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The conservation value of abandoned croplands in Mpumalanga's grasslands

by

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Abstract

The South African grassland biome is one of the most transformed and least protected biomes in the country. The conservation of this species-rich biome is further complicated by the occurrence of abandoned croplands in areas classified as 'natural'. In fact, more than 10 % of the areas classified as natural in Mpumalanga are abandoned croplands. Although it is recognized that they differ from pristine grasslands in species richness and diversity, few studies have assessed the value of abandoned croplands for conservation. The aim of this study was to determine the value of abandoned croplands for conservation in Mpumalanga's grasslands in terms of species composition and landscape connectivity. In the first part of the study the species composition of abandoned croplands was compared to that of pristine natural grassland in the Nooitgedacht Dam Nature Reserve in Mpumalanga. In the second part of the study the contribution of abandoned croplands to overall landscape connectivity in the grassland biome of Mpumalanga was evaluated. It was found that there is a significant difference in species composition, especially for resprouting forb species, between abandoned croplands and pristine natural grasslands. There were also a significant difference in total species richness and forb species richness, while alien plant species richness was significantly higher in abandoned cropland plots. There was no significant difference in medicinal plant species richness. It could be seen that, although different to pristine natural grasslands, the vegetation on these abandoned croplands was not degraded, and can be valuable providers of ecosystem services such as medicinal plants and thatching grass, and can also serve as habitat for different species. It was found that the pristine natural grassland patches in Mpumalanga are already relatively well connected and that abandoned croplands further improved the overall landscape connectivity of grassland habitat patches by 33 %. The results indicated that abandoned croplands have a definite value for conservation by contributing to species richness and connecting the landscape.

Declaration

I declare that the dissertation/thesis, which I hereby submit for the degree M.Sc. Plant Science at the University of Pretoria, is my own work and has not previously been submitted by me for a degree at this or any other tertiary institution.

I further declare that chapter 3 of this dissertation has been submitted to the journal *Austral Ecology.* As a result, content overlap may occur.

L. Fourie

15 July 2014

Signature **Date**

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Chapter 1: Introduction

1.1 The South African Grassland Biome

The South African grassland biome covers about 349 174 square kilometres in the central parts of South Africa (O'Connor and Bredenkamp 1997). It forms part of the global temperate grassland biome, which includes the grasslands of Europe and Asia, the American prairies, and the temperate grasslands of Argentina, Uruguay, Australia and New Zealand (Mucina *et al.* 2006).

In South Africa, the grassland biome is characterised by short vegetation dominated by grasses, and the relative absence of trees. Rainfall ranges from 400 – 2500 mm per year and falls mainly during the summer. Winters are dry with a common occurrence of frost (Mucina *et al.* 2006). The extent of the grassland biome is limited mainly by the relationship between moisture availability and temperature, and also by the interplay of climate, topography, fire and grazing (O'Connor and Bredenkamp 1997). Grassland vegetation is structured by several forces, including competition, grazing, soil type and fire. Grassland is high in species diversity and has 3 378 plant species in the core region (Bredenkamp 2006). If measured by the number of species occurring in 1 000 m^2 sample areas the grassland contains more species than the fynbos biome (Bredenkamp 2006). Forbs are considered to make a very large contribution to the species richness of the grassland (Mucina *et al.* 2006).

Conservation of the grassland biome is threatened by mining, urban development, agriculture, overgrazing, plantation forestry and climate change. According to National Land Cover data, 45 % of the grassland biome area has been transformed, degraded, or severely invaded by alien plants (Neke and Du Plessis 2004). The integrity of the remaining semi-pristine grassland is questionable, as the satellite imagery used does not distinguish between natural grassland and secondary grassland such as abandoned croplands. The portion of grasslands classified as natural but, which are in fact abandoned croplands, are as high as 80 % in the rural areas of the Eastern Cape (Hoare 1997), and 10% in Mpumalanga (Fourie 2010). These secondary grasslands on abandoned croplands are different to pristine grasslands in species richness and composition (Roux 1966; van Oudtshoorn *et al.* 2011;

Zaloumis and Bond 2011), but whether these abandoned croplands are of value to conservation, is still unknown.

1.2 Restoration of abandoned croplands

The abandonment of croplands is a type of land use change that has been exercised by humans since the first agricultural practices. There has been a dramatic increase in the amount of abandoned croplands worldwide, caused mostly by ecological, social or economic change (Cramer and Hobbs 2007). Understanding the vegetation dynamics and plant communities present on these abandoned croplands is becoming increasingly important for conservation management. The alteration of soil properties and biomass causes changes in the recovering vegetation. This either results in a series of successional changes that ultimately lead to vegetation similar to the pre-cultivation conditions (van Aarde *et al.* 1996; Hermy and Verheyen 2007), or cause a more permanent change that can be seen hundreds of years after cropland abandonment (Cramer *et al.* 2008). Cramer *et al.* (2008) grouped vegetation response to cropland abandonment in three types of successional trajectory: *i*) a broadly repeatable successional trajectory; *ii*) a novel or delayed successional trajectory; *iii*) or a persistent degraded state, which shows little resemblance to a natural state. Abandoned croplands of type *ii* and *iii* may be in stable alternate ecosystem states, which are maintained through irreversible changes in ecosystem properties (Cramer *et al.* 2008).

There is limited information on old-field succession in the South African grassland biome, with only three main studies available (Roux 1966; van Oudtshoorn *et al.* 2011; Zaloumis and Bond 2011). It has been found that the vegetation structure on abandoned croplands can be returned with little effort, and even ecosystem services such as forage production can be restored (van Oudtshoorn *et al.* 2011). However, abandoned croplands show little resemblance, in terms of species diversity and composition, to pristine primary grasslands (Roux and Warren 1963; Roux 1966; Roux 1970; Zaloumis and Bond 2011), and forb species richness in recovered areas are affected negatively (van Oudtshoorn *et al.* 2011). There is a serious shortage of studies on the extent to which vegetation on abandoned croplands returns to resemble pristine natural grassland in the South African grassland biome. The value that these abandoned croplands may still have for conservation, such as the provision of ecosystem services and connecting pristine grassland patches is still unknown.

1.3 Abandoned croplands and landscape connectivity

One of the often overlooked values of abandoned croplands may be their role in connecting pristine grassland patches in the grassland biome. The South African grassland biome is highly fragmented, with only 4 % of the remaining natural patches larger than 100 km^2, therefore the extent and locations of abandoned croplands may make them valuable in connecting the landscape. Fragmentation of ecosystems results in landscapes with patches that are more or less isolated from each other and the rest of the landscape, while changing its flow of nutrients, wind, water and radiation (Saunders *et al.* 1991). Habitat patches in a fragmented landscape are isolated from habitat patches in the surrounding landscape to various degrees, and habitat fragmentation intensifies the effects of habitat loss (Fahrig 2003). Organisms have different capabilities of moving between patches. The degree of movement of organisms or processes in a landscape is called connectivity which is responsible for maintaining viable populations in fragmented landscapes (Crooks and Sanjayan 2006). Quantifying connectivity is essential to inform conservation plans and management decisions (Calabrese and Fagan 2004).

Recent advances in connectivity measures based on graph theory have made it possible to quantify connectivity for large landscapes with thousands of habitat patches. Metrics based on graph theory are used to analyse the landscape as a set of habitat patches and connections between habitat patches. In addition to the ability for analysing large landscapes these metrics also allow for the quantification of the contribution of an individual habitat patch for overall connectivity (Saura and Pascual-Hortal 2007; Saura and Torné 2009). This means that for the first time the value of abandoned croplands for overall landscape connectivity can be quantified.

1.4 Aims

This study aims to evaluate the value of abandoned croplands in the grassland biome in Mpumalanga for conservation. The value of abandoned croplands was evaluated first in terms of species composition by comparing the species composition of abandoned croplands to pristine natural grassland at the Nooitgedacht Dam Nature Reserve in Mpumalanga. Secondly, the value of abandoned croplands was evaluated for overall landscape connectivity in Mpumalanga's grasslands using connectivity indices based on graph theory.

1.5 Overview of the dissertation

This dissertation is divided into four chapters: 1) a general introduction (this chapter), 2) an investigation of the conservation value of abandoned croplands on the Nooitgedacht Dam Nature Reserve, 3) quantification of the connectivity between pristine grassland habitat patches and the contribution of abandoned croplands to overall connectivity in Mpumalanga's grasslands and 4) a general conclusion highlighting implications for management and conservation of the Grasslands biome.

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Chapter 2: Species composition and conservation value of secondary grassland after 30 years of cropland abandonment

Abstract

Grasslands have the highest conservation risk of the world's biomes due to its high rate of habitat loss and low protection. The South African grassland biome is notably high in species diversity, but is one of the most transformed and least protected biomes in the country. The biome may be more threatened than realised due to the extensive presence of abandoned croplands in areas classified as 'natural'. These abandoned croplands may have little resemblance to pristine grassland in terms of species diversity and composition, and their conservation value is unknown. The aim of this study was to investigate the conservation value of 30 year old abandoned croplands on the Nooitgedacht Dam Nature Reserve in Mpumalanga by comparing the species composition of abandoned croplands to that of pristine grassland. The results showed a distinct difference in community composition between abandoned cropland and pristine grassland plots. There were also significantly more resprouting forb species on pristine grassland plots than on abandoned croplands. Contrary to the findings of other studies, there were no significant differences in the frequency of forbs, medicinal plants and alien plant species between abandoned croplands and pristine grassland. Pristine grassland plots were more similar to each other in terms of species composition than abandoned croplands. Although the species composition of pristine grasslands does not seem to return after 30 years of cropland abandonment, there are some of the properties and functions of pristine grasslands that do return. The abandoned croplands studied have relatively high species diversity, contribute to landscape heterogeneity and are still valuable providers of important ecosystem services.

2.1 Introduction

The grassland biome occurs on all continents except Antarctica (Gibson 2009). Parts of these grassy biomes have long been viewed as anthropogenic or successional (Bond and Parr 2010), but are in fact very old, and have been present for tens of thousands of years in some places (Mayle *et al.* 2007). Even though they may not be of anthropogenic origin, grasslands are climatically suitable for intensive human settlement and agriculture, and have a history of human impacts (Henwood 1998). Of the world's biomes, grasslands have the highest conservation risk due to its high rate of habitat loss and low level of protection (Hoekstra *et al.* 2005).

The South African grassland biome is part of the global temperate grassland biome, which includes the grasslands of Argentina, Uruguay, Australia and New Zealand (Mucina *et al.* 2006). It occurs mainly in the central parts of the country, as well as areas close to the east coast, the mountains of KwaZulu-Natal and the central parts of the Eastern Cape (Mucina *et al.* 2006). It is believed that the South African grassland biome originated during climatic change during the Oligocene and became widespread in a climate that was 5°C colder than the current temperature. This colder climate created a suitable environment for the replacement of subtropical forests and woodlands by grasslands (Bredenkamp *et al.* 2002). The extent of this biome is limited mainly by the relationship between moisture availability and temperature, and also by the interplay of climate, topography, fire and grazing (O'Connor and Bredenkamp 1997). In South Africa, two types of grasslands are distinguished: Highveld grasslands dominated by C_4 grasses and montane grasslands dominated by C_3 grasses (Mucina *et al.* 2006).

The South African grassland biome is notably high in species diversity, with 3 378 plant species occurring in the core region (Bredenkamp 2006). The biome contains five centres of plant endemism; Drakensberg, Alpine, Barberton, Wolkberg, Sekhukhune and Soutpansberg (Van Wyk and Smith 2001). The C_3 montane grasslands are particularly rich in endemic plant species (Bredenkamp *et al.* 2002). The grassland biome also contains 10 endemic bird species and is an important habitat for 10 of the 14 globally threatened bird species (Neke and Du Plessis 2004). In addition to high species diversity, grassland vegetation plays an important role in sustaining human life by providing key ecosystem services such as grazing, carbon sequestration, and collection of medicinal plants, edible plants and thatch grass (Driver *et al.* 2012). The value of natural grazing in the grassland biome has been valued at over R 8 000 per square kilometre per year (Blignaut *et al.* 2008). Grasslands reduce runoff and erosion and play a crucial role in the hydrological cycle. The South African grassland biome also proves to be one of the biomes with the highest number of medicinal plant species (Driver *et al.* 2012).

The high species diversity and provision of critical ecosystem services make the South African grassland biome one of the most important biomes in the country. However, it is also one of South Africa's most under-protected biomes (Driver *et al.* 2012), and 39.2 % of the biome has been irreversibly transformed (Neke and Du Plessis 2004). Habitat transformation has mainly been a consequence of cultivation (23%), and is also the result of plantation forestry (4%) urbanisation (2%) and mining (1%) (Neke and Du Plessis 2004). The integrity of the remaining semi-pristine grassland is also questionable, as the satellite images do not distinguish between pristine natural grasslands and unnatural grasslands such as planted grasslands and abandoned croplands (Neke and Du Plessis 2004). In Mpumalanga, 16% of Highveld grasslands and 6% of montane grasslands, classified as natural by land cover datasets, are in fact abandoned croplands (Fourie 2010). This portion is even higher in the rural areas of the Eastern Cape, where it has been found that up to 80% of vegetation classified as natural grasslands are in fact abandoned croplands (Hoare 1997).

The classification of abandoned croplands as 'natural grassland' is problematic because the restoration of grasslands, and of the South African grasslands in particular, are reported to be very slow, with restored grasslands showing little resemblance to pristine primary grasslands in species diversity and composition (Roux 1966; Bond and Parr 2010; Zaloumis and Bond 2011). Ecological restoration is defined by the Society for Ecological Restoration International as the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. Restored ecosystems contains assemblages of species characteristic to the reference ecosystem, consist of all functional groups necessary for the stability of the restored ecosystem and are capable of sustaining itself structurally and functionally (Society for Ecological Restoration International Science & Policy Working Group 2004). Rehabilitation differs from restoration by emphasizing the re-establishment of ecosystem processes and services, where restoration has the additional goal of re-establishment of pre-existing species composition and community structure. Revegetation is a component of land reclamation and describes the establishment of a vegetation cover, which may consist of only one or few species (Society for Ecological Restoration International Science & Policy Working Group 2004). The restoration of the complete species diversity after cropland abandonment in Highveld grasslands has not been seen in experiments of up to 30 years (Roux 1966). It is mainly forb species that do not return (Zaloumis and Bond 2011), as there is a trade-off between persistence and colonizing ability and many forb species have traits aimed at persistence, such as underground storage organs (Bond and Midgley 2001).

Although abandoned croplands and old field succession have been relatively well-studied overall (for a comprehensive bibliography, see Rejmánek and van Katwyk 2005), there are very few studies available for the South African Grassland biome. Secondary grasslands resulting from cropland abandonment may reduce the conservation value and biodiversity of the already endangered Grassland biome. However, the conservation value of abandoned croplands in the South African grassland biome is still unknown. There is therefore an urgent need in terms of practical biodiversity conservation to determine the value of these abandoned croplands for conservation.

The aim of this study was therefore to investigate the conservation value of abandoned croplands on the Nooitgedacht Dam Nature Reserve in Mpumalanga. This was done by comparing the species composition of abandoned croplands with the species composition of pristine grassland sites. The following questions were asked:

(a) Is there a difference between the species composition of 30 year old abandoned croplands and pristine grasslands on similar sites at the Nooitgedacht Dam Nature Reserve?
(b) Are there differences in the presence of specific traits that can contribute to ecosystem services associated with species present on abandoned croplands?

2.2 Methods

2.2.1 Study area

The study was conducted at the Nooitgedacht Dam Nature Reserve near Carolina in Mpumalanga (Figure 2.1). The Nooitgedacht Dam was built in 1962 and the surrounding farms were proclaimed as a nature reserve in 1980 (Karl Naudé, Department of Environmental Affairs, pers. comm). It is unclear exactly when during the period 1962 to 1980 the cultivation practices on the croplands were stopped, but it is certain that the abandoned croplands on the reserve are now more than 30 years old. The Nooitgedacht Dam Nature Reserve contains vegetation of the Eastern Highveld Grassland and the KaNgwane Montane Grassland vegetation types (Mucina and Rutherford 2006). The average annual rainfall is 780 mm and the topography of the reserve undulates between 1 500 and 1 668 m above mean sea level. Soils are dystrophic to eutrophic with widespread red soils and the reserve is mainly grazed by black wildebeest, blesbok, Burchell's zebra, red hartebeest and springbok.

2.2.2 Data collection

Sampling took place in February and March 2012, as these months were logistically the most suitable, as well as still in the summer, for easy identification of plants. However, many plants are only visible for a short specific time in the year, and some plants may therefore have been missed. Locations of abandoned croplands were determined using old aerial photographs and maps (see Chapter 2 and Fourie 2010). Vegetation was sampled in 5 m x 5 m quadrats (plots) located on abandoned croplands ('treatment' sites). Each abandoned cropland site was paired to an adjacent pristine grassland (control) site. Plots had similar fire regimes, which were managed by the reserve management. Where possible, plots were placed near the extant long term monitoring points on the reserve (Coordinates in Appendix A). The grass component on these long term monitoring points are surveyed every two to five years, using the step-point method to determine grazing capacity and to aid in reserve management. However, the data from the long term monitoring on the reserve were not used in this study. Sampling was restricted to the area covered by the Eastern Highveld Grassland vegetation type and 26 plots were sampled on this area (13 plots on abandoned croplands and 13 plots on pristine grassland) (Figure 2.1). Plant species on each of these plots were recorded in a Braun-Blanquet type survey. The abundance of each species in the plot was estimated using the following scale:

r	-	single individual with a low cover
+	-	more than one individual, but not abundant, with a cover <1%
1	-	1 – 5 % cover
2a	-	5 – 12 % cover
2b	-	12 – 25 % cover
3	-	25 – 50 % cover
4	-	50 – 75 % cover
5	-	> 75 % cover

Plant species were identified using field guides (Van Wyk and Malan 1998; Van Oudtshoorn 2006) and the herbarium specimens at the H.G.W.J. Schweickerdt Herbarium of the University of Pretoria. For this study, forb species were defined as all non-grass species. Forbs were placed into different categories according to their medicinal properties and growth form characteristics. Species were categorised as medicinal if they are indicated to have medicinal uses on the South African Biodiversity Information Facility developed by SANBI and the Department of Science and Technology of South Africa (See the SIBIS:SABIF species database which can be accessed at http://sibis.sanbi.org/). Forbs were regarded as resprouters if they have any perennial underground storage organs such as rootstocks, rhizomes, bulbs or corms. Alien plant species were also identified. Species that could not be identified up to a species level remained unnamed but were included in calculations.

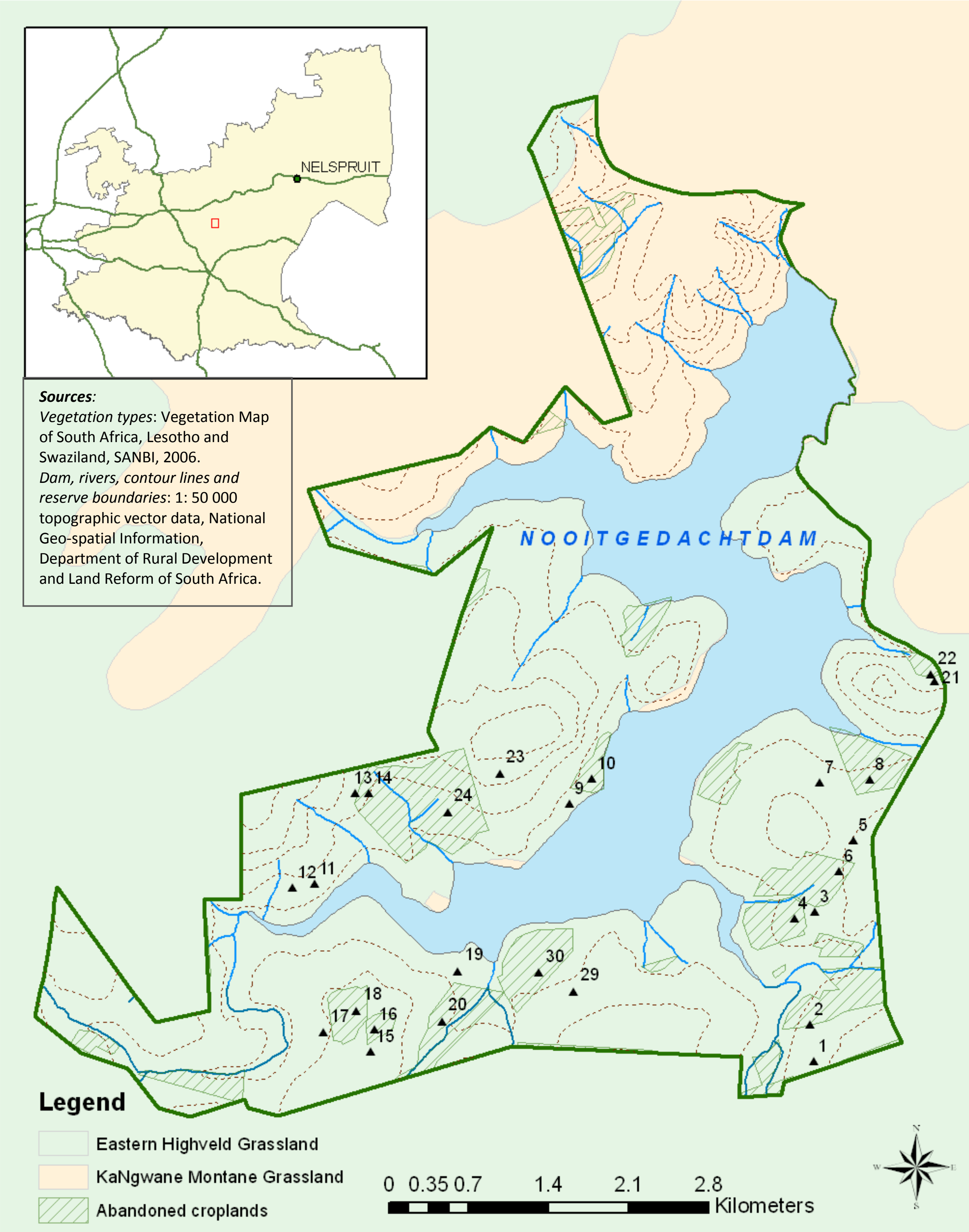


Figure 2.1: Map of the Nooitgedacht Dam Nature Reserve in Mpumalanga showing the locations of abandoned croplands, vegetation types and sites

2.2.3 Data analysis

2.2.3.1 Species richness and diversity

Species richness of all plant species, forb species, resprouting forb species, medicinal plant species and alien plant species on natural grassland and abandoned cropland plots was compared (Figure 2.2). The Wilks-Shapiro statistic was calculated to test for the normality of the frequency distribution of the differences between paired plots in species richness in the above-mentioned categories (Appendix B). As the differences between paired plots in medicinal plant species richness were not normally distributed, the Wilcoxon Signed-rank test was conducted to test for significant differences in all of the categories (Samuels and Witmer 2003). Additionally, the significance of the difference in the number of species exclusive to both natural and abandoned croplands in each of the above categories were tested for with a chi-square test using a two by two contingency table.

2.2.3.2 Similarity

Plots were placed into two different groups; one group containing the 13 pristine natural plots and the other group containing the 13 abandoned cropland plots. Each of these groups were tested for within-group similarities by using the multi-response permutation procedure (MRPP). The multi-response permutation procedure begins by constructing a distance matrix using the Sorensen (Bray-Curtis) distance measure between the plots of each group, producing within-group average distances for each. Next, all sample units are randomly assigned to one of the groups and new distances are calculated. The repetition of this randomization process results in randomized weighted within-group distances. These distances are approximated by a Pearson type III continuous distribution (Mielke and Berry 2001). The comparison of the observed test statistic to the distribution of test statistics obtained through the randomization indicates if sample units belonging to the same groups are significantly more similar to one another than would be expected if they had belonged to other groups. The Sorensen (Bray-Curtis) distance measure was used as it is less prone to exaggerate the influence of outliers than Euclidean distance measures and PC-ORD for Windows, version 6 (McCune and Mefford 2011) was used to conduct the MRPP (McCune and Grace 2002).

2.2.3.3 Ordination and community composition

The plots were ordinated according to community composition of forb and total plant species separately, using nonmetric multidimensional scaling (NMS). NMS was chosen for its independence of a specific distribution type and its flexibility in the choice of distance measure (Peck 2010). The Sorensen (Bray-Curtis) distance measure was used for the ordination which was performed in PC-ORD for Windows, version 6 (McCune and Mefford 2011), with a random starting configuration and 250 runs with real data. From these runs with real data the best starting configuration for two dimensions was used as the starting configuration for the final run. Dimensionality was assessed by the PC-ORD program by comparing the stress of the best solution for each dimensionality. Additional dimensions are considered if they reduce the final stress by 5 or more. For both the forb NMS and total plant species NMS two dimensions has been chosen for the final solution. The instability of the solution was evaluated by PC-ORD through evaluating fluctuations in stress in different iterations. A stability criterion of 0.00010 as standard deviations in stress over the last 15 iterations was chosen. In the final run of the forb species NMS and total plant species NMS 87 and 71 iterations took place respectively. For both NMS's the Monte Carlo test results indicated a 0.0040 proportion of randomized runs with stress smaller or equal to the observed stress.

To test for the influence of location (different places on the reserve) versus land use (abandoned cropland or natural) a permutation-based nonparametric MANOVA (perMANOVA) was carried out using the randomized complete block randomization design (Anderson 2001; McCune and Mefford 2011). In this analysis the matched pairs of plots (one abandoned cropland and one pristine natural) were treated as blocks and land use as groups. The differences among treatments (land use) were tested for by shuffling treatments within blocks (pairs of plots). The Sorensen (Bray-Curtis) distance measure was used. PerMANOVA allows for distance measures other than Euclidean as it averages distances among sample units rather than between sample units and a centroid (Anderson 2001). It also evaluates significance with a permutation test and does not require a normal distribution of data (Anderson 2001). The perMANOVA was performed using PC-ORD for Windows, version 6 (McCune and Mefford 2011).

2.3 Results

2.3.1 Species richness and diversity

A total of 194 plant species were recorded in the 26 plots on abandoned croplands and pristine grassland (Table 2.1; Appendix F). Of these plant species 46 % occurred in both abandoned croplands and pristine grassland, while 30 % were exclusive to pristine grassland. Forb species comprised 80% of the total plant species, with 42 % of forb species occurring in both abandoned croplands and pristine grassland and 34% exclusive to pristine grassland.

Table 2.1: Number of plant species with specific characteristics in total, exclusive to natural, exclusive to abandoned croplands and present in both abandoned cropland and natural grassland plots

	Total species	Exclusive to natural	Exclusive to abandoned croplands	Common in both
Plant species	194	58	46	90
Forb species	156	53	37	66
Resprouting forb species	73	27	11	35
Grass species	38	5	9	24
Medicinal plant species	89	24	23	42
Alien plant species	14	2	6	6
Unidentified plants	24	14	6	4

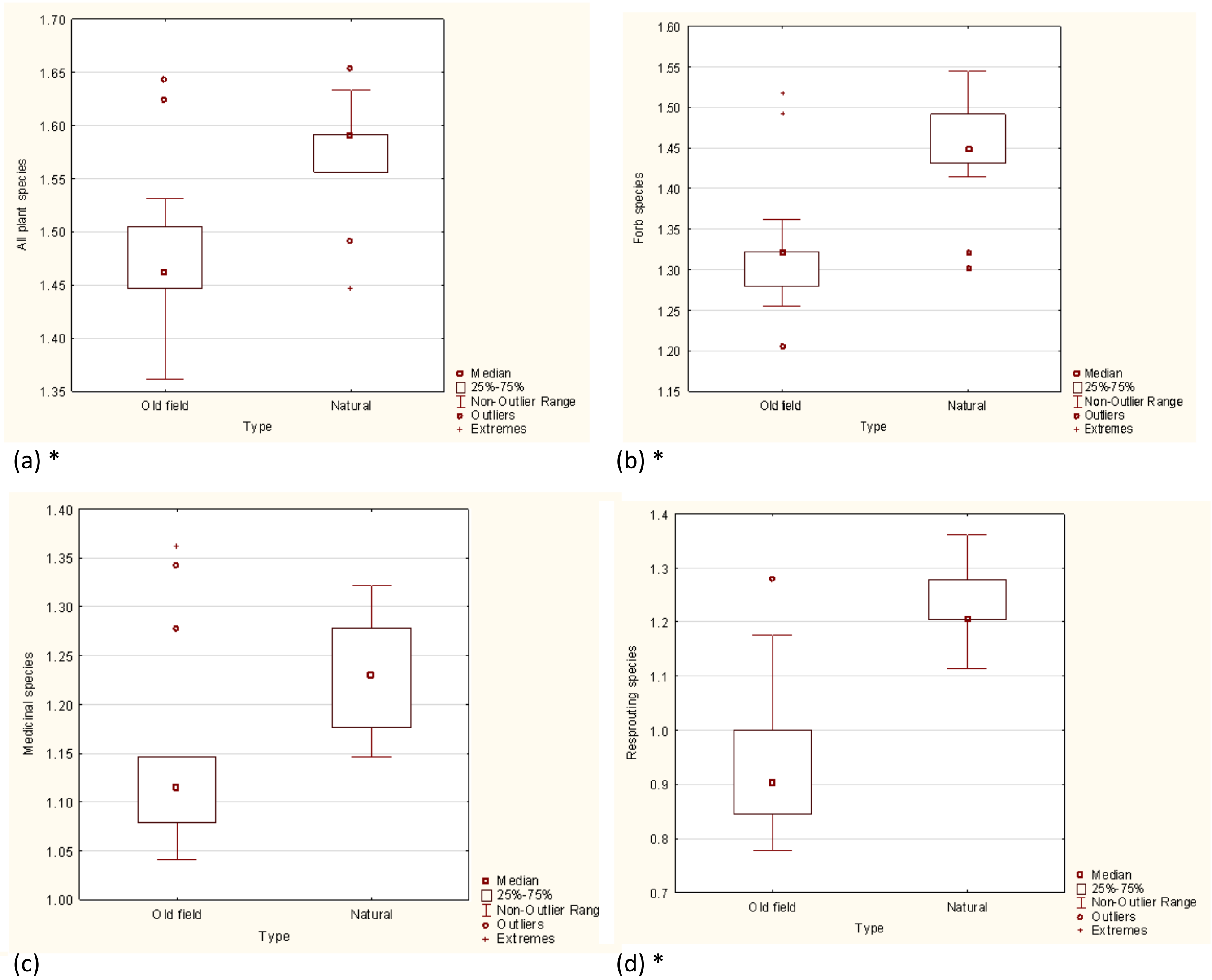


Figure 2.2: Boxplots showing the distribution of a) the total number of plant species, b) forb species, c) medicinal species and d) resprouting forb species observed on abandoned cropland and pristine natural plots. Significant differences are indicated by *(significance according to Wilcoxon Signed-rank test – considered as significant at $p < 0.05$).

Table 2.2: Results of the Wilcoxon Signed-rank tests for significant differences in pristine and abandoned cropland plots in the different categories

Category	W_s test statistic	Directionality	P- value (one tailed)
Total species richness	83	positive	0.005 < *P* < 0.001
Forb species richness	85	positive	0.005 < *P* < 0.001
Resprouting plant species richness	91	positive	*P* < 0.0005
Alien plant species richness	64	negative	0.05 < *P* < 0.025
Medicinal plant species richness	56.5	n/a	*P* > 0.05

The 13 abandoned cropland plots had a total species richness of 136 species, while the pristine natural grassland plots had a total species richness of 148 species (Table 2.1). The Wilcoxon Signed-rank tests indicate that the total species richness, forb species richness, and resprouting plant species richness is significantly higher on the pristine grassland plots than on the abandoned cropland plots (Table 2.2 and Appendix C). The number of alien species on abandoned cropland plots was significantly greater that on pristine grassland plots, and there is no significant difference in the number of medicinal plant species on plots of the two different treatments (Table 2.2 and Appendix C).

According to the Chi-square tests the frequency of resprouting forb species exclusive to pristine grassland plots was significantly greater than the frequency of resprouting forbs exclusive to abandoned croplands (Table 2.1 and Appendix D: $\chi^2_{(1)}$ = 4.02, P<0.05). There was no significant difference in the frequency of forbs (Table 2.1 and Appendix D: $\chi^2_{(1)}$ = 2.64, P>0.05), medicinal plants (Table 2.1 and Appendix D: $\chi^2_{(1)}$ = 0.77, P>0.05) or alien plants (Table 2.1 and Appendix D: $\chi^2_{(1)}$ = 3.33, P>0.05) exclusive to pristine and abandoned croplands sites.

2.3.2 Similarity

The average within group distance of the abandoned croplands group was greater than that of the pristine natural group (Table 2.3). The chance-corrected within-group agreement value of 0.063 indicated that the heterogeneity within groups was greater than expected by chance. This meant that the species composition among natural grassland plots was more homogenous than among abandoned cropland plots.

Table 2.3: Average within-group distance as calculated from the Sorensen distance matrix. Based on a MRPP test, the differences were significant (T= -10.95, A = 0.063, P < 0.001).

	Average within-group distance
Pristine natural	0.703
Abandoned cropland	0.743
Average	0.723

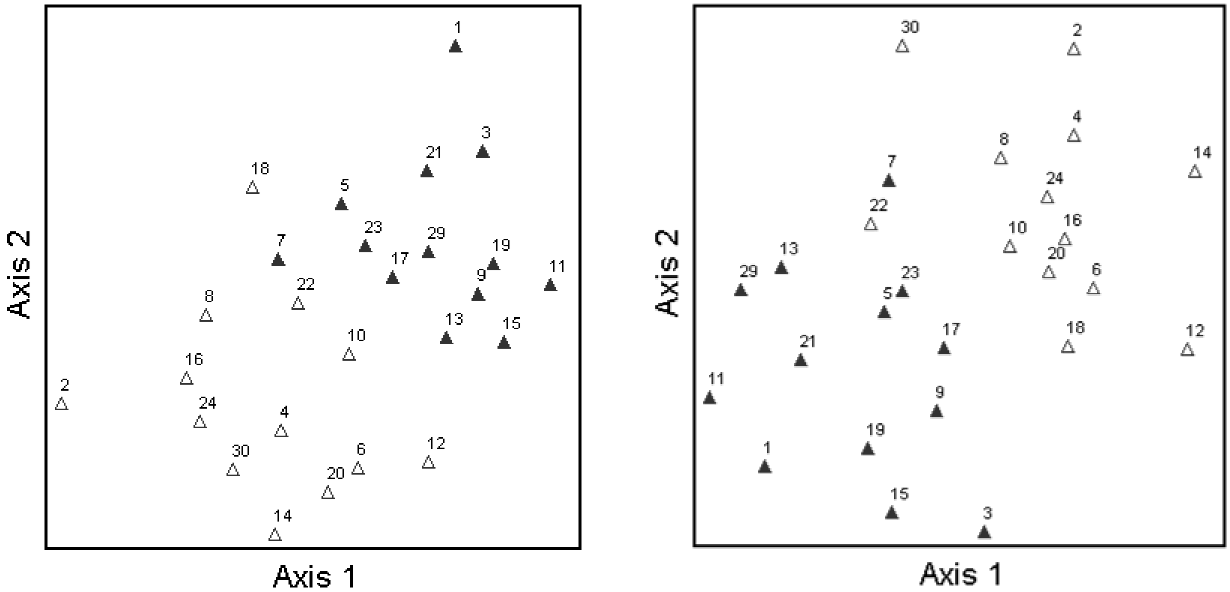
2.3.3 Ordination and community composition

Abandoned croplands and pristine grassland plots were clearly separated into two distinct groups through the NMS ordination (Figure 2.3). The perMANOVA showed a significant difference (F = 5.5309, P<0.01) in species composition between abandoned cropland and natural plots, as well as between blocks consisting of pairs of plots (F = 1.4160, P < 0.01) (Table 2.4). The F ratios indicate a greater difference owing to land use than to location (an F ratio of 1 means that the signal is the same as the noise). The land use components explain 81.92% of the observed variance.

Table 2.4: Blocked PerMANOVA results evaluating the differences in species between groups (abandoned cropland and natural)

Source	d.f.	SS	MS	*F*-value	*p*
Land use	1	1.2093	1.2093	5.5309	0.0008
Location (Blocks)	12	3.7152	0.30960	1.4160	0.0002

Grass species were dominant on all plots with different grass species on the pristine grassland and abandoned cropland sites (See Appendix F for a complete species list). *Themeda triandra* and *Trachypogon spicatus* dominated the pristine grassland plots. *Themeda triandra* occurred exclusively in natural plots, while *Trachypogon spicatus* occurred infrequently in abandoned croplands. The dominant grass species in abandoned cropland plots were *Hyparrhenia hirta, Cymbopogon excavatus,* and *Heteropogon contortus.* These three grass species occurred infrequently on pristine grassland plots as well, but were far more common on abandoned croplands. The pioneer grass species *Aristida congesta,* *Aristida mollissima, Aristida scabrivalvis* and *Paspalum scrobiculatum* were found exclusively on abandoned croplands. There were no pioneer grass species exclusive to pristine natural grasslands. The most common forb species on pristine grassland plots were *Anthospermum rigidum* and *Hermannia transvaalensis*, while the most common forb species on abandoned cropland plots were *Helichrysum polycladum* and *Helichrysum nudifolium*. No threatened species were found and there were 59 species that occurred only once (30 on natural and 29 on abandoned cropland plots).



(a) (b)

Figure 2.3: Nonmetric multidimensional scaling (NMS) ordination graph showing plots separated into abandoned croplands (open triangles) and pristine natural grassland (closed triangles) using total species composition (a) and forb species composition (b). Indication of proportion of variance represented by each axis can be found in Appendix E

2.4 Discussion

This study shows that the community composition of abandoned croplands is still significantly different to pristine grasslands even 30 years after abandonment. There were significantly fewer total plant species, forbs species and resprouting plant species on abandoned croplands than on pristine grasslands. Contrary to the findings of other studies on secondary grasslands (van Oudtshoorn *et al.* 2011; Zaloumis and Bond 2011), this study found that even though the species composition differs significantly, abandoned croplands still have relatively high species diversity (total species richness of indigenous plants on abandoned croplands = 124). The true difference in species richness between abandoned croplands and pristine grassland may however be underestimated in this study, as many grassland forb species are only visible shortly after a fire. Different grassland vegetation types (for example Highveld versus montane) may also respond differently to cultivation and subsequent abandonment.

Differences in community composition between abandoned croplands and pristine grassland have been confirmed by studies on abandoned croplands or restored plantations in different locations in the Grassland biome of South Africa (Roux and Warren 1963; Roux 1966; van Oudtshoorn *et al.* 2011; Zaloumis and Bond 2011). The difference in species composition between abandoned croplands and pristine grassland has also been observed. This was done in phytosociological studies of different areas where disturbed grasslands differ enough form surrounding pristine grasslands that they were recognised as separate communities. These observations were made around Lichtenburg (Bezuidenhout *et al.* 1994), the Suikerbosrand (Bredenkamp and Theron 1980), the Bankenveld (Bredenkamp and Brown 2003) and the Jack Scott Nature Reserve (Coetzee 1974). Bredenkamp and Brown (2003) describe a *Hyparrhenia hirta* Anthropogenic Grassland vegetation type occurring over vast areas of central and southern Gauteng. These *Hyparrhenia* dominated grasslands may appear natural, but are associated with recent or ancient anthropological influences and are characterised by low species richness and low occurrence of forbs (Bredenkamp and Brown 2003). Early studies on the Frankenwald research station of the University of the Witwatersrand focused extensively on investigating the nature of the secondary succession on abandoned croplands. They observed that succession on abandoned croplands proceeds through a series of stages and eventually reaches a stage characterised by dense stands of *Hyparrhenia.* This stage does not resemble pristine undisturbed grassland and after 25 years of observation, a transition from this stage to a stage resembling pristine grassland, has not yet been observed (Davidson 1962). The return of similar species composition to pristine grasslands on a disturbed site in the Grassland biome of South Africa have only been observed once in an iron-age enclosure near the eastern suburbs of Johannesburg (Roux 1966; Roux 1970). This enclosure has either been used as an outer defence for huts or a cattle pen and may be extremely old, but unfortunately the exact time since abandonment is unknown (Roux 1966; Roux 1970).

The significant differences in total number of species, forbs species and resprouting forb species observed in this study is in accordance with a study on grasslands restored after afforestation (Zaloumis and Bond 2011). However, the significant difference in the frequency of medicinal plant species found by Zaloumis and Bond (2011) has not been observed in this study. This difference in findings may be contributed to differences in previous land use, differences in age (17 years since afforestation versus 30 years since cultivation), and different management and environmental conditions.

A possible explanation for the loss of resprouting forb species on abandoned croplands and previously-afforested grasslands could be the proposed trade-off between persistence and seedling recruitment (Bond and Midgley 2001; Bond and Parr 2010). Resprouting plants store resources in underground storage organs, usually at a cost to growth or seed production and seedling survival (Bond and Midgley 2001). It has been determined that climax grasses prefer infertile soils (Roux 1954; Davidson 1962;) and that abandoned croplands have slightly higher levels of available nitrogen (Roux 1966). It has also been proposed that the dense *Hyparrhenia hirta* stands prevent the establishment of the light-demanding climax grasses (Roux 1966).

Active restoration of South African grasslands has achieved limited success. The success of revegetation is documented in mine rehabilitation (Mentis 2006), but full rehabilitation of these sites has not been proved to date. Only one study could be found on active restoration methods of abandoned croplands (van Oudtshoorn *et al.* 2011). This study was conducted on recently abandoned croplands in the Suikerbosrand Nature Reserve, and investigated various seeding methods. The methods tested were mostly successful in establishing grasses normally occurring on pristine grasslands (van Oudtshoorn *et al.* 2011), but less successful in establishing forb species (van Oudtshoorn 2007).

Although the species composition of pristine grasslands does not seem to return after cropland abandonment, there are some of the properties and functions of pristine grasslands that do return. Abandoned croplands may contribute important ecosystem services, for instance, *Hyparrhenia hirta,* the grass species associated with abandoned croplands is used as thatching grass in houses. In this study it was shown that there is no difference in the number of medicinal plants occurring on pristine grassland and abandoned croplands. There is also no difference in the structure of vegetation on pristine grassland and abandoned croplands (Masterson *et al.* 2009) and therefore vegetation on abandoned croplands is also likely to provide the ecosystem services of erosion control and water retention, but further tests are needed to quantify the provisioning of these services.

Although rarely evaluated, abandoned croplands may provide an important habitat for plants and animals. It has however, been determined that croplands, when abandoned for two to six years, support fewer reptile species than equivalent primary grasslands in the Suikerbosrand Nature Reserve (Masterson *et al.* 2009). Through this study, it became clear that the species associated with abandoned croplands differ to those associated with pristine grasslands. As such, abandoned croplands may contribute to landscape heterogeneity, when the landscape consists of a matrix of abandoned croplands and pristine grassland. Abandoned croplands may also play an important role in connecting pristine habitat patches and improving the overall landscape connectivity (see Chapter 3).

There is still no evidence that abandoned croplands in the Grassland biome of South Africa will in time resemble pristine undisturbed grassland and that the original species composition will return. Even though this means that the original biodiversity of pristine grasslands is irreversibly lost, secondary grasslands are still valuable providers of important ecosystem services and should be considered in grassland conservation programmes.

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Chapter 3: Landscape connectivity of the Grassland Biome in Mpumalanga, South Africa

Abstract

The South African Grassland biome is one of the most threatened biomes in South Africa. Approximately 45% of the Grassland biome area is transformed, degraded or severely invaded by alien plants and the remaining natural areas are highly fragmented. In this fragmented landscape, the connectivity between habitat patches is very important to maintain viable populations. The aim of this study was to quantify connectivity of the grassland biome in Mpumalanga using graph theory in order to identify conservation priorities and to direct conservation efforts. Graph theory-based connectivity indices have the ability to combine spatially explicit habitat data with species specific dispersal data and can quantify structural and functional connectivity over large landscapes. These indices were used to quantify the overall connectivity of the study area, to determine the influence of abandoned croplands on overall connectivity, and to identify the habitat patches and vegetation types most in need of maintaining overall connectivity. Natural areas were identified using 2008 land cover data for Mpumalanga. Connectivity within the Grassland Biome of Mpumalanga was analysed for grassland species with dispersal distances ranging from 50 to 1000 metres. The grassland habitat patches were mostly well connected, with 99.6% of the total habitat area connected at a threshold distance of 1000 metres. The inclusion of abandoned croplands resulted in a 33% increase in connectivity at a threshold distance of 500 metres. The habitat patches most responsible for maintaining overall connectivity were the large patches of continuous habitat in the upper and lower centres of the study area and the most important vegetation types were the Wakkerstroom Montane Grassland and the Eastern Temperate Freshwater Wetlands. These results can be used to inform management decisions and reserve design to improve and maintain connectivity in this biome.

3.1 Introduction

The temperate Grassland biome has the highest conservation risk of the world's biomes due to the very high rate of habitat loss and low protection (Hoekstra *et al.* 2005). This biome includes the grasslands of Europe and Asia, the American prairies, the temperate grasslands of Argentina, Uruguay, Australia and New Zealand as well as the South African Grassland biome (Henwood 1998). Historically the most diverse and productive of the world's 15 biomes, its fertile soils and moderate climate has made it one of the best environments for human settlement and agriculture (Henwood 1998).

The South African Grassland biome does not differ significantly from the global trend. High in species diversity, with 3 378 plant species occurring in the core region (Bredenkamp *et al.* 2006), the South African Grassland biome is threatened by mining, urban development, agriculture, overgrazing, plantation forestry and climate change (Neke and Du Plessis 2004). The conservation of the biome is further complicated by the fact that many areas considered as natural are in fact abandoned croplands (Neke and Du Plessis 2004). These abandoned croplands are considered to have lower species richness and especially forb species richness, many of which have not recolonised even after 40 years (Roux 1966, see chapter 2). In Mpumalanga, the biome has been substantially reduced as 44 % has been transformed, mainly through agriculture, plantations and mining (Ferrar and Lötter 2007). The biome is also highly fragmented, with only 4 % of the remaining natural areas bigger than 100 km^2 (Neke and Du Plessis 2004).

Even though habitat loss and fragmentation is seen as the two biggest causes of biodiversity loss worldwide (Wilcox and Murphy 1985; Dirzo and Raven 2003), it is still difficult to separate the effects of habitat fragmentation from the effects of habitat loss (Fahrig 2003). Habitat fragmentation intensifies the effects of habitat loss and can be described as the increased isolation of habitat patches (Fahrig 2003). Habitat connectivity is increasingly used to quantify the isolation of habitat patches through fragmentation (Schumaker 1996), and can therefore be seen as a measure of the effect of fragmentation on the landscape. There is a wide range of definitions and measurements of fragmentation, the choice of which influences our understanding of the effect of fragmentation on biodiversity (Fahrig 2003).

Connectivity refers to the degree of movement of organisms or processes, and is responsible for maintaining viable populations in fragmented landscapes (Crooks and Sanjayan 2006). Connectivity also facilitates juvenile dispersal, recolonization of unoccupied habitat patches, seasonal migration (Hanski 1998), and enables range shifts in response to climate change (Minor and Urban 2008). Quantifying connectivity is therefore essential to inform conservation plans and management decisions (Calabrese and Fagan 2004). However, connectivity measures have not been widely used for conservation planning in South Africa.

Even though an analysis of connectivity of the grassland biome in Mpumalanga is highly necessary to determine and manage the effects of increased habitat fragmentation in this biome, computational limitations previously prevented the quantification of connectivity in this large area. With the recent development of habitat connectivity metrics based on graph theory, it became possible to obtain a detailed quantification of large landscapes such as the grassland biome in Mpumalanga.

The overall aim of this study is therefore to investigate and quantify connectivity of grassland habitat patches in Mpumalanga using graph theory. This is done by: (1) Investigating overall connectivity in Mpumalanga in terms of the number of components, the Integral Index of Connectivity; (2) Investigating the importance of abandoned croplands for maintaining connectivity in the landscape; and (3) Identifying the habitat patches and vegetation types most important for maintaining overall connectivity.

3.2 Methods

3.2.1 Study area

The study was conducted within the South African Grassland biome in the Mpumalanga province of South Africa. Mpumalanga's grasslands occur mainly on fertile soils and the biggest threats to the conservation of the grassland biome is agriculture, plantation forestry, alien plant invasion and open cast mining for coal (Ferrar and Lötter 2007). The biome can be divided into the high-altitude montane grasslands that are dominated by C_3 species and are rich in endemics, and the lower-altitude highveld grasslands that have fewer endemics and are dominated by C_4 species (Mucina *et al.* 2006). The eastern high-rainfall region of the biome is simultaneously the region with the highest diversity of animals and plants, and the area with the highest risk of transformation (Neke and Du Plessis 2004). The grassland biome in Mpumalanga contains a high number of rare and threatened species (Ferrar and Lötter 2007).

3.2.2 Mapping of abandoned croplands

The location of abandoned croplands were determined by digitising the areas mapped as cultivated on the first edition 1:50 000 topographical maps. These first edition topographical maps were obtained from the Chief Directorate: National Geospatial Information of the Department Rural Development & Land Reform of the Republic of South Africa. These maps are generally compiled from aerial photographs, and indicate the locations of, among other things, cultivated lands, orchards and vineyards, trees and bush. The dates on the first edition of these maps range between 1939 and 1986 with a median of 1962. The locations of areas cultivated on the first edition topographical maps were then compared to the 1984 and 2008 land cover datasets of Mpumalanga to identify previously cultivated areas, which are now classified as natural and therefore represent abandoned croplands. The 1984 and 2008 land cover datasets used were produced for the Mpumalanga Parks Board by GeoTerraImage (Pty) Ltd. These land cover datasets distinguish between mined areas, cultivated areas, urban areas, afforested areas and untransformed 'natural' areas and were mapped from Landsat 5 satellite images.

3.2.3 Defining grassland habitat patches

In order to quantify the connectivity between habitat patches of natural grassland in Mpumalanga, the location and the extent of these habitat patches had to be determined from the 2008 land cover for Mpumalanga as well as the abandoned cropland dataset. A habitat patch was considered as any area not transformed by cultivation, plantation forestry, urban development or mining, and a distinction was made between pristine grassland patches and abandoned croplands. The major road network was used to divide the remaining habitat into smaller patches. All habitat patches smaller than 5 ha were removed as computational limitations of the Conefor Sensinode (Saura and Torné 2009) software restricted the number of habitat patches that could be processed. These removed patches were mostly caused by the overlay of the different datasets, and were an insignificant proportion of the total grassland habitat area. However, small patches can play an important role in connecting the landscape, and habitat fragmentation might have been slightly overestimated in this study by removing small patches. The resulting habitat patch layer contained 3 681 grassland habitat patches with a total area of 30 076 km^2, of which 3 056 km^2 was abandoned croplands.

3.2.4 Quantifying connectivity

Connectivity can be described from different perspectives and scales (Crooks and Sanjayan 2006). Landscape connectivity can be seen as a result of both the specific species attributes (dispersal distance) and the spatial arrangement of habitat patches in the landscape (Tischendorf and Fahrig 2000). The arrangement of habitat patches in the landscape determines the structural connectivity. Functional connectivity describes the behavioural response of a specific organism to the landscape structure and is determined using attributes of the specific species, such as dispersal distance (Tischendorf and Fahrig 2000). Although structural connectivity is relatively easy to measure, functional connectivity is a feature of the specific organisms studied and the same landscape can have different levels of connectivity for different organisms (Tischendorf and Fahrig 2000).

There are more than 60 connectivity metrics (Rayfield *et al.* 2011) with various data requirements, information yield and performance depending on the specific ecological situation (Calabrese and Fagan 2004). Some of the most widely used connectivity metrics include the nearest neighbour distance, spatial pattern indices, graph theoretic indices, buffer radius and observed emigration and immigration (Calabrese and Fagan 2004).

Graph theoretic connectivity metrics provide an appropriate balance between initial data requirements and the detail of the results, and are also more computationally efficient than most connectivity metrics (Calabrese and Fagan 2004). Graphs are representations of more complex real systems (Urban *et al.* 2009) and represent habitat as a set of habitat patches (nodes) and connections between habitat patches (links or edges) (Calabrese and Fagan 2004). Graph theory can describe structural or functional connectivity, depending on the way the habitat patches and links are represented (Rayfield *et al.* 2011). Structural connectivity will be represented when the links contain information about the structure and arrangement of habitat patches, and functional connectivity will be represented when additional information such as dispersal distance is used. Nodes and links can be assigned weights representing patch size or quality, or the distance or effective distance of links (Rayfield *et al.* 2011). Graph theory connectivity metrics can be used over broad spatial scales with many habitat patches, and are flexible in the incorporation of additional information (Calabrese and Fagan 2004; Rayfield *et al.* 2011).

In this study, graph theoretic indices were used to quantify a) the overall connectivity of grassland habitat patches in Mpumalanga, b) the importance of individual grassland habitat patches for overall connectivity, and c) connectivity in different grassland vegetation types. The area of habitat patches as well as edge-to-edge Euclidian distances between habitat patches was calculated using ArcGIS 9 and the Conefor inputs GIS extension (www.jennessent.com/arcgis/conefor_inputs; accessed 03-10-2011). The program Conefor Sensinode 2.2 was used to calculate the connectivity indices. This program uses graph structures to calculate indices and also has the ability to determine individual patch importance for overall landscape connectivity (Saura and Pascual-Hortal 2007a; Saura and Torné 2009).

3.2.5 Quantifying overall landscape connectivity with and without abandoned croplands

The Number of Components and the Integral Index of Connectivity were used to determine to what extent the presence of abandoned croplands improve the overall landscape connectivity. Two separate analyses were done: first using only pristine natural habitat patches excluding abandoned croplands, and then using habitat patches consisting of both abandoned croplands and pristine grassland. The Number of Components and Integral Index of Connectivity were chosen because these indices don't demonstrate the same problems associated with many other connectivity indices, where there is an increase in connectivity with increased fragmentation (Tischendorf and Fahrig 2000), or no connectivity predicted for a landscape occupied by one big habitat patch (Tischendorf and Fahrig 2000), or a lack of response of the index to the loss of big isolated habitat patches (Pascual-Hortal and Saura 2006).

A component is a set of habitat patches with a connection between every two habitat patches in the component. As connectivity across the landscape increases, the number of components will decrease (Saura and Pascual-Hortal 2007a). More connected landscapes will also tend to consist of one big component in which all the habitat patches are connected. As the landscape becomes more connected, the percentage of the available habitat area within the largest component will also increase.

The Integral index of connectivity is recommended as the best binary index for landscape connectivity measurements (Pascual-Hortal and Saura 2006; Saura and Pascual-Hortal 2007b). The advantage of the Integral Index of Connectivity is that it incorporates habitat amount (or patch quality) and connectivity into one concept. This means that the habitat patch itself is considered as an area where connectivity occurs (Pascual-Hortal and Saura 2006). The Integral Index of Connectivity (IIC) ranges from 0 to 1, increasing with improved connectivity (Pascual-Hortal and Saura 2006) and is calculated by the following formula:

$$IIC = \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i . a_j}{1 + nl_{ij}}}{A_L^2}$$

In this formula, *n* is the total number of nodes, a_i and a_j are the attributes (area) of nodes *i* and *j*, nl_{ij} is the number of links in the shortest path between patches *i* and *j*, and A_L is the maximum landscape attribute (the total landscape area, consisting of both habitat and non-habitat areas) (Saura and Pascual-Hortal 2007a; Saura and Pascual-Hortal 2007b).

Landscape connectivity is species-specific, and the same landscape has various levels of connectivity for different species, depending on the dispersal ability of each species (Crooks and Sanjayan 2006). To incorporate the responses of different species to the landscape pattern the analysis was repeated with a range of different threshold distances; 50 m, 100 m, 250 m, 500 m and 1 000 m. The threshold distance specifies at which inter-patch distance two patches would be considered as connected or not connected. For example; if the threshold distance for a connectivity analysis is 500 m, every two patches that are less than 500 m apart will be considered as connected.

3.2.6 Quantifying the importance of individual patches for overall connectivity

In order to conserve connectivity in increasingly fragmented landscapes, the conservation of individual habitat patches can be prioritised according to their contribution to overall landscape connectivity (Baranyi *et al.* 2011). Different connectivity indices can be used to identify these key elements in the landscape. The connectivity values for individual patches were calculated by removing each patch in turn and measuring the difference in the Integral Index of Connectivity for the landscape. The difference in the Integral Index of Connectivity was calculated for each patch as:

$$dIIC\ (\%) = 100 \cdot \frac{IIC - IIC_{remove}}{IIC}$$

Here *IIC* is the overall index value when all nodes are present in the landscape and IIC_{remove} is the overall index value after the removal of the specific habitat patch (Pascual-Hortal and Saura 2006; Saura and Pascual Hortal 2007a). The values for individual patches were calculated at a distance threshold of 50 m. Because this study investigates landscape-scale connectivity over a large area for a biome, it is impossible to consider every species' unique dispersal distance and habitat requirements. Therefore a distance threshold of 50 m was used to calculate the values of individual patches. The seed dispersal abilities of plants with long distance dispersal methods is extremely difficult to determine (Cain *et al.* 2000), and is further complicated by the multiple dispersal vectors responsible for seed dispersal of almost any given plant species (Nathan 2007). The dispersal distances of some long distance and short distance wind dispersed grassland forb species can be seen as less than 100 m and less than 10 m respectively (Soons *et al.* 2004). The distance threshold of 50 m can therefore be considered as an intermediate dispersal distance for wind dispersed grassland forb species.

3.2.7 Quantifying connectivity of vegetation types

As the world's ecosystems are increasingly being transformed through human activities it is important to monitor and track the conservation status of ecosystems and identify those most in need of conservation attention (Rodríguez *et al.* 2011). Accordingly, the IUCN developed criteria for identifying such threatened ecosystems, based mostly on the rate of decline and the size of the current distribution of ecosystems (Rodríguez *et al.* 2011). Criteria for the listing of threatened ecosystems have also been developed for South Africa by the South African National Biodiversity Institute and the Department of Environmental Affairs and Tourism (SANBI and DEAT 2009). Even though the habitat fragmentation of an ecosystem is listed as a criterion to identify threatened ecosystems in South Africa, it has not yet been used, as further testing is still needed to determine the workability of this criterion (SANBI and DEAT 2009). Connectivity measures may provide a way to quantify the fragmentation of different vegetation types or ecosystems, and may help with the identification of threatened ecosystems.

In this study the connectivity of vegetation types were quantified in two ways. The weighted importance of each vegetation type for overall connectivity was calculated as:

$$\text{Weighted Importance of vegetation type } = \frac{\sum_{i=1}^{n}(dIIC_i \cdot \mathrm{Area_i})}{\text{Total area of vegetation type}}$$

Here n is the total number of nodes (habitat patches) in the vegetation type, $dIIC_i$ is the percentage difference in the Integral Index of Connectivity for the entire landscape when node i is removed and $Area_i$ is the area of node i in the vegetation type. Connectivity for each vegetation type was also quantified by the percentage of the patch area of the vegetation type that is a part of the largest component in the whole landscape (main landscape component). As a vegetation type becomes less connected, a smaller percentage of the total patch area will be in the main landscape component.

3.3 Results

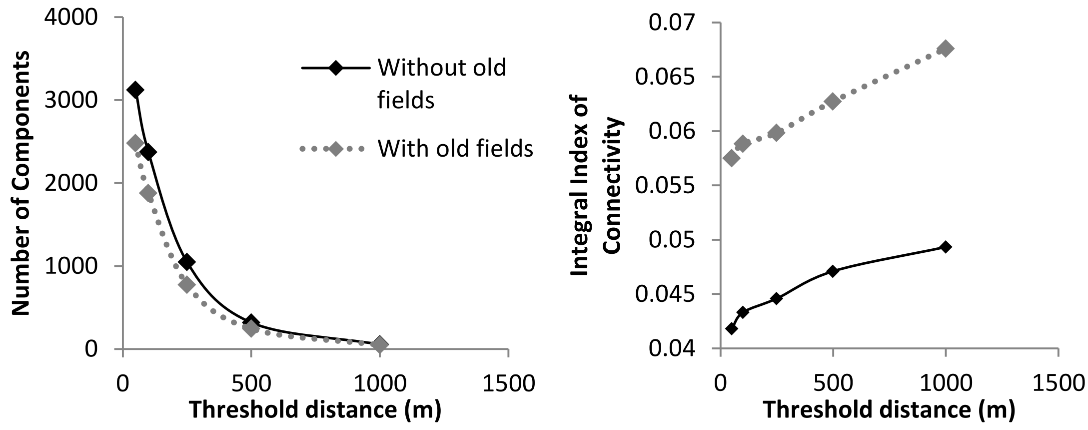
3.3.1 Quantifying overall landscape connectivity and the importance of individual patches

The grassland habitat patches (including both abandoned croplands and pristine grassland) in Mpumalanga were mostly well-connected, with 47 different components and 99.6 % of the total habitat patch area in the main component at a threshold distance of 1000 m (Table 3.1). Although this means that there were still 47 clusters of habitat patches that had no connections between them, most habitat patches were connected in one big component that spanned the entire landscape and occupied a large portion of the total habitat patch area.

Table 3.1: Percentage of the total habitat area in the biggest component and the percentage of the number of patches that are in the biggest component at distance thresholds of 50 m, 100 m, 250 m, 500 m and 1 000 m.

Distance threshold (m)	Patch area in main component (%)	Number of patches in main component (%)
50	93.6	27.6
100	94.8	39.4
250	96.0	72.2
500	98.7	85.9
1000	99.6	96.4

The number of components increased rapidly as the threshold distance decreased (Figure 3.1a), but the largest part of the landscape remained connected, with 94 % of the total habitat patch area in the main component at a threshold distance of 50 m (Table 3.1). Both the number of components as well as the Integral Index of Connectivity showed an increase in connectivity as the threshold distance was increased (Figure 3.1). This was expected, because as the threshold distance was increased, more patches became connected to each other. Three areas became noticeably disconnected as the threshold distance decreased (Figure 3.2). These areas were in the extreme west, north-west and south-west of the study area.



(a) (b)

Figure 3.1: The Number of Components (a) and the Integral Index of connectivity (b) of the grassland habitat patches of Mpumalanga excluding old fields, as well as including both old fields and pristine grasslands at a range of different threshold distances.

The inclusion of abandoned croplands as habitat patches resulted in an improvement in connectivity according to both the number of components and the Integral Index of connectivity (Figure 3.1a &b). Abandoned croplands resulted in a 33 % increase in the Integral Index of Connectivity at a threshold distance of 500 m (Figure 3.1b). Although the inclusion of all abandoned croplands resulted in an improvement in connectivity, no single abandoned cropland patch led to a major improvement in overall connectivity on its own (Figure 3.3). The largest improvement in IIC as a result of the inclusion of a single abandoned cropland habitat patch was 0.2 % (Figure 3.3).

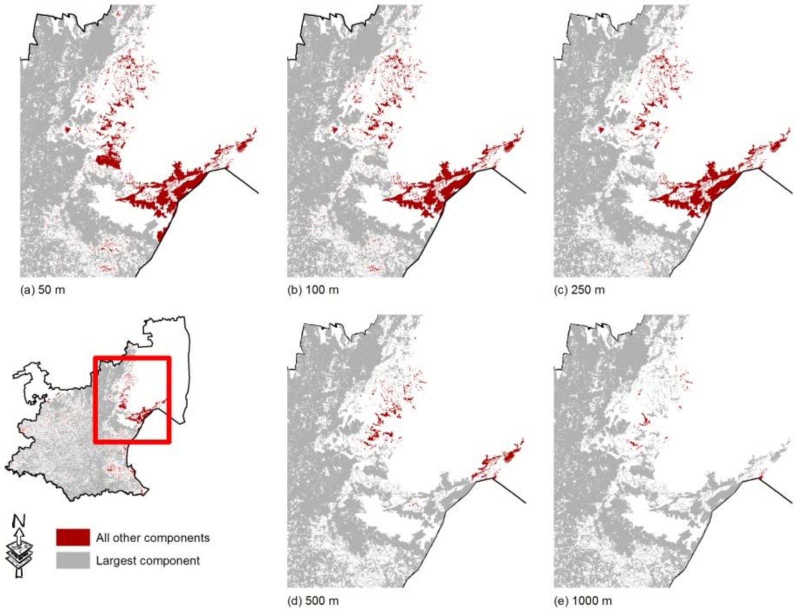


Figure 3.2: The extent of the largest component in the landscape and all other small components at a distance threshold of (a) 50 m, (b) 100 m, (c) 250 m, (d) 500 m and (e) 1000 m.

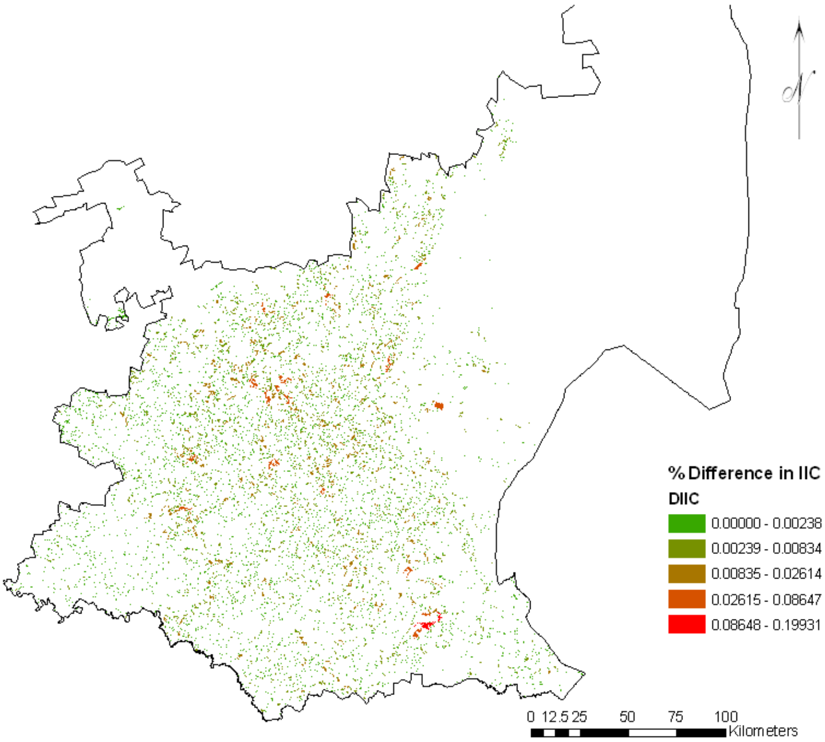


Figure 3.3: The extent of abandoned croplands in the grassland biome of Mpumalanga. The difference in the overall Integral Index of Connectivity caused by the inclusion of each abandoned cropland habitat patch separately range from 0 to 0.3 %.

The most important habitat patches supporting landscape connectivity, as calculated by the difference in the overall Integral Index of Connectivity caused by the removal of the patch, were the large patches of continuous habitat in the upper centre and the lower centre of the study area (Figure 3.4). The largest difference in the overall Integral Index of Connectivity caused by the removal of a single patch was 10.6 %, while the removal of several small patches made no difference.

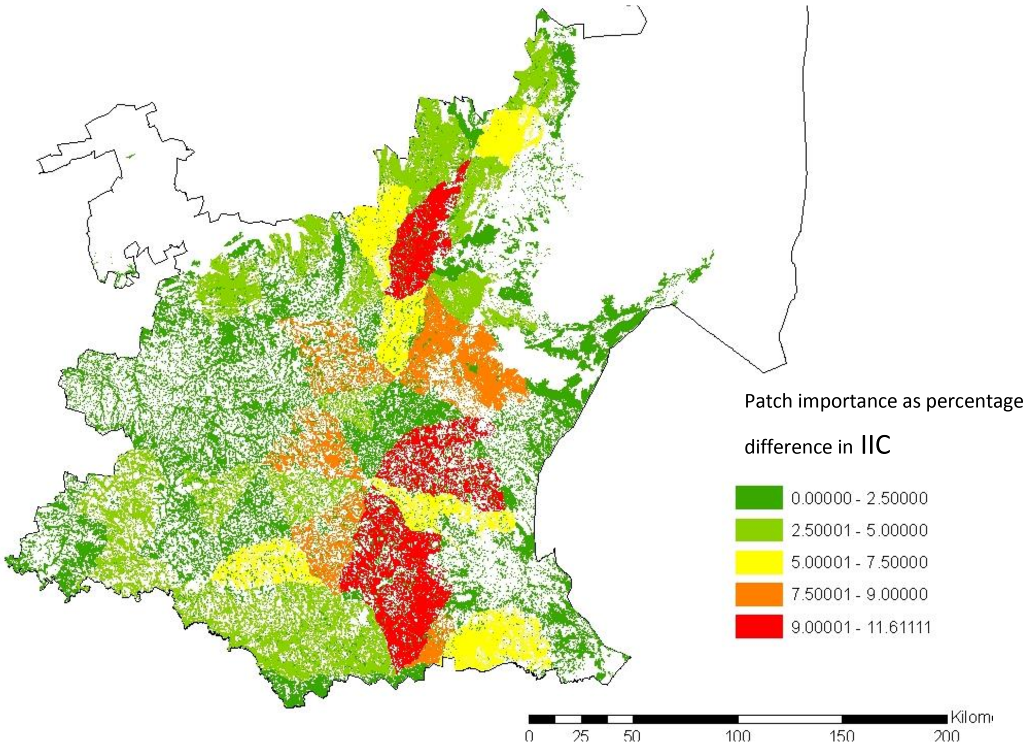


Figure 3.4: Habitat patch importance for overall landscape connectivity as the percentage difference in the Integral Index of Connectivity (IIC) for the removal of each patch at a threshold distance of 50m.

3.3.2 Quantifying connectivity of vegetation types

A distinction was made between the most important vegetation types supporting overall connectivity and the most connected vegetation types. The most important vegetation types for maintaining overall connectivity, as measured by the weighted average of the importance of the patches in the vegetation type, were the Wakkerstroom Montane

Grassland, Eastern Temperate Freshwater Wetlands, Steenkampsberg Montane Grassland and Lydenburg Thornveld (Table 3.2, Figure 3.5). This may be explained by the relatively large habitat patches and central locations of these vegetation types. The least important vegetation types for maintaining overall connectivity were the Northern Escarpment Quartzite Sourveld, the Northern Escarpment Dolomite Grassland and the Barberton Montane Grassland (Table 3.2, Figure 3.5). These vegetation types are mostly located on the borders of the study area and are severely impacted by habitat loss and fragmentation.

Table 3.2: The level of connectivity in the grassland vegetation types of Mpumalanga as the percentage of the total patch area of the vegetation type that is in the largest component and the weighted importance of the patches in each vegetation type for overall connectivity based on a 50 m threshold distance. Vegetation types are ordered by weighted importance values.

Vegetation type	Natural (%)	Total patch area in largest component (%)		Weighted importance
		50 m	500 m	
Barberton Montane Grassland	64.5	6.3	87.4	0.14
Northern Escarpment Quartzite Sourveld	47.4	48.2	77.1	0.31
Northern Escarpment Dolomite Grassland	50.6	73.4	86.8	1.22
Frankfort Highveld Grassland	66.4	99.7	100.0	1.42
Andesite Mountain Bushveld	82.2	99.4	100.0	1.44
Ithala Quartzite Sourveld	76.6	95.4	99.7	1.69
Soweto Highveld Grassland	58.6	98.9	100.0	1.74
Tsakane Clay Grassland	66.6	99.4	100.0	1.78
Rand Highveld Grassland	62.4	95.6	97.4	1.87
Low Escarpment Moist Grassland	97.4	100.0	100.0	2.07
Eastern Highveld Grassland	54.0	95.7	99.9	2.22
KaNgwane Montane Grassland	57.0	91.0	99.5	2.40
Long Tom Pass Montane Grassland	59.9	93.8	98.7	2.58
Sekhukhune Montane Grassland	76.9	99.7	100.0	3.11
Amersfoort Highveld Clay Grassland	67.2	99.8	100.0	3.46
Paulpietersburg Moist Grassland	67.6	97.9	99.9	3.59

Lydenburg Thornveld	85.2	99.9	100.0	3.75
Steenkampsberg Montane Grassland	81.9	97.8	99.5	4.70
Eastern Temperate Freshwater Wetlands	95.4	91.8	99.9	4.80
Wakkerstroom Montane Grassland	86.8	100.0	100.0	6.26

Most vegetation types were well connected as indicated by the percentage of the total patch area of the vegetation type that was within the largest component (Table 3.2). The most connected vegetation types were the Wakkerstroom Montane Grassland, Low Escarpment Moist Grassland and Lydenburg Thornveld (Table 3.2). All the habitat patches of these vegetation types were connected to each other at a threshold distance of 50 m. The least connected vegetation types were the Barberton Montane Grassland, Northern Escarpment Quartzite Sourveld and the Northern Escarpment Dolomite Grassland (Table 3.2). These were the only three vegetation types with less than 90 % patch area connected to the largest patch in the landscape at a threshold distance of 50 m. With less than 40 % of the total patch area in the vegetation type connected to the main landscape component at a threshold distance of 50 m, the Barberton Montane Grassland was the most fragmented vegetation type and deserves further conservation attention. Only 6.3 % of habitat patch area in the Barberton Montane Grassland vegetation type was connected to the study area's main landscape component. This vegetation type was poorly connected to other grassland vegetation types, but its habitat patches were well connected to each other, with 79 % of habitat patch area connected in one component.

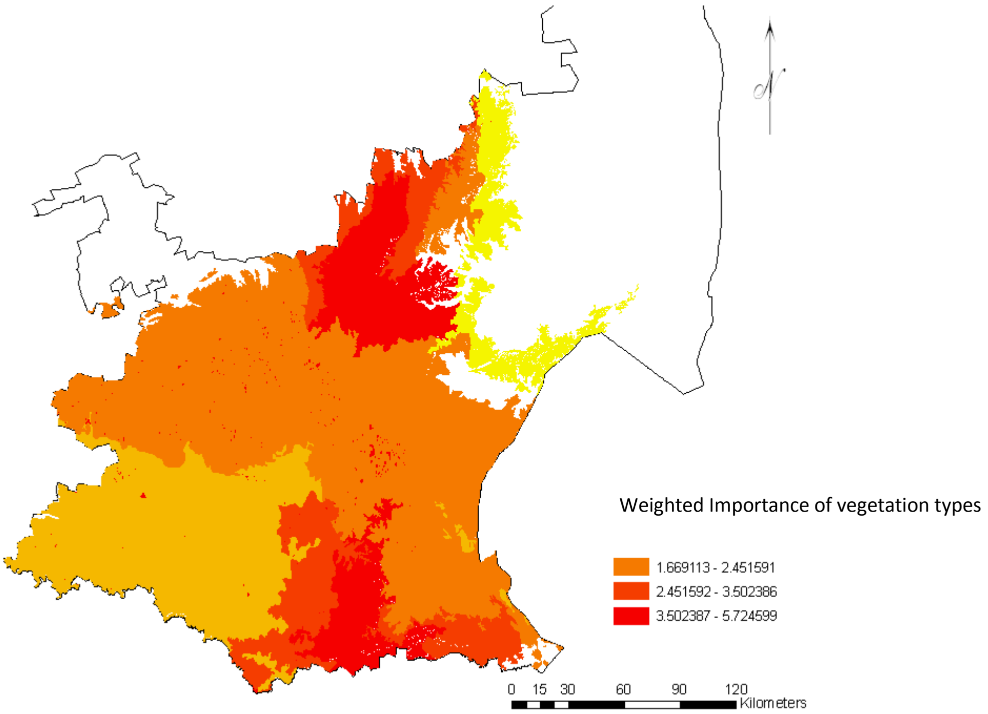


Figure 3.5: The weighted importance of grassland vegetation types in Mpumalanga shown as the weighted average of the percentage difference in the Integral Index of Connectivity.

3.4 Discussion

This study found the grassland biome of Mpumalanga to be relatively well connected despite a high degree of habitat loss. Indeed, 93.6 % of the total grassland habitat patch area (27.6 % of the number of patches) is connected at a threshold distance of 50 m (Table 3.1). This implies that any species with a dispersal distance equal to or larger than 50 m will be able to disperse to 93.6 % of the total habitat patch area in the landscape, with the exclusion of a number of very small isolated patches surrounded by large, well connected habitat patches. The grassland habitat patches of Mpumalanga are well connected compared to European grasslands (Soons *et al.* 2005).

Maintaining connectivity of the grassland habitat patches in Mpumalanga plays an important role in the persistence of organisms and processes when habitat loss and fragmentation increase, and enables range shifts of organisms as an adaptation to climate change (Crooks and Sanjayan 2006). This study identified the habitat patches and vegetation types that are the most critical for the persistence of overall habitat connectivity and can serve as a guideline to direct conservation efforts. The identification of habitat patches and vegetation types supporting overall connectivity should help with the prioritisation of conservation efforts. This process is currently underway in the update of the province conservation plan (see Ferrar & Lötter 2007 for the first version).

The three least important vegetation types for maintaining overall connectivity, according to the weighted difference in Integral Index of Connectivity, were also the least connected vegetation types with the smallest percentage of total patch area in the main patch. These three vegetation types (Northern Escarpment Quartzite Sourveld, Northern Escarpment Dolomite Grassland and Barberton Montane Grassland) are on the eastern edge of the grassland biome distribution in Mpumalanga, are adjacent to and interspersed with savanna vegetation, and are also heavily transformed through plantation forestry (Figure 3.4). At a threshold distance of 50 m, the Frankfort Highveld Grassland vegetation type is the fourth least important vegetation type for overall connectivity, but is 99.7% connected to the main component in the landscape and is therefore well connected. This highlights the difference between the importance of a vegetation type contributing towards overall connectivity, and how well a vegetation type is connected. Areas on the border of the study area have a lower importance for overall connectivity than central areas even though they may be well connected to the rest of the landscape. The most important habitat patches for supporting overall connectivity are in or near ecosystems listed as endangered (the Blyde Quartzite Grassland, Chrissiesmeer Panveld and Dullstroom Plateau Grasslands) (National Environmental Management, 2011). The results confirm the endangered status of these ecosystems and can be an important tool to motivate and direct conservation efforts.

Although this study quantified overall landscape connectivity, functional connectivity is specific to each organism, and the same landscape may be found to be connected for one species and unconnected for another (Bunn *et al.* 2000). Even though this landscape is well connected at the 50 m distance threshold, this is not necessarily true for all the organisms occurring in this landscape. Given the absence of species specific dispersal data this study used a general dispersal distance that can be applied to many species. A separate analysis should incorporate specific species of interest, such as threatened species, but there is very little information available on the dispersal distances of South African grassland species, and further studies in this area would be valuable. The exclusion of species specific data and the broad definition of habitat patches used in this study may be reason for it to be seen as oversimplified. However, for a single analysis of a large landscape with many diverse organisms, it is impossible to account for all dispersal distances and habitat preferences for each species.

This study showed that the percentage of habitat area in the largest component may be used as a measure of the level of connectivity in specific areas. This means that the connectivity of different vegetation types can be compared without doing a separate connectivity analysis for each, but by using connectivity measures calculated for the whole landscape. This method excludes the effects of the natural fragmented structure of some vegetation types on connectivity. In this way the loss of connectivity measured is due to habitat transformation into other forms of land use.

The use of the amount of fragmentation has been suggested as a criterion for identifying threatened terrestrial ecosystems in South Africa but has not been used yet because of insufficient testing (SANBI and DEAT 2009). The loss in habitat connectivity caused by fragmentation can be relatively easy to measure. The percentage of the total habitat area in an ecosystem that is connected to the largest component, can be used to identify the least connected ecosystems and as a criterion to identify threatened ecosystems. Mpumalanga currently has one ecosystem classified as critically endangered, 11 ecosystems as endangered and 20 as vulnerable (SANBI and DEAT 2009). The Northern Escarpment Quartzite Sourveld vegetation type is not considered as endangered, but it is the most fragmented (i.e. least connected) grassland vegetation type in Mpumalanga (Table 3.2). The second and third most fragmented, the Northern Escarpment Dolomite Grassland and the Barberton Montane Grassland, are classified as vulnerable (SANBI and DEAT 2009). These vegetation types or ecosystems may be more threatened than currently realised due to habitat fragmentation and loss of connectivity.

The use of general connectivity analyses play an important role in conservation planning as it identifies areas of the landscape that are connected, it identifies the critical threshold at which the landscape is connected, and it identifies the important connections between patches (Galpern *et al.* 2011). Priority areas for conservation are usually chosen by their ability to contribute to the viability of several species (Visconti and Elkin 2009). This ability is influenced not only by the quality of the habitat, but also by its location with regards to other habitat patches (i.e. connectivity).

Although connectivity measures in conservation planning are mainly used to identify key connector patches (Bodin and Saura 2010; Saura *et al.* 2011b; Vergara *et al.* 2010), these measures have also been used to evaluate temporal changes in connectivity (Saura *et al.* 2011a) and to assess the effects of land use and land use change on connectivity (Theobald *et al.* 2011). Until recently, the use of connectivity metrics to inform conservation decisions, have mainly been species-specific and focused on identifying important connecting habitat patches for specific species. The use of graph theory connectivity indices have great potential in accounting for the loss of specific habitat patches on habitat connectivity for a species or an ecosystem, as well as predicting the success of a protected area network in the conservation of threatened species (Neel 2008). These connectivity characteristics of a landscape can be evaluated even without species- specific dispersal data, by using a range of different threshold distances (Neel 2008).

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Chapter 4: Conclusion

Grasslands have the world's highest conservation risk, due to intensive human settlement and cultivation (Hoekstra *et al.* 2005). The grassland biome is one of the most important biomes in South Africa owing to the high species diversity and the provision of critical ecosystem services (Driver *et al.* 2012). It is also one of the most transformed and least protected biomes (Driver *et al.* 2012). Intensifying the problem, the grassland biome contains a high percentage of grasslands that are classified as natural, but are in fact abandoned croplands (Hoare 1997; Fourie 2010). The aim of this project was to determine if these abandoned croplands have any value for conservation.

The results of a NMS ordination and MRPP indicated a significant difference in the overall community composition, as well as the species richness of species with underground storage organs between abandoned cropland plots and pristine natural plots on the Nooitgedacht Dam Nature Reserve (Chapter 2). There was also a significant difference in the overall species richness and species richness of forb species. It was found that there were significantly more alien plant species on abandoned croplands than on pristine plots, but that there is no difference in the species richness of medicinal plant species (Chapter 2). It was also found that the vegetation on pristine natural plots were more homogenous than the vegetation on abandoned cropland plots. Although both abandoned croplands and pristine grassland plots were dominated by grass species, the dominant grass and forb species were different. The properties of the vegetation on abandoned croplands indicate that they may provide either habitat, or enough cover to serve as corridors for animal movement. Therefore, abandoned croplands may serve a broader cause of connecting highly fragmented landscapes.

Habitat connectivity increases the capacity of fragmented landscapes to support viable populations (Crooks and Sanjayan 2006). Even though the grasslands of South Africa are highly fragmented, the level of connectivity between grassland habitat patches is still unknown. In the third chapter of this study the connectivity of these grassland patches in Mpumalanga were evaluated using graph-theory based indices. It was found that the grassland habitat patches are relatively well connected, even at a threshold distance of 50 m. The inclusion of abandoned croplands resulted in a 33 % increase in the Integral Index of Connectivity at a distance threshold of 500 m. This means that the transformation of the abandoned croplands in Mpumalanga into other forms of land use will cause a 33 % decrease in the connectivity of pristine grassland habitat patches. Abandoned croplands, although highly transformed and fragmented, should be considered as an important factor when designing conservation programmes because they can contribute to the conservation of several species.

The growing amount of abandoned croplands worldwide (Cramer and Hobbs 2007) will cause an even greater need to understand their vegetation dynamics and contributions to conservation. The apparent inability of abandoned croplands in the grassland biome of South Africa to return to pre-cultivation conditions can have several conservation management implications. While decisions concerning the future of these abandoned croplands may lean towards their restoration, previous attempts had only limited success (van Oudtshoorn *et al.* 2011). Considering the high rate of transformation in the grassland biome and the limited extent of remaining pristine habitat patches, abandoned croplands can be seen as priority areas for development. The fact that it has not yet been established whether or not disturbed grassland can return to pristine conditions, highlights the importance of protecting the few pieces of pristine natural grassland left.

The extent, dynamics and conservation value of abandoned croplands in the grassland biome of South Africa is still relatively unknown. Future studies are needed on the vegetation and conservation value of abandoned croplands in different vegetation types within the grassland biome. The influences of rainfall and regular fires on the development and succession of abandoned croplands also need to be investigated.

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Appendix A: Coordinates of plots surveyed on the Nooitgedacht Dam Nature Reserve

Control: Natural Grassland Site		Paired abandoned cropland site	
Site Nr.	Coordinates	Site Nr.	Coordinates
1	26.005917 E 30.079611 S	2	26.003080 E 30.079310 S
3	25.994190 E 30.079720 S	4	25.994690 E 30.078080 S
5	25.988610 E 30.082750 S	6	25.991010 E 30.081630 S
7	25.984030 E 30.080110 S	8	25.985710 E 30.084030 S
9	25.985710 E 30.060440 S	10	25.983690 E 30.062170 S
11	25.991970 E 30.040360 S	12	25.992310 E 30.038640 S
13	25.984830 E 30.043590 S	14	25.984890 E 30.044640 S
15	26.005220 E 30.044830 S	16	26.003440 E 30.045080 S
17	26.003694 E 30.022800 S	18	26.002000 E 30.042690 S
19	25.998880 E 30.051650 S	20	26.002806 E 30.050390 S
21	25.976028 E 30.089110 S	22	25.975556 E 30.088830 S
23	25.983320 E 30.054930 S	24	25.986361 E 30.050830 S
29	26.000472 E 30.060720 S	30	25.999000 E 30.058030 S

Appendix B: Tests for normal distributions of differences between paired plots

Shapiro-Wilk Tests were conducted using the online Shapiro-Wilk Test Calculator from the SciStat Calc website at http://scistatcalc.blogspot.com/2013/10/shapiro-wilk-test-calculator.html

Category (Difference in paired plots)	Calculated Shapiro-Wilk statistic W	Calculated Shapiro-Wilk p-value	Critical value of W at 5 % significance level	Result
Total species richness	0.963872	0.812064	0.866	Accept Null Hypothesis of normal distribution
Total forb species richness	0.947716	0.564117	0.866	Accept Null Hypothesis of normal distribution
Medicinal plant species richness	0.864057	0.043589	0.866	Reject Null Hypothesis of normal distribution
Resprouting plant species richness	0.954288	0.664426	0.866	Accept Null Hypothesis of normal distribution
Alien plant species richness	0.940786	0.467161	0.866	Accept Null Hypothesis of normal distribution

Appendix C: Wilcoxon Signed-rank tests

Total species richness

H_0: The difference between the species richness of natural and abandoned cropland plots is not significant

H_A: The species richness of natural plots is significantly higher than that of abandoned cropland plots

Species richness natural	Species richness abandoned cropland	Signed rank
36	24	11.5
37	29	7.5
44	32	11.5
39	29	10
39	34	4.5
28	29	-1
39	32	6
31	28	2.5
45	26	13
39	31	7.5
37	42	-4.5
37	28	9
41	44	-2.5

Sum of positive ranks: 83

Sum of negative ranks: 8

W_s = 83 (larger of sum of positive ranks and sum of negative ranks)

n_d = 13

Using a table of Critical Values of *W* for the Wilcoxon Signed-rank Test it was determined that (0.005 < *P* < 0.001).

Therefore, there is a significant difference between species richness of natural and abandoned cropland plots (0.005 < *P* < 0.001).

Forb species richness

H_0: The difference between the forb species richness of natural and abandoned cropland plots is not significant

H_A: The forb species richness of natural plots is significantly higher than that of abandoned cropland plots

Forb species richness natural	Forb species richness abandoned cropland	Signed rank
27	18	10.5
26	20	5.5
35	21	12
27	19	8.5
31	23	8.5
20	18	3
30	21	10.5
21	20	1
33	16	13
28	21	7
29	31	-3
27	21	5.5
31	33	-3

Sum of positive ranks: 85

Sum of negative ranks: 6

W_s = 85 (larger of sum of positive ranks and sum of negative ranks)

n_d = 13

Using a table of Critical Values of *W* for the Wilcoxon Signed-rank Test it was determined that (0.005 < *P* < 0.001).

Therefore, there is a significant difference between forb species richness of natural and abandoned cropland plots (0.005 < *P* < 0.001).

Alien species richness

H_0: The difference between the alien species richness of natural and abandoned cropland plots is not significant

H_A: The alien species richness of natural plots is significantly lower than that of abandoned cropland plots

Forb species richness natural	Forb species richness abandoned cropland	Signed rank
3	5	-8
2	4	-8
3	4	-3
3	2	3
2	1	3
0	1	-3
1	3	-8
0	2	-8
4	2	8
0	4	-11
1	1	n/a
4	5	-3
1	7	-12

Sum of positive ranks: 14

Sum of negative ranks: 64

W_s = 64 (larger of sum of positive ranks and sum of negative ranks)

n_d = 12

Using a table of Critical Values of *W* for the Wilcoxon Signed-rank Test it was determined that (0.05 < *P* < 0.025) (one tail).

Therefore, there is a significant difference between alien species richness of natural and abandoned cropland plots (0.05 < *P* < 0.025).

Medicinal plant species richness

H_0: The difference between the medicinal plant species richness of natural and abandoned cropland plots is not significant

H_A: The medicinal species richness of natural plots is significantly higher than that of abandoned cropland plots

Medicinal plant species richness natural	Medicinal plant species richness abandoned cropland	Signed rank
16	11	7
14	19	-7
14	11	2
18	13	7
17	14	2
14	14	n/a
19	14	7
15	12	2
21	13	12
19	13	10.5
19	23	-4
17	12	7
16	22	-10.5

Sum of positive ranks: 56.5

Sum of negative ranks: 21.5

W_s = 56.5 (larger of sum of positive ranks and sum of negative ranks)

n_d = 12

Using a table of Critical Values of *W* for the Wilcoxon Signed-rank Test it was determined that $P > 0.05$ (one tailed).

Therefore, there is not a significant difference between medicinal species richness of natural and abandoned cropland plots ($P > 0.05$).

Resprouting plant species richness

H_0: The difference between the resprouting plant species richness of natural and abandoned cropland plots is not significant

H_A: The resprouting plant species richness of natural plots is significantly higher than that of abandoned cropland plots

Resprouting plant species richness natural	Resprouting plant species richness abandoned cropland	Signed rank
16	7	8.5
16	8	5.5
17	8	8.5
14	8	3.5
20	11	8.5
13	7	3.5
19	7	12
16	7	8.5
14	6	5.5
23	10	13
21	19	1
16	6	11
19	15	2

Sum of positive ranks: 91

Sum of negative ranks: 0

W_s = 91 (larger of sum of positive ranks and sum of negative ranks)

n_d = 13

Using a table of Critical Values of *W* for the Wilcoxon Signed-rank Test it was determined that $P < 0.0005$ (one tailed).

Therefore, there is a significant difference between resprouting species richness of natural and abandoned cropland plots ($P < 0.0005$).

Appendix D: Chi-square tests

Chi-square test: Difference in forb species exclusive to natural and exclusive to abandoned cropland plots

Observed values:

	Forb species	Non-forb species	Total
Exclusive to Natural	53	5	58
Exclusive to abandoned cropland	37	9	46
Total	90	14	104

Expected values:

	Forb	Non-forb	Total
Exclusive to Natural	50.19230769*	7.807692308	58
Exclusive to abandoned cropland	39.80769231	6.192307692	46
Total	90	14	104

* (Total forb x Total Natural)/Total Forb

Chi-square values

0.15705865**	1.009662751
0.198030472	1.273053034

** ((Observed – expected)^2)/expected

Chi-square obtained: 2.637805 (sum of all four chi-square values)

p-value: 0.104348 (derived using the CHITEST function of Microsoft Excel)

df = 1

Therefore: Accept hypothesis that deviation is small enough that chance alone can account for it.

Chi-square test: Difference in resprouting forb species exclusive to natural and exclusive to abandoned cropland plots

Observed values:

	Resprouting species	Non-resprouting species	Total
Exclusive to Natural	27	26	53
Exclusive to abandoned cropland	11	26	37
Total	38	52	90

Expected values:

	Resprouting	Non-resprouting	Total
Exclusive to Natural	22.37777778	30.62222222	53
Exclusive to abandoned cropland	15.62222222	21.37777778	37
Total	38	52	90

Chi-square values

0.954739049*	0.69769392
1.367599178	0.999399399

* ((Observed – expected)^2)/expected

Chi-square obtained: 4.019432 (sum of all four chi-square values)

p-value: 0.044979 (derived using the CHITEST function of Microsoft Excel)

df = 1

Therefore: Reject hypothesis that deviation is small enough that chance alone can account for it.

Chi-square test: Difference in alien species exclusive to natural and exclusive to abandoned cropland plots

Observed values:

	Alien species	Indigenous species	Total
Exclusive to Natural	2	56	58
Exclusive to abandoned cropland	6	40	46
Total	8	96	104

Expected values:

	Alien	Indigenous	Total
Exclusive to Natural	4.461538462	53.53846154	58
Exclusive to abandoned cropland	3.538461538	42.46153846	46
Total	8	96	104

Chi-square values

1.358090186	0.113174182
1.712374582	0.142697882

Chi-square obtained: 3.326337 (sum of all four chi-square values)

p-value: 0.068179 (derived using the CHITEST function of Microsoft Excel)

df = 1

Therefore: Accept hypothesis that deviation is small enough that chance alone can account for it.

Chi-square test: Difference in medicinal species exclusive to natural and exclusive to abandoned cropland plots

Observed values:

	Medicinal species	Non-medicinal species	Total
Exclusive to Natural	24	34	58
Exclusive to abandoned cropland	23	23	46
Total	47	57	104

Expected values:

	Medicinal	Non-medicinal	Total
Exclusive to Natural	26.2115385	31.78846154	58
Exclusive to abandoned cropland	20.7884615	25.21153846	46
Total	47	57	104

Chi-square values

0.18659349	0.153857788
0.23527005	0.193994602

Chi-square obtained: 0.769716 (sum of all four chi-square values)

p-value: 0.380305 (derived using the CHITEST function of Microsoft Excel)

df = 1

Therefore: Accept hypothesis that deviation is small enough that chance alone can account for it.

Appendix E: Indications of the proportion of variance represented by each axis of the NMS ordinations

Forbs NMS:

Coefficients of determination for the correlations between ordination distances and distance in the original n-dimensional space:

	r^2	
Axis	Increment	Cumulative
1	0.164	0.164
2	0.325	0.489

Axis pair	r	Orthogonality
1 vs 2	0.345	88.1

Total plant species NMS:

Coefficients of determination for the correlations between ordination distances and distance in the original n-dimensional space:

	r^2	
Axis	Increment	Cumulative
1	0.397	0.397
2	0.324	0.721

Axis pair	r	Orthogonality
1 vs 2	0.411	83.1

Appendix F: Species list of plots surveyed on the Nooitgedacht Dam Nature Reserve (excluding 22 unknown species)

Species names follow:

Germishuizen G., Meyer N. L., Steenkamp Y. & Keith M. (2006) *A checklist of South African plants*. SABONET, Pretoria.

Acalypha angustata Sond.

Acalypha caperonioides Baill.

Alysicarpus rugosus (Willd.) subsp. *perennifufus* J. Léonard

Anthospermum rigidum Eckl. & Zeyh.

Aristida bipartita (Nees) Trin. & Rupr.

Aristida congesta Roem. & Schult. subsp. *congesta*

Aristida diffusa Trin.

Aristida junciformis Trin. & Rupr.

Aristida mollissima Pilg.

Aristida scabrivalvis Hack.

Asclepias species

Aster harveyanus Kuntze

Aster peglerae Bolus

Athrixia elata Sond.

Berkheya setifera DC.

Berkheya speciosa (DC.) O.Hoffm.

Berkheya zeyheri Oliv. & Hiern

Bewsia biflora (Hack.) Gooss.

Blepharis integrifolia (L.f.) E.Mey. ex Schinz

Brachiaria serrata (Thunb.) Stapf

Bulbostylis burchellii (Ficalho & Hiern) C.B.Clarke

**Centella asiatica* (L.) Urb.

Chaetacanthus species

Chamaecrista comosa E.Mey.

Chlorophytum cooperi (Baker) Nordal

Cineraria parvifolia Burtt Davy

Coleochloa setifera (Ridl.) Gilly

Convolvulus sagittatus Thunb.

**Conyza bonariensis* (L.) Cronquist

**Conyza canadensis* (L.) Cronquist

Conyza podocephala DC.

**Conyza sumatrensis* (Retz.) E.Walker var. *sumatrensis*

Corchorus confusus Wild

Crabbea angustifolia Nees

Crabbea nana Nees

Crepis hypochoeridea (DC.) Thell.

Cymbopogon caesius (Hook. & Arn.) Stapf

Cymbopogon pospischilii (K.Schum.) C.E. Hubb.

Cynodon dactylon (L.) Pers.

Cyperus esculentus L. var. *esculentus*

Cyperus obtusiflorus Vahl

Desmodium dregeanum Benth.

Digitaria diagonalis (Nees) Stapf

Digitaria tricholaenoides Stapf

Diheteropogon amplectens (Nees) Clayton var. *amplectens*

Elephantorrhiza elephantina (Burch.) Skeels

Elionurus muticus (Spreng.) Kunth

Eragrostis capensis (Thunb.) Trin.

Eragrostis chloromelas Steud.

Eragrostis curvula (Schrad.) Nees

Eragrostis gummiflua Nees

Eragrostis plana Nees

Eragrostis racemosa (Thunb.) Steud.

Eragrostis sclerantha Nees

Eragrostis trichophora Coss. & Durieu

Eriosema simulans C.H.Stirt.

Geigeria burkei Harv.

Gladiolus crassifolius Baker

Gnidia caffra (Meisn.) Gilg

Gnidia capitata L.f.

Gnidia species

Haplocarpha scaposa Harv.

Helichrysum aureonitens Sch.Bip.

Helichrysum callicomum Harv.

Helichrysum nudifolium (L.) Less.

Helichrysum oreophilum Klatt

Helichrysum pallidum DC.

Helichrysum polycladum Klatt

Helichrysum rugulosum Less.

Helichrysum spiralepis Hilliard & B.L.Burtt

Helichrysum thapsus (Kuntze) Moeser

Helictotrichon turgidulum (Stapf) Schweick.

Hermannia depressa N.E.Br.

Hermannia transvaalensis Schinz

Heteropogon contortus (L.) Roem. & Schult.

Hibiscus aethiopicus L.

Hibiscus microcarpus Garcke

Hyparrhenia dregeana (Nees) Stapf ex Stent

Hyparrhenia hirta (L.) Stapf

Hypericum aethiopicum Thunb.

Hypochaeris radicata L.

Hypoxis acuminata Baker

Hypoxis iridifolia Baker

Hypoxis rigidula Baker var. *pilosissima* Baker

Hypoxis rigidula Baker var. *rigidula*

Hypoxis species

Indigofera ripae N.E.Br.

Ipomoea bathycolpos Hallier f.

Ipomoea crassipes Hook.

**Ipomoea purpurea* (L.) Roth

Jamesbrittenia aurantiaca (Burch.) Hilliard

Justicia anagalloides (Nees) T.Anderson

Kohautia amatymbica Eckl. & Zeyh.

Kyllinga erecta Schumach.

Lactuca inermis Forssk.

Ledebouria species

Linum thunbergii Eckl. & Zeyh.

Lobelia flaccida (C.Presl) A.DC.

Lotononis calycina (E.Mey.) Benth.

Lotononis eriantha Benth.

Macledium zeyheri (Sond.) S.Ortíz

Monocymbium ceresiiforme (Nees) Stapf

Monopsis decipiens (Sond.) Thulin

Monsonia angustifolia E.Mey. ex A.Rich.

Nemesia fruticans (Thunb.) Benth.

Nesaea sagittifolia (Sond.) Koehne

Nidorella anomala Steetz

Ocimum obovatum E.Mey. Ex Benth.

**Oenothera rosea* L'Hér. ex Aiton

**Oenothera tetraptera* Cav.

**Oxalis corniculata* L.

Oxalis obliquifolia Steud. ex Rich.

Panicum natalense Hochst.

Paspalum scrobiculatum L.

Pearsonia cajanifolia (Harv.) Polhill

Pelargonium dolomiticum R.Knuth

Pelargonium luridum (Andrews) Sweet

Pentanisia angustifolia (Hochst.) Hochst.

Peucedanum magalismontanum Sond.

**Plantago lanceolata* L.

**Plantago virginica* L.

Pogonarthria squarrosa (Roem. & Schult.) Pilg.

Polygala amatymbica Eckl. & Zeyh.

Polygala hottentotta C.Presl

Polygala leendertziae Burtt Davy

Pseudognaphalium oligandrum (DC.) Hilliard & B.L.Burtt

Pseudoselago densifolia (Hochst.) Hilliard

Pygmaeothamnus zeyheri (Sond.) Robyns

Rhynchosia monophylla Schltr.

Rhynchosia totta (Thunb.) DC.

**Richardia brasiliensis* Gomes

Rothia hirsuta (Guill. & Perr.) Baker

Rumex acetosella L.

Scabiosa columbaria L.

**Schkuhria pinnata* (Lam.) Kuntze ex Thell.

Senecio affinis DC.

Senecio cathcartensis O.Hoffm.

Senecio coronatus (Thunb.) Harv.

Senecio erubescens Aiton var. *erubescens*

Senecio inornatus DC.

Senecio lydenburgensis Hutch. & Burtt Davy

Senecio macrocephalus DC.

Senecio polyodon DC. var. *polyodon*

Senecio scitus Hutch. & Burtt Davy

Seriphium plumosum L.

Setaria sphacelata (Schumach.) Stapf. & C.E.Hubb. ex M.B.Moss var. *torta* (Stapf) Clayton

Sida alba L.

Solanum panduriforme E.Mey.

Sonchus dregeanus DC.

Sonchus integrifolius Harv. var. *integrifolius*

Sonchus nanus Sond. ex Harv.

Sonchus species

Sphenostylis marginata E.Mey. subsp. *marginata*

Sporobolus africanus (Poir.) Robyns & Tournay

Syncolostemon concinnus N.E.Br.

Tephrosia capensis (Jacq.) Pers.

Themeda triandra Forssk.

Thesium species

Thunbergia atriplicifolia E.Mey. ex Nees

Trachypogon spicatus (L.f.) Kuntze

Trichoneura grandiglumis (Nees) Ekman

Tristachya biseriata Stapf

Tristachya leucothrix Trin. ex Nees

**Verbena bonariensis* L.

**Verbena brasiliensis* Vell.

Verbena rigida Spreng.

Vernonia natalensis Oliv. & Hiern

Vernonia oligocephala (DC.) Sch.Bip. ex Walp.

Wahlenbergia species

Wahlenbergia undulata (L.f.) A.DC.

Xysmalobium undulatum (L.) Aiton f.

Zornia milneana Mohlenbr.