Grassland ecology along an urban-rural gradient using GIS techniques in Klerksdorp, South Africa

Marié Joey du Toit

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Supervisor: Prof. S.S. Cilliers

Co-supervisor: Mr. T.C. de Klerk

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Abstract

Urban areas represent complex assemblages of unique vegetation communities. The multitude of influences on cities adds to this complexity rendering them an intriguing study object from an ecological point of view. Understanding the underlying patterns and processes operating in urban areas becomes increasingly important with large scale urbanization, making urban areas potential conservation sites of the future. The urban-rural gradient approach often used to study these patterns and processes, aims to quantify the existing gradient allowing comparisons of vegetation at different locations, each with diverse human influences. However, accurately quantifying the urban areas became difficult with the realization that gradients are non-linear and complex.

Most previous studies cannot be compared with each other due to differences in measures used to quantify the gradient and a lack of a well defined definition for urban areas. Our study in Klerksdorp focused on testing a model developed in Melbourne (Australia) in an attempt to contribute towards creating a standard set of broad measures to quantify the urban-rural gradient. The methods used in Melbourne aims to set a general standard with which to globally compare urbanized areas taking into account the entire extent of the study area allowing multidimensional insights into the unknown gradients. Thereby placing individual studies into an urbanization context. At the heart of it, the main objective is to observe if any global patterns emerge to shed light on urbanization influences and drivers of ecological processes.

In our study, SPOT 5 HRV satellite imagery and GIS techniques were used to calculate measures representing demographic and physical variables, as well as landscape metrics. The accuracy of the demographic measures was constrained by the scale of the available census data and subsequently more information is needed. Results showed that density of people, landscape shape index, and the percent urban land cover best quantified the observed gradient. Potential changes in grassland ecology were identified with vegetation surveys studying both the extant and the soil seed bank.

Clear differences were observed in the extant vegetation composition of comparable grassland patches at different locations along the gradient, showing that urbanization does influence grassland vegetation composition and survival in the greater Klerksdorp area. The plant species richness of the existing and the soil seed bank showed significant correlations to the specific soil properties of each sample plot. Both demographic and landscape metrics also correlated significantly to some of the species subsets, emphasizing that both are needed to accurately quantify the urban-rural gradient of the greater Klerksdorp area and identifying potential patterns of species distributions.

The urban-rural gradient described in the greater Klerksdorp area is not associated with an increase of exotic species towards the urban centre, but with a decrease of indigenous species richness as one nears the urban centre. Patterns and processes emerging from the current study could meaningfully influence planning and implementation actions concerning human development and conservation of a critically endangered vegetation type.

Opsomming

Stedelike gebiede verteenwoordig komplekse versamelings van unieke plantgemeenskappe. Die groot verskeidenheid van invloede op stede dra by tot hierdie kompleksiteit en lei daartoe dat stede interessante ekologiese navorsingsmateriaal vorm. Van toenemende belang is begrip van die onderliggende patrone en prosesse wat werksaam is in stedelike gebiede as gevolg van grootskaalse verstedeliking. Dit versterk die rol van stede as moontlike toekomstige bewaringsgebiede. Die verstedelikingsgradiënt benadering word algemeen gebruik om hierdie patrone en prosesse te bestudeer. Kwantifisering van die bestaande gradiënt lei daartoe dat plantegroei op verskillende liggings en onder verskillende menslike invloede, met mekaar vergelyk kan word. Hierdie gradiënte is egter kompleks en nie-reglynig en dit maak akkurate kwantifisering van stedelike gebiede moeilik.

Meeste vorige studies kan nie met mekaar vergelyk word nie omdat hulle verskillende metings gebruik het om die gradiënt te kwantifiseer en ook omdat daar nie 'n goeie definisie vir stedelike gebiede bestaan nie. Ons studie in Klerksdorp is gedoen om 'n model te toets wat in Melbourne (Australië) ontwikkel is in 'n poging om 'n bydrae tot die ontwikkeling van standaardmetings vir die kwantifisering van die verstedelikingsgradiënt, te lewer. Die metodes wat in Melbourne gebruik is het die doel om 'n algemene standaard te stel waarmee stedelike gebiede wêreldwyd vergelyk kan word. Individuele studies word dus in 'n breë verstedelikingskonteks geplaas. Die hoofdoel is die waarneming van enige globale patrone wat te voorskyn mag tree, om duidelikheid te gee oor verstedelikingsinvloede en aandrywers van ekologiese prosesse.

In ons studie is SPOT 5 HRV satellietbeelde en GIS tegnieke gebruik om demografiese, fisiese en landskapsmetings te bereken. Die akkuraatheid van die demografiese metings is egter beperk deur die skaal van die beskikbare sensusdata, en gevolglik word meer inligting benodig. Resultate het getoon dat die menslike bevolkingsdigtheid, landskapsvorm indeks en die persentasie van stedelike grondbedekking die waargenome gradiënt die beste gekwantifiseer het. Moontlike veranderinge in die ekologie van die grasvelde is geïdentifiseer met behulp van plantegroei opnames van beide die bestaande plantegroei en die saadbank. Duidelike verskille is opgemerk in die bestaande plantegroei samestelling van vergelykbare grasveldfragmente by verskillende liggings langs die gradiënt. Dit toon aan dat verstedeliking wel die samestelling en oorlewing van plantegroei in grasvelde in die groter Klerksdorp omgewing beïnvloed. Die spesierykheid van die bestaande en die saadbank plantegroei het betekenisvolle korrelasies met die spesifieke grondeienskappe van elke perseel getoon. Beide die demografiese en die landskapsmetings het ook betekenisvol met van die spesiegroepe gekorreleer. Dit beklemtoon dat beide hierdie tipe metings nodig is om die verstedelikingsgradiënt van die groter Klerksdorp omgewing akkuraat te kwantifiseer en om moontlike patrone van spesie verspreiding te identifiseer.

Die verstedelikingsgradiënt wat beskryf is in die groter Klerksdorp omgewing is nie met 'n toename in uitheemse spesies na die stadskern toe, geassosieer nie, maar wel met 'n afname van inheemse spesies namate die stadskern bereik word. Patrone en prosesse wat in hierdie studie geïdentifiseer is kan 'n noemenswaardige rol speel in beplanning en besluitneming rakende toekomstige stedelike uitbreiding en die bewaring van 'n bedreigde plantegroei tipe.

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Soli Deo Gloria

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Chapter 1: Introduction

1.1 Introduction

"Clearly, human actions dramatically alter the functioning of ecosystems of which humans are a part, and, equally clearly, humans are a part of virtually all ecosystems... Nowhere has this human participation been more intense than in cities..." (Grimm *et al.*, 2000)

With this quote Grimm *et al.* (2000), embodied the fundamental core in the motivation for urban ecological studies. Moreover, Pacione (2005) reminded that our contemporary world is an urban world. In 2008 the proportion of people living in urban areas equalled their rural counterparts and by 2050 it is expected that 70 percent of the world population will inhabit urban areas (United Nations, 2008). Consequently, it is abundantly clear that urbanization is rapidly transforming vast tracts of natural vegetation. In time scientists retaliated to this anthropogenic fuelled habitat alteration by the reconsideration of the widely believed classical paradigm in ecology. This led to the subsequent evolution of the contemporary paradigm where the involvement of humans was realized as an integral part of the study of ecosystems (Pickett *et al.*, 1992). As a result of the widespread acceptance and recognized importance of the new contemporary paradigm in ecology, the science of Urban Ecology was born. In a paper on the early history of Urban Ecology in Europe, Sukopp (2002) stated that the term urban ecology was introduced by the Chicago school of social ecology within sociology (Park *et al.*, 1925 as quoted by Sukopp, 2002) yet the term was not formally defined in ecology until the 1970s. Nonetheless, Sukopp (2002) emphasized that the content had already existed for centuries. Marzluff *et al.* (2008) define Urban Ecology as the study of ecosystems that include humans living in cities and urbanizing landscapes. Studies in Urban Ecology, therefore, specifically aim to elucidate the perceived anthropogenic influences on the previously pristine natural, now transformed urban environments.

The firm acceptance of the new ecology paradigm led to the necessity of quantitatively exploring the social role played in Urban Ecology and its influence on the observed patterns of urban ecosystems. The introduction of a social scientific aspect into the natural sciences research in Urban Ecology explains the development of two directions in Urban Ecology, namely: the basic scientific research and the applied research of planning and management. This theme is broadly discussed in the paper by Pickett *et al.* (2001) where they elaborate on linking terrestrial ecological, physical and socioeconomic components of metropolitan areas as constituents of urban ecological systems. The applied direction in Urban Ecology developed with the realization of the importance of a good scientific underpinning to aspects directly linked with policy making and management decisions of current issues. This subsequently conveyed the responsibility to urban ecologists to involve themselves in these planning and decision making actions. A case study in Halle, Germany by Breuste (2004) serves as an example of an applied study wherein he discussed the decision making, planning and design utilized for the conservation of indigenous vegetation within urban development and how these strategies can be effectively improved.

In the ongoing years of research of urban ecosystems several approaches, some of them linked to each other, developed in the study of the effects of urban environments and the process of urbanization, namely: biotope mapping (Müller, 1997; Cilliers *et al.*, 2004); ecosystem budgets (Pickett *et al.*, 1997; Collins *et al.*, 2000; Kaye *et al.*, 2006); patch dynamics (Pickett *et al.*, 1997; Band *et al.*, 2005); urbanization gradient approach (McDonnell and Pickett, 1990; Pickett *et al.*, 1997; Pouyat *et al.*, 2002; Hahs and McDonnell, 2006); and the recently described mechanistic approach by Shochat *et al.* (2006b).

This dissertation will focus on the urbanization gradient approach as suggested by McDonnell and Pickett (1990). They argued that urban-rural gradients provided an opportunity to explicitly examine the role of humans in urban environmental interactions. In elaboration, Sukopp (1998) stated that while most of the factors which affect urban ecosystems also operate in non-urban areas, the combination of these factors meant that unique ecosystems developed with species combinations peculiar to urban areas. How these factors overlap and influence the urban and surrounding non-urban areas can be effectively examined with urban-rural gradients. Subsequently, the accurate quantification of these unique species assemblages and the processes and influences affecting them became one of the main aims in Urban Ecology.

However, even with this knowledge Theobald (2004) stated that in urban-rural gradient studies there were an immense variability in how the urbanization component of the gradient was quantified. Niemelä (1999b), for example, stated that the term 'urban' was broadly used as an geographical term that characterizes the land use of an area, where an urban area would be described as fairly large with high population densities containing industrial, business and residential districts. McIntyre *et al.* (2000) elaborated on the variability of the use of the term urban with their observation that in most of the urban ecological papers they reviewed the definition of urban was simply assumed to be known to readers and was not clearly defined. It follows that with such vagueness regarding definitions, a natural consequence will be the difficulty of comparing studies with any degree of precision (McIntyre *et al.*, 2000). However, both the terms 'urban' and 'ecology' can have several different meanings especially with regards to the specific research question asked (McIntyre *et al.*, 2000), implying that the term urban ecology, as such, is a diverse and complex concept consisting of different dimensions (Niemelä, 1999a). McIntyre *et al.* (2000) consequently emphasized that in each study the urban environment in question should be quantified in as much detail as possible to facilitate comparisons among different studies and geographical areas.

Hahs and McDonnell (2006) therefore aimed towards contributing to an objective selection of a standard set of broad measures in which to quantify urban ecological studies by defining the broad underlying matrix. Their initial set of measures included aspects deemed important by McIntyre *et al.* (2000) in describing a specific urban area such as demographic variables and physical geographic attributes. The use of a standard set of broad measures would subsequently allow global comparative studies, leading towards the advancement of basic ecological knowledge for a sustainable future. This is on par with the statement made by Marzluff *et al.* (2008) that *"as we study the foundations of Urban Ecology, rarely do we see the various scholars in our field stand back and attempt to place cities into a larger ecological context. That larger-scale vision is now rapidly developing and is the direction in which Urban Ecology, as a field, is clearly headed."* The endeavour of this dissertation is therefore to assist in the advancement of comparative urban ecological studies by documenting the results of the use of the proposed urbanization measures of Hahs and McDonnell (2006) in a South African, Third World urban setting.

1.2 Aims

1.2.1 General objective

The main motivation of the current study is to contribute towards creating a standard set of measures for quantifying the location of a sample point along an urban-rural gradient, by testing its applicability in a South African setting. This will assist in the selection of a universal set of broad measures, which would allow objective comparative studies to be made, between different cities and countries worldwide.

1.2.2 Specific objectives

1. To use SPOT 5 satellite imagery and GIS techniques to determine which of the demographic-, physical variables and landscape metrics identified in Melbourne, Australia can be used to quantify urbanization in the greater Klerksdorp area.
2. To use the identified urbanization measures to quantify the urban-rural gradient of the greater Klerksdorp area.
3. To use vegetation and soil surveys to quantify the influence of human impacts on grassland ecology, investigating aspects such as: plant species composition and diversity; and specific soil properties.

1.3 Study area

The city of Klerksdorp, the town of Orkney, three previous township areas (Jouberton, Alabama and Kanana) and the surrounding rural areas form the 30 km^2 study area (Figure 1.1). The study area is located in the North-West Province, South Africa. Klerksdorp was founded in 1837 on the banks of the Schoon Spruit as an agricultural settlement. Klerksdorp was proclaimed as a town in 1888 and the sporadic development of Klerksdorp began with the discovery and mining of gold in 1889. Orkney was founded afterwards in 1940 on the banks of the Vaal River to provide accommodation for the mineworkers at the nearby Vaal Reefs goldmine (Department of Constitutional Development and Planning, 1986).

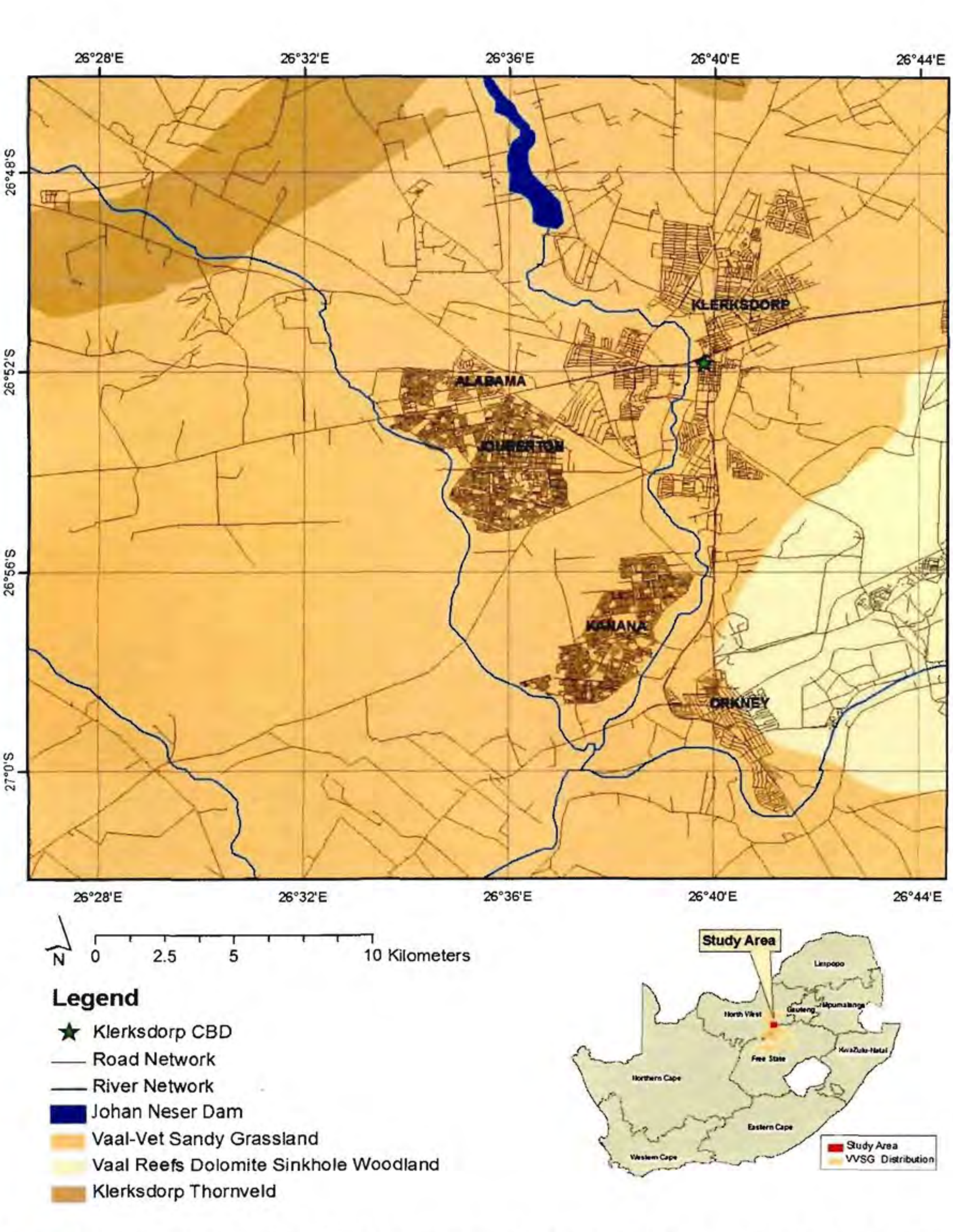


Figure 1.1: Map of the 30 km² study area, showing the vegetation units covering the greater Klerksdorp area as described by Mucina and Rutherford (2006). An overview map shows the location of the study area and the distribution of the Vaal-Vet Sandy Grassland (VVSG) vegetation type in South Africa.

The mining influence is still the major driving force for development in the greater Klerksdorp area with the Vaal Reefs goldmine near Orkney featuring as the largest goldmine in South Africa. As a result one of the main causes of pollution to the surrounding environment in the area is from mining related activities.

The most current accurate population estimates for Klerksdorp and Orkney are those supplied by the Metroplan town and regional planners (2000). According to them the population estimate of Klerksdorp (including Jouberton and Alabama) numbered 216 675 for 2003 with an average annual growth rate of 2.4 % (2000-2003). Orkney (including Kanana) numbered 160 241 with an average annual growth rate of 2.05 % (2000-2003).

The study area is located in the grassland biome. The grassland biome is represented by three different vegetation units in the study area. The research will be done specifically in the Vaal-Vet Sandy Grassland vegetation unit that covers about 78 % of the study area (Mucina and Rutherford, 2006). Research will be restricted to this vegetation type, in order to meaningfully interpret ecological changes along the urbanization gradient.

The Vaal-Vet Sandy Grassland vegetation unit is described by Mucina and Rutherford (2006) as occurring in the North-West and Free State Provinces (Figure 1.1 overview map). The climate is warm-temperate with the region receiving summer-rainfall. The mean annual precipitation is 530 mm. High summer temperatures occur with severe frost in the winter (average of 37 days per year) (Mucina and Rutherford, 2006). The altitude varies between 1220 – 1560 m, where the landscape is dominated by plains with some scattered, slightly irregular undulating plains and hills. The vegetation consists mainly of low-tussock grasslands with an abundant karroid (small xerophytic shrubs) element. The dominance of the grass *Themeda triandra* is an important feature of this vegetation unit. The geology consists of aeolian and colluvial sand overlying sandstone, mudstone and shale of mainly the Ecca Group of the Karoo Supergroup as well as older Ventersdorp Supergroup andesite and basement gneiss. Only 0.3 % of the vegetation type is protected, with the conservation target being 24 % as calculated by Mucina and Rutherford (2006) according to the method proposed by Desmet and Cowling (2004). The conservation status of the Vaal-Vet Sandy Grassland is therefore described as endangered, especially by taking into consideration that more than 63 % of it is transformed for cultivation with only 36.8 % of the natural vegetation remaining (Mucina and Rutherford, 2006).

1.4 Materials and methods

The methods followed in this dissertation will be discussed in the following five chapters.

Chapter 3 explains the satellite classification methods used to create a land-cover map of the greater Klerksdorp area.

Chapter 4 then follows by explaining how the 13 measures of urbanization was calculated for the entire landscape area and the subsequent principal component analysis and factor analysis to objectively determine the most suited measures to quantify the urbanization gradient of the greater Klerksdorp area.

Chapter 5 describes the selection of the fragmented native grassland patches and the calculation of the selected urbanization measures that best quantified the gradient, as discussed in Chapter 4, per fragmented grassland patch.

Chapter 6 elucidates the vegetation surveys done for both the extant vegetation and the soil seed bank components of each grassland patch. Non-metric multidimensional scaling ordination procedures are performed for different species compositions per sample plots for both the extant vegetation and the soil seed bank.

Chapter 7 integrates the urbanization measures with the species data collected at each fragmented native grassland patch. Correlation analysis is used to further identify the most appropriate measures, as selected in Chapter 4, to quantify the urban- rural gradient of the greater Klerksdorp area. The urban-rural gradient is consequently described for the greater Klerksdorp area as well as the characteristics of three grassland patches located along the gradient.

1.5 Dissertation structure and content

The quantification of the urban-rural gradient and the urbanization influence on vegetation patterns of fragmented native grasslands form the two main themes explored in this dissertation. The dissertation can further be divided into five main parts:

Chapter 1 and 2 form the overview, describing the greater Klerksdorp area and reviewing the development and current directions of research of urban-rural gradients and the influence of urbanization on plant patterns and ecosystem functions. The development of urban environmental studies in South Africa is also explored with emphasis on the unique structure of South African cities. These chapters thus describe the broad context on which the rest of the dissertation is based.

Chapter 3 and 4 describe the landscape spatial context, including the spatial patterns of urbanization. 13 of the measures proposed by Hahs (2006) are calculated for the 1 km^2 grid cells enclosed in the greater Klerksdorp area.

Chapter 5 and 6 describe the vegetation of the greater Klerksdorp area. In Chapter 5 the fragmented native grassland patches are identified in the greater Klerksdorp area and their location along the urban-rural gradient is quantified using the selected subset of urbanization measures. In Chapter 6 the patterns of plant species composition and species richness are explored. This was done for the existing vegetation and the soil seed bank of the fragmented natural patches.

Chapter 7 aims to describe how the vegetation data and the selected urbanization measures of Chapter 4 are combined to quantify urbanization as a process causing the observed patterns of vegetation distribution as described in Chapter 6. The observed urban-rural gradient for the greater Klerksdorp area is described together with the attributes of three of the patches located along the urban-rural gradient.

Chapter 8 concludes the dissertation summarizing the results and relating it to the known knowledge as discussed throughout the dissertation and in Chapter 2 specifically. The trends in the vegetation patterns are described as well as the significant anthropogenic influences driving these vegetation patterns.

Chapter 2: Literature review

2.1 Introduction

"...dealing with change – both conceptually and environmentally – is one of ecology's great responsibilities as a science and as a tool for improving the public dialog about the world we live in, care for, and depend on." (Pickett and Grove, 2009)

Since its inception science have tried to explain the world we live in, and as knowledge increased concepts, theories and paradigms have evolved and adapted to suit our understanding. Ecosystem studies in urban environments started in the 1970s (Sukopp, 2002). Pickett and Grove (2009) urged ecologists to incorporate change into fundamental ecological theory by suggesting that the fundamental ecosystem concept need not change in order to successfully describe urban habitats in terms of an urban ecosystem. They affirmed that the explicit inclusion of the human component through a social complex and a built complex into the ecosystem concept will elucidate it for urban use. This is on par with Niemelä (1999b) who, when expanding on the need for a theory of urban ecology; asserted that the existing ecological theories could be applied when studying ecology in the urban setting. Moreover, Niemelä (1999b) also emphasized the inclusion of the human aspect, stating that a holistic view of urban ecosystems is where concepts and approaches satisfying both natural and social scientists as well as managers are integrated.

The recognized importance of human action and interaction in urban ecosystems led scientists to search for adequate ways in which to quantify or study the effects of humans on these ecosystems. Observed changes in plant assemblages across a landscape in association with altered environmental conditions led to the gradient theory in ecology (Hahs, 2006). McDonnell and Pickett (1990) summarized the gradient paradigm as *"the view that environmental variation is ordered in space, and that spatial environmental patterns govern the corresponding structure and function of ecological systems"*. They emphasized that the gradient paradigm is a commanding organizing tool for ecological research on urban influences and that the proposed urban to rural gradient will provide an opportunity for ecologists to explicitly examine the role humans play in urban ecosystems. This proved to be the case, as numerous researchers have since applied the urban-rural gradient approach successfully (Bennett, 2003; Hahs and McDonnell, 2006; Kühn and Klotz, 2006; Weng, 2007; Van Heezik *et al.*, 2008; Burton *et al.*, 2009). Moreover, McDonnell and Hahs (2008) declared that *"we are currently at an appropriate stage in the development and use of the gradient approach to assess what we have learned, and what improvements can be made in the future to achieve better research, management and conservation outcomes"*.

This literature review will expand on (1) the development of urban environmental research in South Africa, (2) the urban morphogenesis of South African cities, (3) the influence of urban areas on native plant patterns and population distributions, and (4) the current research and directions in urban-rural gradient research.

2.2 Urban environmental research in South Africa

Very few urban vegetation surveys has been done in South Africa (Cilliers and Bredenkamp, 2000; Grobler *et al.*, 2006), with urban nature conservation strategies only adopted in certain South African cities over the last 20 years (Cilliers *et al.*, 2004).

Some of the first studies in South Africa documenting the importance of nature in urban areas are that of Poynton and Roberts (1985) where they discussed the importance of urban open space planning in South Africa. They viewed urban nature from a biogeographical perspective emphasizing the optimal functionality of these open spaces as an ecological unit (Roberts and Poynton, 1985). The inclusion of vacant lots, derelict land and road verges, areas typically excluded from conservation programmes was highlighted by their definition of an urban open space as any vegetated area within the city limits, thereby emphasizing its biological potential. In Durban specifically, a Metropolitan Open Space System was implemented to help promote the concept of a viable open space system in Durban (Poynton and Roberts, 1985). To assist in a more ecologically effective open space planning in Durban, Roberts (1993) undertook a comprehensive survey of all the remaining vegetated areas in the city allowing an accurate interpretation of the ecological status and conservation value of the open space resources. However, this was one of the few studies formally describing the vegetation of an urban area in South Africa.

Cilliers *et al.* (2004) stated that the lack of comprehensive ecological data is one of the main problems in the implementation of conservation-oriented policies in urban planning and management, especially since increasing urbanization forms one of the main threats to biodiversity in South Africa and particularly the grassland biome. This lack of detailed ecological data for urban areas in South Africa led to an extensive study of urban open spaces in some cities of the North West Province in South Africa, with the focus on fragmented natural vegetation in urban areas as well as communities directly influenced and caused by anthropogenic factors. These studies include phytosociological and floristic surveys of the ridges (Van Wyk *et al.*, 1997) and wetlands (Van Wyk *et al.*, 2000) of Klerksdorp; and analyses of the railway reserves (Cilliers and Bredenkamp, 1998), wetland communities (Cilliers *et al.*, 1998), spontaneous vegetation of intensively managed urban open spaces (Cilliers and Bredenkamp, 1999a), and the ruderal and degraded natural vegetation on vacant lots (Cilliers and Bredenkamp, 1999b) in Potchefstroom. Additionally, a phytosociological study also focused on the vegetation of road verges along an urbanization gradient in Potchefstroom (Cilliers and Bredenkamp, 2000).

However, even in these first attempts on the study of urban environments it was realized that the conservation of urban open spaces with natural and semi-natural vegetation is continuously in competition with urban development (Cilliers *et al.*, 1999). Additionally, emphasis was put on the necessity of encouraging public awareness of the importance of these natural and semi-natural urban open spaces in promotion of an integrated and participatory approach in the conservation of these areas (Cilliers *et al.*, 1999). Public awareness is of tremendous importance in a world where the perceived differences in the conservation value of natural areas versus remnant patches of 'nature' in urban areas has resulted in tensions between scientists, conservationists and developers in the never ending conflict between development and environmental conservation (McDonnell, 2007).

More recently Cilliers *et al.* (2004), in a paper giving an overview of urban nature conservation in the western-grassland biome of South Africa, stated that urban nature conservation issues in South Africa are overshadowed by the goal to improve human well-being, focusing on such aspects as poverty, equity, redistribution of wealth and wealth creation. Therefore, it is imperative to present urban environmental data in a format that is convincing and useful to decision makers. In the paper the use of urban biotope mapping was proposed as a valuable tool to determine the worthiness of specific biotopes in studied urban areas for conservation purposes. The creation of biotope maps for the urban areas of Potchefstroom placed the city on the forefront of urban nature conservation in the North West Province (Cilliers *et al.*, 2004).

Grobler *et al.* (2006) surveyed the primary grassland communities of urban open spaces in Gauteng, South Africa, where urban areas support approximately 20 % of the country's population. Limited vegetation studies, mostly unpublished, have been done on small areas in urban Gauteng. Therefore, their purpose was to provide conservation authorities with an aid to plan conservation actions when land-use planning initiatives are implemented within the urban environment. In another study, McConnachie *et al.* (2008) was the first to do research on the extent and state of urban green spaces within ten small towns in the Eastern Cape, South Africa. They found that the area and state of current public green space varied markedly between the towns, with the worst situations in poorer towns. The variables of human population density and per capita green space were the best predictors of the proportion and mean area of public green space present in the ten towns (McConnachie *et al.*, 2008).

As a last example of recent studies done in urban environments in South Africa, Cilliers *et al.* (2008) investigated the patterns of exotic plant invasions in fragmented urban and rural grasslands. Their study compared the situation found in South Africa with that of grasslands in Australia in an attempt to describe generalizations regarding the effect of different landscapes on edge responses in grasslands. The results for both South Africa and Australia showed that native grasslands in urban landscapes respond differently to fragmentation than grasslands located in rural landscapes. They identified consistent patterns of exotic species invasions in grasslands surrounded by urban and rural landscapes for both South Africa and Australia.

This short overview was not meant to be a comprehensive review on all the urban vegetation studies done in South African cities, as not all of the research done was discussed. Rather, it briefly described the development of urban ecological studies through the gradual realization of their importance, supplying the main objectives and findings of some of these studies. This overview places the current study as a first to quantify a continuous urban-rural gradient of an entire landscape with a set of urbanization measures as proposed by Hahs and McDonnell (2006) in South Africa. The study will simultaneously quantify the subsequent effect of urbanization processes on plant distributional patterns of fragmented natural grassland patches in the greater Klerksdorp area.

2.3 Urban morphogenesis of South African cities

Distinctive population density distributions occur in the study area and other South African cities. This is as a result of the unique spatial organizational legacy of these cities. In order to understand the observed population patterns in the current study, the spatial organizational legacy of South African cities merits further explanation.

Davies (1981) affirms that in general the structure of the South African city is comparable to that of Western capitalist cities, in that the relationships in space between the land value and land use patterns in the city reflect the functioning of competitive rent bid processes in a capitalist land market. A characteristic framework of zones and sectors such as the central business district (CBD) and industrial zones and major transport routes exists around which residential development takes place on lower value land. Political influence in zoning is generally similarly directed towards preserving a spatial order in land use structure and in particular the segregation of different functions. However, in residential development and distribution of social groups, South African cities depart significantly from conventional spatial models of Western capitalist cities (Davies, 1981).

Davies (1981) identified three main phases in the development of South African cities. (1) The settler-colonial period was from the beginning of white settlement in 1652 until the early years of the formation of the Union in 1910. (2) The second phase commenced after the introduction of the Natives (Urban Areas) Act No. 21 of 1923 which divided South Africa into 'prescribed' (urban) and 'non-prescribed' (rural) areas, and strictly controlled the movement of Black males between the two. This marked the beginning of the conscious nationwide pursuit of urban segregation (Lemon, 2003). As a result, towns became almost exclusively white with the only blacks allowed to live in town being domestic workers. (3) The last and most extensive policy came with urban *apartheid* (separateness) after the introduction of the Group Areas Act of 1950. This act provided for the compulsory zoning of all urban areas into exclusive group areas (Christopher, 2001). Hence, for a 40 year period (1950-1990) during South African history racial groups were forcibly separated into different residential zones involving restricted space and mobility for some groups.

Figure 2.1 shows idealized forms of the segregation city as it existed in South Africa and the subsequent apartheid city as it was established in South Africa after 1950. Lemon (2003) stated that the shift from already highly segregated pre-*apartheid* segregation cities to completely segregated *apartheid* cities was deemed necessary by the state as they saw the different racial groups as incompatible and argued that minimized contact between ethnic groups would prevent mutual friction. For this purpose buffer strips (Figure 2.1) were often proclaimed to act as barriers between different groups.

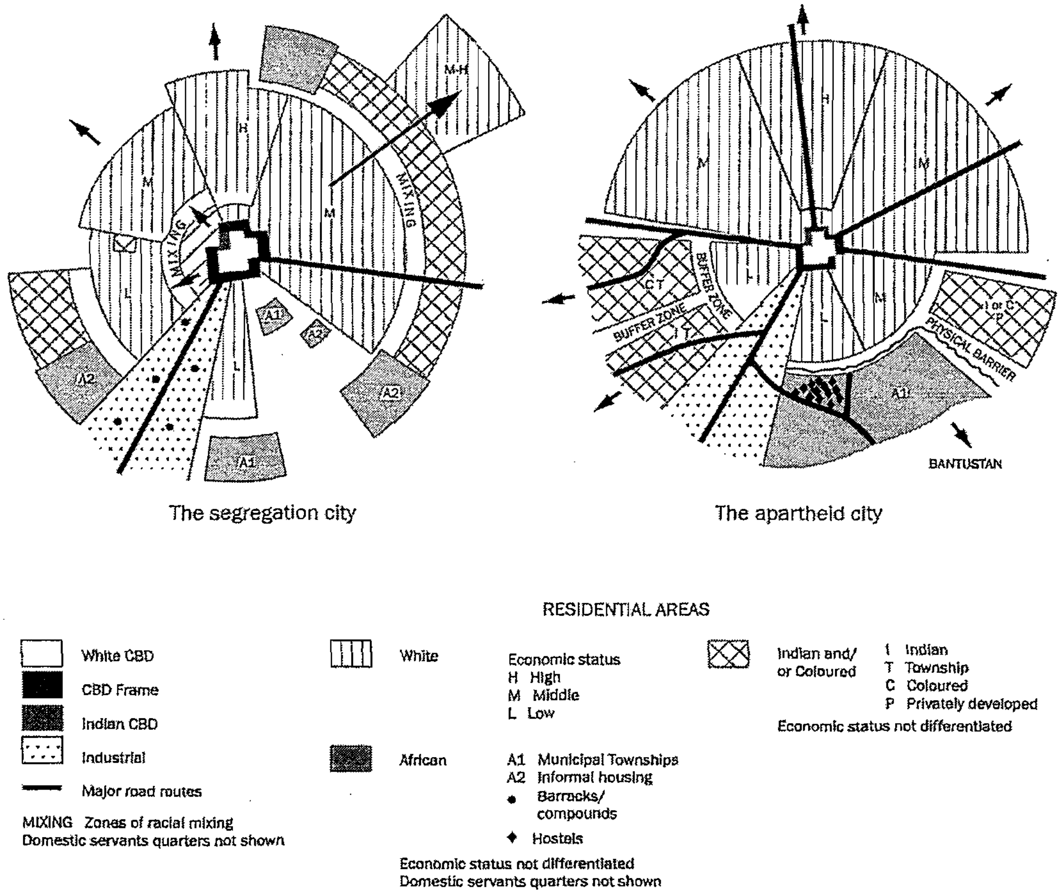


Figure 2.1: Models of the segregation and the apartheid city found in South Africa as shown in idealized form by Davies (1981).

White group areas were characteristically larger (Figure 2.1) than any of the other group areas with ample natural environments, parks and gardens dispersed in the residential areas. The average housing property size was also substantially larger in the white areas than the other residential zones (Figure 2.4). The black townships especially were lacking in green areas and gardens. The average individual plot size in this zone was very small and spatially arranged in a simplistic grid pattern. None of the inhabitants were allowed ownership of any of the properties in the townships. Figure 2.2 illustrates the absence of green areas in the suburb (former township) of Jouberton in a very convincing way. The figure shows the classification of the trees and the grass land cover classes (Chapter 3) for the urban areas in the current study. The distinct absence of vegetation in Jouberton is obvious, which is a legacy of the spatial arrangement of the houses in the area.

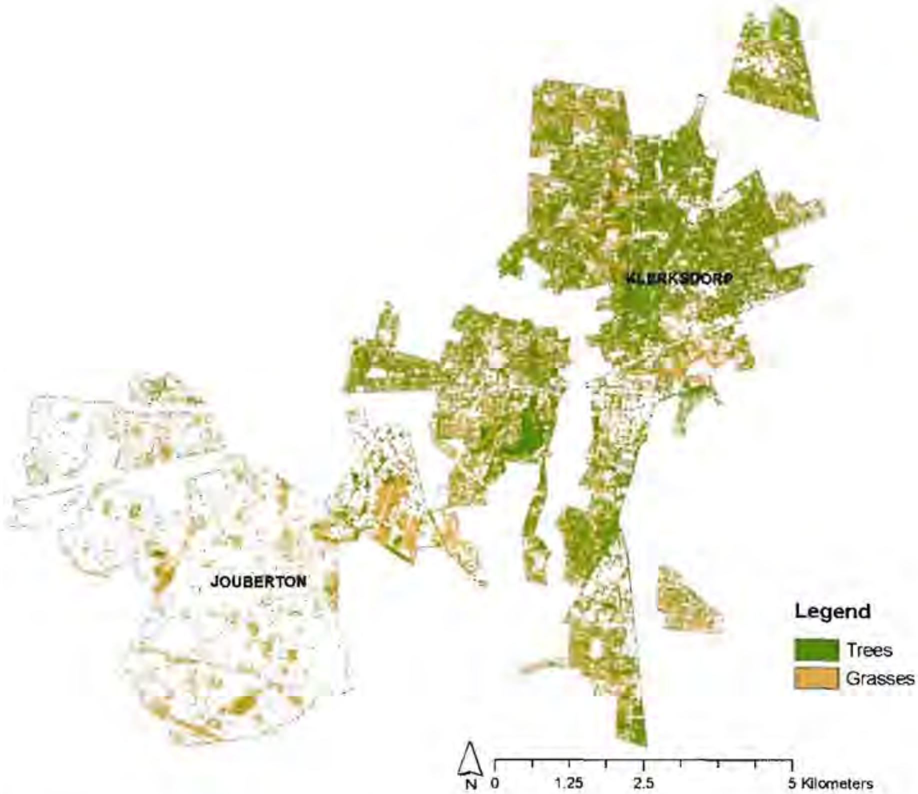


Figure 2.2: The cover of trees and grasses in the urban areas of Klerksdorp and Jouberton in the current study. This was extracted from the classified land cover map B as will be discussed in Chapter 3.

Apartheid officially came to an end in 1994 after the first national democratic election and since then post-*apartheid* cities have aimed at reintroducing race heterogeneity into South African cities. *Apartheid* was perceived as a lasting solution to South Africa's multiracial population and cities were designed to allow group areas to expand indefinitely and endure (Lemon, 2003). The legacy of the effectiveness of the segregation and unlimited growth opportunities for the group areas is still visible in most South African cities today, contributing to the problem of desegregation. However, Christopher (2001) found that rapidly growing towns were more segregated than more stagnant towns, which reflected the lack of economic growth in South Africa and the migration of poverty-stricken people from the rural areas into effectively segregated informal settlements on the urban fringes of, in particular, large urban agglomerations.

Now that all the remnants of legally enforced segregation have disappeared, the current obstacles to desegregation are demographic, social, cultural and above all economic (Lemon, 2003). Christopher (2001) emphasized that even after 7 years of post-*apartheid* the segregation levels in the cities remained exceptionally high and he proposed that rapid integration might require government intervention. This *apartheid* spatial legacy underpins the current situation as found in the greater Klerksdorp area. Figure 2.3 shows the population densities per election ward for the greater Klerksdorp area. Clearly observable is the high population densities of the suburbs (previous formal townships now primarily functioning as informal settlements for poor Africans looking for urban job opportunities, hence the high population densities) of Jouberton and Kanana on the urban fringe of Klerksdorp as opposed to the other residential areas surrounding the CBD. The high population densities are also due to the extreme proximity of the individual dwellings to each other, where each house on a plot has two or more other informal houses sublet to other families or their own extended families adding to the density of people per square kilometer area.

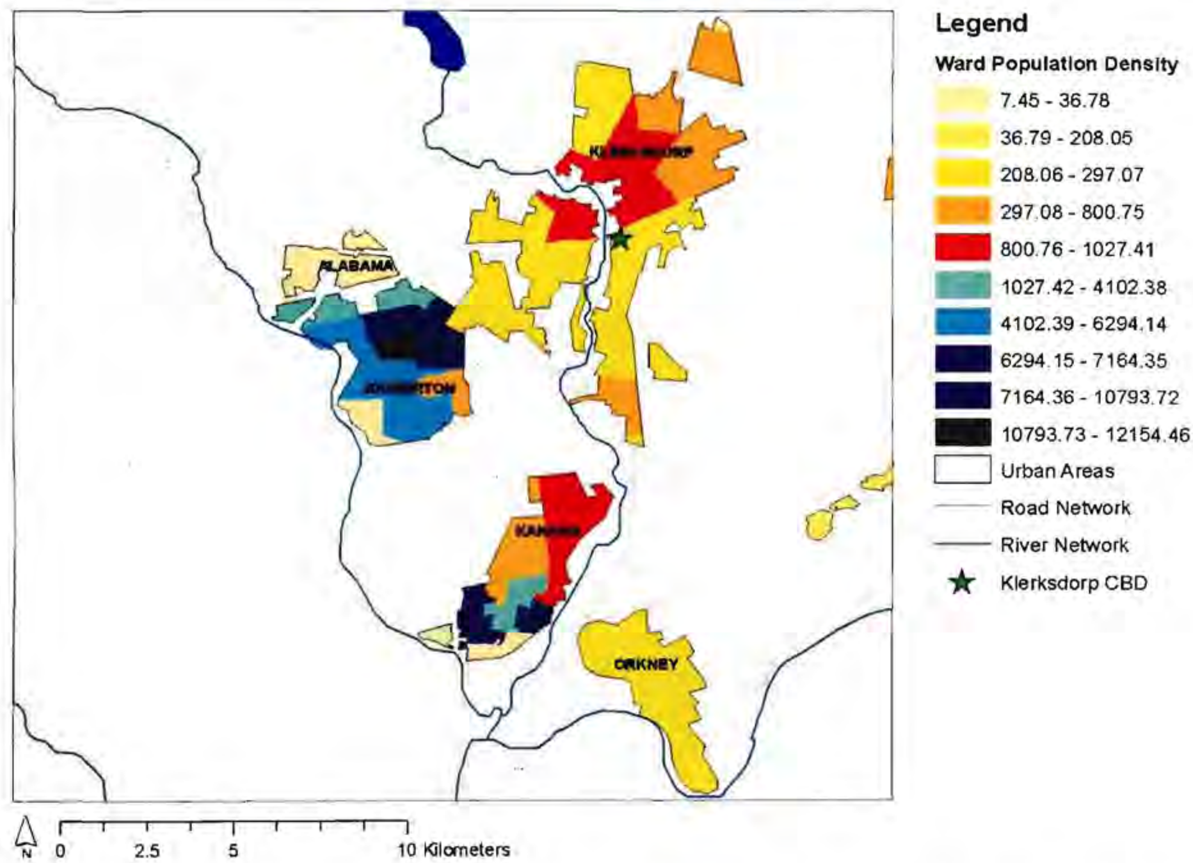


Figure 2.3: The population densities per election ward (total population/ km^2 surface area) for the urban areas of the greater Klerksdorp area. The grey outline indicates the delineation of the urban areas. The green star shows the location of the CBD of Klerksdorp.

This population distribution pattern causes the unique situation that for the greater Klerksdorp area the highest population density is not found around the CBD in the urban core, but on the urban fringe and only in certain specific spatial locations. Figure 2.4 illustrates the unique population distribution situation found in Klerksdorp, both Jouberton (a and b), previously a black township and La Hoff (c and d), a former white residential area, are found on the urban fringe, La Hoff has a mean population density of 250 people per km^2 whereas Jouberton has a mean population density of 7492 people per km^2.

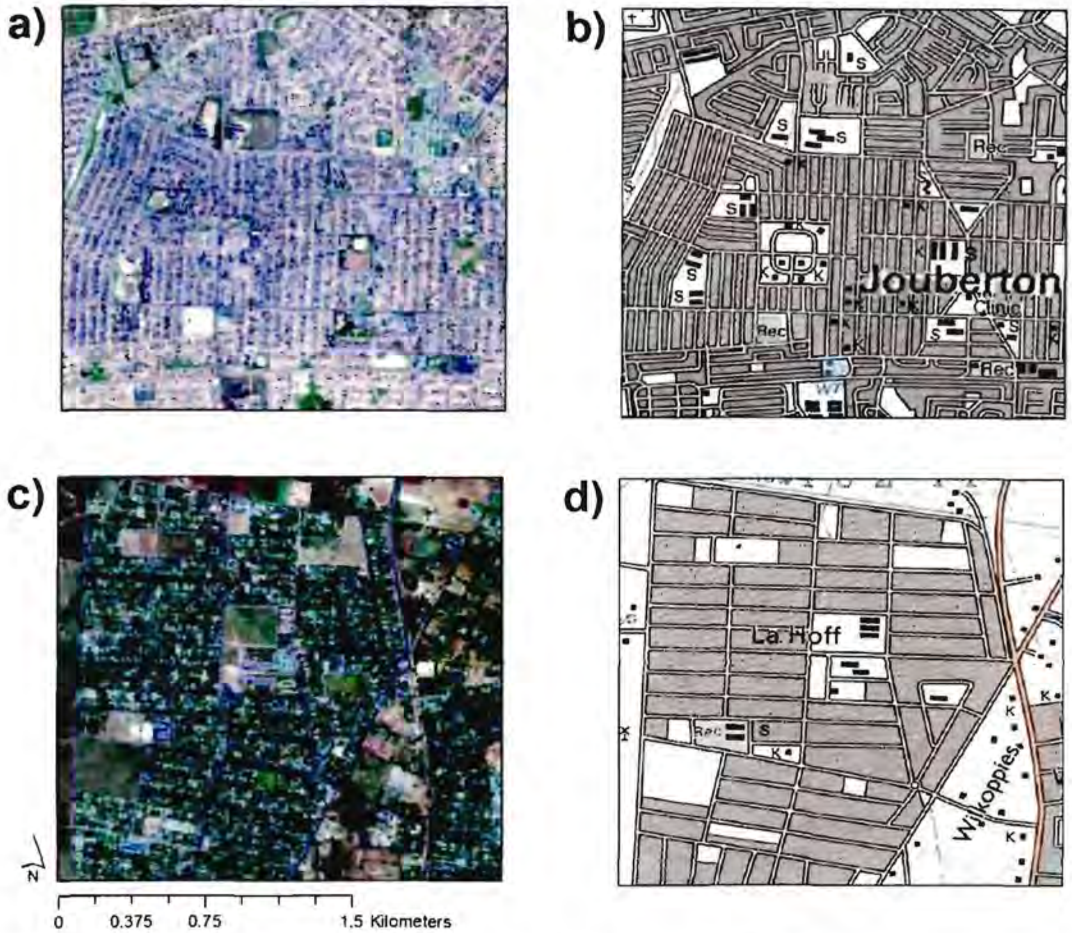


Figure 2.4: SPOT 5 satellite images of the different residential group areas as found in Klerksdorp. Jouberton (former black township) has densely packed small properties (a), clearly visible in the 1996 1:50 000 topographical map of the Klerksdorp area is the spatially constrained street pattern of Jouberton(b) in contrast to the suburb of La Hoff in a formerly white residential area (c) and (d).

The CBD of Klerksdorp still has in its close proximity patches of native grasslands interspersed with other land use zones. This is in contrast with the CBD areas of larger cosmopolitan cities, which is dominated by impervious surfaces and other urban land uses (Phinn *et al.*, 2002).

Contrasting situations of urban development and urban sprawl exist in developed and developing countries. Irwin and Bockstael (2007), in their investigation on urban sprawl in the state of Maryland, USA found that low-density development was the essential footprint of sprawl in the state with the most low-density development occurring relatively far from urban areas. Urbanization in developed countries is primarily fragmenting large areas, with the influence of urbanization extending over the entire landscape (Pauchard *et al.*, 2006). On the other hand, in developing countries the growth is still concentrated around the urban cores, replacing the neighbouring land uses such as agriculture and natural vegetation but the rate of change is much slower than that of sprawl in developed countries (McGranahan and Satterthwaite, 2003). Sutton (2003) added that urban sprawl happens to some extent in most cities but mostly only in specific areas. Pauchard *et al.* (2006) affirmed that in developing countries the expanding areas on the urban fringe also still have a high population density, as opposed to the low-density development in Maryland. These high population densities thus increase the intensity of the urban impacts. However, the quantity and the pattern of the urban fringe developments will also strongly affect its influence on both the indigenous and exotic flora and fauna (Hansen *et al.*, 2005).

Urban sprawl in the current study typically resembles that of developing countries, in that Jouberton and Kanana expand into natural and agricultural areas but the nature of the sprawl is very dense and compact. The expansion is also not necessarily on rubbish dumps and derelict land as in other world cities and as a result poses a huge threat to surrounding natural environments especially trees and shrubs. Additionally, in the residential area itself the area surrounding most of the small houses is predominantly cleared of vegetation, with most trees and shrubs removed for fuel in households. However, low density sprawl as described by Irwin and Bockstael (2007) also affects the current study area, especially in the more affluent areas. The current trend in luxury residential housing development in South Africa is that of 'eco-estates' where houses are built on large individual stands in pristine natural environments (Grey-Ross *et al., in press*). This type of development threatens large areas of natural grasslands and ridges, in and surrounding Klerksdorp, in the current study. This is similar to exurban development in the United States where people settle in rural areas on large holdings within a landscape dominated by native vegetation (Hansen *et al.*, 2005), and is currently the fastest growing form of land use in the United States (Brown *et al.*, 2005).

Therefore, due to both historical and political causes and the large disparity in the economic status of residents living in close proximity to each other in the same city, the current study area allows a unique opportunity to describe a unique and complex urban-rural gradient.

2.4 Vegetation dynamics in urban environments

Alberti (2005) declared that the future of all ecosystems on earth is increasingly dependent on the patterns of urban growth. Urban development fragments, isolates, and degrades natural habitats; simplifies and homogenizes species composition; disrupts hydrological systems; and modifies energy flow and nutrient cycling (Alberti *et al.*, 2003). Pacione (2005) reciprocated with his statement that cities are major agents in depleting the quality of the environment for future generations. Urbanization can even create entire landscapes occupied by anthropogenically created plant communities in which diversity may reflect social, economic and cultural influences as well as those recognized by traditional ecological theory

(Hope *et al.*, 2003). In addition, Alberti (2005) explains that the mechanisms of urban ecosystems such as land cover changes and modifications of natural disturbance regimes affect ecosystem functions which in turn have ecological effects.

Moreover, it should be noted that agriculture is one of the original functional causes of a majority of urban settlements and is often overlooked in quantifying the causes of biodiversity threats in urbanized landscapes. Faggi *et al.* (2006) stated that agriculture can be an even greater threat to biodiversity conservation than urbanization. In their study in the Argentinean Rolling Pampa this proved to be the case with lower native species diversity in 'rural' areas than urban parks. O'Connor and Kuyler (2009) indicated that agricultural and farming practices together with urban settlements had the most severe impact on the biodiversity integrity of the moist sub-biome of the grassland biome in South Africa.

Faggi *et al.* (2006) affirms that generally urban areas are highly dynamic, consisting of distinctive environmental gradients promoted by anthropogenic effects resulting in aspects such as the fragmentation of native vegetation and the introduction of exotic species. In general, urban areas are inhabited by greater numbers of plant species when compared to the surrounding rural areas (Rebele, 1994; McKinney, 2002; Hope *et al.*, 2003; Wania *et al.*, 2006; Millard, 2008; Vallet *et al.*, 2008). Native plants usually decrease towards the urban core with subsequent increases in exotic species richness (Sukopp, 1998; Pickett *et al.*, 2001). This is typically ascribed to the heterogeneity of urban environments, the subsequently higher diversity of potential available habitats, and the high amounts of exotic species introduced into urban areas contributing to the higher species richness. Schwartz *et al.* (2006) in a study on biotic homogenization of the Californian flora in urban and urbanizing regions, found a significant correlation to increased densities of noxious weeds to increases in human population densities. Noxious weeds are good indicators of homogenization as they are generally widespread species and become dominant within natural ecosystems (Rejmanek and Randall, 2004). Niggemann *et al.* (2009) stated that the distribution patterns of plants are typically affected by human activities such as the creation, destruction or modification of habitats, but that the extent to which humans influences plant distributions by acting as dispersal vectors are also important. Their results indicated that exotic species seemed to benefit more from human dispersal than native species in a study of the spread of plant species between human settlements.

A study done on urban biodiversity of urban habitats in Birmingham (England) on derelict sites revealed a positive relation between the site species richness and the site area. No relation could be found between the site species richness and its proximity to vegetation corridors, implying that plants do not use the greenways for dispersal and that those corridors are rather of more importance for plants as a chain of different habitats permeating the urban environment (Angold *et al.*, 2006). The sites with the highest diversity of flora were those that were disturbed in the recent past. Schadek *et al.* (2009) also linked species richness of urban brownfield sites to successional age where their study indicated that species richness was maximized when a community comprises of a mixture of early and mid-successional species. Interestingly, the study of Angold *et al.* (2006) found no evidence of an urban-rural gradient of plant communities of derelict sites, however they did find that the urban sites contained a greater proportion of neophyte exotics (exotic species invading after 1500) than rural sites.

McKinney (2008), in a review focussing on plant species richness patterns along urban-rural gradients, stated that 69.2 % of the studies documented that species richness were the highest at intermediate levels of urbanization, with none of the studies recording increases in species richness levels at the highest levels of urbanization. This is consistent with the results of Cilliers *et al.* (2004) who documented that for Potchefstroom, located in a grassland area, the natural areas situated on the city margin, those included in urban development and along road verges had the highest species richness. Intensively managed parks, pavements, parking areas and smaller ruderal areas displayed the lowest species richness and percentage of native species. A growing number of studies linked increases in soil fertility in urban areas with increases in exotic species (Gilfedder and Kirkpatrick, 1998; Lake and Leishman, 2004). More importantly, Lake and Leishman (2004) found that urban bushland remnants with no disturbance did not support any exotic plant species. Cilliers *et al.* (1999) also linked disturbance and exotic species with human influences, where they declared that the presence of degraded forms of most of the natural communities described for the city of Potchefstroom indicated the influence of various direct or indirect human disturbances on the natural vegetation.

Hahs (2006) specifically reviewed the effects of urbanization on remnant 'natural' vegetation patches in urban areas and found that anthropogenic influences typically resulted in fragmentation of the landscape, reduction in the size of remnant vegetation patches, isolation of remnant vegetation patches, suppression of natural disturbance regimes and disturbance of the patches due to recreational influences. However, Hahs (2006) emphasized that the response of a specific vegetation community to any of the above mentioned influences appears to be largely determined by the history of exposure to past disturbances and to the autecology of the species that form the particular plant community. Stenhouse (2004) investigated 71 native vegetation reserves in the Perth metropolitan area and found that the remnant vegetation in the metropolitan areas tended to be highly fragmented and affected by disturbances. Smaller reserves occurred in the highly populated inner metropolitan area exhibiting high levels of fragmentation, weed infestation and path density. In a study done on exotic plant invasions in fragmented natural grasslands comparing the situations in South Africa and Australia, Cilliers *et al.* (2008) also found that the absolute level of exotic species cover at the grassland edges was higher in most of the urban grasslands in comparison with the rural grasslands.

Native grasslands studied in Melbourne by Williams *et al.* (2005), revealed that over a 15 year period 21 % of the grasslands were destroyed by development with 21 % degraded to non-native grasslands. The remaining grassland patches were increasingly isolated and had lower average patch sizes indicating the increased fragmentation of the landscape, with the distance to the CBD and their proximity to major roads predicting their probability of being destroyed. Smaller patches also had a higher probability of being degraded (Williams *et al.*, 2005). Williams *et al.* (2006) found evidence that for local extinction of grassland species, the landscape surrounding the remnant patches, as well as the quality of the habitat maintained within the remnant patch is potentially more important drivers of fragmentation effects on plant species than the spatial characteristics of the patches such as the area or isolation.

However, despite increasing ecological research in urban areas, little is understood of the relationship between urban patterns and ecosystems dynamics (Alberti, 2005; Hahs, 2006). Shochat *et al.* (2006b) elaborated by stating that "*the key to understanding urban patterns is to balance studying processes at the individual level with an integrated examination of environmental forces at the ecosystem scale*". In the light of this, Williams *et al.* (2009) outlined a conceptual framework for the assembly of all urban floras based on selective filters. The explicit linking of drivers of floristic change to predicted outcomes in urban areas can help towards the sustainable management of urban vegetation and the conservation of biodiversity. The four filters they proposed are: habitat transformation, habitat fragmentation, urban environmental conditions, and human preference. They affirmed that these filters operate generally, with the strength of their effects modified by regional factors. They proposed that the use of a "*structured framework to conceptualize how urban environments act as agents of selection is the best way to move towards a mechanistic understanding of urban floras*" (Williams *et al.*, 2009).

Nonetheless, Pacione (2005) declared that even though people aim towards urban sustainability to protect the environment for future generations; most fundamentally it is unrealistic to expect calls for restraints on economic growth to protect the future environment to be heeded generally in a world where millions of impoverished people face a daily struggle to survive. "*The trade-off between environmental considerations and social, economic and cultural aspirations is clearly a function of general levels of well-being in a society*" (Pacione, 2005). This is increasingly evident in developing countries such as South Africa where environmental concerns generally come in second place behind community upliftment and combating poverty in areas where the majority of the people are poverty-stricken. In these areas people need to be educated and convinced of the advantages of conserving the natural environment, especially the nature in and surrounding the cities.

2.5 Current knowledge and directions in urban–rural gradient research

"*Urban ecologists seek commonalities among city ecosystems, an understanding of how context shapes the socioecological interactions within them, and their role as both drivers and responders to environmental change*" (Grimm *et al.*, 2008). Anthropogenic influences on ecological systems along urban-rural gradients provide an opportunity to address basic questions at various spatial scales. Moreover, urbanization gradients provide an opportunity to explicitly examine the role of humans

(McDonnell and Pickett, 1990). Since its inception the urban-rural gradient approach has been effectively used to study the ecology of urban areas worldwide, with the focus of the studies on understanding the distribution of plants and animals as well as ecosystem processes along gradients of urbanization encompassing the highly urbanized urban core to more rural exurban environments (McDonnell and Hahs, 2008).

Urbanization influences results in the following main environmental changes as summarized by McDonnell and Hahs (2008): changes in land use and land cover (Southworth *et al.*, 2004; Poschlod *et al.*, 2005; Grimm *et al.*, 2008; McDonnell and Hahs, 2008), changes to the chemical and physical environment (Sukopp, 2004; Pouyat *et al.*, 2008), unique species assemblages (Sukopp, 1998; Angold *et al.*, 2006), and changes to disturbance regimes (Williams *et al.*, 2005; Tratalos *et al.*, 2007).

However, Shochat *et al.* (2006a) reminded that urban ecologists have conducted mostly observational, instead of mechanistic studies. Therefore, it is now essential for researchers to move beyond pattern questions and focus on addressing process questions (McDonnell and Hahs, 2008). It is now imperative that current knowledge gaps are addressed in order for the field of urbanization gradient research to reach its full potential (McDonnell and Hahs, 2008). Grimm *et al.* (2008) declared that the next frontier in urban ecology is the understanding of urbanization in the context of biophysical, economic, or political settings.

One of the major setbacks in the assessment of past studies is the difficulty in comparing results between different cities due to the lack of a common urbanization context, a yardstick with which to objectively quantify the various findings of different studies which would allow the identification of global or regional trends. This is essential in a world where there is currently *"a tremendous call for more ecological information in urban and exurban environments by natural resource managers, planners, conservationists, scientists, and professionals associated with human health to inform planning and management decisions required to create more sustainable cities"* (McDonnell and Hahs, 2008). However, Cilliers *et al.* (2004) emphasized that there is an essential need to present urban environmental data in a format that is both convincing and useful to decision makers.

McDonnell and Hahs (2008) recently reviewed gradient analysis studies in an attempt to assess what that current status of the research is and to describe future research directions. A brief outline will be discussed on their main findings. Urbanization gradients are indirect and complex and the quantification of the gradient should be a continuum, rather than categorical. Most studies along gradients utilized a variety of broad measures of urbanization, with specific measures used only to answer particular research questions. However, a better understanding must be obtained on the measures used to define the gradient as well as the measures used as response variables, as Hahs and McDonnell (2006) affirmed that the selection of specific measures can influence the results of the study. McDonnell and Hahs (2008) further proposed the development of a common set of broad measures to facilitate comparisons and better integration between cities. By providing a common measure of urbanization, outcomes of ecology 'in' cities can be used to develop a better understanding of ecology 'of' cities. Finally, they declared that the effectiveness of the urbanization measures in predicting the outcomes of studies should be explored as well as their utility in integrating different studies.

To evaluate the current directions in urbanization gradient research, five recent studies were assessed that focused mainly on vegetation, to test whether they reflected the newest outlook on gradient studies discussed above or whether they simply followed the same traditional observational and isolated ecology 'in' cities approach.

The five studies used in our evaluation were: (1) riparian woody plant traits across an urban–rural land use gradient and implications for watershed function with urbanization (Burton *et al.*, 2009), (2) plant species response to urbanization, the comparison of isolated woodland patches in two cities of North-Western France (Vallet *et al.*, 2008), (3) the response of forest soil properties to urbanization gradients in three metropolitan areas (Pouyat *et al.*, 2008), (4) the ecological integrity of remnant montane forests along an urban gradient in the Sierra Nevada (Heckmann *et al.*, 2008), and (5) mediterranean urban-forest interface classification (MUFIC) as a quantitative method of combining SPOT 5 imagery and landscape ecology indices (Dumas *et al.*, 2008).

Six criteria were chosen with which to evaluate the different studies, namely: (1) the use of satellite images or aerial photographs, (2) quantification of the gradient as a continuum or only categorical, (3) comparativeness of the study, (4) use of broad or specific measures in the quantification of the gradient, (5) correlation of measures or indices to the input biological data, and (6) the type of gradient quantified (e.g. land use).

Three of the studies used aerial photographs; Pouyat *et al.* (2008) used it in combination with topographical maps to select appropriate forest stands, however Burton *et al.* (2009) and Vallet *et al.* (2008) both used it to create land use type maps of the study areas to quantify the gradient. Heckmann *et al.* (2008) exclusively used existing GIS map layers of county parcel data and roads and trail layers in the quantification of a development gradient, with Dumas *et al.* (2008) utilizing SPOT 5 also to quantify the gradient and create land cover maps.

The study of Dumas *et al.* (2008) was the only study to describe a continuous gradient covering the entire study area; however the quantification of the gradient was subdivided into five classes. Pouyat *et al.* (2008) used urban, suburban, and rural categories but did not explicitly define each category. Burton *et al.* (2009), Vallet *et al.* (2008), and Heckmann *et al.* (2008) all used a categorical gradient, but in each of the studies the categories used were well quantified based on specific combinations of land cover or land use, etc. The study of Vallet *et al.* (2008) even utilized PCA in the quantification of the sites as urban or rural.

All of the studies created a land use / land cover gradient of the study areas, with Heckmann *et al.* (2008) specifically defining only a development gradient.

The studies of Vallet *et al.* (2008) and Pouyat *et al.* (2008) were direct comparative studies between cities. Burton *et al.* (2009) specifically used plant traits on order to allow comparisons in other urban environments. Heckmann *et al.* (2008) related the results of their study to findings of other studies and Dumas *et al.* (2008) implied that their techniques could be used as a basis to effectively study a whole spectrum of interests as well as comparing it to other cities.

Dumas *et al.* (2008) used only broad measures of urbanization to create their urban-forest interface maps, with all of the other studies using a combination of broad and specific measures. However, none of the studies explicitly evaluated their measures used on the basis of applying the same set of measures in another study of the same kind. The study of Dumas *et al.* (2008) only focused on the creation of their interface map with no biological data tested. All four of the remaining studies correlated their measures used with environmental data (in some cases) and biological input data, with a subsequent evaluation of the most appropriate measures for each study. These four studies also determined to what extent the selected measures and variables influenced ecological patterns or indicated processes driving these patterns.

Burton *et al.* (2009) was the only study to specifically address how the plant traits they identified might influence ecological processes. Vallet *et al.* (2008), Pouyat *et al.* (2008) and Dumas *et al.* (2008) all mentioned influences on ecological processes or processes driving ecological patterns but none of them explicitly discussed or determined it.

In the evaluation of these five studies trends can be observed on the gradual shift from solely observational individualistic studies towards more comparative studies. The utilization of some form of aerial photographs or satellite imagery in all the studies indicates the widely accepted use and uptake of these technologies into urban ecological studies. Almost all of the studies correlated the urbanization measures or indices with biological data, allowing better insights into appropriate measures and the identification of underlying urbanizational processes driving biological observed patterns. However, all of the studies used the urbanization measures solely to best quantify the study area in question with no regard or mention of its ease of calculation or applicability to a wider audience of studies. None of the above mentioned studies could be directly compared with regards to their broad urbanizational context and the individual quantification of the gradient in each respective study.

Therefore, the call of McDonnell and Hahs (2008) to advance the utilization of a common set of measures to integrate studies and allow more direct comparisons between studies is still a major limitation in the advancement of the field of gradient studies as well as particularly addressing the applicability of the myriad of urbanization measures used by studies. The direct investigation of the processes driving species patterns and distributions should also be addressed in the pursuit of unifying and integrating results and trends observed in biodiversity worldwide to allow the sustainable utilization of cities.

2.6 Summary

The development of urban ecological studies in South Africa as a largely unexplored field was discussed to emphasize the importance of these studies in quantifying the influence of growing urbanization in a species rich country facing typical challenges of poverty, equity and economic aspects overshadowing urban nature conservation. However, the unique spatial organization and urban development patterns in South African cities offer an exceptional opportunity to describe a unique and complex urban-rural gradient.

McConnachie *et al.* (2008) described the delineation of the urban areas as the respective town boundaries in their study. To determine the boundaries of the urban areas they simply digitized the town boundaries as the line joining the edges of the delimited properties for the outermost buildings of the town. The size of the urban areas in the current study was determined in this manner, with the digitized outline used frequently in discussing patterns and certain aspects of the greater Klerksdorp area. This was the outline used in Figures 2.2 and 2.3. For the remainder of the study the term 'urban area' will refer to this digitized outline of the greater Klerksdorp area.

This chapter serves as a broad overview and context in which to evaluate the findings of the current study. The rest of the dissertation will focus on testing the urbanization measures used in Melbourne (Hahs and McDonnell, 2006) in the greater Klerksdorp area and determining how urbanization influences and drives plant species diversity patterns. The following chapters will also include additional and more pertinent literature studies for specific aspects addressed in the current study.

Chapter 3: Satellite land cover classification

3.1 Introduction

"Probably no combination of two technologies has generated more interest and application over a wider range of disciplines than the merger of remote sensing and space exploration." (Lillesand and Keifer, 2000)

There exists a wide and varied range of ecological studies that utilize remote sensing as part of the research process (Aplin, 2005; Fassnacht *et al.*, 2006). Moreover, recent developments in technology and modelling techniques meant that remote sensing is more capable than ever to benefit ecology (Aplin, 2005). Examples of ecological studies utilizing satellite imagery include: aboveground biomass measurements in forests (Anaya *et al.*, 2009); rodent communities composition and diversity in urban ecosystems (Cavia *et al.*, 2009); a satellite-based index for habitat monitoring at continental scales (Coops *et al.*, 2009); abundance of ants in disturbed landscapes (Graham *et al.*, 2009); urban sprawl in peri-urban zones (Huang *et al.*, 2009); estimating ecological status and change of riparian zones (Ivits *et al.*, 2009); assessing rates of forest change and fragmentation (Li *et al.*, 2009); risk assessment of malaria re-emergence (Linard *et al.*, 2009); mapping benthic faunal communities (Mair *et al.*, 2009); role of vegetation fragmentation on aquatic conditions (Shandas and Alberti, 2009); and the modelling of nest-site occurrence for the Northern Spotted Owl (Stralberg *et al.*, 2009).

3.1.1 Overview on remote sensing concepts

Campbell (2006) defines remote sensing as *"the practice of deriving information about the Earth's land and water surfaces using images acquired from an overhead perspective, by employing electromagnetic radiation in one or more regions of the electromagnetic spectrum, reflected or emitted from the Earth's surface."*

Passive satellite sensors, such as the type used in the current study to obtain an image of the study area, only detect energy naturally reflected or emitted by an object (Wade and Sommer, 2006). The satellites specifically detect electromagnetic energy, acquiring their data on the way various earth surface features emit and reflect the electromagnetic energy (Lillesand and Keifer, 2000). Electromagnetic waves are commonly categorized by their wavelength location, in μm, within the electromagnetic spectrum in remote sensing (Figure 3.1). The major ranges utilized for earth resources sensing are between about 0.4 and 12 μm encompassing the visible and infrared range (Richards, 1993).

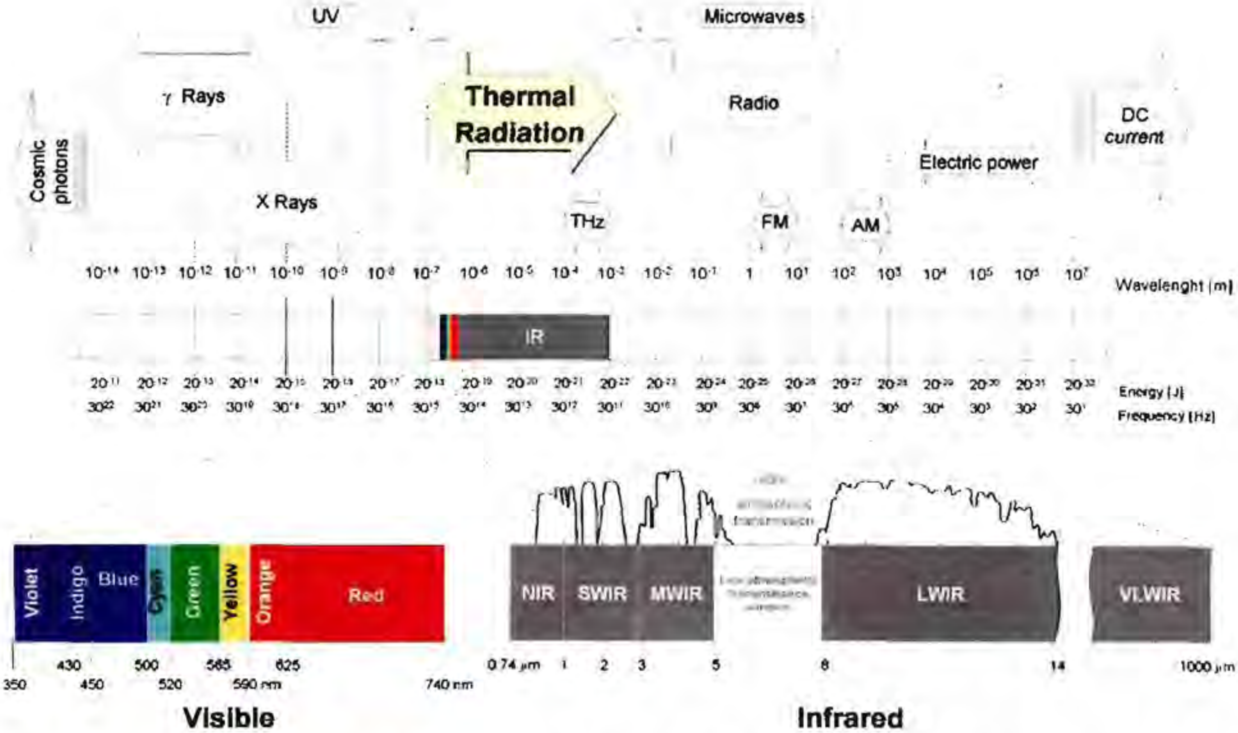


Figure 3.1: The electromagnetic spectrum. Enlarged are the thermal radiation portions of the electromagnetic spectrum mainly used in satellite remote sensing. The atmospheric windows for atmospheric transmission of the electromagnetic spectrum are shown for the infrared (IR) portion of the electromagnetic spectrum (Castanedo, 2005).

In the atmosphere, some gases such as water vapour, carbon dioxide and ozone effectively absorb electromagnetic energy at specific wavelengths making it impossible to detect them at those wavelengths. Therefore, due to this tendency of absorption in specific wavelength bands the atmospheric gases strongly influence which spectral bands are used with any given remote sensing system. The wavelength ranges in which the atmosphere is particularly transmissive of energy is termed atmospheric windows. In Figure 3.1 these atmospheric windows are shown for the infrared portion of the electromagnetic spectrum. This illustrates why spectral bands of remote sensing sensors have certain band widths chosen depending on the features of interest to be analyzed (Lillesand and Keifer, 2000).

The main source of electromagnetic energy detected by passive satellite sensors is the sun; however the earth also emits lower amounts of its own energy. When electromagnetic energy is incident on an earth surface feature, three energy interactions are possible, namely: the incident energy fractions can be reflected, absorbed and/or transmitted (Lillesand and Keifer, 2000). The energy reflected, absorbed and transmitted will vary proportionally for different earth features, depending on their material type and condition. The composition, texture, and shape of matter affect these energy interactions (Kearns, 2006). It is precisely these energy interaction differences that permit the distinguishing of different features on an image. However, each feature is wavelength dependent which means that even with a given feature type the proportional energy interactions will vary at different wavelengths. Therefore, two features can be impossible to be differentiated in one spectral range and be completely different in another wavelength band (Lillesand and Keifer, 2000). For this reason multispectral remote sensor satellites detect in three or more spectral ranges allowing differentiation of more earth features. Each spectral range in which a satellite detects features is called a band; therefore a satellite with seven multispectral bands such as the Landsat 7 sensor (Campbell, 2006) will detect electromagnetic energy in seven different spectral ranges.

Most remote sensing systems operate in wavelength ranges where reflected energy predominates and therefore it is important to know the reflectance properties of earth surface features (Lillesand and Keifer (2000). The surface roughness of the object defines the geometric manner in which an object reflects energy. Two types of reflectance can occur namely: specular, where the surface is smooth relative to the wavelength, redirecting most of the energy in a single direction; and diffuse, where the surface is rough relative to wavelength and the energy is scattered more or less equally in all directions (Campbell, 2006). Lillesand and Keifer (2000) state that diffuse reflections of features contain spectral information on the 'colour' of the reflecting surface, whereas specular reflections do not. Therefore, the diffuse reflectance properties of landscape features are most often measured. By measuring the reflectance characteristics of features unique spectral response curves can be drawn, showing in which wavelengths the feature reflects the best. Most earth features have unique spectral response curves allowing differentiation between features based on their spectral response curves. The specific spectral response characteristics of a feature are often termed its spectral signature (Campbell, 2006). Campbell (2006) further states that in practice, it is now recognized that the spectra of features change both over time, such as plant growth stages through the seasons, and over distance, such as proportional changes of specific tree species in a forest in different locations.

Electronic sensors onboard the satellite platforms subsequently receive the electromagnetic energy reflected from features as an electrical signal that corresponds to the energy variations in the original scene. For each spectral band of the sensor a digital image is created consisting of a two dimensional array of discrete picture elements called pixels (Lillesand and Keifer, 2000). The size of an individual pixel is determined by the spatial resolution of the satellite sensor, in other words, the amount of detail observed by the sensor. Spatial resolution can vary from a couple of centimeters to kilometers; the satellite image used in the current study has a spatial resolution of 10 m for its individual multispectral bands. In satellite imagery the spatial resolution of the pixel is also termed the grain size of the image. Each pixel in an image, matching an exact ground area, is awarded a digital number (DN) corresponding to the average radiance or brightness measured in each pixel. The radiance measured is the amount of energy recorded by the sensor on a binary scale dictated by the bit depth of the image (Kearns, 2006). An 8 bit image will allow a range of brightness values from 0-255. A value of 0 will be awarded to the darkest feature and 255 to the brightest (reflecting the most electromagnetic energy), with all the rest of the feature radiances arranged in between. This is repeated for each band. Depending on the amount of spectral bands a sensor possesses, a specific ground location on the produced satellite image might have four different digital numbers for the corresponding pixel in each image band. Therefore, unique combinations of these different image bands, with subsequent different digital numbers, allow the image analyst to differentiate between different earth features, depending on the band combinations used. This is why satellite sensor specifications typically state what type of features can be distinguished with each spectral band, for example the red (0.63-0.69 μm) band of Landsat 4 is designed to sense in a chlorophyll absorption region aiding in plant species differentiation (Lillesand and Keifer, 2000).

Another important concept in remote sensing is the swath width of the satellite sensor. Swath is the extent of the entire area that the satellite captures per coverage; this is similar to the extent of the produced digital satellite image. For example, the swath of IKONOS satellite sensor is 11.3 km nadir (viewed vertically beneath the satellite sensor) (GeoEye, 2006). In general, the smaller the swath of the satellite sensor the better is its spatial resolution. The spatial resolution of the IKONOS satellite mentioned above is 0.82 m (GeoEye, 2006), whereas the swath of Landsat 7 is 185 km with the majority of the bands having a spatial resolution of 30 m (Williams, 2008).

When the satellite image is subsequently imported into a mapping software program such as ArcView 9.2 (ESRI, 2006a) a map layer is created from the raster image. In ArcMap a single band of data can be displayed or a colour composite image can be created from the available multiple bands. A combination of any three of the available bands in a multiband raster dataset can be used to create Red-Green-Blue (RGB) composites. Figure 3.2 illustrates the creation of a RGB composite image. This display constraint in ArcMap of ArcView 9.2 compels the image analyst to choose between the available satellite bands and decide which band combinations would be the most suitable for the mapping task.

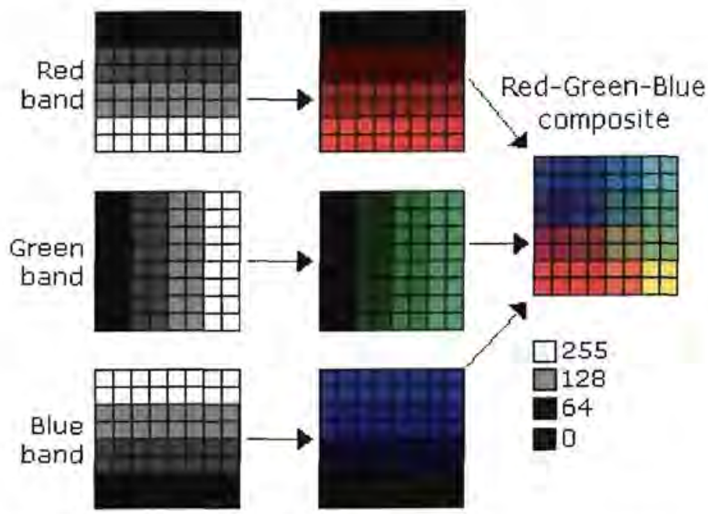


Figure 3.2: Illustration of the creation of a Red-Green-Blue (RGB) composite image for rasters with multiple bands of 8 bit scenes (ESRI, 2008).

The purpose for the satellite image in the current study is to create a land cover map. In order to do this the satellite image will be classified in ArcMap with the use of the Spatial Analyst extension. Classification is the process of sorting or arranging entities into groups or categories on a map (Wade and Sommer, 2006). In satellite image classification two types of classification processes can be used, namely: supervised classification or unsupervised classification (Lillesand and Keifer, 2000). Supervised classification is termed as such because the image analyst supervises the pixel categorization process by specifying training sites of the various land cover types present. Training sites are compiled numerical 'interpretation keys' that describes the spectral attributes for each feature type of interest. Each pixel in the data set is then compared numerically to each category in the training site interpretation key and labeled with the name of the category it most resembles. In unsupervised classification the data are first classified by aggregating them into natural spectral groupings or clusters of pixels present within the image, based on the individual pixel digital numbers per band (Campbell, 2006). The image analyst then determines the land cover identity of the spectral groups by comparing the classified image data to ground reference data (Lillesand and Keifer, 2000). When elements of both supervised and unsupervised classification are combined the procedure is termed hybrid classification. Hybrid classification is usually done to improve the accuracy or efficiency or both of the classification process (Lillesand and Keifer, 2000).

3.1.2 History and ecological applications of remote sensing

The first major US symposium on Remote Sensing of Environment was convened on February 1962 (Anon, 1964 as quoted by Leimgruber *et al.*, 2005). However, ecological and conservation oriented presentations, such as those listed at the beginning of the introduction, only began to appear in the proceedings of remote sensing symposia in any significant numbers in the late 1980s (Leimgruber *et al.*, 2005). The rapid growth and increasing demand of remote sensing software can be illustrated with the approximately 2647 satellites already listed in an index of all successfully launched satellites to date; encompassing commercial earth observation satellites, weather satellites, telecommunications and sun observations (The Satellite Encyclopedia, 2008). In his paper on commercial earth observation satellites, which ecological researchers mainly use, Fritz (1996) anticipated this with his estimation that roughly 100 new satellites will be launched in the period between 1996 and 2006. With time, remotely sensed data has subsequently pervaded increasing numbers of scientific disciplines.

Landsat satellite images specifically, (e.g. Hahs and McDonnell, 2006; Wimberly and Reilly, 2007; Luedeling and Buerkert, 2008; Phua *et al.*, 2008; Röder *et al.*, 2008) are very popular in environmental research. This is largely due to the low cost of Landsat images, the large extent of the images, its relative accuracy and spatial resolution, the large amount of spectral bands and the fact that images of the entire globe exist covering extended time periods of up to 30 years allowing research of change over time (Leimgruber *et al.*, 2005). Most other satellite monitoring programs such as SPOT (Système Pour l'Observation de la Terre) or IRS (Indian remote sensing) do not provide such extensive low-cost multiband imagery and in most cases only images of limited time periods exist. However, studies do utilize satellite images such as SPOT (Cohen and Spies, 1992; Gong *et al.*, 1992; Gluch, 2002). The choice of specific satellite imagery for research are usually concurrent with its local availability and cost as well as its suitability to the research question asked. Furthermore, it is also strongly subject to the availability of software and the limitations in computing power of the researcher using the data. As technology improves and superior resolution images are available, the use of these newer higher resolution satellite images becomes more common. This can, for example, be illustrated with the launch of SPOT 5 in 2002 (CNES, 2007), where increasing research since then has utilized SPOT 5 (2.5-10 m resolution) instead of the traditional Landsat (30 m resolution), especially when only one current satellite image of the study area in question is needed (Jacquin *et al.*, 2005; Inglada, 2007; Zhao *et al.*, 2007; Berthier and Tautin, 2008).

Kerr and Ostrovsky (2003) discussed three main uses of remote sensing in ecological applications namely: land cover classification, integrated ecosystem measurements and change detection. Land cover classification increasingly formed the basis and contextual setting of many urban environmental studies (e.g. Stefanov and Netzband, 2005; Hahs and McDonnell, 2006; Gill *et al.*, 2008; Pennington *et al.*, 2008; Wu *et al.*, 2008). However, Rogan and Chen (2004) declared that the minimum spatial resolution requirement for a USGS level 1 classification (USGS classification levels was proposed by Anderson *et al.*, 1976), namely: urban, agricultural land, rangeland, forest, water, wetland, barren land, tundra, and perennial snow or ice of urban landscapes is 20-100 m. Therefore, even though both SPOT 5 (5 m, 10 m, 20 m) and Landsat (30 m) fall in the category of medium resolution satellites (5-250 m resolution) as described by Rogan and Chen (2004), SPOT 5 would deal better with classification of highly fragmented urban landscapes. Aplin (2005) reminded that the benefit of finer spatial resolution imagery over coarser spatial resolution imagery for ecological investigation is obvious, as the spatial resolution increase the accuracy with which small objects are identified and characterized also increase.

Consequently, SPOT 5 was utilized in the current study firstly, because it was obtained free of charge (source in Table 3.1) and secondly, because of its improved spatial resolution.

Table 3.1: Details of the SPOT 5 HRV image used to classify the greater Klerksdorp area.

SOURCE	*CSIR, Satellite Application Centre (http://www.sac.co.za)*
DATE OF ACQUISITION	*3 September 2007*
GEOGRAPHIC PROJECTION	*UTM35S WGS84 datum*
IMAGE ADJUSTMENT	*Level 3 ortho rectified*
IMAGE NUMBER	*51304040703030815331J*

The unavailability of accurate current data in the correct format compelled the choice of satellite imagery to create the land cover map of the study area. The details of the satellite image used for the classification process are given in Table 3.1.

The first objective listed in Chapter 1 for the study is to use SPOT 5 satellite imagery and GIS techniques to determine which of the demographic-, physical variables and landscape metrics identified in Melbourne, Australia can be used to quantify urbanization in the greater Klerksdorp area. This chapter will partly fulfil the objective by classifying the satellite image of the greater Klerksdorp area that will be used as input for the calculation of the urbanization measures in Chapter 4.

3.2 Methods

The greater Klerksdorp area to be classified is shown in Figure 3.3. Satellite classification was done in ArcView 9.2 (ESRI, 2006a) with the Spatial Analyst extension, due to the unavailability of remote sensing software. However, the exclusive use of ArcView 9.2 and Spatial Analyst for image classification can nevertheless provide valuable lessons to researchers with limited software resources. This would provide a more widely available option to researchers as well as being less expensive.

Table 3.2 lists the spectral band information and the general uses of the chosen satellite sensor. SPOT 5 possesses four multispectral image bands with an additional panchromatic 2.5 m resolution band. To use the satellite image as input for the land cover map creation, the best combination for feature extraction of the four multispectral bands for the RGB composite raster had to be chosen.

All possible three band combinations were explored, adding up to twelve possible combinations which were analyzed. Each of the twelve band combinations of the 60 km x 60 km swath width SPOT 5 image was extracted with a mask of the study area in Spatial Analyst to minimize the classification area to only the relevant 30 km^2 area. At this stage it is important to note that Richards (1993) alternatively suggested that when the data consists of more than three bands and a decision has to be made as to which bands to discard, principal component analysis (PCA) can be used to identify highly correlated bands and those describing the most of the variation in the image. In ArcMap of ArcView 9.2 only the three bands chosen in the display layer can be used as an input for a PCA. Thus the analyst must still choose the best combinations. Therefore, due to this limitation, PCA was not done and all the twelve band combinations as discussed earlier were analyzed.

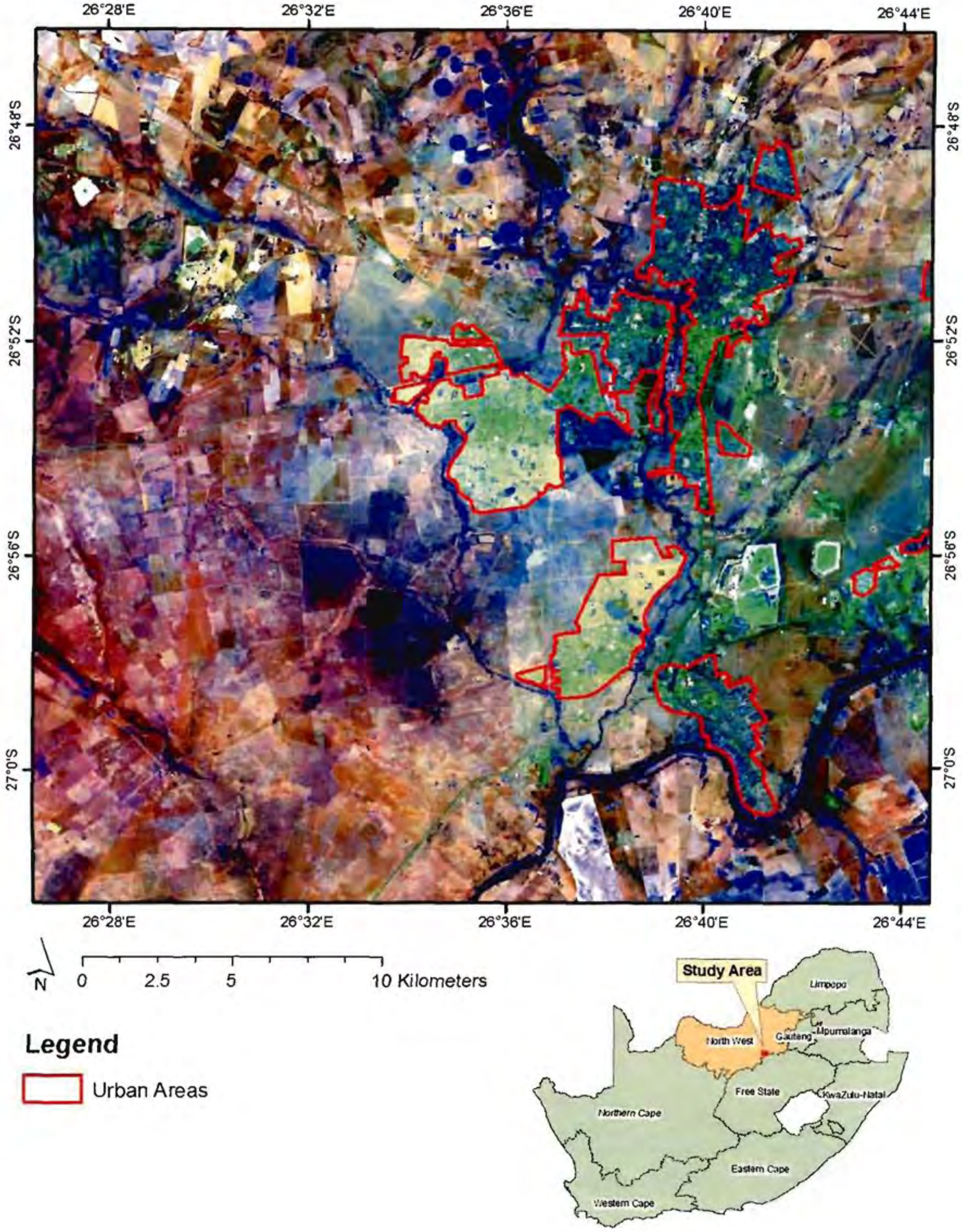


Figure 3.3: Map of the greater Klerksdorp area, showing the SPOT 5 satellite image with the band combinations 423 (RGB) used for the classification of the image. The red outline indicates the urban areas. An overview map shows the location of the greater Klerksdorp area in South Africa.

The nature of the classification typology used for land cover classification determines the questions that can be addressed in a study (Hahs, 2006). Broad-scale questions such as those regarding how ecological systems vary between different forms of land use usually need only a simplified typology to answer research questions (Hahs, 2006). Therefore, only basic land cover classes are required for the classification of the study area to form a broad contextual setting in which to quantify the ecological patterns and processes and as input for some of the urbanization measures. This ensures that land cover maps of different areas can be easily compared, and that the classified image does not add to the complexity of the image for patch delineation and other landscape metrics as used in Chapter 4.

Table 3.2: The spectral band information listed for the SPOT 5 satellite (Spot Image, 2005). The specific multispectral band uses as correlated with their corresponding Landsat bands (Lillesand and Keifer, 2000) and the general uses of SPOT 5 are also listed (Satellite Imaging Corporation, 2007).

SPECTRAL RESOLUTION	*Panchromatic: 0.48-0.71 μm* *Band 1: 0.50-0.59 μm* *Band 2: 0.61-0.68 μm* *Band 3: 0.78-0.89 μm* *Band 4: 1.58-1.75 μm*
SATELLITE BANDS SPECIFICATION	*2 panchromatic: combined to create a 2.5 m resolution panchromatic image band.* *3 multi spectral bands: green, red, narrow infrared (10 m resolution).* *1 short-wave infrared band (10 m resolution).*
SPECIFIC MULTISPECTRAL BAND USES	*B1: Vegetation discrimination, cultural feature identification.* *B2: Plant species differentiation, cultural feature identification.* *B3: Vegetation monitoring, soil moisture discrimination, delineating water bodies.* *B4: Vegetation and soil moisture content monitoring.*
GENERAL USES OF SPOT 5	*Medium-scale mapping.* *Urban and rural planning.* *Oil and Gas Exploration.* *Natural disaster management.* *Stereo pair imagery.*

The classification typology can be decisive to the values of the measurement outcomes. Hahs and McDonnell (2006) used five classes to classify the study area in Melbourne, Australia. Consequently, the chosen classes in the current study resembled those of Hahs and McDonnell (2006) with urban, grass, trees, water, and soil. However, an additional class of mining was added due to the extensive mining activities in the current study area.

3.2.1 Classification process

In the satellite image bands of the study area some feature spectral signature similarities occur. This is a result of similarities in the spectral nature of some feature parts due to their respective high albedos namely: bare soils, tailings dams and large buildings. Parts of these features form highly reflective surfaces that are responsible for the white fraction of specifically the panchromatic band image. For this reason, it would have been very difficult to assign accurate training sites needed for the use of supervised classification. Therefore, unsupervised classification was chosen to create the land cover map of the greater Klerksdorp area. Campbell (2006) however reminds us that two of the disadvantages of unsupervised classification are the fact that spectral properties of specific informational classes will change over time, such as winter vegetation and the summer growth season or irrigated crops and dry crops; and that unsupervised classification identifies spectrally homogenous classes within the data that do not necessarily match the informational categories of interest to the analyst. To overcome this problem all possible class combinations was tested to see which of the class delineations of the unsupervised classification output corresponded best to the actual features. The six broad classes used in the current study will simplify the potential errors associated with unsupervised classification.

The chosen twelve, three band combination rasters were used as input for the unsupervised classification. The unsupervised classification process was started in Spatial Analyst by using the Iso Cluster tool to create signature files which is needed to run a maximum likelihood classification (MLC). The MLC then generates output classified rasters based on the Iso Cluster specifications. The Iso Cluster tool was run with its default settings; respectively changing only the number of classes generated. The Iso Cluster tool was run for six, seven, eight, nine and twelve classes for each of the twelve three band combination images. This was done to test whether recombining classes of , for example, a twelve class classification would generate a more accurate six class reclassified image than only running a six class classification from the start. A total of sixty signature files were generated in this manner. Each of the sixty generated signature files were then used as input for the MLC tool; again used with its default settings. Sixty classified rasters were consequently generated consisting of six, seven, eight, nine and twelve classes per three band combination.

During reconnaissance surveys and the actual vegetation surveys, which took place in March 2008, eighty six GPS waypoints were collected throughout the whole study area, representing different types of land cover. These waypoints were imported into ArcView 9.2, labelled and used as ground reference information to assist in the visual accuracy assessment of the sixty classified images. Visual accuracy assessments were done using the panchromatic band image of the study area overlaid over the respective classified images. The panchromatic band has a spatial resolution of 2.5 m and replaces the need for an aerial photograph of the study area for use in visual and quantitative accuracy assessments of the images.

Figure 3.4 illustrates the successive steps taken in the visual accuracy assessment of two of the classified rasters, namely: 421 and 423. The first step (A) shows the two classified images 421 and 423. The classification is the raw twelve class output generated by the unsupervised classification. Step (B) shows how the classes were visually interpreted with the preliminary reclassification of the classes by allocation the same colour to similar features based on the GPS waypoint references. Step (C) shows the eighty-six GPS waypoints used to aid the identifications and the panchromatic image of the study area. Step (D) illustrates how the panchromatic band was overlain with a 50 % transparency over the classified rasters to verify whether the class delineations as created by the unsupervised classifications are accurate and correspond to actual feature groups.

Early investigations showed that the twelve class classifications conformed best to actual feature delineations. Of the twelve images four were chosen to use in further classifications. The chosen four are the twelve class classifications of the band combinations 341, 423, 321, and 421 respectively representing RGB. Table 3.3 shows a comparison of the different features represented by each class for the four chosen classified images as determined by use of the panchromatic band and the GPS waypoints for referencing.

Table 3.3: A comparison of the different output classes generated by the MLC classification process for 12 classes of the chosen band combinations of 432, 341, 321, and 421.

CLASS	IMAGE 423	IMAGE 341	IMAGE 321	IMAGE 421
1	*Water*	*Water*	*Water*	*Water*
2	*Trees*	*Trees*	*Trees*	*Trees*
3	*Trees*	*Trees*	*Trees*	*Trees*
4	*Urban*	*Trees*	*Grass*	*Trees*
5	*Grass*	*Grass*	*Trees*	*Grass*
6	*Trees*	*Grass*	*Grass*	*Grass*
7	*Grass*	*Grass*	*Grass*	*Grass*
8	*Grass*	*Grass*	*Grass*	*Grass*
9	*Grass*	*Grass*	*Grass*	*Trees*
10	*Grass*	*Urban*	*Soil*	*Urban*
11	*Soil*	*Soil*	*Urban*	*Soil*
12	*Mining*	*Mining*	*Mining*	*Mining*

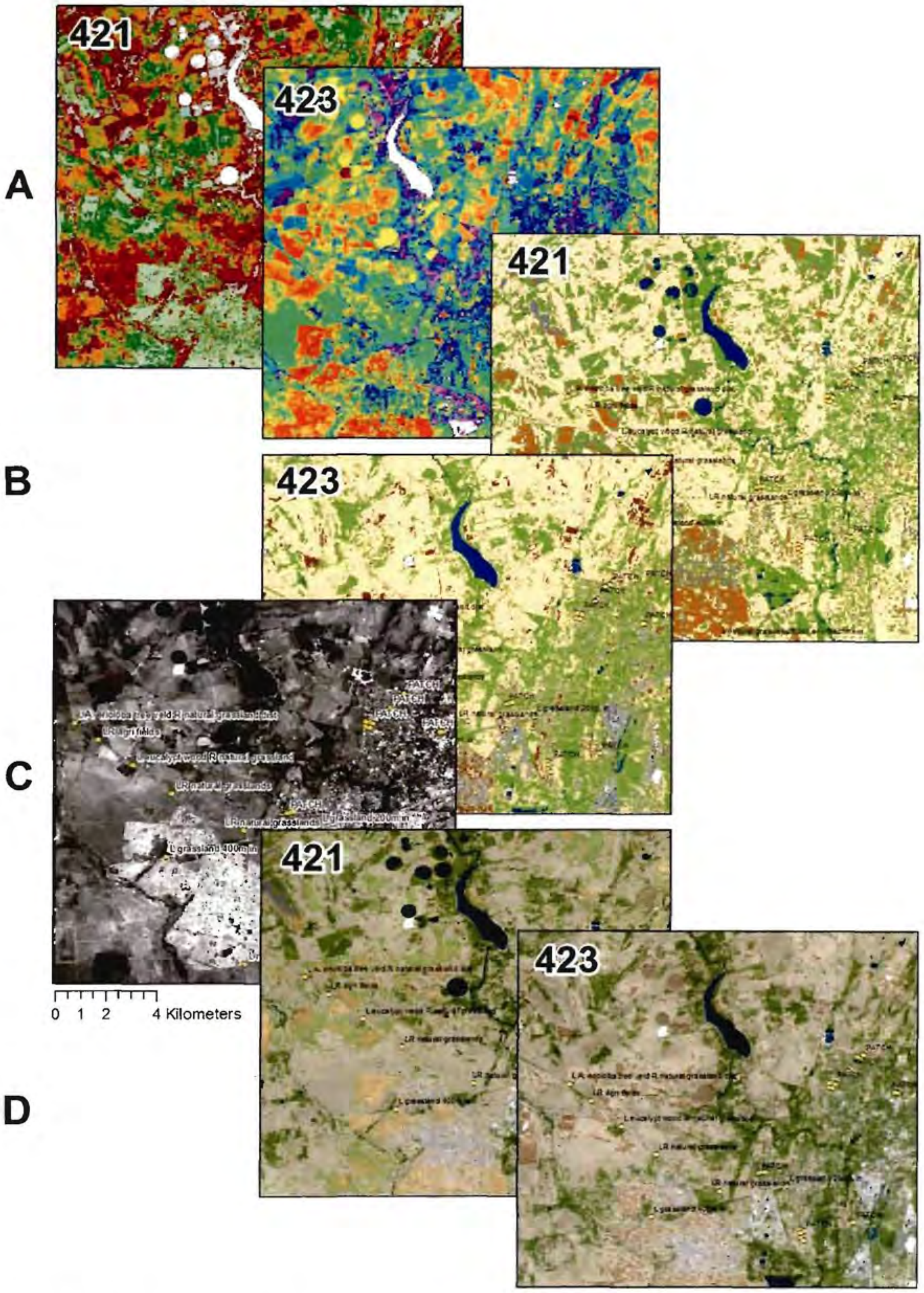


Figure 3.4: The visual observation accuracy process used in the classification of the different RGB composite satellite images. (A) Shows the raw twelve class classified raster images; (B) the preliminary visual interpreted reclassification of the classes; (C) the GPS waypoints with the panchromatic band image, and (D) shows the GPS waypoints and the overlain panchromatic band on the classified images to verify the accuracy of the classified images.

All four rasters were reclassified to six classes by merging them according to the features they represent, as shown in Table 3.3, with the Reclassify tool in Spatial Analyst. Figure 3.5 illustrates the reclassification procedure as done with the band combination 423 classified raster. Steps A-D was followed, namely: classes 3 and 6 were merged as shrubs and classes 5, 7, 8, and 10 were merged as grass (A); in the second step the high moisture content vegetation class were merged with the grass class (B); thirdly, it was decided to merge the shrub class with the trees, as a distinction between the two is not important for the purpose of the current study (C); the last step shows the final six classes namely: water, trees, urban, grass, soil, and mining (D).

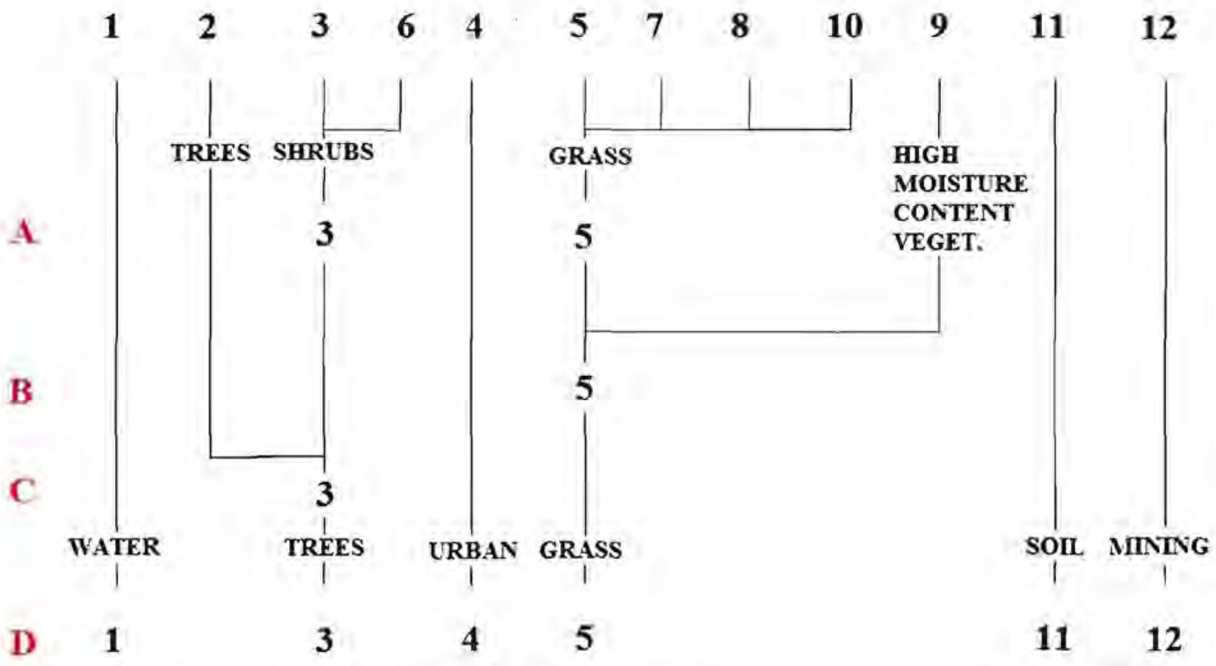


Figure 3.5: Schematic representation of the reclassification process from twelve to six classes for the band combination 423 satellite image of the greater Klerksdorp area.

Figure 3.6 illustrates the proportional size distributions for each of the six classes for the four reclassified images. Accuracy assessment was done for each of the four classified rasters to ensure that the most accurate image is used for the land cover map of the greater Klerksdorp area. The accuracy assessment process used is discussed in detail for land cover map A below. The same ground truthing point layer was used for each raster image respectively. The overall accuracies of the raster images determined which one would be used further. The overall accuracies were: band combination 341, 86 %; combination 423, 88 %; combination 321, 74 %; and combination 421, 76 %. Therefore, the six class classified image of band combination 423 were used further, being the most accurate classification of the greater Klerksdorp area. As listed in Table 3.2 bands 423 of SPOT 5 imagery are specifically used for vegetation differentiation and monitoring, water body delineation and man-made feature identification, which correlates well to the class types used for the current study.

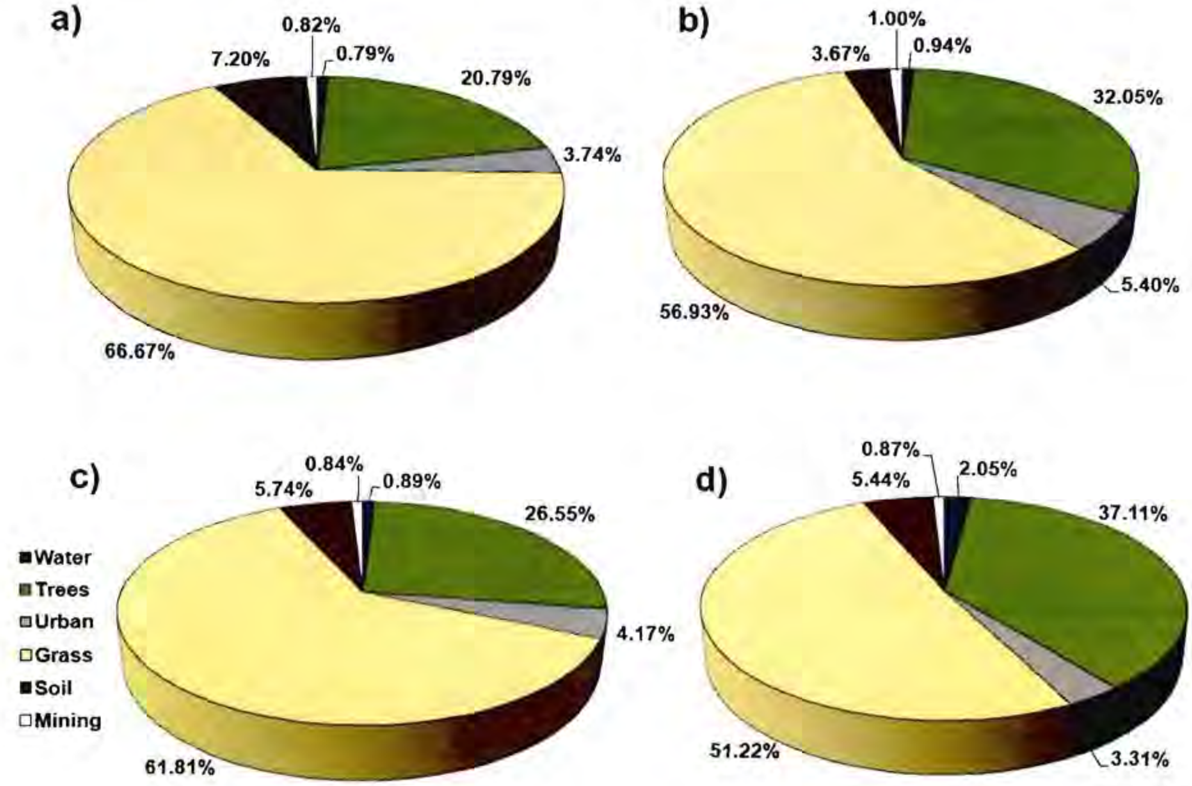


Figure 3.6: Pie charts of the proportional class size distributions for each of the six classes of the four reclassified images of the greater Klerksdorp area, RGB combinations 341 (a), 321 (b), 423 (c), 421 (d).

As a final step, a majority filter was run using its default settings to reduce the amount of isolated single pixel classes in the image resulting from the classification process. The Majority Filter tool identifies single pixels within the raster image surrounded by other land cover types and assigns it to the most represented class of the surrounding eight pixels. The filter was only run once to preserve the ensuing isolated pixels in the urban areas formed as a result of the highly fragmented nature of the urban areas. The output raster of the filter process was called land cover map A.

3.2.2 Accuracy assessment

Accuracy assessment done in the current study will follow the steps as described by Lillesand and Kiefer (2000). They suggested that for an accuracy assessment of a land cover map to be representative at least 50 points per class is needed for the ground truthing, sampled in a stratified random manner. Land cover map A has six different classes therefore, 300 points of known land cover classes was created. The ground truthing points interpreted from the 2.5 m panchromatic band satellite image was overlain on the land cover map and the values of the land cover map extracted with the Extract Values to Points tool in Spatial Analyst. This tool extracts the cell values of a raster based on a set of overlaying points. The accuracy assessment of land cover map A (Table 3.4 in the results and discussion) showed that the respective classes of grass, soil and mining had misclassifications where the pixels were incorrectly classified.

Map investigations showed that urban areas with large concrete buildings also classified in the mining class. This was not represented in the error matrix as samples were randomly chosen per class missing these cases. The spectral nature of the mining tailings dams and waste rock dumps was in some cases identical to bare agricultural soil and urban areas, due to compositional and surface texture similarities. Therefore, missclassified pixels occurred in those three classes. Thus, it was decided to run another classification where the tailings dams and rock dumps were digitized and masked out of the raster to produce a more accurate image. This could easily be done as all tailings dams and rock dumps are simple geometric forms, easily identified and digitized from the panchromatic band image.

3.2.3 Land cover map B

Hybrid classification was used to create land cover map B, as manual digitizing of the mining areas with subsequent steps were done during the unsupervised classification. The study area satellite image, with band combination 423, was used as the initial input of the second classification process. Mining areas were digitized with the use of the Auto-complete polygon tool. This creates and image where the entire study area are included in the created polygon except the mining areas. This then allows the extraction of the areas of the satellite image that does not include mining areas. Therefore, the raster with the removed mining areas were used as input for the classification process. Figure 3.7 shows a schematic representation of the classification process to produce land cover map B.

The classification process, as done with land cover map A, was repeated with the raster where the mining class was masked out. Unsupervised classification was repeated with twelve classes using Iso Cluster and Maximum Likelihood Classification. It was then reclassified to five classes instead of six as in land cover map A. A majority filter was run on the reclassified image. To remerge the mining class, a solid raster was created of the study area and all the pixels given the same value to represent the mining class. The Single Output Map Algebra tool was then used to merge the classified image with the solid raster. The tool works in order of importance with the first specified raster being the most important. Therefore, the solid mining raster only filled in the gaps left by the masked out mining areas. Land cover map B was thus created as the output raster of merging the two raster images.

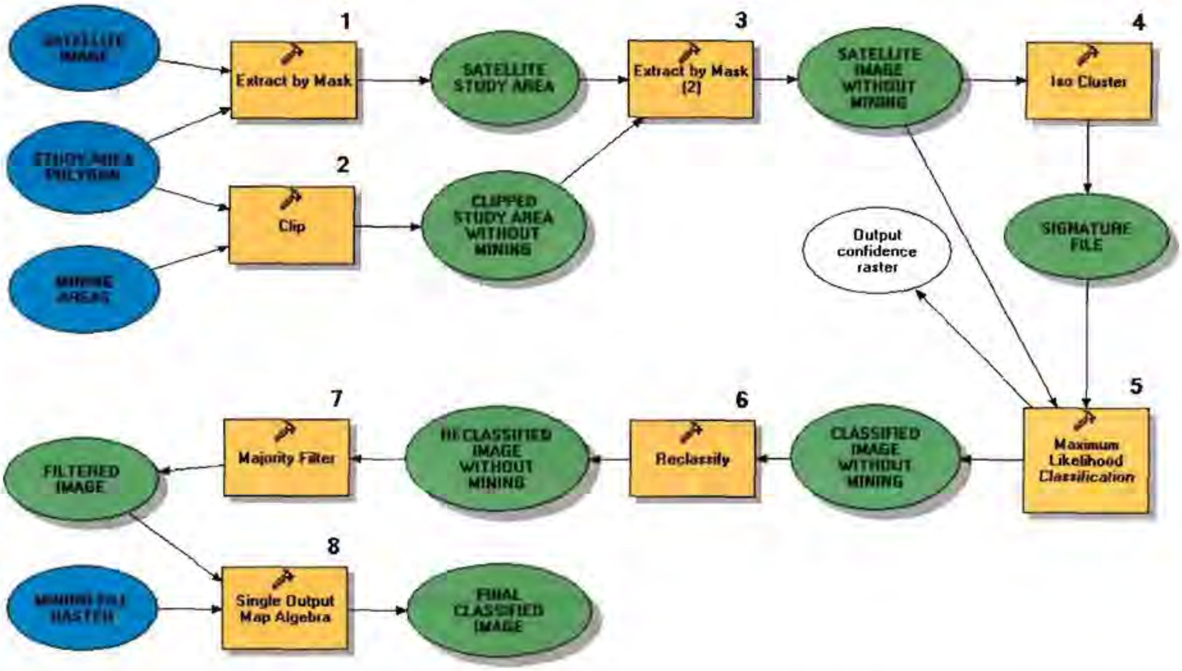


Figure 3.7: A schematic representation of the classification process of the final land cover map B of the greater Klerksdorp area as was done in Spatial Analyst, showing all the tools used. The process was modelled with the ArcView 9.2 ModelBuilder tool. The blue ellipses represent the initial input shapefiles. The tools are numbered to illustrate the sequence in which the steps were performed.

The accuracy assessment of land cover map B was done in the same way as with land cover map A, with the same 300 points used for ground truthing. (Table 3.4 in the results and discussion) This allowed a comparison of the overall accuracies deciding which land cover map will be ultimately used.

3.3 Results and discussion

3.3.1 Land cover map A

The overall classification produced with the use of SPOT 5 satellite imagery was very satisfactory. All six classes can clearly be distinguished. General fuzziness in the class delineations can be ascribed to the 10 m grain of the satellite image where some features are smaller than the grain or overlapping in some of the pixels. Before a detailed description is given of the classified land cover map Campbell (2006) reminded that accuracy assessment is a complex process with both conceptual and practical difficulties and that no clearly superior method exists of sampling an image for accuracy assessment. In creating the map it was clear that even though fifty sampling points per class were used as advised by Lillesand and Keifer (2000) other misclassifications could still be observed which could alter the user's and producer's accuracy of some of the classes, suggesting that more sampling points might be needed. One example of this is the urban class where large concrete buildings classed with mining due to spectral similarities. It is logical to assume that urban areas may pose classification problems as such a variety of component materials exists in urban environments such as concrete, asphalt, plastic, steel, and soils. Therefore, even though the error matrix (Table 3.4) shows very good results it is only at best a rough guide and should not be used as an absolute measure of the map accuracy. Lillesand and Keifer (2000) suggested that the number of samples for each class might be adjusted based on the relative importance of that class for a particular application. The urban areas could possibly have been sampled more intensively for the current study, however the accuracy assessment (Table 3.4) indicate that this is not necessary as the producer's accuracy for the urban areas is 96 %. The created land cover map is shown in Figure 3.8.

The fact that the vegetation was in a dormant state after the winter season resulted in some misclassifications in the trees and grass classes. It should be noted that most of the indigenous trees in the semi natural areas are more shrub-like, with the exotic *Eucalyptus* and urban planted trees predominantly larger. This also added to the mixed grass and trees pixels. An additional fact to be mentioned concerning the trees class is that due to their size and characteristic canopy structure underlying features in urban areas remain hidden from view with aerial views of the city. For example trees growing either side of a narrow road obstruct the view of the road and are then subsequently classified as trees rather than urban. This relationship can change significantly depending on the season in which the image is recorded (deciduous trees) and the resolution of the satellite sensor. Most of the residential houses in the greater Klerksdorp area are single storey structures with gardens and roadsides planted with trees. This can diminish the urban class, obscuring its real extent. In Chapter 4 the measures of urbanization are discussed with some relying on the percentage of urban land cover as input where the above mentioned difficulty can play a deciding role, namely the measures: percentage urban land cover and people per urban unit land cover.

The water bodies in the greater Klerksdorp area were very accurately delineated, with the only main problem occurring where burned vegetation areas were classified as water, resulting in a few such cases. Only the small streams and rivers were not captured by the 10 m spectral resolution, but the trees lining their banks show their location on the land cover map. The profusion of soils in the urban areas is a result of the informal housing settlements forming on the city fringes, where most of the poor and mostly unemployed city residents erect temporary shacks. Few tarred roads exist in these settlements containing hundreds of mostly single room houses. A cultural practice of clearing the yard surrounding the house of vegetation also contributes to the profusion of bare soils found here. Previous city planning in South African towns allocated separate residential zones to each ethnic group, with the whites allocated the prime urban residential areas (Davies, 1981). All other ethnic group zones were situated on the outskirts of the towns. These residential zones on the town fringes now form the influx points of hundreds of unemployed workers and their families. Due to the high poverty levels of the people, high density informal settlements develop where houses are built from any readily available materials. Currently, these areas are expanding at an alarming rate as rural workers flock to cities to find work and better living opportunities (Lemon, 2003).

Lastly, the mining class was added to capture the mining activities in the area. Both Klerksdorp and Orkney were founded as a result of the discovery of large gold deposits in the area and mining activities dominate the landscape surrounding these towns. The main problem that occurred in the classification of the mining activities is the difference in composition of the tailings dams e.g. rock piles and processed gold sludge. In addition, some of them were partly covered with vegetation and also had stagnant water on the top. Their simplistic shapes clearly identified them on the land cover map, but due to their fragmented classification, where some tailings dams had grass, urban, soil and water classes they contributed to synthetic patchiness not genuinely present in those areas. The overall accuracy of land cover map A is however very good, above the 85 % as considered acceptable by Herold *et al.* (2005).

Table 3.4: Error matrix of land cover map A of the greater Klerksdorp area, showing the producer's, user's and overall accuracy.

		Training set data							
		Water	Trees	Urban	Grass	Soil	Mining	Row total	User's accuracy
Classification data	Water	49	1	1				51	96 %
	Trees	1	46		11			58	79 %
	Urban			48				48	100 %
	Grass		3		39	11		53	74 %
	Soil			1		39	8	48	81 %
	Mining						42	42	100 %
Column total		50	50	50	50	50	50	300	
Producer's accuracy		98 %	92 %	96 %	78 %	78 %	84 %		
					Overall accuracy			88 %	

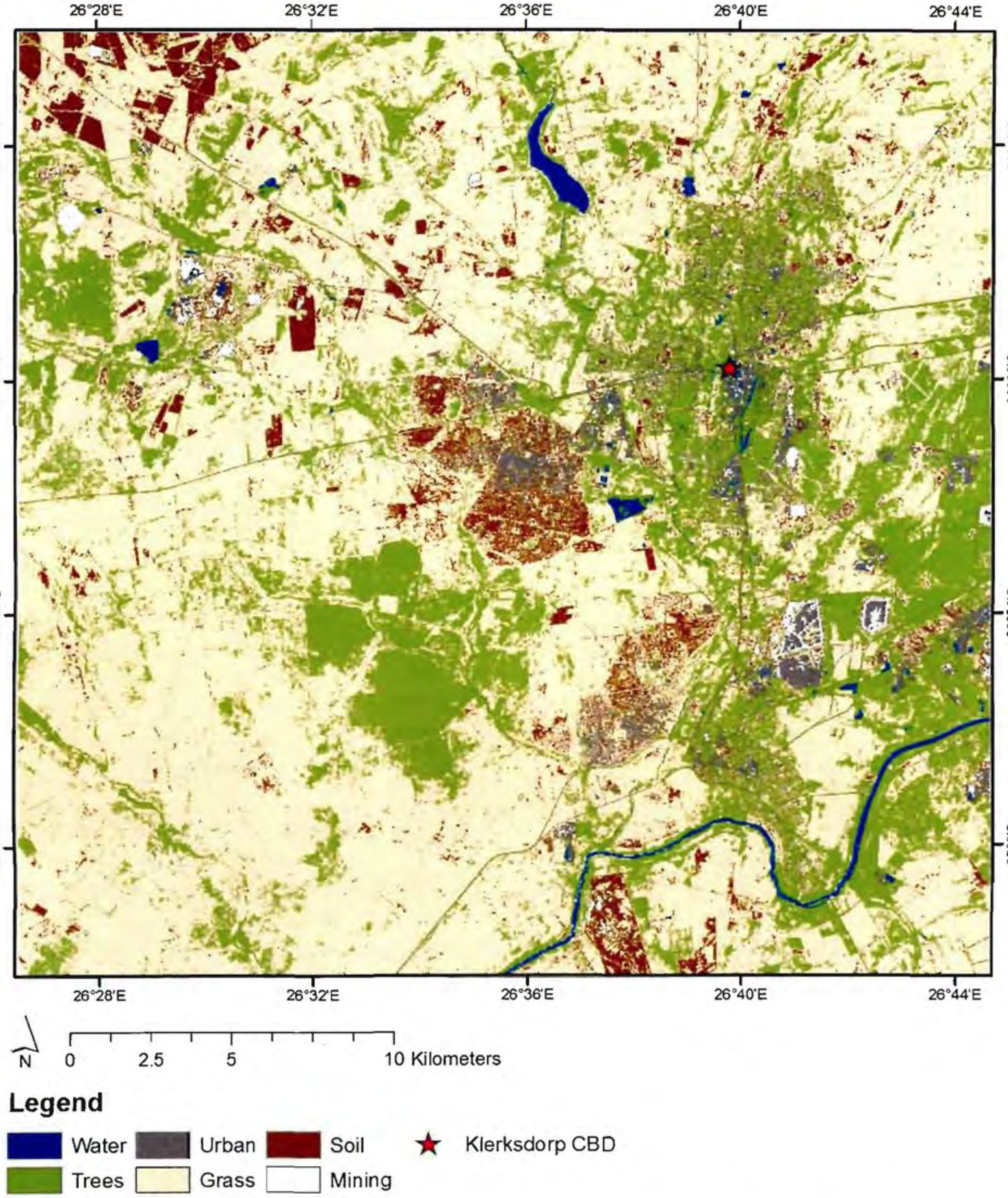


Figure 3.8: Land cover map A, with the central business district (CBD) of Klerksdorp indicated on the map.

3.3.2 Land cover map B

The accuracy improvement in the classes of land cover map B (Figure 3.9) is evident from the error matrix table (Table 3.5) and illustrates how the exclusion of the mining areas in the classification process affected it. This helped to clear some of the mixed pixels in the urban, grass and soil classes. The relative cover of the soil class increased in the classified area; which is due to the exclusion of the mining class from the unsupervised classification procedure, clarifying some of the spectral similarities that occurred. The urban class was also improved. However, a close inspection of some of the larger buildings in the urban areas revealed that where the roof was particularly reflective that part of the building was classified as soil. This is one of the potential problems of using unsupervised classification as well as the constraints in the less sophisticated imaging software used. However, overall the hybrid classification method used in the creation of land cover map B shows excellent potential for use in areas constrained by problems with spectral similarities. However, it is important to note that due to the characteristic outline and simplicity of the shapes it was easy to digitize the mining areas and mask it out. This process can be much more complicated for another type of feature and might even be impossible to edit out of the unclassified image.

As discussed earlier in the creation of land cover map A, the overall accuracy achieved for the image as shown in Table 3.5 should not be used as an absolute value of the map accuracy. Land cover map B will be used further in the dissertation with its overall accuracy of 95 %.

Table 3.5: Error matrix of land cover map B of the greater Klerksdorp area, showing the producer's, user's and overall accuracy.

		Training set data							
		Water	Trees	Urban	Grass	Soil	Mining	Row total	User's accuracy
Classification data	Water	49	1	1				51	96 %
	Trees	1	46		5			52	88 %
	Urban			48				48	100 %
	Grass		3		44	2		49	90 %
	Soil			1	1	48		50	96 %
	Mining						50	50	100 %
Column total		50	50	50	50	50	50	300	
Producer's accuracy		98 %	92 %	96 %	88 %	96 %	100 %		
					Overall accuracy			95 %	

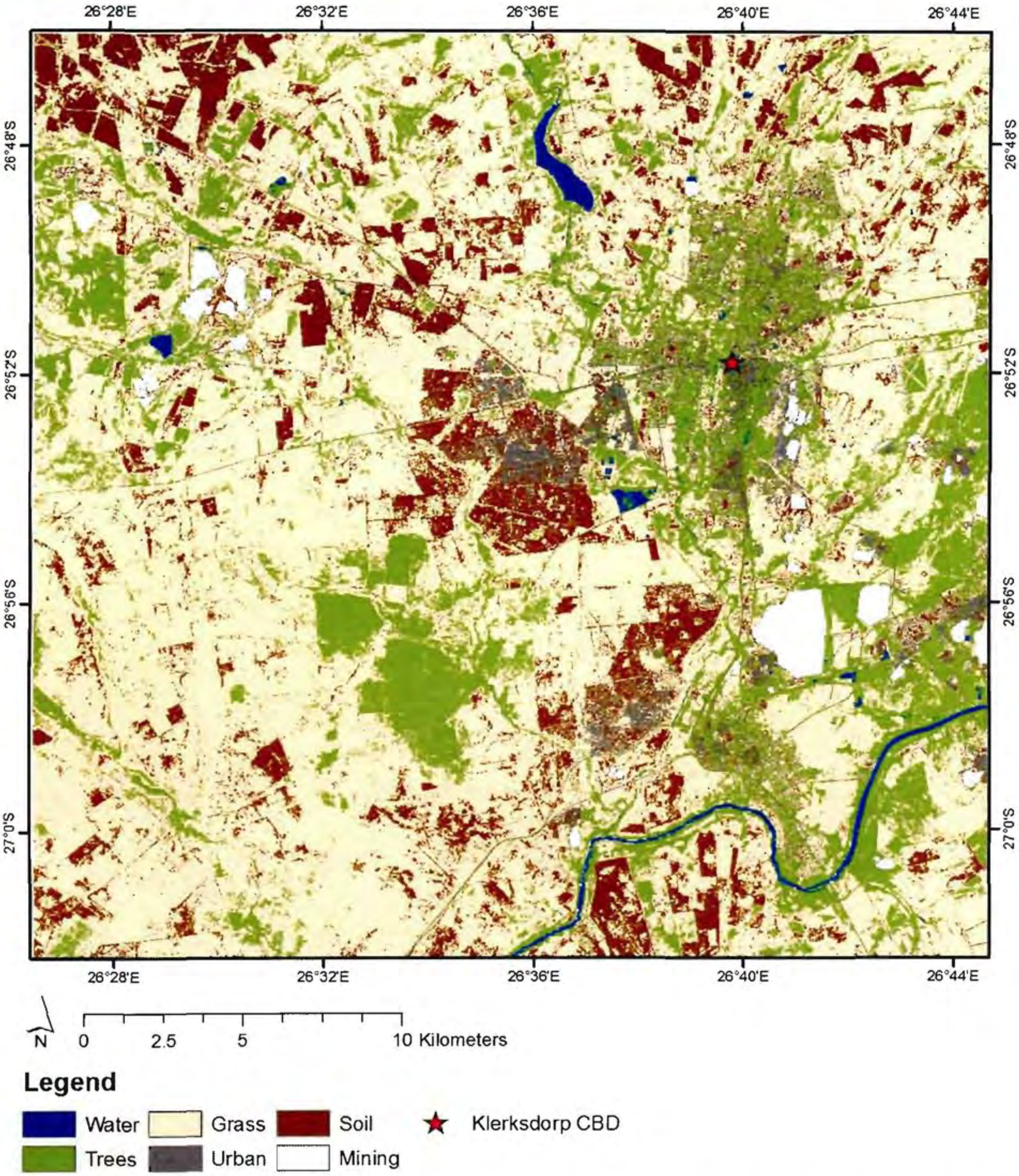


Figure 3.9: Land cover map B, with the central business district (CBD) of Klerksdorp indicated on the map.

3.4. Summary

The objective explored for this chapter was to use SPOT 5 satellite imagery to classify the land cover of the greater Klerksdorp area. The land cover map will then subsequently be used as input for the quantification of the urban-rural gradient of the current study. The objective was successfully achieved and a workable land cover map of the current study was created which can be used effectively as input for the calculation of some of the urbanization measures. The effectiveness of SPOT 5 satellite imagery was also proven to use as a source for the creation of a land cover map. In the current study where the urban areas are not that extensive, the 10 m spatial resolution of the SPOT 5 image, proved to be more than adequate to accurately delineate the urban areas.

The hybrid classification procedure used to create land cover map B with Spatial Analyst, illustrated that ArcView 9.2 can be used successfully to classify satellite images in the absence of better, more specialized imaging software. The 95 % overall accuracy of land cover map B also illustrates the usefulness of Spatial Analyst for classifying satellite images. The land cover map B created from the satellite image will be used further as a basis to calculate the urbanization measures (Hahs and McDonnell, 2006) as discussed later in Chapter 4 and as a spatial reference (Cohen and Goward, 2004) for the chosen natural grassland patches as discussed in Chapter 5.

Chapter 4: Measures of urbanization

4.1 Introduction

"The combination of remote imagery data, geographic information systems software, and landscape ecology theory provides a unique basis for monitoring and assessing large-scale ecological systems." (O'Neill *et al.*, 1999)

The urban-rural gradient approach to the study of urban environments has been effectively used to study the ecology of cities and towns around the world (McDonnell and Hahs, 2008). However, McIntyre *et al.* (2000) stated that in a review of studies using a gradient approach they observed that the quantification of the gradient was largely lacking. The majority of the studies also did not supply any clear definition of what constituted as urban in their study, exacerbating the difficulty of directly comparing the results of different studies with each other. Listed below are four examples of studies done along an urban-rural gradient and the measures they used to quantify the gradient of each respective study. Additionally, the measures are grouped into broad or specific measures (McIntyre *et al.*, 2000; McDonnell and Hahs, 2008)

Alberti *et al.* (2007) investigated the impact of urban development patterns on stream ecological conditions. They quantified the areas of 42 watershed sub-basins with four urban pattern variable subgroups, namely: land use intensity, land cover composition, landscape configuration, and connectivity of the impervious area. The connectivity of the impervious area represented two new specific spatial measures they developed. The other subgroups all contained broad measures. Blair and Johnson (2008) investigated suburban habitats and their role for birds in the urban–rural habitat network. They quantified their urban-rural gradient by defining their site locations along a gradient of increasing urban land use types such as a preserve, golf course, a single-family detached housing residential area and an industrial park. Land use type is an example of a broad measure of urbanization.

Burton *et al.* (2009) investigated riparian woody plant traits across an urban–rural land use gradient and its implications for watershed function with urbanization. The watershed matrix for each site was characterized by using the dominant land cover type, land cover patterns represented by three landscape metrics and specific hydrologic measures. Therefore, they used a combination of broad and specific measures to quantify the gradient. Lastly, Daniel and Lecamp (2004) investigated the distribution of three indigenous fern species along a rural–urban gradient. They divided the entire city into a systematic grid and simply defined the gradient by the distance of the grid cell form the urban centre. However, they did not specifically measure the distances. This is again a broad measure of urbanization.

In review of different gradient studies, Hahs and McDonnell (2006) found that measures used to quantify the urbanization component of the study tended to fall into three general groups, namely: demographic variables, physical variables, and landscape metrics. The exact nature of the respective measures used (broad or specific) are determined by the aims and data types available for the particular study. This could clearly be illustrated in the examples of studies done along an urban-rural gradient, as discussed above.

However, as seen in the above mentioned few examples, the relationship between demographic -, physical variables and landscape metrics have not been well documented, therefore Hahs and McDonnell (2006) aimed towards elucidating the relationships between these three types of measures in an effort to identify an appropriate subset of urbanization measures that best capture variability in the landscape in question, with the least amount of redundancy. Usage of a standard set of measures would then allow comparisons of studies globally, leading towards the advancement of basic ecological knowledge. As McIntyre *et al.* (2000) realized that no single definition of 'urban' is possible or necessary; McDonnell and Hahs (2008) proposed that the use of a standard set of broad urbanization measures would allow an 'urbanization context' for a particular study and facilitate integration between different studies. The universal applicability and importance of this could clearly be illustrated with the four examples given, as none of them can be compared directly.

Table 4.1 shows the 17 broad measures of urbanization Hahs and McDonnell (2006) identified in their study with which they aim to contribute to the establishment of a standard set of measures.

Table 4.1: The 17 measures of urbanization used by Hahs and McDonnell (2006) to characterize the 1 km^2 landscape grid cells of the study area in Melbourne, Australia.

DEMOGRAPHIC AND PHYSICAL VARIABLES:	LANDSCAPE METRICS:
Density of people	*Land cover richness*
Density of dwellings	*Dominant land cover*
Index census	*Simpson's diversity index*
Road network density	*Percent urban land cover*
Distance to central business district (CBD)	*Number of patches*
People per urban unit land cover	*Landscape shape index*
Fraction impervious surface	*Largest patch index*
Index image	*Mean patch fractal dimension*
Index combined	

The current study will consequently assist in the creation of a standard set of broad measures by testing a subset of the 17 measures proposed by Hahs and McDonnell (2006) in a South African setting. The following specific objectives (listed in Chapter 1) will be explored, namely: (1) to use the created land cover map in Chapter 3 and GIS techniques to determine which of the selected demographic and physical variables and landscape metrics can be used to quantify urbanization in the landscape, and (2) to use urbanization measures identified in Melbourne, Australia to quantify the urbanization gradient of the greater Klerksdorp area in South Africa.

4.2 Methods

Transects has traditionally been used to represent changes in urbanization across a landscape (Iwata *et al.*, 2005; Wilcke *et al.*, 2006; George *et al.*, 2007). Patterns identified in this type of linear gradient profile is, however highly dependent on the transect placement, located in a research area where urbanization creates complex non-linear gradients. Therefore, Hahs and McDonnell (2006) proposed the use of multidimensional non-linear methods, such as landscape grids, to provide a more accurate representation of gradients within urban landscapes.

A landscape grid with 1 km^2 cells was created for the 30 km^2 extent of the greater Klerksdorp area, making up a total of 900 individual grid cells for the entire study area (Figure 4.1). The value for each respective urbanization measure was calculated per 1 km^2 grid cell. The calculations were performed using ArcView 9.2 (ESRI, 2006a), Hawth's analysis tools v.3.27 (Beyer, 2007), and Microsoft Office Excel.

To calculate the urbanization measures, the raster land cover map (Chapter 3) was converted to a vector shapefile. This process creates polygons of the class classifications. For some measures a layer was created where all the patches of each class were merged to allow for calculations of measures where the entire class as a whole is analyzed. Hawth's analysis tools (Beyer, 2007) were used to create the landscape grid with the Create Vector Grid tool. The output landscape grid consists of 900 1 km^2 grid cells. To create a grid, the input layer must possess a projected coordinate system, hence the masked out satellite image of the study area was used as input for the vector grid creation.

In order to compare different geographical areas it is crucial that the type of input data should be similar. However, sometimes several situations arise that complicates the matter. The study of Hahs (2006) used a satellite image with a different spatial resolution and the classification topology of their land cover map had one less class. Additionally, some of the input data of the current study was not that accurate. Hahs (2006) used satellite imagery with a spatial resolution of 30 m for the land cover map creation, the current study utilized satellite imagery with a 10 m spatial resolution allowing for better classification accuracy (Chapter 3). However, this increases the average number of patches from 52 of Hahs (2006) to 176 in the current study adding additional complexity to the landscape. The study of Hahs (2006) used a classification typology of five classes. Because the current study had large mining areas an additional class of mining was added. Inclusion of these areas into the urban class would have erroneously enlarged the extent of the urban areas, influencing some of the urbanization measures and as a result the accurate quantification of the gradient. The accuracy of the census data used in the current study was restricted by the available local level data. As a result, some measures relying on the census data as input was influenced by the problem of accuracy as well.

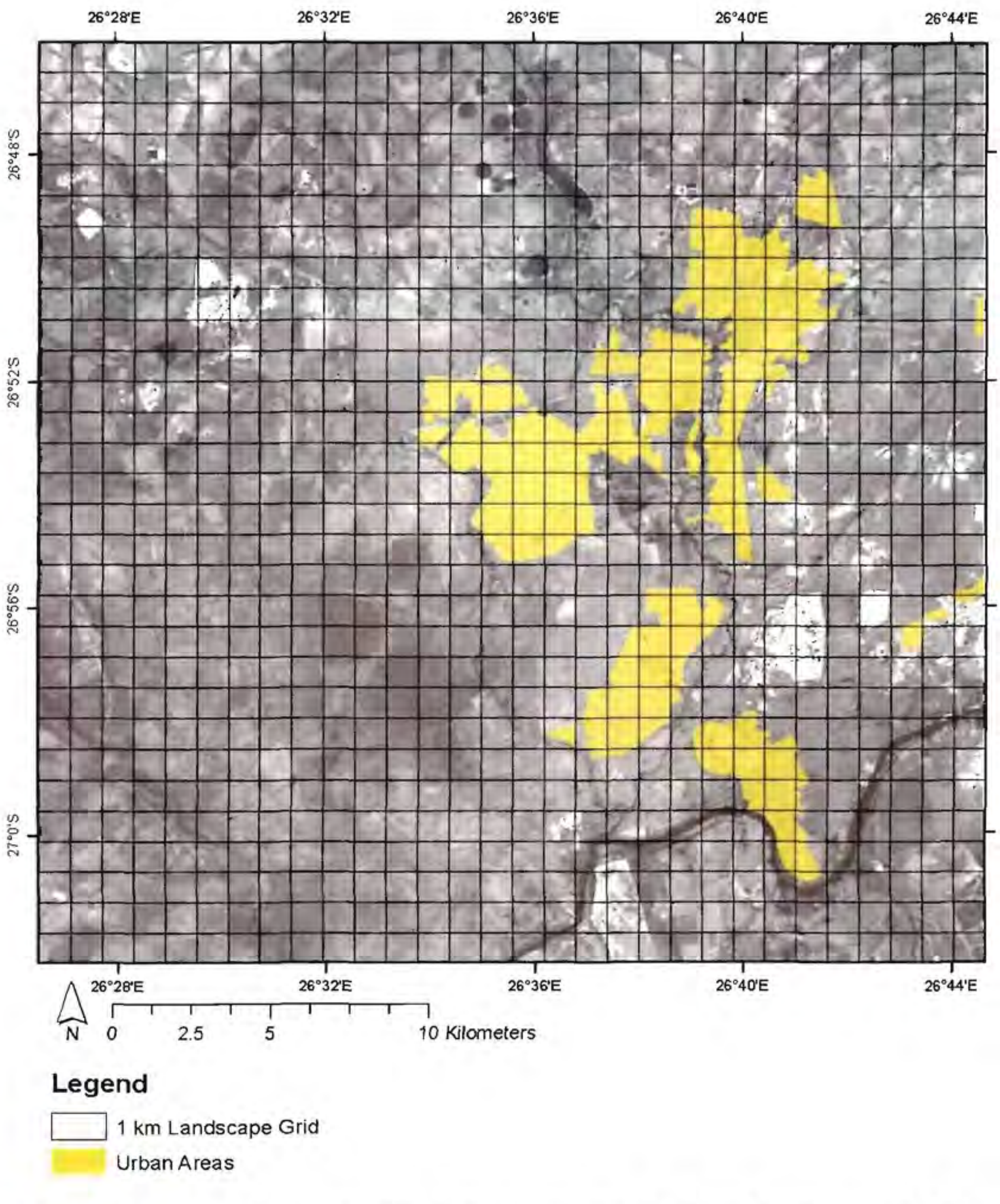


Figure 4.1: Map of the greater Klerksdorp area with the 1 km^2 landscape grid. The urban areas are highlighted in yellow.

At the outset all 17 urbanization measures as used by Hahs and McDonnell (2006) were evaluated for calculation in the current study. However, of the seventeen measures used by Hahs and McDonnell (2006) listed in Table 4.1 only 13 were eventually used in the current study. Fraction impervious surface, index image and index combined could not be calculated because specialist remote sensing software packages are needed to extract fraction images from the satellite multiband image. The measure of people per urban unit land cover was also removed from the list. In the following pages it will be explained why we decided to omit the people per urban unit land cover from the gradient analysis, even though it was suggested by Hahs and McDonnell (2006) as an important urbanization measure.

4.2.1 People per urban unit land cover

Hahs and McDonnell (2006) explained that this newly created measure will more accurately relate census data to the amount of urban land cover present in the landscape, thereby indicating the intensity of urban land use. The new measure proposed by Hahs and McDonnell (2006) was calculated in the current study, but certain discrepancies were discovered which made the measure impractical as an urban to rural gradient measure, in the current study.

People per urban unit land cover (PEOP) measures the ratio of the density of people (DENSPEOP) divided by the proportion of urban land cover (PURBLC). Hahs and McDonnell (2006) added a value of 0.5 to the denominator to allow the measure to be calculated for areas where there was no urban land cover present (Equation 1).

$$\mathrm{PEOP} = \frac{\mathrm{DENSPEOP}}{(\mathrm{PURBLC} + 0.5)} \tag{1}$$

Hahs and McDonnell (2006) described the measure as *"directly measuring the variability in the nature of the urban environment when percent urban land cover is held constant"*. It describes the variation in the density of people in different areas where the amount of urban land cover is the same. It signifies the urban land use intensity within the urban portion of the landscape, giving an indication of different levels of human use at specific locations. People per urban unit land cover values of less than one signifies relatively low land use intensities, with values greater than one signifying increasingly higher land use intensities. Areas with high percentages of urban land cover and high densities of people can be distinguished from areas with equivalent percentages of urban land cover having lower densities of people such as industrial areas (Hahs and McDonnell, 2006).

The problems encountered with the calculation of PEOP in the current study will be discussed. This is in support of the decision to reject the further use of the measure for the current study.

The density of people measure needed as input for the PEOP measure was relatively inaccurate in the current study. The accuracy of density of people is dependent upon the scale of its available census data. The scale of the census data used in Melbourne was calculated per suburb (Hahs and McDonnell, 2006). In South Africa the best local scale census data that is freely available is that of ward delineation. Wards are clusters of voting district polygons obtained from the Independent Electoral Commission of South Africa, created in 2000 by the Municipal Demarcation Board (Statistics South Africa, 2004). Most of South Africa is divided into wards, often containing areas with human habitation and natural and agricultural areas lumped together. The selection of ward boundaries is therefore not dependent on specific population distributions but on divisions based on political choices. Consequently, some wards have an enormous extent with a low average population density. But the main problem is the inclusion of rural areas with urban areas containing high population densities. The average population density for the affected rural area will therefore reflect the urban population densities and not the true situation found in the rural area in question. This is evident in the current study where some of the census tracts span urban as well as the surrounding natural and agricultural areas (Figure 4.2). This results in rural areas awarded high population density distributions that in reality contains no human habitation.

The census data sampling size of the current study (as shown in Figure 4.2) influenced the calculation of the population per landscape grid cell based on the percentage area it occupies relative to the ward it is located in. This allowed cells to have a relatively high population density even though it contained no houses or urban structures as the entire ward population number is divided between the grid cells it contains. Thereby, allowing generalizations about the population distribution per cell specifically across the surrounding rural landscape. The resulting population numbers per cell is therefore at a variance to the actual situation encountered in each specific cell.

Additionally, the classification typology of the land cover map can play a deciding role in the output values of the measure. The accuracy of the percentage of urban land cover measure depends upon the spatial resolution of the satellite image used for the land cover map creation, the classification process, and the nature of the urban areas being classified. As the classified land cover map class delineations are very accurate with the satellite image having an appropriate spatial resolution, the potential problem for the greater Klerksdorp area is the nature of the urban areas.

Figure 4.3 (a) shows the urban areas as found in the greater Klerksdorp area with a pie chart (b) illustrating the proportional sizes of the different classification classes as determined for the urban areas. Only 27.63 % of the urban areas classed as urban specifically, this includes the industrial areas, CBD, large houses, roads and other urban commercial zones. The pie chart shows the dominance of the trees class in the urban areas as well. This is as a result of the abundance of trees lining the roads and the trees found especially in the residential gardens. The average South African residence has a single storey house with a large garden. Therefore the trees lining the roads and those surrounding the houses cover large parts of the roads and houses, hiding the actual extent of the urban land cover when viewed directly from above as with a satellite image. The inset map in Figure 4.3 (c) clearly illustrates this situation for Klerksdorp.

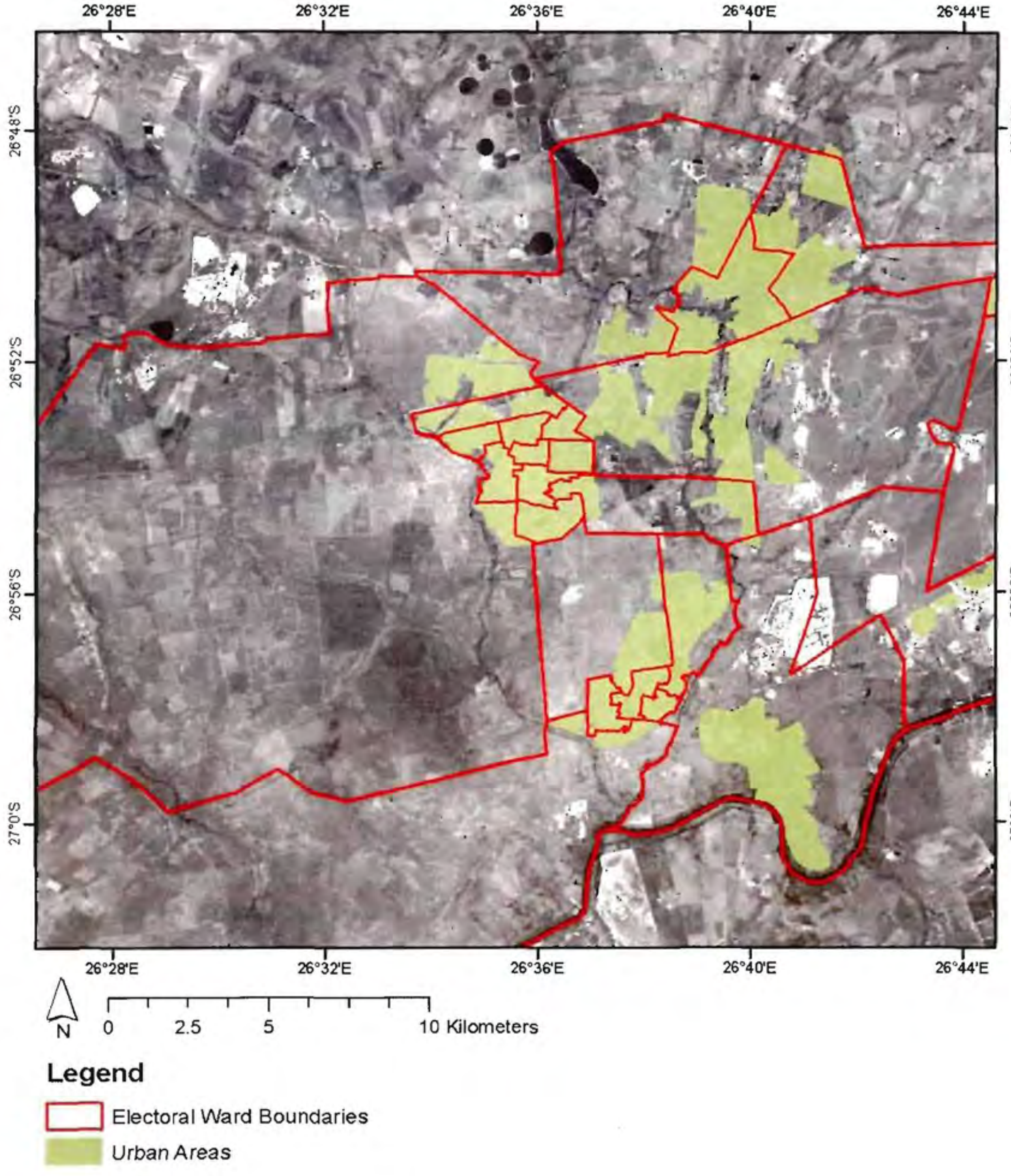


Figure 4.2: Map of the greater Klerksdorp area showing the delineation of the electoral wards.

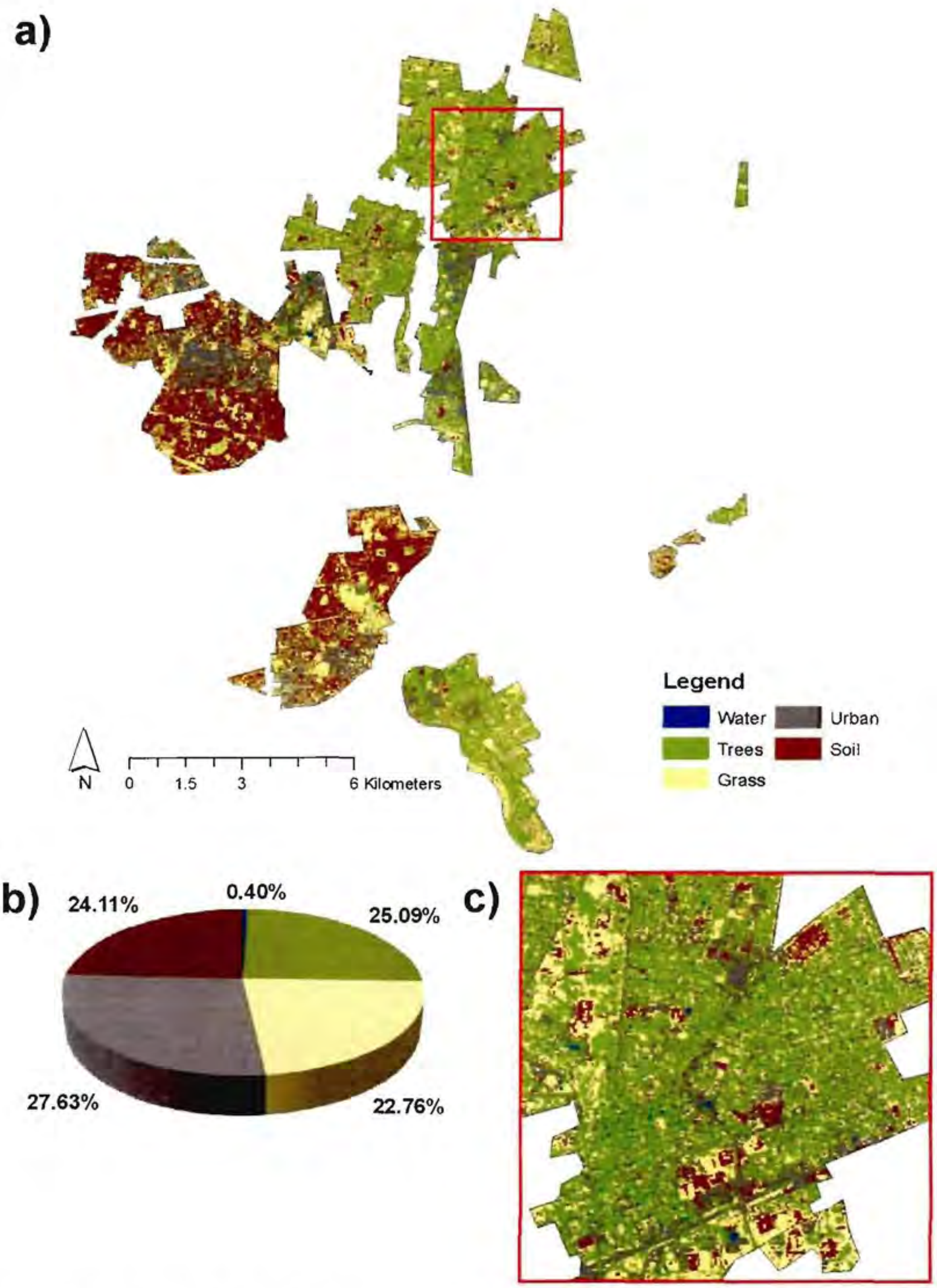


Figure 4.3: Map of the urban areas found in the greater Klerksdorp area (a). The pie chart shows the relative size proportions of the classification classes for the entire urban areas (b). The colours of the pie chart are the same as indicated in the legend for the urban areas (a). The inset map of a part of Klerksdorp was added to show the classification detail (c).

In the informal settlements on the urban fringes a profusion of the soil class is found. This occurs due to the large amount of dirt roads, the clearing of the yard surrounding their small informal houses of vegetation, and the overall removal of vegetation in these areas. Trees are used in particular as fuel for heat and cooking. The informal houses, 'shacks' built by the people has an average size of 3 m x 5 m, and as a result the 10 m spatial resolution of the multispectral satellite bands does not detect some of these houses if the majority of the pixel consists of bare ground instead of the house. The bulk of the informal houses built in these areas are also constructed from corrugated iron, which has a highly reflective surface that can cause spectral reflection similarities between itself and bare ground causing misclassified pixels. This was discussed in Chapter 3 as well.

Table 4.2 lists nine scenarios spread across the greater Klerksdorp area, that illustrate the inconsistencies of the people per urban unit land cover output values. These illustrate the differences occurring in the landscape from that of its intended meaning as described by Hahs and McDonnell (2006). In Figure 4.4 the location of the nine chosen grid cells are shown (a-h), labelled as listed in Table 4.2. These cells will be used to explain why the measure of people per urban unit land cover is not a practical option for this particular study area.

Table 4.2: List of some examples of people per urban land cover values found across the greater Klerksdorp area.

SCENARIOS		FIG 4.2 NO	DENSPEOP	PURBLC	PEOP
Highest PEOP value	*1*	*(a)*	*793*	*0.06*	*1416.07*
Lowest PEOP value	*2*	*(b)*	*18*	*32.56*	*0.54*
Highest % urban land cover	*3*	*(c)*	*10517*	*70.94*	*147.21*
Similar PEOP values	***4***	***(d)***	***177***	***0.09***	***300***
	5	***(e)***	***7504***	***24.49***	***300.28***
	6	*(f)*	*246*	*45.66*	*5.33*
	7	*(g)*	*8*	*1*	*5.33*
	8	***(e)***	***5825***	***48.75***	***118.27***
	9	***(h)***	***297***	***2.01***	***118.33***

The first scenario describes a grid cell located on the juncture of three wards. Due to the high density areas included in some of the wards the cell has a high population density; however the value for percent urban land cover is only 0.06 %. The combination of these two is the reason for the high value of PEOP. Therefore, this cell does not represent high land use intensity as suggested by the value interpretation as discussed by Hahs and McDonnell (2006), in fact, just the opposite as shown in Figure 4.4. The grid cell with the highest PURBLC and the highest population density of the entire study area only has a relatively low PEOP value of 147.21 (Table 4.2 scenario 3).

Scenarios four to nine in Table 4.2 all illustrate cases where widely dissimilar situations, with regards to values of DENSPEOP and PURBLC, have the same value for PEOP. Some of them will be discussed. Cells four (Figure 4.4 d) and nine (Figure 4.4 h) are both cases where low percent urban land cover cells are located near a high density population area, where the high densities are ascribed to the cells due to ward delineations. However, cells six (Figure 4.4 f) and seven (Figure 4.4 g) are normal cases that would be found in an area where the census data is extremely accurate. Almost half of cell six is classified as urban and the population density is moderate. Cell seven on the other hand has only a one percent urban land cover with a population density of 8, typical of rural areas. However, both of these cells have a PEOP value of 5.33.

The people per urban unit land cover values for these three cell pairs mentioned above and listed in Table 4.2 are respectively, exactly the same even though the situations are widely dissimilar. An arbitrary look at a map of people per urban unit land cover for the greater Klerksdorp area would identify these situations as the same, where the viewer will have no idea of the underlying differences. Figure 4.4 (d-h) clearly shows the differences in situations of the six grey shaded scenarios (Table 4.2) for the respective 1 km^2 landscape grid cells in the greater Klerksdorp area.

The above mentioned problems occur due to the fact that the percentage of urban land cover values ranges between 0 and 100, allowing for fractions and integer values e.g. 0.4 % or 60 %. This discrepancy in dividing with fractions and integers allows situations such as illustrated in Figure 4.4. Although Hahs and McDonnell (2006) stated that the measure illustrates differences in land use intensities between cells with the same amount of urban land cover, the sole use of the PEOP values will not elucidate whether similar PEOP values genuinely have the same land use intensities or if it is a case of different input values as with some of the scenarios discussed previously. Therefore, this measure is impractical for urban to rural gradient quantifications in the current study and it was discarded from the list of measures to be used in further analysis.

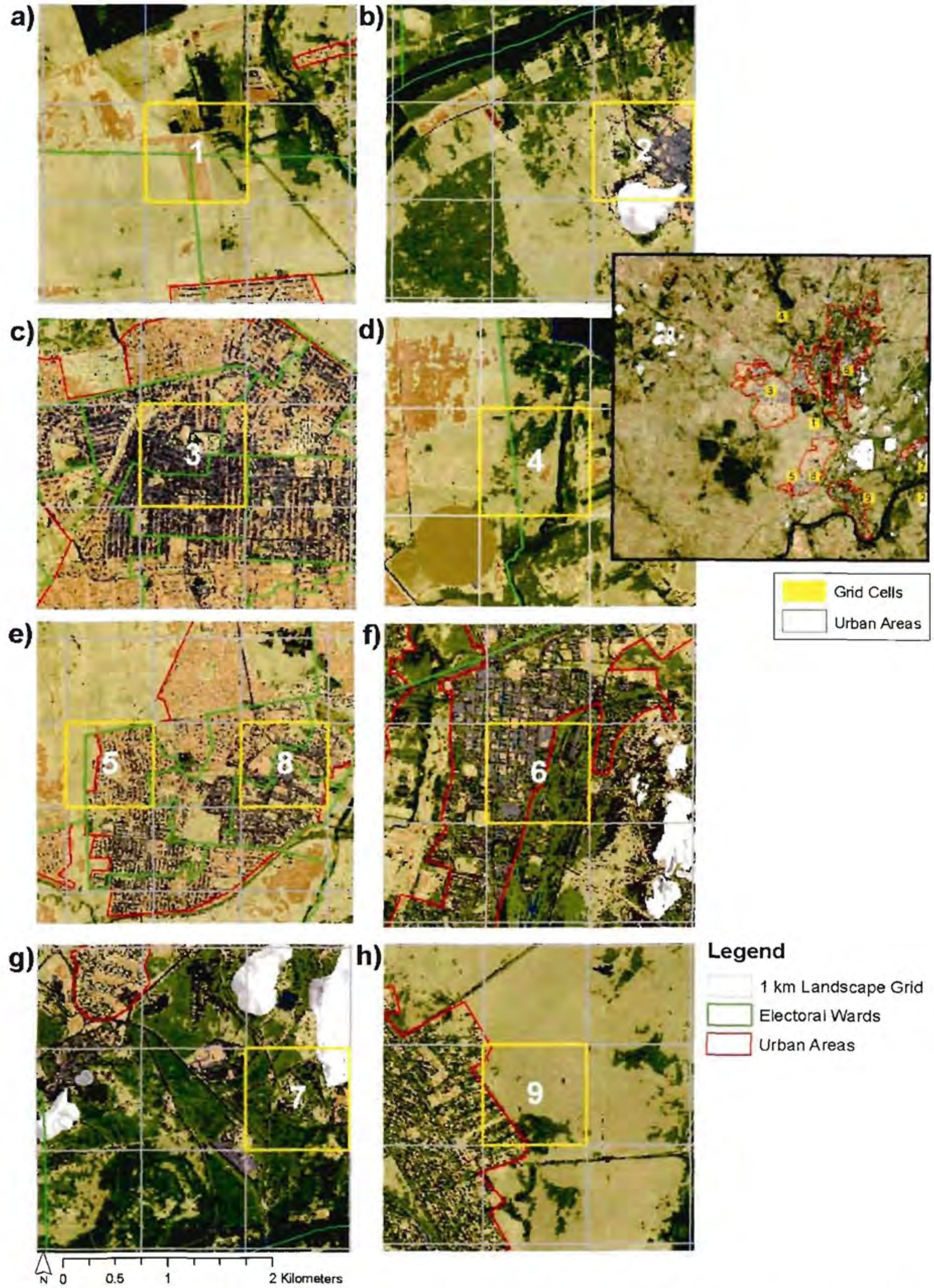


Figure 4.4: Enlarged sections of the land cover map of the greater Klerksdorp area. The map illustrates the nine PEOP scenarios (a-h) described in Table 4.2. The map detail was improved by overlaying the 2.5 m resolution panchromatic band image of the SPOT 5 satellite image on the land cover classification (land cover map B, Chapter 3) with a 50 % transparency.

4.2.2 Calculation of the used urbanization measures

Table 4.3 lists the measures that were calculated in the current study out of the seventeen measures proposed by Hahs and McDonnell (2006). The calculation method of each measure will be discussed, supplying a brief description of each measure.

Table 4.3: List of 13 of the measures proposed by Hahs and McDonnell (2006) that was calculated for the greater Klerksdorp area.

DEMOGRAPHIC AND PHYSICAL VARIABLES:	LANDSCAPE METRICS:
Density of people (DENSPEOP)	*Land cover richness (LCR)*
Density of dwellings (DENSDWEL)	*Dominant land cover(DOMLC)*
Index census (INDEXCEN)	*Simpson's diversity index (SIDI)*
Road network density (RND)	*Percent urban land cover (PURBLC)*
Distance to central business district (CBDkm)	*Number of patches (NP)*
	Landscape shape index (LSI)
	Largest patch index (LPI)
	Mean patch fractal dimension (MPFD)

4.2.2.1 Demographic- and physical variables

Density of people (DENSPEOP). This measure describes the total number of people calculated, in the current study, on the local electoral wards level, expressed as the number of people per km^2 (Hahs and McDonnell, 2006). The ward and census data were obtained from the website of the Municipal Demarcation Board of South Africa (Municipal Demarcation Board, 2006). The census data used was those of the last national census in 2001. The study area overlaps two local municipalities. To calculate the population numbers for the grid cells, the two different municipality ward shapefiles were merged with the Union tool in ArcMap of ArcView 9.2 (ESRI, 2006a). To calculate the population per grid cell, a free ArcScript was downloaded from the Internet, namely the Calculate Demographics tool (Muzslay, 2006). This tool computes the demographics of selected polygons or a polygon layer from another layer's data. The tool then adds the output fields to the destination layer if the fields do not already exist. Attributes are assigned proportionally based on their area of intersection, for example, a source polygon 100 % within the destination polygon will transfer 100 % of the values; a source polygon 93 % contained will transfer 93 % and so on. The source layer was the merged wards, with the destination layer being the 30 km^2 vector landscape grid. As the grid cell has a grain size of 1 km^2 the generated values were used directly as the density of people per grid cell.

Density of dwellings (DENSDWEL). This measure describes the total number of dwellings enumerated, in the current study, on the local electoral wards level, expressed as the number of dwellings per km^2 (Hahs and McDonnell, 2006). The measure was calculated in exactly the same way as the density of people from the 2001 census data.

Index census (INDEXCEN). This measure is a census-based index of urbanness as proposed by Weeks *et al.* (2003). The measure describes the total number of people multiplied by the proportion of males employed in non-agricultural work (Hahs and McDonnell, 2006), as enumerated in the 2001 census. It represents a description of not only the number of people living within an area, but also the degree to which they are dependent upon the urban agencies for employment (Hahs, 2006). (Equation 2)

$$Index\ census = \left[\frac{\ln(density\ of\ people * prop.of\ males\ non - agri\ jobs)}{12.85}\right] * 100 \qquad (2)$$

A description of the formula is given in Weeks *et al.* (2003). The measure of density of people was used as input together with the 2001 census occupation data per ward. The occupation data was calculated per grid cell by the Calculate Demographics tool described earlier. The density of people and the values of the proportion of males engaged in non-agricultural jobs were calculated in ArcMap with the Field Calculator in the attribute Table of the vector grid. The data was then exported to Microsoft Office Excel where the final calculations were completed.

Road network density (RND). This measure describes the total length of all public roads calculated per landscape grid cell, expressed as the length of roads per km^2. All the tarred and dirt roads found in the greater Klerksdorp area were manually digitized from the panchromatic band image. This polyline layer was then used as input for the Sum Line Lengths in Polygons tool, in Hawth's Analysis tools (Beyer, 2007). This tool adds a field in the source layer, which is in this case the vector grid, containing the length of roads computed in meters. These values were then converted to kilometres. As the grid cells have an extent of 1 km^2 the values were used directly as input for RND.

Distance to central business district (CBDkm). The linear distance from the central business district (Hahs and McDonnell, 2006). The location of the CBD was digitized in ArcMap, chosen as the police station at the corner of Commisioner - and Noord Street in the New Town suburban area in Klerksdorp (Department of Constitutional Development and Planning, 1986). Centroids for each grid cell were created to calculate the distance of each centroid from the CBD. Two new fields were added in the vector grid attribute table, namely X coordinate, and Y coordinate. The Calculate Geometry function was used to respectively calculate the X coordinate of centroid and the Y coordinate of centroid. These two fields were then used as input for the Export XY data tool to create a centroid point layer. Spatial Join was done with the CBD point layer and the centroids point layer. Spatial Join creates a field in the attribute table of the centroids layer. This field contains the computed distances from the CBD. The tool computes the values in meters; we then converted it to kilometres. These values were then used as input for the CBDkm measure.

4.2.2.2 Landscape metrics

All the landscape metrics used in the current study are described by McGarigal and Marks (1995). Appendix A lists the definitions of the specific notation used in Equations 3 – 10.

Land cover richness (LCR). The number of land cover types present in the landscape (Hahs and McDonnell, 2006). (Equation 3)

$$LCR = m \tag{3}$$

The land cover richness equals the number of different patch types present within the landscape boundary (McGarigal and Marks, 1995). The classified raster version of the land cover map was used as input for calculation of this measure. In Spatial Analyst, the Block Statistics tool was used, with the VARIETY statistics type calculated for an area equal to the 1 km^2 grid cells. This tool calculates the variety (the number of unique values) of the inputs. As each classification class has its own unique grid code, this determines the amount of unique values found in the equivalent 1 km^2 grid area; hence the land cover richness per landscape grid cell is computed. The resulting raster was then converted to a polygon layer and intersected with the landscape grid to get the values of LCR per grid cell of the landscape.

Dominant land cover (DOMLC). This measure represents the most extensive land cover type (Hahs and McDonnell, 2006). This measure is adapted from measures described by McGarigal and Marks (1995). (Equation 4)

$$DOMLC = m_{\text{max area}} \tag{4}$$

The measure describes the class with the most extensive total area in the landscape. The classified raster version of the land cover map was used as input for calculation of this measure. In Spatial Analyst, the Block Statistics tool was used, with the MAJORITY statistics type calculated for an area equal to the 1 km^2 grid cells. This tool calculates the majority (the value that occurs most often) of the inputs. Therefore, the value of the most extensive class type will be allocated to the area equal to the 1 km^2 grid cells. The resulting raster was then converted to a polygon layer and intersected with the landscape grid to get the values of DOMLC per grid cell of the landscape.

Simpson's diversity index (SIDI). This measure is an index of the land cover diversity based on both richness and abundance (Hahs and McDonnell, 2006). (Equation 5)

$$SIDI = 1 - \sum_{i=1}^{m} Pi^2 \tag{5}$$

When there is only one patch in the landscape the value of SIDI equals 0, thus there is no diversity. SIDI approaches 1 as the number of patch types, the patch richness, increases and the proportional distribution of area among the different patch types evens out (McGarigal and Marks, 1995). The vector version of the land cover map was used as input, where the different patch classes were merged. This merged land cover map was intersected with the vector grid. The square meters area of each patch type was computed in ArcMap. These values were then divided by the grid cell square meters area to calculate the proportional area of each patch. The proportional area values of each patch type where then squared. The Summarize tool in the attribute table was then used to sum all the squared area values per grid cell. These subsequent values were then used as input for the SIDI measure.

Percent urban land cover (PURBLC). This measure calculates the area of urban land cover divided by the total area sampled (Hahs and McDonnell, 2006). (Equation 6)

$$PURBLC = \frac{\sum_{j=1}^{n} a_{ij}}{A} (100) \tag{6}$$

The percent urban land cover equals the percentage of the landscape comprised by the urban land cover class. The merged patch class vector land cover map was used as input for the measure. The urban class was selected in the attribute table and exported as a separate layer. This urban layer was intersected with the vector grid. The respective areas of the urban land cover class was computed in ArcMap and divided by the grid cell area. These values were then multiplied by a hundred to get the percentage urban land cover per grid cell. The subsequent values were then used as input for the PURBLC measure.

Number of patches (NP). This measure is a count of the number of patches in the landscape (Hahs and McDonnell, 2006). (Equation 7)

$$NP = N \tag{7}$$

To calculate this measure the vector land cover map was intersected with the vector grid of the landscape. The Summarize tool in the attribute table of the newly created layer was used to sum the amount of patches per grid cell. The output attribute table was joined with the vector grid attribute table to add the values to the vector grid layer. The subsequent values were used as input for the NP measure.

Landscape shape index (LSI). This measure is an index of the amount of irregularity of the shape of the landscape patches (Hahs and McDonnell, 2006). The raster version of the measure was used, because the vector land cover map was converted from a raster retaining the raster patch boundaries. (Equation 8)

$$LSI = \frac{0.25\, E'}{\sqrt{A}} \tag{8}$$

The LSI equals 1 when the landscape consists of a single square (raster) patch, and value increases without limit as the landscape shape becomes more irregular or as the length of edge within the landscape increases, or both (McGarigal and Marks, 1995). The intersected land cover map (with the vector grid) was used as input for the measure. The perimeter (in meters) per patch of the intersected layer was calculated in ArcMap with the Calculate Geometry tool in the attribute table. This value was multiplied by 0.25. With the Summarize tool the new perimeter values was summed per grid cell. The output attribute table was joined with the vector grid and exported to create a vector grid with the values added in its attribute table. The area of each grid cell was then computed in square meters with the Calculate Geometry tool. A new field was then added to the table and the LSI values were calculated by dividing the sum of the new perimeters per cell by the squared area of the grid cell. The subsequent values were then used as input for the LSI measure.

Largest patch index (LPI). This measure calculates the percentage area of the largest patch in the landscape (Hahs and McDonnell, 2006). (Equation 9)

$$LPI = \frac{{}_{j=1}^{n}\max(a_{ij})}{A}(100) \tag{9}$$

The LPI nears 0 as the size of the largest patch in the landscape decreases. The value is 100 when the entire landscape consists of only one patch. The intersected land cover map (with the vector grid) was used as input for the measure. The area in hectares was calculated for all the patches of the intersected layer with the Calculate Geometry tool. The Summarize tool in the attribute table was then used to summarize the maximum values for the areas in hectare per grid cell. The output table was joined with the vector grid layer to add the values to the vector grid attribute table. As the values is in hectare and the grid cell area equals 100 hectares, the values were used without dividing it with the grid cell area and multiplying it with a hundred. The maximum area values were used as input for the LPI measure.

Mean patch fractal dimension (MPFD). This measure describes the patch shape complexity, which is also an indication of the scaling within the landscape (Hahs and McDonnell, 2006). The raster version of the measure was used in the current study. (Equation 10)

$$MPFD = \frac{\sum_{i=j}^{m}\sum_{j=1}^{n}\left(\frac{2\ln(0.25p_{ij})}{\ln a_{ij}}\right)}{N} \tag{10}$$

The values for MPFD approach 1 for shapes with very simple perimeters, such as a square, and near 2 for shapes with highly complex plane-filling perimeters (McGarigal and Marks, 1995). The intersected land cover map (with the vector grid) was used as input for the measure. The perimeter and area in meters of each patch was computed with the Calculate Geometry tool in the attribute table of the intersected layer. The attribute table was then exported to Microsoft Excel to calculate the fractal dimension per patch. The calculated values were then copied back into the attribute table of the intersected layer and the Summarize tool was subsequently used to compute the sum of the fractal dimensions per grid cell. The output table was joined with the vector grid attribute table where the values were then divided by the NP values to calculate the MPFD per grid cell of the landscape.

4.2.3 Data analysis

The 900 values for each of the 12 urbanization measures were then used as input for the principal components analysis (PCA) and the subsequent factor analysis (FA). However, the measure dominant land cover (DOMLC) was excluded from PCA and FA procedures, as it contains nominal values. PCA and FA require interval scale data as input values, e.g. values representing quantities where 2 is bigger than 1, etc. Nominal values are associated with measurement systems used to identify one instance from another. These values are qualitative, not quantitative, with no relation to a fixed point or linear scale (ESRI, 2006b). Therefore, the water class, for example, could have been given any number. The allocated value 3 is not larger than 1 etc. there is no linear scale of values. The measure was still retained, however, as it supplies valuable information about the major matrix type of the entire landscape area as well as each grid cell. This measure will be used in Chapter 5 to describe the chosen grassland patches found in the greater Klerksdorp area.

4.2.3.1 Principal components analysis

A PCA was done with the 12 measures (excluding DOMLC), since the different urbanization measures are highly correlated (Hahs, 2006). The main aim of a PCA is to reduce the dimensionality of a data set consisting of a large number of interrelated variables, while retaining as much as possible of the variation present in the data set (Jolliffe, 2002). Newly formed components are ordered in such a way that the first few retain most of the variation present in all of the original variables. Each variable are given a coefficient on each of the components representing the weighting of the original variables on each component in explaining the maximum amount of variance in the data (Kent and Coker, 1992). All of the variables that contribute to the variability will be more or less equally loaded on the first, most important component.

Because a PCA uses covariance or correlations as a measure of variable association, PCA is more effective as a variable reduction procedure when there are linear relationships between variables (Quinn and Keough, 2002). Therefore, if the variables are normally distributed, the solution is enhanced. However, to the extent that normality fails, the solution is degraded but may still be meaningful (Tabachnick and Fidell, 2001). This can easily be the case, since nonlinear relationships are commonly found between biological variables (Quinn and Keough, 2002). In such cases, transformations can often improve the linearity of the relationships between variables. Normal distribution plots of the 12 urbanization measures were drawn in STATISTICA 8.0 (StatSoft, Inc., 2008) to test the variables for normality. All the measures were subsequently transformed using Box-Cox transformations (Box and Cox, 1964). This is an objective way of determining a transformation for the values of a measure. Box-Cox transformation is a power transformation used to convert a sample of observations to normality. The best transformation, in terms of normality and homogeneity of variance, is obtained by determining the maximum likelihood estimate of λ in an iterative process (Thode, 2002). The Box-Cox method is an easy and effective way to select the form of the variable transformation in making the distribution of the response variables closer to the normal distribution (Montgomery, 2001).

The PCA was done in STATISTICA 8.0 as an initial step to identify the amount of factors needed as input for factor analysis (FA) (Johnson, 1998; Tabachnick and Fidell, 2001). With a typical PCA the analyst tries to extract components that explain the variability in the original variables (Quinn and Keough, 2002). However, one cannot explain the correlations among the original variables, as with a FA. As we want to see which variables explain the most variability without using redundant variables explaining the same underlying phenomenon, a FA is needed.

4.2.3.2 Factor analysis

The goal of FA is to assess the extent to which the various input variables in the data set can be interpreted as measures of one or more underlying constructs or latent variables (Warner, 2008). An advantage of FA over PCA is that in the creation of new factors/components, the factors created by FA are generally much easier to interpret than those created by PCA because of the rotation of factors (Johnson, 1998). Patterns of correlation among the variables are examined to see whether the variables can be interpreted as measures of a single underlying latent variable or whether a better understanding can be obtained of the patterns of correlations among the variables (Warner, 2008). FA can help in the decision of how many latent variables are needed to understand the responses to test items and to decide which items are good indicators for each variable. FA is also useful for re-examination of existing measures (Warner, 2008). Tabachnick and Fidell (2001) stated that factors are thought to 'cause' variables; therefore it is the underlying factors that are responsible for the scores on the variables. Whereas the components of PCA are simply aggregates of correlated variables, the variables 'cause' the component. Therefore, to decide whether both landscape metrics and demographic measures are needed to describe the urbanization gradient; and in deciding which of the existing measures used in literature best measures the underlying variability a FA was executed, also using STATISTICA 8.0.

The transformed variables (urbanization measures) were used as input for the FA. In the PCA only the first two components had eigenvalues above 1 (Table 4.5), therefore the FA was done retaining only two factors. The factors were rotated with the Varimax raw rotation method. Varimax rotation is a variance maximizing procedure, this maximizes the variance of the factor loadings by making high loadings higher and low ones lower for each factor. The initial output summary table showed that some variables, namely: CBDkm, MPFD, RND did not load very high on either of the factors, therefore the FA was repeated exclusive of these variables to allow the correlations between the high loading variables and the factor to be better defined.

The measures identified with the FA will be used further to quantify the urban-rural gradient of the greater Klerksdorp area (Chapter 7).

4.3 Results and discussion

4.3.1 Calculated urbanization measures

4.3.1.1 Demographic- and physical variables

Table 4.4 supplies a summary of the measures as calculated by Hahs (2006) and in our subsequent study. Note that the maximum value of the density of people (DENSPEOP) values of the current study exceeds that of Hahs (2006). The census 2001 count of the total number of people in the greater Klerksdorp area is 243 088, whereas Hahs (2006) stated that in 2001 the number of people in Melbourne numbered 3.2 million. The reason for the higher maximum value in the current study is the unique spatial distribution of the population in the greater Klerksdorp area. In the suburbs of Jouberton and Kanana 63.2 % of the population of the greater Klerksdorp area lives on only 5.34 % of the total land area. However, the more urbanized nature of Melbourne can clearly be distinguished by the substantially higher mean population density of 700, in comparison with 270.1 of the current study.

In density of dwellings (DENSDWEL), the same trend is observed for the mean of the measure as in DENSPEOP. However, interestingly the highest maximum value here is found in the study of Hahs (2006) resulting in the conclusion that in the current study the average people per dwelling ratio is much higher. These situations are particularly evident in the high density areas of Jouberton and Kanana where an entire family sometimes shares a small one room house.

Index census (INDEXCEN) is an urbanness measure (Weeks *et al.*, 2003); the higher the values of the population and the proportion of males engaged in non-agricultural activities, the higher is the value of the measure. The higher maximum value of the current study is because it measured higher population densities for a few landscape grid cells. However, the slightly higher mean values for Melbourne indicate that Melbourne is a more urbanized area. One reason for the current study's mean value being so close to that of Melbourne, can be ascribed to the relatively few people living in the surrounding rural areas and the fact that only approximately 2.3 % of all the employed people in the study area are engaged in agricultural activities. Both the measures of road network density (RND) and the distance to central business district (CBDkm) show no exceptions, and in relation to the larger study area of Hahs (2006), the values of the measures are both proportionally similar for the two studies.

Table 4.4: Comparative table of the mininum, maximum, and mean values for the 13 measures as calculated by Hahs (2006) for Melbourne and in the current study in the greater Klerksdorp area.

	Hahs (2006)			Current study		
Measure	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**
Demographic and physical variables						
Density of people (DENSPEOP)	*0*	*8403*	*700*	*8*	*10517*	*270.1*
Density of dwellings (DENSDWEL)	*0*	*3524*	*280*	*2*	*2466*	*70.24*
Index census (INDEXCEN)	*0*	*68.07*	*29.94*	*10.79*	*71.95*	*28.99*
Road network density (RND)	*0*	*22.7*	*4*	*0*	*20.08*	*2.26*
Distance to CBD (CBDkm)	*0.4*	*53.9*	*26.5*	*0.45*	*28.57*	*13.42*
Landscape metrics						
Land cover richness (LCR)	*1*	*5*	*4*	*2*	*6*	*4*
Dominant land cover (DOMLC)	*1*	*5*	*3*	*1*	*6*	*3*
Simpson's diversity index (SIDI)	*0*	*1*	*0.37*	*0*	*0.75*	*0.42*
Percent urban land cover (PURBLC)	*0*	*97.9*	*26.9*	*0*	*70.94*	*4.06*
Number of patches (NP)	*1*	*170*	*52*	*7*	*1053*	*176.57*
Landscape shape index (LSI)	*1.13*	*15.40*	*6.01*	*1.22*	*38.38*	*11.56*
Largest patch index (LPI)	*13.1*	*100*	*78.6*	*6.38*	*99.76*	*62.09*
Mean patch fractal dimension (MPFD)	*1*	*1.38*	*1.22*	*1.03*	*1.74*	*1.16*

4.3.1.2 Landscape metrics

The fact that the additional mining class in the land cover map of the current study did not have a big effect in the comparison between the two studies, can be seen in the similarities between the measures of land cover richness (LCR) and dominant land cover (DOMLC) for the respective studies. Especially in the case of LCR (Table 4.4), where both studies have the same mean value, even though an additional class was added to our study. The dominant matrix type in both the studies, as described by DOMLC, is grasses.

The effect of the satellite spatial resolution differences can be seen in the values of Simpson's diversity index (SIDI). The mean SIDI is higher in the current study than that found by Hahs (2006). This is due to the higher number of classified patches per grid cell, as described earlier.

This increase in spatial resolution in the current study also affected the landscape shape index (LSI), which is the reason for the mean and maximum values being larger than those of Hahs (2006). As LSI measures the total length of edge in the grid cell divide by the square root of the total area, the higher the measured value of LSI, the more irregular is the landscape shape or the longer the total length of edge. The higher mean and lower minimum values can therefore be ascribed to the higher accuracy of the land cover map (Figure 3.9, Chapter 3) resulting in a more 'fragmented' classification.

The higher mean and maximum values for the percent urban land cover (PURBLC) measure by Hahs (2006) clearly illustrates the differences in the intensity and the extent of urbanization between Melbourne and the greater Klerksdorp area. However, in the current study as referred to earlier, the nature of the urban areas influenced the classification typology, resulting in only 27.63 % (Figure 4.3) of the urban areas being classified in the urban class. But in comparison to Melbourne, even if the urban areas were classified more accurately according to their real extent, it would probably not have made a significant difference. Nevertheless, this classification situation can be a common problem in any residential area supporting large trees.

The most substantial difference in any of the compared measures is that of the number of patches (NP). The mean of the current study is higher than the maximum value of that described by Hahs (2006). This is as a result of the higher land cover accuracy due to the 10 m spatial resolution of the input satellite image allowing more detail to be observed. As a result, the biggest influence of the higher resolution was seen in this measure. Therefore, more detailed patches of different land cover features are captured in the current study, especially in the highly fragmented and heterogeneous urban areas.

The mean value for largest patch index (LPI) in the current study is smaller than that of Hahs (2006). However, this measure was also influenced by the spatial accuracy of the satellite image, therefore the higher spatial accuracy accounted for more patches and subsequently the mean LPI decreased. This might be the reason for the lower value of LPI in the current study. However, the lower value might also indicate that the overall fragmentation of the landscape in the current study is higher. The greater Klerksdorp area is dominated with farms and agricultural area which results in the fragmentation of the landscape.

The maximum mean patch fractal dimension (MPFD) value of the current study was higher than that of Hahs (2006). This measure describes the complexity of the landscape patches and therefore was also influenced by the higher spatial accuracy which might be the reason for the higher values recorded for the current study. However, the measure clearly showed that in the urban areas the landscape reveals a complex arrangement of a mosaic of patch types with highly heterogeneous patch shapes. This measure is therefore an indication of the fragmentation and simultaneous complexity of the urban environment.

4.3.2 Data analysis

4.3.2.1 Principal components analysis

The table showing the respective formulas used by each measure for the Box-Cox transformations for normality of the values is listed in Appendix B together with their respective original and transformed normal distribution plots. The fact that transformation of the measures was needed can be expected as it is known that urban areas are complex and form non-linear gradients. Urban areas are subjected to multiple anthropogenic influences, as well as having an influence on the surrounding countryside and natural vegetation.

Table 4.5 shows the output summary table for the PCA results. The eigenvalue of the first component represents the highest possible degree of correlation of all the variables with the principal axis and thus is a measure of the amount of variation in the data set accounted for by the first axis. The forthcoming axes all show lower eigenvalues as the axes are extracted in descending order of importance in terms of their contribution to the total variation in the data set (Kent and Coker, 1992).

One of the common methods used in deciding how many components to retain are the exclusive use of only eigenvalues with scores larger than 1. Johnson (1998) elaborates on this by stating that the reason the eigenvalues are compared to 1 is because the analysis uses the correlation matrix (standardized) data, and the variance of each standardized value is equal to 1. Therefore, they stated that the belief is that if a principal component cannot account for more variation than a single variable can by itself, then it is probably not important and hence components with eigenvalues less than 1 are ignored.

On the other hand, the individual loading of each respective variable greater than 0.3 are interpretable (Tabachnick and Fidell, 2001). The nearer the loading is to ± 1.0 the more important that variable is regarding weighting that component (Kent and Coker, 1992). However, when many of the loadings for a specific variable have intermediate values, that is, most loadings on the different components are in the order of + 0.3 the interpretation of the component can be difficult (Warner, 2008).

The PCA results in Table 4.5 shows that only the first two principal components had eigenvalues above 1. These two components explained 55.3 % and 18.4 % of the respective total variance. Combined they explain 73.6 % of the overall variance in the data. However, all of the values had only intermediate eigenvector scores on the first component and thus made interpretation of the components difficult. As the first component in a PCA explains the maximum amount of variance in the data, it is not surprising that all the variables have relatively equal contributions to this first component. Warner (2008) argues that for each variable the loading should be large enough in absolute value to be able to state that this variable corresponds to this component; preferably each variable should only load high on one component. Both SIDI and LPI load equally on the first two components.

Table 4.5: PCA results of all the input urbanization measures of the greater Klerksdorp area. Eigenvalues of the correlation matrix, the total percentage of variance and the cumulative variance % are given per factor to indicate the total amount of variance of each factor as well as explaining its relative importance. The eigenvectors for each variable per component is also listed. Values < 0.3 are highlighted.

Eigenvalue	*6.630708*	*2.207021*	*0.877687*
% Total variance	*55.25590*	*18.39184*	*7.31406*
Cumulative %	*55.2559*	*73.6477*	*80.9618*
	Comp 1	**Comp 2**	**Comp 3**
DENSPEOP	*-0.272972*	***-0.467710***	*0.000808*
DENSDWEL	*-0.269820*	***-0.467330***	*0.001438*
INDEXCEN	*-0.253386*	***-0.488686***	*0.017481*
RND	*-0.287861*	*0.043679*	*-0.028199*
CBDkm	*0.262181*	*0.113362*	*0.151312*
LCR	*-0.259688*	*0.186596*	*-0.283868*
SIDI	***-0.301552***	***0.328577***	*-0.105313*
PURBLC	***-0.341847***	*0.053640*	*-0.068282*
NP	***-0.354226***	*0.142377*	*0.097686*
LSI	***-0.344158***	*0.208351*	*0.044033*
LPI	***0.301413***	***-0.300211***	*0.099733*
MPFD	*0.163934*	*-0.099054*	***-0.926412***

The PCA results therefore showed that the first two components explained a significant amount of variance and the correlations between the variables are large enough to form a compact factor. Tabachnick and Fidell (2001) explained that if no correlation exists the use of FA would be questionable because there would be nothing to factor analyze.

4.3.2.2 Factor analysis

The factor analysis was subsequently done with two factors, possessing eigenvalues greater than one. Because the first component of the PCA showed only intermediate loadings for all the variables the FA results were rotated by a varimax raw rotation, in order to maximize the loadings. Table 4.6 clearly shows the significant improvement in the variable loadings on the two factors. As stated earlier, FA explains the correlations among the original variables with the factors, thereby highlighting the contributions of each variable to the interpretation of the underlying variance explained by each factor. In FA the interest is primarily in the variables that cause the variance, and not in explaining the maximum amount of variance on one component as with PCA. Therefore in interpreting the underlying patterns in FA the factors 'cause' the variable loadings (Tabachnik and Fidell, 2001). For the current study, it explains exactly what we are seeking, to identify the measures that best describes the urbanization gradient. The greater the loading, the more the variable is a pure measure of the factor, loadings in the excess of 0.71 (50 % overlapping variance) are considered excellent (Tabachnik and Fidell, 2001).

Table 4.6: FA results of all the input urbanization measures for the first two components of the greater Klerksdorp area. The factor loadings for each variable per component are listed. Values > 0.7 are highlighted in bold. The data was varimax raw rotated. The eigenvalues are the same as for the PCA (Table 4.5).

	Factor 1	**Factor 2**
DENSPEOP	*0.182540*	***0.971366***
DENSDWEL	*0.176187*	***0.966282***
INDEXCEN	*0.123334*	***0.968285***
RND	*0.646429*	*0.368494*
CBDkm	*-0.459282*	*-0.522696*
LCR	***0.707609***	*0.152626*
SIDI	***0.916290***	*0.040539*
PURBLC	***0.769157***	*0.435441*
NP	***0.870391***	*0.345184*
LSI	***0.904853***	*0.249839*
LPI	***-0.892012***	*-0.074985*
MPFD	*-0.430851*	*-0.119240*
Expl.Var	*5.197943*	*3.639786*
Prp.Totl	*0.433162*	*0.303316*

As shown in Table 4.6 the landscape metrics and the demographic and physical variables each respectively correlate extremely well to the two extracted factors. SIDI and LPI now load almost exclusively on only one component. At this stage it is important to note that when designing a study, measures should be included that have reasonably large correlations with the factors. Restricted range on the variable tends to reduce the size of the correlations, which would in turn lead to a less clear factor structure (Warner, 2008). Therefore, the better the normal distribution of the variable ranges the better it correlates to the factors or components.

The FA results clearly indicate the better distribution of the explained variance for the two factors with factor one now explaining 43.3 % of the total variance in the data and factor two explaining 30.3 %. However, CBDkm, MPFD and RND did not load exceptionally high on either factor; therefore it was removed to improve the correlations of the other high scoring variables to supply a more unambiguous picture of the underlying formed variables (factors). Therefore, the FA was repeated without these variables. Table 4.7 shows the results of this analysis.

Table 4.7: FA results of a selection of all the input urbanization measures for the first two components (low scoring variables were removed) of the greater Klerksdorp area. The factor loadings for each variable per component are listed. Values > 0.7 are highlighted in bold. The data was varimax raw rotated.

Eigenvalue	5.538244	2.170264
% Total variance	61.53604	24.11404
Cumulative %	61.53604	85.65009
	Factor 1	**Factor 2**
DENSPEOP	0.186479	**0.977605**
DENSDWEL	0.178480	**0.971082**
INDEXCEN	0.129201	**0.979660**
LCR	**0.722935**	0.148046
SIDI	**0.933797**	0.050564
PURBLC	**0.756702**	0.426302
NP	**0.863636**	0.351208
LSI	**0.908348**	0.258827
LPI	**-0.907974**	-0.087216
Expl.Var	4.445909	3.262599
Prp.Totl	0.493990	0.362511

The eigenvalues, percentage of the total variation and cumulative percentages are shown for the input data. The overall explained variance has increased from 73.6 %, initially, to 85.7 %. This illustrates that without the low scoring variables more variation in the data is explained with the first two factors, as the additional variance caused by the low scoring variables is omitted. A clearer picture of the variables correlated to the underlying factors is now obtained. The first factor now explains 49.4 % of the variance, with the second factor explaining an additional 36.3 % of the variance. SIDI, LSI and LPI are highly correlated with the first component, with DENSPEOP, DENSDWEL and INDEXCEN highly correlated to the second component. Table 4.8 describes the two factors and their correlated variables listed in the order of highest correlation. The factors were named based on the type of variables most highly correlated with the factor.

The first factor can be described as the landscape structure, with the second defined as demographic attributes. All of the relevant variables loading on the two factors respectively describe aspects of the landscape structure and the demographic attributes of the greater Klerksdorp area. In FA variables loading on the same factor are assumed to correlate with each other and can be taken as an indication that both of the variables may measure the same underlying feature, therefore they are different measures of the same underlying pattern or process (Warner, 2008). As a result any of the variables with high loadings on the respective factors can be chosen to further represent the gradient, thus irrespective if e.g. SIDI or LSI is chosen even if they are measured differently, both of them represent the landscape structure. The choice then lies with which one is the simplest to calculate or as done in the current study which one of the measures shows the highest correlation to each other in a basic correlation matrix (created in STATISTICA 8.0).

Table 4.8: The two factors extracted from the FA. Each factor is named according to the type of variables it is most highly correlated with. The factor loadings are also given, only values above 0.71 listed.

LANDSCAPE STRUCTURE (FACTOR 1)		DEMOGRAPHIC ATTRIBUTES (FACTOR 2)	
SIDI	*0.933797*	INDEXCEN	*0.979660*
LSI	*0.908348*	DENSPEOP	*0.977605*
LPI	*-0.907974*	DENSDWEL	*0.971082*
NP	*0.863636*		
PURBLC	*0.756702*		
LCR	*0.722935*		

Table 4.9 shows the correlation matrix for the transformed variables. The grey blocks highlights the respective measures correlated to each variable. The black outline in the grey shaded areas indicates the measures with the highest respective correlations to the others.

Landscape shape index showed the highest correlations with other variables in the first factor, with the density of people chosen to represent the second factor on the same grounds. As the loadings of both LSI and DENSPEOP are in the excess of 0.9, both can be taken as a pure measure of the underlying latent variable (factor). Any additionally used measures would be redundant as they are all highly correlated with each other. Therefore, the urban-rural gradient in the greater Klerksdorp area can be described by using only these two measures without being concerned about omitting a critical variable.

Table 4.9: Correlation matrix of the variables with high loadings to examine which variable of each factor group has shows the best correlation to the rest in the group. All the correlations were significant at $p < 0.05$. The bold highlighted measures showed the best correlations to their respective correlated measures as identified in the FA.

	PURBLC	NP	SIDI	LCR	**LSI**	LPI	**DENSP...**	DENSD...	INDEXC...
PURBLC	*1.00*	*0.81*	*0.63*	*0.63*	*0.75*	*-0.66*	*0.53*	*0.53*	*0.47*
NP	*0.81*	*1.00*	*0.76*	*0.61*	***0.95***	*-0.73*	*0.48*	*0.48*	*0.44*
SIDI	*0.63*	*0.76*	*1.00*	*0.60*	***0.85***	*-0.93*	*0.25*	*0.23*	*0.20*
LCR	*0.63*	*0.61*	*0.60*	*1.00*	***0.58***	*-0.55*	*0.28*	*0.27*	*0.23*
LSI	*0.75*	*0.95*	*0.85*	*0.58*	*1.00*	*-0.81*	*0.41*	*0.41*	*0.37*
LPI	*-0.66*	*-0.73*	*-0.93*	*-0.55*	***-0.81***	*1.00*	*-0.28*	*-0.26*	*-0.23*
DENSPEOP	*0.53*	*0.48*	*0.25*	*0.28*	*0.41*	*-0.28*	*1.00*	*0.98*	*0.99*
DENSDWEL	*0.53*	*0.48*	*0.23*	*0.27*	*0.41*	*-0.26*	***0.98***	*1.00*	*0.96*
INDEXCEN	*0.47*	*0.44*	*0.20*	*0.23*	*0.37*	*-0.23*	***0.99***	*0.96*	*1.00*

4.3.3 The selected gradient measures

Based on the FA results and the correlation matrix LSI and DENSPEOP were chosen to respectively represent the landscape metrics and demographic variables. Hahs and McDonnell (2006) were the first to combine landscape and demographic variables, and in their study as in this one it is clear that both are needed to describe the urban environment as each of them explains a different type of variance in the data. Table 4.10 summarizes the landscape measures highly correlated with each other and gives a description of what each respective measure quantifies. It is apparent that these measures all describe the landscape structure.

Table 4.10: Summary of the correlated landscape metrics

MEASURE	DESCRIPTION
SIDI	*Index of land cover diversity*
LSI	*Index of irregularity of the landscape patches*
LPI	*Largest patch area of landscape*
NP	*Number of patches*
PURBLC	*Area of urban land cover*
LCR	*Number of land cover types present*

The LSI, chosen to represent the landscape structure is a measure of the complexity of the landscape (Hahs, 2006), the higher the values the more complex the landscape. The highest values correspond to grid cells with the highest number of patches, therefore the most fragmented grid cells. The lowest values represent grid cells with one dominant large patch and a few other smaller patches. All the measures listed in Table 4.10 primarily rely on the number of patches present in the landscape grid cell, hence their high correlations. This measure possibly correlates the best as it not only describes the amount of patches, but also their complexity by calculating the total length of edge in the grid cell as well.

Table 4.11 on the other hand; summarizes the highly correlated demographic measures, also giving a description of what each respective measure quantifies. It is apparent that these measures all describe demographic attributes. In the study of Hahs and McDonnell (2006) more demographic and physical measures were used, for reasons explained earlier these measures could not be repeated in the current study. Therefore, the current study's measures might be biased based on the diminished variability it describes. However, those demographic measures loading high in the study of Hahs and McDonnell (2006) were included in our study, with the exception of Index combined.

DENSPEOP is a direct representation of the available census data used as input for the measure. Both DENSDWEL and INDEXCEN also use census based input. Index census supplies additional information on the number of people employed in non-agricultural occupations. However, in an urban environment most of the people have urban occupations. In a study area evaluating only one urban area the density of people will measure nearly completely the same variation; index census might however be extremely informative for a comparison of two equally sized cities located in different areas. Nonetheless, DENSPEOP is the simplest to calculate and the data is the easiest to obtain.

Table 4.11: Summary of the correlated demographic variables

MEASURE	DESCRIPTION
DENSPEOP	*Density of people*
INDEXCEN	*Index of urbanness based on census data*
DENSDWEL	*Density of dwellings*

Figure 4.5 and Figure 4.6 show the landscape grid and the distribution of the two chosen measures. Each measure was illustrated using a ten class interval. The urban areas are displayed over the grid to add detail and to orientate the observed patterns in relation to the urban areas. These two measures will be used in the following chapter to quantify the location of the selected grassland patches along the observed urban-rural gradient.

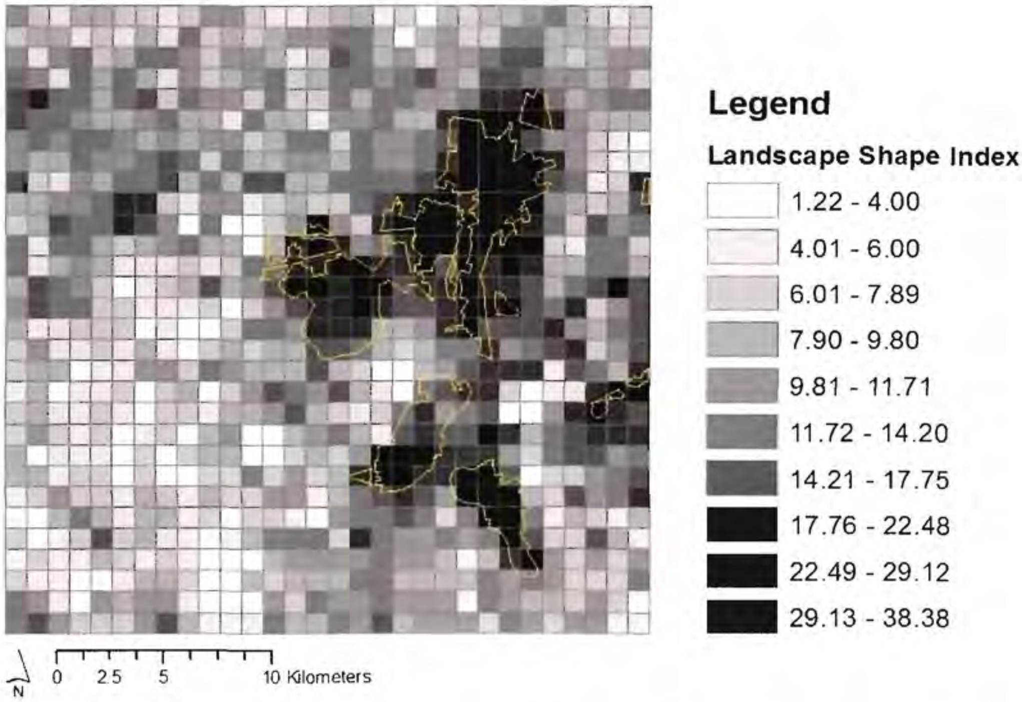


Figure 4.5: Landscape grid showing the distribution of the values for LSI for the greater Klerksdorp area. The yellow outline indicates the urban areas in the landscape.

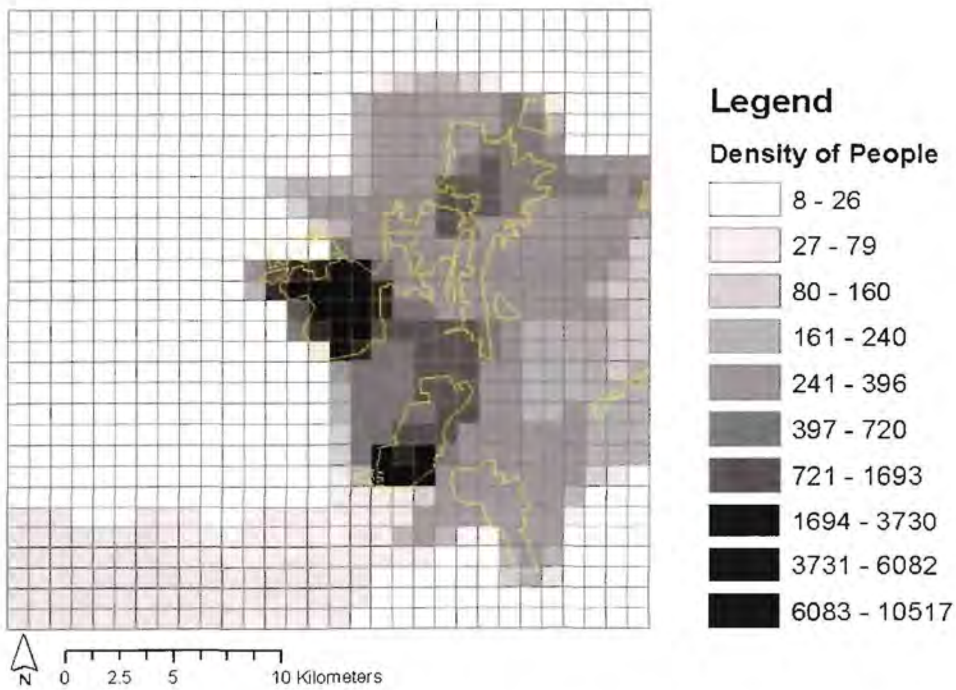


Figure 4.6: Landscape grid showing the distribution of the values for DENSPEOP for the greater Klerksdorp area. The yellow outline indicates the urban areas in the landscape.

4.4 Summary

Two objectives were explored in this chapter, namely: (1) to use SPOT satellite imagery and GIS techniques to determine which of the selected demographic-, physical variables and landscape metrics can be used to quantify urbanization in the landscape, and (2) to use the urbanization measures identified in Melbourne, Australia to quantify the urbanization gradient of the greater Klerksdorp area.

Two urbanization measures; landscape shape index and density of people, were identified as being informative in quantifying the urban-rural gradient in the current study and these two will be used in Chapter 7 to further quantify the urbanization gradient. By using the same measures identified in the study of Hahs and McDonnell (2006) we contributed to the creation of a standard set of measures by evaluating how the measures explained the variability in the greater Klerksdorp area. Landscape shape index, chosen in the current study was also one of the four measures chosen by Hahs and McDonnell (2006).

However, direct comparisons of studies require exactly the same type and scale of input data sets. In the current study some of the input data had differences and influenced some of the measures, such as the number of patches influenced by the spatial resolution of the satellite image. Nevertheless, the response of variables such as in the comparison of the current study with that of Hahs (2006) illustrates that the absolute values of some variables might not be as informative, as for instance using the proportional distributions of the respective values in comparison with themselves and the values in other studies.

Importantly, the range variability of the input measures also influences its loadings. Therefore, good quality variables are needed to compute the FA and PCA; otherwise the PCA/FA might be biased towards the better distributed value even if the others might be better indicators. However, it is clear that both demographic and landscape measures are needed to define the urban environments more accurately, as was also stated by Hahs and McDonnell (2006) and Tavernia and Reed (*in press*).

The next chapter will explore the selection of grassland patches in the greater Klerksdorp area as well as describing the location of the chosen patches along the quantified urbanization gradients.

Chapter 5: Selection of grassland patches

5.1 Introduction

"*...because most ecosystem processes are scale dependent, the choice of boundaries is of profound importance to the conceptualization of an ecosystem and to the scope and validity of questions being asked within that ecosystem*" Post *et al.* (2007).

Murphy and Lovett-Doust (2004) declared that the nature of the composite landscape mosaic (including both the patches and the matrix) is the key determinant of the fate of plant populations. Traditional plant metapopulation theory focus on suitable habitat patches determining population dynamics occurring in an inhospitable matrix. Consequently, emphasis is put on the importance of connectivity between the suitable habitat patches. However, Murphy and Lovett-Doust (2004) discussed the fact that many plants exist in situations where populations are not that readily describable (no clear habitat patches), therefore describing gradients of habitat suitability would be more appropriate. By proposing the use of a landscape grid in describing population dynamics they emphasized the value of an integrative, landscape based approach to understanding regional scale dynamic without the need to explicitly define matrix or patch habitats. This is strengthened by Hahs (2006) who emphasized the importance of the spatial context of a study area in explaining and understanding underlying ecological patterns and processes.

Figure 5.1 illustrates the spatial location of the greater Klerksdorp area with regards to its setting in South Africa. South Africa is divided into nine biomes of which the grassland biome is the second largest, covering 27.97 % of the total area (Figure 5.1 a). The greater Klerksdorp area is located in the grassland biome, which is defined by its unique vegetation structure and local environmental factors, primarily the amount of summer rainfall and the minimum winter temperatures (Mucina and Rutherford, 2006). Fire plays a big role in the grassland biome and is essential in maintaining the vegetation structure (Tainton, 1999). An additional major influence on the canopy structure and species composition in grasslands is the effect of grazing (Mucina and Rutherford, 2006). Table 5.1 summarizes the extent of the biome and its protection status as discussed by Van Wilgen *et al.*, (2008) and Mucina and Rutherford (2006). Additionally, 29.2 % of the biome is permanently transformed as a result of anthropogenic influences, with an additional 7 % of the biome severely degraded (Van Wilgen *et al.*, 2008).

O'Conner and Kuyler (2009) stated that the "*grassland biome encapsulates much of what confronts biodiversity conservation in the developing world: ongoing land transformation to serve economic development and limited opportunity for increasing the area of protected land*".

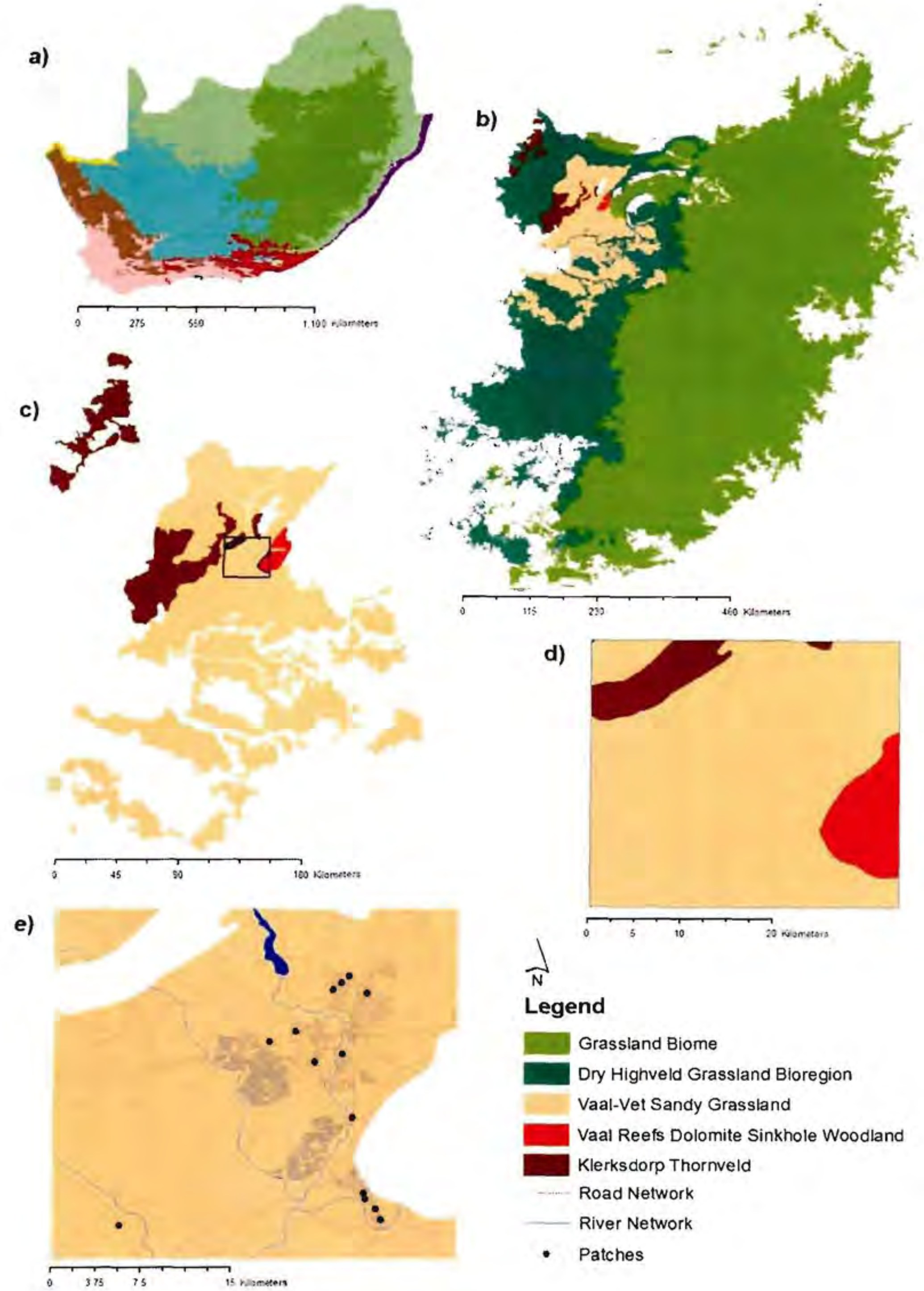


Figure 5.1: The spatial contextual setting of the greater Klerksdorp area. The greater Klerksdorp area is situated in the Grassland Biome (a); the Dry Highveld Grassland Bioregion (b); and includes three vegetation types, namely the Vaal-Vet Sandy Grassland, Vaal Reefs Dolomite Sinkhole Woodland and the Klerksdorp Thornveld (c). The Vaal-Vet Sandy Grassland forms the majority of the greater Klerksdorp area (d), and subsequently the patches were all selected in this vegetation type as illustrated in (e).

Even though the percentage area remaining is high (Table 5.1), Mucina and Rutherford (2006) affirmed that untransformed areas are often highly fragmented, where as much as half of the remaining grassland areas may be composed of fragments of only a few hectares in extent. This could be illustrated for the greater Klerksdorp area, where the mean number of patches (NP) of 176.57 per 1 km^2 and the mean largest patch index (LPI) of 62.09 per 1 km^2 grid cell clearly revealed the high fragmentation of the landscape (Chapter 4, Table 4.4).

Table 5.1: Comparative table of the extent of the grassland biome as described by Van Wilgen *et al.*, (2008) and Mucina and Rutherford (2006). The percentage remaining natural areas and the percentage conserved areas are also listed.

	Van Wilgen *et al.* (2008)	Mucina and Rutherford (2006)
Total area (km^2)	*349 190*	*354 593*
% Remaining natural area	*70.8*	*64.96*
% Protected area	*2.1*	*1.68*

The Dry Highveld Grassland bioregion forms the western region of the grassland biome with a mean annual precipitation of 496 mm (Figure 5.1 b). The environmental factors controlling the vegetation distribution as well as the recognition of different vegetation types are the annual rainfall, which forms an east to west gradient. Three of the vegetation types of this bioregion are included in the greater Klerksdorp area (Figure 5.1 c), namely: Vaal-Vet Sandy Grassland, Vaal Reefs Dolomite Sinkhole Woodland and the Klerksdorp Thornveld (Mucina and Rutherford, 2006). The bioregion is highly threatened by anthropogenic pressures as it is particularly suitable for agricultural activities and it contains significant mineral deposits as well as having high urban densities (Mucina and Rutherford, 2006). The existing high urban densities with ever increasing urbanization is one of the main threats to biodiversity in the grassland biome as the natural vegetation in and around influenced cities is destroyed at an alarming rate (Cilliers *et al.*, 2004).

The dominant vegetation type in the greater Klerksdorp area, covering 84.6 % of the area, is the Vaal-Vet Sandy Grassland (Figure 5.1 d). The vegetation type has a mean annual precipitation of 530 mm. Mucina and Rutherford (2006) classifies this vegetation type as having an endangered status (less than 60 % natural habitat left). More than 63 % of the Vaal-Vet Sandy Grassland type is transformed for cultivation with the remaining areas experiencing strong grazing pressure from cattle and sheep. The highly fragmented nature of this vegetation type can be seen in the greater Klerksdorp area from the number of roads in the area (Figure 5.1 e) and the previously discussed values for NP and LPI. A comparison of aerial photographs for the period between 1976 and 2002 revealed that 26.18 % of the grasslands found in the urban areas of Klerksdorp were transformed into various urban land uses during the twenty-six year period (Barnard *et al.*, 2008). The location of the selected grassland patches are also shown in Figure 5.1 (e).

Mucina and Rutherford (2006) compiled a habitat conservation priority map of the entire South Africa based on values for ecosystem status, protection levels and irreplaceability of each respective vegetation type. The greater Klerksdorp area has a habitat conservation priority map score of 81-90 %, therefore studies done in this area is of tremendous importance for potential conservation of the Vaal-Vet Sandy Grassland. Remaining habitats of critically endangered and endangered ecosystems are usually fragmented and likely situated in a matrix of productive agricultural, industrial and urban land uses. This almost entirely prohibits the establishment of large protected areas, creating situations where the conservation sector needs to proactively engage with industry, civil society and the local government (Mucina and Rutherford, 2006). However, Cilliers *et al.*, (2004) emphasized that the lack of detailed ecological data is one of the main problems in the implementation of conservation orientated policies in urban planning and management. The current study aims to contribute to this larger body of knowledge to assist in informed decision making.

The city of Klerksdorp was founded in 1837 as an agricultural settlement and the sporadic development of Klerksdorp began with the discovery and mining of gold in 1889. Orkney was afterwards founded in 1940 to provide accommodation for mineworkers (Department of Constitutional Development and Planning, 1986). All the grassland patches in the greater Klerksdorp area would therefore probably have had grazing influences and large parts of the region had prior agricultural influences. The natural grasslands in the landscape is characterized by the dominance of the grass *Themeda triandra;* and one of the criteria for selection of the patches for the current study was the presence of *T. triandra* (Table 5.2).

However, in *Themeda triandra* grasslands old agricultural fields are clearly identifiable and have distinct disturbed communities that persist for many years. Tainton (1999) found evidence in Kwa-Zulu Natal (one of the provinces in South-Africa) of land last cultivated approximately 80 years ago that is still covered with thatch grass (*Hyparrhenia hirta*). Furthermore, even where well-managed *T. triandra* veld adjoins *H. hirta* dominated veld of areas cultivated 30 years ago; Tainton (1999) affirmed that there was no sign of the *T. triandra* establishing in the previously cultivated areas. This is tremendously important for the current study and future research in the grassland biome as much of this biome has been disturbed by past cultivation, livestock grazing or the disruption of natural fire regimes, resulting in a severe decrease in the plant species diversity (Cilliers *et al.*, 2004).

The aim of this chapter is to use five measures to quantify the location of the patches along the urban-rural gradient, the two measures of density of people (DENSPEOP) and landscape shape index (LSI), selected as the most suitable measures in Chapter 4, together with three additional measures dominant land cover (DOMLC), land cover richness (LCR), and percent urban land cover (PURBLC). The additional measures are used to further describe the patches, particularly in order to allow a comparison of the patch characteristics and its location along the gradient with the patches identified in Melbourne as these three measures were used by Hahs (2006) as well.

5.2 Methods

5.2.1 Selecting the grassland patches

In the current study the term patch will be used to describe homogenous grassland areas delineated by physical boundaries such as roads or fences. Hahs (2006) declared that by understanding how the remaining patches of natural vegetation are responding to urbanization in the surrounding landscape it can assist in ensuring that these remnant plant communities are conserved for future generations.

To enable the quantification of the influence of urbanization on the grassland patches, a set of criteria was created to ensure that the selected patches will have a relatively similar environmental setting. This will ensure that any observed variation in the plant communities is not due to different environmental influences, but can be ascribed to differences in the location of the patch along the urban-rural gradient. The set of criteria used in the current study to ensure that the patches can be directly compared is listed in Table 5.2.

Table 5.2: The selection criteria used to select appropriate grassland patches in the greater Klerksdorp area.

	CRITERIA
1	*All the patches should be located in the Vaal-Vet Sandy Grassland vegetation type.*
2	*Patches should be natural grasslands with minimum disturbance.*
3	*Themeda triandra should be present.*
4	*Patches should be located on the same terrain type (lower-lying level ground).*
5	*No tree cover or only the minimum (able to be excluded from the sample plots).*
6	*Patches located in urban and rural areas.*
7	*Patches should be easily accessible.*
8	*Patch size greater than 0.5 ha.*

No current formal maps exist of the extent of natural grasslands in the greater Klerksdorp area. Therefore, with the aid of the 2.5 m resolution panchromatic band of the satellite image, reconnaissance surveys were done to identify potential patches complying with the chosen criteria. During these trips the 86 GPS waypoints used in the classification process of the land cover map described in Chapter 3 were also collected. The selected patches were then mapped in ArcMap of ArcView 9.2 (ESRI, 2006a) using the panchromatic band image to accurately digitize patches boundaries and to calculate the patch sizes.

Cilliers *et al.* (2008) examined patterns of exotic species invasion in the grasslands of Klerksdorp and Potchefstroom to describe the edge effects found in these fragmented grasslands. The depth of penetration of the exotic species from the edge of the respective patches varied from 3-18 m for rural sites to 1-66 m for urban sites. Consequently, no definite edge depth could be ascribed to these patches. Therefore, no edge to interior ratio was calculated for the patches as were done in the study of Hahs (2006).

5.2.2 Quantifying the surrounding landscape

The five urbanization measures, DOMLC, LCR, PURBLC, DENSPEOP, and LSI were calculated for the patches to allow the quantification of their location along the urban-rural gradient. In order to ensure that the patches were comparable to the landscape grid cells, a circle with a 565 m radius was created around the patches to equal the 1 km^2 area of the landscape grid cells. The selected urbanization measures were then calculated for these circular areas (Hahs, 2006).

To create the 1 km^2 area circles the Generate polygon centroid points tool of Hawth's analysis tools v.3.27 (Beyer, 2007) was used. This created a point layer of the centroids of each respective patch. In ArcMap of ArcView 9.2 (ESRI, 2006a) the centroids layer was then used as input layer to create a buffer around each point. The Buffer tool creates buffers with the selected radius of 565 m around the centroid points; the output is a polygon layer of circles with an area of 1 km^2.

These circles were then used as input to calculate the urbanization measures. Three of the 1 km^2 buffer circles of the patches overlap with other buffer circles where the patches are less than 1 km from each other. Two separate layers were subsequently created for the three buffer circles and the rest to allow calculation of the measures. All of the measures were calculated as described in Chapter 4, except DOMLC.

The raster based method used to create the DOMLC values of the landscape grid cannot be used with a vector shapefile. Therefore, the land cover map described in Chapter 4, where the polygons of each class were merged into one polygon was used as input for the calculation of the measure. The merged land cover map layer was clipped with both of the buffer layers. The clipped land cover layers were then intersected with the respective buffer layers. The areas of each class were computed with the Calculate Geometry tool in the attribute table of the intersected layer. The Summarize tool in the attribute table was then used to compute the maximum area for each buffer circle. The output values were then corresponded to the land cover class they represent and these values were then used directly as the DOMLC values for each patch.

5.3 Results and discussion

5.3.1 Selecting the grassland patches

During the reconnaissance survey 19 potential sites were identified of which only 14 were chosen (Table 5.3). This was done to ensure that adequate time was available to complete the vegetation surveys as well as the seedling emergence study of the soil seed bank analysis in the glasshouse. This differs from the study in Melbourne where the patches selected by Hahs (2006) represented all of the existing documented localities that met their selection criteria.

The general attributes for each of the 14 patches are listed in Table 5.3. Most of the patches are open spaces in the residential areas of Klerksdorp and Orkney, but the majority of them functions as recreational areas. Only a few of them such as patch 1 are potentially designated for residential development in the immediate future.

Table 5.3: Attributes of the 14 grassland patches of *Themeda triandra* in the greater Klerksdorp area.

Patch Nr	Patch Name	Suburb/Location	Patch Size (ha)	Distance to CBD (km)
1	*Steinbeck Boulevard*	*Vaalpark, Orkney*	*1*	*16.16*
2	*Carlyle Street*	*Vaalpark, Orkney*	*1.66*	*15.2*
3	*Dickens Street*	*Golf Park, Orkney*	*0.93*	*14.26*
4	*Milton Street*	*Rotary Park, Orkney*	*10.06*	*13.77*
5	*R30, Ariston Station*	*Outskirts of Orkney*	*1.69*	*7.24*
6	*Markotter Stadium*	*Roosheuwel, Klerksdorp*	*25.42*	*3.48*
7	*Park Street*	*Neserhof, Klerksdorp*	*1.53*	*1.89*
8	*Wilken Street*	*Meiringspark, Klerksdorp*	*4.3*	*3.93*
9	*Maricelle Street*	*Wilkoppies, Klerksdorp*	*0.9*	*3.7*
10	*Odendaal Street*	*Flimieda, Klerksdorp*	*9.93*	*3.91*
11	*Mostert Street*	*La Hoff, Klerksdorp*	*0.83*	*4.34*
12	*Jansen Street*	*La Hoff, Klerksdorp*	*4.69*	*4.86*
13	*R503*	*Next to Alabama, Klerksdorp*	*36.89*	*5.9*
14	*Yzerspruit Farm*	*South-west corner of the study area*	*67.95*	*23.65*

All of the 14 patches had clear boundaries delineated by fences, residential areas or roads, but three of the patches were closer than 1 km to each other, namely patches 1, 4 and 11. The relatively small size of the urban areas of the greater Klerksdorp area made it difficult to find suitable grassland patches that were far enough removed from each other. The land cover map of the greater Klerksdorp area, classified in

Chapter 3, was used as a background to show the location of the selected 14 patches in the greater Klerksdorp area. The map is shown in Figure 5.2. A map of each of the individual patches is given in Appendix C. Take note that no suitable grassland patches were found in the densely populated areas suburbs of Jouberton and Kanana. Almost all of the available space in these areas is used for housing. The few remaining open space in these areas is highly disturbed and most are used for dumping or burning refuse. The grasslands immediately surrounding these suburbs act as communal grazing lands and overgrazing by cattle and goats occur.

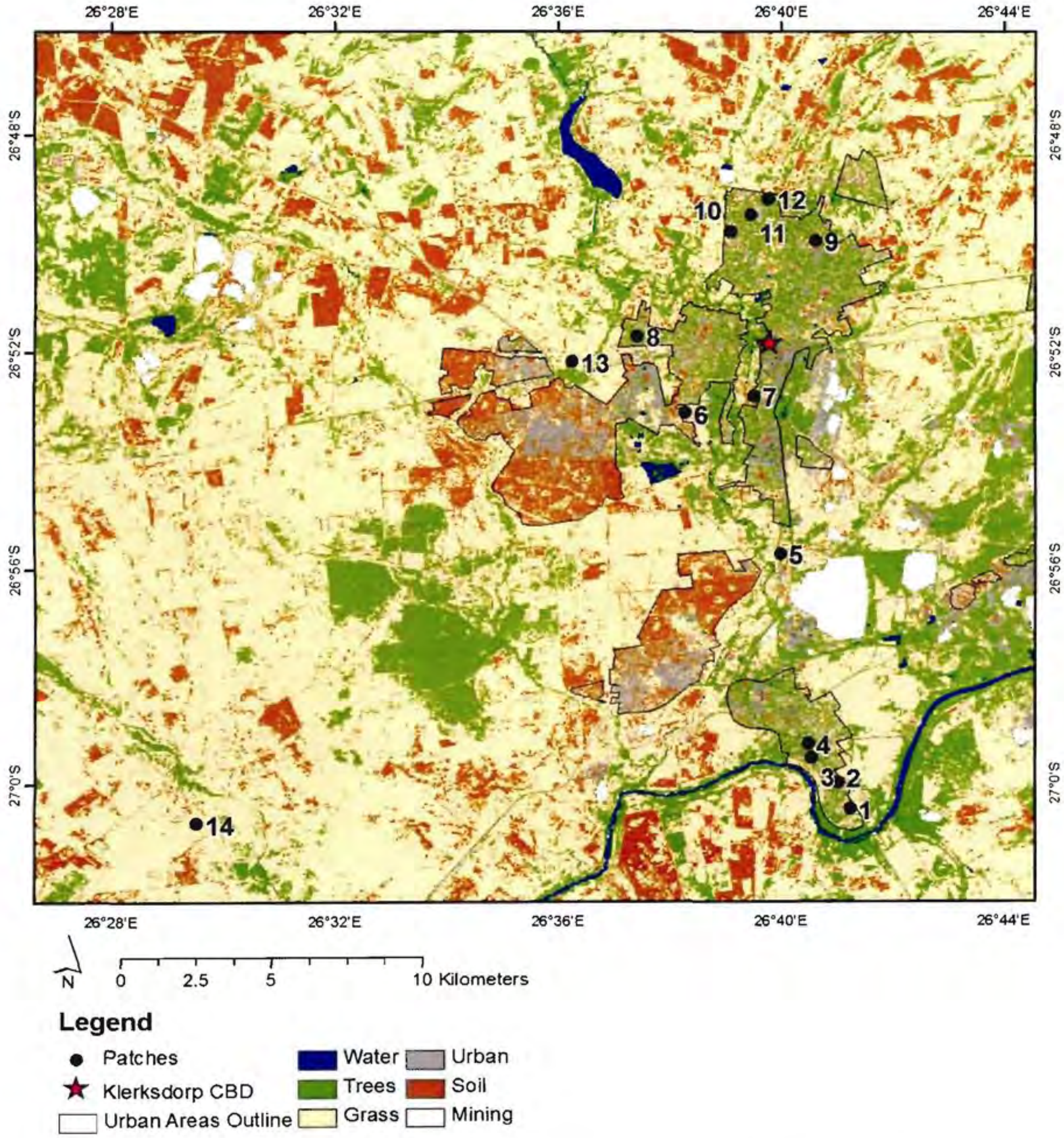


Figure 5.2: Land cover map of the greater Klerksdorp area, showing the location of the selected grassland patches.

The location of patch 14 is far from the other patches in the study area (Figure 5.2), because no suitable patches could be found nearer to the city. This patch was selected to be the rural control plot. Patch 13 was also considered as rural even though it is located close to Klerksdorp. Large farms and grassland areas border patch 13, hence its classification as a rural plot.

Large areas with extensive shrub and tree cover extend towards the south western part of the study area, interspersed with open grassland patches. However, many of these grassland patches are disturbed due to overgrazing by cattle. Large cultivated lands where mainly maize is grown also cover extensive areas of the greater Klerksdorp area. Patch 14 is a large farm camp used as grazing for cattle, but the vegetation show minimum disturbance and the camp is allowed to rest part of the year and it had a low stocking rate when the surveys were done.

Figure 5.3 shows photographs of six of the patches as examples of the typical vegetation communities found in the 14 selected patches. The dense stands of *T. triandra* can clearly be seen; especially in patches 13, 10 and 8 (Figure 5.3 (b), (d) and (e) respectively). Two of the patches, namely: 12 (Figure 5.3 c) and 9 (Figure 5.3 f) were mown when the pictures were taken.

The patches cover a total area of 167.78 ha. The individual patches vary in size from 0.9 ha (patch 9) to 67.95 ha (patch 14) (Table 5.3). However, most of the patches are smaller than 10 hectares which can be expected as urban areas are known to be highly fragmented. The distance from the CBD ranges from 1.89 km to 23.65 km.

One important fact to highlight is that because Orkney was included in the greater Klerksdorp area and the selection of grassland patches, the distance from the CBD of the patches located in Orkney will reflect the distance from the CBD in Klerksdorp. Nonetheless, the attributes of the patches are typically similar to those found in Klerksdorp located 1-5 km from the CBD.

Figure 5.4 shows the location of the patches along the observed gradients of the patch size distribution and the respective distances of the patches from the CBD. In Figure 5.4 (b) the Orkney patches (patch 1-5) can clearly be distinguished from the rest with an average of 15 km from the CBD of Klerksdorp.

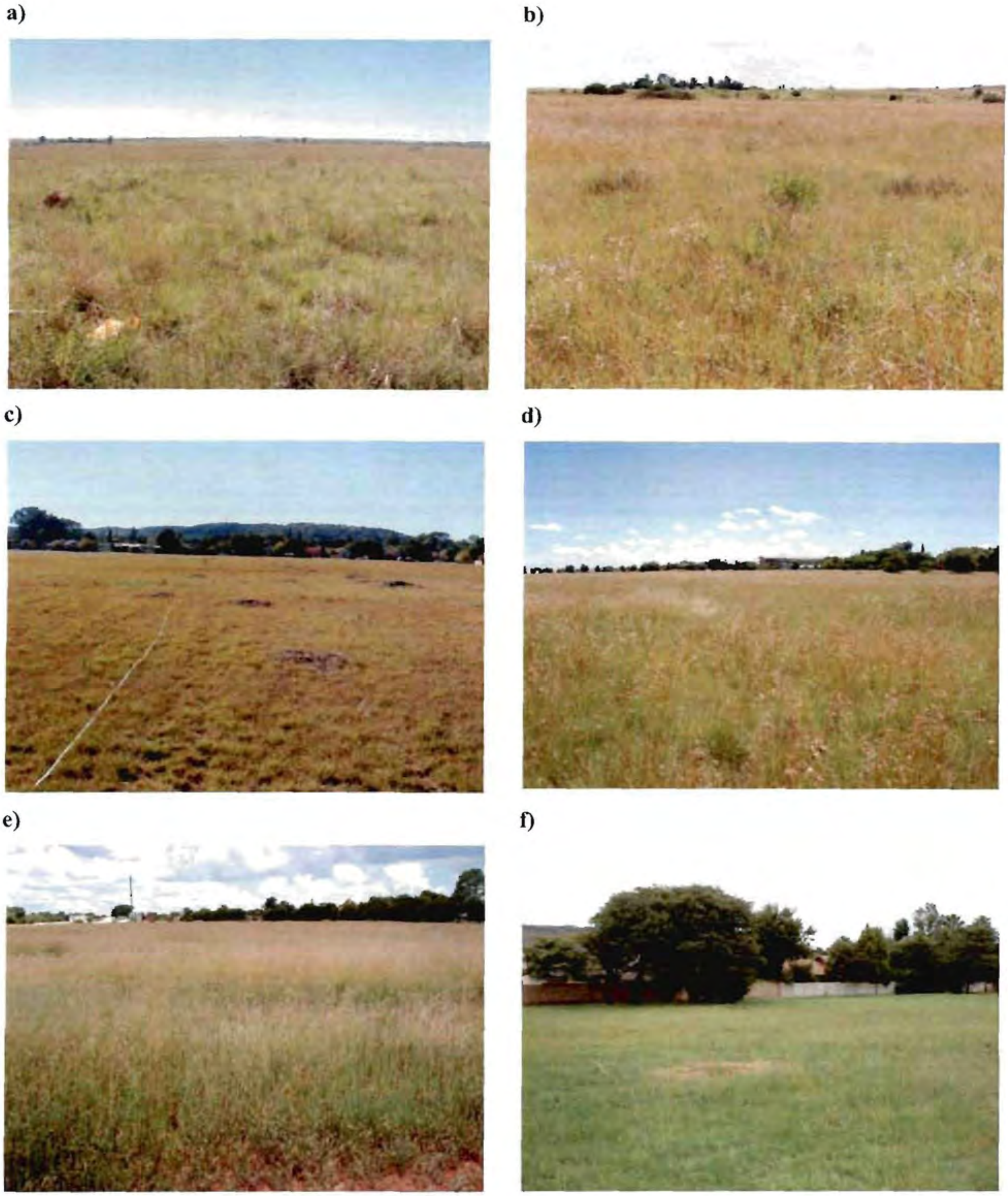


Figure 5.3: Photographs of some of the *Themeda triandra* grassland patches in the greater Klerksdorp area. The patches are arranged from the rural (patch 14) to the most urbanized site (patch 9). The photographs show patch 14 (a), patch 13 (b), patch 12 (c), patch 10 (d), patch 8 (e), and patch 9 (f).

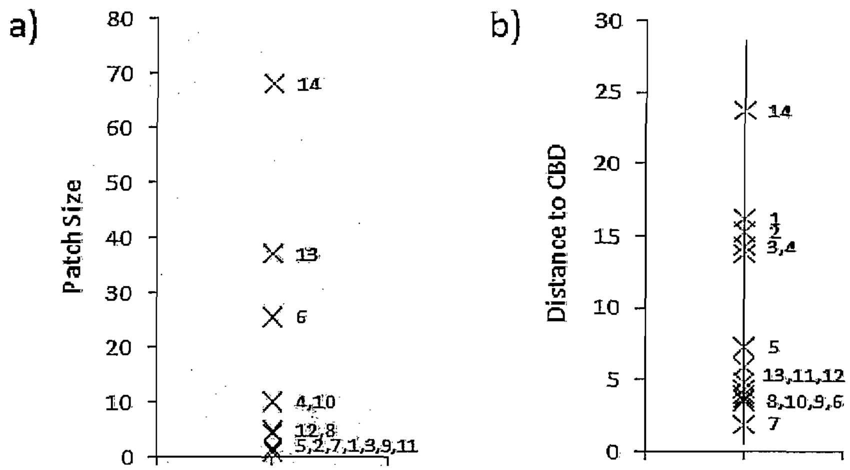


Figure 5.4: Graphs of the distribution of the fourteen selected grassland patches along the two gradients of patch size (a) and the distance to the CBD (b). The solid line in the distance to CBD graph (b) represents the range of values recorded for the entire landscape grid. The crosses represent the patches and are labelled according to the patch numbers.

To illustrate the general correlation, typically found in most studies in urban areas between the patch size and the distance from the CBD, the urban patches of the Klerksdorp area including the rural patches were compared. Figure 5.5 illustrates the positive linear relationship that can be observed. The Orkney patches were omitted because of the discrepancy in the distance from the CBD measure described earlier.

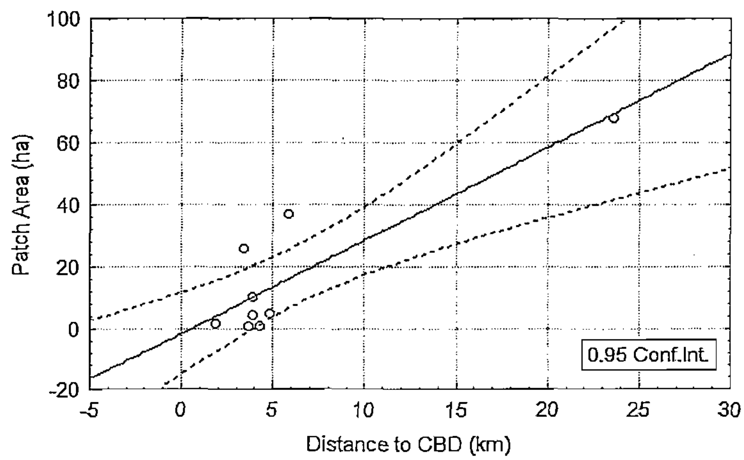


Figure 5.5: Graph showing the positive correlation between the distance to the CBD and the patch size for the patches located in Klerksdorp and the surrounding rural areas.

5.3.2 Quantifying the surrounding landscape

The two measures of urbanization, DENSPEOP and LSI that were identified in Chapter 4 as describing the variation in the landscape the best, were used to quantify the locations of the grassland patches along the observed gradients. The two measures, used in combination (Hahs, 2006), adequately quantify the placement of the patches along the entire observed landscape gradient where each patch can be distinguished from one another based on the respective values of the two measures. However, the two measures did not describe the patch locations in enough detail to specifically elaborate on their locations. Therefore, the additional measures of DOMLC, LCR, and PURBLC also used by Hahs (2006) to describe the patches in Melbourne were used to further describe the individual patches. Table 5.4 shows the values of the selected measures as calculated for each patch.

The measure of DOMLC is an indication of the types of disturbances and influences that are potentially present in the landscape (Hahs, 2006). The values for DOMLC range between 3 (grass) and 2 (trees). The majority of the patches have a value of 3, which is also the overall average value for all the urban areas in the landscape. Most of the urban patches are located in residential areas. Almost all of the streets in Klerksdorp and Orkney are lined with trees and nearly all of the gardens in the residential areas have trees. Therefore, as explained in Chapter 4, a third of the urban areas of Klerksdorp and Orkney are classified as trees. The tree cover increases towards the residential areas; hence the more urbanized smaller patches have a DOMLC of trees. The LCR clearly distinguishes between the urban and rural sites. The urban area sites have a LCR value of 5 and the two rural patches (patch 14 and 13) a LCR value of 4. However, the additional class in the urban areas are repeatedly the water class. Some areas have water, but the majority of the classed water is areas of shade with a minor amount of burned patches. The satellite image records these areas, together with water bodies as the same class due to spectral signature similarities (Rashed *et al.*, 2001). Therefore, no real distinction could be made between the remaining patches (1-12) with regards to the values of LCR.

The values of PURBLC range between 0.03 % (patch 14) and 29.55 (patch 9). The average value for all the patches is 15.66 % which is below the average of all the urban areas in the landscape, namely 19.76 %. The values of PURBLC were useful in distinguishing between the different patches. The values of DENSPEOP range between 584 (patch 9) and 36 (patch 14). The average for all the patches is 274.57; though the average for all the urban areas of the landscape is 1412.6. There is a clear distinction between the values of the rural farm patch and the urban sites. Patch 13 has rural surroundings, but it is situated quite close to a residential area hence its high value for DENSPEOP. Some of the patches have exactly the same values; this is due to the census data scale as discussed in Chapter 4.

LSI, however shows a well distributed range of values and each patch can be distinguished from each other based on its LSI value. The values range between 4.4 (patch 14) and 36.3 (patch 11). This measure is an indication of the landscape complexity with the distinction between urban and rural areas clearly observable, highlighting the highly fragmented nature of urban areas. The average value for the patches is 26.07 which are above the average for all the urban areas of the landscape of 23.79.

Table 5.4: Landscape attributes for the 14 grassland patches of *Themeda triandra* grassland in the greater Klerksdorp area

Patch Nr	DOMLC	LCR	PURBLC	DENSPEOP	LSI
1	*3*	*5*	*13.75*	*297*	*28.57*
2	*2*	*5*	*11.19*	*297*	*27.48*
3	*2*	*5*	*15.9*	*297*	*33.11*
4	*2*	*5*	*18.69*	*297*	*35.55*
5	*3*	*5*	*10.92*	*297*	*21.19*
6	*3*	*5*	*13.61*	*246*	*20.81*
7	*3*	*5*	*19.39*	*246*	*29.03*
8	*2*	*5*	*21.12*	*251*	*32.69*
9	*2*	*5*	*29.55*	*584*	*32.2*
10	*3*	*5*	*15.64*	*251*	*24.99*
11	*2*	*5*	*28.19*	*251*	*36.3*
12	*3*	*5*	*16.59*	*251*	*29.34*
13	*3*	*4*	*4.71*	*243*	*9.43*
14	*3*	*4*	*0.03*	*36*	*4.4*

Figure 5.6 shows the locations of the patches along the gradients of the measures listed in Table 5.4. The locations of the patches are also compared to the rest of the landscape by including the range of values encountered for each measure as was calculated for the greater Klerksdorp area. The value range is shown in Figure 5.6 as a solid black line on each graph. The range for DENSPEOP measured for the entire landscape was not included because the maximum value reaches 10517 compared to only 584, which is the maximum value for the selected grassland patches. Therefore in order to allow a visual comparison of the values of DENSPEOP for each patch the entire landscape range for the values was omitted.

Only the values for LSI of the patches were distributed along the gradient observed for the entire landscape. This illustrates the measure's usefulness in distinguishing between the patches and in describing their location along the urban-rural gradient. Nonetheless, only using the values of LSI to describe the gradient would give a false impression of the representation of the patches for the entire landscape. The values for the patches of DENSPEOP are for example all located at the lower half of the observed DENSPEOP gradient for the entire landscape.

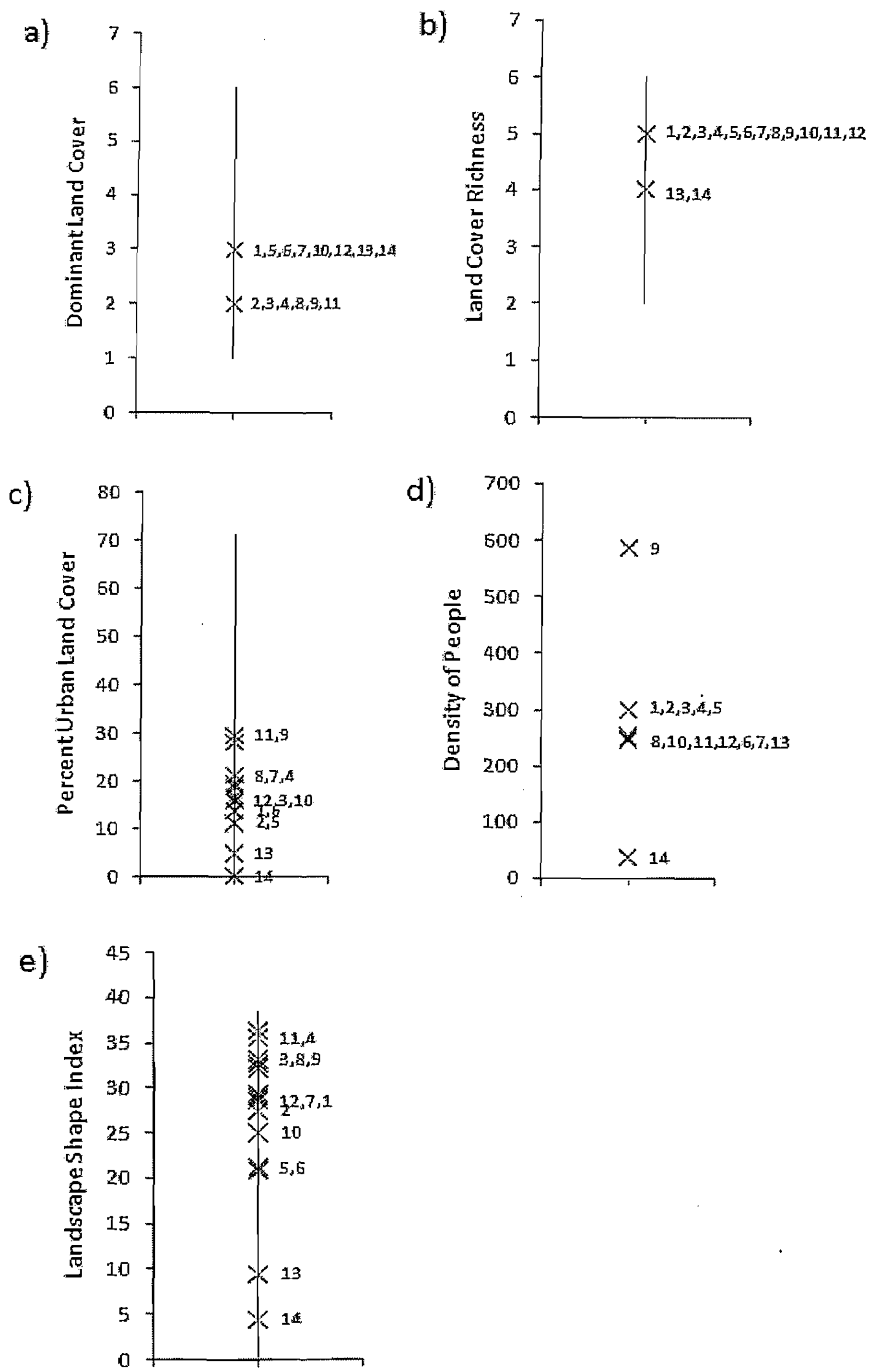


Figure 5.6: Graphs of the distribution of the fourteen selected grassland patches of the greater Klerksdorp area along the four gradients of DOMLC (a), LCR (b), PURBLC (c), DENSPEOP (d), and LSI (e). The solid line in the graphs represents the range of values recorded for the entire landscape grid. The range of values for DENSPEOP was omitted to allow differentiation of the individual patch values. The crosses represent the patches and are labelled according to the patch numbers.

Figure 5.7 gives a graphical illustration of groupings of the value range of DENSPEOP recorded throughout the landscape. The red bars indicate the groupings of values that include those that were recorded for the selected patches. It is clear that the patches are poor representatives of the entire DENSPEOP gradient. This is due to the fact that no vegetation has been sampled in the areas of the landscape with the highest population density, namely the residential areas of Jouberton and Kanana. The values of PURBLC also reflect a similar situation to DENSPEOP, as Figure 5.6 (c) illustrates that the patches are distributed in the lower half of this gradient as well.

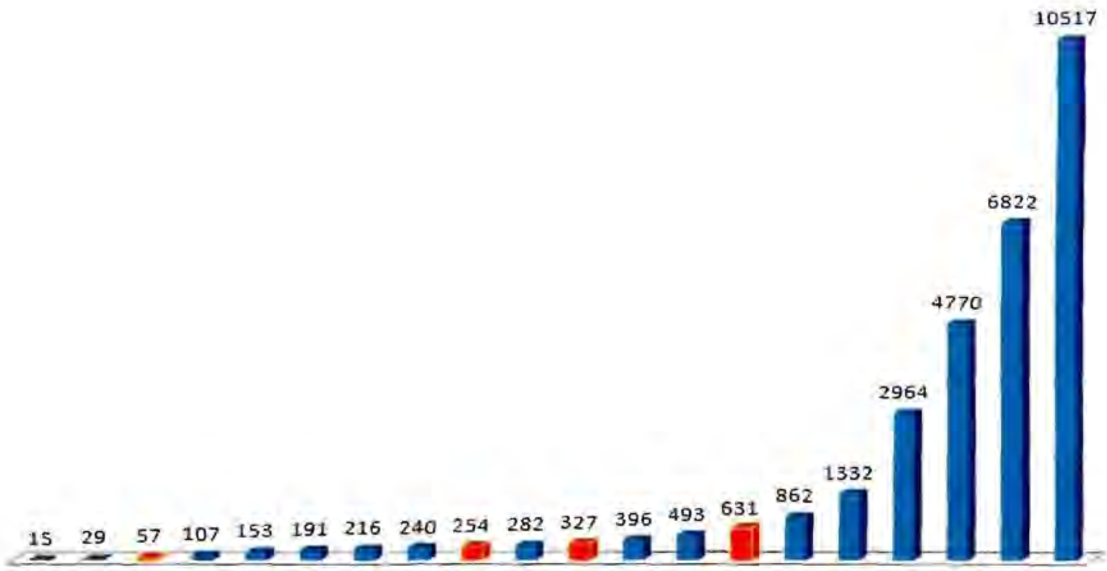


Figure 5.7: Graph of the values observed for DENSPEOP in the landscape. Each bar represents a range of values beginning from the end of the value shown on top of the previous bar. The first column represents values ranging from 8-15. The proportion of grid cells attributed to each grouping is not indicated on the graph. The length of each bar only indicates the size of the last value of the group. The red bars are the groups with values represented by the selected grassland patches. The values on top of each bar indicate the maximum value for each group.

Table 5.5 shows the relative gradients formed by the patches for each of the calculated measures as represented in Figure 5.6. The green highlighted patches are those with the lowest values for the five measures matching the rural control plot (patch 14). Red indicates patches located in the urban areas, possessing similar characteristics. Some of the measures show an added subdivision of the urban patches with yellow indicating moderate urbanization and orange showing higher urbanization scores for this specific study. Moderate and higher are used to describe the urbanization intensity, because in context with enormous urban agglomerates such as Melbourne the entire urban area of Klerksdorp is small.

However, the interpretation of the gradients represented by the patches is not that straightforward, especially with regards to the demographic variable of DENSPEOP. Patch 9 and 11 are indicated by

PURBLC and DENSPEOP as having higher urbanization scores, but the DENSPEOP value of patch 9 is 584, which is only half of the average value of the urban areas. Therefore, according to this measure all the sampled urban areas are only moderately urbanized with regards to the total proportion of urbanization in the landscape. However, patch 13 and 14 can clearly be distinguished as rural areas in most of the other observed gradients of the different measures.

Table 5.5: List showing arrangement of the individual patches along the observed gradients of the five urbanization measures and the patch size of each patch, deduced from the groupings of the patches for each measure as in Figure 5.6. The white columns respectively show the gradient values (v) for the different measures as used to create the patch groupings. The different colours represent, urban (red), rural (green), high urbanization (orange) and moderate urbanization (yellow).

DOMLC	v	LCR	v	PURBLC	v	DENSPEOP	v	LSI	v	Size	v
2		*1*		*9*		*9*	*300*	*11*		*11*	*0*
3		*2*		*11*	*28*	*1*		*4*		*9*	
4		*3*		*8*		*2*		*3*		*3*	
8		*4*		*7*		*3*		*8*		*1*	
9		*5*		*4*		*4*		*9*	*30*	*7*	
11	*2*	*6*		*12*		*5*		*12*		*2*	
1		*7*		*3*		*8*		*7*		*5*	
5		*8*		*10*		*10*		*1*		*8*	*4*
6		*9*		*1*		*11*		*2*		*12*	
7		*10*		*6*		*12*		*10*		*10*	
10		*11*		*2*		*6*		*5*		*4*	
12		*12*	*5*	*5*	*5*	*7*		*6*	*10*	*6*	*20*
13		*13*		*13*		*13*	*100*	*13*		*13*	
14	*3*	*14*	*4*	*14*	*0*	*14*	*0*	*14*	*0*	*14*	

These situations as found with LSI and DENSPEOP described above, corroborates the importance of using a combination of measures to quantify the urban-rural gradient as declared by Hahs (2006). A recent study by Tavernia and Reed (*in press*) also confirmed the importance of using a combination of urbanization measures.

Figure 5.8 shows the specific patches located at both ends of the DENSPEOP gradient. Patch 9 represents the highest value (Figure 5.8 a) with patch 14 representing the lowest observed values (Figure 5.8 b). However, the true range extremities as observed for the entire landscape are also indicated. The highest value is found in a grid cell located in Jouberton where no suitable patches were found, the absence of trees and gardens is also evident (Figure 5.8 c). The lowest value is found in the northern parts of the landscape (Figure 5.8 d) forming part of the surrounding rural areas of the greater Klerksdorp region. Important to note is that the lowest value found in the landscape is shared by 209 grid cells which are included in the same census collection ward. One of the grids was randomly chosen to represent the lowest value.

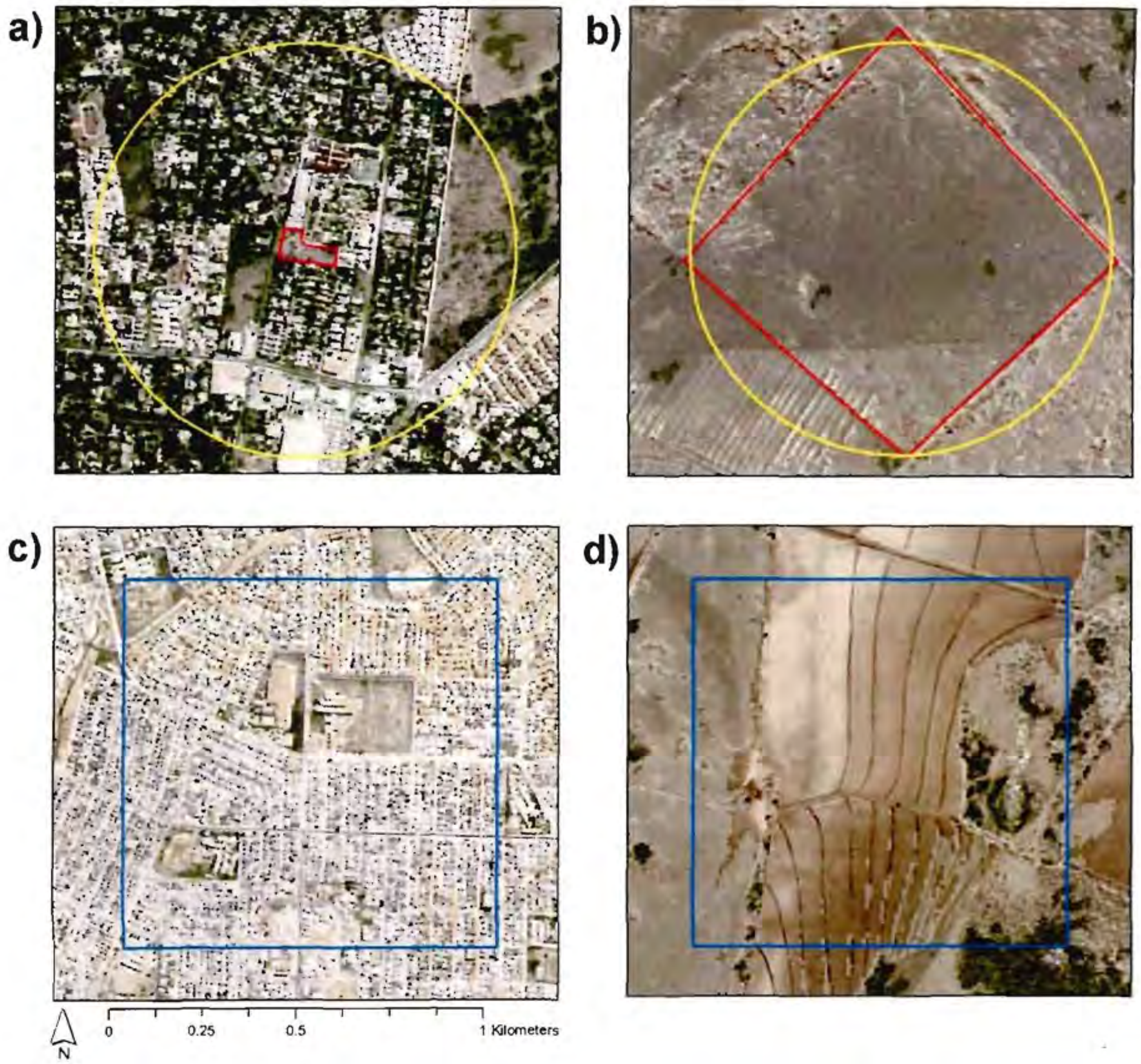


Figure 5.8: The grassland patches at both ends of the density of people gradient for the selected patches: highest DENSPEOP (patch 9) (a), and lowest DENSPEOP (patch 14) (b). The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The grid cells representing the range of values for the entire landscape are the highest DENSPEOP (c) and the lowest DENSPEOP (d). The grid cell boundary is shown in blue. The background for all the maps is the overlaid panchromatic band image on the land cover classification map.

5.4 Summary

The selected grassland patches represent the majority of the grassland patches found in the landscape; however they are not representative of the complete gradient for the demographic measures of urbanization. This is because there were no suitable grassland patches to sample in the areas with the highest population densities.

The heterogeneity of the values of the measures per patch illustrates the importance of using more than one measure to quantify the location of the patches along the urban-rural gradient. Land cover richness and dominant land cover did not allow any detailed distinctions between the patches. The values of landscape shape index allowed the most detailed differentiations between the patches. However, the values of density of people are needed to accurately describe the urban context.

As basic measures for quantifying the location of the patches along the urban-rural gradient the measures of density of people and landscape shape index work well and can be used without the addition of other measures. However, percent urban land cover allows a more instantly understandable measure of the gradient. A quick impression can be formed of the surroundings based on the amount of urban land cover measured. Therefore, as an additional and universal measure, percent urban land cover can be used as a specific measure to describe the patches.

Tavernia and Reed (*in press*) recently examined the effect of varying spatial extents and habitat context on correlations between demographic and landscape measures. Principal component analysis (PCA) was used to determine the correlation between measures; however as with Hahs (2006) all the variables on the first component loaded almost equally with low values. This happened because a PCA explains the maximum amount of variation in the data starting with the first component. A factor analysis, on the other hand, shows how the variables individually describe underlying factors causing the variation in the data and how they correlate with each other. This subsequently identifies the redundant measures as done on the current study (Chapter 4). But, in emphasis "*it should be noted that the ability to serve as a proxy in a statistical sense does not suggest that these urban features will have similar biological effects*" (Tavernia and Reed, *in press*).

However, even though Tavernia and Reed (*in press*) quoted Hahs and McDonnell (2006) on the importance of using both landscape metrics and demographic variables, none of the landscape metrics indentified in Hahs and McDonnel (2006) was used in their study. The landscape scale measures used by Tavernia and Reed (*in press*) were different cover estimates (ha) of different land cover types with only population density, road length and impervious surface cover to represent the demographic and physical measures. None of these measures were included in the subset of measures proposed by Hahs and McDonnell (2006) and in the current study road network density had low scores on the factor analysis. However, density of people was identified by the current study as being a useful measure.

To conclude, even though both landscape and demographic measures were examined in Tavernia and Reed (*in press*) no big contribution was made by them on the elucidation of a general set of measures to describe the broad underlying matrix of urban-rural gradient studies, especially with regards to the landscape metrics where none of the measures identified by Hahs and McDonnell (2006) were included.

The following chapter (6) will discuss the vegetation patterns and distributions observed for the different patches. Chapter 7 will then subsequently explore how the different species richness values of the described vegetation correlate to the observed urbanization gradients.

Chapter 6: Patterns of plant diversity and species composition

6.1 Introduction

"The relative influences of urban and natural environmental factors on ecosystem patterning, and the extent to which ecosystem processes are also influenced, could be examined most easily along urban-rural gradients, where human influences can be directly quantified" (McDonnell and Pickett, 1990).

The spatial heterogeneity in urban landscapes is a major feature of urban vegetation; this is created by the vast array of building densities and types, various land uses and different social contexts (Pickett *et al.*, 2001). Furthermore, individual plant species of remnant natural vegetation communities in urban areas reveal diverse responses to urbanization based on the longevity of the plant species, the individual species involved and the nature of the changes to the remnant patch (Hahs, 2006).

In general, urban areas are inhabited by greater numbers of plant species when compared to the surrounding rural areas (Rebele, 1994; McKinney, 2002; Hope *et al.*, 2003; Wania *et al.*, 2006; Millard, 2008; Vallet *et al.*, 2008). This is usually ascribed to the heterogeneity of urban environments, the subsequently higher diversity of potential available habitats, and the high amounts of exotic species introduced into urban areas. However, McKinney (2008) found that in 69.2 % of the studies along urban-rural gradients that he reviewed focusing on plant species richness, the species richness was the highest at intermediate levels of urbanization, with none of the studies recording increases in the species richness levels at the highest levels of urbanization.

The process of urbanization has resulted in the increase of exotic plant species (Roy *et al.*, 1999; Sukopp, 2004; Kühn and Klotz, 2006; Cilliers *et al.*, 2008; Heckmann *et al.*, 2008) and the decline of native species in areas subjected to urbanization influences (Kent *et al.*, 1999; Sukopp, 2004; Kühn and Klotz, 2006; Williams, 2007). In studies focusing on remnant patches of vegetation in urban areas larger patches had higher native and overall species diversities than smaller patches (McDonnell *et al.*, 1997; Godefroid and Koedam, 2003; Hahs and McDonnell, 2007), with higher species diversity also found in patches located at the urban part of the gradient.

However, to obtain a complete understanding of the dynamics of vegetation in urban environments studies of the soil seed bank should be included as well. Investigating the composition of the soil seed bank will give an indication of the temporal dynamics of the soil seed bank, indicating the possible response of a plant community after disturbance (Hahs, 2006). The proportion of exotic species present in the soil seed bank represents an indicator of the intensity and length of exposure to urbanization influences, where the relative abundance of the indigenous and exotic constituents in the soil seed bank could indicate potential changes in the composition of the future plant community based on the available propagules (Hahs, 2006).

In a paper on the seasonal variation of germinable seed banks of grass species in an urban forest reserve, Odgers (1999) indicated that many exotic species typically have persistent soil seed banks with native species having small transient seed banks that are seasonally dependent. Therefore, if regular disturbance occurs small transient seed banks may be replaced by larger persistent seed banks of exotics, where the exotic species could then eventually replace native species in the above ground vegetation. The quantification of the soil seed bank within grassy woodlands along an urban-rural gradient in Melbourne revealed that exotic monocots dominated the soil seed bank (Hahs, 2006). This is similar to the findings of Kostell-Hughes *et al.* (1998) and Stewart *et al.* (2004) where exotic species also dominated the soil seed banks in urban areas.

In general, studies of soil seed banks revealed that soil seed banks constitute a species poor version of the higher diversity above ground vegetation community (Kostell-Hughes *et al.*, 1998; Morgan, 1998; De Villiers *et al.*, 2003; Hahs, 2006; Solomon *et al.*, 2006; Vécrin *et al.*, 2007). However, it should be remembered that Gross (1990) warned that the use of the germination method in estimating soil seed banks allows the risk of underestimating the number of seeds in the seed bank owing to the detection of only the easily germinable part of the seed bank.

The under representation of existing species in the soil seed bank suggests that the soil seed bank does not play a major contributing role in maintaining species diversity within existing vegetation communities (Hahs, 2006). Morgan (1998) reciprocated by stating that the size of the long-term soil seed bank of grassland vegetation suggested that it has little functional importance for many native species and it probably contributes little to seeding regeneration processes following disturbance. He elaborates by suggesting that the high number of indigenous species with perennial life-forms and persistent buds and tubers, may reduce the need for a soil seed bank.

This chapter will therefore explore the following specific objective, namely: to use vegetation and soil surveys to quantify the influence of human impacts on grassland ecology, investigating aspects such as plant species composition and diversity of existing and soil seed bank vegetation; and specific soil properties for the grassland fragments of the greater Klerksdorp area.

6.2 Methods

Vegetation surveys were done for each of the identified grassland patches in the landscape. The characteristics of these patches and their location along the measured gradients were described in

Chapter 5. Table 6.1 shows the selected patches with their respective patch size and the number of sample plots surveyed in the patch. A total of 45 sample plots were surveyed for the 14 patches.

Table 6.1: The 14 patches of *Themeda triandra* grassland in the greater Klerksdorp area, indicating the patch size and the number of sample plots per patch.

Patch Nr	Patch Name	Suburb/Location	Patch Size (ha)	Number of sample plots
1	*Steinbeck Boulevard*	*Vaalpark, Orkney*	*1*	*2*
2	*Carlyle Street*	*Vaalpark, Orkney*	*1.66*	*2*
3	*Dickens Street*	*Golf Park, Orkney*	*0.93*	*2*
4	*Milton Street*	*Rotary Park, Orkney*	*10.06*	*4*
5	*R30, Ariston Station*	*Outskirts of Orkney*	*1.69*	*2*
6	*Markotter Stadium*	*Roosheuwel, Klerksdorp*	*25.42*	*4*
7	*Park Street*	*Neserhof, Klerksdorp*	*1.53*	*2*
8	*Wilken Street*	*Meiringspark, Klerksdorp*	*4.3*	*3*
9	*Maricelle Street*	*Wilkoppies, Klerksdorp*	*0.9*	*2*
10	*Odendaal Street*	*Flimieda, Klerksdorp*	*9.93*	*4*
11	*Mostert Street*	*La Hoff, Klerksdorp*	*0.83*	*2*
12	*Jansen Street*	*La Hoff, Klerksdorp*	*4.69*	*3*
13	*R503*	*Next to Alabama, Klerksdorp*	*36.89*	*6*
14	*Yzerspruit Farm*	*South west corner of the study area*	*67.95*	*7*

6.2.1 Existing vegetation

Vegetation surveys were done in February and March of 2008, in the flowering season of most of the plants, to aid in the identification of the species. As done by Hahs and McDonnell (2007), standard quadrat sampling techniques were used. Sample plots were randomly located throughout the patch with a minimum distance of 20 m from the edge of the patch. As discussed in Chapter 5, Cilliers *et al.* (2008) examined patterns of exotic species invasion in the grasslands of Klerksdorp and Potchefstroom to describe the edge effects found in these fragmented grasslands. No definite edge depth could be ascribed to these patches. However, to reduce the effects of any potential edges none of the sample plots were placed in a 20 m area from the edge of each grassland patch.

A minimum of 2 sample plots were surveyed per patch, the number of additional plots increased with increasing patch sizes from 3 to 7 added plots. The largest patch (patch 13) had 7 sample plots (Table 6.1). The location of each sample plot per patch is shown on the map of each patch in Appendix C.

Sample plots of 20 m x 20 m in size were used as done in the study of Hahs and McDonnell (2007). However, only the size of the sample plots was similar, as Hahs and McDonnell (2007) used sampling methods especially designed to sample forests. In the current study surveys were done in grasslands with no tree cover, and consequently the sampling method used was designed to optimally sample grassland species. Figure 6.1 illustrates the design of the sample plot. Each plot was divided into 5 parallel transects along which sampling were done. The first transect was 2 m from the edge of the sample plot with the next 4 transects spaced 4 m apart (Figure 6.1 a). Along each transect sampling were done in 1 m intervals, resulting in 20 sampling points per transect with 100 sampling points in total for each plot. At each sampling point the nearest tree, shrub, grass and forb was noted. A forb is a non-grass plant without any secondary thickening, in other words non-woody (Low and Rebelo, 1998). The plant species had to be within a 30 cm range from a sampling point to be recorded. The area taken into consideration in the identification of the species for each sampling point is shown in Figure 6.1 (b). A 1 x 4 m area around each sampling point was used; with the first and the last sampling point in each transect representing a 0.5 x 4 m area as illustrated in Figure 6.1(b).

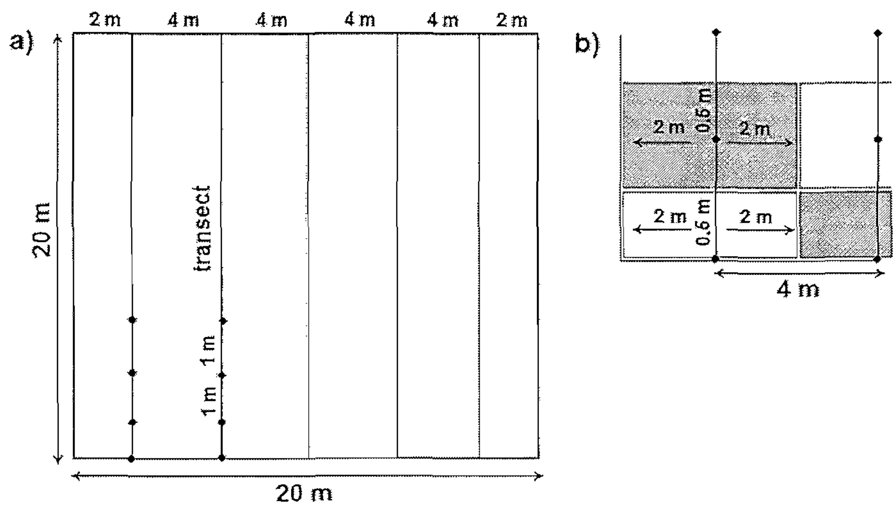


Figure 6.1: The design of the sample plots used to survey the existing vegetation. Each plot was divided into 5 parallel transects with 20 sampling points along each transect (a). A 1 x 4 m area around each sampling point was used to identify the nearest tree, shrub, grass and forb (b). However, the first and the last sampling point in each transect only represented a 0.5 x 4 m area (b).

After all 5 transects were surveyed the entire plot was searched and any additional tree, shrub, grass or forb recorded and included in the data analysis. Any unknown species recorded in the plot were collected and pressed for later identification. The location of each sample plot was recorded with a GPS to allow the soil seed bank and laboratory soil samples to be obtained for each plot at a later stage.

6.2.1.1 Data analysis

The unknown plant species were identified with the use of herbarium specimens and books and field guides of the vegetation of the region. The plant names follow Germishuizen *et al.* (2006). The species were also grouped into general categories, namely: all species, indigenous, exotic, annual, perennial, dicot, grasses, monocots (excluding grasses), and geophytes. The exotic species were also identified from Germishuizen *et al.* (2006).

The frequency of each species recorded per sample plot was used for subsequent non-metric multidimensional scaling (NMDS) analysis. The number of individuals counted per sample plot could be used directly as there were 100 sampling points per plot. The additional tree, shrub, or forb species recorded in each sample plot, were each given a frequency of 1.

Non-metric multidimensional scaling

Following Hahs and McDonnell (2007), the species composition of the sample plots of all the patches was compared using the ordination method of non-metric multidimensional scaling (Clarke, 1993). The software program Primer 5 (Clarke and Gorley, 2001) was used to perform the ordinations. Similarity between the sample plots were analyzed using the Bray-Curtis dissimilarity index (Hahs and McDonnell, 2007) on square root transformed data to allow a greater contribution from the rarer species (Clarke, 1993; Robertson and James, 2007). The ordinations were performed using the frequency data recorded during the vegetation surveys. Ordinations were performed using all the species and subsets of the data representing indigenous, exotic, annual, perennial, dicot and grass species.

The results of the ordination show the sample plots arranged in the ordination space according to the vegetation composition of each plot. The further the distance between the points (sample plots) in the ordination space the higher the degree of dissimilarity between the points (Kent and Coker, 1992). Therefore, plots closely grouped together are more similar. To assess the success of the ordination in preserving the relationships between the samples, Clarke (1993) stated that the simplest indicator is that of the stress value computed for each ordination. Stress values below 0.1 corresponds to a good ordination, stress below 0.2 is still useful, however ordination values with values near 0.2 can potentially be misleading and too much reliance should not be placed on the details of the sample arrangements in the ordination. By the time the value reached 0.35 the samples are effectively randomly placed. However, Clarke (1993) also warned that the use of the suggested stress value guidelines, as discussed, are simplified as the stress values tends to increase with increased numbers of samples used.

The effect of increased stress values with increased numbers of samples used was observed in the current study as ordinations where some plots were removed showed a drastic decrease in the observed stress value. Plots were removed when they formed outliers that caused the rest of the plots to densely cluster in one place or when no species were recorded in the plots for the subset of the data in question.

6.2.2 Soil seed bank

Samples for the soil seed bank analysis were collected in April 2008. Soil samples were collected for all of the selected grassland patches. Four soil samples were collected per sampling plot, as identified in the surveys of the existing vegetation. During the soil seed bank survey soil samples were also collected for laboratory analysis, but this will be discussed in the next chapter.

To collect the soil samples, a 1 x 1 m square quadrat was randomly placed in the 20 x 20 m sampling plot. An area of 50 x 50 cm, 5 cm deep was removed with a shovel. Each sample was put in a plastic bag and marked. For each of the four soil samples the quadrat was moved to another part of the sample plot. Therefore, four random 50 x 50 x 5 cm samples were collected for each sampling plot (marked A – D). A 1 m^2 area of soil was removed per sample plot. In total 180 soil samples was taken representing 45 sample plots and a sample area of 180 m^2. Figure 6.2 illustrates the soil sampling method (a), showing a photograph (b) of the quadrat with some of the 50 x 50 cm area removed.

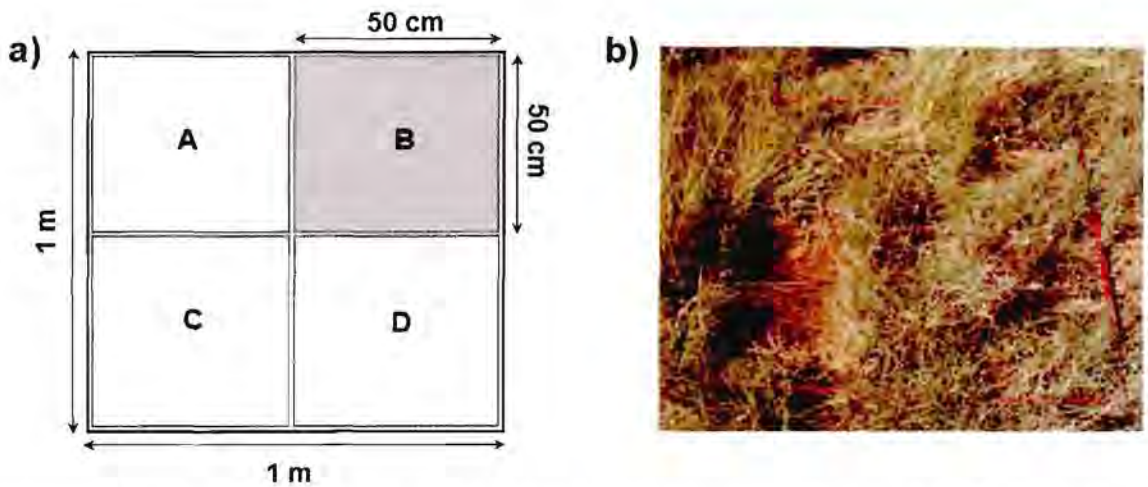


Figure 6.2: The soil sampling method. A 1 x 1 m square quadrat was subdivided into four 50 x 50 cm areas A-D (a). A photograph of the quadrat with some of the 50 x 50 cm area removed (b).

The seedling emergence method was used for the soil seed bank analysis (Ter Heerdt *et al.*, 1999). In this method soil samples are spread in trays in a greenhouse and kept under optimal conditions to promote the germination of as many species and individuals as possible (Ter Heerdt *et al.*, 1996). The collected soil samples for the seed bank analysis were sorted to remove any unwanted organic material. Figure 6.3 shows the experimental layout and the dimensions of the trays.

The bottom third of each tray was filled with perlite or vermiculite. The next layer was filled with sterile Hygromix© growth medium (Hygrotech, 2008) to supply nutrients to the germinating seeds. A part of each of the sorted soil samples were then added on top in a 3 cm thick layer. Each tray was divided for 2 soil samples per plot (A + B) or (C + D) as shown in Figure 6.3 (b). In total, for the 45 sample plots there were 90 trays in the glasshouse.

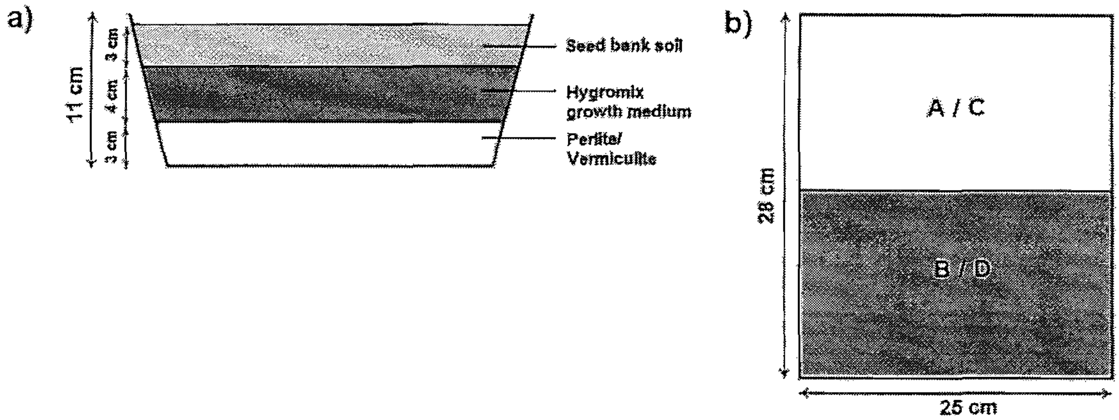


Figure 6.3: The experimental layout and the dimensions of the trays used for the germination of the seed bank samples. Each tray contained two soil samples of the same sample plot labelled A, B, C or D (b).

The trays were randomly placed on the tables in the glasshouse (Figure 6.4 a) and 5 control trays were placed on each table filled with vermiculite and Hygromix©. The glasshouse temperature was set to 26 °C during the day and 20 °C during the night. Fluorescent lights control the day length; the day length was set as 05:00 to 20:00. The trays were manually watered daily. The trays were randomly moved once a week to minimize the effect of local conditions within the glass house (Hahs, 2006). This was very important in the current study, since the one side of the glasshouse received direct sunlight for the majority of the day. Soil samples in the trays located on this table dried faster than in the rest of the glasshouse.

The glasshouse soil seed bank analysis was done from June to September 2008. The trays were monitored twice a week to see if any plants germinated. The moment a plant germinated it was marked by inserting a toothpick next to the plant (Figure 6.4 b). The toothpicks were coded to identify the species as a tree (two black rings), grass (one black ring) and forb (unmarked toothpick). The toothpicks allowed a quick look at the tray to identify any new seedlings and to record any seedlings which might have died between counting periods. Each seedling of a different species was photographed and given a number and its location in the tray was noted by using a makeshift ruler (Figure 6.4 b). Photographs were subsequently taken each week of the different numbered species. Their development could then be monitored. This allows quick identifications of similar species at early developmental stages. All the individuals recorded in the trays could then be identified and counted from an early developmental stage.

a)

b)

Figure 6.4: The layout of the glasshouse soil seed bank analysis. Samples were randomly placed on the tables (a). Example of the toothpicks used in the identification and counting of seedlings in each tray, the makeshift ruler (red) was used to record the location of the numbered and photographed species (b).

Figure 6.5 shows the photograph sequence of two species, *Hibiscus trionum* (a) monitored over a 46 day period, and *Pseudognaphalium luteo-album* (b) monitored over a 54 day period. The moment a plant could be identified it was removed from the trays to minimize the effects of competition (Hahs, 2006).

Because each species was numbered all the individuals with that code could then also be removed without waiting for the full development of all the seedlings. The photographs of each species allowed identification of the individual seedlings at a very early stage. The photographs of each species recorded in the soil seed bank will be of great assistance to future seed bank studies in the region.

a)

b)

Figure 6.5: Photograph sequence of two of the recorded species monitored during their development in the glasshouse. *Hibiscus trionum*, monitored over a 46 day period (a), *Pseudognaphalium luteo-album*, monitored over a 54 day period (b).

The species which could not be identified were re-potted into additional trays and allowed to grow to maturity. Some of the species were only identified to genus level and one geophyte species did not produce any distinguishing features during the germination period.

6.2.2.1 Data analysis

As with existing vegetation, the species were grouped into general categories namely: all species, indigenous, exotic, annual, perennial, dicot, grasses, monocots (excluding grasses) and geophytes.

The number of individuals counted for each species in the sampling plots (lumped for A + B and C + D, as indicated in Figure 6.3) were used as input for NMDS ordinations. The ordination was done in the same way as described previously in the data analysis discussion of the existing vegetation surveys.

6.2.3 Comparison of the soil seed bank and the existing vegetation

The species composition of the existing vegetation and the soil seed bank was compared with the use of the different NMDS ordinations created for the different species groupings. Following Hahs (2006) the Sørenson's Index of Similarity (Magurran, 1988) was computed for each general category, namely: all species, indigenous, exotic, annual, perennial, dicot and grass species per grassland patch. The geophyte and monocot (excluding grasses) grouping were not compared because neither of them had strong representation in the existing vegetation or the soil seed bank (Table 6.3).

The Sørenson similarity coefficient is a very simplistic measurement and is one of the most useful measures from a vast range of existing similarity indices (Magurran, 1988). The Sørenson (Cs) index is calculated with the following formula:

$$Cs = \frac{2j}{(a+b)} * 100 \qquad (11)$$

where (j) = the number of species occurring in both the soil seed bank and the existing vegetation, (a) = the number of species in the soil seed bank, with (b) = the number of species in the existing vegetation. The measure was modified by multiplying it with 100 to calculate the values as a percentage as done by Hahs (2006). The nearer the value is to 100 the more similar is the composition of the species and existing vegetation of the grassland patch in question.

The soil seed bank and the existing vegetation were also compared with regards to the total species richness per patch as recorded for the species and the existing vegetation per patch. For this purpose surface interpolation rasters were created in ArcMap of ArcView 9.2 (ESRI, 2006a) using the Spatial Analyst Extention tools. Surface interpolation rasters is commonly used in geographical applications to generate maps illustrating aspects such as topography, rainfall or temperature gradients. The IDW

(Inverse Distance Weighted) interpolate tool was used to create the rasters. IDW is a method of interpolation that estimates cell values by averaging the values of sample data points in the neighbourhood of each processing cell. The nearer a point (input data) is to the centre of the cell being estimated, the higher the influence or weight the point has in the averaging process (ESRI, 2006c).

The input data used to create the surface rasters was the patch centroid layer. This is the same point layer as created to generate the buffers in Chapter 5. In the attribute table of the point layer the species richness of the respective patches were added, grouped according to the general categories of all species, indigenous, exotic, annual, perennial, dicot and grasses. The geophyte and monocots (excluding the grasses) were left out for the same reasons as discussed earlier. The IDW tool generates a raster with the exact extent of the input point layer which is why some of the points are located on the extreme edge of the output raster image. Figure 6.6 illustrates the generated raster for the total species richness recorded per patch of the existing vegetation.

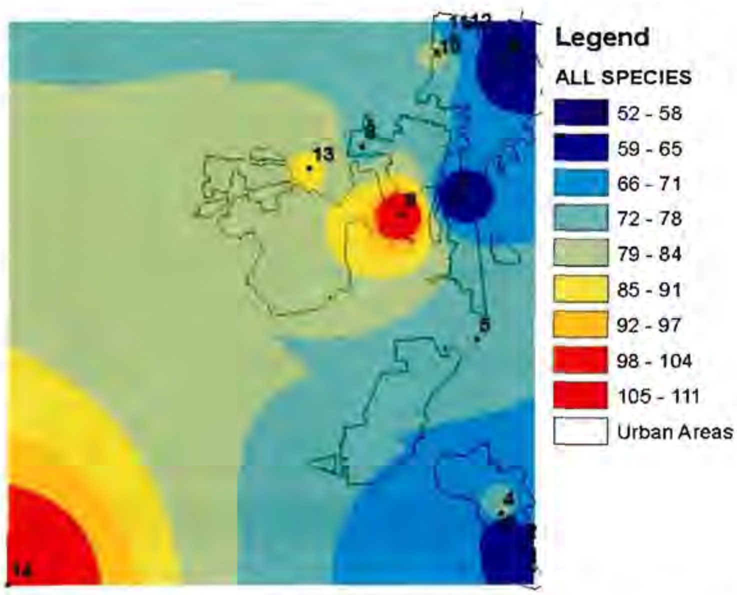


Figure 6.6: An example of a created IDW interpolate image. The figure illustrates the total species richness per patch as recorded in the existing vegetation of the current study.

The image is created using gradients of the input data (see the legend in Figure 6.6) and therefore only the values immediately surrounding the patch point are the corresponding species richness of that patch. For example, the species richness of patch 14 illustrated in the 105-111 gradient, is 108. Note that the surfaces between the patches should not be taken as a correct representation of the real situation as the values are interpolated from the patches. However, it is informative to see the contributions of each patch towards the overall picture giving information on potential patterns present in the entire landscape. One can clearly see by the colour of the areas, such as the blue gradient in Figure 6.6, that certain areas have similar species richness thereby potentially indicating similar patch attributes or landscape influences. In Figure 6.6 it is clear that some of the urban areas have similar species richness patterns.

The use of interpolation rasters allows spatially accurate comparisons of species distributions across the landscape. The visualization of the specific spatial locations in accordance with sampled species data of the patch could allow quick identifications of possible influences affecting vegetation at that specific point in the landscape only, allowing identifications of anomalies or point sources of pollution or disturbances. Road networks, soil classifications or any other data sources available for the landscape could be overlaid on the rasters to identify the possible effects driving the species distributional patterns and therefore ascribing possible mechanisms and processes that cause the observed species patterns.

6.3 Results and discussion

6.3.1 Existing vegetation

A total of 228 species were recorded for the existing vegetation of which 201 were indigenous. Only 27 exotic species were recorded. As found in the study of Hahs (2006) the largest patches (>25 ha) had the highest plant diversity, which could be due to the fact that more sample plots were surveyed in these patches (Hahs, 2006). The table with the specific species richness for the existing vegetation surveyed as recorded for each patch is given in Appendix D.

During the vegetation surveys the human management practices were also noted for each patch. This assisted in explaining possible influences experienced by the patch which could potentially determine the patterns and distributions of the vegetation communities of the patch. Table 6.2 lists the management practices for each patch as noted during the reconnaissance and vegetation surveys.

Cilliers *et al.* (1999) stated that over-management of vegetation due to the long term human influences in urban environments; especially frequent mowing as observed for some of the patches in the current study, keeps the grasslands in a sub-climax condition, decreasing the species diversity of the plant communities. However, for the selected patches in the current study no definite indication could be found that the management activities significantly altered observed species composition patterns in the respective affected patches.

Table 6.2: The management activities occurring within each fragmented grassland patch in the greater Klersdorp area.

Patch Nr	Management activities
1	*Regularly mown (3 times per year)*
2	*Regularly mown (3 times per year)*
3	*Regularly mown (3 times per year)*
4	*Parts of the patch regularly mown*
5	*None*
6	*None*
7	*Regularly mown (3 times per year)*
8	*Regularly mown (3 times per year)*
9	*Regularly mown (3 times per year)*
10	*Regularly mown (3 times per year)*
11	*Regularly mown (3 times per year)*
12	*Regularly mown (3 times per year)*
13	*None*
14	*Cattle grazing, some areas of patch are slightly overgrazed*

The NMDS ordinations of all the species (Figure 6.7 a), the indigenous species (Figure 6.7 b), the grass species (Figure 6.7 c) and the perennial species (Figure 6.7 d) all revealed a similar pattern. The observable gradient indicates that the species composition alters due to differences in the intensity of disturbances on the respective patches, ranging from patch 13 and 14 at one end of the gradient through to patch 9 and 3 which grouped separately from the rest of the patches (grouped on the left hand side of the ordinations in Figure 6.7 (a-d) and Figure 6.8 a). The dicot species composition also revealed this pattern, but with a slightly higher stress value (Figure 6.8 a) indicating that the pattern is not that definite in comparison with the ordinations with lower stress values.

The most dominant species recorded in grassland patches were all indigenous perennial species namely: *Themeda triandra*, *Eragrostis chloromelas*, *Felicia muricata* subsp. *muricata*, *Gazania krebsiana* subsp. *serrulata*, *Hermannia depressa*, *Heteropogon contortus*, *Lippia scaberrima* and *Pollichia campestris*. Of these species only 10 individuals of some of the above mentioned species were recorded in patch 9. Therefore, patch 9 grouped separately due to the absence of the dominant species found in varying proportions in the other patches. Five of the dominant species were also absent in patch 3 which is why this patch also grouped separately with patch 9. The presence of respectively *Delosperma herbeum*, *Gomphrena celosioides*, *Schkuhria pinnata* and *Sporobolus africanus* in patch 9 and their absence from patch 3 is why they are not more closely associated. Furthermore, patch 9 also grouped separately due to the dominance of the grass *Urochloa mosambicensis* in the patch, a typical grass found in disturbed areas (Van Oudtshoorn, 2004).

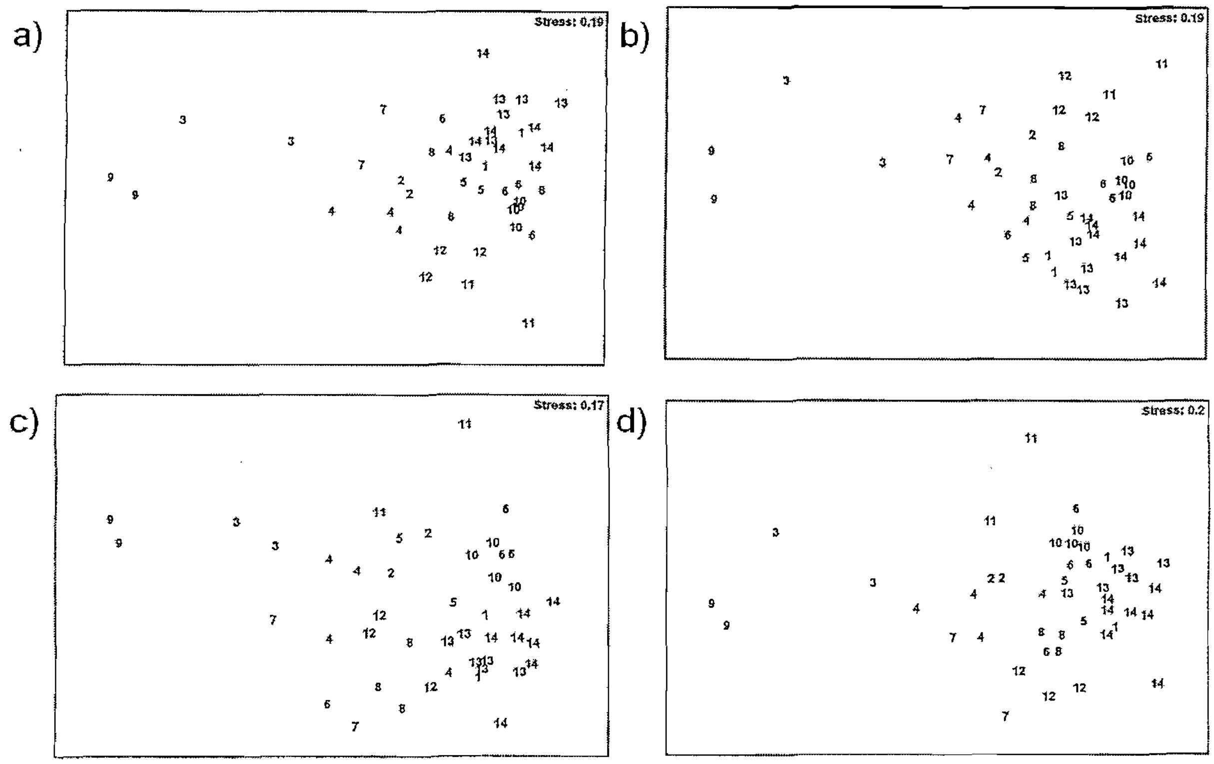


Figure 6.7: NMDS ordinations of all the species (a), the indigenous species (b), the grass species (c), and the perennial species composition (d) of the existing (extant) vegetation of the fragmented grassland patches in the greater Klerksdorp area. The points on the ordinations represent the sample plots, labelled according to the patch in which they were sampled.

The observed gradient (Figure 6.7) could therefore be ascribed to disturbances associated with urbanization in general as patch 9 experienced the most urbanization influences with decreasing influence until patches 13 and 14 are reached. The quantification of the observed urban-rural gradient will be applied to the vegetation data in Chapter 7. The absence of more distinct clusters may be due to the fact that the Klerksdorp urban areas are not very large and show only moderate urbanization influences in comparison with large urban agglomerates such as Melbourne in the study of Hahs (2006).

The ordination of the grass species had the lowest stress value for the sample plots with all the sample plots included in the ordination. This might be due to the fact that grasses have shorter life spans, than for example trees, and are known to respond to disturbance gradients (Van Outshoorn, 2004). The absence in patch 9 of the climax grass *Elionurus muticus* and the dominance of the increaser grass *Urochloa mosambicensis* clearly illustrate the previous statement. Therefore, the urbanization gradient could best be described by the respective grass species compositions of each patch.

The absence of the observed pattern in the previous ordinations in the exotic and annual species (Figure 6.8 (b) and (c)) might be because only 27 exotic species were recorded in the entire landscape area sampled. The highest species richness for exotics was recorded in patch 9 (16 species).

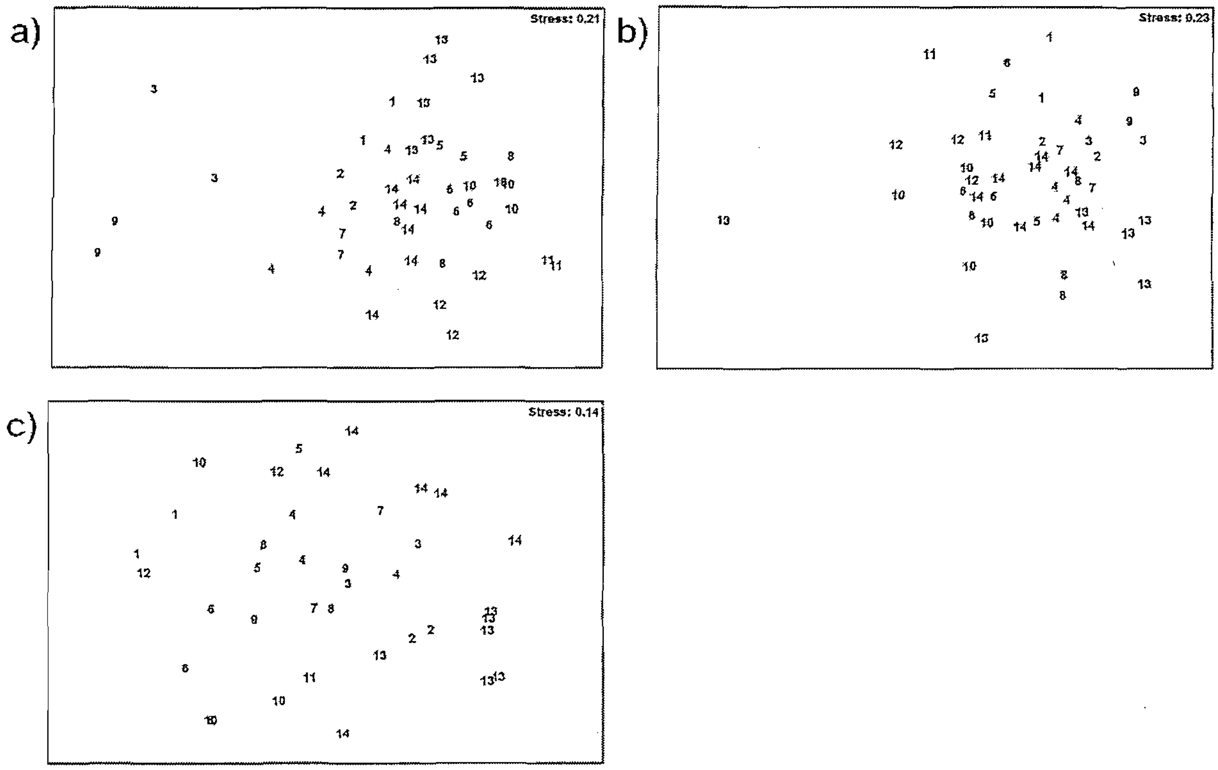


Figure 6.8: NMDS ordinations of the dicot species composition (a), the annual species (b), and the exotic species composition (c) of the existing vegetation of the fragmented grassland patches in the greater Klerksdorp area. The points on the ordinations represent the sample plots, labelled according to the patch in which they were sampled. The following sample plots were removed in the ordination of the exotic species, namely: 6.2, 4.3, 12.3, 10.1, and 11.2.

In four of the sample plots no exotic species were recorded and they were removed from the ordination. One of the sample plots in patch 11 had only one exotic species (*Euphorbia hirta*) which was shared with only one other patch therefore the sample plot formed an outlier and was removed to better observe the patterns in the ordination.

6.3.2 Soil seed bank

A total of 58 species were recorded for the soil seed bank of which 38 were indigenous and 20 were exotic (Table 6.3). 1106 seedlings were counted for all the patches in the seed bank analysis (Table 6.3). However, 9 of the sample plots had 6 individuals or less counted for the entire duration of the analysis. A list of species for the soil seed bank composition as recorded for each patch is given in Appendix D. The results of Hahs (2006) for seedling emergence in grassy woodlands indicated that the majority of individuals were exotic (51.2 %) and monocots (75.4 %), however in our study the majority of the individuals were indigenous (64.6 %) and dicots (78 %) (Table 6.3). The observed pattern in the current study is similar to the findings of Hill and French (2003). In the study of Hahs (2006) 55 exotic species were recorded in the soil seed bank in comparison to the 20 exotic species recorded in the soil seed bank of the current study.

Hahs (2006) applied heat and smoke treatments to the samples in the study in Melbourne; however no treatments were applied in the current study. This might have had an effect on the germination of some of the species present in the seed bank. Christoffoleti and Caetano (1998) also discussed the potential influence of seed dormancy on the results of the emergence method. The area sampled in the sample plot might also have had fewer amounts of viable seeds in the seed bank than in another part of the sample plot.

Table 6.3: The soil seed bank composition of the fragmented grassland patches of the greater Klerksdorp area. The total number of species and the number of seedlings recorded in the soil seed bank.

	Number of Species (%)	Number of Seedlings (%)
Total	*58*	*1106*
Indigenous	*38 (65.5 %)*	*714 (64.6 %)*
Exotic	*20 (34.5 %)*	*392 (35.4 %)*
Annual	*24 (41.4 %)*	*387 (35 %)*
Perennial	*34 (58.6 %)*	*719 (65 %)*
Dicot	*45 (77.6 %)*	*862 (78 %)*
Grasses	*9 (15.5 %)*	*194 (17.5 %)*
Monocot (excluding grasses)	*1 (1.7 %)*	*1 (0.09 %)*
Geophytes	*3 (5.2 %)*	*49 (4.4 %)*

The seedlings that emerged first were typical pioneer species occurring after disturbances in the landscape, with few indigenous forbs germinating. The germination period was monitored for only four months with no pre-treatment of the soil samples, which might have been responsible for the result of the study. Morgan (1998), in a study in Australia monitored untreated seedling emergence for 17 months as an effective method to assess both the transient and persistent components of the seed bank. However, Morgan (1998) reminded that the technique still emphasised the readily-germinable seed content rather than species with dormancies that may not have been broken under the experimental regime imposed.

The most dominant species occurring in the soil seed bank was *Conyza bonariensis*, *Gamochaeta pennsylvanica*, *Verbena bonariensis*, *Conyza podocephala* and *Wahlenbergia undulata*. The first three species are exotic annuals and the last two are indigenous perennials. *Gamochaeta pennsylvanica* was one of the species found exclusively in the soil seed bank. None of the ordinations showed any distinguishable patterns. Interestingly, no distinct relationship existed between the species richness for the seed bank and the patch sizes. However, the ordinations of all the species, the indigenous and the perennial species showed clearer groupings of the sample plots of each grassland patch (Figure 6.9 a-c). This might be due to the fact that 65.5 % of all the species counted in the soil seed bank were indigenous with 58.6 % perennial species.

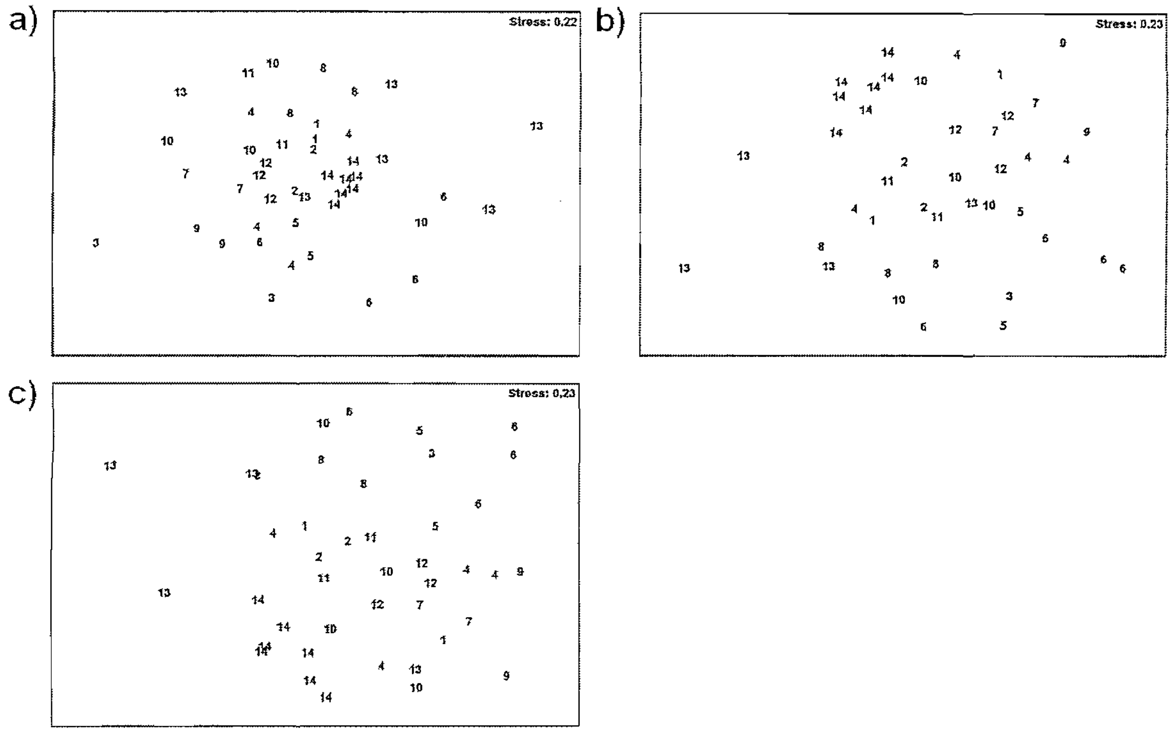


Figure 6.9: NMDS ordinations of all the species (a), the indigenous species (b), and the perennial species composition (c) in the soil seed bank of the fragmented grassland patches in the greater Klerksdorp area. The points on the ordinations represent the sample plots, labelled according to the patch in which they were sampled. The following sample plots were removed in the ordinations of the indigenous and perennial species, namely: 3.1, 13.4, and 13.6.

In all of the ordinations, except the ordination showing the groupings of all species (Figure 6.9 a), some sample plots were removed. Most of the plots removed had no species counted for the specific categories. Some plots were also removed due to the sole occurrence of a single species counted only in those plots, making them outliers. Plot 3.1 had a single individual of *Dactyloctenium aegyptium,* and plot 13.4 a single individual of *Hermannia depressa.*

The dicot, annual, and exotic (Figure 6.10 a - c) ordinations all showed sample plots grouped far from the rest. The separate grouping of these sample plots was as a result of the absence of some general species (counted in many of the sample plots) in some of these plots. Additionally, three species were counted in only four of the sample plots, namely: *Solanum nigrum* (annual, recorded in only three sample plots), *Physalis viscosa* (exotic, three individuals recorded in a single sample plot only) and *Taraxicum officinale* (exotic, recorded on only 3 sample plots).

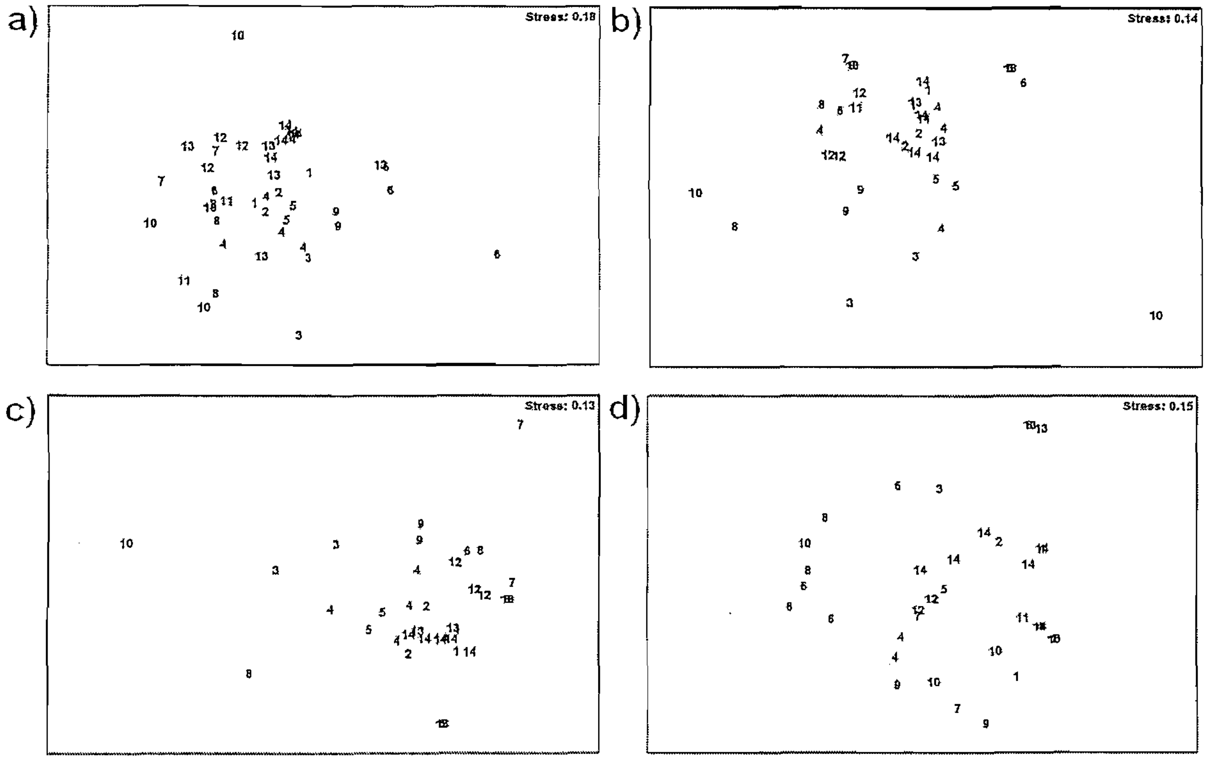


Figure 6.10: NMDS ordinations of the dicot species composition (a), the annual species (b), the exotic (c), and the grass species composition (d) in the soil seed bank of the fragmented grassland patches in the greater Klerksdorp area. The points on the ordinations represent the sample plots, labelled according to the patch in which they were sampled. The following sample plots were removed in the ordinations of the exotic and the annual species, namely: 11.2, 10.1, 6.3, and 13.5. Additionally, 10.1 was removed from the exotic and 7.1 from the annual ordination. 13.5 was also removed in the dicot ordination.

Overall, no patterns in terms of distinct gradients are observable in the ordinations for the soil seed bank. This might be due to the low total number of species recorded in the current study for the seed bank. The germination conditions and time period allowed for identification of seedlings might also not have been adequate. The most striking difference between the existing and the soil seed bank vegetation observed in the ordinations of the sample plots are the ordinations of their grass species composition, from a definite gradient in the existing vegetation too no observable gradient for the soil seed bank (Figure 6.10 d).

In a study by Morgan (1998) in Australia, the seedling emergence technique was used (seedlings were counted over a 17 month period) to determine the size and taxonomic composition of the soil seed bank of five *Themeda triandra* grasslands that had different fire histories. This was compared to the existing vegetation at each sampled site. No correlation between seed bank richness and fire history was found.

However, Morgan (1998) found that the richness of the seed bank was significantly lower than the vegetation at all spatial scales. 47 % of the recorded existing species were not detected in the seed bank. Therefore, as these species were almost entirely absent from the persistent seed bank, despite their abundance in the existing vegetation, Morgan (1998) suggested that it seemed that the conservation of many perennial native plants of species-rich grasslands were crucially dependent on the conditions that maintained the standing flora. He concluded that "*restoration management of degraded grasslands will be dependent on the reintroduction of propagules by direct seeding or planting rather than the potential exploitation of the soil seed bank*" (Morgan, 1998).

6.3.3 Comparison of the soil seed bank and the existing vegetation

6.3.3.1 Plant species composition

189 species were recorded in the existing vegetation that were absent in the soil seed bank. 173 of these were indigenous species and only 16 of them exotic. 19 species recorded in the soil seed bank did not occur in the existing vegetation. Of these 10 were indigenous and 9 exotic. *Gamochaeta pennsylvanica* (a widespread annual weed) was the most abundant in the soil seed bank with 135 individuals counted, but it was limited to patches 6, 2, 1, 4, 5, 14 and 13. Table 6.4 shows the differences in the vegetation composition of the existing and the soil seed bank vegetation. The percentages listed are the percentages of each category in relation to the total number of species in the existing and the soil seed bank respectively.

Table 6.4: A comparison of the composition of the existing vegetation and the soil seed bank of fragmented grasslands in the greater Klerksdorp area.

	Existing vegetation		Soil seed bank	
Indigenous species	201	88.2 %	38	65.5 %
Exotic species	27	11.8 %	20	34.5 %
Annual species	54	23.7 %	24	41.4 %
Perennial species	174	76.3 %	34	58.6 %
Dicot species	162	71.1 %	45	77.6 %
Grass species	42	18.4 %	9	15.5 %
Monocot species (excl. grasses)	4	1.8 %	1	1.7 %
Geophyte species	20	8.8 %	3	5.2 %
All species	228		58	

Figure 6.11 illustrates the relative percentage differences in the vegetation compositions shown in Table 6.4. This is supplied as a visual aid to quickly distinguish the compositional differences between the existing vegetation and the soil seed bank. The most apparent differences are the lower percentage of indigenous species and the higher percentage of exotic species in the soil seed bank in comparison to the

existing vegetation. The higher percentage pattern for the soil seed bank is repeated for the annual species and to a certain extent, the dicot species.

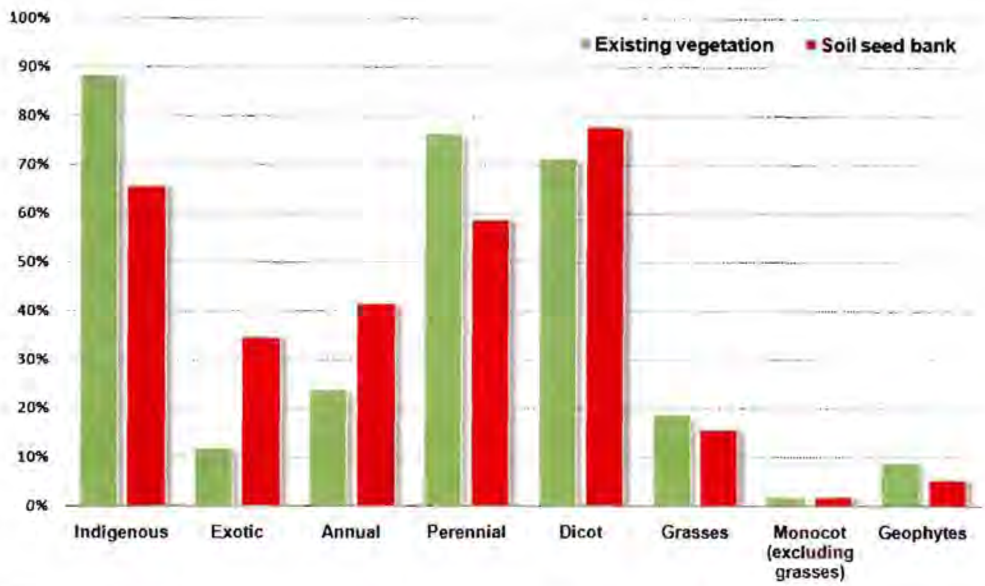


Figure 6.11: Comparison of the relative percentages of species representation of the soil seed bank and existing vegetation composition of fragmented grassland patches in the greater Klerksdorp area.

6.3.3.2 Species richness patterns

Table 6.5 indicates the Sørenson's index of similarity computed for each patch. No evident similarities occurred. The highest similarity occurred for the grasses of patch 9. Ten species were recorded in the existing vegetation, five in the soil seed bank and four of the species occurred in both.

The annual and the indigenous species had the lowest average similarities across all the patches (Table 6.5). In both the current study and the study of Hahs (2006) the similarity percentage for the indigenous species of all the patches was the lowest. This is on par with the finding of Morgan (1998) as discussed earlier. This is a cause for concern as the soil seed bank is the repository for future plant communities, especially where existing communities will experience future disturbances. This would influence the species with an absence of seed in the permanent soil seed bank. However, a more comprehensive seed bank study might be needed to obtain the true picture.

Table 6.5: Sørenson's Index of Similarity between the soil seed bank and the existing vegetation within each vegetation patch in the greater Klerksdorp area.

Patch	All species	Indigenous	Exotic	Annual	Perennial	Dicot	Grasses
1	*10.81*	*11.59*	*0.00*	*0.00*	*12.12*	*12.00*	*14.29*
2	*17.50*	*20.59*	*0.00*	*0.00*	*21.21*	*16.95*	*28.57*
3	*29.85*	*24.00*	*47.06*	*47.06*	*24.00*	*28.00*	*36.36*
4	*27.52*	*22.99*	*43.48*	*23.08*	*28.92*	*27.85*	*31.58*
5	*9.76*	*7.89*	*33.33*	*13.33*	*8.96*	*11.76*	*9.52*
6	*4.88*	*5.17*	*0.00*	*0.00*	*5.71*	*0.00*	*12.90*
7	*20.29*	*20.69*	*20.00*	*22.22*	*20.00*	*18.18*	*33.33*
8	*16.09*	*15.19*	*25.00*	*11.11*	*17.39*	*16.95*	*18.18*
9	*29.85*	*26.67*	*36.36*	*23.08*	*34.15*	*26.09*	*53.33*
10	*14.58*	*13.33*	*33.33*	*0.00*	*17.72*	*10.34*	*24.00*
11	*7.89*	*8.82*	*0.00*	*0.00*	*8.96*	*4.35*	*9.52*
12	*23.68*	*17.14*	*33.33*	*40.00*	*19.67*	*22.64*	*31.58*
13	*14.00*	*15.05*	*0.00*	*0.00*	*16.67*	*14.29*	*25.00*
14	*19.05*	*17.39*	*18.18*	*25.00*	*17.65*	*19.05*	*27.59*
Average	*17.55*	*16.18*	*20.72*	*14.63*	*18.08*	*16.32*	*25.41*
Total all patches	*27.27*	*23.43*	*46.81*	*28.21*	*26.92*	*29.13*	*27.45*

The following figures (Figures 6.12 - 6.15) show the IDW surface rasters created by the interpolation tool. The different patches are compared with regards to their total patch species richness. As discussed in the methods, the surfaces between the patches should not be taken as a correct representation of the real situation since the values are interpolated from the patches. However, it is informative to see the contributions of each patch towards the overall picture giving information on potential patterns present in the entire landscape. The blue parts of the colour gradient used to illustrate the species richness patterns represent the lower species diversity with the red part indicating higher species diversities. All the figures were created in this manner.

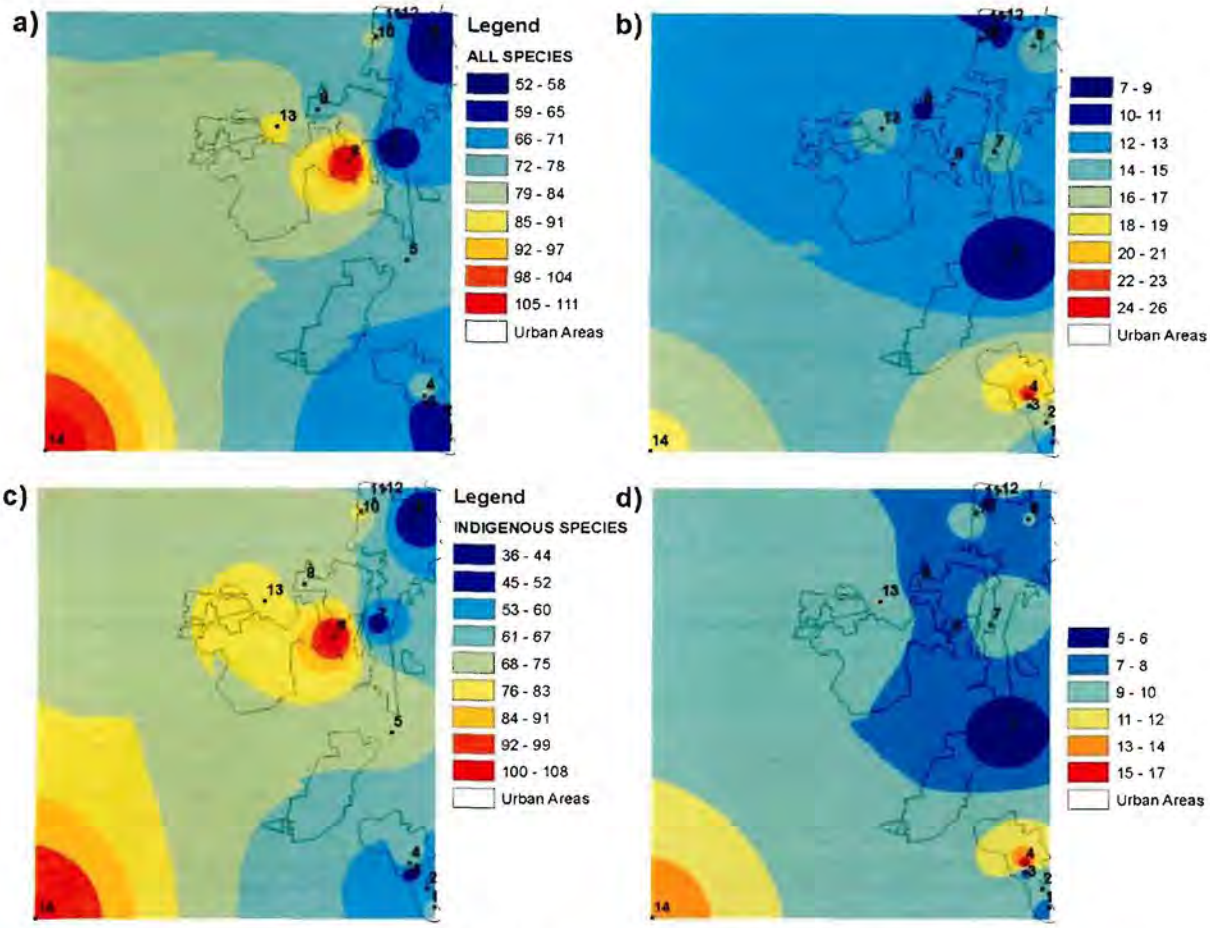


Figure 6.12: IDW surface rasters of the species richness of all the species recorded per patch for the existing vegetation (a), and the soil seed bank (b); and the indigenous vegetation, existing (c) and soil seed bank (d) in the fragmented grasslands of the greater Klerksdorp area.

The patterns for the total species (Figure 6.12 a), the indigenous species (Figure 6.12 c), the perennial species (Figure 6.13 a) and the dicot species (Figure 6.13 c) are very similar with regards to the existing vegetation. In these figures patch 14, 13 and 6 had the highest species diversity. These patches also had the largest patch size. The patches with sizes below four hectares all had the lowest species diversity (blue shading). However, in the soil seed bank the contribution of patch 6 and 13 for the total, indigenous, perennial and dicot species richness are substantially lower (Figure 6.12 (b) and (d); Figure 6.13 (b) and (d) respectively). Patch 7, patch 14 and patch 4 contributes here to the higher soil seed bank diversity, with patch 4 having the highest diversity for the soil seed bank vegetation.

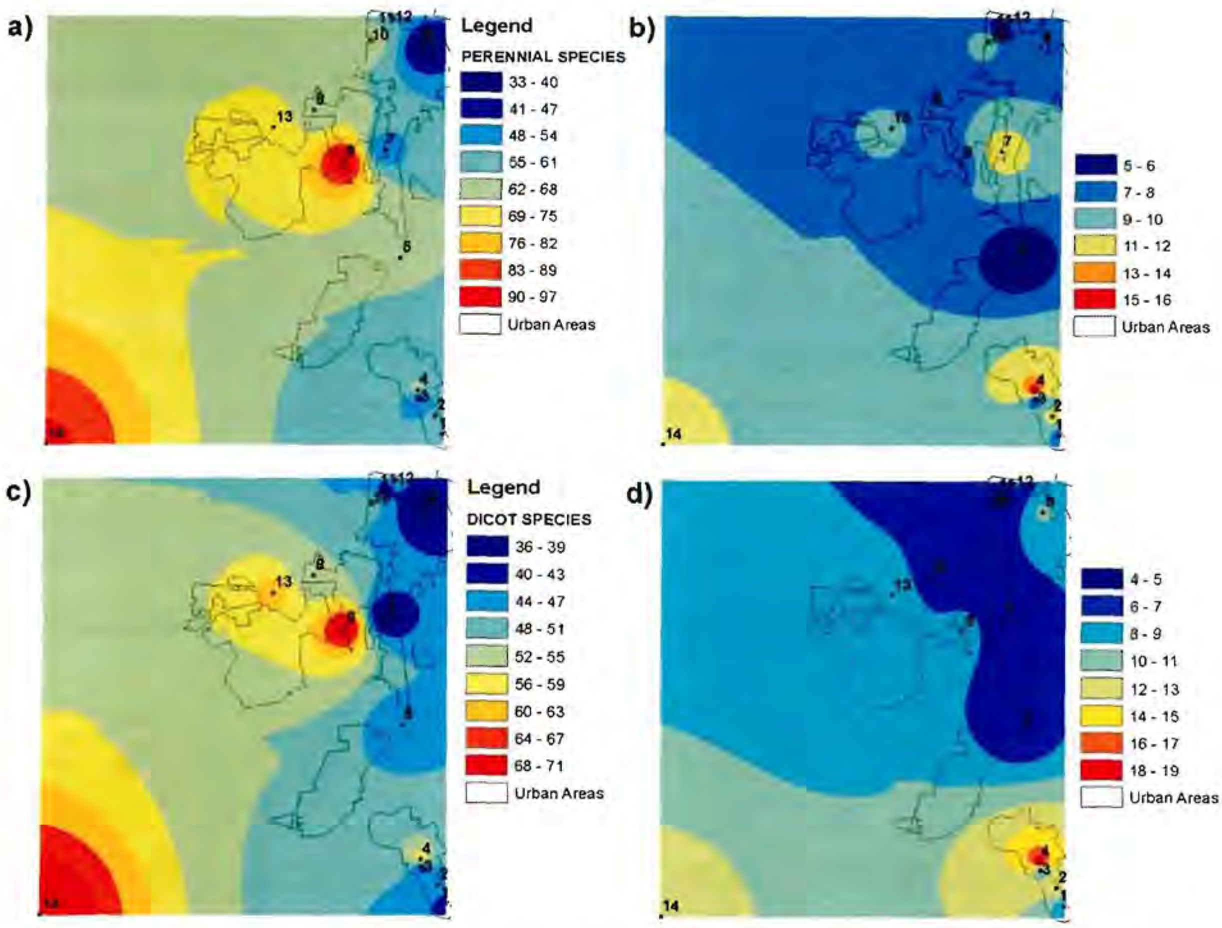


Figure 6.13: IDW surface rasters of the species richness of the perennial species recorded per patch for the existing vegetation (a), and the soil seed bank (b); and the dicot vegetation, existing (c) and soil seed bank (d) in the fragmented grasslands of the greater Klerksdorp area.

Comparison of the grass species richness of the existing vegetation (Figure 6.14) illustrates an immense decline in the species richness of patches 1- 4, 7, 9, 11-13. Patch 6 and 10 were large patches which could account for the high grass diversity still encountered here. However, patch 13 (the second largest patch) had low diversity reminding us of the complexity of urban landscapes and the danger of making generalities in urban areas. Bredenkamp *et al.* (1994) indicated that the presence of high numbers of the dwarf shrub *Ziziphus zeyheriana* indicated past disturbances which could explain the lower grass diversity in this patch.

An interesting situation is found in the soil seed bank where the most disturbed patch 9 and the large rural relatively undisturbed patch 14 had similar number of grass species in the soil seed bank. However, of extreme importance is the overall low occurrence of grass species in the soil seed bank (maximum of 5 species recorded), which could be ascribed to the possible general lack of seeds for these perennial species in the permanent soil seed bank. Van den Berg (2002) stated that indigenous native species rarely establish in disturbed grasslands because of often specific germination requirements with regards to environmental conditions, especially moisture, temperature and light. Additionally, Van den Berg (2002) affirmed that in South African grasslands the soil seed banks of degraded areas contribute minimally to successional processes. In the current study the maximum indigenous species richness recorded in a patch for the soil seed bank was 17 (patch 4) compared to 108 indigenous species recorded for the existing vegetation of patch 6 (Figure 6.12 (c) and (d)). Patch 4 had the highest species richness for all except the grass species richness of the soil seed bank.

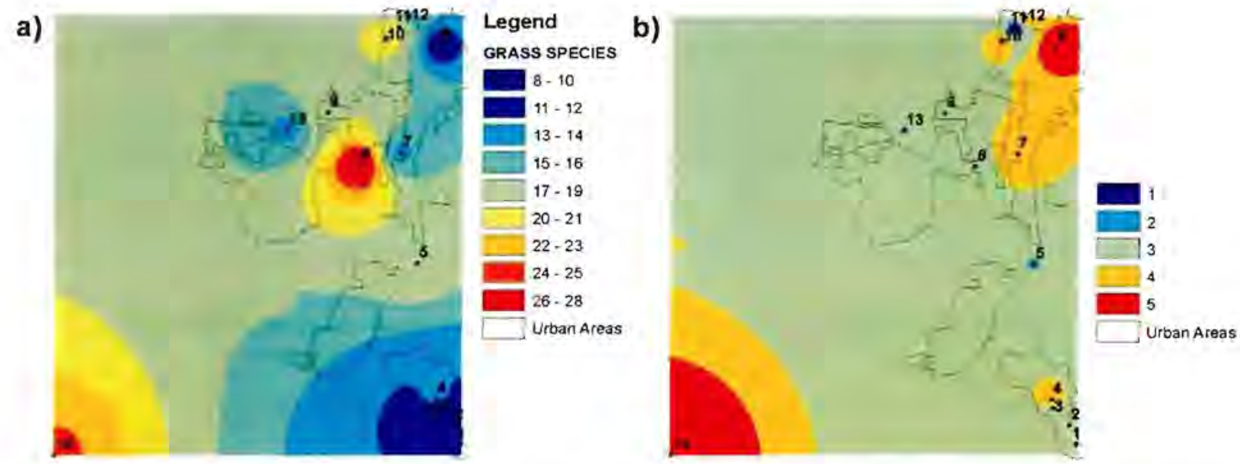


Figure 6.14: IDW surface rasters of the species richness of the grass species recorded per patch for the existing vegetation (a), and the soil seed bank (b) in the fragmented grasslands of the greater Klerksdorp area.

The 46.81 % similarity (Table 6.5) between all the patches for exotic species can be seen in Figure 6.15 (a-b). Note that the general pattern for both the existing and the soil seed bank are similar, with patch 9, 4 and 3 having the highest diversity. The lower number of exotic species in comparison to the indigenous species of the patch in all of the urban patches might be an indication of the low urbanization pressures on these patches as well as the higher competitiveness of the remaining existing indigenous vegetation in the respective patches. Even the most disturbed patches with the highest exotic species richness still had more indigenous species in both their existing vegetation and the soil seed bank (Figure 6.12 (c-d) and Figure 6.15 (a-b) or compare the existing and soil seed bank species richness tables in Appendix D).

In the study of Hahs and McDonnell (2007) no clear urbanization influences were found and the majority of indigenous vegetation species persisted in even the most disturbed remnant woodland patches, also serving as an indication of the resilience of indigenous plant species. However, in the current study the indigenous diversity (Figure 6.12 c) declined from 108 species in the most diverse patch (patch 14) to only 36 species in the least diverse patch (patch 9) in the existing vegetation, indicating the impact of urbanization related disturbances on fragmented grassland patches.

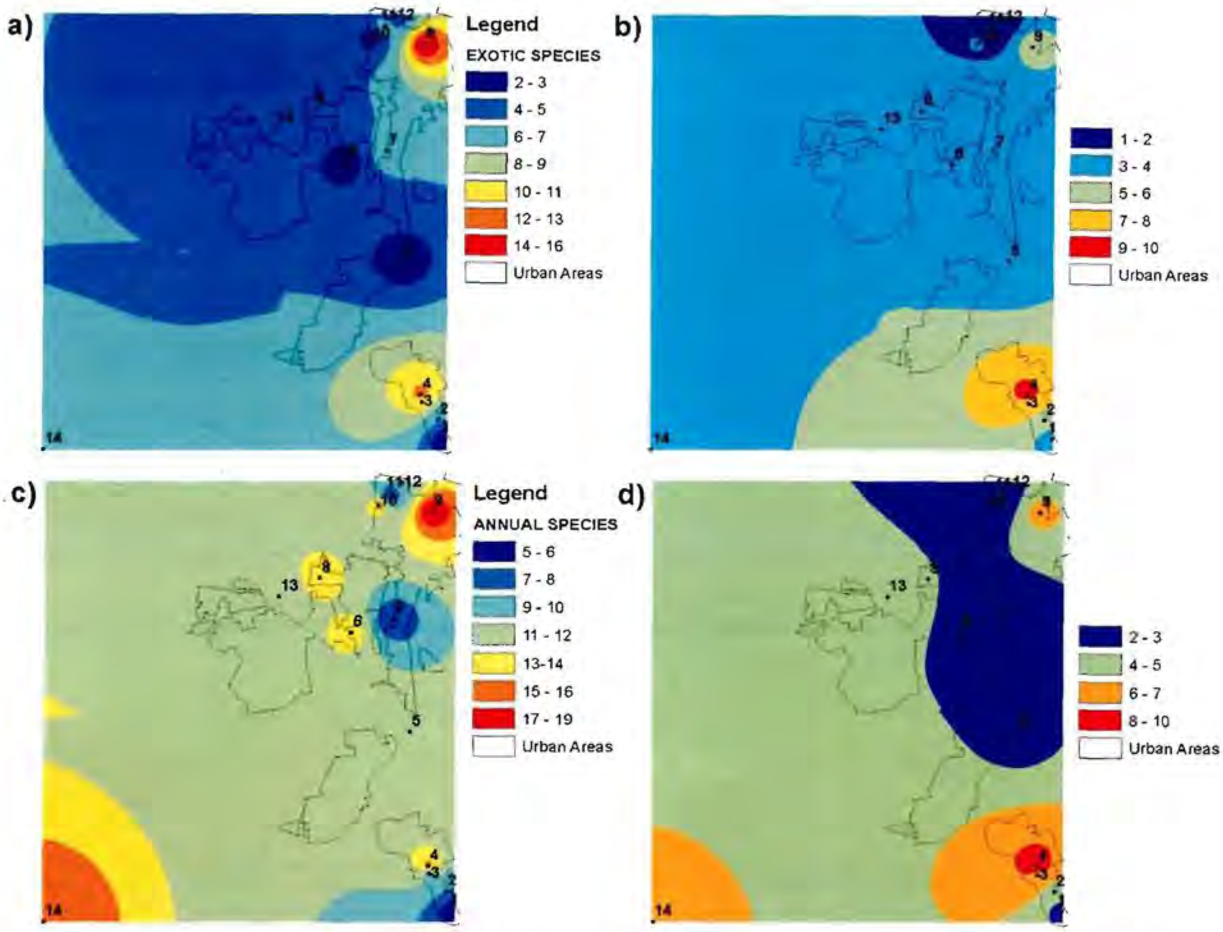


Figure 6.15: IDW surface rasters of the species richness of the exotic species recorded per patch for the existing vegetation (a), and the soil seed bank (b); and the annual vegetation, existing (c) and soil seed bank (d) in the fragmented grasslands of the greater Klerksdorp area.

No clear patterns were observed for the annual species composition (Figure 6.15 c-d). However, because most of the exotic species were annuals, patch 9 and patch 3 had higher species diversity for both the existing and the soil seed bank. By comparing the indigenous and the exotic species diversity patterns it can be seen that the general pattern of native plant diversity decreasing towards the urban core with subsequent increases in exotic species richness (Sukopp, 1998; Pickett *et al.*, 2001) is true for the greater Klerksdorp area as well.

6.4 Summary

Overall the low species richness of the exotic species for the existing vegetation (27) and the soil seed bank (20), indicate that the observed gradients of disturbances related to urbanization for the greater Klerksdorp area are not associated with the increase in exotic species, but overwhelmingly with the decrease of indigenous species towards the urban centre (from 111 to 36 for the most disturbed patch).

It is clear that urbanization influences the indigenous and especially the grass species richness the most. This was also clear in the NMDS ordinations observed for these species categories (Figure 6.7 (b) and (c) respectively). Due to the dominance of indigenous species in the existing vegetation the total species diversity show the same patterns. In the vegetation survey it was not apparent that the regular mowing of some of the patches had any distinct effects on plant species richness or the species composition. Future studies should include additional suitable grassland patches not sampled in the current study to see if the general patterns identified in this chapter hold for a more detailed look at the urban grassland patches. Surveying the entire patch and not only representative sample pots as suggested by Hahs (2006), could be informative and could confirm or negate the patterns observed for the sample plots.

A gradient showing differences in the vegetation composition of fragmented grasslands was identified in this chapter, especially in terms of the existing vegetation with emphasis on the indigenous and grass species compositions of the patches. However, it is difficult to determine to which extent urbanization is driving the species composition and plant species richness of these patches.

The following chapter will therefore examine how the urbanization measures correlate to the biological diversity described in this chapter. This will be done in an attempt to identify the processes, or indicators of the processes, driving patterns of plant diversity. The urban-rural gradient of the greater Klerksdorp area will be quantified and the characteristics of the patches found along the gradient will be described. Corry and Nassauer (2005) stated that several landscape ecology studies have demonstrated the applicability of landscape pattern indices in the characterization of landscapes, however there is a lack of evidence that pattern indices imply ecological processes. Therefore, the next chapter aims to answer both the applicability of the selected urbanization measures in the current study to indicate ecological processes and how well they correlate to these processes.

Chapter 7: The urban-rural gradient of the greater Klerksdorp area

7.1 Introduction

"The ultimate goal of landscape pattern analysis should be to achieve better explanations and predictions of ecological phenomena based on established relationships between pattern and process" (Li and Wu, 2004)

Grimm *et al.* (2008) described urban ecology as integrating the theory and methods of both natural and social sciences to study the patterns and processes of urban ecosystems. However, in a recent survey of the use of gradient analysis, McDonnell and Hahs (2008) found that 63 % of the 300 articles they reviewed focused on the response of organisms to urbanization. Only in recent years has research progressed to mechanistic urban ecology where it is becoming clear that understanding the relevant ecological processes operating in cities is essential to conservation practices.

Therefore, the selection of urbanization measures to quantify the urban-rural gradient should be based on both its potential in explaining the maximum amount variation present in the landscape, e.g. accurately quantifying the gradient (Hahs and McDonnell, 2006), as well as its biological or ecological interpretability (McDonnell and Hahs, 2008). Only when focusing on measures useful as ecological or biological indicators will the underlying drivers of species patterns and responses be elucidated. Olsen *et al.* (2007) emphasized that "*identifying ecological indicators is an important component in describing an ecological system, establishing potential metrics of change, and building an effective environmental monitoring system*".

The current emphasis in urban ecology is the integration of different studies to begin to develop an understanding of the patterns and processes operating in an 'ecology of cities' approach (Pickett *et al.*, 2001). A common measure of urbanization would facilitate integration between studies as well as contributing to the understanding of ecology of cities (McDonnell and Hahs, 2008). However, McDonnell and Hahs (2008) reminded that the trade-offs between broad and specific measures of urbanization relate to the resources required to calculate the measure and in their applicability to examine different ecological responses. Burton *et al.* (2009) and Pouyat *et al.* (2008) indicated that both broad and specific measures were significantly correlated to the biological data they analyzed. Ultimately the choice of measures will be dependant on the scale of analysis as well as the specific research question asked (McIntyre *et al.*, 2000).

This chapter will attempt to answer two of the listed objectives in Chapter 1, namely: (1) to use the urbanization measures identified in Melbourne (Hahs and McDonnell, 2006) to quantify the urbanization gradient of the greater Klerksdorp area, and (2) to use the vegetation and soil surveys to quantify the influence of human impacts on the ecology of grassland patches investigating aspects such as: plant species composition and diversity; and specific soil properties.

Therefore, the relationship between the calculated measures of urbanization and the vegetation composition, species richness and soil characteristics of the selected grassland patches will be explored. The most appropriate measures will be discussed and used to quantify the urban-rural gradient of the greater Klerksdorp area and its influence on vegetation community composition by describing three grassland patches located along the observed gradient.

7.2 Methods

7.2.1 Soil sample analysis

During the soil seed bank surveys, discussed in Chapter 5, soil samples for laboratory analysis were also collected. In each sample plot two soil samples were randomly collected and mixed to form one soil sample per sample plot. The soil samples were collected with a soil auger at a depth of 10 cm, the auger had a diameter of 7 cm. The samples were sent to the North-West University Eco-Analytica laboratory for analysis. The pH, electrical conductivity (EC) and the percentage base saturation (BASE SAT) of each sample were measured.

The percent base saturation was derived from calculation of the amount of exchangeable cations, and subsequently the cation exchange capacity, in a 1 M ammonium acetate solution at a fixed pH of 7. The displacement of the cations is effected by mass action (White, 1997). The percent base saturation is the ratio between the sum of exchangeable basic cations and the cation exchange capacity (Gobat *et al.*, 2004) The exchangeable cations reflect the nutritional status of the soil. The percent base saturation is a good measure of how much of the cation exchange capacity is being utilized to store plant nutrients (Thompson and Troeh, 1978). The higher the base saturation the more nutrients are available to the plant. In general, the higher the percent base saturation the higher the pH (Black, 1968; Thompson and Troeh, 1978) and the higher the soil fertility. Foth and Turk (1972) stated that the most garden and agricultural crops grew best with base saturations of 80 and more with a pH of 6 or greater. Additionally, they found that the most demanding tree species required a high base saturation and a high pH which illustrates the great importance of cation exchange relationships for plant growth.

The pH was measured in a 1:2.5 soil/water suspension extract on a mass basis. This is not the 'true' pH of the soil, but the most useful in soil-plant relations. It is an indication of the readily available cations, and therefore, an understanding of the short-term functional processes in the soil such as leaching of mobile cations (Gobat *et al.*, 2004)

The electrical conductivity (EC) was measured in a saturated extract of de-ionized water. The EC is an indication of the total dissolved solids in the extract and therefore the dissolved solids in the soil. The EC is a measure of the salinity of the soil, the higher the EC values the more saline the soil (White, 1997).

7.2.2 Correlation of soil properties, species richness and urbanization measures

A correlation matrix was used to describe the correlations between average and total species richness per patch with the urbanization measures and the measured soil properties. The correlations were done in STATISTICA 8.0 (StatSoft, Inc., 2008) with the Pearson r correlation coefficient. This coefficient determines the extent to which values of two variables are linearly related. Values range between -1.00 to 1.00 respectively indicating a perfect negative and a perfect positive relationship. Values of 0.00 indicate a lack of correlation (StatSoft, Inc., 2008). Strong correlations are indicated by $r \geq 0.8$ or $r \leq -0.8$, moderate correlations are indicated by $0.5 < r < 0.8$ or $-0.8 < r < -0.5$, and weak correlations exist for $-0.5 \leq r \leq 0.5$ (Devore and Peck, 1993). However, even though a significance test indicates the existence of a correlation between x and y, it does not signify a cause-and-effect relationship. Thus, even after a significant correlation between variables has been recognized, the cause of the correlation must be identified and whether the correlation is high enough to be of practical value (Brase and Brase, 1999). In the current study values of $r \geq 0.6$ or $r \leq -0.6$ were used as the minimum value for good correlations, indicating possible direct or indirect correlations to underlying processes driving species diversity.

It is important to note that due to the small sample size, significant correlations should not be interpreted as definitive but rather as indicating a potential trend (Dytham, 2001). Significant correlations should be supported with additional research incorporating more samples. This is a common problem in urban ecological studies, where there are mostly only limited study sites available. Therefore, Hahs (2006) proposed the use of Bayesian statistics as a means of overcoming small sample sizes in ecological research. However, this approach was not used in the current study.

Scatterplots were produced in STATISTICA 8.0 for some of the significant correlations computed in the correlation matrices to better describe the relationships and identify the potential influence of outliers on the linear regression line. The differences in the average and total species richness values for the respective patches were also tested to evaluate the effectiveness of the use of samples plot to describe vegetation composition in urban areas.

The non-metric multidimensional scaling (NMDS) ordinations for the indigenous vegetation composition of the existing and the soil seed bank, discussed in Chapter 5, were also used to evaluate the influence of the observed gradients on the vegetation patterns.

7.2.3 The location of three grassland patches along the urban-rural gradient

Three grassland patches were chosen, not to show idealized forms of a perfect gradient, but rather to describe the general patterns that are observable for the greater Klerksdorp area as one moves from the rural outskirts to the urban centre.

Each of the three grassland patches quantified along the urban-rural gradient was described according to their total species composition as recorded for each patch (summed for all the sample plots). To describe the actual species composition of each patch the overall number of individual plants recorded in the surveys were used. Therefore, for example, to calculate the proportion of indigenous species, the indigenous individuals were divided by the overall total individuals recorded for the patch and multiplied by 100 to obtain the percentage indigenous species per patch. This was done to obtain the indigenous and the exotic proportional composition of each patch for the existing and the soil seed bank vegetation. Additionally, the proportional life form composition of each patch was also determined in terms of its grass, geophyte, dicot and monocot (excluding grasses) composition. This was done in the same manner as described in the example given. However, take note that these patch estimations are only for the vegetation recorded in each sample plot of the respective patch and that the actual plant composition of the entire patch might be different.

7.3 Results and discussion

7.3.1 The influence of urbanization on vegetation along an urban-rural gradient

The complete report for the results of the soil laboratory analysis is supplied in Appendix E. Table 7.1 shows the correlations of the average species richness of the existing grassland patches with the urbanization measures, the soil properties and the patch area.

The pH and the percent base saturation (BASE SAT) were significant with correlations above $r = 0.6$ for all the species subsets except the average annual and dicot species richness. However, the exotic species had a positive correlation to the BASE SAT and pH with the rest of the significant correlations being negative. As discussed in the methods the pH and the BASE SAT generally increase relative to each other (Thompson and Troeh, 1978). In the current study, average (Table 7.1) and total (Table 7.3) exotic species richness of the existing vegetation increased in more nutrient rich soils, with an accompanying decrease in the indigenous species richness. Disturbance and change of soil properties and its effect on vegetation in urban areas and along urban to rural gradients have been well documented (Pouyat *et al.*, 2002; Bennett, 2003; Kaye *et al.*, 2006). Moreover, Sukopp (2004) stated that "*typically, disturbance of urban ecosystems leads to a decrease in the number of species native to the region and an increase of introduced non-native species.*" This was generally true for the current study, however too much reliance should not be placed on the correlations, as the sample size was limited.

Table 7.1: Correlation matrix of average species richness per patch for the existing vegetation (EXS) of different subsets of species with regards to the urbanization measures (Table 4.3, Chapter 4), patch area and the soil properties of the greater Klerksdorp area. Correlations highlighted in bold are significant at $p < 0.05$.

	EXS_all	EXS_indig	EXS_exotic	EXS_perenn	EXS_annual	EXS_dicot	EXS_grass
pH	***-0.62***	***-0.71***	***0.75***	***-0.66***	0.3	-0.47	***-0.74***
EC	-0.11	-0.13	0.16	-0.23	0.43	-0.01	-0.05
BASE SAT	***-0.69***	***-0.76***	***0.73***	***-0.72***	0.26	-0.5	***-0.82***
PATCH SIZE	0.3	0.34	-0.34	0.37	-0.3	0.28	0.2
DENSPEOP	-0.43	***-0.56***	***0.73***	***-0.6***	***0.67***	-0.46	-0.39
DENSDWEL	-0.44	***-0.56***	***0.73***	***-0.62***	***0.7***	-0.5	-0.36
INDEXCEN	-0.29	-0.37	0.47	-0.39	0.39	-0.29	-0.32
RND	-0.3	-0.39	0.52	-0.4	0.42	-0.22	-0.16
CBDkm	0.02	0.02	0.01	0.08	-0.2	0.25	-0.22
LCR	-0.11	-0.16	0.23	-0.21	0.35	-0.09	0.03
SIDI	0.31	0.32	-0.26	0.36	-0.25	0.33	0.17
PURBLC	-0.39	-0.47	0.53	-0.49	0.42	-0.5	-0.07
NP	-0.42	-0.47	0.48	-0.47	0.27	-0.34	-0.24
LSI	-0.42	-0.47	0.44	-0.47	0.27	-0.38	-0.23
LPI	0.23	0.2	-0.06	0.23	-0.05	0.31	0.12
MPFD	0.33	0.39	-0.44	0.38	-0.25	0.25	0.42

Both the density of people (DENSPEOP) and the density of dwellings (DENSDWEL) (highly correlated to each other, as illustrated in Table 4.9) show negative significant relationships to the existing indigenous and the existing perennial average species richness, with the correlation of perennial species being above $r \leq -0.6$ (Table 7.1). The exotic and annual average species richness showed a positive relationship to the two measures. This is similar to the correlations observed with the BASE SAT and the pH values. After this observation the correlation between the BASE SAT and DENSPEOP was tested to see whether DENSPEOP could be used as a proxy for both the measures, however no relationship exists between DENSPEOP and the BASE SAT (not shown). This reveals that the underlying processes driving urbanization influences, is probably not directly related to DENSPEOP with regards to urban soil properties as described for the current study. There was also no correlation between landscape shape index (LSI) and BASE SAT (not shown). None of the other measures revealed any significant correlations to the average species richness of any of the subsets in Table 7.1.

The average species richness for the existing grass species showed the highest individual correlation in Table 7.1 in its relationship to pH and BASE SAT. This possibly reflects the sensitivity of some of the indigenous grass species occurring in undisturbed natural areas to altered soil conditions and therefore to disturbance. However, this should be studied in more detail to prove any significant trends.

Figure 7.1 illustrates the correlations of the average species richness of the indigenous existing vegetation (Figure 7.1 a), the exotic existing vegetation (Figure 7.1 b), and the grass existing vegetation (Figure 7.1 c) with the BASE SAT. The scatterplots (Figure 7.1 a-c) indicate that no distinct outliers influence the correlation to the BASE SAT, emphasizing the strength of the correlations.

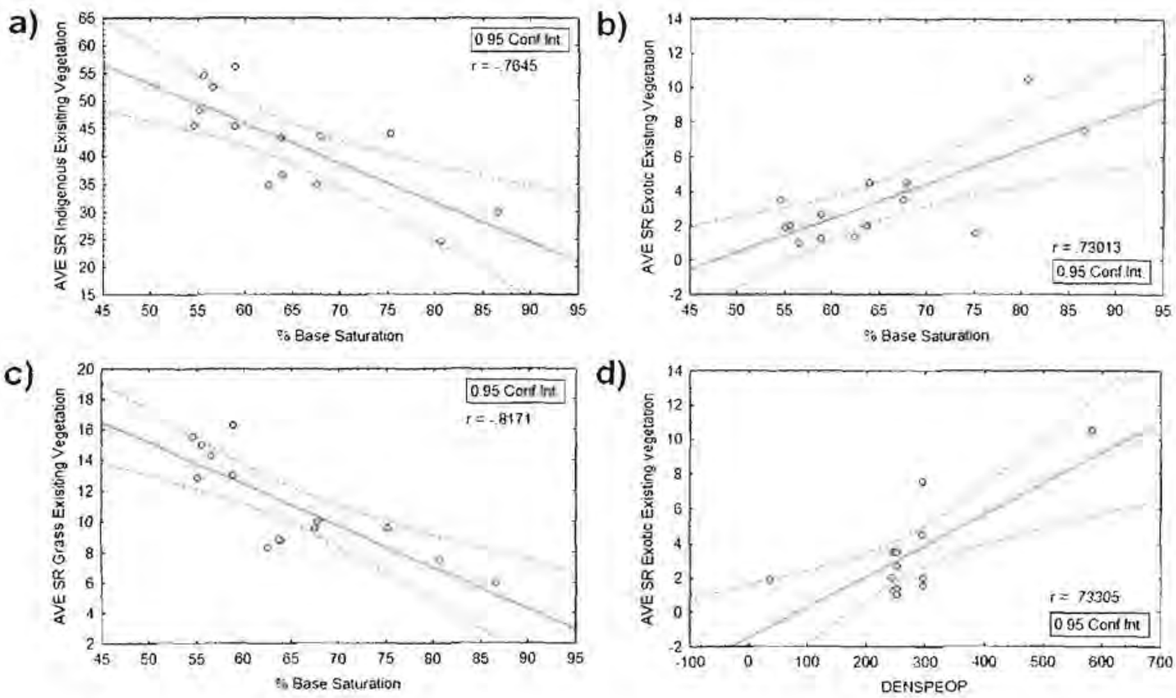


Figure 7.1: Scatterplots of the correlations of the average indigenous species richness (SR) vs. BASE SAT (a), average exotic SR vs. BASE SAT (b), average grass SR vs. BASE SAT (c), and the correlation of the average exotic SR vs. DENSPEOP (d). All of the correlations are the relationships observed for the existing vegetation. The red dashed line indicate the 95 % confidence interval computed for each graph and the solid red line the least-squares line indicating the strength of the correlation.

However, the correlation of the average exotic species richness of the existing vegetation to the DENSPEOP measure (Figure 7.1 d) indicates the influence of outliers. The census data accuracy problems described in Chapter 4 influence the correlations of DENSPEOP, DENSDWEL and index census (INDEXCEN) to species richness of the grassland patches. Figure 7.1 (d) clearly shows a clustering of data points in the middle of the plot with two outliers. The two outliers represent the values for the rural and the most disturbed patches respectively. An important consideration in the treatment of these two outliers is the fact that the DENSPEOP values for the patches do not represent the entire DENSPEOP gradient observed for the current study (this is discussed in detail in Chapter 5). Therefore, the non-normal distribution is largely due to the absence of values in between the two outliers and above the outlier with the highest DENSPEOP value. As a result the outliers were not removed and the correlation was treated as informative of a possible relationship between the numbers of exotic species to the population density, rather than a definite correlation. This opinion was held for all the correlations to the demographic variables (DENSPEOP, DENSDWEL and INDEXCEN).

Table 7.2 lists the correlations computed for the average species richness of the soil seed bank. The pH and BASE SAT again showed significant correlations to the average species richness of all species, exotic, annual, and dicot species per patch. However, all of the correlations to the seed bank subgroups were positive. Further research should be done on the possible significance of this. Only one urbanization measure revealed significant correlations to the average species richness of the soil seed bank. Mean patch fractal dimension (MPFD) showed negative relations to all the species subgroups of which the correlations to all the species, the exotics and the dicot species richness were significant with moderate correlations. However, Li and Wu (2004) warned that complex indices such as fractal dimension might be difficult to interpret biologically and should be avoided in correlation analysis.

Table 7.2: Correlation matrix of average species richness per patch for the soil seed bank (SSB) of different subsets of species with regards to the urbanization measures (Table 4.3, Chapter 4), patch area and the soil properties of the greater Klerksdorp area. Correlations highlighted in bold are significant at $p < 0.05$.

	SSB_all	SSB_indig	SSB_exotic	SSB_perenn	SSB_annual	SSB_dicot	SSB_grass
pH	***0.56***	0.26	***0.66***	0.26	***0.63***	0.51	0.35
EC	-0.04	-0.04	-0.03	-0.07	0	-0.1	-0.02
BASE SAT	***0.56***	0.26	***0.66***	0.22	***0.68***	***0.55***	0.34
PATCH SIZE	-0.28	-0.25	-0.2	-0.28	-0.11	-0.21	-0.07
DENSPEOP	0.41	0.2	0.49	0.18	0.47	0.39	0.34
DENSDWEL	0.35	0.18	0.42	0.14	0.43	0.32	0.38
INDEXCEN	0.29	0.09	0.39	0.17	0.28	0.3	0.02
RND	0.41	0.33	0.36	0.3	0.3	0.34	0.14
CBDkm	0.36	0.14	0.48	0.09	0.49	0.53	-0.06
LCR	0.3	0.32	0.16	0.32	0.06	0.18	0.17
SIDI	-0.15	-0.19	-0.04	-0.23	0.05	-0.03	-0.08
PURBLC	0.23	0.31	0.05	0.25	0.05	0.03	0.34
NP	0.39	0.38	0.28	0.33	0.24	0.29	0.17
LSI	0.38	0.38	0.24	0.35	0.18	0.24	0.19
LPI	-0.04	-0.13	0.1	-0.2	0.19	0.08	-0.03
MPFD	***-0.57***	-0.4	***-0.55***	-0.45	-0.4	***-0.59***	-0.14

The low correlations of the average and the total species richness (table not included as only one individual correlation occurred) of the soil seed bank to the urbanization measures might also be due to the fact that too few sites were sampled, or too few species were recorded in the soil seed bank. The fact that there were significant correlations with pH and BASE SAT could imply that the urbanization measures simply cannot be used as direct indicators of processes influencing soil seed bank species distributions.

The total species richness of the grassland patches was also correlated to the soil properties and urbanization measures to test whether different correlations exist in comparison to the average species richness values calculated for each species subset per patch. Table 7.3 lists the results for the computed correlations to the total species richness recorded for the existing vegetation. As stated previously, the correlations for the total species richness of the soil seed bank are not shown as only one significant correlation existed. This was the correlation of the dicot species richness to distance to central business district (CBDkm) (r = 0.60). The pH and the BASE SAT showed significant correlations to the total species richness as well, however the correlation to the total exotic species richness is distinctly lower than for the average species richness. This might indicate that the BASE SAT and the pH is more sample plot specific than patch specific and that intra patch dynamics also drives species richness patterns at sample plot scale. However, studies surveying the entire patch will be needed to prove this assumption.

A significant observation is the fact that the patch size correlates highly to the total species richness, the indigenous species richness, the perennial species richness and the dicot species richness. All of them had a strong positive correlation to the patch size. This partially confirms the statement of McDonnell (2007) that "*remnant patches of 'nature' in cities and towns are typically small in size contain a high diversity and abundance of non-indigenous species and a low number of indigenous species*" as being true for the current study with regards to the influence of patch size on the indigenous species richness. However, the exotic species richness showed no correlation to the patch size or any of the landscape metrics. This potentially indicates that the landscape structure as such do not influence the exotic species distributions, but rather that direct anthropogenic influences on the patch such as specific soil properties and underlying processes related to the population density drives exotic species richness. Increasing numbers of studies linked increases in soil fertility in urban areas with increases in exotic species (Gilfedder and Kirkpatrick, 1998; Lake and Leishman, 2004).

The correlations of the overall species richness for most of the species subsets per patch were significant for both the landscape metrics and the demographic variables. This is contradictory to the situations for the average species richness as shown in Table 7.1 and Table 7.2. This infers that the landscape structure and anthropogenic influences impacts the vegetation dynamics in the current study on a patch scale. The strong correlation of the patch size to all the landscape metrics (Table 7.3) also emphasizes this, which as a result points to the potential importance of the landscape context and configuration to the persistence of indigenous species in fragmented urban areas. However, this might only be an aspect of grasslands in this study as Hahs (2006) found that LSI did not appear to influence any aspect of the existing plant community in Melbourne.

Table 7.3: Correlation matrix of total species richness per patch for the existing vegetation (EXS) of different subsets of species with regards to the urbanization measures (Table 4.3, Chapter 4), patch area and the soil properties of the greater Klerksdorp area. Correlations highlighted in bold are significant at $p < 0.05$.

	EXS_ All	EXS_ Indig	EXS_ Exotic	EXS_ Perenn	EXS_ Annual	EXST_ Dicot	EXS_ Grass	PATCH SIZE
pH	***-0.58***	***-0.65***	***0.57***	***-0.61***	-0.06	-0.47	***-0.75***	*-0.34*
EC	-0.15	-0.16	0.10	-0.23	0.27	-0.11	-0.09	*-0.23*
BASE SAT	***-0.66***	***-0.71***	0.53	***-0.68***	-0.13	-0.52	***-0.83***	*-0.38*
PATCH SIZE	***0.79***	***0.74***	-0.13	***0.75***	0.41	***0.83***	0.51	*1.00*
DENSPEOP	***-0.62***	***-0.67***	***0.53***	***-0.70***	0.16	***-0.61***	***-0.57***	***-0.64***
DENSDWEL	***-0.59***	***-0.65***	***0.54***	***-0.69***	0.23	***-0.59***	-0.52	***-0.59***
INDEXCEN	***-0.64***	***-0.63***	0.24	***-0.64***	-0.20	***-0.62***	***-0.57***	***-0.85***
RND	-0.47	-0.53	0.49	-0.52	0.03	-0.45	-0.33	***-0.73***
CBDkm	0.20	0.16	0.11	0.21	0.03	0.30	-0.12	***0.45***
LCR	-0.52	-0.49	0.10	-0.51	-0.22	***-0.59***	-0.19	***-0.89***
SIDI	***0.68***	***0.64***	-0.09	***0.65***	0.35	***0.71***	0.41	***0.95***
PURBLC	***-0.57***	***-0.62***	0.48	***-0.61***	-0.02	***-0.63***	-0.25	***-0.73***
NP	***-0.57***	***-0.61***	0.47	***-0.58***	-0.12	***-0.53***	-0.40	***-0.76***
LSI	***-0.66***	***-0.68***	0.39	***-0.66***	-0.23	***-0.66***	-0.42	***-0.87***
LPI	***0.62***	***0.53***	0.15	***0.55***	0.51	***0.67***	0.35	***0.87***
MPFD	0.50	0.52	-0.33	0.51	0.12	0.43	***0.58***	***0.62***

Figure 7.2 shows the correlation of the total species richness of the indigenous vegetation to LSI and DENSPEOP. These two measures showed some of the best correlations to the species richness patterns observed for the patches.

However, although DENSDWEL and INDEXCEN show similar high correlations with the same subsets of species as DENSPEOP (Table 7.3), DENSPEOP was selected based on the ease of calculation and its high correlation to the other demographic measures as discussed in Chapter 4. Simpson's diversity index (SIDI), on the other hand, also showed the same high correlations as LSI, with the other signifcant correlations of the rest of the landscape measures having strong negative correlations to the species richness (Table 7.3). However, LSI is a less complex measure than SIDI and therefore more readily interpretable.

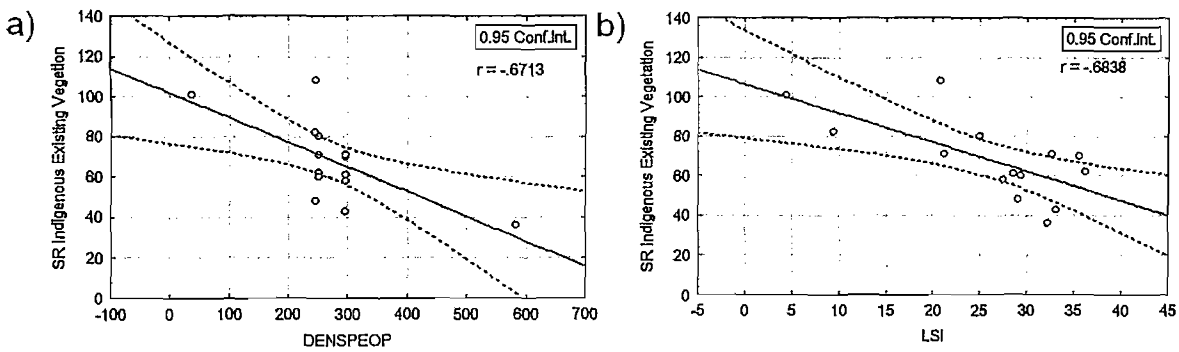


Figure 7.2: Scatterplots of the correlations of the total indigenous species richness (SR) with DENSPEOP (a), and LSI (b) for the greater Klerksdorp area. Both of the correlations are the relationships observed for the existing vegetation. The red dashed line indicate the 95 % confidence interval computed for each graph and the solid red line the least-squares line indicating the strength of the correlation.

The correlations of the total species richness per patch to the urbanization measures showed an overall increase in the number of significant correlations. This highlights the importance of the type of input data used to quantify the influence of urbanization on urban vegetation. Tavernia and Reed (*in press*) stressed the importance of the measures used to estimate the intensity of urbanization influences on vegetation. However, for the current study it was found that the type of vegetation input data also significantly influenced the biological correlations to the urbanization measures. This could have a serious impact on the ultimate selection of measures for a specific study. Therefore, the maximum number of biological input data available should be used to test the applicability of the measures for a specific study in the quantification of urban-rural gradients.

In contrast to the previous discussion, Li and Wu (2004) warned that the inadequate understanding of the scale-dependent relationship between pattern and process, sometimes indiscriminately ascribed to observed correlations of landscape metrics to biological patterns, contributes to confusion in landscape ecology. They emphasized that a common misuse of landscape indices is the quantification of an observed pattern without consideration of the underlying process causing the pattern. *"If landscape analysis is done only at a single scale that does not match the characteristic scale of the phenomenon of interest, the result may be a failure to detect pattern in a landscape even though pattern exists, or the "manufactured pattern" when none exists"* (Li and Wu, 2004). This relays to the importance of understanding the ecological interpretation of a measure as a direct or indirect indicator of a cause-effect relationship driving species diversity or as being ecologically irrelevant. Additionally, an understanding of the scale at which the biological pattern in question is occurring will also indicate which type of biological data to collect as well as at which scale the pattern will be optimally identified or quantified by urbanization measures.

To test whether the average and the total species richness values for the individual patches differ, their similarity was calculated by simply dividing the average patch species richness with the total species richness for the patch. The generated tables are listed in Appendix F. All the patches larger than four hectares irrespective if it were the existing or the seed bank's species richness had similarities of less than 50 % for the proportion of average to total species richness. In the smaller patches the size of all the sample plots surveyed per patch covered, on average, were more than half of the total patch area. This signified that the odds of not recording a species present in the patch were small. However, the total species diversity was also less which enhanced the chances of that specific species occurring in all the sample plots surveyed for the patch. The sample plots for the larger patches did not cover such a proportionally large area in the patch. Therefore, many of the species occurring in the patch might have been missed. The intra species distribution in larger patches might also not be that homogenous as in smaller patches due to more variability in the local patch conditions. Therefore the odds of recording the majority of the species occurring in the entire patch in a single sample plot becomes significantly lower than for smaller patches.

Hahs (2006) anticipated this in her study by stating that *"predictions of species richness were based on the sample plot area, rather than the entire remnant patch, and therefore are likely to under-represent the true species richness at the patch scale"*. The significance of this in the current study can be seen in the fact that the total species richness showed better correlations to the measures than the average species richness calculated per sample plot. Therefore, as suggested by Hahs (2006) future studies should, if possible sample the entire patch and test whether the correlations identified in the current study hold at the entire patch scale or is strengthened.

The correlations that proved significant for the specific species richness subsets were the same species subsets identified in Chapter 6 as displaying distinct vegetation composition patterns along a disturbance gradient, now identified as processes related to anthropogenic influences. To test whether the species compositions also grouped according to the observed gradients for the measures, the sample plots were symbolized according to their groupings along the gradients for DENSPEOP and LSI as described in Chapter 5 (Table 5.5). The indigenous vegetation composition was selected as its NMDS ordination shows some of the lowest stress values and it is well documented in previous studies that indigenous species diversity decline along an urban-rural gradient.

Figure 7.3 illustrates the groupings of the species composition of the indigenous vegetation symbolized according to the gradients of DENSPEOP (a) and LSI (b). A distinct gradient can be observed for both of the measures indicating the presence of an urbanization gradient in the greater Klerksdorp area. This proves for the current study that urbanization influences grassland species composition and that specific urbanization measures can be used to quantify the location of a vegetation patch along an urban-rural gradient. The red triangles in the figure indicate patches with higher urbanization influences; the yellow triangles indicate moderate urbanization, and the blue circles identify rural areas.

The fact that the moderate and higher urbanization plots occur together in Figure 7.3 (b) indicate that highly fragmented areas are not limited to the urban core and that the landscape structure is determined by more than just the direct processes of urbanization. Additional processes present in the greater Klerksdorp area include mining and agricultural practices. However, it should be emphasized that the urbanization related anthropogenic influences on the landscape in question is not that intense as the greater Klerksdorp area is relatively small, with the urban areas covering a surface area of only 78.22 km^2 (as determined by the digitizing of only the densely populated areas in the landscape, not including the formal city boundary).

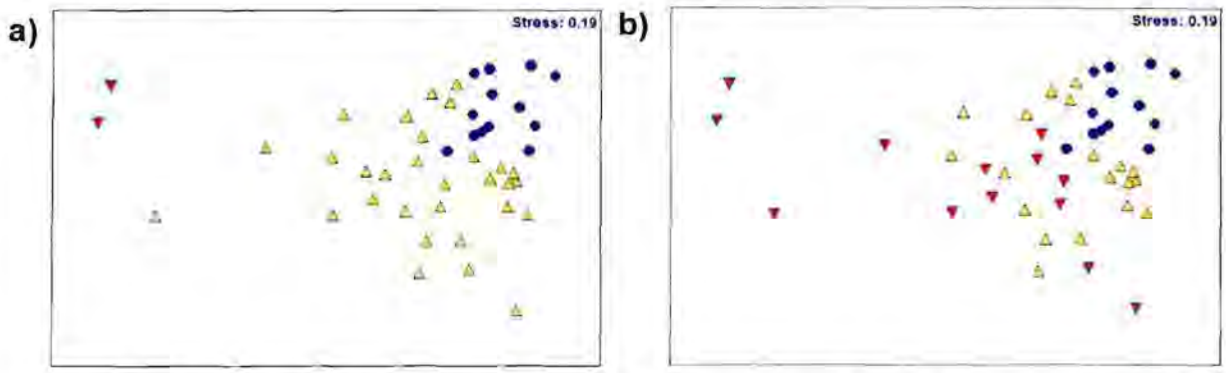


Figure 7.3: Gradient according to the vegetation composition for the NMDS ordination of the existing indigenous vegetation of the greater Klerksdorp area. The patch arrangements are shown for the DENSPEOP (a) and the LSI (b) gradients. Higher urbanization (red triangles), moderate urbanization (yellow triangles), and rural (blues circles).

By combining DENSPEOP, LSI, percent urban land cover (PURBLC) and the plant species composition information in Chapter 6, the indigenous species ordination were symbolized once more according to the observed urbanization gradient. In this instance the similarities of the species groupings are also considered as well as the overall groupings of the patches along the different gradients as described in Table 5.5 (Chapter 5). Patches 3 and 9 form the higher urbanization patches and patch 13 and 14 the rural patches with the rest grouped as moderately urbanized patches.

Figure 7.4 shows the gradient, according to the NMDS ordinations, for the indigenous species of the existing vegetation (a) and the soil seed bank (b). The plots area symbolized according to their groupings of higher urbanization (red triangles), moderate urbanization (yellow triangles), and rural areas (blue circles) as determined by the consideration of a combination of the observed gradients and the plant species composition. The plots of the soil seed bank were symbolized with the specific sample plots in the groupings as identified for the existing indigenous vegetation. A clear gradient is lacking in the soil seed bank data (Figure 7.4 b) which is strengthened by the few significant correlations to the urbanization measures as observed in Table 7.2. However, a gradient is still implied in comparison to the indigenous composition of the existing vegetation as the higher urbanized plots and the rural plots group on opposite sides of the ordination.

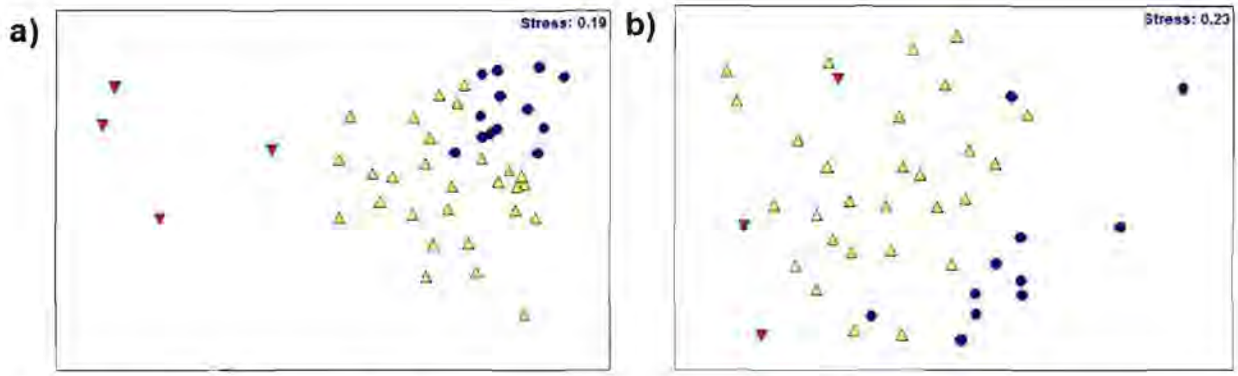


Figure 7.4: Gradient according to the vegetation composition for the NMDS ordinations of the existing indigenous vegetation (a) and the soil seed bank (b) of the greater Klerksdorp area. Patch 3 and 9 higher urbanization (red triangles), moderate urbanization (yellow triangles), and patches 13 and 14 rural (blues circles). The following sample plots were removed in the ordinations of the indigenous soil seed bank species, namely: 3.1, 13.4, and 13.6.

7.3.2 Quantification of the urban-rural gradient of the greater Klerksdorp area

The urban-rural gradient of the greater Klerksdorp area was defined using the measures of DENSPEOP, LSI and PURBLC as quantified for the patches in Chapter 5. All three of these measures showed significant correlations to the species richness values of the existing and in some instances the soil seed bank (Table 7.1 – 7.3). The patterns of vegetation composition as described in Chapter 6 also showed distributions along a gradient that can be ascribed to urbanization as shown in Figure 7.3 and 7.4 when the plots were symbolized according to their distributions as quantified in Chapter 5 for the individual gradients.

Figure 7.5 illustrates the observed urban-rural gradient for the selected measures of DENSPEOP (a), LSI (b), and PURBLC (c). The delineations of the urban areas in the landscape can best be described by the gradients of LSI and PURBLC. The values of the DENSPEOP gradient are constrained by the census data scale, as discussed in Chapter 4.

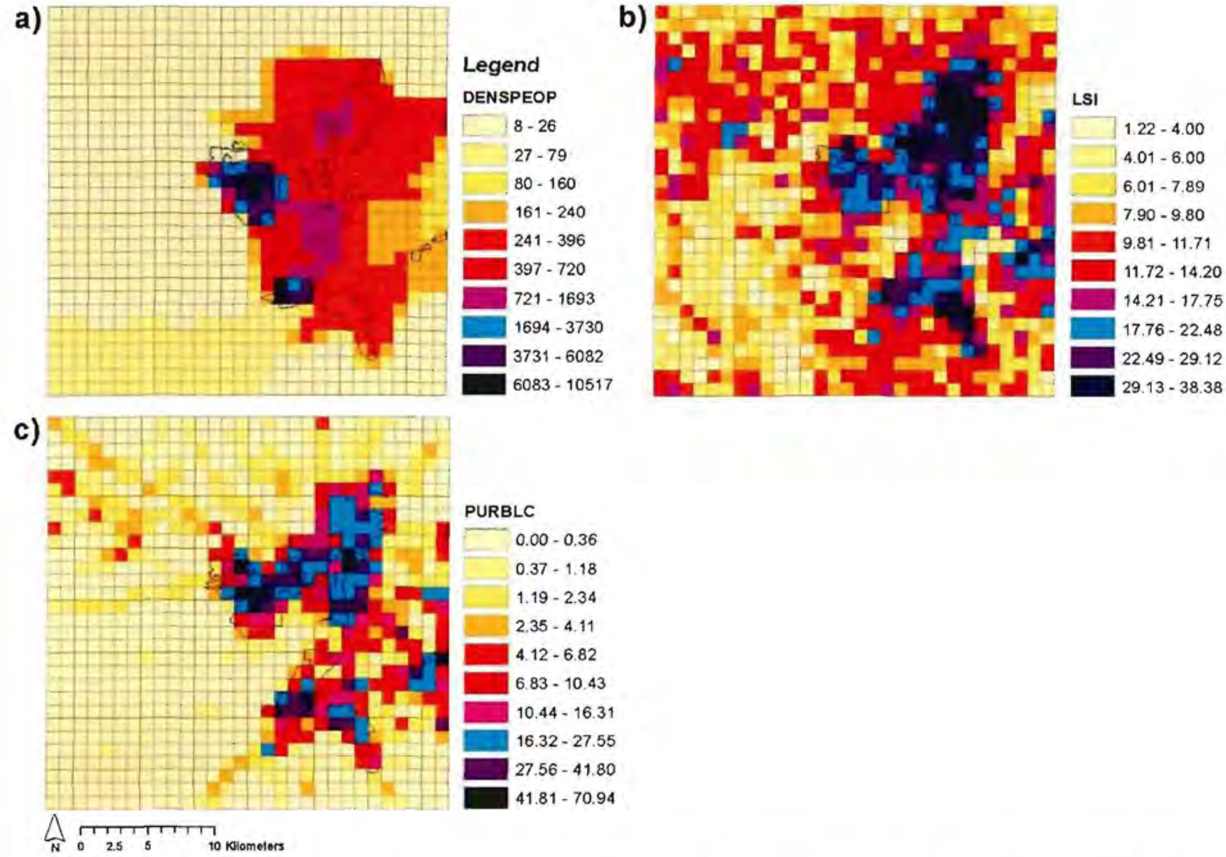


Figure 7.5: Landscape grids showing the distribution of the values for the observed gradients of DENSPEOP (a), LSI (b) and PURBLC (c) across the greater Klerksdorp area. The black outline indicates the urban areas in the landscape.

Figure 7.6 describes the gradient of the density of people (DENSPEOP) as it varies across the landscape. The grid cell shown in Figure 7.6 (b) indicates the cell with the median value for the gradient (not the mean value).

The values represented by each of the grids are listed in Table 7.4, also shown is the accompanying values of LSI and PURBLC for each respective grid cell to provide the maximum amount of detail for each grid cell and to evaluate each respective gradient in relation to the other measures. The urban areas had values ranging between 241 and 10517 people per km^2, with the rural areas having values between 8 and 240.

Figure 7.6: The range of values recorded for the DENSPEOP gradient across the landscape of the greater Klerksdorp area. The 1 km^2 grid cells are arranged from the lowest (a) to the highest value (c). The blue square indicate the delineation of the landscape grid cell in which it was measured.

The upper half of the range from the median to the highest value (5273 -10517) is represented by only 17 cells of the 900 grid cells in the landscape grid. These 17 cells are distributed between Jouberton and Kanana respectively (Figure 7.5 a). The mean value for the DENSPEOP values of the entire landscape is only 270.1. Therefore, to allow a more detailed description of the urban areas the DENSPEOP gradient were further subdivided into moderate urbanization (values 241-500) and higher urbanization areas (500-10517) to allow a more equitable distribution of the gradient.

Table 7.4: The values for the DENSPEOP gradient as observed for the entire landscape area. Also listed are the corresponding LSI and PURBLC values.

	Low (a)	Median (b)	High (c)
DENSPEOP	*8*	*5273*	*10517*
LSI	*5.23*	*23.69*	*19.76*
PURBLC	*0*	*20.59*	*70.94*

Figure 7.7 describes the gradient of the landscape shape index (LSI) as it varies across the landscape. The grid cell shown in Figure 7.7 (b) indicates the cell with the median value for the gradient. The values represented by each of the grids are listed in Table 7.5, with the accompanying values of DENSPEOP and PURBLC.

The values for LSI varied between 17.76 and 38.38 for the urban areas. However, some rural areas are included in that range as well. The rural areas showed distributions of values between 1.22 and 17.75. Interesting to note is the fact that the LSI describes a different aspect of the urban-rural gradient than DENSPEOP as indicated by Table 7.5 where the grid cell with the highest LSI had a DENSPEOP value of 338.

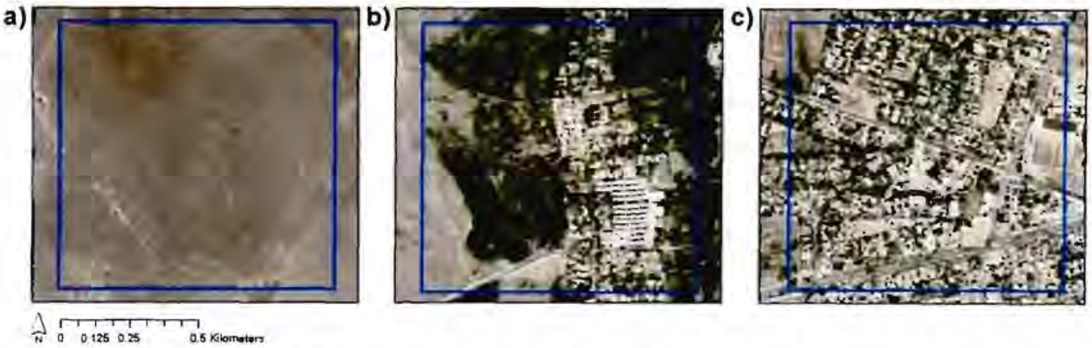


Figure 7.7: The range of values recorded for the LSI gradient across the landscape of the greater Klerksdorp area. The 1 km^2 grid cells are arranged from the lowest (a) to the highest value (c). The blue square indicate the delineation of the landscape grid cell in which it was measured.

The LSI is an index of the amount of irregularity of the shape of the landscape patches (Hahs and McDonnell, 2006). The LSI equals 1 when the landscape consists of a single square (raster) patch, and value increases without limit as the landscape shape becomes more irregular or as the length of edge within the landscape increases, or both (McGarigal and Marks, 1995). Therefore, it is a good indication of fragmentation in the landscape. Highly fragmented areas are characteristic of urban environments, hence the applicability of landscape metrics in quantification of the urban-rural gradient. Weng (2007) elaborated on this by declaring that calculating landscape metrics along an urbanization gradient can help to identify the fringes of urban expansion where the development pressure is most intense. The areas with the highest LSI values corresponded to the residential areas of Klerksdorp and Orkney (Figure 7.5 b).

Table 7.5: The values for the LSI gradient as observed for the entire landscape area. Also listed are the corresponding DENSPEOP and PURBLC values.

	Low (a)	Median (b)	High (c)
LSI	*1.22*	*19.16*	*38.38*
DENSPEOP	*36*	*246*	*338*
PURBLC	*0*	*11.44*	*34.23*

Figure 7.8 describes the gradient of the percent urban land cover (PURBLC) as it varies across the landscape. The grid cell shown in Figure 7.8 (b) indicates the cell with the median value for the gradient.

Figure 7.8: The range of values recorded for the PURBLC gradient across the landscape of the greater Klerksdorp area. The 1 km^2 grid cells are arranged from the lowest (a) to the highest value (c). The blue square indicate the delineation of the landscape grid cell in which it was measured.

The values represented by each of the grids are listed in Table 7.6 including the respective values of LSI and DENSPEOP for the grid cells in question. As broad measures for quantifying the location of the patches along the urban-rural gradient the measures of DENSPEOP and LSI work well and can be used without the addition of other measures. However, PURBLC, allow a more instantly understandable measure of the gradient. A quick impression can be formed of the surroundings based on the amount of urban land cover measured. Therefore, as an additional and universal measure, PURBLC was used as a specific measure to describe the location of the patches.

Table 7.6: The values for the PURBLC gradient as observed for the entire landscape area. Also listed are the corresponding DENSPEOP and LSI values.

	Low (a)	Median (b)	High (c)
PURBLC	*0*	*34.49*	*70.94*
DENSPEOP	*36*	*280*	*10517*
LSI	*3.58*	*26.94*	*19.76*

Parts of the suburbs of Jouberton and Kanana are of such a nature that they classify as soil (discussed in detail in Chapter 3). This creates the unique situation that for an area with a DENSPEOP value of 5972 only 13.13 % of the cell is classified as urban (Table 7.7). This illustrates some of the unique features of the urban-rural gradient in the greater Klerksdorp area. The profusion of soils are due to the dirt roads in the area and the small size of the houses found there, as a consequence the 10 m spatial resolution of the satellite image classified these areas as soil.

During the calculation of the urbanization measures and the quantification of the grid cells it was discovered that even though LSI is a very accurate measure to quantify the landscape structure it is not indicative of a typical urban-rural gradient from the rural area (lowest values) to the urban core (highest values). Due to the fact that the patches used in the calculation of the measure is derived from the land cover map, cells containing a substantially larger patch than the rest will have a lower value for LSI. This is sensitive to the spatial resolution of the satellite image, hence the substantially larger values for a measure such as the number of patches in the current study in comparison to Hahs (2006) (discussed in Chapter 4). Table 7.7 lists four scenarios found for LSI in the landscape of the current study, these correlate to the numbered maps in Figure 7.9. These scenarios were chosen to show the dominance of soil (Figure 7.9 a), grass (Figure 7.9 c), trees (Figure 7.9 e) and urban (Figure 7.9 g) respectively, and the similarity in the LSI values of these cells (18.88 – 19.76). For each scenario two maps are shown the land cover map and the panchromatic image overlaid on the land cover map to show the specific detail of the grid cell. Figure 7.9 shows the grid cell in question on the left of the figure (a, c, e, g) with the panchromatic image overlain to add detail to the map (b, d, f, h). Maps a, c, e, and g are the classified land cover map B (Chapter 3) of the landscape indicating the size of each class in the grid cell.

Table 7.7: List of different scenarios for cells along the LSI gradient. The dominant land cover (DOMLC) of cell is listed, as well as the corresponding values of PURBLC and DENSPEOP for each cell.

Fig 7.9 No.	DOMLC	DENSPEOP	LSI	PURBLC
a)	*Soil*	*5972*	*18.91*	*13.13*
c)	*Grass*	*320*	*19.27*	*12.05*
e)	*Trees*	*208*	*18.88*	*18.85*
g)	*Urban*	*10517*	*19.76*	*70.94*

The DENSPEOP values for these situations are widely dissimilar (Table 7.7). Therefore, by only using LSI to quantify the urban-rural gradient the most urbanized cell in the entire landscape would not be recognized. By using the PURBLC values in conjunction with the DENSPEOP a most accurate representation of the situation in the grid cell is formed. Therefore, we chose to use PURBLC as well to more accurately quantify the gradient by supplying additional information of each grid cell.

The situation that the most urbanized areas in the landscape exhibit the same LSI values as cells located in the rural areas and on the urban fringe reflects the fact that as urbanization intensifies the urban land cover becomes more homogeneous across the grid cell. Therefore, the higher the urbanization and the more uniform the urban land cover, as typically found in the CBD, the more it resembles its natural counterparts in terms of LSI values forming large 'patches' of urban land cover surrounded by highly fragmented residential areas. This situation was found in Melbourne as well where Hahs and McDonnell (2006) documented that the CBD had lower values of LSI than cells located on the urban fringe. In the current study the DENSPEOP values varied from 8 – 10517 for cells with LSI values of 18 – 20.

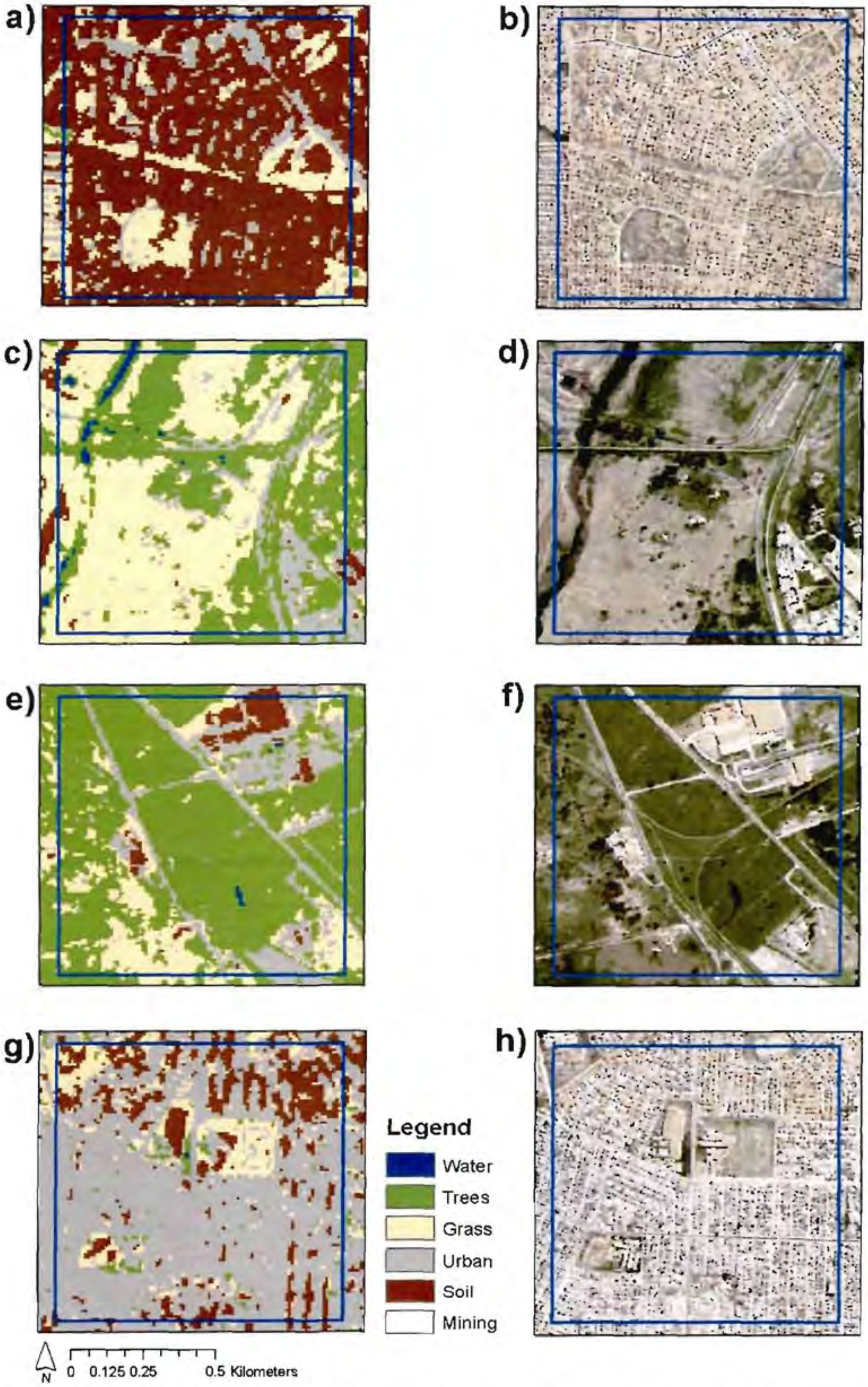


Figure 7.9: Maps of four grid cells found in the landscape of the greater Klerksdorp area that illustrates different scenarios where the LSI values are similar along the gradient. Each respective map on the left has a corresponding map on the right showing the panchromatic image overlay to add detail to the map. The boundaries of the grid cells in question are shown in blue.

Weng (2007) also illustrated this phenomenon by describing that in areas where more compact development is favoured previously highly fragmented areas will become more homogenous as small undeveloped gaps will be filled, which would effectively lower the fragmentation of the specific area as typical for the CBD. "*However, the urban matrix is different from the matrix in rural or natural landscapes and so are their respective ecological characteristics. Therefore, when comparing different landscapes, the background matrix should be explicitly defined since landscapes having the same metric value may represent rather different landscapes ecologically*" (Weng, 2007).

This highlights the importance of using multiple measures to quantify the urban-rural gradient.

7.3.3 The location of three grassland patches along the urban-rural gradient

Figure 7.10 illustrates three patches located along the quantified urban-rural gradient. Patch 14, represents the rural areas (Figure 7.10 a), patch 12 moderate urbanization (Figure 7.10 b), and patch 9 higher urbanization (Figure 7.10 c).

Figure 7.10: The representation of the urban-rural gradient as formed by three of the grassland patches in the landscape of the greater Klerksdorp area. Patch 14, represents the rural areas (a), patch 12 the moderately urbanized areas (b), with patch 9 representing the higher urbanization (c) endpoint. The background of each map is the panchromatic image overlaid on the land cover map. The red outline indicates the patch boundary with the yellow indicating the 1 km^2 buffer boundary.

Table 7.8 lists some of the attributes of each patch including the values of each of the urbanization measures. The values for DENSPEOP, LSI and PURBLC all increase towards the urban centre, with a subsequent decrease in the patch size and species richness of the existing vegetation of the respective patches. The total indigenous, dicot and grass species richness decrease towards the urban centre with respect to the existing vegetation. However, there is a general increase of exotic species but as described in chapter 6 no clear trend could be described for the exotic species composition. This possibly relates to the existence of a general ubiquitous seed rain across the landscape (Bastin and Thomas, 1999). As described previously in this Chapter and Chapter 6 the soil seed bank vegetation composition showed no definite patterns. Therefore, the values of average and total species richness for the soil seed bank in Table 7.8 should not be taken as indicative of a general trend for the quantified urban-rural gradient. As a result no inferences can be made of the observed soil seed bank patterns along the urban-rural gradient.

Table 7.8: The attributes of each of the patches representing the gradient of the greater Klerksdorp area. The average (AVE) and total species richness values (SR) of the existing vegetation (EXS) and the soil seed bank (SSB) are listed for each patch. The total indigenous, exotic, dicot and grass SR for the existing vegetation and the patch size are also listed.

	Patch 14	**Patch 12**	**Patch 9**
DENSPEOP	*36*	*251*	*584*
LSI	*4.4*	*29.34*	*32.2*
PURBLC	*0.03*	*16.59*	*29.55*
Patch size	*67.95*	*4.69*	*0.9*
Ave SR all EXS	*50.14*	*36*	*35*
Total SR all EXS	*108*	*64*	*52*
Total indigenous SR EXS	*101*	*60*	*36*
Total exotic SR EXS	*7*	*4*	*16*
Total dicot SR EXS	*71*	*45*	*36*
Total grass SR EXS	*24*	*15*	*10*
Ave SR SSB	*7.29*	*6.33*	*10.50*
Total SR SSB	*18*	*12*	*15*

An inspection of the gradient allows an arbitrary subdivision of the gradient into groupings of rural, moderate urbanization and higher urbanization. The values denoting each subdivision is given in Table 7.9. However, these values are not the exact description of each subdivision as some exceptions occur, but it indicates the general gradient within which most of the landscape cells can be quantified.

Figure 7.11 illustrates that the number of indigenous species for the existing vegetation (calculated as a proportion of all the counted individuals in the patch) decrease towards the urban centre, with a subsequent increase in the percentage of exotic species.

Table 7.9: The values for the subdivisions of the urban-rural gradient quantified for the greater Klerksdorp area. The values follow those identified in table 5.5 (Chapter 5).

	Rural	Moderate urbanization	Higher urbanization
DENSPEOP	*0 – 100*	*100 – 300*	*300 +*
LSI	*0 – 10*	*10 – 30*	*30 – 40*
PURBLC	*0 – 5*	*5 – 28*	*28 +*

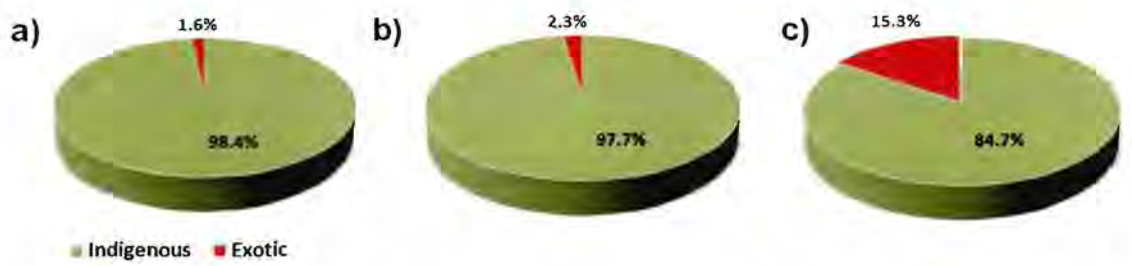


***Figure 7.11:* The *proportion* individuals of indigenous and exotic plant species occurring within the respective patches in the existing vegetation, namely: patch 14 (a), patch 12 (b), and patch 9 (c).**

The composition of the existing vegetation shifts from an almost equal dominance of grasses and dicots as found in patch 14 (Figure 7.12 a) to subsequent increase in the dominance of grasses towards the urban centre (Figure 7.12 c). This can be ascribed to the disappearance of large numbers of indigenous diversity as one move towards the urban centre. Pioneer species and species thriving in disturbed habitats typically take over, as evident from the dominance of *Urochloa mosambicensis* in patch 9 (Table 7.12). Total species richness loss is from 108 to only 52 recorded in patch 9 (Table 7.8). The decrease in dicots from 71 to 36 species and the grasses from 24 to 10 species are also evidence of this (Table 7.8).

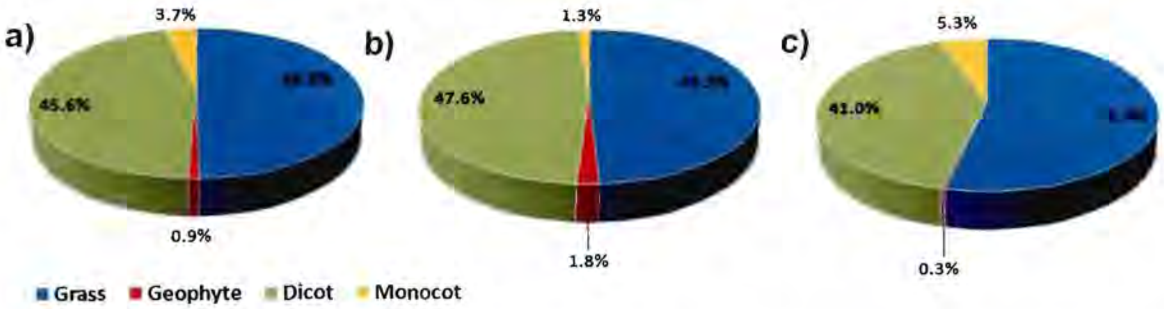


Figure 7.12: The proportional composition of the life forms recorded at each patch for the existing vegetation. Patch 14 (a), patch 12 (b), and patch 9 (c).

Figure 7.13 clearly indicates that in the soil seed bank the proportion of exotic species significantly increases towards the urban centre, with a subsequent decrease in the percentage of indigenous species. It is clear that there are much higher percentages of exotic species in the soil seed bank than the exotic species counted in the existing vegetation.

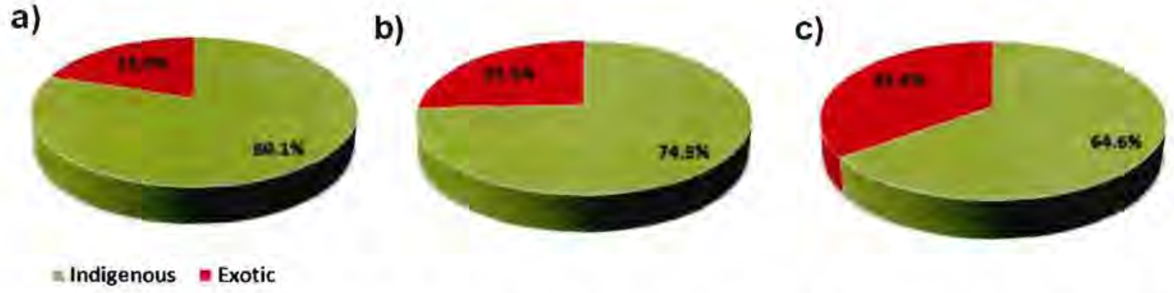


Figure 7.13: The proportion of indigenous and exotic plant species occurring within the respective patches in the soil seed bank analysis, namely: patch 14 (a), patch 12 (b), and patch 9 (c).

Figure 7.14, in turn indicates the increasing dominance of grass species in the soil seed bank as the urban centre is reached. This is very interesting, as the species richness of the grass species drastically decreased towards the urban centre in the existing vegetation (Figure 6.14 a, Chapter 6). However, the dominance of grass in patch 9 (Figure 7.14 c) is ascribed mainly to only one species, *Urochloa mosambicensis*, a grass occurring in disturbed areas, producing a profusion of viable seeds (Van Outshoorn, 2004).

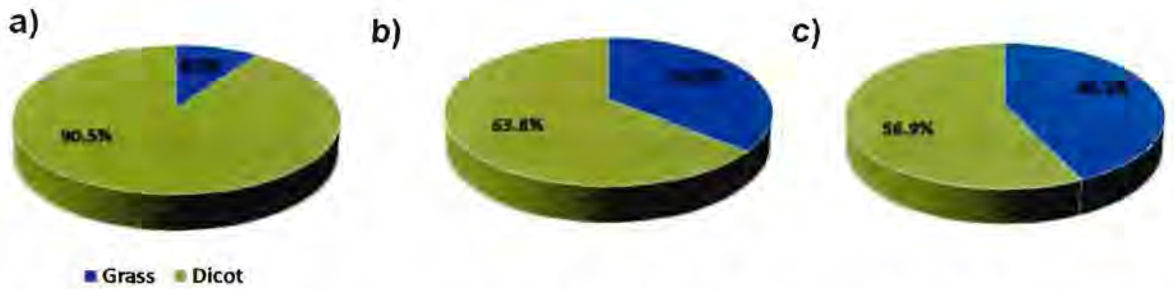


Figure 7.14: The proportional composition of the life forms recorded at each patch in the soil seed bank analysis. Patch 14 (a), patch 12 (b), and patch 9 (c).

Tables 7.10 to 7.12 show the species composition of each patch for the existing and the soil seed bank vegetation. Listed in descending order of importance of the frequency of occurrence are the four most dominant extant species with their percentage contribution to the total amount of individuals recorded in the patch, as well as the three most dominant species in the soil seed bank.

Table 7.10 indicates that *Themeda triandra* is the most dominant species in patch 14 accounting for 17.4 % of all the sampled individuals, with *Elionurus muticus* and *Aristida canescens* subsp. *canescens* respectively in the second and third position. Patch 14 is the only patch of the three discussed here where the grasses are the most dominant species of the patch. However, it should be noted that both *Elionurus muticus* and *Aristida canescens* subsp. *canescens* increase with disturbance such as overgrazing (Van Oudtshoorn, 2004) which might have been part of the reason for their dominance, as patch 14 was subjected to grazing.

Table 7.10: A list of the dominant species occurring in the existing vegetation and the soil seed bank of patch 14. The relative proportions of each species to all the counted individuals in the patch are given. The origin, life cycle and life form of each species are also listed.

Existing vegetation				
Themeda triandra	17.4 %	Indigenous	Perennial	Grass
Elionurus muticus	14.1 %	Indigenous	Perennial	Grass
Aristida canescens subsp. *canescens*	6.5 %	Indigenous	Perennial	Grass
Lippia scaberrima	5 %	Indigenous	Perennial	Dicot
Soil seed bank				
Wahlenbergia undulata	67 %	Indigenous	Perennial	Dicot
Gamochaeta pennsylvanica	12.1 %	Exotic	Annual	Dicot
Eragrostis chloromelas	5.8 %	Indigenous	Perennial	Grass

Table 7.11 lists the prominent species for patch 12, *Themeda triandra* is still the most dominant individual, but its proportions have increased significantly from that recorded for patch 14.

Table 7.11: A list of the dominant species occurring in the existing vegetation and the soil seed bank of patch 12. The relative proportions of each species to all the counted individuals in the patch are given. The origin, life cycle and life form of each species are also listed.

Existing vegetation				
Themeda triandra	24.5 %	Indigenous	Perennial	Grass
Pollichia campestris	14.2 %	Indigenous	Perennial	Dicot
Heteropogon contortus	11.6 %	Indigenous	Perennial	Grass
Eragrostis chloromelas	7.2 %	Indigenous	Perennial	Grass
Soil seed bank				
Conyza bonariensis	23.4 %	Exotic	Annual	Dicot
Eragrostis chloromelas	19.1 %	Indigenous	Perennial	Grass
Panicum maximum	12.8 %	Indigenous	Perennial	Grass

The second most prominent species of the patch is the indigenous forb *Pollichia campestris*. However, the most dominant species in the soil seed bank is an exotic species, indicating the influence of anthropogenic effects on the patch and its location in the urban areas.

Table 7.12 lists the prominent species for patch 9, the contribution of the most dominant species increased to 38.8 % with the next species only representing 7.7 % of the community composition. It is interesting to note that in this patch *Urochloa mosambicensis* dominates both the existing and the soil seed bank component.

Table 7.12: A list of the dominant species occurring in the existing vegetation and the soil seed bank of patch 9. The relative proportions of each species to all the counted individuals in the patch are given. The origin, life cycle and life form of each species are also listed.

Existing vegetation				
Urochloa mosambicensis	38.8 %	Indigenous	Perennial	Grass
Euphorbia inaequilatera var. *inaquilatera*	7.7 %	Indigenous	Annual	Dicot
Sporobolus africanus	7.4 %	Indigenous	Perennial	Grass
Delosperma herbeum	5.5 %	Indigenous	Perennial	Dicot
Soil seed bank				
Urochloa mosambicensis	30.8 %	Indigenous	Perennial	Grass
Conyza bonariensis	12.3 %	Exotic	Annual	Dicot
Delosperma herbeum	10.8 %	Indigenous	Perennial	Dicot

The continued increase of the contribution of the dominant species as one move towards the urban centre indicates that in natural areas the plant communities are more diverse with equal contributions of a large number of species. These communities are far more stable and resilient towards periodic disturbances and stress conditions than the plant communities of the urban patches. The higher urbanized patches typically reflect the dominance of one or two species which could more readily be replaced or affected by disturbances. Urbanization clearly influences the species composition of communities and within cities biological communities are often dissimilar to surrounding communities as the urban species become reshuffled into novel communities (Sukopp, 1998; Angold *et al.*, 2006).

This creates important conservation issues for grassland fragments persisting in urban areas. However, the potential for unique species assemblages to form on these disturbed patches should not be underestimated towards their possible conservation value.

7.4 Summary

It is clear that urbanization does influence grassland vegetation communities along the urban-rural gradient described for the greater Klerksdorp area.

The majority of research in urban environments documents the effect of anthropogenic influences on the species richness of remnant natural vegetation and neglect the importance of the overall species composition. In the current study, it was clear that an urban-rural gradient could be observed for both the patterns of species richness and the patterns of the overall vegetation composition of the patches.

Hahs and McDonnell (2006) emphasized that the identification of a subset of appropriate uncorrelated measures that best quantified the variability in the landscape is one of the major challenges for current research. In the current study the measures of density of people and landscape shape index fulfilled such a role with the added measure of percent urban land cover. All of these measures showed significant correlations to the observed species richness patterns of the recorded species for the existing vegetation, but less so for the soil seed bank. This is as a result of the lack of clear patterns observed for the soil seed bank as discussed in Chapter 6.

The location of three grassland patches could be quantified along the observed urban-rural gradient, where the increasing urbanization influences on their vegetation composition could clearly be observed. Therefore, the selected measures can be used, if not as a direct indicator, than as an indirect indicator of the processes driving vegetation pattern distributions in this landscape on 1 km^2 composite landscape grid scale. This is on par with Li and Wu (2004) when they affirmed that "*to reduce uncertainty and increase predictability, future research in landscape pattern analysis must go beyond the mere quantification of landscape pattern and emphasize its relationships to ecological processes*".

The three grassland patches discussed showed that compositionally urbanization creates communities where a few individual species dominate in contrast to the more equitable composition of the rural areas. The rural areas represent more stable communities with the urban areas possessing more dynamic communities. This could imply that with disturbances or large influences on the communities the composition could change drastically, especially if the dominant species are replaced.

The final chapter forms the conclusion and summarizes the findings of the current study, discussing the contribution made by the study as well as recommendations for future studies.

Chapter 8: Conclusion

The quantification of the urban-rural gradient and the urbanization influence on vegetation patterns of fragmented native grasslands formed the two main themes explored in this dissertation. The results of the dissertation could further be divided into three main parts, namely: landscape spatial context (Chapter 3 and 4), patterns of plant diversity and species composition (Chapter 5 and 6), and the quantification of the urban-rural gradient (Chapter 7).

8.1 Landscape spatial context

Two objectives were explored in this section, namely: (1) to use SPOT 5 satellite imagery and GIS techniques to determine which of the demographic-, physical variables and landscape metrics identified in Melbourne, Australia can be used to quantify urbanization in the greater Klerksdorp area, and (2) to use the identified urbanization measures to quantify the urban-rural gradient of the greater Klerksdorp area.

Satellite imagery was used to create the land cover map of the study area. The hybrid classification procedure used to create land cover map B with Spatial Analyst, illustrated that ArcView can be used successfully to classify satellite images in the absence of better, more specialized software. The 95 % overall accuracy of land cover map B also illustrated the usefulness of Spatial Analyst for classifying satellite images. The use of SPOT 5 satellite imagery was successful in accurately delineating the urban areas of the greater Klerksdorp area; nonetheless the limited number of spectral bands (4) put minor constraints on the urban land cover classification. In the current study parts of some of the large buildings classified as soil, which influenced the amount of classified urban areas in the land cover map, but it was of little consequence as the land cover map was classified with broad classes. The overall accuracy of the urban classification was, therefore, more than adequate for the objectives of the study, but in a more detailed study of urban areas where there might be subdivisions of urban zones this can be a potential problem. The 7 spectral bands of Landsat TM allows better delineation between urban and soil classes because it detects in a much wider range where combinations of the different bands can delineate between different soil types as illustrated in the classification procedure of Hahs (2006). However, the main drawback of Landsat imagery is its 30 m resolution which would have had a large effect on the classification of the urban areas in the current study. Landsat imagery would have considerably diminished the urban classification in areas where grasses and trees are dominant on a 30 m scale, a scenario that is quite common on the urban fringes and residential areas in the greater Klerksdorp area.

Two urbanization measures were identified namely: density of people and landscape shape index as being informative in quantifying urban-rural gradients. They were easy to calculate and the input data needed are readily available, regarded by Hahs and McDonnell (2006) as important aspects for inclusion into a standard set of measures. By using the same measures identified in the study of Hahs (2006) a contribution was made to the creation of a standard set of measures by evaluating how the measures explained the variability in the greater Klerksdorp area. Landscape shape index, chosen in the current study was also one of the four measures chosen by Hahs and McDonnell (2006). The paper by Cushman *et al.* (2008) also identified landscape shape index as part of a subset of independent components to quantify the maximum variation in the landscape structure. However, they emphasized that their analysis did not focus on the relation of the measures to ecological processes and that the specific studies using the measures should elucidate this.

However, direct comparisons of studies require exactly the same type and scale of input data sets. In the current study the scale of the demographic input data and the spatial resolution of the satellite image influenced the calculation of the measures. Nevertheless, the response of variables such as in the comparison of the current study with that of Hahs (2006) illustrates that the absolute values of some variables might not be as informative, as for instance using the proportional distributions of the respective values in comparison with themselves and the values in other studies.

Importantly, the range variability of the input measures also influences its loadings on principal components analysis (PCA) or factor analysis (FA). Therefore, good quality variables are needed to compute the FA and PCA; otherwise the PCA/FA results might be biased towards the values with a better distribution despite its appropriateness as an ecological indicator. However, in the current study the FA showed that both landscape and the demographic measures are needed to describe the maximum variation in the data, together they explained 85.7 % of the variance with the landscape metrics explaining 49.4 % of the variation and the demographic variables 36.3 %. This strengthened the findings of Hahs and McDonnell (2006) on the importance of using both demographic and landscape measures to quantify the urban-rural gradient and contributes to the understanding that a standard set of measures should include both types of urbanization measures.

Tavernia and Reed (*in press*) recently examined the effect of varying spatial extents and habitat context on correlations between demographic and landscape measures. PCA was used to determine the correlation between measures; however as with Hahs (2006) all the variables on the first component loaded almost equally with low values. We proposed the use of FA with varimax rotated components as in the current study, because this clearly identified independent and uncorrelated measures. Chen *et al.* (2008) and Schindler *et al.* (2008) also used FA with a varimax rotation to identify independent variables. The type of analysis used can influence the outcomes of the study and future studies should therefore identify which analysis is the most appropriate. This could have a potentially important impact on the correct choice of a truly independent subset of standard measures.

8.2 Patterns of plant diversity and species composition

The objective (Chapter 1) explored in this section was to use vegetation and soil surveys to quantify the influence of human impacts on grassland ecology, investigating aspects such as: plant species composition and species richness; and specific soil properties.

The selected grassland patches represented the majority of the patches found in the landscape; however they were not representative of the entire gradient for the demographic measures of urbanization. The areas of Jouberton and Kanana contained the highest population densities however they were not sampled as there were no suitable grassland patches to sample.

The observed vegetation patterns indicated the definite presence of a disturbance gradient along the grassland patches, which is in contrast to the study of Hahs (2006) where no clear urbanization influence on the remnant woodlands of Melbourne could be discerned. In the current study the fragmented grassland patches in the urban areas had lower overall species richness than the rural patches. The species richness therefore did not increase in the urban areas as generally documented (Rebele, 1994; McKinney, 2002; Hope *et al.*, 2003). It is clear that urbanization influenced the overall species composition, the indigenous, the grass, the perennial and the dicot species composition of the existing vegetation in the current study. All the listed species subsets showed a similar pattern of a decrease in species richness towards the urban centre. McKinney (2008) stated that the species richness of plants decrease from rural to urban areas as was indicated for the current study, where it was particularly evident in the patterns of indigenous and the grass species richness. The exotic and annual species composition of the existing vegetation showed no clear gradient. Overall 201 indigenous and 27 exotic species were recorded for the existing vegetation.

In the soil seed bank 28 indigenous and 20 exotic species were recorded. In the study of Hahs (2006) the majority of the individual seedlings that were counted were exotic (51.2 %) and monocots (75.4 %), in the current study the majority were indigenous (64 %) and dicots (78 %). However, species richness of the soil seed bank is not associated with the described urban-rural gradient. The disturbance gradient found in the greater Klerksdorp area is, therefore, not due to an increase of exotic species towards the urban centre, but due to the decrease of indigenous species richness as one nears the urban centre.

The vegetation pattern elucidation was based on species recorded in the individual sample plots. Surveying the entire patch and not only representative sample pots as suggested by Hahs (2006), could be informative and confirm or negate the patterns observed for the sample plots.

By observing the influence of urbanization on the patterns of species composition and species richness in the current study, the fact that the vegetation type in which the study area is located is critically endangered (Mucina and Rutherford, 2006) was emphasized for the greater Klerksdorp area. The clear effect of urbanization related influences on the occurrence of indigenous grassland species is disconcerting. The current study illustrated that larger patches of fragmented grasslands should be conserved in and around urban areas to conserve the indigenous biodiversity of the vegetation type. This emphasizes the importance of doing ecological studies in urban areas to elucidate the actual status of the biodiversity in the region and the importance of presenting ecological data in a manner useful and understandable to policy makers, stakeholders and urban planners (Cilliers *et al.*, 2004).

8.3 Quantification of the urban-rural gradient

The objective (Chapter 1) explored in this section was to use the identified urbanization measures to quantify the urban-rural gradient of the greater Klerksdorp area. The majority of research in urban environments documents the effect of anthropogenic influences on the species richness of remnant natural vegetation and neglect the importance of the overall plant species composition. In the current study, it was clear that an urban-rural gradient could be observed for both the patterns of species richness and the patterns of the overall plant species composition of the patches. However, a comparison of the total species richness and the average species richness of the patches showed that for patches larger than four hectares the similarity for the species richness was below 50 %. This again adds to the importance of surveying the entire patch to observe which patterns would then be apparent.

The average and the total species richness of the subsets of species of the existing vegetation and the soil seed bank was significantly correlated to the pH and the percentage base saturation, indicating the importance of the specific soil properties to the vegetation composition of a specific patch. The average species richness of the annual and the exotic species showed a strong positive correlation to the density of people and the density of dwellings, with the indigenous and the perennial species richness a strong negative correlation.

The total species richness of the existing vegetation showed strong correlations with almost all of the landscape metrics with the best correlations with Simpson's diversity index and landscape shape index. All the species subsets correlated to the landscape metrics except the exotic, annual and grass species richness. The patch size correlated positively to all the species, the indigenous, perennial and dicot species richness.

Hahs and McDonnell (2006) stated that the selection of the measures can influence the findings of the study, our study emphasised that the type of biological input data could also influence the significance of the measures used. Li and Wu (2004) reflected on the importance of this by matching the scale of observation to the scale of analysis in determining processes that influence ecological observed patterns. The determination of the scale of landscape pattern formation of an ecological community may provide indications as to the processes influencing its spatial and temporal dynamics (Fonseca *et al.*, 2008).

Hahs and McDonnell (2006) emphasized that the identification of a subset of appropriate uncorrelated measures that best quantified the variability in the landscape is one of the major challenges for current research. In the current study the measures of density of people and the landscape shape index fulfilled such a role with the added measure of percent urban land cover. All of these measures showed significant correlations to the observed species richness patterns of the recorded species for the existing vegetation. The selected measures were successfully used to quantify the urban-rural gradient and by combining all three measures, the specific location of each grid cell along the gradient could be accurately described. Each of the measures showed potential problems if used as the sole measure to quantify the gradient, emphasizing that the quantified gradient should be as descriptive as possible (McIntyre *et al.*, 2000; Weng, 2007).

The location of three grassland patches could be quantified along the observed urban-rural gradient, where the increasing urbanization influences on their vegetation composition and species richness could clearly be observed. In the three patches there were decreases in the patch size and the total species richness of the indigenous, dicot and grass species subsets towards the highly urbanized centre with a general accompanying increase in the exotic species. The vegetation composition of the patches all favoured indigenous species, but for the existing and the soil seed bank vegetation the proportion of exotic species increased towards the urban centre. Interestingly the amount of indigenous species recorded along the gradient drastically decreases towards the urban centre; however the life form composition of the patches stays relatively the same. However, the dominance of individual species in the vegetation community increases towards the urban centre indicating more dynamic and disturbed communities.

The influence of urbanization could be successfully quantified along an urban-rural gradient and the selected urbanization measures were also biologically meaningful.

8.4 Recommendations on further research

The sole use of ArcView and its Spatial Analyst Extension allowed a novel way of classifying the satellite image and future studies might benefit from using it, in the absence of more sophisticated software. The classification procedure outlined in the current study could also be refined to further enhance the usefulness of Spatial Analyst as an alternative to sophisticated remote sensing software.

Future studies could benefit from including additional grassland patches not sampled in the current study to test if the general patterns identified in this dissertation hold for a more detailed look at the remaining urban grassland patches in the greater Klerksdorp area. The surveying of the entire patch could also yield additional information on the processes affecting grassland ecology along an urbanization gradient.

Possible point sources of pollution driving intra patch dynamics could also be identified in this manner. The use of surface interpolation rasters visualizing the entire patch and the specific species richness at different points in the patch could also potentially lead to a better understanding of plant species patterns and the processes influencing it.

The soil seed bank showed no observed trends in species composition and richness as discussed previously. This might be due to the low number of species counted in the germination period. Therefore, future studies should extend the germination period and explore the influence of pre-treating the samples. Our samples were also spread in a 3 cm layer in the trays which might have affected the germination of the seeds in the bottom part of this layer, potentially responsible for the low number of seedlings recorded.

Future studies should also investigate the organic matter breakdown rates of the different patches to further explore the soil properties of the patches and its possible influence on species composition, potentially elucidating a process driving the species composition of the grassland patches in the greater Klerksdorp area.

The influence of mining was not directly quantified in the landscape and additional soil surveys explicitly testing the influence of mining to the soil properties of the area could identify additional drivers of species diversity patterns in the greater Klerksdorp area.

The identified urbanization measures should also be tested to identify the possible processes that can be directly linked with them, possibly using additional specific measures to quantify the drivers of vegetation patterns in the current study.

Lastly, the findings of the study should also be tested in additional urban areas in South Africa to elucidate potential general trends and strengthen the basis of ecological knowledge needed to adequately combat biodiversity loss and native grassland fragmentation in South Africa.

8.5 Summary

McDonnell and Hahs (2008) stated that further research should expand the range of cities being investigated, particularly to include cities in other parts of the world where development patterns may differ in testing the applicability of a standard set of measures. The endeavour of this dissertation was therefore to assist in the advancement of comparative urban ecological studies by documenting the results found in a South African, Third World setting. By quantifying the urban-rural gradient of Klerksdorp this dissertation also contributed to the general knowledge of urban ecological studies in South Africa. The study area allowed an exceptional opportunity to describe an unique and complex urbanization gradient.

The same measures were apparent, especially with regards to the importance of using both landscape metrics and demographic and physical measures, with a different development pattern for the urban areas in the greater Klerksdorp area as observed in Melbourne. This reflects the possible attainability of a standard set of broad measures with which to quantify the urbanizational context of a wide variety of studies and geographical areas.

To evaluate the contribution of this study to the wider body of urban ecological knowledge it is important to note that the current study addressed some of the knowledge gaps identified by McDonnell and Hahs (2008) in their paper on the use of gradient studies. The following knowledge gaps were identified: (1) identification of the most informative combinations of urbanization measures; (2) the influence of the landscape classification system; (3) the level of redundancy of correlation between selected measures and (4) the ability to attach a biological or ecological interpretation to the selected measure.

By testing the measures proposed by Hahs and McDonnell (2006) in a South African city the usefulness of some of the measures were again demonstrated. Importantly, the current study also identified the importance of using both demographic measures and landscape metrics to quantify the gradient of an urban area. The use of 10 m resolution SPOT 5 imagery with the selected measures of Hahs and McDonnell (2006), identified the influence of the enhanced scale on the comparativeness of the two studies and the sensitivity of some of the measures to changes in the grain size of the input data, However, the urban areas could be accurately delimited which is important in areas constrained by size as the urban areas of the current study. The level of redundancy between the various measures in this study was unequivocally identified by the use of factor analysis and correlation matrices. The correlations between some of the measures and the species richness data elucidated the biological and ecological meaningfulness of some of the measures. However, the extent to which they are direct indicators of underlying processes driving species diversity patterns should be tested further.

Therefore, the current study contributed towards the advancement of urban ecological knowledge and locally identified the patterns of grassland species distributions in response to urbanization influences. This knowledge will help to sustainably manage the fragmented grasslands in the Vaal-Vet Sandy Grassland in an effort to conserve a critically endangered vegetation type. This study could also serve as a potential model of elucidating the effects of urbanization in grasslands of South Africa.

"*We know that the totality of human activity occurs on a biophysically constrained planet, and urban ecology can elucidate the connections between city dwellers and the biogeophysical environment in which they reside*" (Grimm *et al.*, 2008).

Bibliography

ALBERTI, M. 2005. The effects of urban patterns on ecosystem function. *International Regional Science Review*, 28(2):168-192.

ALBERTI, M., BOOTH, D., HILL, K., COBURN, B., AVOLIO, C., COE, S. & SPIRANDELLI, D. 2007. The impact of urban patterns on aquatic ecosystems: an empirical analysis in Puget lowland sub-basins. *Landscape and Urban Planning*, 80:345-361.

ALBERTI, M., MARZLUFF, J., SHULENBERGER, E., BRADLEY, G., RYAN, C. & ZUMBRUNNEN, C. 2003. Integrating humans into ecology: opportunities and challenges for studying urban ecosystems. *Bioscience*, 53(12): 1169-1179.

ANAYA, J.A., CHUVIECO, E. & PALACIOS-ORUETA, A. 2009. Aboveground biomass assessment in Colombia: a remote sensing approach. *Forest Ecology and Management*, 257:1237-1246.

ANDERSON, J.R., HARDY, E.E., ROACH, J.T. & WITMER, R.E. 1976. A land use and land cover classification system for use with remote sensor data. Geological Survey Professional Paper 964. Washington: United States Government Printing Office.

ANGOLD, P.G., SADLER, J.P., HILL, M.O., PULLIN, A., RUSHTON, S., AUSTIN, K., SMALL, E., WOOD, B., WADSWORTH, R., SANDERSON, R. & THOMPSON, K. 2006. Biodiversity in urban habitat patches. *Science of the Total Environment*, 360:196-204.

APLIN, P. 2005. Remote sensing: ecology. *Progress in Physical Geography*, 29(1):104-113.

BAND, L.E., CADENASSO, M., GRIMMOND, S., GROVE, M. & PICKETT, S.T. 2005. Heterogeneity in Urban Ecosystems: Patterns and Processes, Chaper 13. (*In* G.M. Lovett, C.G. Jones, M.G. Turner & K.C. Weathers, *eds*. Ecosystem Function in Heterogeneous Landscapes. New York: Springer-Verlag. p. 257-278)

BARNARD, F.J., CILLIERS, S.S. & DE KLERK, T.C. 2008. Fragmentation and plant invasions in native grasslands in and around cities in the North-West province, South Africa. (*In* Müller, N., Knight, D. & Werner, P., *eds*. Urban biodiversity & design. Implementing the Convention on Biological Diversity in towns and cities: Third Conference of the Competence Network Urban Ecology, Erfurt, 21-24 May 2008. Bonn: Bundesamt für Naturschutz. p. 34.)

BASTIN, L. & THOMAS, C.D. 1999. The distribution of plant species in urban vegetation fragments. *Landscape Ecology,* 14:493-507.

BENNETT, E.M. 2003. Soil phosphorus concentrations in Dane County, Wisconsin, USA: an evaluation of the urban-rural gradient paradigm. *Environmental Management,* 32(4):476-487.

BERTHIER, E. & TOUTIN, T. 2008. SPOT5-HRS digital elevation models and the monitoring of glacier elevation changes in North-West Canada and South-East Alaska. *Remote Sensing of Environment,* 112:2443-2454.

BEYER, H. 2007. Hawth's analysis tools v.3.27. [Web:] http://www.spatialecology.com/htools/index.php [Date of use: 21 Jul 2008]

BLACK, C.A. 1968. Soil-plant relationships. 2nd ed. New York: Wiley. 792 p.

BLAIR, R.B. & JOHNSON, E.M. 2008. Suburban habitats and their role for birds in the urban–rural habitat network: points of local invasion and extinction? *Landscape Ecology,* 23:1157-1169.

BOX G.E.P. & COX, D.R. 1964. An analysis of transformations. *Journal of the Royal Statistical Society. Series B (Methodological),* 26(2):211-252.

BRASE, C.H. & BRASE, C.P. 1999. Understandable statistics: concepts and methods. 6th ed. Boston: Houghton Mifflin Company. 775 p.

BREDENKAMP, G.J., BEZUIDENHOUT, H., JOUBERT, H. & NAUDE, C. 1994. The vegetation of the Boskop Dam Nature Reserve, Potchefstroom. *Koedoe,* 37(1):19-33.

BREUSTE, J.H. 2004. Decision making, planning and design for the conservation of indigenous vegetation within urban development. *Landscape and Urban Planning,* 68: 439-452.

BROWN, D.G., JOHNSON, K.M., LOVELAND, T.R. & THEOBALD, D.M. 2005. Rural land-use trends in the conterminous United States, 1950-2000. *Ecological Applications,* 15:1851-1863.

BURTON, M.L., SAMUELSON, L.J. & MACKENZIE, M.D. 2009. Riparian woody plant traits across an urban-rural land use gradient and implications for watershed function with urbanization. *Landscape and Urban Planning,* 90:42-55.

CAMPBELL, J.B. 2006. Introduction to remote sensing. 4th ed. London: Taylor & Francis. 626 p.

CASTANEDO, C.I. 2005. Appendix A. Fundamentals of infrared radiation. [Web:] http://archimede.bibl.ulaval.ca/archimede/files/545db9f1-c794-47ff-9ebf-0af00068a7b5/apb.html [Date of use: 6 March 2009]

CAVIA, R., CUETO, G.R. & SUÁREZ, O.V. 2009. Changes in rodent communities according to the landscape structure in an urban ecosystem. *Landscape and Urban Planning*, 90:11-19.

CHEN, T., LIU, X., ZHU, M., ZHAO, K., WU, J., XU, J. & HUANG, P. 2008. Identification of trace element sources and associated risk assessment in vegetable soils of the urban-rural transitional area of Hangzhou, China. *Environmental Pollution*, 151: 67-78.

CHRISTOFFOLETI, P.J. & CAETANO, R.S.X. 1998. Soil seed banks. *Scientia Agricola, Piracicaba*, 55(special issue):74-78.

CHRISTOPHER, A.J. 2001. Urban segregation in post-apartheid South Africa. *Urban Studies*, 38(3):449-466.

CILLIERS, S.S. & BREDENKAMP, G.J. 1998. Vegetation analysis of railway reserves in the Potchefstroom municipal area, North-West Province, South Africa. *South African Journal of Botany*, 64(5):271-280.

CILLIERS, S.S. & BREDENKAMP, G.J. 1999a. Analysis of the spontaneous vegetation of intensively managed urban open spaces in the Potchefstroom municipal area, North West Province, South Africa. *South African Journal of Botany*, 65(1):59-68.

CILLIERS, S.S. & BREDENKAMP, G.J. 1999b. Ruderal and degraded natural vegetation on vacant lots in the Potchefstroom municipal area, North West Province, South Africa. *South African Journal of Botany*, 65(2):163-173.

CILLIERS, S.S. & BREDENKAMP, G.J. 2000. Vegetation of road verges on an urbanisation gradient in Potchefstroom, South Africa. *Landscape and Urban Planning*, 46:217-239.

CILLIERS, S.S., MŰLLER, N. & DREWES, E. 2004. Overview on urban nature conservation: situation in the western-grassland biome of South Africa. *Urban Forestry and Urban Greening*, 3:49-62.

CILLIERS, S.S., SCHOEMAN, L.L. & BREDENKAMP, G.J. 1998. Wetland plant communities in the Potchefstroom Municipal Area, North-West, South Africa. *Bothalia*, 28(2):213-229.

CILLIERS, S.S., VAN WYK, E. & BREDENKAMP, G.J. 1999. Urban nature conservation: vegetation of natural areas in the Potchefstroom municipal area, North West Province, South Africa. *Koedoe*, 42(1):1-30.

CILLIERS, S.S., WILLIAMS, N.S.G. & BARNARD, F.J. 2008. Patterns of exotic plant invasions in fragmented urban and rural grasslands across continents. *Landscape Ecology*, 23:1243-1256.

CLARKE, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, 18: 117-143.

CLARKE, K.R. & GORLEY, R.N. 2001. 'PRIMER v.5: User Manual/ Tutorial.' PRIMERE Ltd.: Plymouth, U.K.

CNES. 2007. SPOT. [Web:] http://www.geoimage.com.au/geoweb/spot/spot_overview.htm [Date of use: 12 Oct. 2007]

COHEN W.B. & GOWARD, S.N. 2004. Landsat's role in ecological applications of remote sensing. *BioScience*, 54(6):535-545.

COHEN, W.B. & SPIES, T.A. 1992. Estimating structural attributes of Douglas-fir/western hemlock forest stands from Landsat and SPOT imagery. *Remote Sensing of Environment*, 41:1-17.

COLLINS, J.P., KINZIG, A., GRIMM, N.B., FAGAN, W.F., HOPE, D., WU, J. & BORDER, E.T. 2000. A New Urban Ecology. *American Scientist*, 88: 416-425.

COOPS, N.C., WULDER, M.A. & IWANICKA, D. 2009. Demonstration of a satellite-based index to monitor habitat at continental-scales. *Ecological Indicators*, 9:948-958.

CORRY, R.C. & NASSAUER, J.I. 2005. Limitations of using landscape pattern indices to evaluate the ecological consequences of alternative plans and designs. *Landscape and Urban Planning*, 72:265-280.

CUSHMAN, S.A., MCGARIGAL, K. & NEEL, M.C. 2008. Parsimony in landscape metrics: strength, universality, and consistency. *Ecological Indicators*, 8:691-703.

DANIEL, H. & LECAMP, E. 2004. Distribution of three indigenous fern species along a rural–urban gradient in the city of Angers, France. *Urban Forestry and Urban Greening*, 3:19-27.

DAVIES, R.J. 1981. The spatial formation of the South African city. *GeoJournal*, 2(Supplement 2): 59-72.

DEPARTMENT of Constitutional Development and Planning ***see*** SOUTH AFRICA. Department of Constitutional Development and Planning.

DESMET, P. & COWLING, R. 2004. Using the species–area relationship to set baseline targets for conservation. *Ecology and Society*, 9(2):11. http://www.ecologyandsociety.org/vol9/iss2/art11

DE VILLIERS, A.J., VAN ROOYEN, M.W. & THERON, G.K. 2003. Similarity between the soil seed bank and the standing vegetation in the strandveld succulent Karoo, South Africa. *Land Degradation and Development*, 14(6):527-540.

DEVORE, J. & PECK, R. 1993. Statistics: the exploration and analysis of data. 2nd ed. Belmont: Duxbury Press. 881 p.

DUMAS, E., JAPPIOT, M. & TATONI, T. 2008. Mediterranean urban-forest interface classification (MUFIC): a quantitative method combining SPOT5 imagery and landscape ecology indices. *Landscape and Urban Planning*, 84:183-190.

DYTHAM, C. 2001. Choosing and using statistics: a biologist's guide. Oxford: Blackwell Science Ltd. 218 p.

ESRI (Environmental Systems Research Institute). 2006a. ArcView 9.2. www.esri.com Redlands, CA: USA

ESRI (Environmental Systems Research Institute). 2006b. ArcGIS desktop 9.2 help. Values and what they represent. www.esri.com Redlands, CA: USA

ESRI (Environmental Systems Research Institute). 2006c. ArcGIS desktop 9.2 help. Implementing Inverse Distance Weighted (IDW). www.esri.com Redlands, CA: USA

ESRI (Environmental Systems Research Institute). 2008. ArcGIS desktop 9.3 help. Raster bands. [Web:] http://webhelp.esri.com/arcgisdesktop/9.3/index.cfm?TopicName=Raster_bands [Date of use: 10 March 2009]

FAGGI, A.M., KRELLENBERG, K., CASTRO, R., ARRIAGA, M. & ENDLICHER, W. 2006. Biodiversity in the Argentinian rolling pampa ecoregion: changes caused by agriculture and urbanisation. *Erdkunde*, 60:127-138.

FASSNACHT, K.S., COHEN, W.B. & SPIES, T.A. 2006. Key issues in making and using satellite-based maps in ecology: a primer. *Forest Ecology and Management*, 222:167-181.

FONSECA, M.S., KENWORTHY, W.J., GRIFFITH, E., HALL, M.O., FINKBEINER, M. & BELL, S.S. 2008. Factors influencing landscape pattern of the seagrass *Halophila decipiens* in an oceanic setting. *Estuarine, Coastal and Shelf Science,* 76:163-174.

FOTH, H.D. & TURK, L.M. 1972. Fundamentals of soil science. New York: Wiley. 454 p.

FRITZ, L.W. 1996. The era of commercial earth observation satellites. *Photogrammetric Engineering and Remote Sensing*, 62:39-45.

GEOEYE. 2006. GeoEye imagery products: IKONOS [Web:] http://www.geoeye.com/products/imagery/ikonos/default.htm [Date of use: 7 March 2007]

GEORGE, K., ZISKA, L.H., BUNCE, J.A. & QUEBEDEAUX, B. 2007. Elevated atmospheric CO_2 concentration and temperature across an urban–rural transect. *Atmospheric Environment,* 41:7654-7665.

GERMISHUIZEN, G., MEYER, N.L., STEENKAMP, Y. & KEITH, M. *eds*. 2006. A checklist of South African plants. Southern African Botanical Diversity Network Report No. 41. Pretoria: SABONET. 1126 p.

GILFEDDER, L. & KIRKPATRICK, J. B. 1998. Factors influencing the integrity of remnant bushland in subhumid Tasmania. *Biological Conservation,* 84:89-96.

GILL, S.E., HANDLEY, J.F., ENNOS, A.R., PAULEIT, S., THEURAY, N. & LINDLEY, S.J. 2008. Characterising the urban environment of UK cities and towns: a template for landscape planning. *Landscape and Urban Planning,* 87:210-222.

GLUCH, R. 2002. Urban growth detection using merged Landsat TM and SPOT-P data. *Photogrammetric Engineering and Remote Sensing*, 68(12):1283-1288.

GOBAT, J., ARAGNO, M. & MATTHEY, W. 2004. The living soil: fundamentals of soil science and soil biology. Enfield: Science Publishers, Inc. 602 p.

GODEFROID, S. & KOEDAM, N. 2003. Distribution pattern of the flora in a peri-urban forest: an effect of the city-forest ecotone. *Landscape and Urban Planning,* 65:169-85.

GONG, P., MARCEAU, D.J. & HOWARTH, P.J. 1992. A comparison of spatial feature extraction algorithms for land-use classification with SPOT HRV data. *Remote Sensing of Environment,* 40:137-151.

GRAHAM, J.H., KRZYSIK, A.J., KOVACIC, D.A., DUDA, J.J., FREEMAN, D.C., EMLEN, J.M., ZAK, J.C., LONG, W.R., WALLACE, M.P., CHAMBERLIN-GRAHAM, C., NUTTER, J.P & BALBACH, H.E. 2009. Species richness, equitability, and abundance of ants in disturbed landscapes. *Ecological Indicators,* 9:866-877.

GREY-ROSS, R., DOWNS, C.T. & KIRKMAN, K. *In press.* Using housing estates as conservation tools: a case study in KwaZulu-Natal, South Africa. *Applied Geography.*

GRIMM, N.B., FAETH, S.H., GOLUBIEWSKI, N.E., REDMAN, C.L., WU, J., BAI, X. & BRIGGS, J.M. 2008. Global change and the ecology of cities. *Science,* 319:756-760.

GRIMM, N.B., GROVE, J.M., PICKETT, S.T.A. & REDMAN, C.L. 2000. Integrated approaches to long-term studies of urban ecological systems. *BioScience,* 50(7):571-584.

GROBLER, C.H., BREDENKAMP, G.J. & BROWN, L.R. 2006. Primary grassland communities of urban open spaces in Gauteng, South Africa. *South African Journal of Botany,* 72:367-377.

GROSS, K.L. 1990. A comparison of methods for estimating seed numbers in the soil. *Journal of Ecology,* 78:1079-1093.

HAHS, A.K. 2006. Measures of urbanisation and the ecology of remnant woodlands along an urban-rural gradient. Melbourne: The University of Melbourne. (Thesis: Ph. D) 264 p.

HAHS, A.K., & MCDONNELL, M.J. 2006. Selecting independent measures to quantify Melbourne's urban-rural gradient. *Landscape and Urban Planning,* 78:435- 448.

HAHS, A.K., & MCDONNELL, M.J. 2007. Composition of the plant community in remnant patches of grassy woodland along an urban-rural gradient in Melbourne, Australia. *Urban Ecosystems,* 10:355-377.

HANSEN, A.J., KNIGHT, R.L., MARZLUFF, J.M., POWELL, S., BROWN, K., GUDE, P.H. & JONES, K. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. *Ecological Applications*, 15(6):1893-1905.

HECKMANN, K.E., MANLEY, P.N. & SCHLESINGER, M.D. 2008. Ecological integrity of remnant montane forests along an urban gradient in the Sierra Nevada. *Forest Ecology and Management*, 255:2453-2466.

HEROLD, M., COUCLELIS, H. & CLARKE, K.C. 2005. The role of spatial metrics in the analysis and modeling of urban land use change. *Computers, Environment and Urban Systems*, 29(4):369-399.

HILL, S.J. & FRENCH, K. 2003. Response of the soil seed-bank of Cumberland Plain woodland to heating. *Austral Ecology*, 28(1):14-22.

HOPE, D., GRIES, C., ZHU, W., FAGAN, W.F., REDMAN, C.L., GRIMM, N., NELSON, A.L., MARTIN, C. & KINZIG, A. 2003. Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Science*, 100:8788-8792.

HUANG, S., WANG, S. & BUDD, W.W. 2009. Sprawl in Taipei's peri-urban zone: responses to spatial planning and implications for adapting global environmental change. *Landscape and Urban Planning*, 90:20-32.

HYGROTECH. 2008. Hygrotech sustainable solutions. [Web:] http://www.hygrotech.com/Contacts.asp [Date of use: 23 March 2008]

INGLADA, J. 2007. Automatic recognition of man-made objects in high resolution optical remote sensing images by SVM classification of geometric image features. *ISPRS Journal of Photogrammetry and Remote Sensing*, 62:236-248.

IRWIN, E.G. & BOCKSTAEL, N.E. 2007. The evolution of urban sprawl: evidence of spatial heterogeneity and increasing land fragmentation. *Proceedings of the National Academy of Sciences of the United States of America (PNAS)*, 104(52):20672-20677.

IVITS, E., CHERLET, M., MEHL, W. & SOMMER, S. 2009. Estimating the ecological status and change of riparian zones in Andalusia assessed by multi-temporal AVHHR datasets. *Ecological Indicators*, 9:422-431.

IWATA, K., EGUCHI, K. & YAMANE, S. 2005. A case study on urban ant fauna of southern Kyusyu, Japan, with notes on a new monitoring protocol (Insecta, Hymenoptera, Formicidae). *Journal of Asia-Pacific Entomology*, 8(3):263-272.

JACQUIN, A., CHÉRET, V., DENUX, J., GAY, M., MITCHLEY, J. & XOFIS, P. 2005. Habitat suitability modelling of Capercaillie (*Tetrao urogallus*) using earth observation data. *Journal for Nature Conservation,* 13:161-169.

JOHNSON, D.E. 1998. Applied multivariate methods for data analysts. Pacific Grove: Duxbury Press. 567 p.

JOLLIFFE, I.T. 2002. Principal components analysis. 2nd ed. New York: Springer-Verlag. 487 p.

KAYE, J.P., GROFFMAN, P.M., GRIMM, N.B., BAKER, L.A. & POUYAT, R.V. 2006. A Distinct Urban Biogeochemistry? *TRENDS in Ecology and Evolution,* 21(4): 192-199.

KEARNS, T. 2006. Remote sensing. (*In* Wade, T. & Sommer, S., *eds*. 2006. A to Z GIS: an illustrated dictionary of geographic information systems. Redlands: ESRI Press. p. 261-265.)

KENT, M. & COKER, P. 1992. Vegetation description and analysis: a practical approach. Chichester: Wiley. 363 p.

KENT, M., STEVENS, R.A. & ZHANG, L. 1999. Urban plant ecology patterns and processes: a case study of the flora of the City of Plymouth, Devon, U.K. *Journal of Biogeography,* 26:1281-1298.

KERR, J.T. & OSTROVSKY, M. 2003. From space to species: ecological applications for remote sensing. *TRENDS in Ecology and Evolution*, 18(6):299-305.

KOSTELL-HUGHES, F., YOUNG, T.P. & MCDONNELL, M.J. 1998. The soil seed bank and its relationship to the aboveground vegetation in deciduous forests in New York City. *Urban Ecosystems,* 2:43-59.

KÜHN, I. & KLOTZ, S. 2006. Urbanization and homogenization – comparing the floras of urban and rural areas in Germany. *Biological Conservation,* 127:292-300.

LAKE, J.C. & LEISHMAN, M.R. 2004. Invasion success of exotic plants in natural ecosystems: the role of disturbance, plant attributes and freedom from herbivores. *Biological Conservation*, 117:215-226.

LEIMGRUBER, P., CHRISTEN, C.A. & LABORDERIE, A. 2005. The impact of Landsat satellite monitoring on conservation biology. *Environmental Monitoring and Assessment*, 106:81-101.

LEMON, A. 2003. Urbanization and urban forms. (*In* Fox, R. & Rowntree, K., *eds*. The geography of South Africa in a changing world. Cape Town: Oxford University Press Southern Africa. p. 186-210.)

LI, M., HUANG, C., ZHU, Z., SHI, H., LU, H. & PENG, S. 2009. Assessing rates of forest change and fragmentation in Alabama, USA, using the vegetation change tracker model. *Forest Ecology and Management*, 257:1480-1488.

LI, H. & WU, Y. 2004. Use and misuse of landscape indices. *Landscape Ecology*, 19:389-399.

LILLESAND, T.M. & KEIFER, R.W. 2000. Remote sensing and image interpretation. 4th ed. New York: Wiley. 724 p.

LINARD, C., PONÇON, N., FONTENILLE, D. & LAMBIN, E.F. 2009. A multi-agent simulation to assess the risk of malaria re-emergence in southern France. *Ecological Modelling*, 220:160-174.

LOW, A.B. & REBELO, A.G. *eds*. 1998. Vegetation of South Africa, Lesotho and Swaziland: a companion to the vegetation map of South Africa, Lesotho and Swaziland. 2nd ed. Pretoria: Department of environmental Affairs & Tourism. 85 p.

LUEDELING, E. & BUERKERT, A. 2008. Typology of oases in northern Oman based on Landsat and SRTM imagery and geological survey data. *Remote Sensing of Environment*, 112:1181-1195.

MAGURRAN, A.E. 1988. Ecological diversity and its measurement. Princeton, New Jersey: Princeton University Press. 179 p.

MAIR, J.M., CUNNINGHAM, S.L., SIBAJA-CORDERO, J.A., GUZMAN, H.M., ARROYO, M.F., MERINO, D. & VARGAS, R. 2009. Mapping benthic faunal communities in the shallow and deep sediments of Las Perlas Archipelago, Pacific Panama. *Marine Pollution Bulletin*, 58:375-383.

MARZLUFF, J.M., SHULENBERGER, E., ENDLICHER, W., SIMON, U., ZUMBRUNNEN, C., ALBERTI, M., BRADLEY, G. & RYAN, C. 2008. An introduction to Urban Ecology as an interaction between humans and nature. (*In* Marzluff, J.M., Shulenberger, E., Endlicher, W., Simon, U., ZumBrunnen, C., Alberti, M., Bradley, G. & Ryan, C., *eds*. Urban Ecology: an international perspective on the interaction between humans and nature. New York: Springer. p. vii-xi.)

MCCONNACHIE, M.M., SHACKLETON, C.M. & MCGREGOR, G.K. 2008. The extent of public green space and alien plant species in 10 small towns of the sub-tropical thicket biome, South Africa. *Urban Forestry and Urban Greening*, 7:1-13.

MCDONNELL, M.J. 2007. Restoring and managing biodiversity in an urbanizing world filled with tensions. *Ecological Management and Restoration*, 8(2):83-84.

MCDONNELL, M.J. & HAHS, A.K. 2008. The use of gradient analysis studies in advancing our understanding of the ecology of urbanizing landscapes: current status and future directions. *Landscape Ecology*, 23:1143-1155.

MCDONNELL, M.J. & PICKETT, S.T.A. 1990. Ecosystem structure and function along urban-rural gradients: an unexploited opportunity for ecology. *Ecology*, 71(4):1232-1237.

MCDONNELL, M.J., PICKETT, S.T.A., GROFFMAN, P., BOHLEN, P., POUYAT, R.V., ZIPPERER, W.C., PARMELEE, R.W., CARREIRO, M.M. & MEDLEY, K. 1997. Ecosystem processes along an urban-to-rural gradient. *Urban Ecosystems,* 1:21-36.

MCGARIGAL, K. & MARKS, B. J. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. Gen. Tech. Rep. PNW-GTR-351. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 122 p.

MCGRANAHAN, G. & SATTERTHWAITE, D. 2003. Urban centres: an assessment of sustainability. *Annual Review of Environmental Resources*, 28:243-274.

MCKINNEY, M.L. 2002. Urbanization, biodiversity, and conservation. *Bioscience,* 52:883-890.

MCKINNEY, M.L. 2008. Effects of urbanization on species richness: a review of plants and animals *Urban Ecosystems*, 11:161-176

MCINTYRE, N.E., KNOWELS-YÁNEZ, K. & HOPE, D. 2000. Urban ecology as an interdisciplinary field: differences in the use of "urban" between the social and natural sciences. *Urban Ecosystems*, 4:5-24.

METROPLAN TOWN AND REGIONAL PLANNERS. 2000. Annexure Southern district council: district perspective. Unpublished report. 128 p.

MILLARD, A. 2008. Semi-natural vegetation and its relationship to designated urban green space at the landscape scale in Leeds, UK. *Landscape Ecology*, 23:1231-1241.

MONTGOMERY, D.C. 2001. Design and analysis of experiments. 5th ed. New York: Wiley. 684 p.

MORGAN, J.W. 1998. Composition and seasonal flux of the soil seed bank of species-rich *Themeda triandra* grasslands in relation to burning history. *Journal of Vegetation Science*, 9(2):145-156.

MUCINA, L. & RUTHERFORD, M.C. *eds*. 2006. The vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19. Pretoria: South African National Biodiversity Institute. 805 p.

MÜLLER, N. 1997. Biotope mapping and nature conservation in cities – Part 1: Background and methods as basis for a pilot study in the urban agglomeration of Tokyo (Yokohama City). *Bulletin Institute of Environmental Science and Technology, Yokohama National University*, 23(1):47-62.

MUNICIPAL DEMARCATION BOARD, 2006. Municipal demarcation board, South Africa. [Web:] http://www.demarcation.org.za/ [Date of use: 12 April 2008]

MURPHY, H.T. & LOVETT-DOUST, J. 2004. Context and connectivity in plant metapopulations and landscape mosaics: does the matrix matter? *OIKOS*, 105:3-14.

MUZSLAY, J. 2006. Calculate Demographics. [Web:] http://arcscripts.esri.com/details.asp?dbid=13866 [Date of use: 14 May 2008]

NIEMELÄ, J. 1999a. Ecology and urban planning. *Biodiversity and Conservation*, 8:119-131.

NIEMELÄ, J. 1999b. Is there a need for a theory of urban ecology? *Urban Ecosystems*, 3:57-65.

NIGGEMANN, M., JETZKOWITZ, J., BRUNZEL, S., WICHMANN, M.C. & BIALOZYT, R. 2009. Distribution patterns of plants explained by human movement behavior. *Ecological Modelling*, 220:1339-1346.

O'CONNOR, T.G. & KUYLER, P. 2009. Impact of land use on the biodiversity integrity of the moist sub-biome of the grassland biome, South Africa. *Journal of Environmental Management*, 90:384-395.

ODGERS, B.M. 1999. Seasonal variation in buried germinable seed banks of grass species in an urban eucalypt forest reserve. *Australian Journal of Botany*, 47:623-638.

OLSEN, L., DALE, V.H. & FOSTER, T. 2007. Landscape patterns as indicators of ecological change at Fort Benning, Georgia, USA. *Landscape and Urban Planning*, 79:137-149.

O'NEILL, R.V., RIITTERS, K.H., WICKHAM, J.D., & JONES, K.B. 1999. Landscape pattern metrics and regional assessment. *Ecosystem Health*, 5(4):225-233.

PACIONE, M. 2005. Urban geography: a global perspective. 2nd ed. NY : Routlegde. 686 p.

PAUCHARD, A., AGUAYO, M., PEÑA, E. & URRUTIA, R. 2006. Multiple effects of urbanization on the biodiversity of developing countries: the case of a fast-growing metropolitan area (Concepción, Chile). *Biological Conservation*, 127:272-281.

PENNINGTON, D.N., HANSEL, J. &BLAIR, R.B. 2008. The conservation value of urban riparian areas for landbirds during spring migration: land cover, scale, and vegetation effects. *Biological Conservation*, 141:1235-1248.

PHINN, S., STANFORD, M., SCARTH, P., MURRAY, A. T. & SHYY, P. T. 2002. Monitoring the composition of urban environments based on the vegetation–impervious surface–soil (VIS) model by subpixel analysis techniques. *International Journal of Remote Sensing*, 23(20):4131-4153.

PHUA, M., TSUYUKI, S., FURUYA, N. & LEE, J.S. 2008. Detecting deforestation with a spectral change detection approach using multitemporal Landsat data: A case study of Kinabalu Park, Sabah, Malaysia. *Journal of Environmental Management*, 88:784-795.

PICKETT, S.T.A., BURCH, W.R., DALTON, S., FORESMAN, T.W., GROVE, J.M. & ROWNTREE, R. 1997. A conceptual framework for the study of human ecosystems in urban areas. *Urban Ecosystems*, 1: 185-199.

PICKETT, S.T.A., CADENASSO, M.L., GROVE, J.M., NILON, C.H., POUYAT, R.V., ZIPPERER, W.C. & COSTANZA, R. 2001. Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecological Systems*, 32:127-157.

PICKETT, S.T.A. & GROVE, J.M. 2009. Urban ecosystems: what would Tansley do? *Urban Ecosystems*, 12(1):1-8.

PICKETT, S.T.A., PARKER, V.T. & FIEDLER, P.L. 1992. The new paradigm in Ecology: Implications for Conservation Biology above the species level. (*In* Fiedler, P.L. & Jain, S.K., *eds*. Conservation

Biology, the theory and practice of nature conservation preservation and management. New York : Chapman & Hall. p. 65-88.)

POSCHLOD, P., BAKKER, J.P. & KAHMEN, S. 2005. Changing land use and its impact on biodiversity. *Basic and Applied Ecology*, 6:93-98.

POST, D.M., DOYLE, M.W., SABO, J.L. & FINLAY, J.C. 2007. The problem of boundaries in defining ecosystems: a potential landmine for uniting geomorphology and ecology. *Geomorphology*, 89:111-126.

POUYAT, R., GROFFMAN, P., YESILONIS, I. & HERNANDEZ, L. 2002. Soil carbon pools and fluxes in urban ecosystems. *Environmental Pollution*, 116:S107-S118.

POUYAT, R.V., YESILONIS, I.D., SZLAVECZ, K., CSUZDI, C., HORNUNG, E., KORSÓS, Z., RUSSELL-ANELLI, J. & GIORGIO, V. 2008. Response of forest soil properties to urbanization gradients in three metropolitan areas. *Landscape Ecology*, 23:1187-1203.

POYNTON, J.C. & ROBERTS, D.C. 1985. Urban open space planning in South Africa: a biogeographical perspective. *South African Journal of Science*, 81:33-37.

QUINN, G.P. & KEOUGH, M.J. 2002. Experimental design and data analysis for biologists. Cambridge: Cambridge University Press. 537 p.

RASHED, T., WEEKS, J.R., GADALLA, M.S. & HILL, A.G. 2001. Revealing the anatomy of cities through spectral mixture analysis of multispectral satellite imagery: a case study of the greater Cairo region, Egypt. *Geocarto International*, 16(4):7-18.

REBELE, F. 1994. Urban ecology and special features of urban ecosystems. *Global Ecology and Biogeography Letters*, 4:173-187.

REJMANEK, M. & RANDALL, J.M. 2004. The total number of naturalized species can be a reliable predictor of the total number of alien pest species. *Diversity and Distributions*, 10:367-369.

RICHARDS, J.A. 1993. Remote sensing digital image analysis: an introduction. 2nd ed. Berlin: Springer-Verlag. 340 p.

ROBERTS, D.C. 1993. The vegetation ecology of municipal Durban, Natal. Floristic classification. *Bothalia*, 23(2):271-326.

ROBERTS, D.C. & POYNTON, J.C. 1985. Central and peripheral urban open spaces: need for biological evaluation. *South African Journal of Science*, 81:464-466.

ROBERTSON, H.A. & JAMES, K.R. 2007. Plant establishment from the seed bank of a degraded floodplain wetland: a comparison of two alternative management scenarios. *Plant Ecology*, 188:145-164.

RÖDER, A., HILL, J., DUGUY, B., ALLOZA, J.A. & VALLEJO, R. 2008. Using long time series of Landsat data to monitor fire events and post-fire dynamics and identify driving factors. A case study in the Ayora region (eastern Spain). *Remote sensing of Environment*, 112:259-273.

ROGAN, J. & CHEN, D. 2004. Remote sensing technology for mapping and monitoring land-cover and land-use change. *Progress in Planning*, 61:301–325.

ROY, D.B., HILL, M. 0. & ROTHERY, P. 1999. Effects of urban land cover on the local species pool in Britain. *Ecography*, 22:507-515.

SATELLITE IMAGING CORPORATION. 2007. SPOT-5 Satellite Images. [Web:] http://www.satimagingcorp.com/gallery-spot5-images.html [Date of use: 6 March 2007]

SCHADEK, U., STRAUSS, B., BIEDERMANN, R. & KLEYER, M. 2009. Plant species richness, vegetation structure and soil resources of urban brownfield sites linked to successional age. *Urban Ecosystems*, 12(2):115-126.

SCHINDLER, S., POIRAZIDIS, K. & WRBKA, T. 2008. Towards a core set of landscape metrics for biodiversity assessments: a case study from Dadia National Park, Greece. *Ecological Indicators*, 8:502-514.

SCHWARTZ, M.W., THORNE, J.H. & VIERS, J.H. 2006. Biotic homogenization of the California flora in urban and urbanizing regions. *Biological Conservation*, 127:282-291.

SHANDAS, V. & ALBERTI, M. 2009. Exploring the role of vegetation fragmentation on aquatic conditions: linking upland with riparian areas in Puget Sound lowland streams. *Landscape and Urban Planning*, 90:66-75.

SHOCHAT, E., WARREN, P.S. & FAETH, S.H. 2006a. Future directions in urban ecology. *TRENDS in Ecology and Evolution*, 21(12):661-662.

SHOCHAT, E., WARREN, P.S., FAETH, S.H., MCINTYRE, N.E. & HOPE, D. 2006b. From patterns to emerging processes in mechanistic urban ecology. *TRENDS in Ecology and Evolution*, 21(4):186-191.

SOLOMON, T.B., SNYMAN, H.A. & SMIT, G.N. 2006. Soil seed bank characteristics in relation to land use systems and distance from water in a semi-arid rangeland of southern Ethiopia. *South African Journal of Botany*, 72:263-271.

SOUTH AFRICA. Department of Constitutional Development and Planning. 1986. Klerksdorp – Orkney – Stilfontein draft guide plan. Pretoria: Government Printer. 119 p.

SOUTHWORTH, J., MUNROE, D. & NAGENDRA, H. 2004. Land cover change and landscape fragmentation-comparing the utility of continuous and discrete analyses for a western Honduras region. *Agriculture, Ecosystems and Environment*, 101:185-205.

SPOT IMAGE. 2005. SPOT satellite technical data. [Web:] http://www.spotimage.fr/automne_modules_files/standard/public/p299_3a1cd2cb59b76fc7se20286a6abb7efesatSPOT_E.pdf [Date of use: 6 March 2007]

STATISTICS SOUTH AFRICA. 2004. Metadata: geography hierarchy and attributes. Report no. 03-02-25 (2001). Version 3: January 2004. [Web:] http://www.statssa.gov.za/census01/html/Geography_Metadata.htm [Date of use: 19 March 2009]

STATSOFT, INC. 2008. STATISTICA (data analysis software system), version 8.0. www.statsoft.com.

STEFANOV, W.L. & NETZBAND, M. 2005. Assessment of ASTER land cover and MODIS NDVI data at multiple scales for ecological characterization of an arid urban center. *Remote Sensing of Environment* 99: 31-43.

STENHOUSE, R.N. 2004. Fragmentation and internal disturbance of native vegetation reserves in the Perth metropolitan area, Western Australia. *Landscape and Urban Planning*, 68:389-401.

STEWART, G.H., IGNATIEVA, M.E., MEURK, C.D. & EARL, R.D. 2004. The re-emergence of indigenous forest in an urban environment, Christchurch, New Zealand. *Urban Forestry and Urban Greening*, 2:149-158.

STRALBERG, D., FEHRING, K.E., POMARA, L.Y., NUR, N., ADAMS, D.B., HATCH, D., GEUPEL, G.R. & ALLEN, S. 2009. Modeling nest-site occurrence for the Northern Spotted Owl at its southern range limit in central California. *Landscape and Urban Planning*, 90:76-85.

SUKOPP, H. 1998. Urban ecology: scientific and practical aspects. (*In* Breuste, J., Feldmann, H. & Uhlmann, O., *eds*. Urban ecology. Berlin: Springer-Verlag. p 3-16.)

SUKOPP, H. 2002. On the early history of urban ecology in Europe. *Preslia Praha*, 74(4):373-393.

SUKOPP, H. 2004. Human-caused impact on preserved vegetation. *Landscape and Urban Planning*, 68:347–355.

SUTTON, P.C. 2003. A scale-adjusted measure of "urban sprawl" using nighttime satellite imagery. *Remote Sensing of Environment*, 86:353-369.

TABACHNICK, B.G. & FIDELL, L.S. 2001. Using multivariate statistics. 4th ed. Boston: Allyn and Bacon. 966 p.

TAINTON, N.M. 1999. The ecology of the main grazing lands of South Africa (*In* Tainton, N.M., *ed*. 1999. Veld management in South Africa. Pietermaritzburg: University of Natal Press. p. 23-53.)

TAVERNIA, B.G. & REED, J.M. *In press*. Spatial extent and habitat context influence the nature and strength of relationships between urbanization measures. *Landscape and Urban Planning*.

TER HEERDT, G.N.J., SCHUTTER, A. & BAKKER, J.P. 1999. The effect of water supply on seed-bank analysis using the seedling emergence method. *Functional Ecology*, 14:428-430.

TER HEERDT, G.N.J., VERWEIJ, G.L., BEKKER, R.M. & BAKKER, J.P. 1996. An improved method for seed-bank analysis: seedling emergence after removing the soil by sieving. *Functional Ecology*, 10:144-151.

THEOBALD, D.M. 2004. Placing exurban land-use change in a human modification framework. *Frontiers in Ecology and the Environment*, 2:139-144.

THE SATELLITE ENCYCLOPEDIA. 2008. Alphabetical satellite index. [Web:] http://www.tbs-satellite.com/tse/online/index_alpha.html [Date of use: 22 Nov 2008]

THODE, H.C. 2002. Testing for normality. New York: Marcel Dekker, Inc. 479 p.

THOMPSON, L.M. & TROEH, F.R. 1978. Soils and soil fertility. 4th ed. New York: McGraw-Hill Book Company. 516 p.

TRATALOS, J., FULLER, R.A., WARREN, P.H., DAVIES, R.G. & GASTON, K.J. 2007. Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning*, 83:308-317.

UNITED NATIONS. 2008. World urbanization prospects: the 2007 revision. Executive Summary. [Web:] http://www.un.org/esa/population/publications/wup2007/2007WUP_ExecSum_web.pdf [Date of use: 20 Feb 2008]

VALLET, J., DANIEL, H., BEAUJOUAN, V. & ROZÉ, F. 2008. Plant species response to urbanization: comparison of isolated woodland patches in two cities of North-Western France. *Landscape Ecology*, 23:1205-1217.

VAN DEN BERG, L. 2002. The evaluation of a number of technologies for the restoration of degraded rangelands in selected arid and semi-arid regions of South Africa. Potchefstroom: North-West University. (Dissertation: M. Sc) 179 p.

VAN HEEZIK, Y., SMYTH, A. & MATHIEU, R. 2008. Diversity of native and exotic birds across and urban gradient in a New Zealand city. *Landscape and Urban Planning*, 87:223-232.

VAN OUDTSHOORN, F. 2004. Gids tot Grasse van Suider Afrika. 2de uitgawe. Pretoria: Briza publikasies. 288 p.

VAN WILGEN, B.W., REYERS, B., LE MAITRE, D.C., RICHARDSON, D.M. & SCHONEGEVEL, L. 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Journal of Environmental Management*, 89:336-349.

VAN WYK, E., CILLIERS, S.S. & BREDENKAMP, G.J. 1997. Ranteveldplantegroei van Klerksdorp, Noordwes Provinsie, Suid-Afrika. *Suid-Afrikaanse Tydskrif vir Natuurwetenskap en Tegnologie*, 16(2):74-85.

VAN WYK, E., CILLIERS, S.S. & BREDENKAMP, G.J. 2000. Vegetation analysis of wetlands in the Klerksdorp Municipal Area, North West Province, South Africa. *South African Journal of Botany*, 66(1):52-62.

VÉCRIN, M.P., GRÉVILLIOT, F. & MULLER, S. 2007. The contribution of persistent soil seed banks and flooding to the restoration of alluvial meadows. *Journal for Nature Conservation*, 15:59-69.

WADE, T. & SOMMER, S., *eds*. 2006. A to Z GIS: an illustrated dictionary of geographic information systems. Redlands: ESRI Press. 268 p.

WANIA, A., KÜHN, I. & KLOTZ, S. 2006. Plant richness patterns in agricultural and urban landscapes in central Germany-spatial gradients of species richness. *Landscape and Urban Planning*, 75:97-110.

WARNER, R.M. 2008. Applied statistics: from bivariate through multivariate techniques. Los Angeles: Sage Publications. 1100 p.

WEEKS, J.R., LARSON, D. & RASHED, T. 2003. Contrast or continuum? The creation and application of an urban gradient index using remotely sensed imagery and GIS. Paper presented at the annual meeting of the population association of America, Minneapolis, p. 1-44.

WENG, Y. 2007. Spatiotemporal changes of landscape pattern in response to urbanization. *Landscape and Urban Planning*, 81:341-353.

WHITE, R.E. 1997. Principles and practice of soil science: the soil as a natural resource. 3rd ed. Oxford: Blackwell Science Ltd. 348 p.

WILCKE, W., KRAUSS, M., SAFRONOV, G., FOKIN, A.D. & KAUPENJOHANN, M. 2006. Polychlorinated biphenyls (PCBs) in soils of the Moscow region: concentrations and small-scale distribution along an urban-rural transect. *Environmental Pollution*, 141:327-335.

WILLIAMS, D. 2008. Landsat 7 Science data users handbook. [Web:] http://landsathandbook.gsfc.nasa.gov/handbook/handbook_toc.html [Date of use: 10 March 2009]

WILLIAMS, N.S.G. 2007. Environmental, landscape and social predictors of native grassland loss in western Victoria, Australia. *Biological Conservation*, 137:308-318.

WILLIAMS, N.S.G, MCDONNELL, M.J. & SEAGER, E.J. 2005. Factors influencing the loss of an endangered ecosystem in an urbanising landscape: a case study of native grasslands from Melbourne, Australia. *Landscape and Urban Planning*, 71:35-49.

WILLIAMS, N.S.G., MORGAN, J.W., MCCARTHY, M.A. & MCDONNELL, M.J. 2006. Local extinction of grassland plants: the landscape matrix is more important than patch attributes. *Ecology*, 87(12):3000-3006.

WILLIAMS, N.S.G., SCHWARTZ, M.W., VESK, P.A., MCCARTHY, M.A., HAHS, A.K., CLEMANTS, S.E., CORLETT, R.T., DUNCAN, R.P., NORTON, B.A., THOMPSON, K. & MCDONNELL, M.J. 2009. A conceptual framework for predicting the effects of urban environments on floras. *Journal of Ecology*, 97:4-9.

WIMBERLY, M.C. & REILLY, M.J. 2007. Assessment of fire severity and species diversity in the southern Appalachians using Landsat TM and ETM+ imagery. *Remote Sensing of Environment*, 108:189-197.

WU, C., XIAO, Q. & MCPHERSON, E.G. 2008. A method for locating potential tree-planting sites in urban areas: a case study of Los Angeles, USA. *Urban Forestry and Urban Greening*, 7:65-76.

ZHAO, Y., ZANG, G., ZEPP, H. & YANG, J. 2007. Establishing a spatial grouping base for surface soil properties along urban-rural gradient – a case study in Nanjing China. *Catena*, 69:74-81.

Appendix A: Notation used in the landscape metrics equations

Table A.1: List of the relevant notation used in the landscape metrics Equations 3 – 10 (Chapter 4), as described by McGarigal and Marks (1995).

Term	Definition
Subscripts	
i	1, … , m or m' patch types (classes)
j	1, … , n patches
Symbols	
A	Total landscape area (m^2)
a_{ij}	Area (m^2) of patch ij
p_{ij}	Perimeter (m) of patch ij
E'	Total length (m) of edge in landscape; includes entire landscape boundary and background edge segments regardless of whether they represent true edge
N	Total number of patches in the landscape, excluding any background patches
m	Number of patch types (classes) present in the landscape, excluding the landscape border if present
P_i	Proportion of the landscape occupied by patch type (class) i

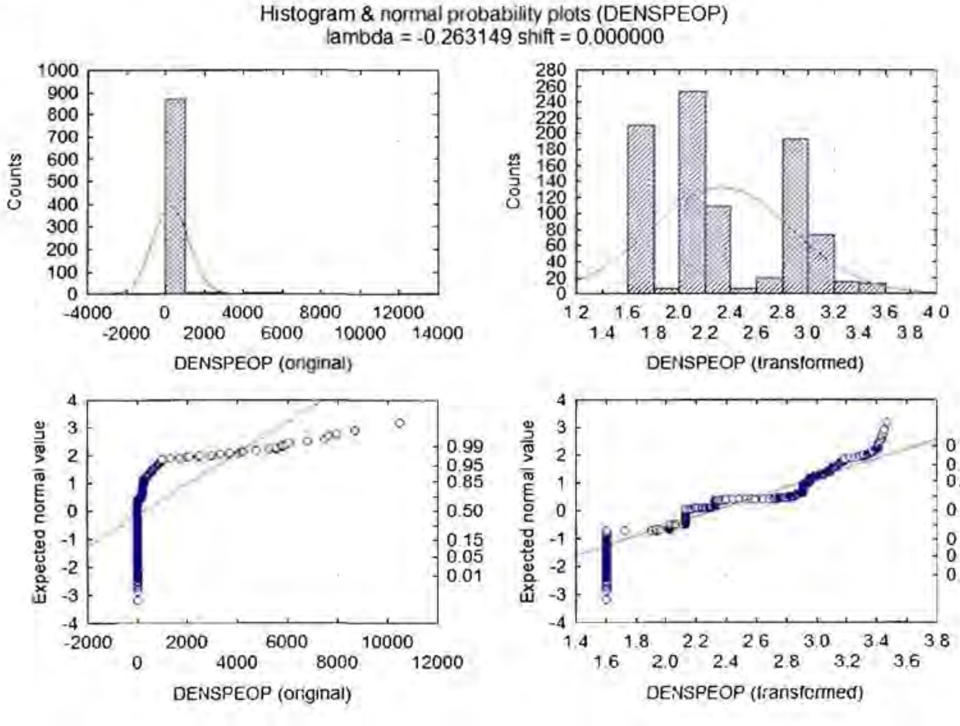
Appendix B: Box-Cox transformations (normality)

Table B.1: The Box-Cox transformations with the individual formulas as listed for each urbanization measure to achieve maximized normality of their value distributions. Also given is the standard deviation (SD), lower confidence limit (LCL), upper confidence limit (UCL) for the individual measures as computed for the transformations. The bold print value of MPFD indicates that the search for optimal lambda did not yield a satisfactory result within the specified range.

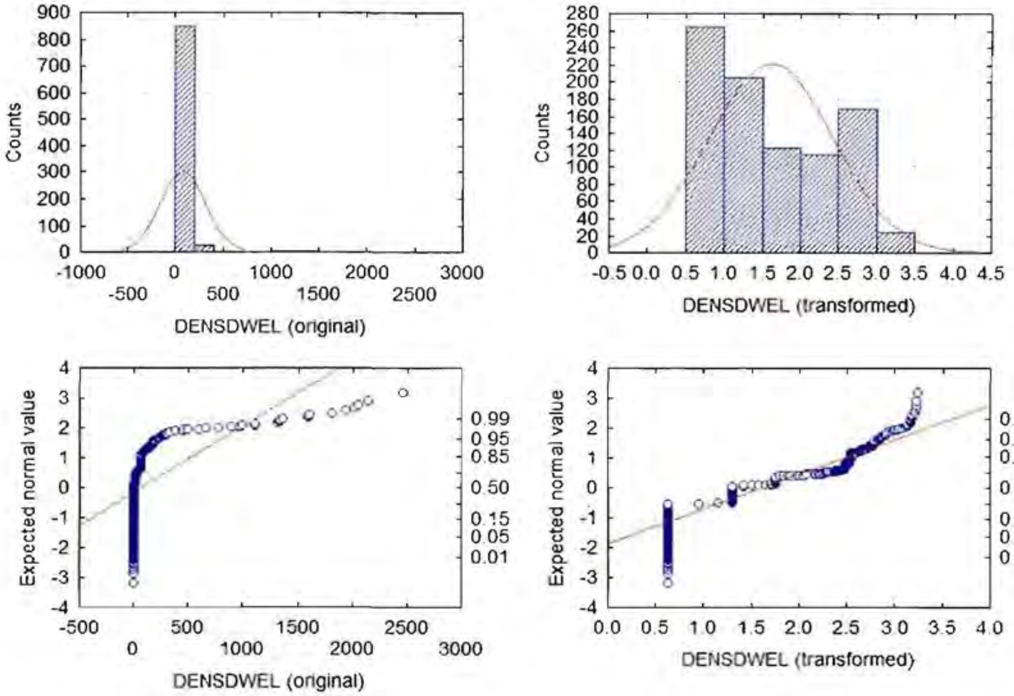
	Lambda	Shift	SD	LCL	UCL	Formula for Box-Cox transformation
DENSPEOP	*-0.26315*	*0.000*	*0.54511*	*-0.309498*	*-0.217739*	*((v11^(-0.263149))-1)/(-0.263149)*
DENSDWEL	*-0.27024*	*0.000*	*0.81087*	*-0.316758*	*-0.224759*	*((v12^(-0.270235))-1)/(-0.270235)*
INDEXCEN	*0.37222*	*0.000*	*1.78982*	*0.245608*	*0.498819*	*((v13^(0.372216))-1)/(0.372216)*
RND	*-0.50763*	*1.000*	*0.46482*	*-0.600457*	*-0.417113*	*(((v9+(1.000000))^(-0.507634))-1)/(-0.507634)*
CBDkm	*0.78192*	*0.550*	*3.50511*	*0.661885*	*0.905216*	*(((v2+(0.550000))^(0.781921))-1)/(0.781921)*
LCR	*1.27739*	*0.000*	*1.35352*	*1.021521*	*1.537799*	*((v5^(1.277387))-1)/(1.277387)*
SIDI	*2.55668*	*1.000*	*0.30506*	*2.037318*	*3.081376*	*(((v4+(1.000000))^(2.556676))-1)/(2.556676)*
NP	*-0.14229*	*0.000*	*0.39651*	*-0.205982*	*-0.078088*	*((v3^(-0.142292))-1)/(-0.142292)*
PURBLC	*-0.82345*	*1.000*	*0.41917*	*-0.911973*	*-0.737521*	*(((v1+(1.000000))^(-0.823453))-1)/(-0.823453)*
LSI	*0.12531*	*0.000*	*0.75029*	*0.034590*	*0.216826*	*((v6^(0.125309))-1)/(0.125309)*
LPI	*1.01908*	*0.000*	*23.89391*	*0.860810*	*1.182238*	*((v7^(1.019079))-1)/(1.019079)*
MPFD	***-5.00000***	*0.000*	*0.02773*			*((v8^(-4.999997))-1)/(-4.999997)*

The following graphs are the original and the transformed normal distribution plots for the urbanization measures created in STATISTICA 8.0, as described and used in Chapter 4.

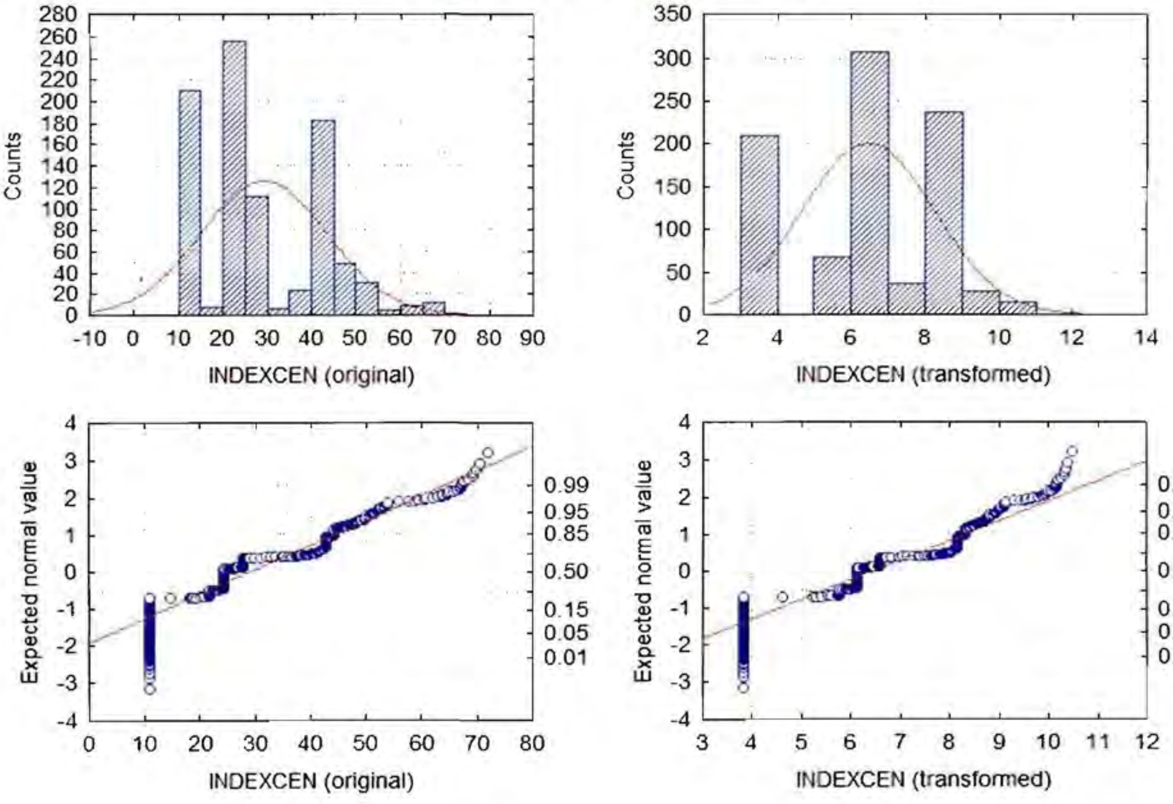
DEMOGRAPHIC- AND PHYSICAL VARIABLES



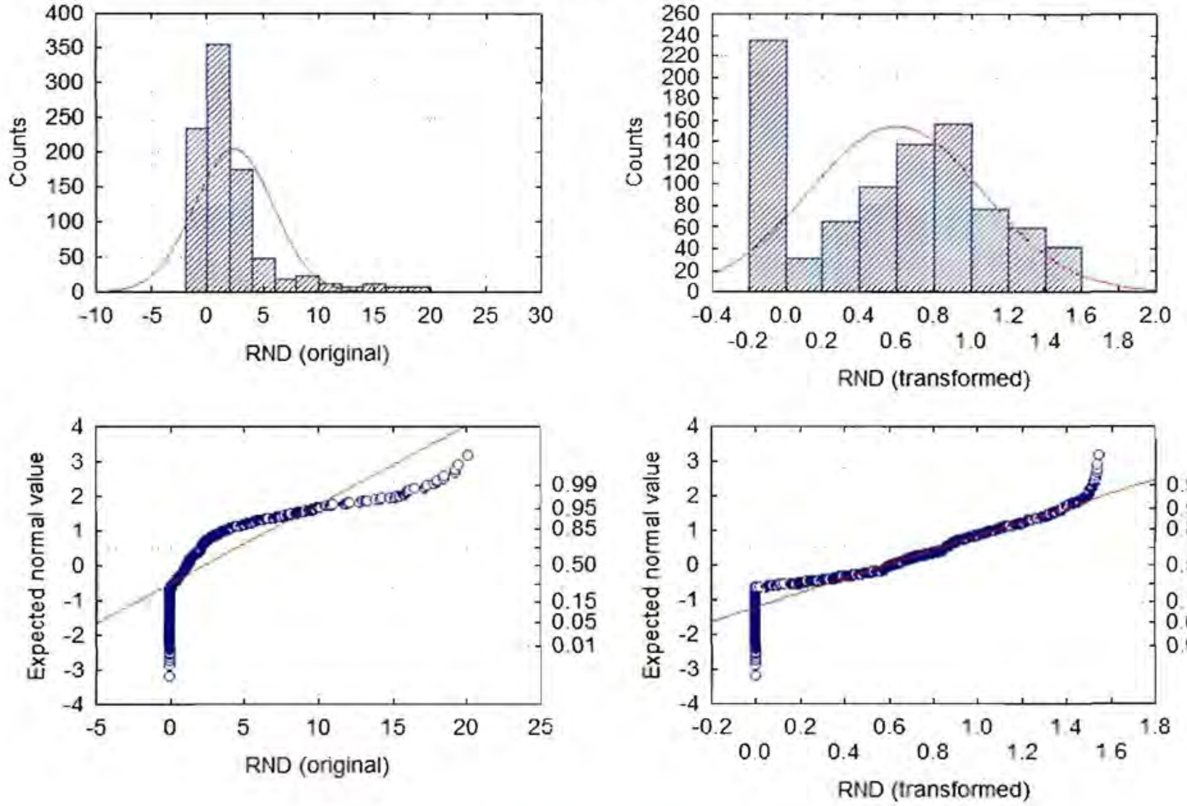
Histogram & normal probability plots (DENSDWEL)
lambda = -0.270235 shift = 0.000000



Histogram & normal probability plots (INDEXCEN)
lambda = 0.372216 shift = 0.000000



Histogram & normal probability plots (RND)
lambda = -0.507634 shift = 1.000000



Histogram & normal probability plots (CBDkm)
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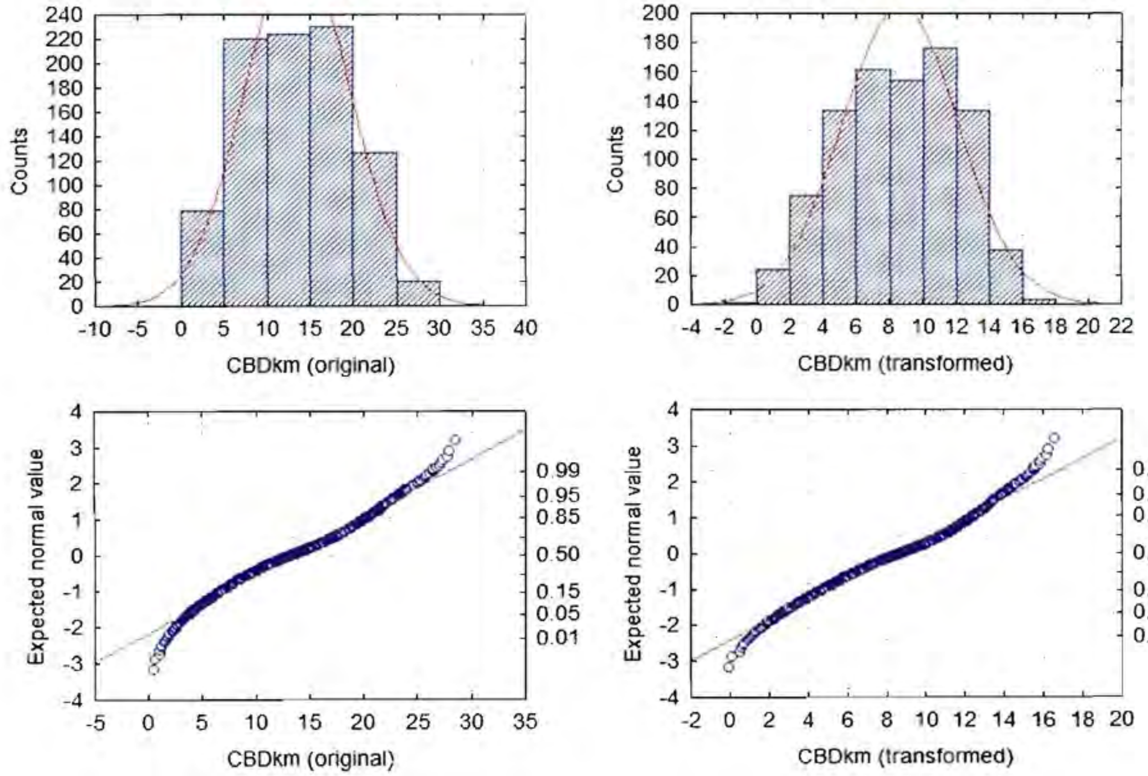

LANDSCAPE METRICS

Histogram & normal probability plots (LCR)
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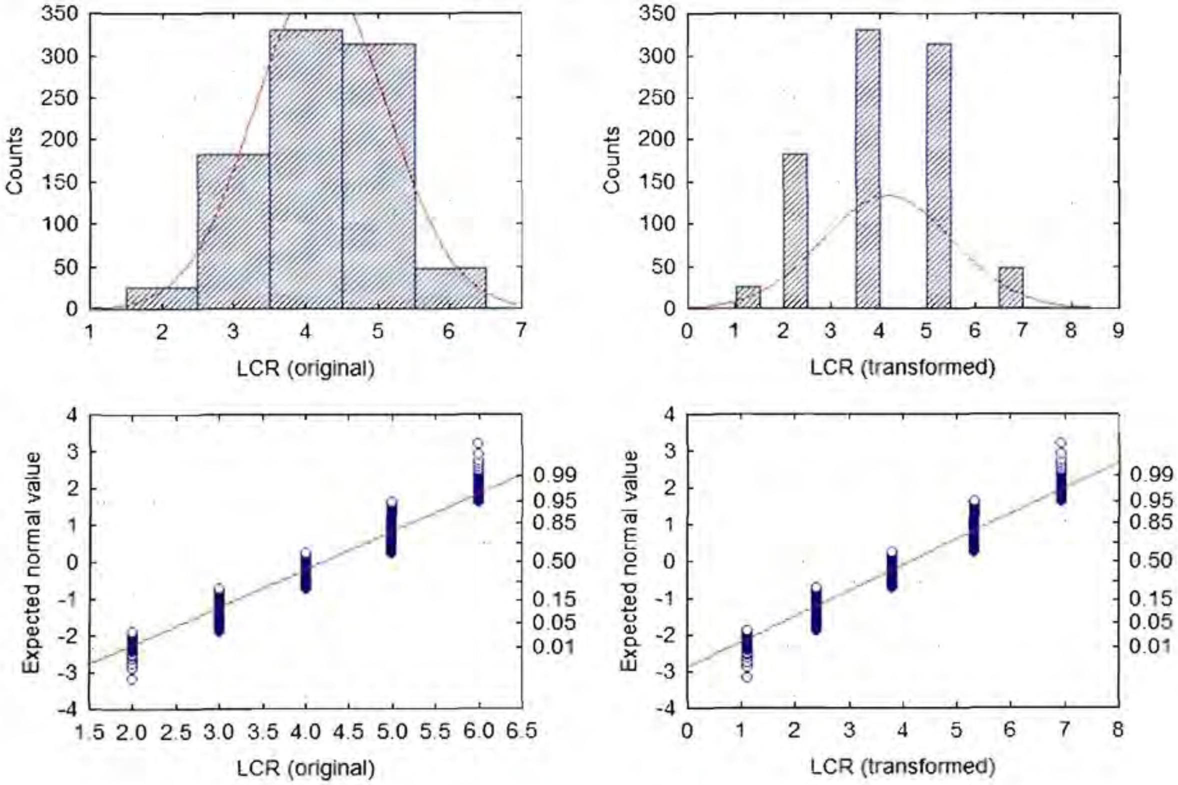

Histogram & normal probability plots (SIDI)
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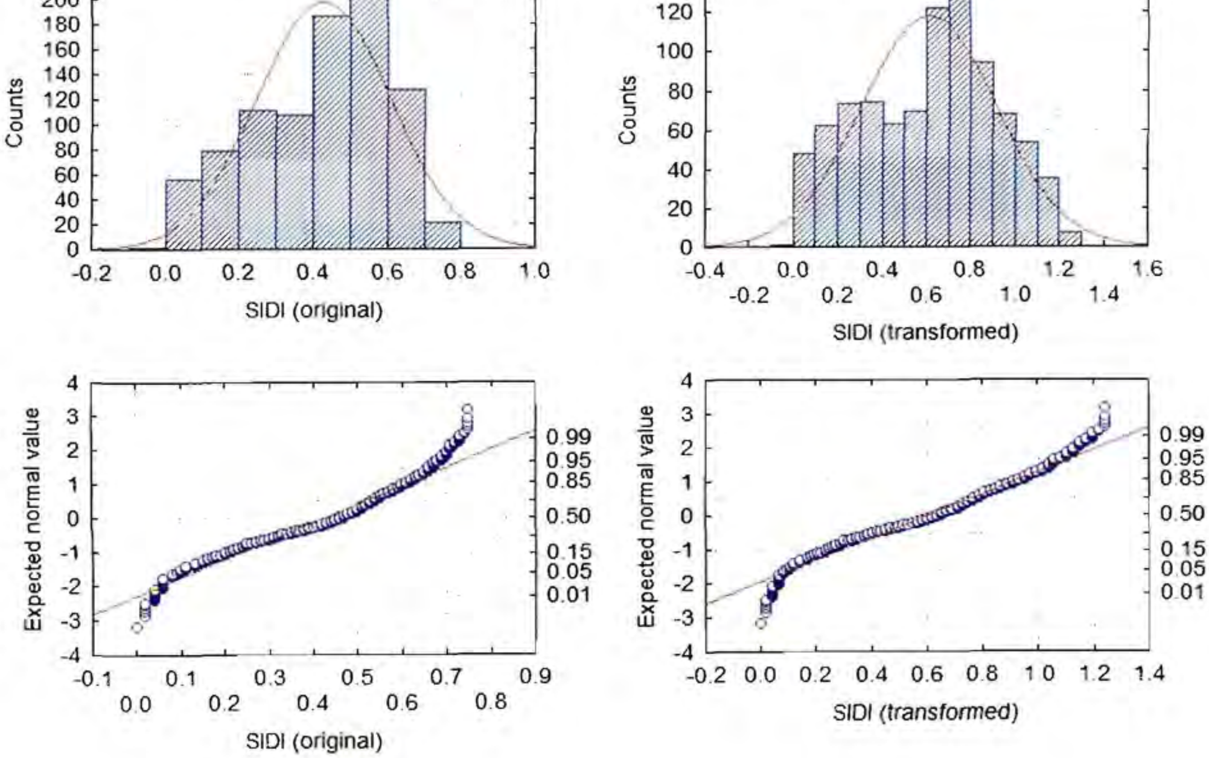
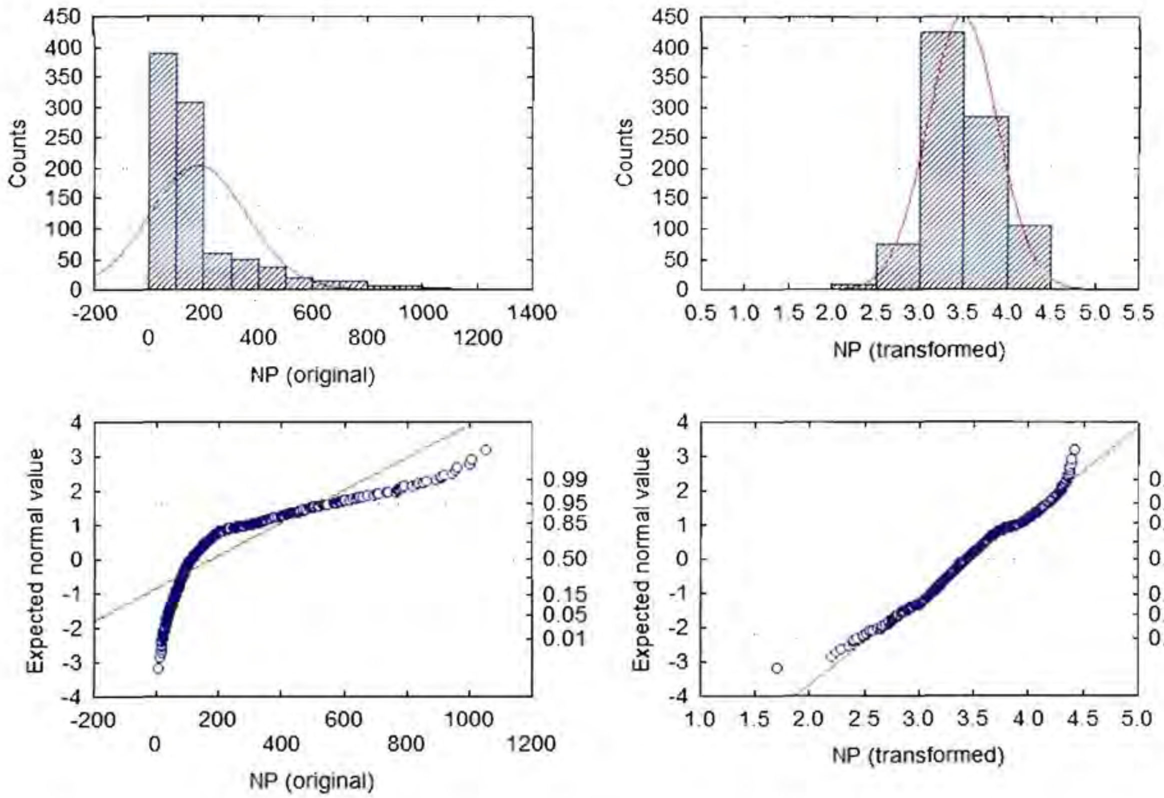


Histogram & normal probability plots (NP)
lambda = -0.142292 shift = 0.000000



Histogram & normal probability plots (PURBLC)
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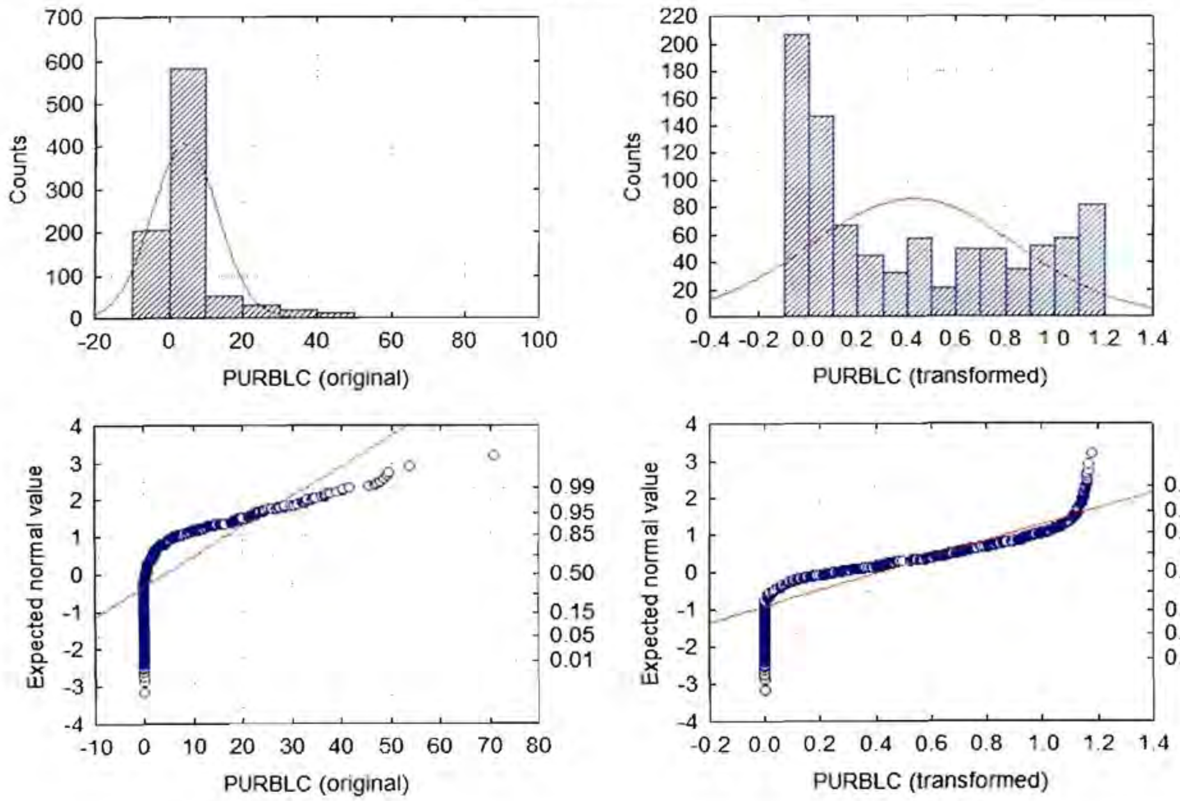
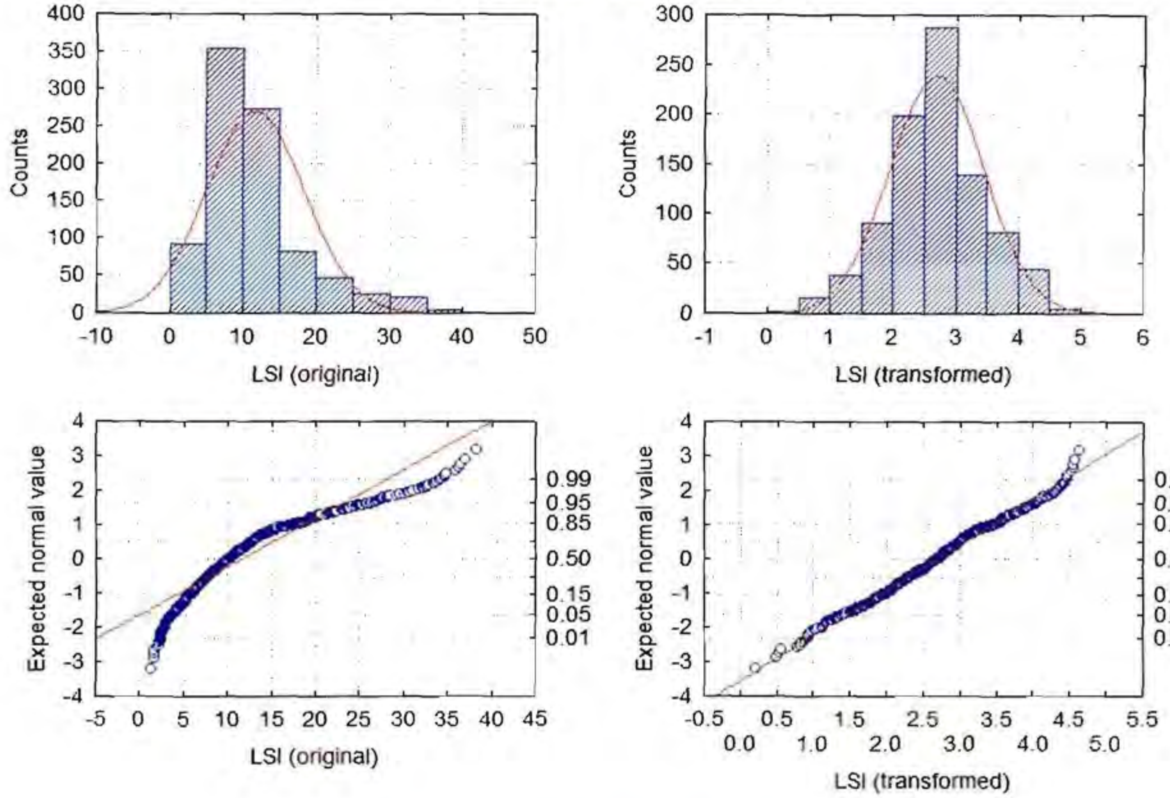


Histogram & normal probability plots (LSI)
lambda = 0.125309 shift = 0.000000



Histogram & normal probability plots (LPI)
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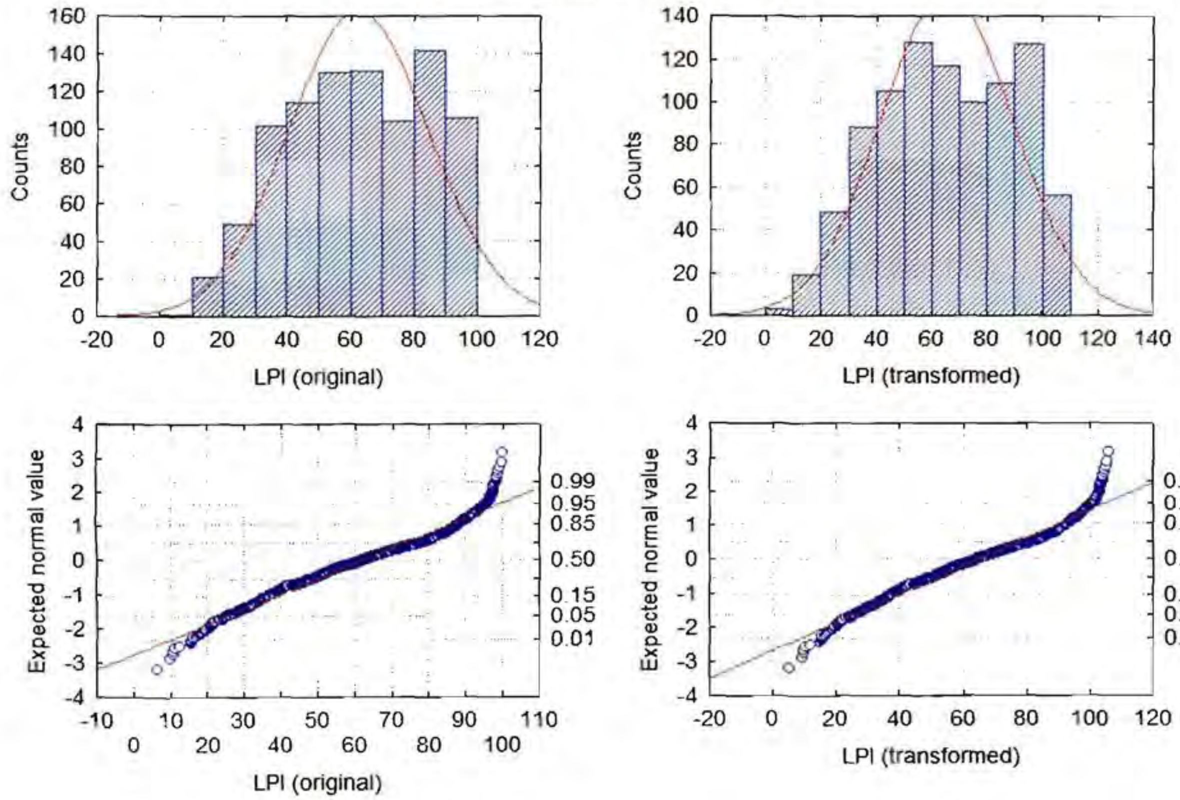

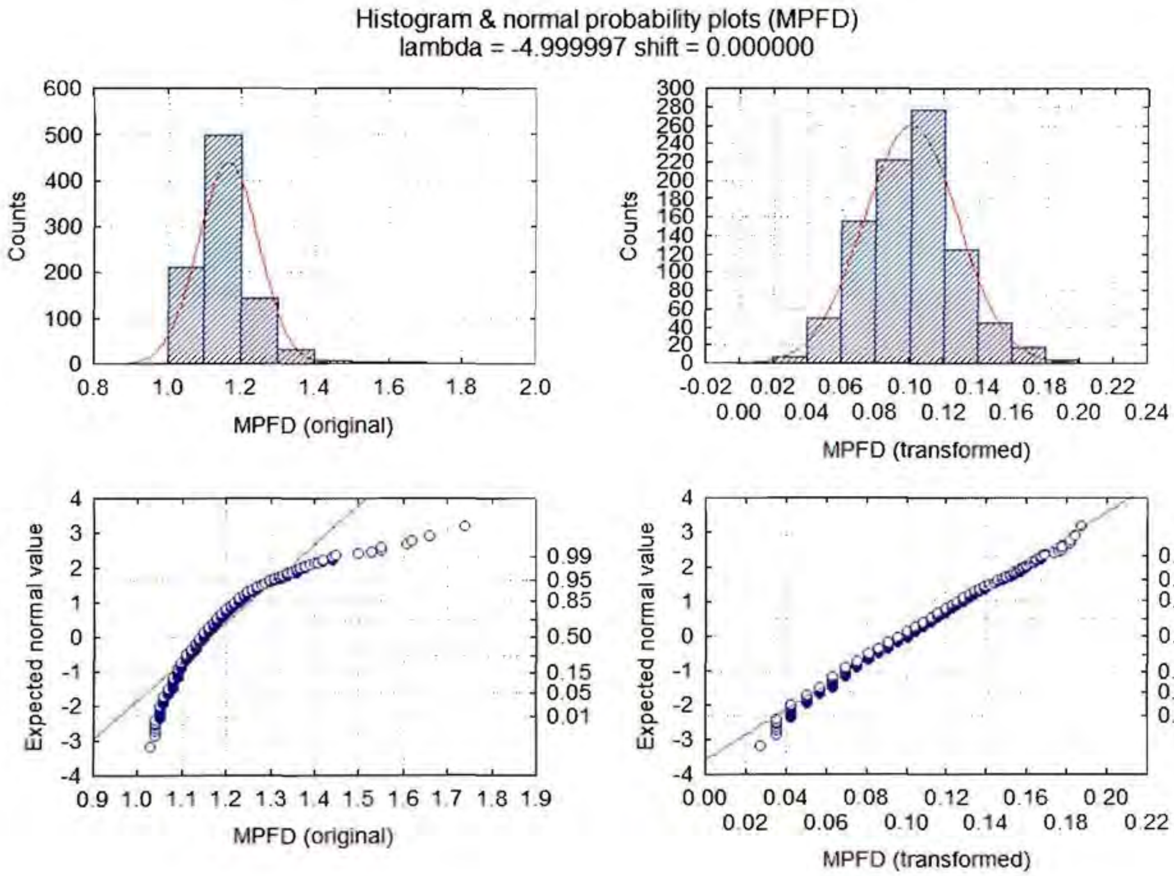
Histogram & normal probability plots (MPFD)
lambda = -4.999997 shift = 0.000000
Counts
MPFD (original)
MPFD (transformed)
Expected normal value
0.99
0.95
0.85
0.50
0.15
0.05
0.01

Appendix C: Maps of the selected grassland patches

Figures C.1 to C.14 are maps of each of the respective selected grassland patches (discussed in Chapter 5) in the landscape, showing the immediate surroundings and the number and location of the sample plots in each patch.

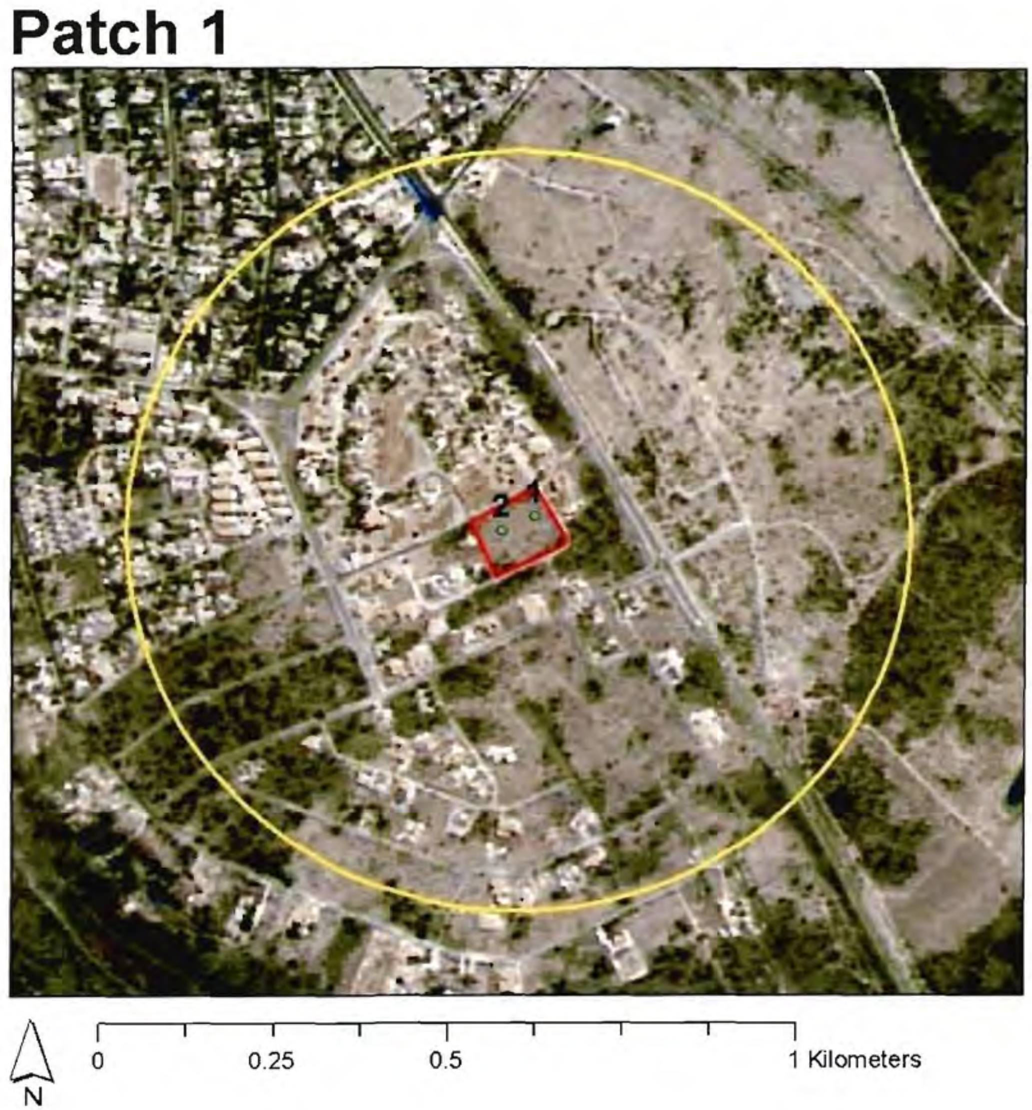

Figure C.1: Patch 1. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 2

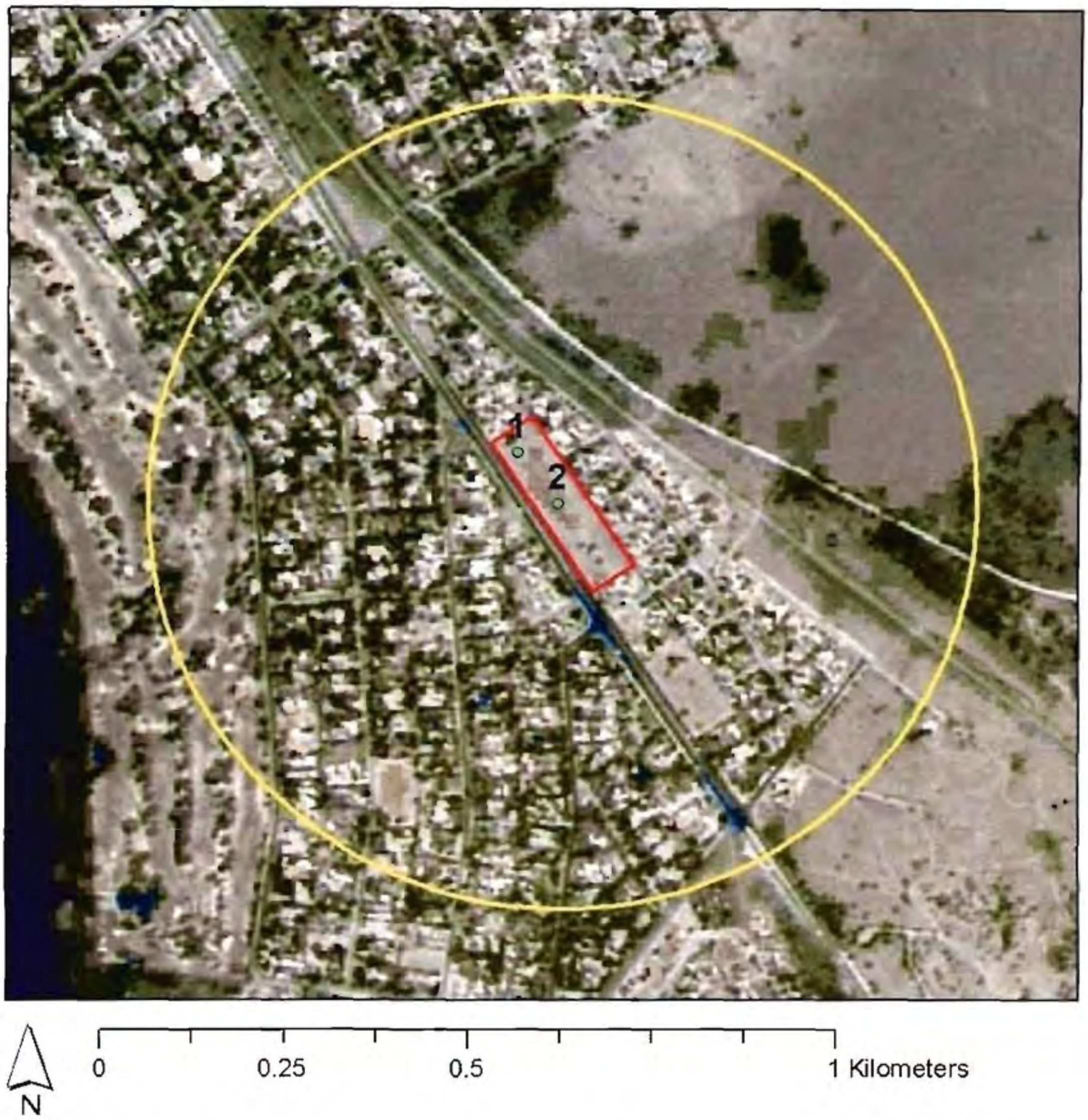

Figure C.2: Patch 2. The outline of the grassland patch is shown in red, and the boundary of the 1 km² area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 3

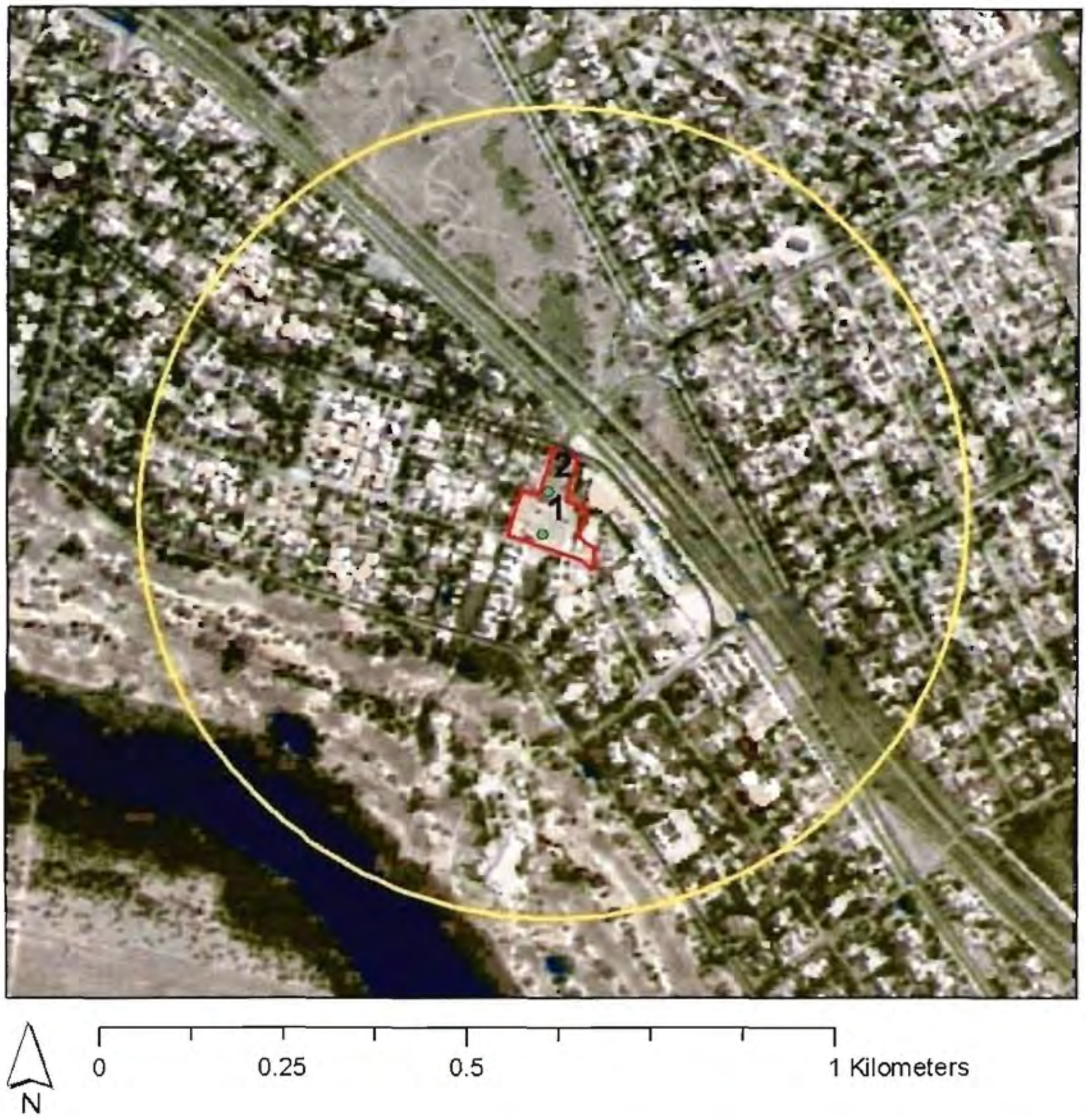

Figure C.3: Patch 3. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 4

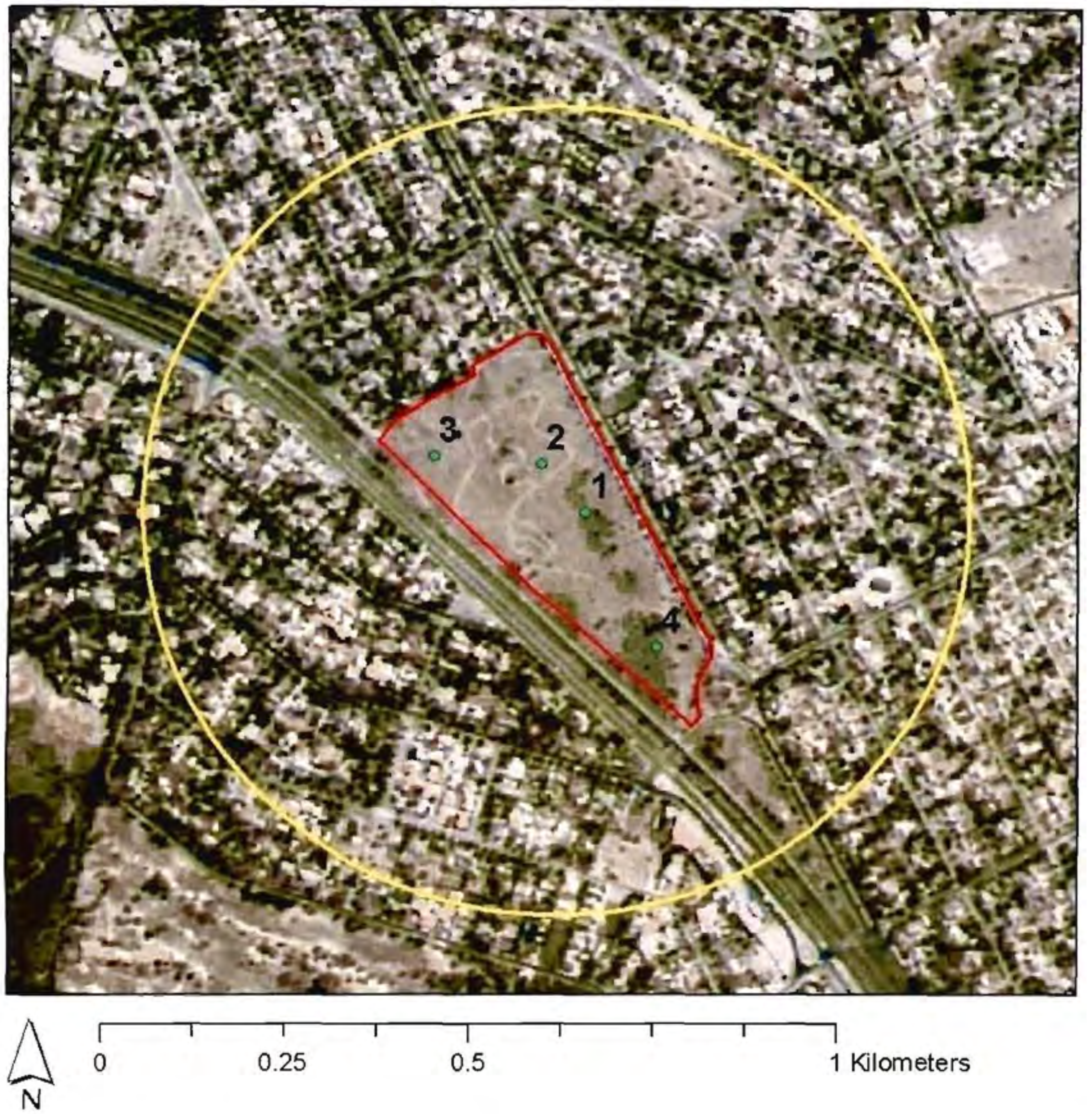

Figure C.4: Patch 4. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 5

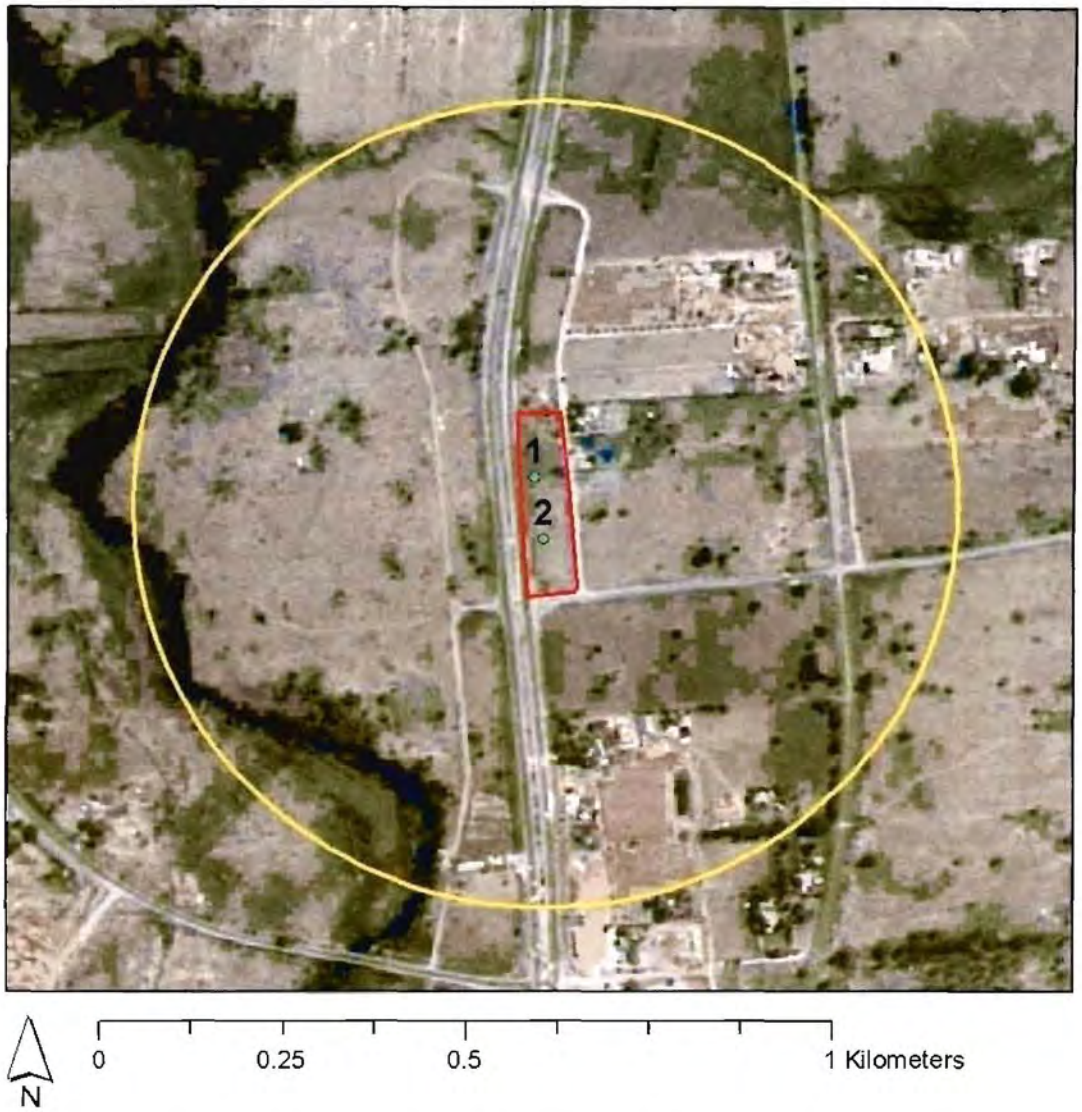

Figure C.5: Patch 5. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 6

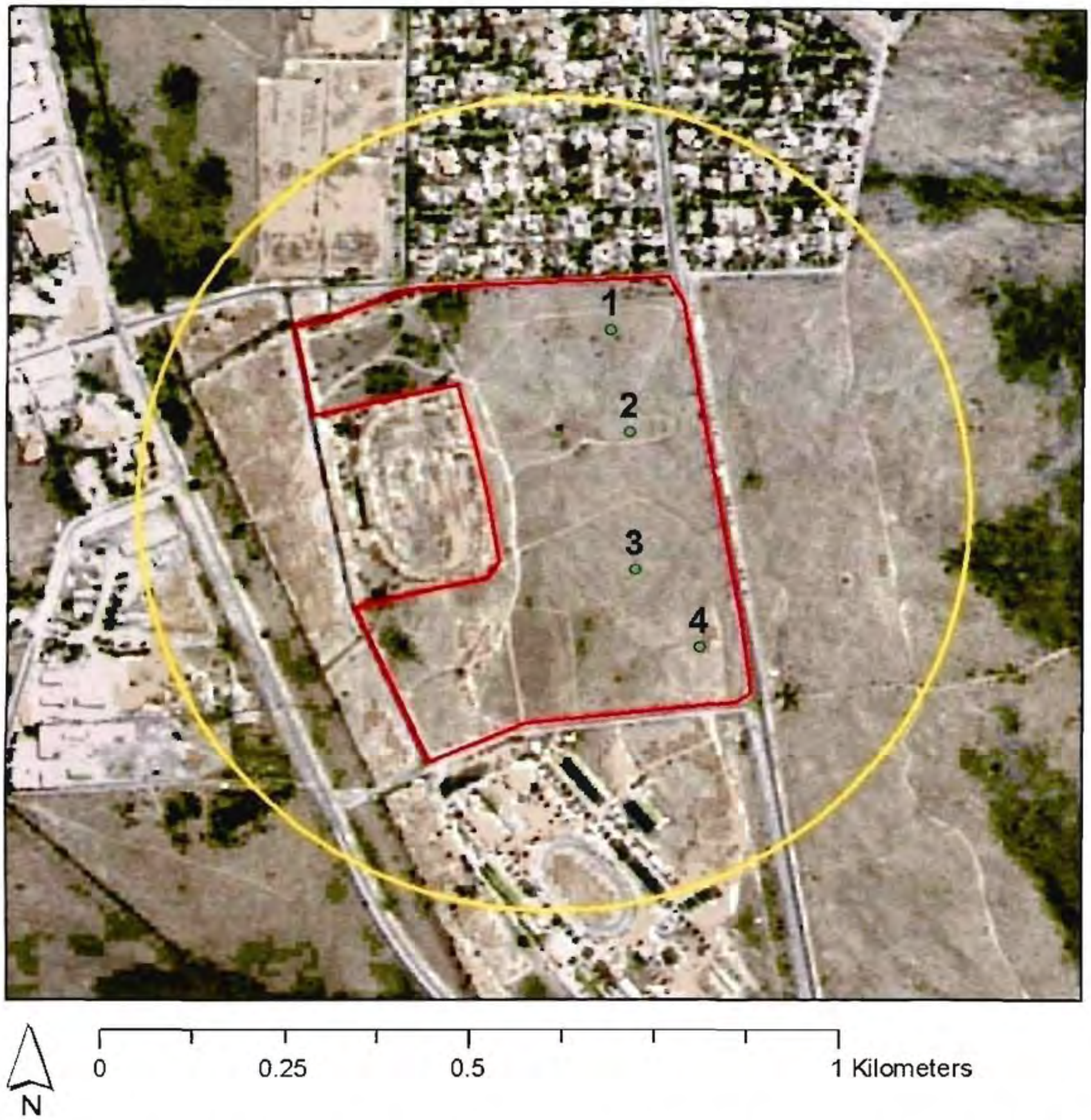

Figure C.6: Patch 6. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 7

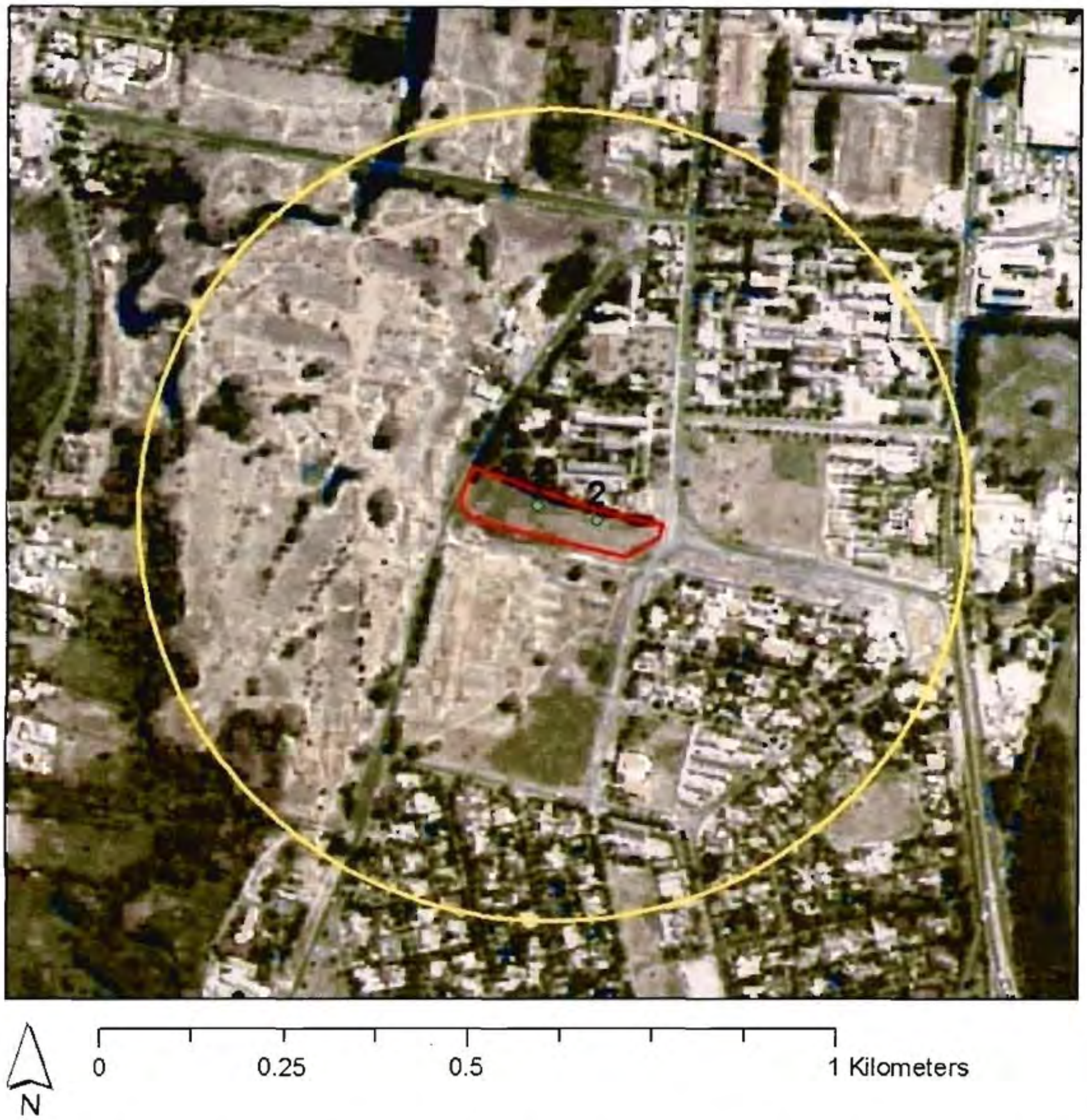

Figure C.7: Patch 7. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 8

Figure C.8: Patch 8. The outline of the grassland patch is shown in red, and the boundary of the 1 km² area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 9

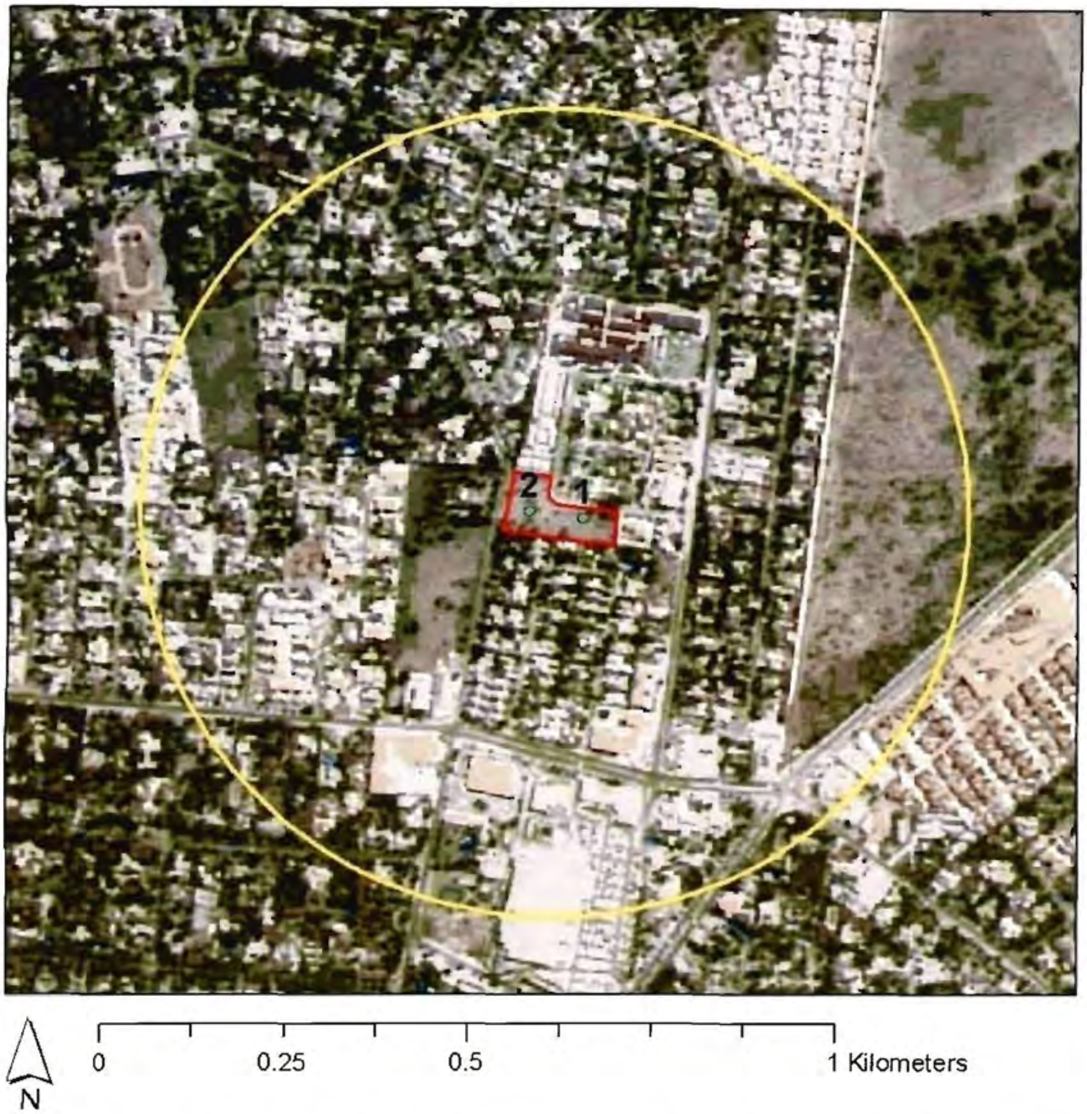

Figure C.9: Patch 9. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 10

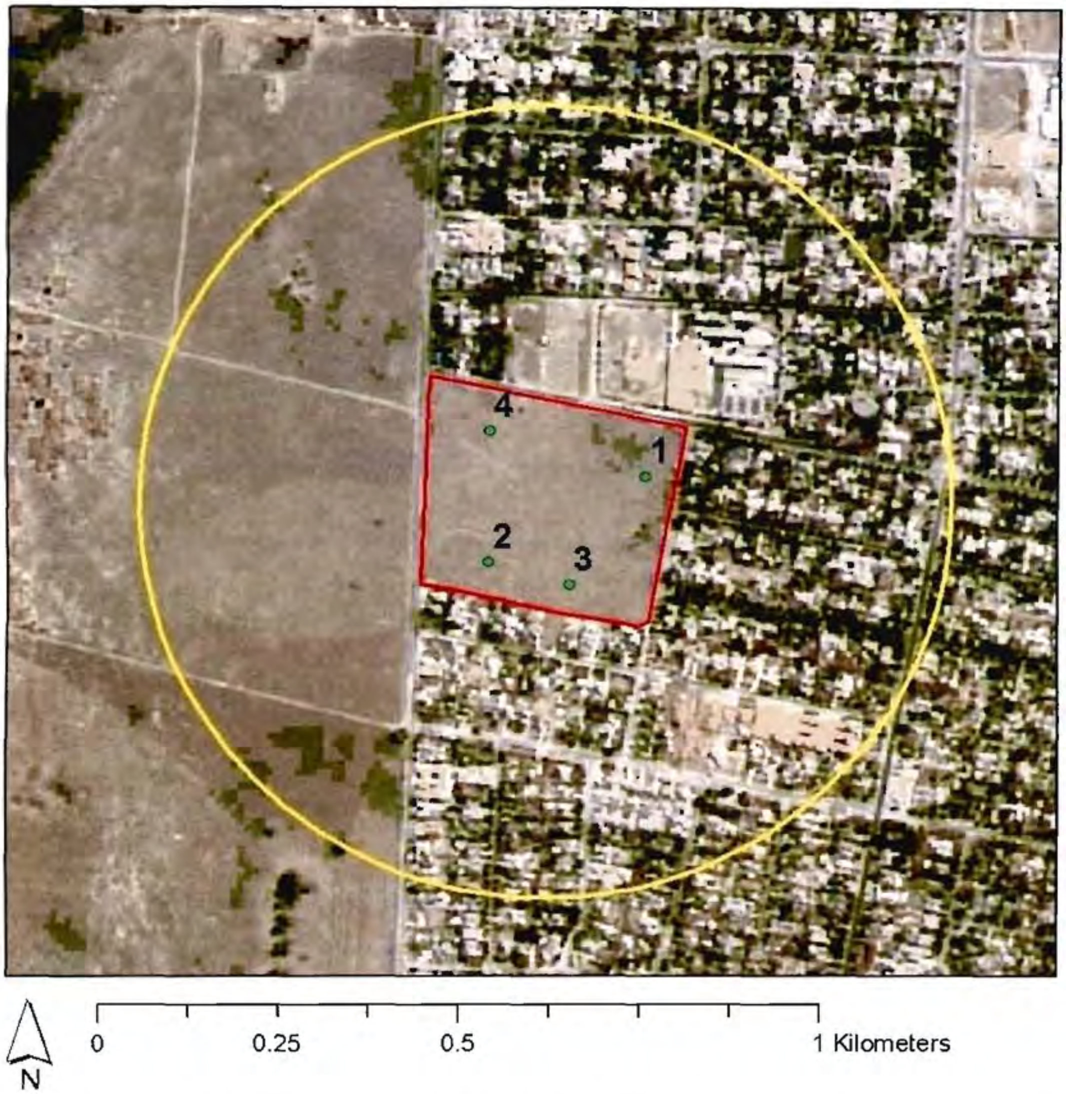

Figure C.10: Patch 10. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 11

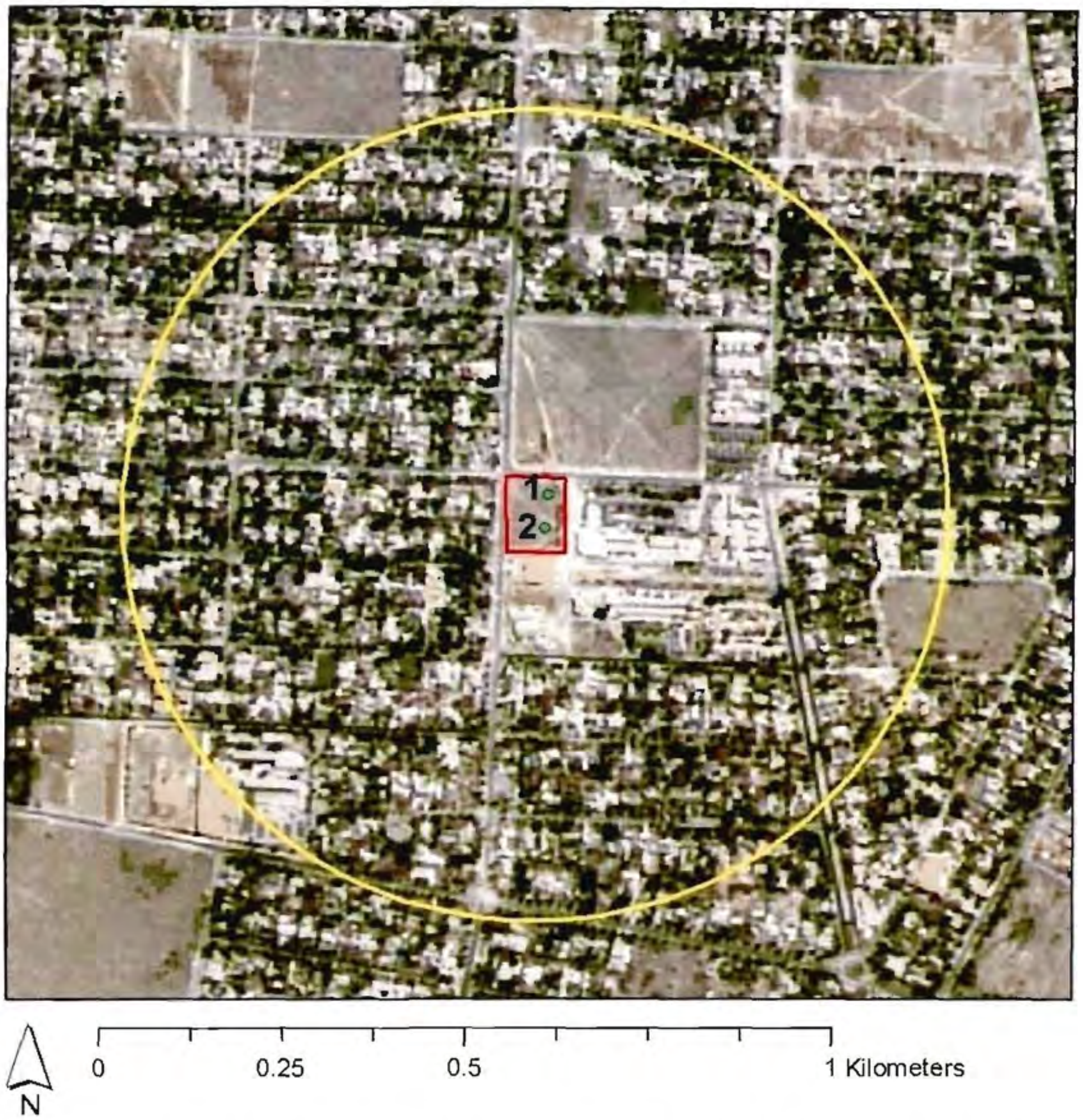

Figure C.11: Patch 11. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

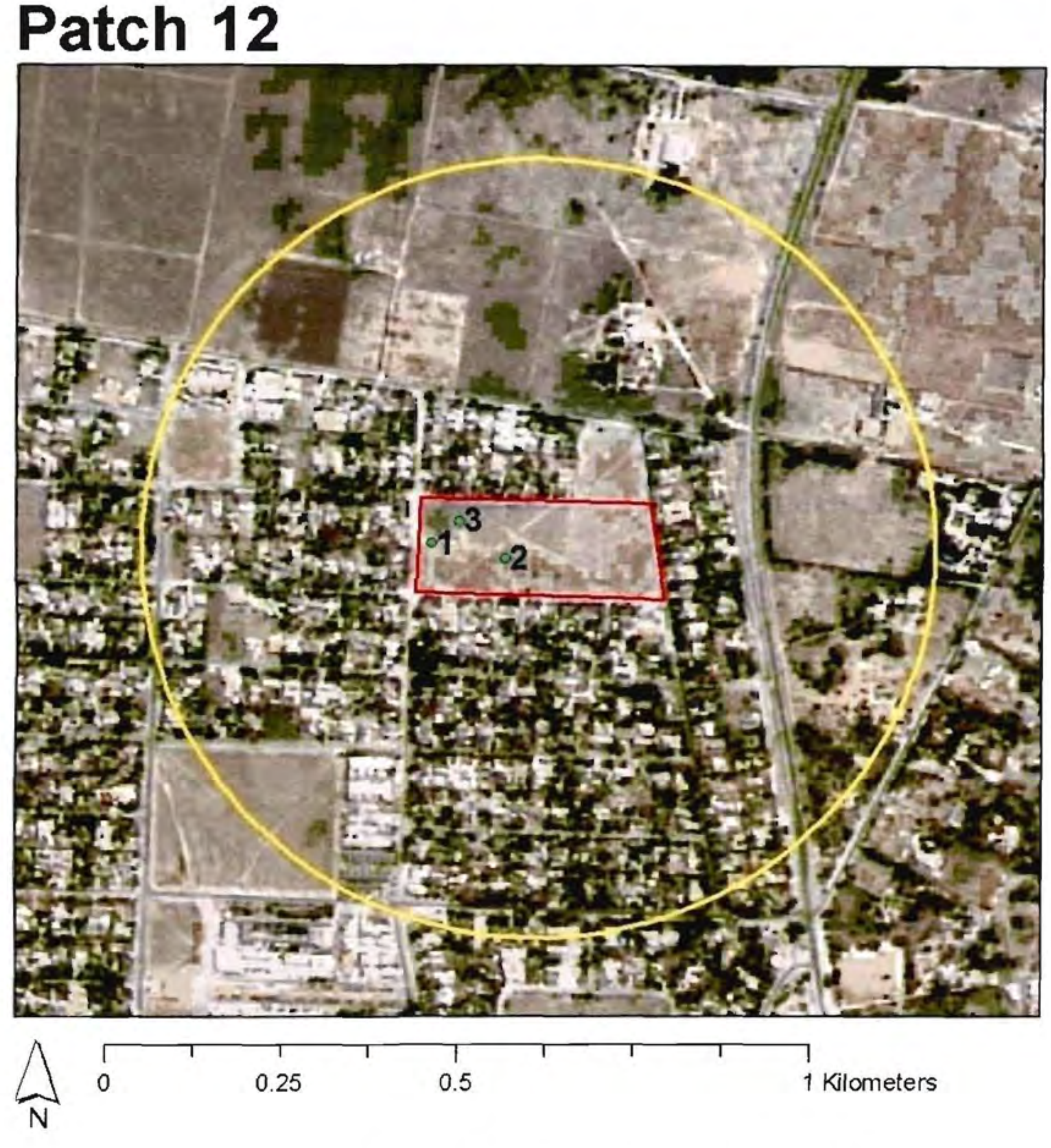

Figure C.12: Patch 12. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 13

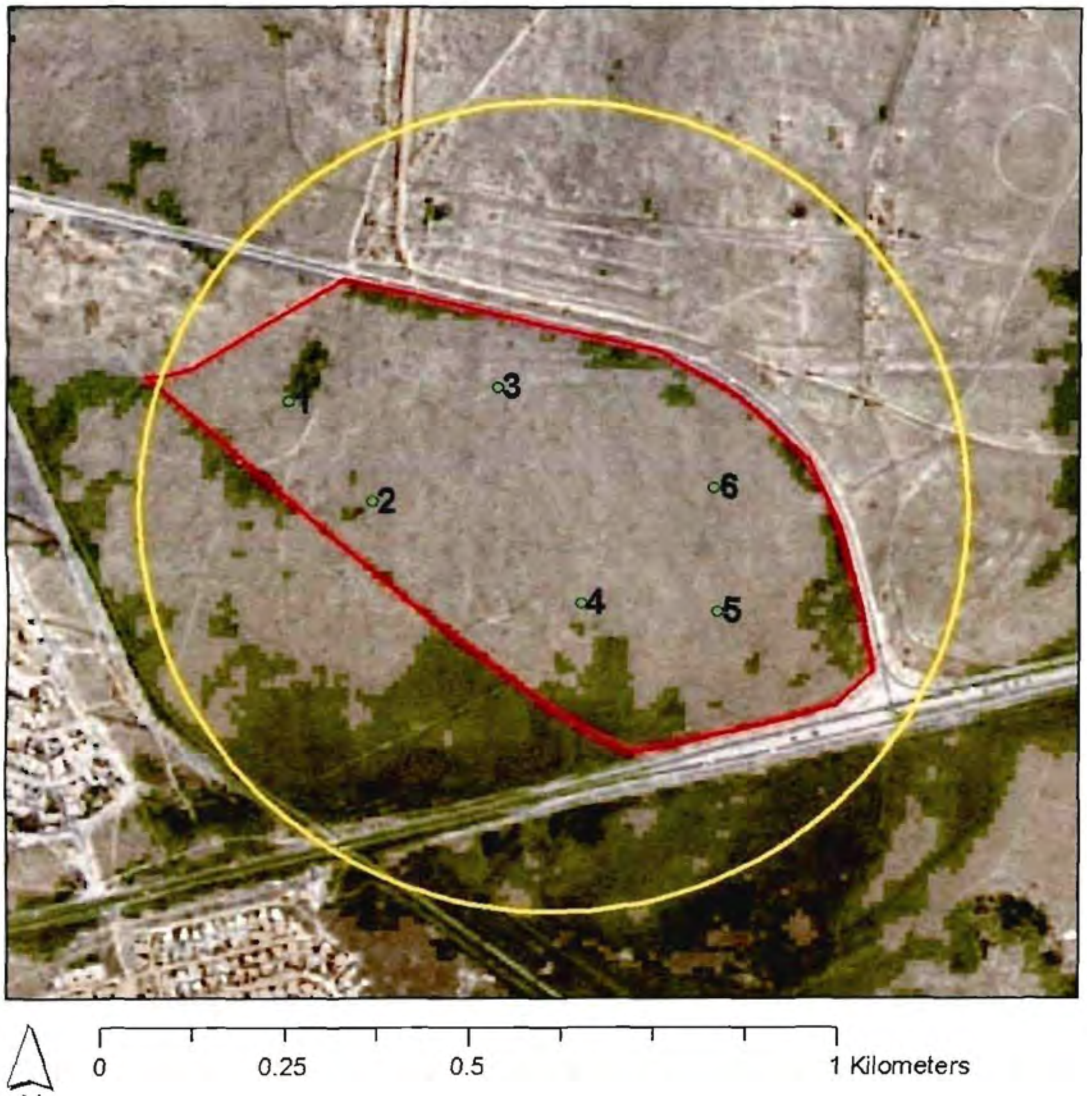

Figure C.13: Patch 13. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Patch 14

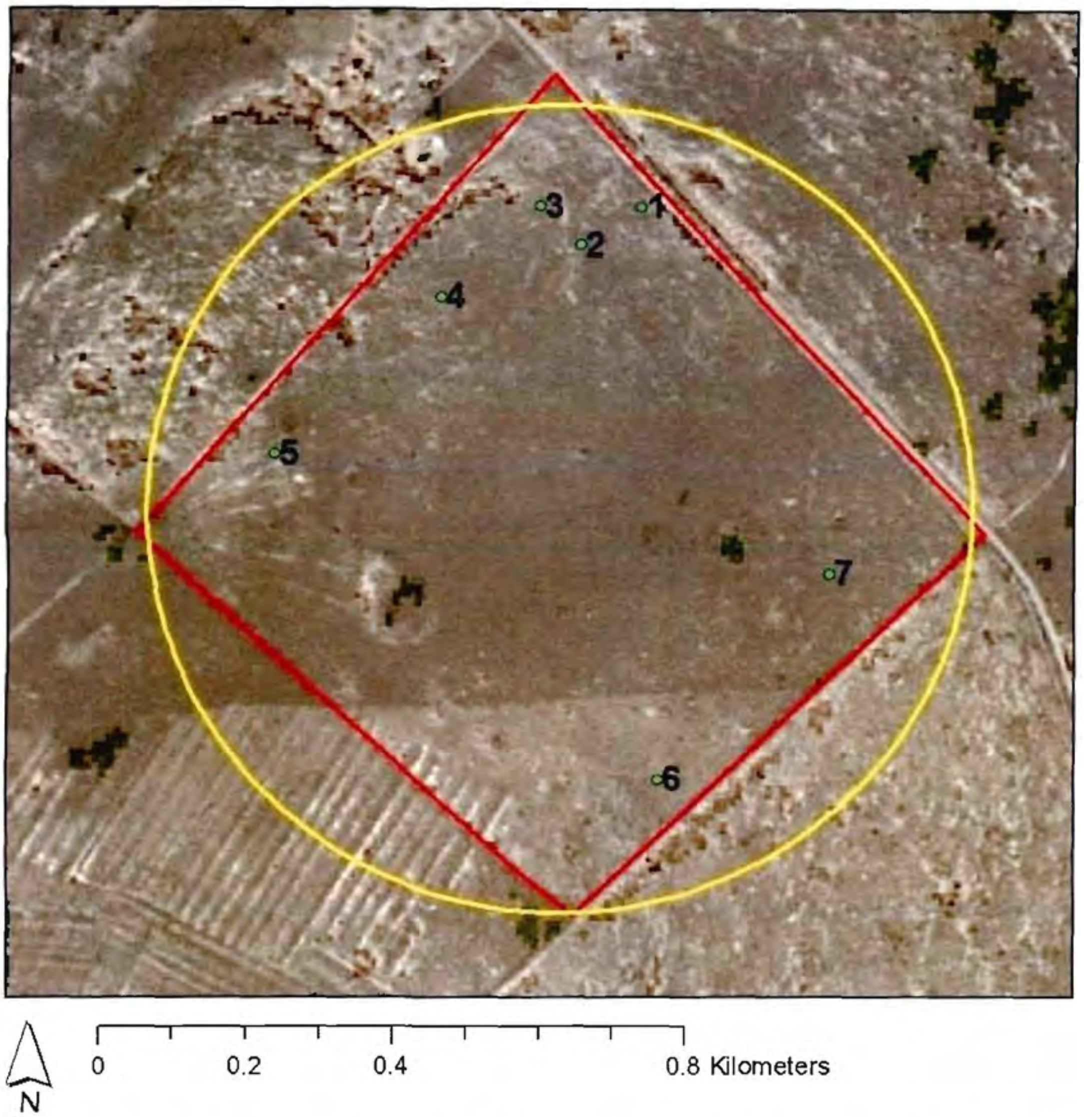

Figure C.14: Patch 14. The outline of the grassland patch is shown in red, and the boundary of the 1 km^2 area surrounding the patch is shown in yellow. The location of the sample plots is indicated as green dots. The background for the map is the, 50 % transparency, overlaid panchromatic band image on the land cover classification map.

Appendix D: Specific patch species richness and plant species lists

Table D.1: The total number of plant species and the grouping categories as surveyed for the existing vegetation per patch (Chapter 6) of the greater Klerksdorp area.

Patches	All species	Indigenous	Exotic	Annual	Perennial	Dicot	Grasses
1	63	61	2	5	58	42	12
2	64	58	6	9	55	46	12
3	54	43	11	10	44	41	8
4	83	70	13	16	67	60	15
5	74	71	3	12	62	45	19
6	111	108	3	14	97	70	28
7	55	48	7	7	48	37	14
8	76	71	5	14	62	52	19
9	52	36	16	19	33	36	10
10	83	80	3	14	69	51	21
11	69	62	7	7	62	42	20
12	64	60	4	11	53	45	15
13	86	82	4	11	75	61	14
14	108	101	7	17	91	71	24
Total	228	201	27	54	174	162	42

Table D.2: The total number of plant species and the grouping categories as surveyed for the soil seed bank vegetation per patch (Chapter 6) of the greater Klerksdorp area.

Patches	All species	Indigenous	Exotic	Annual	Perennial	Dicot	Grasses
1	11	8	3	3	8	8	2
2	16	10	6	5	11	13	2
3	13	7	6	7	6	9	3
4	26	17	10	10	16	19	4
5	8	5	3	3	5	6	2
6	12	8	4	4	8	8	3
7	14	10	3	2	12	7	4
8	11	8	3	4	7	7	3
9	15	9	6	7	8	10	5
10	13	10	3	3	10	7	4
11	7	6	1	2	5	4	1
12	12	10	2	4	8	8	4
13	14	11	3	5	9	9	2
14	18	14	4	7	11	13	5
Total	58	38	20	24	34	44	9

Table D.3: List of the plant species recorded during the vegetation survey of the existing vegetation (Chapter 6). Plant names follow Germishuizen *et al.* (2006).

Botanical name	Origin	Life span	Life-form
Acacia karroo	Indigenous	Perennial	Dicot
Acacia tortilis subsp. *heteracantha*	Indigenous	Perennial	Dicot
Acalypha angustata	Indigenous	Perennial	Dicot
Acanthospermum glabratum	Exotic	Annual	Dicot
Aloe greatheadii var. *davyana*	Indigenous	Perennial	Monocot
Alternanthera pungens	Exotic	Perennial	Dicot
Amaranthus thunbergii	Indigenous	Annual	Dicot
Anthospermum rigidum subsp. *rigidum*	Indigenous	Perennial	Dicot
Aptosimum elongatum	Indigenous	Perennial	Dicot
Arctotis arctotoides	Indigenous	Perennial	Dicot
Arctotis microcephala	Indigenous	Perennial	Dicot
Aristida canescens subsp. *canescens*	Indigenous	Perennial	Grass
Aristida congesta subsp. *congesta*	Indigenous	Perennial	Grass
Aristida diffusa subsp. *burkei*	Indigenous	Perennial	Grass
Aristida stipitata	Indigenous	Perennial	Grass
Asclepias aurea	Indigenous	Perennial	Geophyte
Asclepias gibba	Indigenous	Perennial	Geophyte
Asparagus suaveolens	Indigenous	Perennial	Dicot
Atriplex semibaccata var. *typica*	Exotic	Annual	Dicot
Barleria macrostegia	Indigenous	Perennial	Dicot
Berkheya radula	Indigenous	Perennial	Dicot
Bidens bipinnata	Exotic	Annual	Dicot
Blepharis angusta	Indigenous	Perennial	Dicot
Blepharis integrifolia var. *integrifolia*	Indigenous	Perennial	Dicot
Boerhavia erecta	Exotic	Annual	Dicot
Boophone disticha	Indigenous	Perennial	Geophyte
Brachiaria serrata	Indigenous	Perennial	Grass
Bulbine abyssinica	Indigenous	Perennial	Geophyte
Bulbine narcissifolia	Indigenous	Perennial	Geophyte
Chaetacanthus costatus	Indigenous	Perennial	Dicot
Chamaecrista biensis	Indigenous	Perennial	Dicot
Chascanum adenostachyum	Indigenous	Perennial	Dicot
Chascanum hederaceum	Indigenous	Perennial	Dicot
Chloris virgata	Indigenous	Annual	Grass
Chlorophytum cooperi	Indigenous	Perennial	Monocot
Chlorophytum fasciculatum	Indigenous	Perennial	Monocot
Ciclospermum leptophyllum	Exotic	Annual	Dicot
Clematis brachiata	Indigenous	Perennial	Dicot
Commelina africana	Indigenous	Perennial	Dicot
Commelina benghalensis	Indigenous	Annual	Dicot
Convolvulus arvensis	Exotic	Perennial	Dicot
Convolvulus sagittatus	Indigenous	Perennial	Dicot
Conyza aegyptiaca	Indigenous	Annual	Dicot
Conyza bonariensis	Exotic	Annual	Dicot
Conyza podocephala	Indigenous	Perennial	Dicot
Corchorus asplenifolius	Indigenous	Perennial	Dicot
Crabbea acaulis	Indigenous	Perennial	Dicot
Crabbea angustifolia	Indigenous	Perennial	Dicot
Crabbea hirsuta	Indigenous	Perennial	Dicot
Crassula lanceolata subsp. *transvaalensis*	Indigenous	Perennial	Dicot
Crotalaria lotoides	Indigenous	Perennial	Dicot
Crotalaria sphaerocarpa subsp. *sphaerocarpa*	Indigenous	Annual	Dicot
Cucumis hirsutus	Indigenous	Perennial	Dicot
Cucumis zeyheri	Indigenous	Perennial	Dicot
Cyanotis speciosa	Indigenous	Perennial	Dicot
Cymbopogon caesius	Indigenous	Perennial	Grass
Cymbopogon pospischilii	Indigenous	Perennial	Grass
Cynodon dactylon	Indigenous	Perennial	Grass
Cyperus indecorus	Indigenous	Perennial	Monocot
Delosperma herbeum	Indigenous	Perennial	Dicot
Deverra burchellii	Indigenous	Perennial	Dicot
Dianthus basuticus subsp. *basuticus* var. *basuticus*	Indigenous	Perennial	Dicot
Dicoma anomala	Indigenous	Perennial	Dicot
Digitaria argyrograpta	Indigenous	Perennial	Grass

Botanical name	Origin	Life span	Life-form
Digitaria eriantha	Indigenous	Perennial	Grass
Digitaria ternata	Indigenous	Annual	Grass
Diheteropogon amplectens	Indigenous	Perennial	Grass
Elephantorrhiza elephantina	Indigenous	Annual	Dicot
Elionurus muticus	Indigenous	Perennial	Grass
Eragrostis biflora	Indigenous	Annual	Grass
Eragrostis chloromelas	Indigenous	Perennial	Grass
Eragrostis gummiflua	Indigenous	Perennial	Grass
Eragrostis inamoena	Indigenous	Perennial	Grass
Eragrostis lehmanniana var. *lehmanniana*	Indigenous	Perennial	Grass
Eragrostis obtusa	Indigenous	Perennial	Grass
Eragrostis plana	Indigenous	Perennial	Grass
Eragrostis racemosa	Indigenous	Perennial	Grass
Ergrostis superba	Indigenous	Perennial	Grass
Eriospermum porphyrium	Indigenous	Perennial	Geophyte
Erythrina zeyheri	Indigenous	Perennial	Dicot
Euphorbia hirta	Exotic	Annual	Dicot
Euphorbia inaequilatera var. *inaequilatera*	Indigenous	Annual	Dicot
Eustachys paspaloides	Indigenous	Perennial	Grass
Felicia fascicularis	Indigenous	Perennial	Dicot
Felicia muricata subsp. *muricata*	Indigenous	Perennial	Dicot
Gazania krebsiana subsp. *serrulata*	Indigenous	Perennial	Dicot
Gerbera piloselloides	Indigenous	Perennial	Dicot
Gladiolus permeabilis subsp. *edulis*	Indigenous	Perennial	Dicot
Glossochilus burchellii	Indigenous	Perennial	Dicot
Gnidia capitata	Indigenous	Perennial	Dicot
Gnidia sericocephala	Indigenous	Perennial	Dicot
Gomphrena celosioides	Exotic	Perennial	Dicot
Guilleminea densa	Exotic	Perennial	Dicot
Gymnosporia heterophylla	Indigenous	Perennial	Dicot
Harpagophytum procumbens subsp. *procumbens*	Indigenous	Perennial	Dicot
Helichrysum caespititium	Indigenous	Perennial	Dicot
Helichrysum dregeanum	Indigenous	Perennial	Dicot
Helichrysum nudifolium var. *nudifolium*	Indigenous	Perennial	Dicot
Helichrysum rugulosum	Indigenous	Perennial	Dicot
Helichrysum zeyheri	Indigenous	Perennial	Dicot
Hermannia depressa	Indigenous	Perennial	Dicot
Heteropogon contortus	Indigenous	Perennial	Grass
Hibiscus microcarpus	Indigenous	Annual	Dicot
Hibiscus pusillus	Indigenous	Perennial	Dicot
Hibiscus trionum	Indigenous	Annual	Dicot
Hyparrhenia hirta	Indigenous	Perennial	Grass
Hypochaeris radicata	Exotic	Perennial	Dicot
Hypoxis argentea	Indigenous	Perennial	Geophyte
Hypoxis hemerocallidea	Indigenous	Perennial	Geophyte
Hypoxis rigidula	Indigenous	Perennial	Geophyte
Indigastrum costatum subsp. *macrum*	Indigenous	Annual	Dicot
Indigofera comosa	Indigenous	Perennial	Dicot
Indigofera cryptantha var. *cryptantha*	Indigenous	Perennial	Dicot
Indigofera daleoides var. *daleoides*	Indigenous	Perennial	Dicot
Indigofera filipes	Indigenous	Perennial	Dicot
Indigofera heterotricha	Indigenous	Perennial	Dicot
Indigofera rhytidocarpa subsp. *rhytidocarpa*	Indigenous	Annual	Dicot
Indigofera vicioides var. *vicioides*	Indigenous	Perennial	Dicot
Ipomoea oblongata	Indigenous	Perennial	Dicot
Ipomoea obscura var. *obscura*	Indigenous	Perennial	Dicot
Ipomoea oenotheroides	Indigenous	Perennial	Dicot
Kohautia caespitosa subsp. *bracyloba*	Indigenous	Annual	Dicot
Kohautia virgata	Indigenous	Annual	Dicot
Kyphocarpa angustifolia	Indigenous	Annual	Dicot
Lactuca inermis	Indigenous	Perennial	Dicot
Lactuca serriola	Exotic	Perennial	Dicot
Lantana rugosa	Indigenous	Perennial	Dicot
Ledebouria cooperi	Indigenous	Perennial	Geophyte
Ledebouria ovatifolia	Indigenous	Perennial	Geophyte
Ledebouria revoluta	Indigenous	Perennial	Geophyte
Lepidium bonariense	Exotic	Annual	Dicot
Leucas capensis	Indigenous	Perennial	Dicot

Botanical name	Origin	Life span	Life-form
Lippia scaberrima	Indigenous	Perennial	Dicot
Lotononis calycina	Indigenous	Perennial	Dicot
Lotononis listii	Indigenous	Perennial	Dicot
Macledium zeyheri subsp. *zeyheri*	Indigenous	Perennial	Dicot
Medicago laciniata var. *laciniata*	Exotic	Annual	Dicot
Melhania prostrata	Indigenous	Perennial	Dicot
Melinis repens	Indigenous	Annual	Grass
Merremia palmata	Indigenous	Perennial	Dicot
Microchloa caffra	Indigenous	Perennial	Grass
Monsonia angustifolia	Indigenous	Annual	Dicot
Nidorella anomala	Indigenous	Annual	Dicot
Nidorella hottentotica	Indigenous	Annual	Dicot
Nidorella resedifolia subsp. *resedifolia*	Indigenous	Annual	Dicot
Oenothera tetraptera	Exotic	Perennial	Dicot
Oldenlandia capensis var. *capensis*	Indigenous	Annual	Dicot
Ornithogalum flexuosum	Indigenous	Perennial	Geophyte
Ornithogalum setosum	Indigenous	Perennial	Geophyte
Osteospermum muricatum subsp. *muricatum*	Indigenous	Perennial	Dicot
Oxalis corniculata	Exotic	Annual	Dicot
Oxalis depressa	Indigenous	Perennial	Geophyte
Oxalis obliquifolia	Indigenous	Perennial	Geophyte
Pachycarpus rigidus	Indigenous	Perennial	Geophyte
Panicum coloratum var. *coloratum*	Indigenous	Perennial	Grass
Pennisetum clandestinum	Exotic	Perennial	Grass
Pentzia globosa	Indigenous	Perennial	Dicot
Phyllanthus maderaspatensis	Indigenous	Perennial	Dicot
Physalis viscosa	Exotic	Perennial	Dicot
Plantago lanceolata	Indigenous	Perennial	Dicot
Pogonarthria squarrosa	Indigenous	Perennial	Grass
Pollichia campestris	Indigenous	Perennial	Dicot
Polygala hottentotta	Indigenous	Perennial	Dicot
Portulaca kermesina	Indigenous	Annual	Dicot
Portulaca oleracea	Exotic	Annual	Dicot
Portulaca quadrifida	Indigenous	Annual	Dicot
Pseudognaphalium oligandrum	Indigenous	Annual	Dicot
Pterodiscus speciosus	Indigenous	Perennial	Dicot
Pteronia tricephala	Indigenous	Perennial	Dicot
Raphionacme velutina	Indigenous	Perennial	Geophyte
Rhynchosia confusa	Indigenous	Perennial	Dicot
Ruschia hamata	Indigenous	Perennial	Dicot
Salvia radula	Indigenous	Perennial	Dicot
Salvia runcinata	Indigenous	Perennial	Dicot
Scabiosa columbaria	Indigenous	Perennial	Dicot
Schizachyrium sanguineum	Indigenous	Perennial	Grass
Schkuhria pinnata	Exotic	Annual	Dicot
Searsia ciliata	Indigenous	Perennial	Dicot
Sebaea exigua	Indigenous	Annual	Dicot
Sebaea grandis	Indigenous	Annual	Dicot
Seddera suffruticosa	Indigenous	Perennial	Dicot
Selago densiflora	Indigenous	Perennial	Dicot
Senecio affinis	Indigenous	Perennial	Dicot
Senecio consanguineus	Indigenous	Annual	Dicot
Senecio inaequidens	Indigenous	Perennial	Dicot
Senna italica subsp. *arachoides*	Indigenous	Perennial	Dicot
Seriphium plumosum	Indigenous	Perennial	Dicot
Setaria sphacelata var. *torta*	Indigenous	Perennial	Grass
Sida dregei	Indigenous	Annual	Dicot
Sida ovata	Indigenous	Perennial	Dicot
Sida rhombifolia subsp. *rhombifolia*	Indigenous	Annual	Dicot
Sida spinosa var. *spinosa*	Indigenous	Annual	Dicot
Solanum lichtensteinii	Indigenous	Perennial	Dicot
Solanum rubetorum	Indigenous	Perennial	Dicot
Sporobolus africanus	Indigenous	Perennial	Grass
Sporobolus discosporus	Indigenous	Perennial	Grass
Sporobolus fimbriatus	Indigenous	Perennial	Grass
Sporobolus pyramidalis	Indigenous	Perennial	Grass
Striga asiatica	Indigenous	Annual	Dicot
Tagetes minuta	Exotic	Annual	Dicot

Botanical name	Origin	Life span	Life-form
Talinum caffrum	Indigenous	Annual	Dicot
Tephrosia burchellii	Indigenous	Annual	Dicot
Tephrosia capensis	Indigenous	Perennial	Dicot
Tephrosia longipes subsp. *longipes* var. *longipes*	Indigenous	Annual	Dicot
Teucrium trifidum	Indigenous	Perennial	Dicot
Themeda triandra	Indigenous	Perennial	Grass
Thesium racemosum	Indigenous	Perennial	Dicot
Thesium utile	Indigenous	Perennial	Dicot
Trachyandra asperata var. *macowanii*	Indigenous	Perennial	Geophyte
Tragus berteronianus	Indigenous	Annual	Grass
Tribulus terrestris	Indigenous	Annual	Dicot
Trichoneura grandiglumis	Indigenous	Perennial	Grass
Trifolium repens	Exotic	Perennial	Dicot
Tripteris aghillana var. *aghillana*	Indigenous	Perennial	Dicot
Triraphis andropogonoides	Indigenous	Perennial	Grass
Urochloa mosambicensis	Indigenous	Perennial	Grass
Vahlia capensis	Indigenous	Annual	Dicot
Verbena aristigera	Exotic	Perennial	Dicot
Verbena bonariensis	Exotic	Annual	Dicot
Verbena officinalis	Exotic	Annual	Dicot
Vernonia oligocephala	Indigenous	Perennial	Dicot
Vigna unguiculata subsp. *stenophylla*	Indigenous	Perennial	Dicot
Vigna vexillata	Indigenous	Perennial	Dicot
Wahlenbergia undulata	Indigenous	Perennial	Dicot
Zinnia peruviana	Exotic	Annual	Dicot
Ziziphus zeyheriana	Indigenous	Perennial	Dicot
Zornia linearis	Indigenous	Perennial	Dicot

Table D.4: List of the plant species recorded during the vegetation survey of the soil seed bank (Chapter 6). Plant names follow Germishuizen et al. (2006).

Botanical name	Origin	Life span	Life-form
Acacia karroo	Indigenous	Perennial	Dicot
Aptosimum elongatum	Indigenous	Perennial	Dicot
Arctotis arctotoides	Indigenous	Perennial	Dicot
Argemone ochroleuca subsp. *ochroleuca*	Exotic	Annual	Dicot
Chenopodium carinatum	Exotic	Annual	Dicot
Chenopodium sp.	Exotic	Annual	Dicot
Ciclospermum leptophyllum	Exotic	Annual	Dicot
Convolvulus arvensis	Exotic	Perennial	Dicot
Conyza bonariensis	Exotic	Annual	Dicot
Conyza canadensis	Exotic	Annual	Dicot
Conyza podocephala	Indigenous	Perennial	Dicot
Corchorus asplenifolius	Indigenous	Perennial	Dicot
Cynodon dactylon	Indigenous	Perennial	Grass
Cyperus esculentus var. *esculentus*	Indigenous	Perennial	Geophyte
Cyperus indecorus	Indigenous	Perennial	Monocot
Dactolyctenium aegyptium	Indigenous	Annual	Grass
Delosperma herbeum	Indigenous	Perennial	Dicot
Deverra burchellii	Indigenous	Perennial	Dicot
Emex australis	Indigenous	Annual	Dicot
Eragrostis chloromelas	Indigenous	Perennial	Grass
Eragrostis lehmanniana var. *lehmanniana*	Indigenous	Perennial	Grass
Eragrostis superba	Indigenous	Annual	Grass
Felicia muricata subsp. *muricata*	Indigenous	Perennial	Dicot
Gamochaeta pennsylvanica	Exotic	Annual	Dicot
Gomphrena celosioides	Exotic	Perennial	Dicot
Helichrysum argyrosphaerum	Indigenous	Annual	Dicot
Hermannia depressa	Indigenous	Perennial	Dicot
Hibiscus trionum	Indigenous	Annual	Dicot
Kohautia sp.	Indigenous	Annual	Dicot
Lepidium bonariense	Exotic	Annual	Dicot
Oenothera tetraptera	Exotic	Perennial	Dicot
Oxalis corniculata	Exotic	Annual	Dicot
Oxalis obliquifolia	Indigenous	Perennial	Geophyte
Panicum maximum	Indigenous	Perennial	Grass
Physalis viscosa	Exotic	Perennial	Dicot
Plantago lanceolata	Indigenous	Perennial	Dicot
Pollichia campestris	Indigenous	Perennial	Dicot
Polygonum sp.	Indigenous	Annual	Dicot
Pseudognaphalium luteo-album	Indigenous	Annual	Dicot
Salvia runcinata	Indigenous	Perennial	Dicot
Schinus terebinthifolius	Exotic	Perennial	Dicot
Selago densiflora	Indigenous	Perennial	Dicot
Senecio apiifolius	Indigenous	Annual	Dicot
Senecio consanguineus	Indigenous	Annual	Dicot
Solanum nigrum	Exotic	Annual	Dicot
Tagetes minuta	Exotic	Annual	Dicot
Taraxacum officinale	Exotic	Perennial	Dicot
Tephrosia longipes subsp. *longipes* var. *longipes*	Indigenous	Annual	Dicot
Themeda triandra	Indigenous	Perennial	Grass
Tragopogon porrifolius	Exotic	Perennial	Dicot
Tragus berteronianus	Indigenous	Annual	Grass
Unknown geophyte	Indigenous	Perennial	Geophyte
Urochloa mosambicensis	Indigenous	Perennial	Grass
Verbena aristigera	Exotic	Perennial	Dicot
Verbena bonariensis	Exotic	Annual	Dicot
Vernonia oligocephala	Indigenous	Perennial	Dicot
Wahlenbergia undulata	Indigenous	Perennial	Dicot
Zornia milneana	Indigenous	Perennial	Dicot

Appendix E: Soil laboratory analysis results

The following tables supplies the entire report of soil sample analysis of the soil properties measured for each respective sample plot as calculated by the Eco-Analytica laboratory.

NORTH-WEST UNIVERSITY ECO-ANALYTICA	Eco Analytica PO Box 19140 NOORDBRUG 2522 Phone: (018) 293 3900

Nutritional Status

Sample	Ca	Mg	K	Na	pH(H_2O)	EC
no.	(mg/kg)					(mS/m)
1.1	618.5	104	187	47.5	6.57	25
1.2	640	99	129	27.5	6.49	28
2.1	540	88.5	103.5	29	6.2	16
2.2	849	157	262.5	54.5	6.48	27
3.1	1509	397.5	168.5	44.5	6.59	30
3.2	1941	454.5	147	62.5	6.95	41
4.1	555.5	68	144	26.5	6.03	28
4.2	736.5	166	223	18.5	6.39	27
4.3	820	171.5	117	35	6.23	26
4.4	804	166.5	145.5	68	6.76	33
5.1	596	116.5	118	20.5	6.33	25
5.2	577.5	122.5	176.5	103	5.95	24
6.1	428	81	350.5	36	6.22	20
6.2	447.5	96.5	71.5	14.5	6.26	16
6.3	552.5	95.5	109.5	53.5	6.23	25
6.4	788.5	186	208	17	6.31	22
7.1	499	82	197.5	22.5	6.25	26
7.2	652.5	106.5	177	37.5	6.62	27
8.1	748.5	150.5	135	14	6.1	21
8.2	797	212.5	176	28	6.41	28
8.3	674.5	185.5	291.5	47.5	6.07	192
9.1	799.5	66.5	179	6	6.75	47
9.2	708.5	102	149.5	13	6.61	30
10.1	457.5	67.5	80.5	25	6.26	42
10.2	497	68.5	88.5	21	6.31	29
10.3	501.5	85	156	14.5	5.93	38
10.4	547	94.5	123	31.5	6.35	32
11.1	476.5	78	100	19	6.37	22
11.2	398	55	55	15	5.87	17
12.1	540	102.5	182	46.5	5.54	30
12.2	604.5	124	183	28.5	6.36	18
12.3	699	121.5	207.5	6.5	6.58	25
13.1	842.5	160	220.5	40	6.47	40
13.2	874	200	212.5	63.5	6.09	21
13.3	657.5	143	120	57	6.23	19
13.4	456.5	195	122	83.5	6.28	20
13.5	825.5	231.5	183	79	6.3	39
13.6	846	202.5	168	89.5	6.42	22
14.1	607	176	175.5	63	6.1	29
14.2	520.5	64.5	98	22.5	5.96	19
14.3	542.5	167.5	150.5	64.5	6.19	18
14.4	568.5	188.5	124	38	6.2	21
14.5	554.5	161.5	137	82.5	6.14	16
14.6	580.5	144	178	38.5	6.23	24
14.7	546	124.5	173	25.5	6.15	19

Exchangeable cations

Sample no.	Ca	Mg	K	Na	CEC	S-value	Base saturation (%)	pH(H_2O)
	(cmol(+)/kg)							
1.1	3.09	0.86	0.48	0.21	5.96	4.63	77.7	6.57
1.2	3.19	0.81	0.33	0.12	6.11	4.46	72.94	6.49
2.1	2.69	0.73	0.27	0.13	6.16	3.81	61.96	6.2
2.2	4.24	1.29	0.67	0.24	8.71	6.44	73.9	6.48
3.1	7.53	3.27	0.43	0.19	14.86	11.43	76.89	6.59
3.2	9.69	3.74	0.38	0.27	14.63	14.08	96.23	6.95
4.1	2.77	0.56	0.37	0.12	7.4	3.82	51.57	6.03
4.2	3.68	1.37	0.57	0.08	8.72	5.69	65.28	6.39
4.3	4.09	1.41	0.3	0.15	9.53	5.96	62.49	6.23
4.4	4.01	1.37	0.37	0.3	7.91	6.05	76.47	6.76
5.1	2.97	0.96	0.3	0.09	6.98	4.32	61.93	6.33
5.2	2.88	1.01	0.45	0.45	9.73	4.79	49.23	5.95
6.1	2.14	0.67	0.9	0.16	6.71	3.86	57.46	6.22
6.2	2.23	0.79	0.18	0.06	5.65	3.27	57.92	6.26
6.3	2.76	0.79	0.28	0.23	6.87	4.06	59.05	6.23
6.4	3.93	1.53	0.53	0.07	9.94	6.07	61.1	6.31
7.1	2.49	0.67	0.51	0.1	6.56	3.77	57.49	6.25
7.2	3.26	0.88	0.45	0.16	6.12	4.75	77.58	6.62
8.1	3.74	1.24	0.35	0.06	10.29	5.38	52.31	6.1
8.2	3.98	1.75	0.45	0.12	8.82	6.3	71.44	6.41
8.3	3.37	1.53	0.75	0.21	11.03	5.85	52.98	6.07
9.1	3.99	0.55	0.46	0.03	6.11	5.02	82.15	6.75
9.2	3.54	0.84	0.38	0.06	6.09	4.81	79.1	6.61
10.1	2.28	0.56	0.21	0.11	5.37	3.15	58.68	6.26
10.2	2.48	0.56	0.23	0.09	5.5	3.36	61.18	6.31
10.3	2.5	0.7	0.4	0.06	7.88	3.67	46.52	5.93
10.4	2.73	0.78	0.32	0.14	6.63	3.96	59.76	6.35
11.1	2.38	0.64	0.26	0.08	5.24	3.36	64.06	6.37
11.2	1.99	0.45	0.14	0.07	5.85	2.64	45.2	5.87
12.1	2.69	0.84	0.47	0.2	8.55	4.21	49.22	5.54
12.2	3.02	1.02	0.47	0.12	7.57	4.63	61.13	6.36
12.3	3.49	1	0.53	0.03	6.54	5.05	77.2	6.58
13.1	4.2	1.32	0.57	0.17	8.62	6.26	72.65	6.47
13.2	4.36	1.65	0.54	0.28	12.66	6.83	53.93	6.09
13.3	3.28	1.18	0.31	0.25	8.33	5.01	60.18	6.23
13.4	2.28	1.6	0.31	0.36	7.4	4.56	61.6	6.28
13.5	4.12	1.91	0.47	0.34	10.57	6.84	64.66	6.3
13.6	4.22	1.67	0.43	0.39	9.63	6.71	69.69	6.42
14.1	3.03	1.45	0.45	0.27	9.67	5.2	53.79	6.1
14.2	2.6	0.53	0.25	0.1	7.63	3.48	45.6	5.96
14.3	2.71	1.38	0.39	0.28	8.15	4.75	58.32	6.19
14.4	2.84	1.55	0.32	0.17	8.41	4.87	57.93	6.2
14.5	2.77	1.33	0.35	0.36	8.51	4.81	56.46	6.14
14.6	2.9	1.19	0.46	0.17	7.97	4.71	59.08	6.23
14.7	2.72	1.02	0.44	0.11	7.86	4.3	54.75	6.15

Exchangeable cations:	1 M NH_4-acetate pH=7	**Phosphate:**	Bray 1 - Extract
CEC:	1 M NH_4-acetate pH=7	**pH H_2O/KCl:**	1:2,5 - Extract
		EC:	Saturated extract

Appendix F: Patch species richness similarity tables

Table F.1: Percentage similarity between the average and the total species richness per patch for the existing vegetation of the fragmented grassland patches of the greater Klerksdorp area.

Patch	All species	Indigenous	Exotic	Annual	Perennial	Dicot	Grasses
1	*72.22*	*72.13*	*75*	*80*	*71.55*	*69.05*	*79.17*
2	*75*	*75*	*75*	*72.22*	*75.45*	*75*	*83.33*
3	*69.44*	*69.77*	*68.18*	*75*	*68.18*	*69.51*	*75*
4	*49.4*	*52.14*	*34.62*	*40.63*	*51.49*	*46.67*	*58.33*
5	*76.35*	*76.76*	*66.67*	*75*	*76.61*	*74.44*	*78.95*
6	*51.58*	*51.85*	*41.67*	*42.86*	*52.84*	*49.29*	*58.04*
7	*70*	*72.92*	*50*	*64.29*	*70.83*	*70.27*	*67.86*
8	*63.16*	*63.85*	*53.4*	*59.5*	*63.98*	*61.54*	*68.42*
9	*67.31*	*68.06*	*65.63*	*65.8*	*68.18*	*66.67*	*75*
10	*64.16*	*65.31*	*33.33*	*55.36*	*65.94*	*63.73*	*67.86*
11	*71.01*	*73.39*	*50*	*64.29*	*71.77*	*69.05*	*77.5*
12	*56.25*	*57.78*	*33.25*	*48.45*	*57.87*	*55.56*	*55.53*
13	*52.71*	*52.84*	*50*	*33.36*	*55.56*	*47*	*63.07*
14	*46.43*	*47.81*	*26.57*	*31.94*	*49.13*	*45.68*	*53.58*

Table F.2: Percentage similarity between the average and the total species richness per patch for the soil seed bank vegetation of the fragmented grassland patches of the greater Klerksdorp area.

Patch	All species	Indigenous	Exotic	Annual	Perennial	Dicot	Grasses
1	*68.18*	*62.5*	*83.33*	*83.33*	*62.5*	*62.5*	*75*
2	*65.63*	*65*	*66.67*	*70*	*63.64*	*65.38*	*75*
3	*61.54*	*50*	*75*	*71.43*	*50*	*66.67*	*50*
4	*40.38*	*35.29*	*45*	*40*	*40.63*	*38.16*	*50*
5	*75*	*60*	*100*	*100*	*60*	*83.33*	*50*
6	*37.5*	*40.63*	*31.25*	*31.25*	*40.63*	*31.25*	*50*
7	*67.86*	*80*	*50*	*50*	*70.83*	*64.29*	*75*
8	*51.55*	*54.13*	*44.33*	*41.75*	*57.14*	*47.57*	*44.33*
9	*70*	*66.67*	*75*	*71.43*	*68.75*	*70*	*70*
10	*34.62*	*37.5*	*25*	*25*	*37.5*	*32.14*	*37.5*
11	*64.29*	*66.67*	*50*	*50*	*70*	*62.5*	*100*
12	*52.75*	*50*	*66.5*	*58.25*	*50*	*45.88*	*66.75*
13	*23.79*	*18.18*	*44.33*	*33.4*	*22.22*	*29.67*	*30*
14	*40.5*	*35.71*	*57.25*	*42.86*	*39*	*38.46*	*45.8*