.61	30.72	37.25
		10-20	13.46	35.26	38.46
		20-30	17.86	32.86	37.86
	R	0-10	8.05	47.13	37.93
		10-20	6.10	34.15	48.17
		20-30	8.07	39.13	42.24
Young Field	D	0-10	23.24	34.51	36.62
		10-20	36.73	36.73	24.49
		20-30	36.67	38.33	20.00
	S	0-10	24.00	68.00	16.00
		10-20	34.78	32.61	25.00
		20-30	42.59	26.85	25.00
	R	0-10	28.35	41.24	25.26
		10-20	33.33	39.25	23.66
		20-30	36.69	35.25	20.86

Appendix B

List of plant species identified from recorded species on old agricultural fields at Elandsberg Private Nature Reserve. Life form, species & genus, as well as family are ascribed according to Germishuizen & Meyer (2003). The undesirable plant species are represented by *, and √ indicates the presence on young field and old field.

Life form	Species	Family	Young field	Old field
Grasses	<i>Avena fatua</i> L.*	Poaceae	√	√
	<i>Briza maxima</i> L.*	Poaceae	√	√
	<i>Briza minor</i> L.*	Poaceae	√	√
	<i>Bromus catharticus</i> Vahl*	Poaceae	√	
	<i>Bromus pectinatus</i> Thunb.*	Poaceae	√	
	<i>Cynodon dactylon</i> (L.) Pers.*	Poaceae	√	√
	<i>Hordeum murinum</i> L.*	Poaceae	√	√
	<i>Imperata cylindrica</i> (L) Raeusch	Poaceae	√	√
	<i>Lolium perenne</i> L.*	Poaceae	√	√
	<i>Pentaschistis curvifolia</i> (Schrad.) Stapf	Poaceae	√	
	<i>Pentaschistis densifolia</i> (Nees) Stapf	Poaceae		√
	<i>Pentaschistis pallida</i> (Thunb.)	Poaceae	√	√
	H.P.Linder			
	<i>Tribolium hispidum</i> (Thunb.) Desv.	Poaceae	√	√
<i>Vulpia myuros</i> (L.) C.C. Gmel.*	Poaceae	√		
Forbs	<i>Arctotheca calendula</i> (L.) Levyns	Asteraceae	√	√
	<i>Aristea africana</i> (L.) Hoffmanns	Iridaceae	√	√
	<i>Conyza bonariensis</i> (L.) Cronquist*	Asteraceae	√	
	<i>Cotula coronopifolia</i> L.	Asteraceae	√	
	<i>Cotula turbinata</i> L.	Asteraceae	√	√
	<i>Dischisma capitatum</i> (Thunb.) Choisy	Scrophulariaceae	√	√
	<i>Drosera trinervia</i> Spreng.	Droseraceae	√	√
	<i>Erodium botrys</i> (Cav.) Bertol.*	Geraniaceae	√	√
	<i>Felicia bergeriana</i> (Spreng.) O. Hoffm.	Asteraceae	√	√
	<i>Ficinia indica</i> (Lam.) Pfeiff.	Cyperaceae		√
	<i>Hypochaeris radicata</i> L.*	Asteraceae	√	
	<i>Rumex cordatus</i> Poir.	Polygonaceae		√
	<i>Trifolium repens</i> L.*	Fabaceae	√	
	<i>Ursina anthemoides</i> (L.) Poir.	Asteraceae	√	√
Geophytes	<i>Cyanella hyacinthoides</i> L.	Tecophilaeaceae	√	√
	<i>Geissorhiza confusa</i> Goldblatt	Iridaceae	√	√
	<i>Moraea miniata</i> Andrews	Iridaceae		√
	<i>Moraea umbellata</i> Thunb.	Iridaceae	√	
	<i>Moraea villosa</i> (Ker Gawl.)	Iridaceae	√	
	<i>Ornithogalum thyrsoides</i> Jacq.	Hyacinthaceae	√	√
	<i>Oxalis commutata</i> Sond.	Oxalidaceae	√	√
	<i>Oxalis pes-caprae</i> L.	Oxalidaceae	√	
	<i>Oxalis purpurea</i> L.	Oxalidaceae	√	√
	<i>Oxalis versicolor</i> L.	Oxalidaceae	√	√
	<i>Satyrium erectum</i> Sw.	Orchidaceae	√	

Appendix B

List of plant species identified from recorded on old agricultural fields at Elandsberg Private Nature Reserve. Life form, species & genus, as well as family are ascribed according to Germishuizen & Meyer (2003). The undesirable plant species are represented by *, and √ indicates the presence on young field and old field.

Life Form	Species & Genus	Family	Young field	Old field
Geophytes	<i>Spiloxene capensis</i> (L.) Garside	Hypoxidaceae	√	
	<i>Spiloxene flaccida</i> (Nel) Garside	Hypoxidaceae	√	
	<i>Wurmbea recurva</i> B. Nord.	Colchicaceae	√	
Shrubs	<i>Aspalathus Ciliaris</i> L.	Fabaceae		√
	<i>Elytropsappus rhinocerotis</i> (Thunb.) Fourc.	Asteraceae		√
	<i>Eriocephalus africanus</i> L.	Asteraceae		√
	<i>Eriocephalus punctulatus</i> DC.	Asteraceae		√
	<i>Leysera gnaphalodes</i> (L.) L.	Asteraceae		√
	<i>Metalasia muricata</i> (L.) D.Don	Asteraceae		√
	<i>Stoebe plumosa</i> (L.) Thunb.	Asteraceae		√
Dwarf shrubs	<i>Helichrysum asperum</i> (Thunb.) Hilliard & B.L. Burt	Asteraceae	√	√
	<i>Hermannia scabra</i> Cav.	Sterculiaceae		√
Hydrophyte	<i>Crassula natans</i> Thunb.	Crassulaceae		√

