Understanding Impacts of Water Supplementation in a Heterogeneous Landscape

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I hereby confirm that this work is my own except where I have explicitly indicated otherwise. I have followed the required conventions in referencing the thoughts and ideas of others.

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13 OCTOBER 2010

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Abstract

Artificial water provision is a contentious management issue in southern African savanna conservation areas. Supplementation of permanent water leads to higher herbivore population sizes which can generate greater profits. However, water supplementation can lead to detrimental effects on soils and vegetation surrounding waterpoints. Currently, the impact of artificial waterpoints across properties is understood in terms of the piosphere model: concentric circles with differing degradation levels, focused on waterpoints. Southern African savannas are highly heterogeneous so the suitability of a homogeneity based approach in management is questionable. Provision of water currently follows a relatively high degradation risk strategy on many properties so a sound basis for management is essential.

This study assessed the general applicability of the piosphere model by testing the relationship between distance to water and ecological variables (soil functionality and herbaceous and woody vegetation). 23 variables were tested across 22 waterpoints from five properties within the Great Limpopo Transfrontier Conservation Area. Statistical approaches used matched those of previous piosphere studies but only 14% of tests were significant. Although utilisation gradients were found for some variables and some transects, there was no generally applicable pattern. This means that results from previous piosphere studies cannot be transferred to unsampled waterpoints or scaled up across properties. The level of heterogeneity in southern African savannas disrupts the piosphere pattern to such an extent that the model does not form an appropriate basis for management.

In order to develop an alternative approach to understanding the functionality of properties which takes into account both water supplementation and heterogeneity, the influence of a range of environmental and management variables on degradation and species composition were tested using ordinations. The best explanation of variation was a combination of environmental and management variables. Broader scale variables such as natural and artificial water availability were more important than finer scale variables such as distance to water. These results were used to develop a basic approach to evaluating property functionality.

Acknowledgements

This project resulted from my interest in the application of ecological theory in conservation management. This interest was first encouraged by Sue Milton and further developed by Harry Biggs who got me to the Kruger Network Meeting in 2005. At that meeting, Mike Peel highlighted the unique opportunity offered by the Great Limpopo Transfrontier Conservation Area for comparing the impacts of different conservation management approaches and Ed Witkowski told him he needed a PhD student to develop a project. After a lunchtime discussion at the meeting, Ed and Mike took on the task of supervising my PhD. I would like to extend a really enormous thank you to them for the time, energy and commitment they have put into supporting me through the project.

Conversations with Norman Owen-Smith and Mohammed Said contributed to the initial development of the project and Viv Williams provided a much appreciated tutorial on the use of EstimateS. My committee of Dave Mycock, Barend Erasmus, David Tongway, Rina Grant and Kevin Rogers have provided valuable insights and tricky questions to keep me focused. In the latter stages of the project, long conversations with Carola Cullum were key in fine-tuning my understanding, and keeping me sane! Annual conversations with Kate Matchett on piosphere theory, fieldwork problems and life in general have also been of great help.

The fieldwork for this project had a long duration and the logistical support I received from the School of Animal, Plant and Environmental Sciences was really appreciated. A special thanks to Landie and Herman for their help. Also to Kevin Balkwill who let me take the KTK bakkie for such an extended period of time. The loan of camping equipment from the ARC made life a lot more comfortable in the field.

Having not worked in savanna ecosystems prior to this project, I had to undergo a rapid learning phase. The time and effort put in by John and Jakes (John Peel and Andre Jacobs of the ARC Nelspruit) to teach me grass and woody vegetation identifications is immeasurable. Associated with this, thanks to the private reserves which allowed me to join John and Jakes for field experience. In particular to York

Private Nature Reserve, Sabie Sands and Thornybush Private Nature Reserve where I had the opportunity to stay overnight and really maximise my field time with John and Jakes.

Valuable field time was also spent with my supervisors, hours driving between private reserves and chatting about every conceivable angle of management with Mike and several days tramping through the bush collecting as many woody vegetation species as possible with Ed. I was privileged to spend time in the field with David Tongway, I will never forget his surprise at the amount of grass there was! The few days I spent with David were an incredible experience and very influential on the way I now see ecosystems. Mike Pingo and Mike Pieterse gave me an amazing opportunity to take part in the helicopter game count of Thornybush Private Nature Reserve. A really fun day and very informative to see waterpoints from the air.

The nine properties I worked in were each wonderful in their own way. In alphabetical order…

Many thanks to the owners and management of the Greater Olifants River Conservancy who gave permission for the research. Mario and Meagan Cesare made life comfortable during fieldwork. A really special thank you to the shareholders who let us use their houses, this meant that after fieldwork each day we relaxed in the lap of luxury! A big thank you also, to the field rangers who spent days out with us while we were sampling. Colin Rowles of Klaserie Private Nature Reserve was of great help to the project. His GIS database made my life much easier! Colin also had to play the role of coordinator for the APNR (Associated Private Nature Reserves), a job that proved to be tougher than expected. Finally the project is over with interesting results! A thank you also to the field rangers from Klaserie who accompanied us during fieldwork.

Many people from Kruger National Park contributed to various aspects of the project. Special thanks for help from Rina Grant (my coordinator), Sandra MacFadyen, Izak Smit, Sharon Lemos, Jacques Venter, Nick Zambatis and Richard Sowry. The data I received from Kruger made Chapter 2 possible, a big thanks for that. The watchful eyes and ears of the game guards made fieldwork a more relaxed experience and the added expertise of the field rangers of Letaba who acted as game guards was greatly appreciated. A special thank you to Joe Nkuna, section ranger of Letaba, for going out of his way to assist in fieldwork logistics and to the field rangers of Houtboschrand for their help with waterpoint histories.

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Thanks to the owners and management of the Mohlabetsi Association of Landowners for permission to do the project and access to data. Glen Thomson spent several hours driving me around the property and explaining the management system and water provision approach. Francois Oberholzer was a great help in data collection both on Mohlabetsi and in Mozambique. Also, thanks to Alistair Potts who gave me free use of his shapefiles for the map of Mohlabetsi.

Scott Ronaldson (then of Timbavati Private Nature Reserve) gave a lot of input towards the development of the project. Many thanks to the owners and management of Timbavati Private Nature Reserve for making data available for the project. In particular, a special logistical thank you to Paul White and Michéle Pietersen. Mike Pieterse and Jonathan Beretta of Thornybush Private Nature Reserve were also involved in development of the project. Their assistance with logistics and data acquisition was much appreciated. Thank you also to the owners and management of Thornybush for allowing me to conduct preliminary fieldwork at one of their waterpoints. The assistance of Georgette Lagendijk during this data collection was appreciated. Errol Pietersen (then of Umbabat Private Nature Reserve) spent a day with me explaining the management and running of Umbabat. This was a great learning experience and was really appreciated. Thanks also to the owners and management of Umbabat Private Nature Reserve for access to the data used in this project.

York Private Nature Reserve was always a fun place to visit. Riaan Ahlers and Melodie Bates provided many amusing evenings of conversation alongside a lot of helpful information and knowledge. The time spent on York doing the rapid waterpoint assessment contributed immeasurably to my understanding of the water provision issue in the private reserves. Many thanks to the owners and management of York for allowing me to do the fieldwork and to use the data I collected in this study.

Throughout the fieldwork in 2006 and 2007 I was assisted by various people. The tireless contributions of Innocentia Mahlabegwane, Joost van Munster, Robert Buitenwerf, Francois Oberholzer and Noor de Laat were greatly appreciated. Without their help I would not have been able to collect the amount of data and cover the number of waterpoints that I did. For my return to the field in 2008 I was joined by Nathan Arendse and Carola Cullum. Their determination to make me get it right and the discussion on the tree at Jim No. 1 resulted in the change in direction of the project and the testing of piosphere theory. Their input to this field trip and enthusiasm in mapping waterpoints and dodging elephants was greatly appreciated.

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Summary and Thesis Structure

Water supplementation is a controversial management issue in southern African savanna conservation areas. Currently, the impacts of supplementation are understood in terms of the piosphere model which describes functional degradation as concentric circles focused on waterpoints, with higher degradation closer to waterpoints. Due to the high level of heterogeneity in southern African savannas, the applicability of a method which removes variation has been questioned. This study was initiated to better understand the use of the piosphere model and to test its general applicability in the heterogeneous southern African savanna. This was achieved through a set of aims split into three groups:

Ecological Theory

1. To critically examine the assumptions, limitations and modern application of piosphere theory.
2. To understand how savanna waterpoint ecology, spatial heterogeneity and ecosystem resilience interact.

Application of Theory

3. To determine water distribution in the study area.
4. To develop an understanding of the impacts of artificial water supplementation at the landscape extent.

Synthesis and Applications

5. To understand the implications of a heterogeneity paradigm for understanding and managing the effects of artificial water supplementation.
6. To increase information available for management decisions regarding artificial water provision levels and waterpoint monitoring.

In order to address the aims, this study was performed in a section of the Great Limpopo Transfrontier Conservation Area: seven South African private reserves, Kruger National Park and Limpopo National Park. The artificial water distribution was determined for all properties except Limpopo National Park as Limpopo National

Park does not have artificial waterpoints. Fieldwork to develop an understanding of the impacts of artificial water provision was performed in the two national parks and three of the private reserves. Due to time constraints for fieldwork, not all private reserves could be sampled. The three reserves selected were the most logistically easy for fieldwork and represented the full spectrum of management intensities found in the private reserves.

This thesis gives the results of the study and is set out as a collection of seven chapters, each written as a scientific paper addressing an element from the aims. The first chapter is a literature review that deals with the rationale for the project. The second chapter presents data analysis to determine water supplementation levels. The third chapter is a literature review that deals with ecological concepts of importance to water provision as an issue in conservation management. The fourth and fifth chapters present data analysis to test the piosphere theory and the development of an alternative approach to understanding waterpoint impact across properties. The final two chapters both synthesise the study. The sixth chapter presents a standard ecological synthesis of the first five chapters. The seventh chapter presents an alternative synthesis, aimed at a management audience, that specifically details the implications of the results of the study and the application of these results in conservation management.

Chapter One addresses the origin of the piosphere model and how its application has changed over time. In 1969 the terminology was created (Lange, 1969) and in 1978 statistics to analyse piospheres were developed (Graetz & Ludwig, 1978). In 1988 two papers were written, a review which did not mention heterogeneity (Andrew, 1988) and a report on a grazing pattern study which states vegetation type is important (Pickup & Chewings, 1988). In the 1990s a split appears between researchers who use ‘piospheres’ and those who use ‘grazing gradients’. At the end of the 1990s two review papers were published which illustrate the low importance of heterogeneity in piosphere work (Thrash & Derry, 1999; James *et al.*, 1999). In the 2000s acknowledgement of heterogeneity is often found in methods sections but not in the interpretation of results. In 2009 it was concluded that data cannot be averaged around waterpoints to generate gradients as heterogeneity has ecological and management importance (Chamaillé-Jammes *et al.*, 2009).

The first step in understanding the impact of water provision is to consider the level of water provision. Chapter Two gives the results of an artificial and natural water availability study in Kruger National Park and seven South African Private Reserves. The private reserves had higher levels of artificial water provision than Kruger National Park and this provision did not follow natural water availability patterns. Areas of the landscape that are naturally wetter have vegetation that is better adapted to handle high herbivore impact (Milchunas *et al.*, 1988). Artificial water supplementation is therefore at potentially degrading levels and in high risk areas of the landscape.

Chapter Three is a second literature review that addresses the importance of spatial heterogeneity and resilience in conservation areas and looks at how these concepts link with water management. Spatial heterogeneity has important functional links with resilience (Suding *et al.*, 2004), so it is important that reserve management is aware of how ecosystem function varies with spatial heterogeneity. Currently, ecosystem function with regards to water provision is understood in terms of concentric circular patterns focused on waterpoints with low function close to water (Gaylard *et al.*, 2003). Due to the high level of heterogeneity in the southern African savannas (Pickett *et al.*, 2003) it is unlikely that this is an appropriate way to understand the functionality of the landscape.

When properties have a high risk approach to water supplementation, it is essential that there is a sound ecological basis for management. The applicability of the piosphere model in the southern African savannas is questionable so Chapter Four deals with testing the general broad-scale applicability of the approach in the heterogeneous southern African savannas. Initial piosphere papers state a requirement for homogeneity (Lange, 1969; Graetz & Ludwig, 1978) but the southern African savannas are highly heterogeneous (Pickett *et al.*, 2003). Testing the effect of distance to water across 23 ecological variables (soil functionality and herbaceous and woody vegetation) across 22 waterpoints from five properties revealed that only 14% of tests were significant. It was therefore concluded that the piosphere model is not generally applicable in the southern African savanna and as such, does not form a good basis for water management.

Informing property management that their basis of management is not applicable has no value unless an alternative approach is made available. Chapter Five presents work on the development of an alternative approach to understanding the functionality of southern African savanna conservation areas with regards to water provision. Environmental and management factors were both found to be influential on degradation levels across landscapes. Broad-scale factors such as property water supplementation level and natural water availability were found to be more important than fine-scale factors such as distance to an artificial waterpoint or catenal position. A basic characterisation system for properties and landscapes was developed.

The final two chapters both present a synthesis of the work from the first five chapters. Chapter Six is focused on an ecological synthesis of the study covering the current use of piosphere theory and problems with this approach through to development of an alternative approach which incorporates the understanding of savannas as a heterogeneous system. Chapter Seven focuses on a management synthesis of the study. It is very important that results of scientific studies are translated into management (Underwood, 1998; Roux *et al.*, 2006) so this shortened synthesis was written for a management audience.

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Acronyms

GRC	Greater Olifants River Conservancy
KLA	Klaserie Private Nature Reserve
KNP	Kruger National Park
MAL	Mohlabetsi Association of Landowners
LNP	Limpopo National Park
THB	Thornybush Private Nature Reserve
TIM	Timbavati Private Nature Reserve
UMB	Umbabat Private Nature Reserve
YRK	York Private Nature Reserve
GLTP	Great Limpopo Transfrontier Park
GLTFCA	Great Limpopo Transfrontier Conservation Area
SANP	South African National Parks
CA	Correspondence Analysis
CCA	Canonical Correspondence Analysis
DCA	Detrended Correspondence Analysis
DCCA	Detrended Canonical Correspondence Analysis
PCA	Principal Components Analysis
RDA	Redundancy Analysis
SD	Standard deviation
n	Sample size

Chapter 1

Origins of the piosphere concept and its modern application in savanna conservation: a critical review

Helen Farmer

ABSTRACT

Management of water provision is important in agriculture and conservation. In southern African savanna conservation areas, water supplementation is a contentious management intervention. The piosphere model has been used to understand the effects of water supplementation in these systems for the last 19 years. The piosphere model is based on a trade-off between water and forage requirements of water-dependent herbivores and was developed in homogeneous systems as an ecological and management unit. From 1988 there was a split between ‘piosphere’ studies which tend to not acknowledge heterogeneity and ‘grazing gradient’ studies which do acknowledge it. Through the 1990s, work continued around the world on piospheres and grazing gradients. There were varying degrees of success in application with some studies highlighting important disruptive effects caused by heterogeneity. Two reviews at the end of the 1990s reveal the weak focus on heterogeneity. The piosphere model became firmly entrenched in southern African conservation areas, despite their high levels of ecosystem heterogeneity. In the 2000s, waterpoint studies began to take more note of homogeneity when sampling, though the impact of this on interpretation of results was often not discussed. In 2009 it was highlighted that the spatial heterogeneity within the surroundings of waterpoints has ecological relevance. This paper reviews developments and changes in the use and application of piospheres and grazing gradients with particular reference to conservation areas in the highly heterogeneous southern African savanna.

KEY WORDS

Grazing gradient; herbivore impact; heterogeneity; homogeneity; water management

INTRODUCTION

Water management is important in agriculture and conservation. In many systems in the world with domestic and/or wild animals, water is a limiting resource (Valentine, 1947; Chamaillé-Jammes *et al.*, 2007). In domestic livestock systems, water management is important in paddock planning for maximum use of resources (Stafford Smith, 1991; Aucamp *et al.*, 1992). In conservation areas, water is currently supplemented for two broad reasons. The first reason is profit generation: water provision is used to boost herbivore population sizes through maximum use of vegetation resources (Aucamp *et al.*, 1992; Grossman *et al.*, 1999). The second reason is biodiversity enhancement: water provision is used to increase habitat heterogeneity and therefore biodiversity (Thrash, 1998b; Gaylard *et al.*, 2003).

In order to manage water, it is important to understand the impacts that supplementation has on the ecosystem. Prior to the late 1960s, impacts of water supplementation tended to be understood in terms of the effects on stocking rate (Valentine, 1947). Lange (1969) investigated impact on vegetation and coined the term 'piosphere' to describe the impact pattern created by water-dependent herbivores around waterpoints in water limited ecosystems. The piosphere theory states that degradation around a waterpoint is determined by the trade-off between forage and water requirements of the animals (Lange, 1969; Graetz & Ludwig, 1978). Piosphere size is limited only by the distance herbivores can travel between times when they must drink (Lange, 1969) and therefore becomes related to quality and quantity of forage around a waterpoint (Adler & Hall, 2005; Smit *et al.*, 2007).

When conservation areas in southern African savannas began managing water in 1933, the only available theory was agricultural (Aucamp *et al.*, 1992; Mabunda *et al.*, 2003). These theories were concerned with maximising production through complete use of forage resources (Grossman *et al.*, 1999; Mabunda *et al.*, 2003). Piosphere research began in the early 1990s in these systems (Thrash *et al.*, 1991a,b). The approach offered a way to model the system and determine herbivore utilisation levels across vast areas with multiple waterpoints (Redfern *et al.*, 2003; Smit *et al.*, 2007). Application of the piosphere model in management was seen as an opportunity to manage water provision to enhance biodiversity (Owen-Smith, 1996; Thrash, 2000). Today, water provision is a major management tool in southern African savannas

(Owen-Smith, 1996; Gaylard *et al.*, 2003). Waterpoint impact is understood in terms of circular piospheres with the management variables of interest being (1) the degradation level around a single waterpoint, and (2) the proximity of waterpoints, i.e. the probability that piospheres will merge.

The use of piosphere theory moved from research to conservation management very rapidly. Studies using the approach in the savanna found it to be successful, applicable and useful (Thrash, 1997, 1998a,b). However, studies in other areas highlighted problems with using piospheres in heterogeneous landscapes (du Plessis *et al.*, 1998; Nash *et al.*, 1999; Nangula & Oba, 2004). The savanna system is highly heterogeneous (Skarpe, 1992; Pickett *et al.*, 2003; Sankaran *et al.*, 2004) with functional mosaics at multiple scales (Scoones, 1995; Augustine, 2003). The level of heterogeneity found in southern African savannas suggests that greater care should be taken in using the piosphere approach. This paper aims to examine the development and changes in application of the piosphere model with specific reference to its assumptions and limitations. The focus of the paper is the southern African savannas, statistics will be provided based on a collection of waterpoint literature (80 papers) from 1932 to 2009 (Appendix 1).

FROM GRAZING PATTERNS TO PIOSPHERES

Historically, vegetation management was based on assessment of potential stocking rates (Valentine, 1947). In the early 1900s, range ecologists noted uneven utilisation of livestock paddocks which was dependent on waterpoint location (Valentine, 1947) and radial patterns of grazing symmetry (Osborn *et al.*, 1932). Zonation of impact was first described in a wildlife system by van der Schijff (1959). However, this study was published in Afrikaans in a local South African journal. Through the early to mid 1900s, the importance of vegetation state increased in animal management.

R.T. Lange, an Australian botanist, was interested in vegetation management based on vegetation state rather than animal condition. He therefore did a study to help management and understanding of vegetation in arid areas (Lange, 1969). A piosphere was originally thought of as an ecological unit for management in arid lands (Lange, 1969). To illustrate the piosphere effect, a study was done in South Australia in a paddock chosen for its smooth landscape and uniform vegetation

(Lange, 1969). Lange (1969) concludes that the piosphere model has a general applicability in any area where there is grazing by water-dependent herbivores from a central waterpoint.

In 1978 a paper was published that detailed an operational approach for investigating range condition based on piospheres which are taken as an ecological and management unit (Graetz & Ludwig, 1978). A logistic relationship was proposed for understanding the cumulative impacts of herbivores on vegetation (Graetz & Ludwig, 1978). This relationship states that over a constant vegetation type, the greatest degradation occurs close to the waterpoint and then there is a zone of change to a point far from the waterpoint where degradation tails off to negligible levels and the system shows its full ecological potential (Figure 1) (Graetz & Ludwig, 1978). The logistic relationship was tested across five sites with a variety of variables and a consistently good relationship was found (Graetz & Ludwig, 1978).

(i) **(ii)**

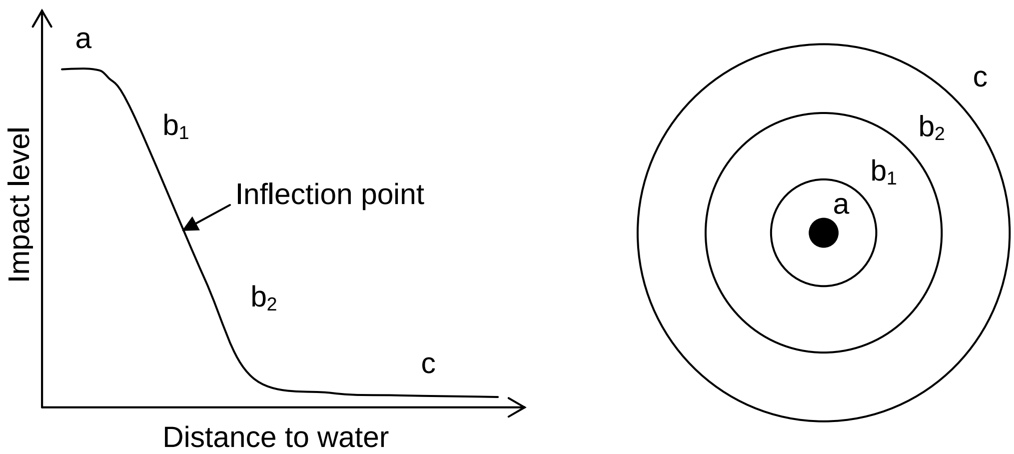

Figure 1: **(i)** The logistic curve of Graetz & Ludwig (1978) used to describe piosphere zones and **(ii)** concentric rings of different impact levels with rings corresponding to the logistic curve around a waterpoint indicated by a black circle. Zones are labelled as, a: sacrifice zone, poor condition; b_1: changing impact, fair condition; b_2: changing impact, good condition; and c: very little impact, excellent condition.

In the 1980s waterpoint studies were done in Australia (Foran, 1980; Lange, 1985; Pickup & Chewings, 1988), South Africa (Collinson, 1983) and Botswana (Tolsma *et* *al.*, 1987). Collinson (1983) presents a description of how variation in wildlife herbivore assemblages could cause circular impact patterns around waterpoints. In a study considering the effects of livestock on vegetation in Australia, vegetation types were found to be disruptive of the piosphere pattern but because piospheres are a controllable and measurable part of utilisation, the study concluded in their favour (Foran, 1980).

In 1988, two pivotal studies were published. Andrew (1988) wrote a review about the use of piospheres in domestic livestock systems with a small reference to application in conservation. Pickup & Chewings (1988) reported a study on modelling grazing and cattle distribution in a large paddock. Andrew (1988) published in *Trends in Ecology and Evolution* while Pickup & Chewings (1988) published in the *International Journal of Remote Sensing*. Availability of these journals resulted in widespread availability of the piosphere review but not of the modelling study. Andrew (1988) has been cited 90 times compared to Pickup & Chewings (1988) which has been cited 34 times (Source: Web of Science Citation Index) A key difference between the studies is the mention of heterogeneity: in the piosphere review it is not mentioned (Andrew, 1988) but in the grazing model it is important (Pickup & Chewings, 1988). Both studies conclude that distance to water is an important factor in determining herbivore impact through either piospheres (Andrew, 1988) or grazing gradients (Pickup & Chewings, 1988).

Following 1988 there was a split between usage of the terms ‘piosphere’ and ‘grazing gradient’ with only two studies using both terms. Forty-four percent of subsequent studies use ‘piosphere’, the other 56% use ‘grazing gradient’. Over time, the use of ‘piosphere’ has increased while the use of ‘grazing gradient’ peaked in the 1990s. The term ‘piosphere’ came to imply a concentric circular pattern (Tolsma *et al.*, 1987; Owen-Smith, 1996; Adler & Hall, 2005) whilst the term ‘grazing gradient’ was used with less implication of spatial patterning (Pickup, 1994; Rietkerk *et al.*, 2000). Both approaches use distance from water as a dominant variable in determining herbivore pressure on soils and vegetation. This paper will use the ‘grazing gradient’ terminology unless the author being referenced is specifically investigating an aspect of piospheres. Most research, even on piospheres, is done in a single direction and therefore can be better described as characterisation of a grazing gradient than description of a concentric circular pattern.

Over the following 20 years (1989 – 2009), Andrew and Pickup were both important authors with 36% of waterpoint studies quoting Andrew (1988) and 37% quoting studies by Pickup. Because of the low availability of Pickup & Chewings' (1988) paper, many subsequent studies quote Pickup (1994) published in *Journal of Applied Ecology*. These values can be compared with the 51% of waterpoint studies that quote Lange (1969). The proportional usage of Lange (1969) has declined over time with the increase in use of other studies (Figure 2). Many papers appear to reference Lange (1969) simply as the source of the term 'piosphere'.

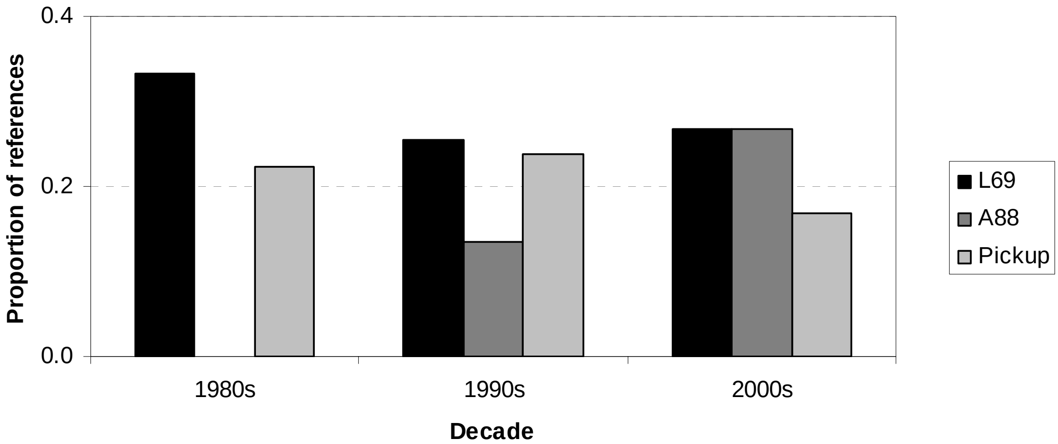

Figure 2: Proportion of references used in each decade by waterpoint studies that is made up by each key reference. L69 = Lange (1969), A88 = Andrew (1988), Pickup = any reference with Pickup as the primary author.

1990s: SPREAD OF WATERPOINT STUDIES

At the beginning of the 1990s, piosphere and grazing gradient studies start appearing from more countries (Figure 3). Studies appear from Senegal in 1991 (Hanan *et al.*, 1991), the United States in 1995 (Fusco *et al.*, 1995), and Namibia and Mali in 1998 (Turner, 1998; du Plessis *et al.*, 1998). Work continued in Australia (Stafford Smith, 1990), South Africa (van Rooyen *et al.*, 1990) and Botswana (Kalikawa, 1990). The dominance of studies from Australia decreased (Figure 4).

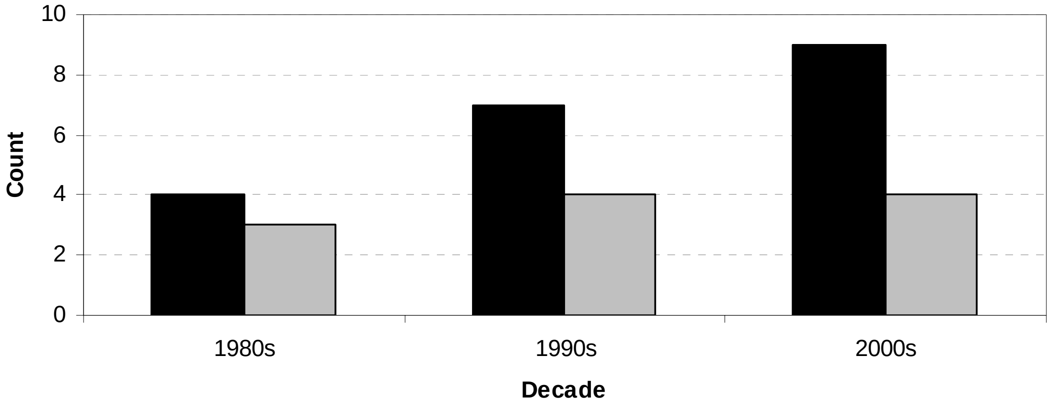

Figure 3: Number of countries continuing (black bars) and starting (grey bars) studies on waterpoints using the piosphere or grazing gradient approach.

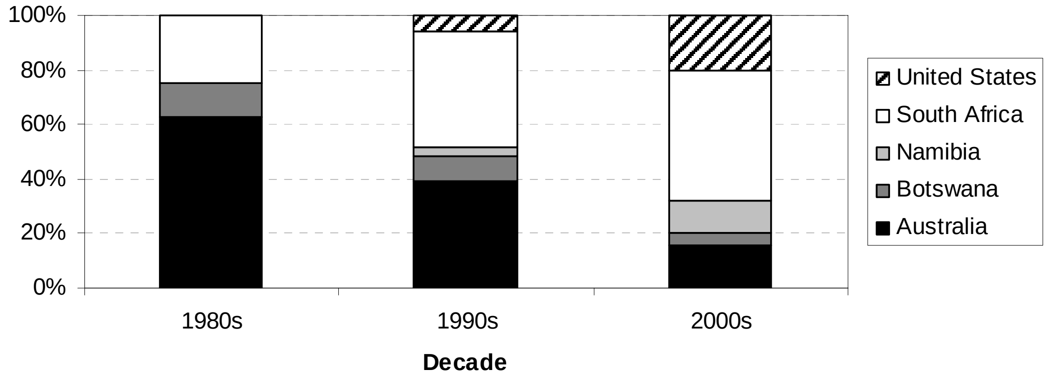

Figure 4: Representation of countries in the waterpoint literature (limited to countries represented in more than one decade). Sample sizes: 1980s = 8 studies, 1990s = 33 studies, 2000s = 25 studies.

The proportion of studies in wildlife systems nearly doubled from the 1980s to the 1990s (Figure 5). This increase is due to studies by I. Thrash in Kruger National Park, South Africa (Thrash *et al.*, 1991a,b; Thrash *et al.*, 1993, 1995; Thrash, 1997, 1998a,b). This decade marks the incorporation of grazing gradients into management of southern African conservation areas (Owen-Smith, 1996; Thrash, 1998b). Of the livestock studies performed in the 1990s, 60% are from Australia.

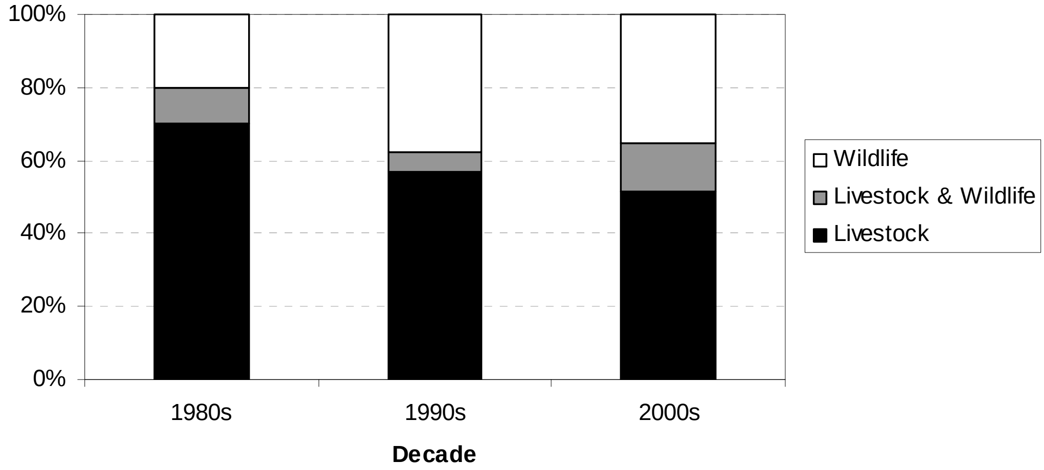

Figure 5: Differences between decades in animal systems studied. Sample sizes: 1980s = 10 studies, 1990s = 37 studies, 2000s = 31 studies.

Prior to the 1980s, waterpoint studies tended to focus either on herbivores (generally livestock) or on vegetation (Valentine, 1947; Lange, 1969). In the 1980s studies on soil and using remote sensing appear and in the 1990s a full set of variables appears with the start of waterpoint impact modelling (Figure 6). Modelling was used to determine optimal positioning of waterpoints (Stafford Smith, 1991; Owen-Smith, 1996) and to investigate vegetation change around waterpoints (Jeltsch *et al.*, 1997; Weber *et al.*, 1998). It was shown that piospheres could be modelled with clear patterns growing over time (Jeltsch *et al.*, 1997) and that vegetation heterogeneity can affect grazing patterns (Weber *et al.*, 1998).

Remote sensing studies dramatically increased in the 1990s with seven times the number of studies published in the 1980s. Remote sensing studies are considered separately from other vegetation studies here because of their different spatial scale, study extent and methodology. All the remote sensing studies investigated livestock impacts and covered large distances, at least 5km from waterpoints. The importance of remote sensing for detecting long-term trends was stressed (Pickup *et al.*, 1998). There were six studies from Australia (Cridland & Stafford Smith, 1993; Bastin *et al.*, 1993a; Bastin *et al.*, 1993b; Pickup, 1994; Pickup & Bastin, 1997; Pickup *et al.*, 1998) and one from Senegal (Hanan *et al.*, 1991).

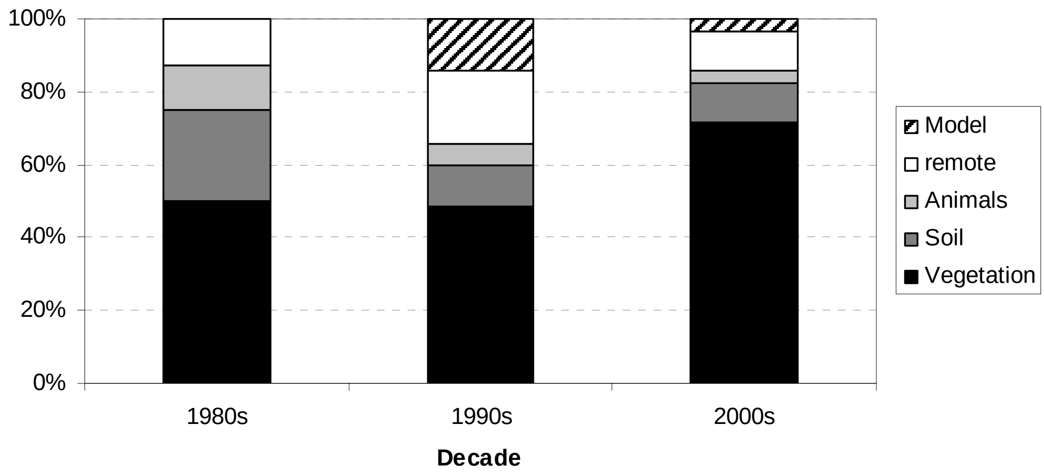

Figure 6: Variation in variables studied in waterpoint investigations. Sample sizes: 1980s = 8 studies, 1990s = 35 studies, 2000s = 28 studies.

In the 1990s a split formed between waterpoint studies that reference Andrew (1988) and those that reference papers by Pickup. The split is most obvious when considering references cited by studies focusing on different variables (Figure 7). Remote sensing studies cite work by Pickup while studies on animals and vegetation tend to reference Andrew (1988). The split causes a potential problem because of the lack of mention of heterogeneity in Andrew (1988).

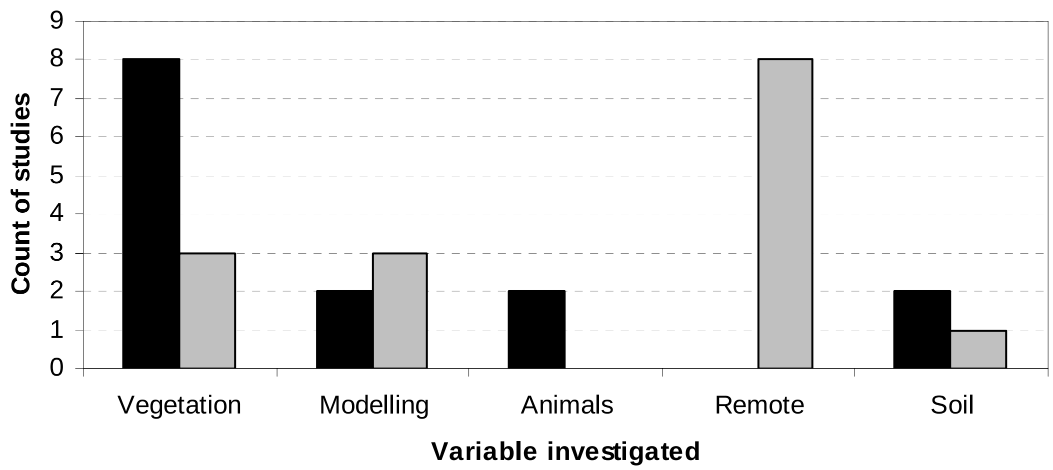

Figure 7: Differences in variables investigated by studies referencing Andrew (1988) (black bars) and papers by Pickup (grey bars). Studies that quoted both authors (9 studies, 16%) were removed.

Considering papers that reference Andrew (1988) and not Pickup in the 1990s, there are four studies with a variety of distances, variables and systems under consideration. All the studies conclude in favour of grazing gradients and all lack consideration of heterogeneity (Thrash *et al.*, 1995; Owen-Smith, 1996; Moleele & Perkins, 1998; Thrash, 1998a). Conversely, all nine studies which reference Pickup without Andrew (1988) discuss the importance of heterogeneity. Grazing gradients were found to be useful for management of livestock systems (Stafford Smith, 1990; Stafford Smith, 1991).

Although several studies referencing Pickup without Andrew (1988) found problems with underlying environmental heterogeneity (Hanan *et al.*, 1991; Hodgins & Rogers, 1997; Weber *et al.*, 1998), it was generally agreed that a grazing gradient can be superimposed over environmental heterogeneity (Bastin *et al.*, 1993a; Pickup, 1994; Pickup & Bastin, 1997; Pickup *et al.*, 1998). The only study from a wildlife system that referenced Pickup without Andrew (1988) was Parker & Witkowski (1999) who specifically avoided topographical gradients when investigating the impact of waterpoints on vegetation. They conclude in favour of grazing gradients with no reference to the implications of heterogeneity (Parker & Witkowski, 1999).

There were nineteen studies in the 1990s that referenced neither Andrew (1988) nor Pickup. Eighty-four percent of these concluded in favour of grazing gradients. Study scales varied from 0.06km (Thrash, 1997) to 7km (Thrash *et al.*, 1991a,b) in wildlife systems and 1.5km (Navie *et al.*, 1996) to 5km (Turner, 1998) in livestock systems. All studies either made no mention of heterogeneity or controlled for homogeneity in their methods. Herbivore impact levels were found to vary between different soils and vegetation but authors concluded that grazing gradients were still apparent (Kalikawa, 1990). When heterogeneity did cause a problem in the results, its effects were removed to make grazing gradients an appropriate model (Bosch & Gauch, 1991).

Studies that concluded against grazing gradients had problems with environmental gradients that were stronger than grazing gradients (van Rooyen *et al.*, 1994; Friedel, 1997; du Plessis *et al.*, 1998; Nash *et al.*, 1999; Turner, 1999). These studies came from both wildlife and livestock systems over a variety of variables. Some papers specifically concluded that heterogeneity has important disruptive effects on grazing gradient impact patterns (Cridland & Stafford Smith, 1993; Nash *et al.*, 1999). A remote sensing study that referenced both Andrew (1988) and Pickup found that vegetation type affects detection of grazing gradients (Bastin *et al.*, 1993b). Although grazing gradients were understood to be a simplistic approach (Pickup, 1994), the need for management information drove their continued use.

At the end of the 1990s, two major reviews on piospheres were published that both referenced Andrew (1988) and work by Pickup. One came from work done in South Africa and focused on describing piospheres from conservation areas (Thrash & Derry, 1999). This review does not explicitly mention heterogeneity and concludes in favour of piospheres as a generally applicable concept (Thrash & Derry, 1999). The other review was from Australia and focuses on the message that piospheres have a wide variety of effects throughout an ecosystem (James *et al.*, 1999). Heterogeneity is mentioned in this review as being important because it can distort the piosphere pattern (James *et al.*, 1999). Both of these reviews tend towards the Andrew (1988) perspective that piospheres are generally applicable and good for research and management. James *et al.* (1999) reference Pickup & Chewings (1988) for their mention of vegetation type distorting the piosphere pattern. Thrash & Derry (1999) reference multiple Pickup papers but only as a methodology source.

HOMOGENEITY vs. HETEROGENEITY

The recognition of heterogeneity in models applied in management is of particular importance in the southern African savanna. Spatial heterogeneity refers to the amount and spatially explicit configuration of environmental resources and constraints across a landscape (Pickett *et al.*, 2003). Herbivores perceive functional heterogeneity in vegetation and can respond by altering their utilisation patterns (Bailey, 1995; Gómez *et al.*, 2004). Spatial heterogeneity is scale dependent with different agents at different scales (Urban *et al.*, 1987; Coughenour, 1991; Pickett *et al.*, 2003). Interaction between top-down and bottom-up drivers results in spatial patterning and the creation of a series of ecosystem mosaics at different scales (Scoones, 1995; Augustine, 2003; Bestelmeyer *et al.*, 2006).

Homogeneity refers to a lack of landscape or environmental variation such as topography, soil or vegetation that could affect animal distribution or behaviour

(Bailey *et al.*, 1996). There are references to homogeneity in early piosphere studies: Lange (1969) notes that most piospheres are not as regular as the one he studied and that a change in vegetation can disrupt the piosphere pattern. Using the logistic relationship to describe piospheres requires a homogeneous environment where herbivore movement and utilisation is affected solely by the trade-off between water and forage requirements (Graetz & Ludwig, 1978).

Through the 1990s, studies looking for grazing gradients in heterogeneous areas concluded with varying results on their applicability (Thrash, 1997; Verlinden *et al.*, 1998; du Plessis *et al.*, 1998; Thrash, 1998b). African savanna conservation areas have a higher level and importance of spatial heterogeneity (Skarpe *et al.*, 2000; Pickett *et al.*, 2003; Augustine, 2003) than the relatively simple system within which the piosphere approach was developed (Andrew, 1988). Heterogeneity is important in waterpoint studies as 92% of rejections of grazing gradients have problems with heterogeneity. Only 17% of acceptances of grazing gradients mention difficulties caused by heterogeneity.

Because of the interactions between drivers and mosaics, scale is an important factor to consider alongside heterogeneity. In studying grazing gradients in heterogeneous areas, scale is important to note in two areas: (1) the sampling distance covered, and (2) the variable under consideration (Brown & Allen, 1989). Different sampling distances can lead to differing dominance of ecological and grazing gradients (Friedel, 1997). Broad scale variables such as vegetation productivity can give different patterns to smaller scale species effects (Hanan *et al.*, 1991; Riginos & Hoffman, 2003). This can be seen in different studies from heterogeneous areas which had different levels of consideration of heterogeneity (e.g. Fernandez-Gimenez & Allen-Diaz, 2001; Heshmatti *et al.*, 2002).

2000s: CONSIDERING HOMOGENEITY AND HETEROGENEITY

Use of grazing gradients and piospheres spread further in the 2000s (Figure 3) with new studies from Burkina Faso (Rietkerk *et al.*, 2000), Mongolia (Fernandez-Gimenez & Allen-Diaz, 2001), Tanzania (Tobler *et al.*, 2003) and Zimbabwe (Chamaillé-Jammes *et al.*, 2007). There was a further drop in the proportion of studies from Australia and an increase in the proportion of studies from South Africa (Figure

4). The percentage of studies performed in mixed wildlife/livestock systems or comparing the impacts of wildlife and livestock more than doubled from 5% in the 1990s to 13% in the 2000s (Figure 5). There was an increasing dominance of vegetation studies during this decade (Figure 6). Only one modelling study was performed, showing how simple grazing rules can lead to development and growth of piospheres over time (Adler & Hall, 2005).

In this decade there were seven vegetation studies that reference Andrew (1988) without Pickup, one from a wildlife system and six from livestock systems. The complexity of variables under consideration increased to variables such as system patchiness (Rietkerk *et al.*, 2000) and vegetation composition and reproduction (Riginos & Hoffman, 2003). In all studies, homogeneity was controlled for in methods (e.g. soil type kept constant during sampling (Brits *et al.*, 2000)). There was a wide scale range from 0.2km to 10km but all seven studies concluded in favour of grazing gradients. Controlling for homogeneity appears to result from an increased general awareness of homogeneity and heterogeneity (Kotliar & Wiens, 1990; Coughenour, 1991) as the impacts of heterogeneity on interpretation of results were not discussed.

There were three other studies that referenced Andrew (1988) without Pickup. A soil study compared differing effects of wildlife and livestock (Smet & Ward, 2006) and an animal and a modelling study looked at effects in wildlife systems (Redfern *et al.*, 2003; Ryan & Getz, 2005). Homogeneity and heterogeneity were not mentioned in any of these studies. For the modelling and animal studies, the landscapes were smoothed with water acting as the only attractor (Redfern *et al.*, 2003; Ryan & Getz, 2005), removing the potential impact of heterogeneity completely.

There were only three studies from this decade that referenced Pickup without Andrew (1988). Two were on vegetation and the third was remote sensing, all in livestock systems. They all concluded in favour of grazing gradients with grazing gradients being labelled as good (Tobler *et al.*, 2003) and piospheres as "universally accepted" (Hunt, 2001). One study notes that a grazing gradient was not found at seasonal waterpoints (Harris & Asner, 2003).

Studies referencing neither Andrew (1988) nor papers by Pickup decreased to only eight, all from outside Australia. These were split between four livestock and four wildlife studies. Only two studies controlled for homogeneity (Brits *et al.*, 2002; Getzin, 2005) and one study considered heterogeneity (Makhabu *et al.*, 2002). In general, longer transects or transects with more replicates found more problems with heterogeneity where environmental gradients were stronger than herbivore gradients (Fernandez-Gimenez & Allen-Diaz, 2001; Makhabu *et al.*, 2002). One study removed noise (caused by heterogeneity) in order to obtain a grazing gradient (Getzin, 2005). A study from South Africa with no acknowledgement of heterogeneity concluded in favour of grazing gradients (Beukes & Ellis, 2003).

A further eight studies referenced both Andrew (1988) and papers by Pickup, an increase from the 1990s. Five of these studies controlled for homogeneity (Thrash, 2000; Legget *et al.*, 2003; Landsberg *et al.*, 2003; Adler & Hall, 2005; Smet & Ward, 2005), one specifically included heterogeneity (Nangula & Oba, 2004) and the remaining two do not mention homogeneity in the method (Washington-Allen *et al.*, 2004; Chamaillé-Jammes *et al.*, 2007). Again, when vegetation type was considered it was found to have a stronger effect than grazing gradients (Nangula & Oba, 2004).

The increasing control for homogeneity (Figure 8) led to an increased acceptance of grazing gradients as an appropriate approach to understanding herbivore impact. 84% of studies in the 2000s concluded in favour of grazing gradients compared to 70% in the 1990s. Authors state that grazing gradients are a strong, well proven concept, accepted everywhere (Thrash, 2000; Hunt, 2001; Landsberg *et al.*, 2003). By this stage, grazing gradients were deeply entrenched in waterpoint management of savanna conservation areas. Studies on water availability on these properties simply investigated distance to water, the basis of grazing gradients (Cronje *et al.*, 2005; Ryan & Getz, 2005; McDonald, 2005; Chamaillé-Jammes *et al.*, 2007). Piospheres were also linked to the evolution of African herbivores (Derry & Dougill, 2008).

There is increasing recognition of the importance of heterogeneity in savanna systems (Venter *et al.*, 2003) and it is important that ecosystem management be based on the best current models available (Christensen *et al.*, 1996; Roux *et al.*, 2006). Currently, management of water in southern African savanna conservation areas is not in line with ecological theories (Friedel, 1991). Understanding of waterpoint impact and subsequent waterpoint management in the southern African savannas needs to acknowledge heterogeneity. Averaging values to remove the effects of heterogeneity (Bosch & Gauch, 1991; Getzin, 2005) is not appropriate as it does not make ecological sense (Chamaillé-Jammes *et al.*, 2009). Heterogeneity within the surroundings of waterpoints has ecological relevance and is important to management (Chamaillé-Jammes *et al.*, 2009).

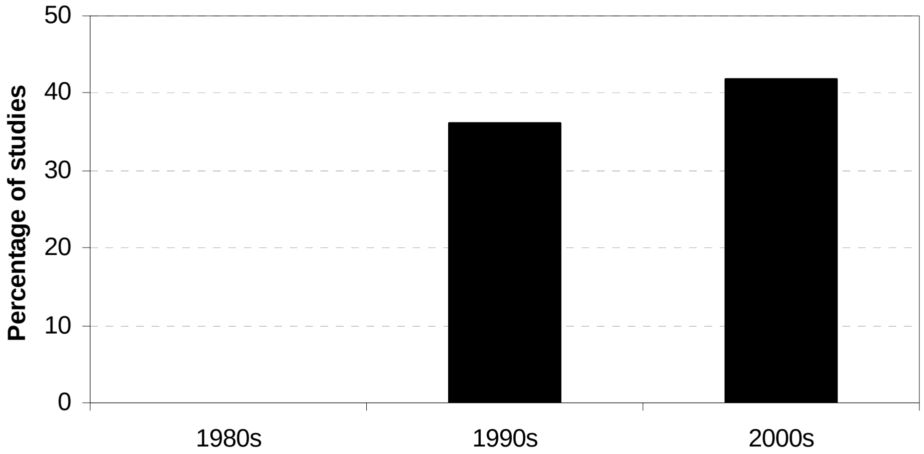

Figure 8: The percentage of studies which specifically controlled for homogeneity in their sampling design when investigating impact patterns around waterpoints.

CONCLUDING REMARKS

Water management is important in agricultural and conservation systems and understanding the impacts of installation of artificial waterpoints on the ecosystem is crucial for proper management. Understanding of waterpoint impact has changed over the past 60 years (Figure 9). Management and understanding of waterpoints using piospheres and grazing gradients works in homogeneous systems or in areas where heterogeneity can be removed from analyses as it lacks management importance. In the southern African savanna conservation areas, heterogeneity has both ecological and management importance. Therefore, it is very likely that the piosphere/grazing gradient approach is not suitable as a basis for understanding and management in these systems. This statement needs to be formally tested, and if correct, a new approach to understanding and managing impact around waterpoints in the southern African savanna needs to be developed.

The problems highlighted in this review lead to a number of recommendations for future waterpoint studies:

1. Terminology must be considered before use. The explicit 'distance to water' is recommended instead of 'piosphere' or 'grazing gradient'. If one of these terms is to be used, 'grazing gradient' is preferred as it lacks the implication of concentric circle patterning unless concentric circle patterning is truly being investigated.
2. The scale of the study (distances from water that sampling occurred at and extended to) needs to be explicitly stated in the methods. Twenty-eight percent of studies in this database do not state their distance sampled.
3. Homogeneity and heterogeneity need to be specifically considered and their impact on interpretation of results should be stated.
4. An explicit statement of whether the method is suitable for moving from the description of one (or a few) transect(s) to discussion of 'a radial grazing pattern extending out from waterpoints' should be included.
5. An explicit statement of whether the study results are suitable for scaling up across properties for management purposes should be included. This is particularly important where waterpoints are not separated by fences.

Grazing gradients and piospheres form an important part of rangeland and conservation research and management. However, their application needs to be considered in relation to the heterogeneity of the study and/or management area. Both approaches are simplifications of the system, based on underlying assumptions of homogeneity. The suitability of such a simplified approach should be considered carefully before it is used.

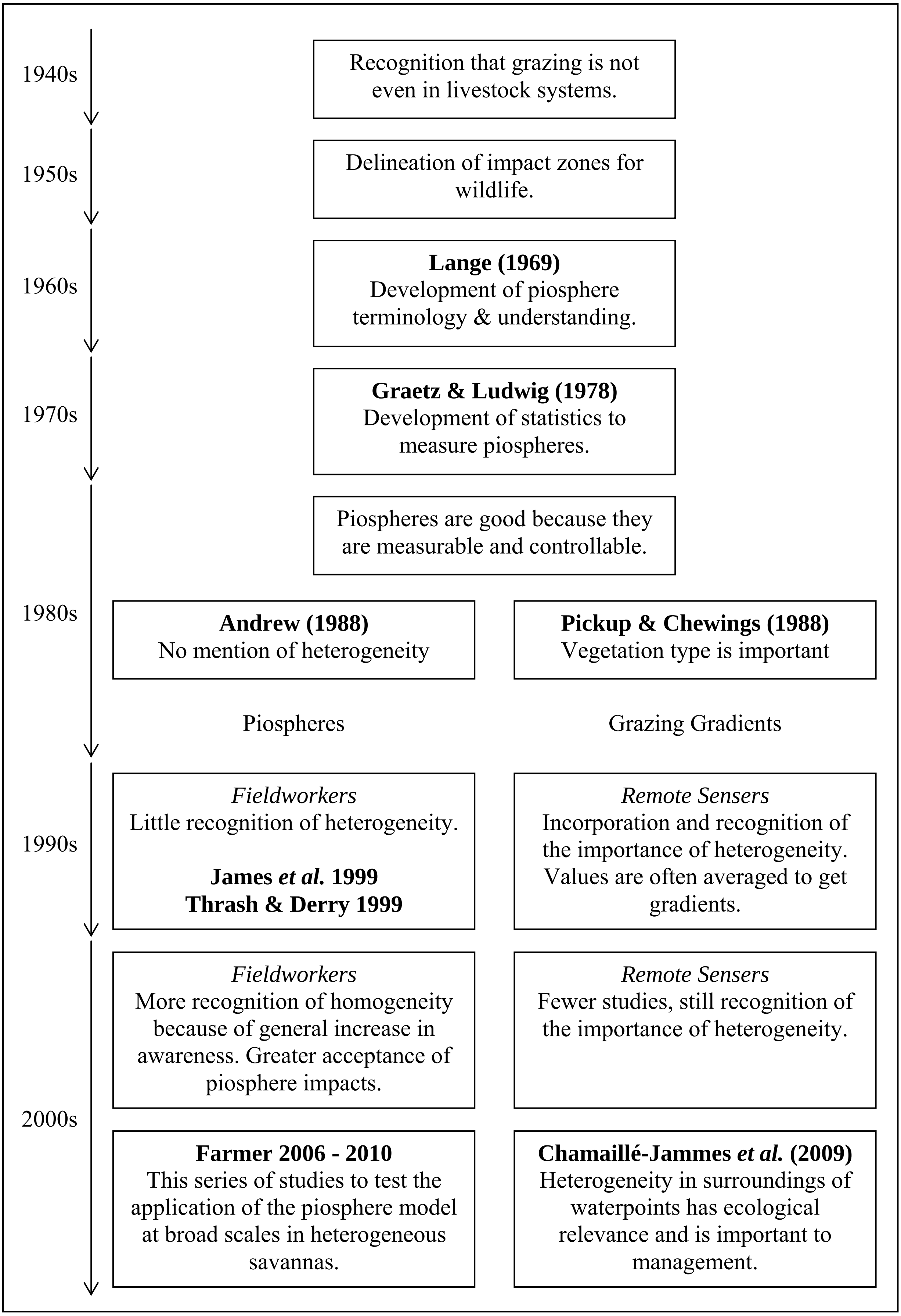

Figure 8: Timeline of the development of understanding in studies of the impacts of waterpoints and application of their results. Key studies are highlighted in bold.

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Chapter 2

Conservation management and water supplementation: artificial waterpoint density and positioning

Helen Farmer

ABSTRACT

Artificial supplementation of permanent water is a key management intervention in southern African savannas. This study analysed databases from seven private reserves (intensively managed) and a national park (medium management intensity) in South Africa to determine how artificial waterpoint density reflects conservation management and whether artificial water availability patterns follow natural patterns. Aerial census records of water at the end of the dry season were used to generate natural water availability scores for 35 landscape types. Comparison with drainage lines from topographic maps was used to determine artificial waterpoint catenal position. Artificial waterpoint density varied from 0.008 to 0.492 points/km^2 and increased with management intensity (Spearman $r = 0.946$, $n = 7$, $p<0.05$). Property maximum nearest neighbour distance was negatively correlated with management intensity (Spearman $r = -0.933$, $n = 8$, $p<0.05$) and smaller properties had more regularly distributed waterpoints (Spearman $r = 0.952$, $n = 8$, $p<0.05$). Artificial water provision followed natural patterns in the national park (Ephemeral: $\chi^2_{34} = 0.452$, $p>0.05$; Permanent: $\chi^2_{34} = 0.547$, $p>0.05$) but not in the private reserves (Ephemeral: $\chi^2_{20} = 1408$, $p<0.01$; Permanent: $\chi^2_{20} = 2464$, $p<0.01$) because they consist primarily of dry landscape types. Percentage of waterpoints within drainage lines varied from 13% to 54% per property but did not correlate with management intensity (Spearman $r = 0.618$, $n = 8$, $p>0.05$). More intensely managed properties had lower natural water availability and therefore were forced to supplement in drier landscape types. However, waterpoints on these properties tended to be found in more natural catenal positions.

KEY WORDS

Catena; degradation; piosphere; savanna; water availability

TERMINOLOGY

Property water availability: natural and artificial water sources available to animals at the end of the dry season (i.e. permanent water sources)

Artificial water availability: permanent artificial water sources available to animals at the end of the dry season

Natural water availability: ephemeral and permanent natural water sources available to animals at the end of the dry season

Natural water availability score: numeric value that represents the natural water availability

Ephemeral water availability: natural water sources available to animals at the end of the dry season whose occurrence is not reliable between years

Ephemeral water availability score: numeric value that represents the ephemeral water availability

Permanent water availability: natural water sources available to animals at the end of the dry season whose occurrence is reliable between years

Permanent water availability score: numeric value that represents the permanent water availability

Water patterns: the spatial distribution of water availability across a property (artificial or natural)

INTRODUCTION

The southern African savannas are a key conservation area with the Great Limpopo Transfrontier Conservation Area (GLTFCA) forming the most extensive large mammal conservation area in the world (Peace Parks, 2005). Originally part of a large scale migratory system, Kruger National Park (KNP) and bounding privately owned properties have been gradually fenced off into smaller conservation areas (Walker *et al.*, 1987; Mabunda *et al.*, 2003). Creation of the GLTFCA involves removal of fences between properties with the objective of re-establishing migration routes and aligning conservation management approaches (Peace Parks, 2005). One of the key management actions to address in achieving these aims is artificial water supplementation (Owen-Smith, 1996; Gaylard *et al.*, 2003).

Currently, two broad objectives are found for water provision. The first, profit generation, is found on properties where management aim to boost herbivore population sizes through maximum use of vegetation resources (Aucamp *et al.*, 1992; Grossman *et al.*, 1999). The second, biodiversity enhancement, is found on properties where management aim to increase habitat heterogeneity to boost biodiversity (Thrash, 1998a; Gaylard *et al.*, 2003). Although all properties subscribe to the same broad conservation objective, factors that differ between them affect management decisions (Peel *et al.*, 1999). There is great variation in property size with a range of $50km^2$ to $20\ 000km^2$. Smaller properties do not have space for a natural disturbance regime (e.g. fires) and therefore require more intensive management (Baker, 1992; Peel *et al.*, 1999). Fencing interrupts natural ecosystem function, leading to a requirement for greater management intensity (Forman & Godron, 1981).

Variations in management intensity have been linked to variations in management objectives. Extensively managed properties emphasise biodiversity conservation objectives, intensively managed properties emphasise objectives that facilitate tourism. Management practices to enhance tourism are not always the same as those to enhance biodiversity (Aucamp *et al.*, 1992; Peel *et al.*, 1999; Craine *et al.*, 2009). Variations in management objectives are reflected in water provision levels. Properties emphasising biodiversity conservation strive for a level of water provision which increases habitat heterogeneity and therefore biodiversity (Thrash, 1998a; Smit *et al.*, 2007). Properties emphasising tourism have a higher level of water provision to ensure full use of property resources (Aucamp *et al.*, 1992; Grossman *et al.*, 1999).

Waterpoint density, the level of water supplementation, is an important consideration because of the effects that herbivores using waterpoints can have on surrounding vegetation and soils. As water constrains herbivore population size by limiting available forage area in the dry season, artificial supplementation uncouples herbivore populations from their natural limitations (Illius & O'Connor, 1999; Cronje *et al.*, 2005; Chamaillé-Jammes *et al.*, 2007a). Repetitive grazing, browsing and trampling around permanent waterpoints leads to degradation of soil and vegetation (Lange, 1969; Graetz & Ludwig, 1978; Adler & Hall, 2005). Density of artificial waterpoints on a property indicates how much of the landscape is potentially under unnatural pressure from herbivores. Properties with a higher density of waterpoints theoretically have a greater spread of unnatural pressure.

Previous studies in the GLTFCA area have often only considered property waterpoint density as important when discussing water supplementation (Owen-Smith, 1996; Thrash & Derry, 1999; Thrash, 2000; Smit *et al.*, 2007). At a finer scale, the position of waterpoints within a landscape could affect the level of degradation around them. Some landscape types naturally hold water longer into the dry season because of rainfall and soil types (Rietkerk *et al.*, 1997; Gaylard *et al.*, 2003; Venter *et al.*, 2003). Within landscapes, the position of the waterpoint with reference to the topography (the catenal position) is also important. Vegetation within drainage lines is more resistant to higher repetitive pressure from herbivores (Milchunas *et al.*, 1988; Milton, 1991; Rietkerk *et al.*, 1997).

Water supplementation is a contentious management issue in the southern African savanna, so it is important to understand both differences in water supplementation between properties and reasons for the differences. To fully understand the importance of water management decisions, we need to understand the risk associated with these decisions. In order to determine the potential degradation risk associated with artificial supplementation, this study investigated (1) how artificial water availability reflects management intensity, and (2) how closely artificial waterpoint density and positioning follow natural water patterns. This provides the baseline risk assessment of water provision in the study area.

METHODS

Study area

The study was performed within the South African portion of the GLTFCA. Limpopo National Park (Mozambique) was not included in this section of the study because it has no artificial water provision. The study area has mild dry winters and hot wet summers with rainfall generally decreasing on a south to north and east to west gradient (Venter *et al.*, 2003). There is a slight temperature cooling trend from the north to the south (Venter *et al.*, 2003). The geology of the area is granitic in the west and basaltic in the east separated by a thin sedimentary strip (Venter *et al.*, 2003). The granitic landscapes have an undulating topography (Venter *et al.*, 2003) with a higher stream density than the basalts (Gaylard *et al.*, 2003). Basalts are generally flat with high nutrients and high water holding capacity (Redfern *et al.*, 2003; Venter *et al.*, 2003; Smit *et al.*, 2007). Granites are characterised by strong catenas with sandy gravely crests and clay bottomlands (Venter *et al.*, 2003). The granite crests have low nutrients and do not hold water well whereas the bottomlands have a moderate level of nutrients and high water holding capacity (Venter *et al.*, 2003).

Management Intensity Index

The management intensity index is a general reflection of the intensity of management on a property. The presence of fencing, the size of the property and the emphasis on tourism or biodiversity were used to infer management intensity. A higher score in the management intensity index means that the property has less natural management. Fencing was scored as a 0 for absent or a 1 for present. Size of the property was split into four classes with the lowest score going to the largest size class: <100km^2 scored 4, 100-500 km^2 scored 3, 500-1000 km^2 scored 2, and >1000 km^2 scored 1. Management emphasis was scored as a 0 for a biodiversity emphasis and a 1 for a tourism emphasis. Scores for each variable were summed to give the management intensities of each property (Figure 1). Property details are given in Table 1 and their locations in Figure 2.

Table 1: Summary of management intensity and emphasis and property size for properties of the Great Limpopo Transfrontier Park included in the study. See text for details on calculation of management intensity score.

Name	Abbreviation	Size (km^2)	Management intensity score	Management emphasis
Mohlabetsi Association of Landowners	MAL	48	6	Tourism
York Private Nature Reserve	YRK	49	5	Tourism
Thornybush Private Nature Reserve	THB	115	5	Biodiversity/ Tourism
Greater Olifants River Conservancy	GRC	105	4	Biodiversity/ Tourism
Umbabat Private Nature Reserve	UMB	195	4	Biodiversity/ Tourism
Timbavati Private Nature Reserve	TIM	533	3	Biodiversity/ Tourism
Klaserie Private Nature Reserve	KLA	578	3	Biodiversity/ Tourism
Kruger National Park	KNP	18956	1	Biodiversity

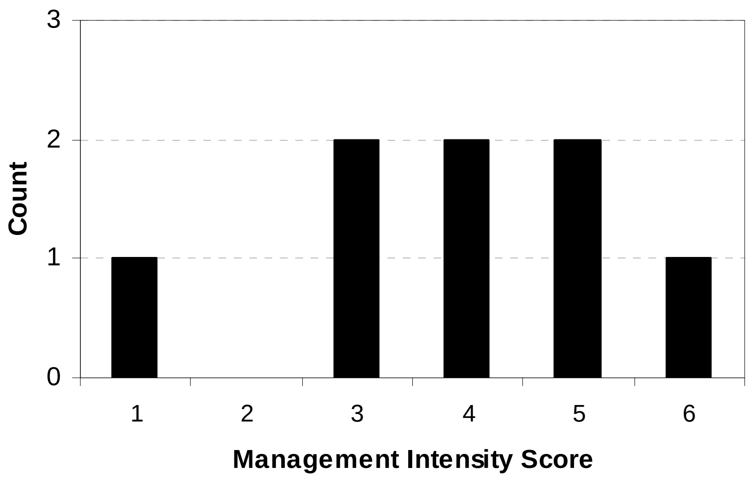

Figure 1: Frequency distribution of management intensity scores (see text for details on calculation) for properties in the study area.

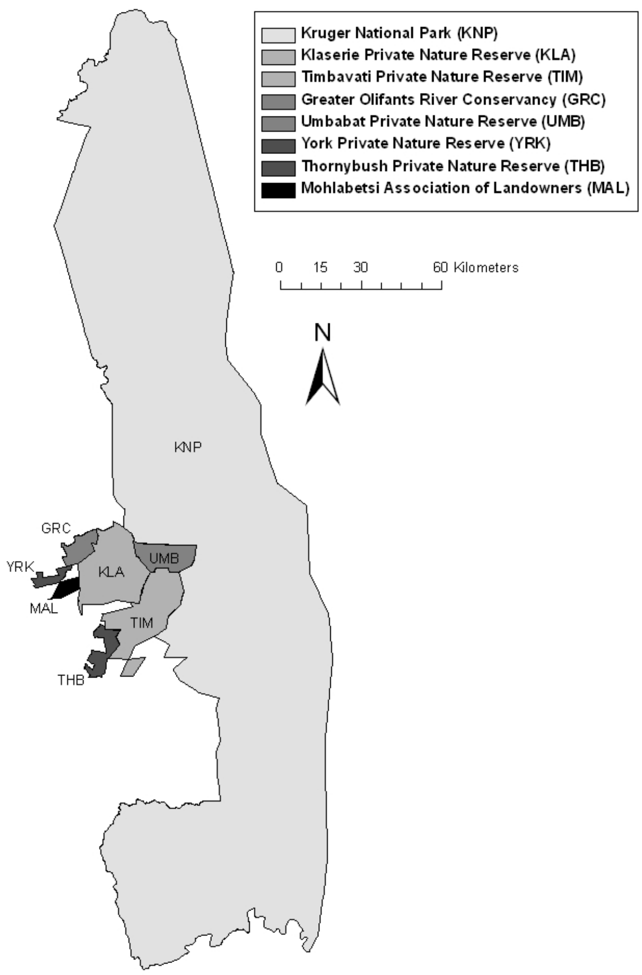

Figure 2: Map of the study area with darker shades representing more intensive management.

Data collection

This study was based on analysis of property artificial waterpoint databases. The most accurate data were from GPS records of waterpoint location. Assuming the data collector took the reading as accurately as possible, the points have a 10m error. The only dataset for dams in KNP that was available at the time of the study was created by digitising from 1:50 000 topographic maps, and thus has a lower accuracy.

Only permanent waterpoints, those which last throughout the annual wet-dry cycle of a moderate rainfall year, were used for analysis (Figure 3). Databases of KNP, GRC, THB, YRK, UMB and MAL clearly stated waterpoint permanence. For KLA and TIM, data from five years of aerial census of water points at the end of the dry season were used to determine the reliability of waterpoints. Waterpoints recorded in more than 75% of censuses were taken as permanent.

Aerial census data from KNP were used to calculate natural water availability (obtained from KNP Scientific Services in 2006). Aerial censuses were flown at the end of the dry season and water present was recorded from 1981 to 2001. During the census, water was recorded as 'river flow', 'pools', 'rivers' and 'pans'. For this study, these data types were simplified into two groups: ephemeral and permanent. Ephemeral and permanent water types were analysed separately because of their different geomorphologies (Ayeni, 1977; Gaylard *et al.*, 2003; Venter *et al.*, 2003). Census data recorded as 'pans' was used for the ephemeral data set and census data recorded as 'river pools', 'pools' or 'rivers' was used for the permanent data set. There were 14 years of data available for ephemeral data sources (1981, 1982, 1984, 1986-1990, 1992, 1993, 1997-1999, 2001) and 17 years for permanent data sources (1981-1984, 1986-1993, 1997-2001). Artificial water sources, recorded as 'waterholes', were excluded from this analysis because they corresponded with borehole records.

Where possible, digital property boundaries were obtained from managers. In other cases, boundaries were determined from digital cadastral data and were confirmed with reserve management. The area of each reserve was calculated in ArcGIS (ESRI, 1999).

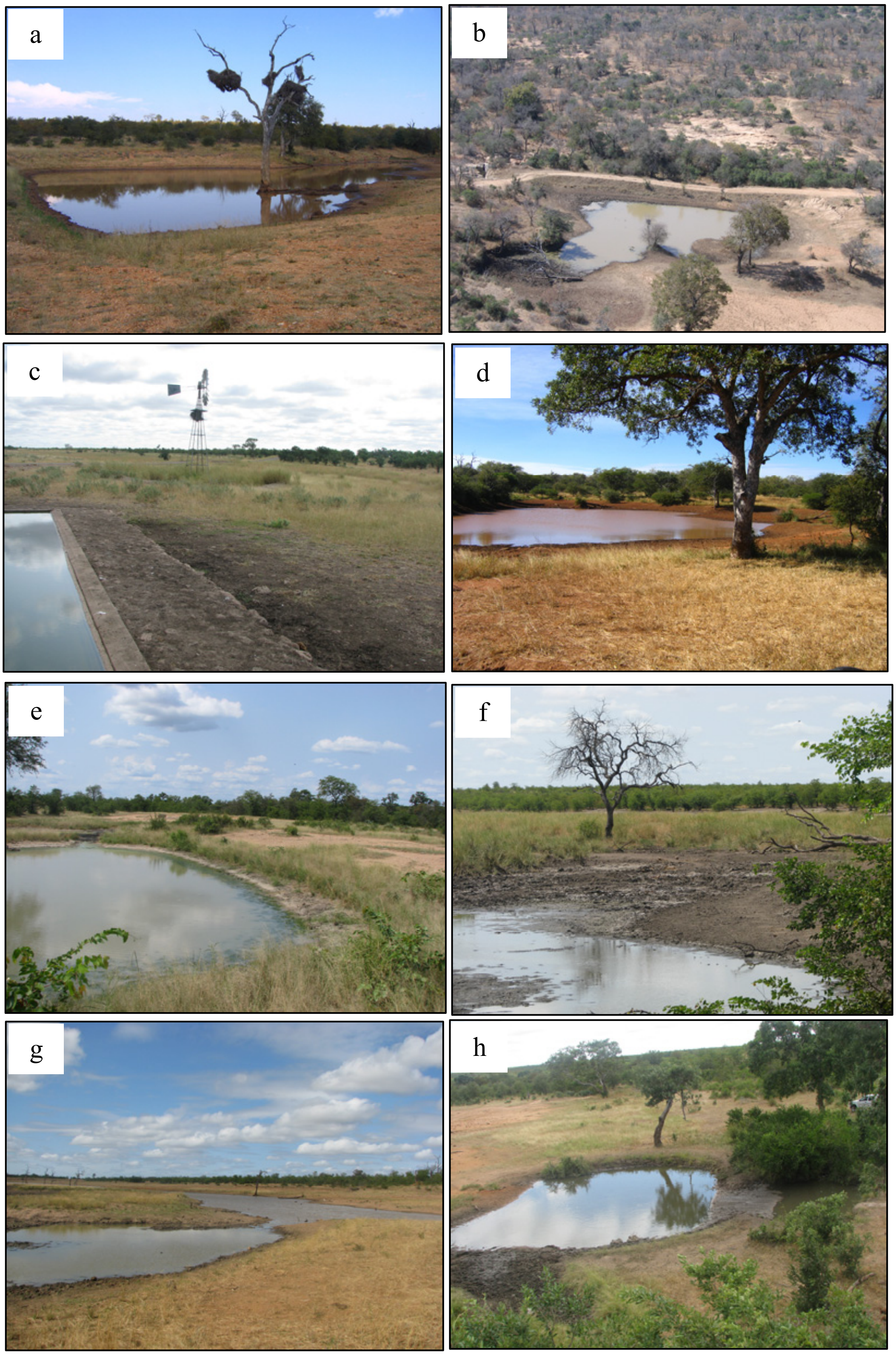

Figure 3: Photographs of examples of permanent waterpoints from the study area. a = permanent dam from UMB, b = aerial photograph of permanent dam from THB, c = permanent trough from KNP, d = permanent dam from MAL, e - h = permanent pumped dams from KNP.

Landscape types were used to subdivide properties and compare the natural and artificial water patterns. Landscapes for KNP were taken from Gertenbach (1983) and for the private reserves from Peel *et al.* (2007). Landscape types were determined by a combination of geomorphology, soil, climate, vegetation and faunal characteristics (Gertenbach, 1983) and therefore give a better representation of differences in potential to hold surface water into the dry season than vegetation type alone (Ayeni, 1977; Gaylard *et al.*, 2003).

Data analysis

Description of artificial water availability

In order to compare current conceptions of differing management objectives, three measurements of artificial water availability were calculated: density, nearest neighbour (NN) distances and nearest neighbour standard deviation (NN_{SD}). Density, waterpoints/km^2, gives an overall indication of the level of supplementation. NN distances indicate an estimate of the probability of impact stretching between waterpoints. Across properties, NN_{SD} indicates the irregularity of waterpoint spacing as it increases when regularity decreases.

The relationships between management intensity and (1) density of water provision, (2) average NN distance, (3) maximum NN distance, and (4) NN_{SD} were tested using Spearman Rank Order Correlations. The relationship between property size and NN_{SD} was also tested using Spearman Rank Order Correlation.

Comparison of artificial and natural water availability

a) Location (landscape type)

Natural water availability of landscape types was calculated from KNP census data. Census points from each year were converted into density rasters which were then converted into two classes: presence or absence of water. Reclassified rasters were added together to give a value for each cell which represented the total number of years that water was present, the water recurrence. Water recurrence rasters were converted into polygons and then each landscape type was clipped out.

The proportion of each landscape type covered by different water recurrence years was calculated. This was used to calculate the natural water availability score for both ephemeral and permanent water:

$$\text{Ephemeral/Permanent water availability score} = \frac{\sum_{0 \to n} \text{Proportion of landscape in recurrence year} \times \text{Recurrence year value}}{\text{Years of available data}}$$

where n is the maximum number of years that water was present in the landscape. These natural water availability scores therefore take into account the inter-annual variability in water availability as well as the coverage of water and have a minimum value of 0 and a maximum value of 1. A value of 0 would indicate that no water of that type is ever found in the area at the end of the dry season. A value of 1 (very unlikely) would indicate the area is covered in permanent water (e.g. a very large dam). The closer to 1 a natural water availability score is, the higher the quantity and/or reliability of the water. The distinction between quantity and reliability can be made by comparing the ephemeral and permanent water availability scores.

Natural water availability scores were tested for a relationship between ephemeral water availability and permanent water availability using correlation. Natural availability scores were compared with artificial water availability to investigate whether artificial water patterns followed natural water patterns. Distribution of artificial waterpoints between landscape types was compared to a predicted distribution based on ephemeral and permanent water availability scores with χ^2 tests. Private reserves were tested as a group due to the much lower number of landscape types present on each property. Regression was used to test if artificial waterpoint distribution was related to ephemeral or permanent natural water availability.

b) Position (catenal)

Analysis of waterpoint position within the catena was limited to waterpoint locations recorded by GPS. Buffers were calculated for waterpoint positions based on GPS accuracy (10m). Data collected in YRK were used to analyse the accuracy of drainage line position and therefore to determine the drainage line buffer size. Distance was measured from each waterpoint to the nearest drainage line in ArcGIS. Photographs of the waterpoints were then assessed to determine waterpoint location in relation to drainage lines. These notes were used to determine a buffer size of 20m for the drainage lines. Buffer sizes were tested on the detailed data and all waterpoints known to be located within drainage lines were selected.

Waterpoint buffers were intersected with drainage line buffers to determine the percentage of a property's waterpoints that were located within the drainage line. This was compared between management intensities using a Spearman Rank Order Correlation. This analysis did not differentiate between types of drainage line (perennial or non-perennial) because vegetation differs in all drainage lines.

RESULTS

Artificial water availability

Waterpoint densities varied widely between properties from 0.008 (KNP) to 0.492 (YRK) (Table 2). The overall average density was 0.20 points/km^2 (sd = 0.16). The lowest private reserve waterpoint density (THB) was from a reserve with a management intensity score of 5. However, rainfall and natural water availability are both higher on this property than on most other private reserves (Table 2). When all reserves were included, management intensity and density of artificial waterpoints were not significantly correlated (Spearman r = 0.655, n = 8, $p>0.05$). When THB was removed from the analysis, the correlation was significant (Spearman r = 0.946, n = 7, $p<0.05$), reserves with more intense management had a higher water provision level.

Average NN distances vary from 1.98 km to 21.9 km but do not reflect management intensity (Spearman r = -0.478, n = 8, $p>0.05$). Several properties have very low minimum NN distances (e.g. minimum = 0 km in UMB) because different types of waterpoint are installed in close proximity (Figure 4). Maximum NN distance is negatively correlated with management intensity (Spearman r = -0.933, n = 8, $p<0.05$).

NN_{SD} varied from 0.53 km to 3.35 km and was negatively correlated to management intensity (Spearman r = -0.837, n = 8, $p<0.05$). It was found that NN_{SD} is also related to property size (Spearman r = 0.952, n = 8, $p<0.05$) with smaller properties having more regularly distributed waterpoints. Average NN distance and NN_{SD} were not significantly correlated (Spearman r = 0.524, n = 8, $p>0.05$; Figure 5).

Table 2: Property management intensity, artificial waterpoint density and natural water availability. See text for details on management intensity score and Table 1 for reserve full names.

Property	Management intensity score	Artificial density (points/km^2)	MAR [a] (mm)	Natural availability [b]
KNP	1	0.008	375-925	0.042
GRC	4	0.143	394	0.022
YRK	5	0.492	405	0.022
UMB	4	0.277	413	0.022
MAL	6	0.357	431	0.010
KLA	3	0.118	462	0.023
TIM	3	0.130	481	0.052
THB	5	0.113	616	0.035

[a] Mean Annual Rainfall

[b] Natural availability is calculated as the average value of the permanent water availability scores for landscapes of the property

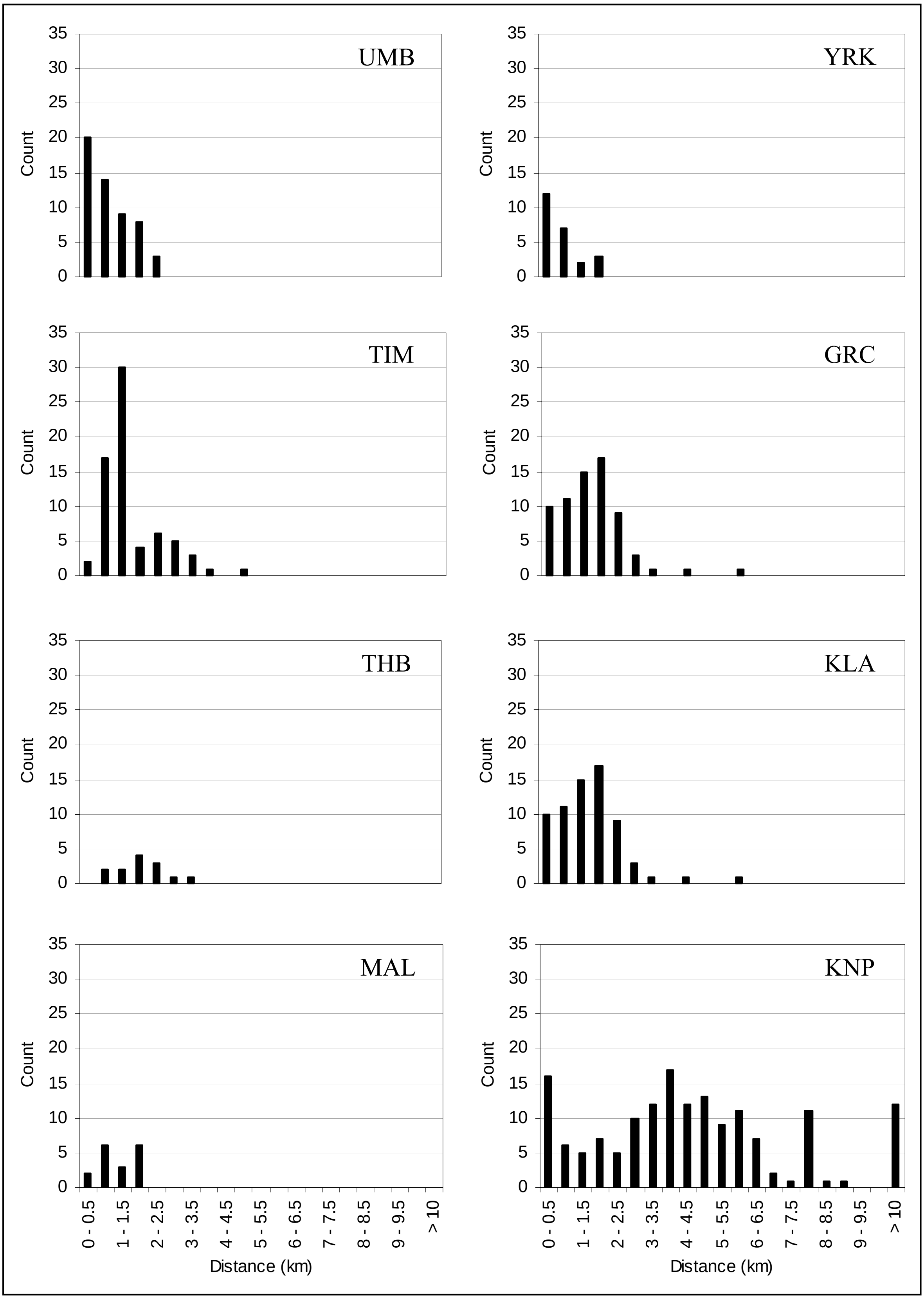

Figure 4: Frequency distributions of nearest neighbour distances for each property. See Table 1 for full names of properties.

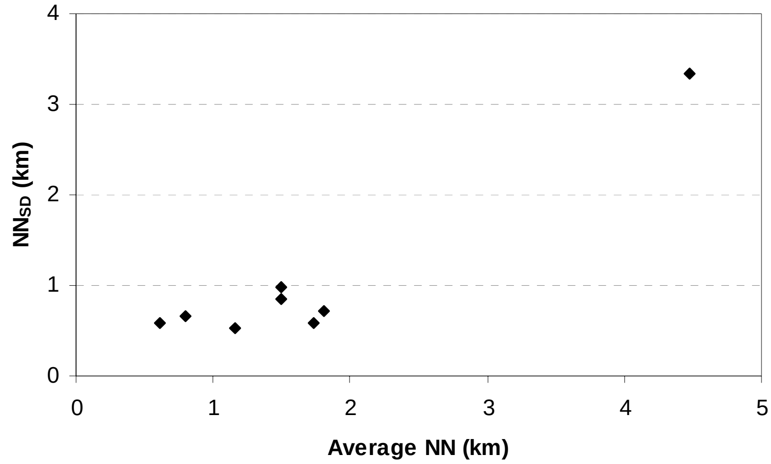

Figure 5: The relationship between property average nearest neighbour distance (NN) and nearest neighbour standard deviation (NN_{SD}).

Comparison of artificial and natural water availability

There was a wide variation in ephemeral and permanent water availability scores for KNP landscape types (Table 3; Figure 6, 7; Appendix 2). All landscape types had ephemeral water at some point in the data collection period but not all had permanent water. There was no significant relationship between ephemeral natural water availability and permanent natural water availability of a landscape type ($r = 0.019$, $n = 35$ $p>0.05$). The difference in reliability is illustrated by ephemeral water sources having a maximum recurrence of 50% of years whilst permanent water sources have a maximum recurrence of 94% of years (Figure 8). Average recurrence was 32% of years for ephemeral water sources and 66% of years for permanent water sources.

Table 3: Descriptive statistics for natural water availability scores calculated from aerial census of natural water at the end of the dry season over thirty-five landscapes in the Kruger National Park, South Africa. See text for detailed description of, and calculation methods for, water availability scores.

	Ephemeral water availability score	Permanent water availability score
Maximum landscape score	0.062	0.206
Minimum landscape score	0.002	0
Average landscape score	0.018	0.042
Standard deviation	0.012	0.040
Median	0.016	0.034

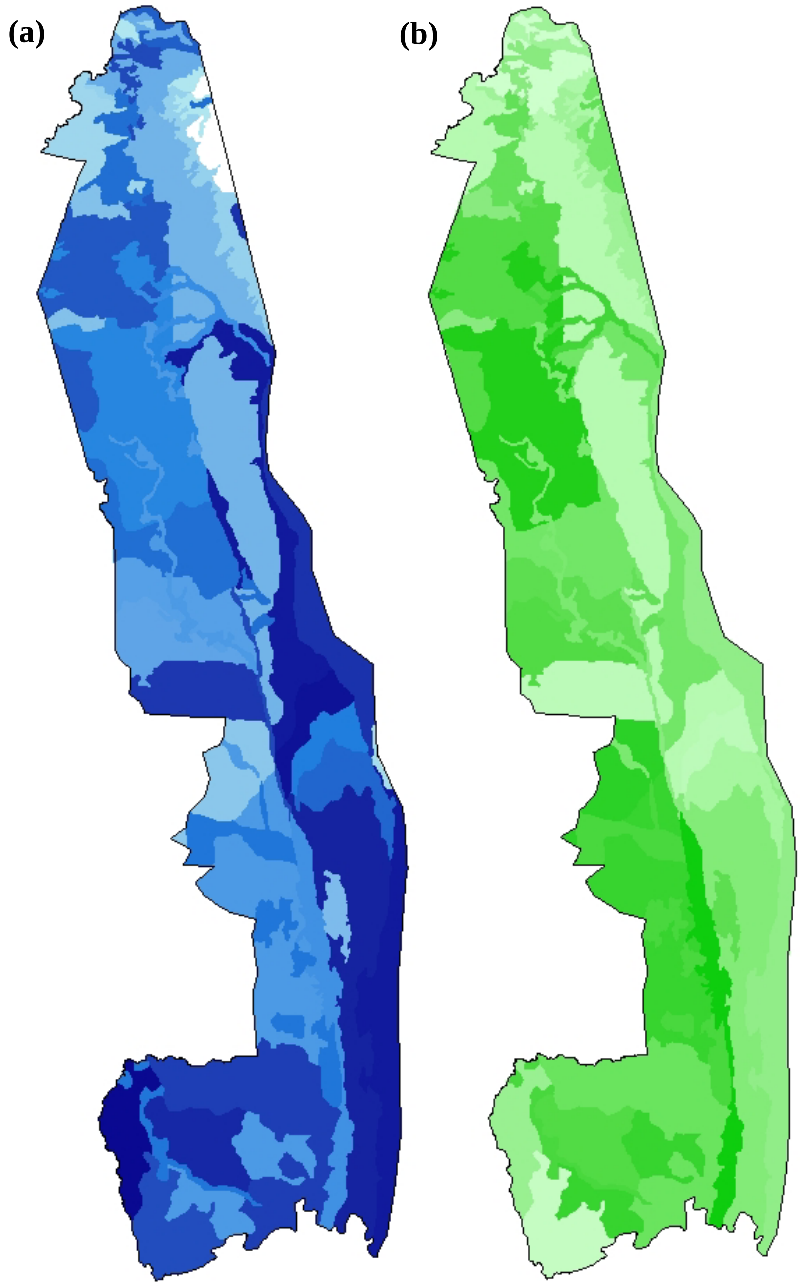

Figure 6: Natural water availability scores for landscape types of Kruger National Park based on **(a)** permanent water availability, and **(b)** ephemeral water availability. Darker shading indicates wetter landscapes. See text for detailed description of, and calculation methods for, water availability scores.

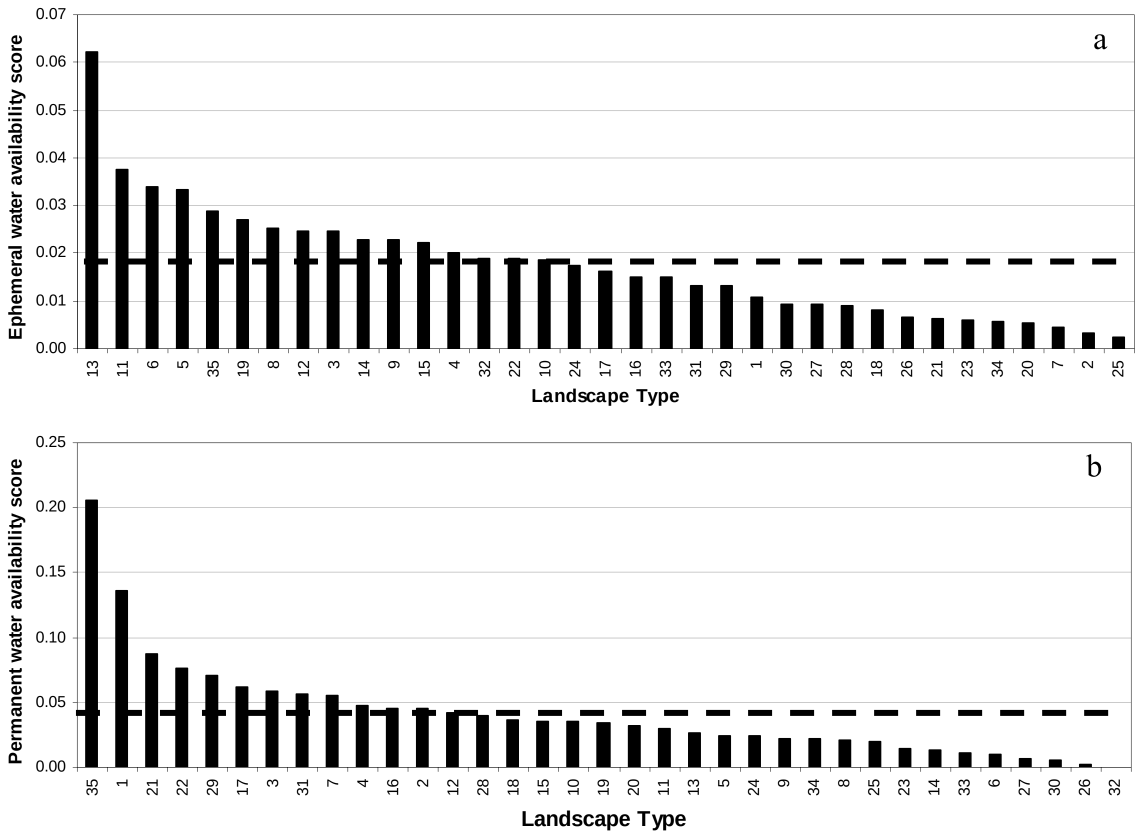

Figure 7: Landscape type water availability scores for (a) ephemeral natural water availability, and (b) permanent natural water availability for the 35 landscapes of Kruger National Park, South Africa. Dashed lines indicate average scores. Note different y-axis scales. Full names for landscapes can be found in Appendix 2.

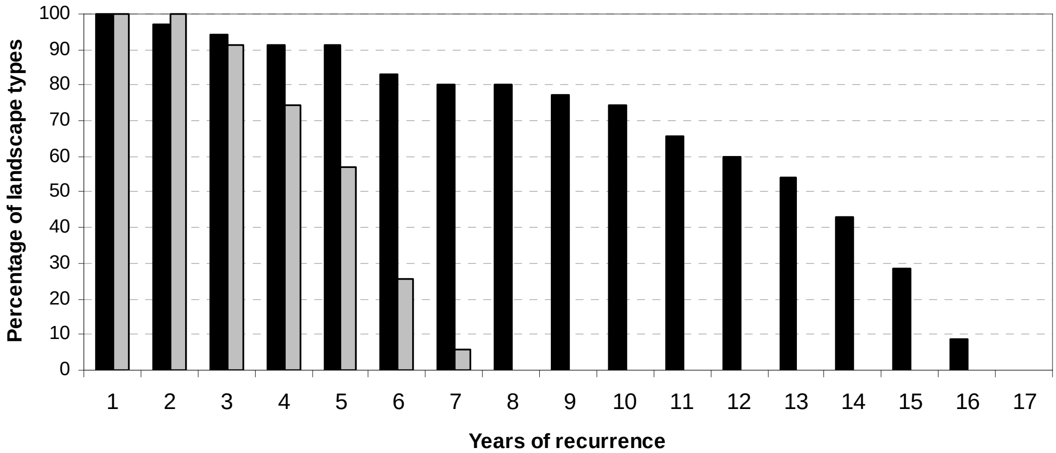

Figure 8: Reliability of ephemeral and permanent natural water sources shown through years of recurrence over landscapes in Kruger National Park, South Africa. Black bars represent permanent natural water (maximum 17 years of data); grey bars represent ephemeral natural water (maximum 14 years of data).

a) Location (landscape type)

The distribution of artificial waterpoints within KNP is not significantly different to the expected distribution calculated based on natural water availability (Ephemeral: $\chi^2 = 0.452$, df = 34, p>0.05; Permanent: $\chi^2 = 0.547$, df = 34, p>0.05). It was found that the artificial water distribution follows permanent water availability rather than ephemeral water availability (Ephemeral: r = 0.234, $F_{1,33} = 1.91$, p>0.05; Permanent: r = 0.396, $F_{1,33} = 6.13$, p<0.05). Following low impact conservation management objectives, artificial water supplementation follows natural permanent water patterns though there is considerable variation around the relationship (Figure 9).

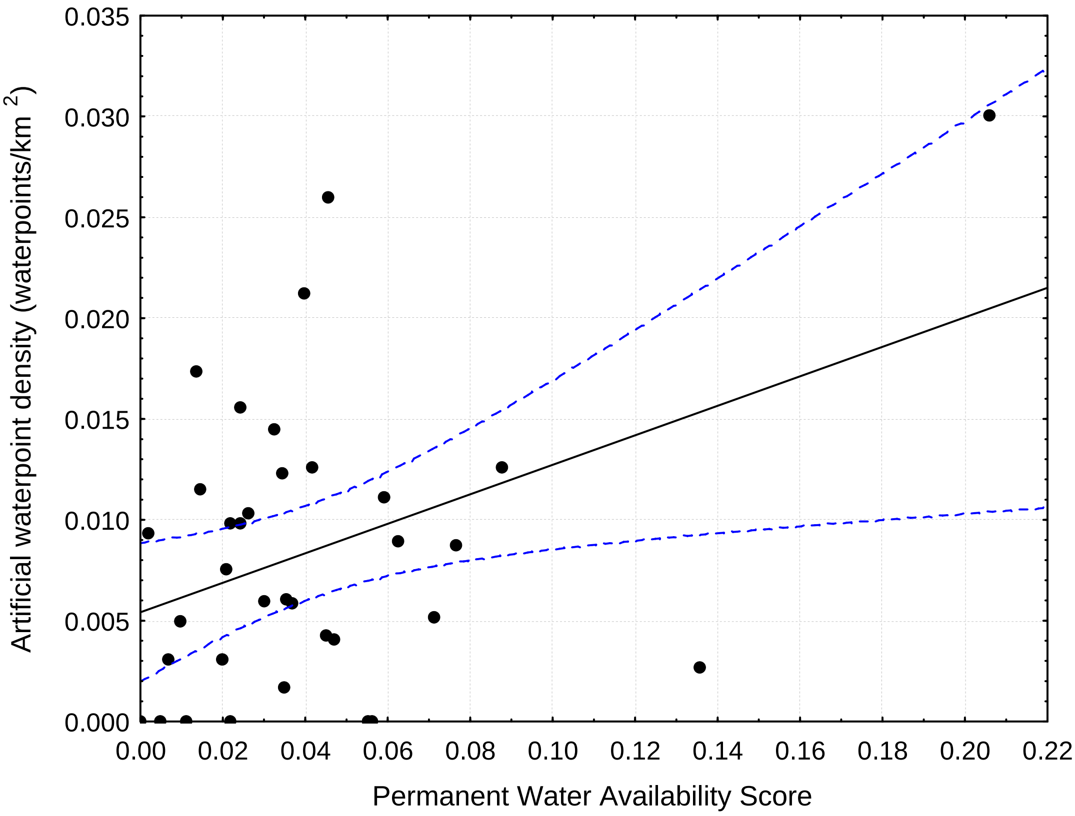

Figure 9: The relationship between landscape type natural permanent water availability score and density of artificial waterpoints in Kruger National Park, South Africa.

In the private reserves, the distribution of waterpoints is significantly different to what would be expected if waterpoints were distributed according to natural water availability (Ephemeral: $\chi^2 = 1408$, df = 20, p<0.01; Permanent: $\chi^2 = 2464$, df = 20, p<0.01). Expected waterpoint densities for landscapes in the private reserves vary from 0.042 to 1068 points/km^2 based on ephemeral water availability and 0.006 to 1862 points/km^2 based on permanent water availability. The private reserves are primarily made up of drier landscape types. The expected waterpoint densities of 1068 and 1862 points/km^2 occur because of the very small areas of wetter landscape types on the private reserves. 83% of the private reserves (over 99% for four individual properties) is made up of Landscape 6 (*Combretum*/*Colophospermum mopane* woodland on granite) which ranks 3rd wettest for ephemeral water availability but only 31st for permanent water availability (Figure 7). Only three of the top ten wettest landscape types for permanent water availability are represented in the private reserves and their combined area accounts for only 3%.

Catenal Position

There was a wide range of the proportion of waterpoints within a drainage line from 13% to 54% (Table 4). Proportion of waterpoints in drainage lines did not correlate with property management intensity (Spearman $r = 0.618$, $n = 8$, $p>0.05$) though there was a general increase in percentage in drainage line with increasing management intensity (Figure 10). There were two obvious outliers, THB had a much higher proportion of waterpoints in the drainage line and GRC a much lower proportion than anticipated.

Table 4: The percentage of waterpoints occurring within drainage lines and management intensity of properties (restricted to waterpoints with locations recorded by GPS). See Table 1 for full reserve names.

Reserve	Management intensity	Total waterpoints	% in drainage
THB	5	13	54
MAL	6	17	41
UMB	4	54	30
KLA	3	68	26
YRK	5	24	25
TIM	3	69	23
KNP	2	125	17
GRC	4	15	13

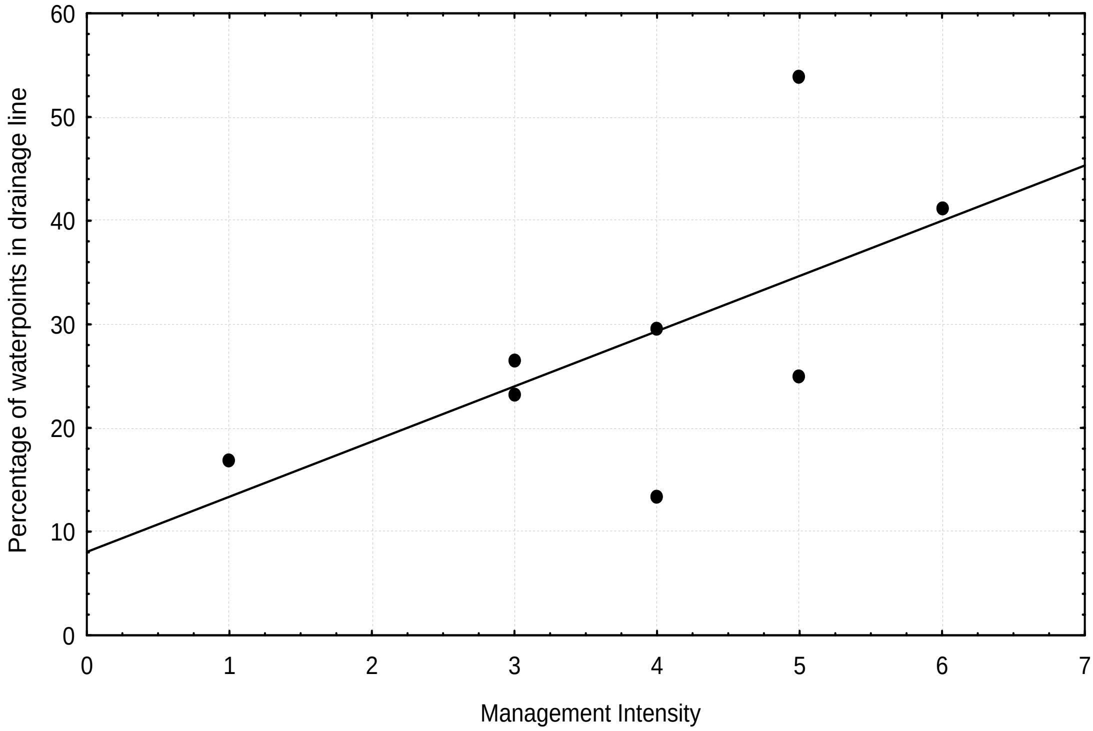

Figure 10: The percentage of waterpoints in drainage lines plotted against management intensity for eight properties of the Great Limpopo Transfrontier Conservation Area.

DISCUSSION

It is often generally stated that artificial water availability increases with intensity of management (Owen-Smith, 1996; Thrash & Derry, 1999; Grossman *et al.*, 1999). This study found conclusively that properties with higher intensity management had a higher density of waterpoints. The most intensely supplemented landscape of KNP had an artificial waterpoint density nearly four times below that of the more intensely managed private reserves. In conjunction, nearest neighbour distances are shorter on more intensively managed properties. Previous research has stated that inter-waterpoint distances of over 10km are required to maintain vegetation outside the dry season foraging distance of water-dependent herbivores (Owen-Smith, 1996). Close positioning of waterpoints leads to full use of property resources but the long-term sustainability of this approach has been questioned (Craine *et al.*, 2009).

Previous studies have recommended the use of clumped waterpoint distributions to maximise the area outside the impact of herbivores during the dry season (Owen-Smith, 1996; Thrash, 2000). Management intensity and property size affected regularity of waterpoints. Waterpoints with an inter-waterpoint distance of 0km were not combined for analyses because different types of waterpoints attract different types of herbivores (McDonald, 2005) and therefore have potentially different impacts on surrounding vegetation and soils. Smaller properties rarely have the space to work with a 10km inter-waterpoint distance, for example MAL is approximately 11km by 5km and includes separately owned private properties. Internal property boundaries can influence waterpoint locations as each owner wants a waterpoint on their property (Farmer, 2007).

Compression of plant and animal communities into small fenced areas can lead to homogenisation of vegetation structure and composition (Owen-Smith, 1996; Peel *et al.*, 1999) and intensification of management actions (Forman & Godron, 1981; Baker, 1992; Peel *et al.*, 1999). The lowest private reserve waterpoint density was found in THB, a fenced reserve. This reserve is naturally wetter than the other private reserves indicating that comparisons between properties need to be aware of other factors that contribute to management decisions.

Calculation of natural water availability revealed that landscape types vary considerably in their ability to provide water for herbivores into the dry season with no relationship between permanent and ephemeral water availability. Patterns for permanent water availability are consistent with those found by Redfern *et al.* (2005) through analyses based on distance to water. A combination of the techniques used in this study and those of Redfern *et al.* (2005) could lead to a more accurate water distribution description. Redfern *et al.* (2005) highlight the potential importance of ephemeral water sources in herbivore distribution patterns. The lack of relationship between permanent and ephemeral water sources suggests that more research into patterns of availability of ephemeral water sources is urgently needed.

The naturally wettest landscape type of KNP was Landscape 35, *Salvadora angustifolia* Floodplains. This landscape type had the highest permanent water availability score, 152% of the second wettest landscape type and nearly five times the average permanent water availability score. Landscape 13, *Acacia welwitschii* Thickets on Karoo Sediments, had the highest score for ephemeral water availability. The driest landscape in terms of ephemeral water availability was Landscape 25, *Adansonia digitata*/*Colophospermum mopane* Rugged Veld which is closely related to Landscape 26, *Colophospermum mopane* Shrubveld on Calcrete, the second driest for permanent water availability. The driest landscape in terms of permanent water availability was Landscape 32, Nwambiya Sandveld. This landscape consists of recent sand plains with *Baphia massaiensis* bush savanna. The closely related Landscape 30, Pumbe Sandveld, also had very low permanent water availability.

When managing for a natural system in conservation it is important for artificial water supplementation patterns to follow natural patterns as closely as possible. Southern African savanna systems have evolved with herbivore pressure for millions of years (Bouchenak-Khelladi *et al.*, 2009). Under a natural system, areas of the landscape with permanent water would have received higher utilisation pressure from herbivores during the dry season (Chamaillé-Jammes *et al.*, 2007a,b). Supplementing water in less natural areas leads to an increased risk of potential degradation.

Variations in soil and vegetation sensitivity to herbivore pressure means that location of waterpoints in different landscape types is important. Despite this, investigations of water supplementation often only determine property density (Thrash & Derry, 1999; Thrash, 2000; Smit *et al.*, 2007), with reference to different landscape types being found only in the study site description (Thrash, 1997, 1998b; Brits *et al.*, 2002). This study found that the KNP artificial water distribution follows what would be expected if the distribution was based on natural permanent water occurrence, in line with their biodiversity conservation objectives. Four landscape types were identified with high densities of artificial water, all of which are in the group of the ten smallest landscapes of KNP.

In the private reserves, the artificial water pattern did not follow the natural water pattern. The private reserves consist primarily of Landscape 6 (*Combretum*/*Colophospermum mopane* woodland of Timbavati) which has high ephemeral but low permanent water. In some years the reserves will be very wet and in others very dry. Because of the dominance of Landscape 6, the majority (86%) of waterpoints are installed in this landscape. It is important to note that this unnatural supplementation pattern is due simply to available landscape area. Again, this highlights the fact that external factors contributing to management actions must be considered when understanding and interpreting the effects of management. The smaller size, and therefore limited landscape availability, results in more unnatural management.

At a finer scale, within landscapes, topographic position can influence the risk of degradation of soil and vegetation around a waterpoint. Granitic landscapes in the area are characterised by strong catenas with shallow, sandy/gravely and low nutrient soil with unpalatable vegetation on crests and deep clayey soil with moderate nutrients and palatable vegetation on bottomlands (Witkowski & O'Connor, 1996; Venter *et al.*, 2003). Palatable vegetation of the drainage lines is characterised by adaptations to handle consistent herbivore pressure with features such as grazing-tolerance, fast regeneration times and herbivore defence strategies (Milchunas *et al.*, 1988; Prins & van der Jeugd, 1992; Mushove *et al.*, 1995).

Sodic sites are areas of the landscape with high nutrient value and high soil salts occurring near to drainage lines (Khomo & Rogers, 2005). Herbivores selectively use these areas for wallowing, nutrition and relief from predation (Khomo & Rogers, 2005; Jacobs *et al.*, 2007). Sodic sites naturally grow through soil movement but installation of waterpoints can exacerbate erosion and growth of patches (Khomo & Rogers, 2005). Erosion is common in central piosphere zones (James *et al.*, 1999) where trampling reduces microtopography (Nash *et al.*, 2003) and protective biotic crusts (Belnap & Gillette, 1998). Installation of waterpoints on sodic sites should therefore be avoided. Erosion can lead to loss of vegetation cover, including trees, as the system can no longer absorb climatic variations when soil health is compromised (MacGregor & O'Connor, 2002).

Accuracy of waterpoint locations in this study is neither consistent nor high in some cases. In the calculations of artificial waterpoint statistics and natural water availability scores this is unlikely to have caused a problem. However, the catenal position analysis requires more accurate data. The higher proportion of waterpoints within the drainage line for more intensely managed properties may simply be a reflection of better quality data from these properties. Use of a larger buffer size would lead to a higher proportion of waterpoints in drainage lines but because of the presence of sodic sites in close proximity to drainage lines, this analysis needs to be as accurate as possible.

CONCLUSION

More intensely managed properties have less natural water densities and high water supplementation in naturally drier landscapes. However, this landscape supplementation is constrained by available area and is therefore beyond the control of management. More intensely managed properties had a higher level of natural catenal positioning of waterpoints. This may be a reflection of better quality data capture but may also reflect a greater degree of sensitivity when planning waterpoint locations. The environmental impacts on intensely managed private reserves should be interpreted with regards to alternate factors contributing to management. The density of artificial waterpoints and their landscape and catenal positioning within properties suggests that there is a high risk of degradation associated with water supplementation in the southern African savannas.

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Chapter 3

A review of waterpoint management in African savannas: the need for acknowledgement of spatial heterogeneity

Helen Farmer

ABSTRACT

Conservation is increasingly focused on broad scales with incorporation of spatial heterogeneity and ecosystem resilience. Establishment of transboundary conservation areas involves properties with differing management approaches working together for regional conservation. Water provision is a major management intervention in the southern Africa savanna. The first records of water management in the area are from 1933. Research through the late 1900s highlighted long term detrimental effects of artificial waterpoints on vegetation and soils and led to revision of the approach to water management. Current levels of water supplementation are strongly linked to management objectives. Consideration of ecosystem resilience is becoming a major part of conservation management. Herbivore impacts around waterpoints have effects on local (around the waterpoint) and broad (across the property) scale resilience. Incorporating spatial heterogeneity into water management is challenging as the current basis for management, piospheres, assumes homogeneity. African savannas are highly heterogeneous and therefore the piosphere model is potentially inappropriate. More factors than are currently considered would need to be included when understanding the impact of water supplementation using a heterogeneity approach.

KEYWORDS

Conservation; degradation; piospheres; resilience; transfrontier

INTRODUCTION

Conservation today is increasingly focused on broad scales, emphasising ecosystem processes and complexity, with management to retain critical natural variation and adaptive capacity in ecosystems (Baker, 1992; Walker *et al.*, 2002; Rogers, 2003; Boyd *et al.*, 2008). The expansion in objectives from species conservation to ecosystem conservation is accompanied by an increasing requirement for the incorporation of ecological theory into conservation management (Baker, 1992; Holling & Meffe, 1996; Rogers, 2003). Spatial heterogeneity and ecosystem resilience are perceived to have a positive effect on the maintenance of biodiversity (Fuhlendorf & Engle, 2001) and in turn, biodiversity can have a strong, positive influence on ecosystem function (Elmqvist *et al.*, 2003; Tylianakis *et al.*, 2008).

The Great Limpopo Transfrontier Park (GLTP), established in 2002, is situated in the savanna region of north-eastern South Africa, south-eastern Zimbabwe and south-western Mozambique (SANP, 2005). The GLTP is made up of Gonarezhou National Park, Limpopo National Park and Kruger National Park and forms the core area of the Great Limpopo Transfrontier Conservation Area (GLTFCA) which encompasses management areas ranging from communal lands to national conservation areas and private nature reserves. The scale at which biodiversity is managed varies greatly between the GLTFCA properties resulting in different management challenges, even within the same objective (Peel *et al.*, 1999). Management approaches vary because in smaller conservation areas, actions need to be intensified because the system becomes more limited and therefore less natural (Baker, 1992; Peel *et al.*, 1999; Illius & O'Connor, 1999). Previously, management decisions and actions were limited to single properties which therefore acted as closed systems, in isolation from surrounding areas. This contributed to sharpening of boundaries between properties (Forman & Godron, 1981). The establishment of transboundary and transfrontier conservation areas and subsequent dropping of fences enables planning for landscape and regional conservation management. Regional conservation management can soften artificial boundaries across the landscape.

The main areas of management variation between GLTFCA properties are fire (van Wilgen *et al.*, 2004; Higgins *et al.*, 2007), water provision (Owen-Smith, 1996; Gaylard *et al.*, 2003), and elephant density (Cumming, 1982; Owen-Smith &

Danckwerts, 1997). The level of water supplementation varies from none in Limpopo National Park, Mozambique (Grossman & Holden, 2003), to high levels in the South African private reserves (Chapter 2). Current waterpoint management is based on piosphere theory, which sees the landscape as a homogeneous template over which a standardised utilisation pattern focused on waterpoints appears (Lange, 1969; Thrash & Derry, 1999; Smit *et al.*, 2007). On properties where waterpoints are not fenced off from each other, the piosphere of one waterpoint can link to the piosphere of the next waterpoint, depending on the distance between them (Owen-Smith, 1996; Thrash, 2000). Impact of artificial waterpoints is therefore understood in terms of (1) the degradation caused around a single waterpoint, and (2) the distance between neighbouring waterpoints, i.e. the probability of herbivore impact extending between the two.

In recent years there has been a shift in savanna conservation management towards basis on a heterogeneity paradigm (Holling & Meffe, 1996; Rogers, 2003; Sinclair *et al.*, 2007). An understanding of the role of heterogeneity in landscape function suggests an alternative approach to understanding the impact of artificial water provision may be required. This paper aims to (1) review the history of water management in a southern African savanna, highlighting how the theory behind management has developed over time, and (2) discuss the issues surrounding modernisation of waterpoint management: incorporation of resilience and spatial heterogeneity. As water supplementation is associated with a high degradation risk (Chapter 2), it is important to understand how the approach to management has changed over time and how current ecological theories can contribute to understanding and managing the impacts of artificial water supplementation.

WATER MANAGEMENT

One of the management interventions that is said to be able to alter spatial heterogeneity in southern African savannas is artificial water provision (Owen-Smith, 1996; Gaylard *et al.*, 2003). Artificial water provision alters herbivore distributions, and therefore impacts on vegetation and soils (Owen-Smith, 1996; Chamaillé-Jammes *et al.*, 2007; Smit *et al.*, 2007). Formal conservation management in the southern African savanna dates back to 1902 with the establishment of the Sabi Game Reserve (Mabunda *et al.*, 2003). Water management began in Kruger National Park in 1933 in order to alleviate herbivore water stress and to spread game across the park (Mabunda *et al.*, 2003). The western veterinary fence of the Kruger National Park was completed by the early 1960s, and by 1980 Kruger National Park was fully fenced. This led to the requirement for more waterpoints as there was greater pressure on existing water sources and migration routes had been disrupted (Pienaar, 1985). These justifications are tightly linked to agricultural approaches (Aucamp *et al.*, 1992), the only management theory available at the time.

In 1983, severe drought led to mass mortality within Kruger National Park and adjacent private reserves (Walker *et al.*, 1987). The mortality was linked to lack of forage resources due to widespread herbivore utilisation, enabled because of water provision (Walker *et al.*, 1987). Further research on vegetation and soils in the 1990s highlighted the long-term detrimental effects of artificial waterpoints (Thrash *et al.*, 1991a,b; Thrash, 1997, 1998a). Extensive and long-term provision of artificial water can lead to degradation of soils and vegetation, compromising production and biodiversity objectives (Thrash & Derry, 1999; Parker & Witkowski, 1999; Illius & O'Connor, 1999; James *et al.*, 1999). Stabilising water availability reduces variability in access to forage resources and therefore reduces the likely natural cause of fluctuations in herbivore abundance (Cronje *et al.*, 2005; Chamaillé-Jammes *et al.*, 2007).

Greater understanding of the detrimental effects of water provision led to the revision of water management. Research revealed that the presence of artificial waterpoints could be detrimental to rare antelope populations (Grant & van der Walt, 2000). Further research done around waterpoints assessed impact by herbivores (see Thrash & Derry 1999) and a series of guidelines were developed for management decisions (Owen-Smith, 1996). The guidelines suggested inter-waterpoint distances of over 10km in order to maintain dry season unutilised vegetation between waterpoints (Owen-Smith, 1996; Redfern *et al.*, 2003; Smit *et al.*, 2007). This led to the decision to close many waterpoints in Kruger National Park (Gaylard *et al.*, 2003; Venter *et al.* 2008). The period between 1996 and 2007 saw the closure of boreholes and destruction of dams in Kruger National Park (Venter *et al.* 2008).

Currently, permanent water is managed to increase water availability in the dry season, to stabilise water availability and increase herbivore population size (Aucamp *et al.*, 1992; Grossman *et al.*, 1999; Chamaillé-Jammes *et al.*, 2007), or to increase habitat heterogeneity (Owen-Smith, 1996; Thrash, 1998b; Grant & van der Walt, 2000; Venter *et al.*, 2008). Management objectives determine the level of water supplementation (Owen-Smith, 1996; Underwood, 1998). Small private properties tend to have a higher profit requirement and therefore have higher levels of water provision leading to maximum utilisation of forage resources (Aucamp *et al.*, 1992; Grossman *et al.*, 1999). As properties get larger there is greater scope for more natural and broader scale management regimes (Baker, 1992; Peel *et al.*, 1999). Properties emphasising biodiversity conservation have a water provision level which is considered to increase habitat type heterogeneity and therefore biodiversity (Thrash, 1998b; Smit *et al.*, 2007; Tylianakis *et al.*, 2008). Management emphasising wilderness conservation has no provision of artificial water (Grossman & Holden, 2003).

PIOSPHERES

In order to manage waterpoints, it is necessary to understand the impact that they have on the surrounding ecosystem. Herbivore impact around a waterpoint is currently understood in terms of piosphere formation through grazing, browsing and trampling. The piosphere concept originated in Australia in domestic livestock systems (Lange, 1969) and has been applied in Kruger National Park and surrounding areas since the early 1990s (Thrash *et al.*, 1991b; Owen-Smith, 1996; Thrash, 1998a). The piosphere theory states that herbivore impact is highest close to the water and decreases with increasing distance from water because of the forage-water trade-off of water dependent herbivores (Lange, 1969; Graetz & Ludwig, 1978; Adler & Hall, 2005). The impact pattern is described using concentric circles with the waterpoint at the centre and each ring having a different impact level (Figure 1) (Graetz & Ludwig, 1978; Adler & Hall, 2005). Over time, the zone of highest utilisation moves away from the waterpoint and the piosphere grows (Adler & Hall, 2005).

Excessive trampling around a waterpoint leads to soil compaction and an associated decline in water infiltration (Thrash, 1997; Snyman & du Preez, 2005; Castellano & Valone, 2007) and seedling establishment (Bassett *et al.*, 2005). Clay crusts form, further reducing infiltration and subsequent water availability to plants (Mills & Fey, 2004). Biological crusts are quickly broken by mechanical disturbance such as herbivore trampling and this increases susceptibility to wind erosion (Belnap & Gillette, 1998; Eldridge & Leys, 2003). Trampling also has a negative effect on soil nutrients (Smet & Ward, 2006). These effects culminate in a reduction in biomass production (Ludwig *et al.*, 2005) close to the waterpoint.

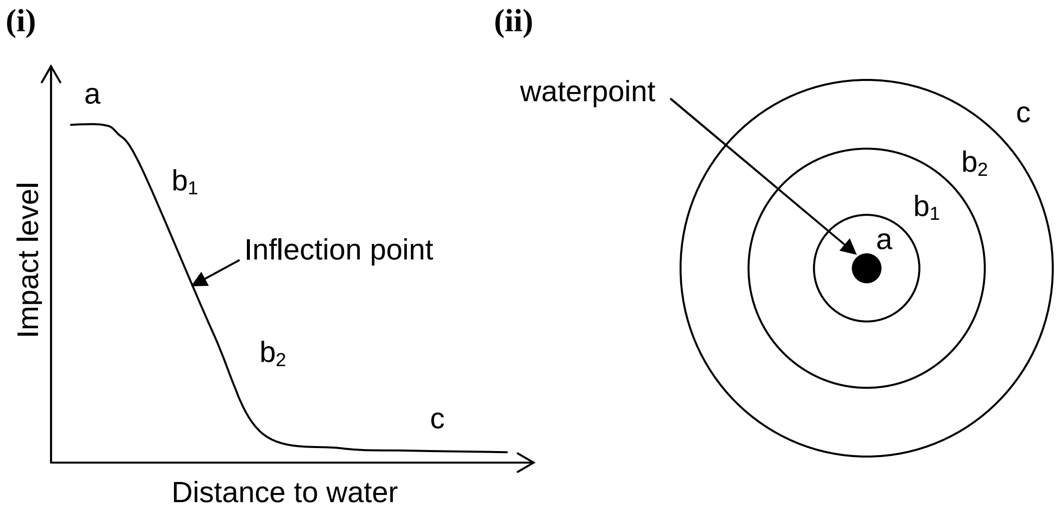

Figure 1: (i) The logistic curve of Graetz & Ludwig (1978) used to describe the zones of a piosphere and, **(ii)** the piosphere as concentric rings of different impact levels with rings corresponding to the logistic curve. Zones are labelled as (a) sacrifice zone, poor condition, (b_1) changing impact, fair condition, (b_2) changing impact, good condition and (c) very little impact, excellent condition.

Heavy grazing reduces grass plant density (Bestelmeyer *et al.*, 2006) and changes the species composition of the vegetation with a reduction in palatable and an increase in unpalatable plants (O'Connor, 1994; Parsons *et al.*, 1997). There is also a shift from perennial to annual vegetation (Nangula & Oba, 2004; Dorrough *et al.*, 2004). Grass recruitment is affected by seed availability, soil moisture and gaps in vegetation (O'Connor, 1994) and seedlings have a greater sensitivity to grazing impacts than adult plants (Hunt, 2001). Savanna grasses have low seed production and smaller seed banks than moister vegetation types (Skoglund, 1992; O'Connor, 1994) and vegetative reproduction is important (Scholes, 1997). Grazing reduces both vegetative growth and seed production (O'Connor, 1994).

Heavy browsing has a variety of effects on woody vegetation. Stability of adult population sizes is important for population growth in perennial herbs and shrubs (Hunt, 2001; Bruna, 2003). Adult plants are reduced in size and growth (van Langevelde *et al.*, 2003; Levick & Rogers, 2008; Fornara & du Toit, 2008) which can lead to increased losses of woody vegetation to fire (Mills & Fey, 2005). There is also a shift towards unpalatable woody species as reproduction by (Fornara & du Toit, 2008) and seedling survival of (Shaw *et al.*, 2002) palatable trees are reduced

RESILIENCE

Resilience, the ability of the system to absorb disturbance without changing into an alternative stable state (Holling, 1973; Holling & Meffe, 1996; Gunderson, 2000), and water management are linked primarily at two scales. At a relatively fine scale, herbivore impact around a waterpoint can alter vegetation and soil states and their resilience. At a broader scale, resilience of vegetation and soils around each waterpoint contributes to broader scale property resilience (Carpenter *et al.*, 2001; Cumming *et al.*, 2005). In the last few decades there has been an increasing awareness of the dynamism of ecological systems and the importance of ecosystem resilience for long-term functioning and sustainability of ecosystems (Gunderson, 2000). Currently, savanna conservation is moving towards incorporation of resilience (du Toit *et al.*, 2003).

Resilience of vegetation and soils surrounding a waterpoint will be affected by factors such as vegetation type, soil type and age of the waterpoint (Westoby *et al.*, 1989; Friedel, 1991; Suding *et al.*, 2004). If the implications of these factors on resilience can be determined then there is an opportunity for improved management of artificial water provision through placement of waterpoints. Thrash (2000) found that the piosphere effect is larger in areas with lower density water provision. In areas with low density water provision animals tend to use a single waterpoint. Piospheres are delimited only by herbivores ability to forage away from water (Lange, 1969) so they will continue to grow as herbivores need to move further to obtain sufficient forage (Redfern *et al.*, 2003; Adler & Hall, 2005).

High provision of permanent waterpoints creates a multiplicity of degradation nodes resulting in a potential decrease in resilience across the property (Figure 2). In areas with homogenised piospheres or old piospheres, the central zones of the piosphere become isolated and it is likely that in these areas recovery from the degraded state will be difficult. Regeneration capacity of this area is low due to lack of existing plants and the re-colonisation ability of surrounding vegetation is limited due to isolation (Owen-Smith, 1996; Eriksson, 1996; Suding *et al.*, 2004; Kolb & Diekmann, 2004).

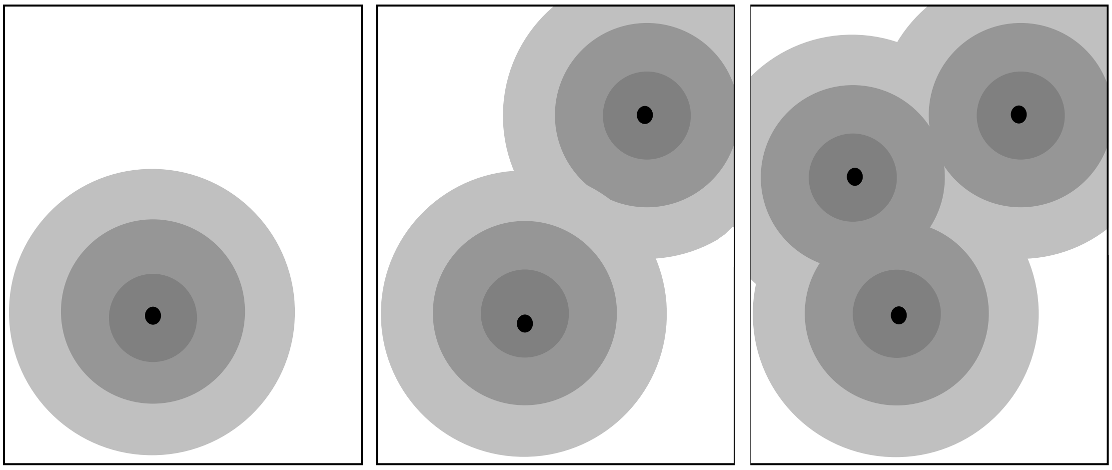

Figure 2: The relationship between resilience, heterogeneity and increasing water provision on a property. Black dots indicate waterpoints; grey bands indicate piosphere zones with darker shading indicating increased degradation. As waterpoint density increases, the more degraded piosphere zones merge and the distance to surrounding vegetation increases.

Previous work has shown that large differences in vegetation, steep gradients of change or large patch areas can lead to a slower return to the original vegetation state (Forman & Godron, 1981; Dunning *et al.*, 1992; Babaasa *et al.*, 2004; Huggett, 2005; Johst & Huth, 2005). Dispersal ability of plants has a large impact on re-colonisation (Freestone & Inouye, 2006). In peripheral zones of the piosphere, resilience of the original vegetation is likely to be high because ecological memory, source areas and mobility links are present (Nystrom & Folke, 2001). Ecological memory is represented by persisting plants, source areas are represented by unutilised vegetation and mobility links are present because the distances are likely to be within those of plant dispersal (Figure 3).

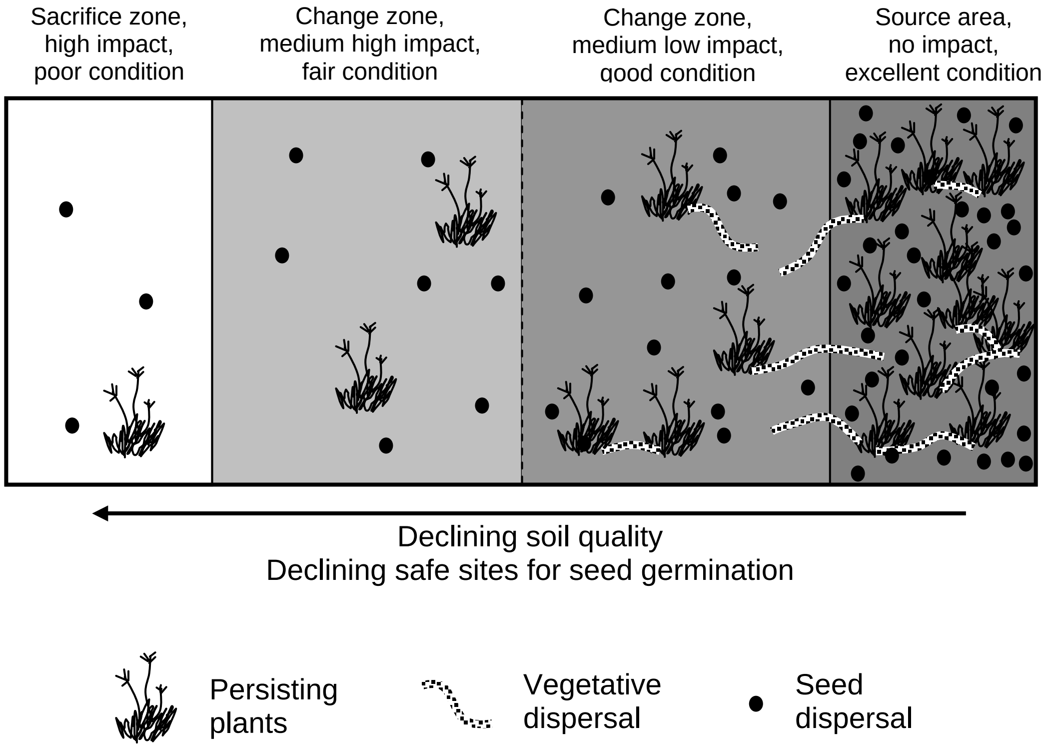

Figure 3: Theoretical ecological memory (persisting plants), source area and mobility links (vegetative and seed dispersal) within a grazing gradient.

SPATIAL HETEROGENEITY

Considering the importance of vegetation pattern in resilience of an area, understanding the spatial heterogeneity of a property is important for predicting its ecological resilience (Eriksson, 1996; Nystrom & Folke, 2001; Suding *et al.*, 2004). Spatial heterogeneity refers to the degree of difference between parts of a landscape including the amount and spatially explicit configuration of the environmental resources and constraints (Pickett *et al.*, 2003). This can be split into *functional heterogeneity*, which ecological entities perceive, relate to and respond to, and *measured heterogeneity*, which is a product of the observer's arbitrary perspective (Li & Reynolds, 1995). In terms of this study, spatial heterogeneity is multi-scaled and refers to intrinsic differences between areas of the biophysical template.

African savannas are highly spatially heterogeneous (Skarpe, 1992; Pickett *et al.*, 2003; Sankaran *et al.*, 2004). Variation arises at broad scales from geology and climate and at finer scales from fire and herbivory (Scholes, 1990; Skarpe, 1992; Wiegand *et al.*, 2006). Soil variation provides a template for vegetation variation

(Ben-Shahar, 1991). Vegetation is then further modified by invertebrate (Mobaek *et al.*, 2005) and vertebrate (Verweij *et al.*, 2006; Anderson *et al.*, 2007; Waldram *et al.*, 2008) herbivores, rainfall (Wiegand *et al.*, 2006) and fire (Parr & Anderson, 2006; Govender *et al.*, 2006; Higgins *et al.*, 2007). The interaction of multiple drivers of heterogeneity gives rise to a savanna ecosystem mosaic at various scales (Scoones, 1995; Augustine, 2003).

Herbivores are able to alter heterogeneity within the topographical and vegetation type mosaic. Differences between vegetation patches can be reduced or enhanced depending on the intensity and scale of grazing and the scale of heterogeneity (Abrams, 2000; Adler *et al.*, 2001; Fuhlendorf & Engle, 2001). By simple repeated preferential use of one vegetation patch over another, vegetation boundaries are promoted (Farnsworth & Anderson, 2001). Enhancement of heterogeneity by one species, for example rhino (Waldram *et al.*, 2008) or termites (Mobaek *et al.*, 2005), can have subsequent effects on utilisation of an area by other species. Variation in herbivore size results in a wide variation of perceived templates and decision making scales (Arditi & Dacorogna, 1988; Law & Dickman, 1998; Bowyer & Kie, 2006).

The ecosystem mosaic is a pattern in space and time (Montana, 1992; Traill, 2004; Ludwig *et al.*, 2005; Shrader *et al.*, 2006; Chamaillé-Jammes *et al.*, 2007) and forms the template upon which herbivore foraging decisions are made (Arditi & Dacorogna, 1988; Bailey *et al.*, 1996; Farnsworth & Anderson, 2001; Boone *et al.*, 2006). Heterogeneity in resource availability and quality leads to forage area preferences where intake rate is maximised (Senft *et al.*, 1987; Bailey *et al.*, 1996; Dolman & Sutherland, 1997; Skarpe *et al.*, 2000; Augustine, 2003). Forage area preferences will also change temporally, for example with creation of newly burnt areas (Mills & Fey, 2005; Archibald *et al.*, 2005; Klop *et al.*, 2007) and rainfall patterns (Boone *et al.*, 2006; Chamaillé-Jammes *et al.*, 2007). Herbivores have been shown to have a spatial memory of foraging patches and will return to the same area repeatedly to feed (Coughenour, 1991; Cid & Brizuela, 1998; Skarpe *et al.*, 2000; Verweij *et al.*, 2006).

When managing waterpoints across a property, the scale of interest is related to the distance between locations where water is available as the area surrounding a waterpoint becomes a management unit within a water limited landscape (Lange, 1969). Initial studies on piospheres focused on single herbivore species systems in homogeneous landscapes with palatable vegetation (Lange, 1969; Andrew, 1988; Stafford Smith, 1990). In these areas, herbivore movement and utilisation is affected solely by the trade-off between water and forage requirements (Graetz & Ludwig, 1978). Heterogeneity with functional relevance to medium to large water-dependent herbivores within the inter-waterpoint distance is of importance when understanding herbivore impact because it can affect movement and utilisation patterns, thereby disrupting or preventing the formation of the piosphere impact pattern (Nash *et al.*, 1999; Nangula & Oba, 2004).

LOOKING FORWARD

Current management of waterpoints and water availability in southern African savanna conservation areas is based on the piosphere approach (Owen-Smith 1996; Thrash & Derry 1999; Thrash 2000). Water management and the understanding of impact across a property is based simply on the number of waterpoints and distance between them (Redfern *et al.* 2003, Smit *et al.* 2007). Circular impact zone patterns are used to approximate herbivore impact on a homogeneous landscape (Gaylard *et al*. 2003). However, there has been a shift in savanna conservation to incorporate spatial heterogeneity in management (Rogers, 2003). The high level of heterogeneity of southern African savannas affects herbivore movement patterns and utilisation (Bailey *et al.*, 1996; Skarpe *et al.*, 2000; Mills & Fey, 2004). There is also a shift towards management based on dynamism and resilience (du Toit *et al.*, 2003). Water management in savanna conservation areas needs to be altered to come into line with relevant current ecological theories.

Piosphere studies in savanna conservation areas have focused on homogeneous landscape units (Thrash et al. 1991a; Thrash 1997, 1998; Brits et al. 2002) and found that the piosphere approach works. However, heterogeneity has been found to disrupt piosphere patterns (Lange 1969; Cridland & Stafford Smith 1993; Nash et al. 1999; Nangula & Oba 2004; Washington-Allen et al. 2004) and at management scales in the savanna, the assumption of homogeneity is invalid. The basic landscape template over which herbivore impact is superimposed interacts with herbivores (Bailey *et al.*, 1996; de Knegt *et al.*, 2008); the water/forage trade-off is not the only factor influencing animal movement. In moving towards a new approach to understanding artificial waterpoint impact, factors such as landscape vegetation type heterogeneity (Nangula & Oba, 2004), herbivore foraging area preferences (Bailey *et al.*, 1996), artificial waterpoint placement (Ayeni, 1977; Owen-Smith, 1996), and waterpoint type (Washington-Allen *et al.*, 2004) need to be considered in addition to artificial waterpoint density.

Vegetation types across a natural landscape have differing levels of resilience to disturbance by herbivores. Landscape areas which have evolved with a higher pressure from herbivory will be more resilient to permanent herbivore impact. Moister areas of the landscape often have different vegetation and higher levels of defence against herbivory (Milton, 1991; Stalmans *et al.*, 2004). The availability of seasonal and permanent water should be assessed separately as they have differing impacts (Parker & Witkowski, 1999; Chamaillé-Jammes *et al.*, 2007).

Water management has a long and varied history within the GLTFCA, from supplementation in the early 1900s to reduction in the late 1900s and early 2000s. Levels of water provision now tend to reflect management objectives and vary between properties. Supplementation of permanent waterpoints for herbivores is associated with a high risk of degradation (Chapter 2). This in turn affects the resilience of the property. In order to optimally manage properties for regional conservation, it is important that the link between water supplementation and property resilience is understood. Using a piosphere approach to understanding degradation patterns and linking them to property resilience is likely to be too simplistic. The piosphere approach needs to be tested for its general applicability, and if it is not generally applicable, a new approach to understanding the links between artificial water provision and degradation across properties needs to be developed.

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Chapter 4

A test of piosphere theory: can it explain herbivore impact patterns in a heterogeneous savanna?

Helen Farmer

ABSTRACT

The level of water supplementation in southern African savanna conservation areas depends on property objectives. Current water management is based on the piosphere model which assumes homogeneity but southern African savannas are highly heterogeneous. This study tests the general applicability of the piosphere model over a wide range of waterpoints across different properties. Data were collected on soil functionality and herbaceous and woody vegetation along transects from natural, closed artificial and open artificial waterpoints in Limpopo National Park (Mozambique), Kruger National Park (South Africa) and three privately owned South African reserves. Variables for testing were selected based on previous piosphere studies. A total of 778 analyses were performed with 106 (14%) showing a significant relationship with distance to water. Linear and logistic regressions on 19 waterpoints (three excluded due to small sample sizes) revealed that soil infiltration, nutrient cycling and stability indices, bare ground cover, herbaceous vegetation basal cover and density, woody vegetation density, species richness, canopy cover, proportion in height classes, density in height classes and height diversity did not have consistent relationships with distance to water within or between waterpoints. RDA and CCA ordinations on herbaceous and woody vegetation data from 22 waterpoints revealed only one significant relationship between herbaceous vegetation and distance to water. These results led to the conclusion that the piosphere model is not generally applicable in heterogeneous southern African savannas. Results from small scale studies cannot be generalised between waterpoints or across properties so the piosphere model does not form a good basis for management. The lack of general applicability is most likely due to high levels of heterogeneity in soils and vegetation within inter-waterpoint distances. Although utilisation gradients were detected for some variables (e.g. herbaceous tuft size and density), the low level of consistency means a new approach to understanding and managing the effects of water supplementation needs to be developed.

KEY WORDS

Conservation; grazing gradient; herbaceous; Landscape Function Analysis (LFA); soil; vegetation; waterpoint; woody

INTRODUCTION

Creation of transboundary conservation areas involves removal of fences to join properties with different management approaches into biodiversity conservation areas that enable landscape conservation and promote natural processes (Peace Parks, 2005). In the southern African savanna, the Great Limpopo Transfrontier Conservation Area (GLTFCA) was created to, among other objectives, harmonise environmental management procedures with the removal of artificial barriers and re-establishment of historical animal migration routes (Peace Parks, 2005). When properties were fenced off through the mid to late 1900s, natural migrations driven by water availability were prevented and waterpoints were installed for herbivores (Mabunda *et al.*, 2003). Subsequent research highlighted detrimental effects of water supplementation (Walker *et al.*, 1987; Thrash & Derry, 1999). Current levels of water provision are linked to management objectives (Owen-Smith, 1996; Thrash & Derry, 1999) and management intensity (Chapter 2).

Artificial waterpoints affect vegetation through alteration of herbivore movement and utilisation patterns (Owen-Smith, 1996; Smit *et al.*, 2007). Areas that were previously unused are opened up and the utilisation period is prolonged (Chamaillé-Jammes *et al.*, 2007). A variety of effects of waterpoint installation have been found on ecosystems throughout the world (e.g. Thrash & Derry, 1999; James *et al.*, 1999; Fernandez-Gimenez & Allen-Diaz, 2001). Generally, the piosphere model is used to understand the pattern of herbivore impact around waterpoints (Lange, 1969; Graetz & Ludwig, 1978; Chapter 1,3). This model is based on the trade-off between water and forage requirements of herbivores and states that degradation declines with increasing distance from water (Lange, 1969; Graetz & Ludwig, 1978). The relationship between degradation and distance from water can be modelled with a logistic equation under constant environmental variables (Graetz & Ludwig, 1978). The piosphere approach has been used in southern African savannas since the early 1990s (Thrash *et al.*, 1991a, b).

Savanna piosphere studies have assessed impacts on soil, herbaceous vegetation and woody vegetation. Soil infiltration rate increased with distance from water, a relationship not caused by changes in soil particle size which remained relatively constant (Thrash, 1997). Basal cover of herbaceous vegetation increases with greater distance from water (Thrash *et al.*, 1991a) as well as standing crop, forage production and fuel production (Thrash, 1998a), with no differences between artificial and natural waterpoints (Thrash, 1998b). Species composition and functional characteristics of herbaceous vegetation reveal gradients of herbivore utilisation from waterpoints (Parker & Witkowski, 1999). Stocking rate affects the distance of impact on herbaceous vegetation (Thrash, 2000). Woody vegetation survival is higher at greater distances from water though patterns vary between species (Thrash *et al.*, 1991b). Tree and shrub densities increased with distance from water (Brits *et al.*, 2002).

Although the piosphere approach has been applied frequently in southern African savannas, it has not been as broadly tested as in the Australian rangelands (Graetz & Ludwig, 1978; Bastin *et al.*, 1993; Pickup, 1994; Hunt, 2001; Heshmatti *et al.*, 2002; Landsberg *et al.*, 2003). In general, studies cover small numbers of waterpoints (e.g. Brits *et al.*, 2002) with only one study covering more than ten waterpoints (Thrash, 2000). All the studies are based on small sections of the landscape with many waterpoints being sampled with single transects of less than 300m in length (e.g. Thrash, 1997, 1998b). These landscape sections are chosen for homogeneity, often of soil type (e.g. Thrash, 1997, 1998a,b, 2000). The results of these studies of homogeneous areas are generalised between waterpoints and projected across landscapes for management. At management scales, the southern African savanna is highly heterogeneous (Skarpe, 1992; Pickett *et al.*, 2003; Sankaran *et al.*, 2004) so the applicability of results from studies of homogeneous areas is questionable.

Supplementation of water is associated with a high degradation risk (Chapter 2) and patterns of degradation are important for estimating property resilience (Chapter 3). This study was therefore initiated to test the general applicability of the piosphere model across southern African savanna waterpoints and therefore to determine whether results from short transect, low waterpoint number studies can be scaled up across properties. Soil functionality and herbaceous and woody vegetation variables were investigated against distance to water gradients for a wide variety of waterpoints (artificial and natural) over a range of properties with differing conservation management objectives and intensities. Statistical methods used in previous piosphere studies (linear and logistic regression and ordination) were used to test for a relationship between 23 variables (e.g. soil nutrient cycling, herbaceous vegetation density and woody vegetation composition) and distance to water. If the piosphere model is generally applicable, a high proportion of significant statistical tests should be found for variables and waterpoints.

METHOD

Study area

The GLTFCA is located in Zimbabwe, Mozambique and South Africa. Over 50% of the GLTFCA consists of national parks and privately owned conservation land (Peace Parks, 2005). The South African private reserves are important for the east-west representation of the GLTFCA as the national parks are oriented north-south. This study targeted three privately owned properties and two national parks with different management approaches (Figure 1, Table 1). Management intensity is highest in the private reserves and lowest in Limpopo National Park (see Chapter 2 for more detail on management intensity and objectives). Due to the limited time available for fieldwork, the three most logistically easy private reserves were selected for fieldwork. These reserves also represent the full spectrum of management intensities from Chapter 2.

The study area has mild dry winters and hot wet summers with rainfall generally decreasing on a south to north and east to west gradient (Venter et al., 2003; Stalmans et al., 2004). There is a slight temperature trend from hotter in the north to cooler in the south (Venter et al., 2003; Stalmans et al., 2004). The geology is dominated by granite in the west, basalt in the centre and calcaric sedimentary in the east (Venter *et al.*, 2003; Stalmans *et al.*, 2004). A thin sedimentary strip separates the granite from the basalt and a thin rhyolite strip separates the basalt from the calcaric sedimentary (Venter *et al.*, 2003; Stalmans *et al.*, 2004). Basalts are generally flat with high nutrients and high water holding capacity (Redfern et al., 2003; Venter et al., 2003; Smit et al., 2007). Granites are characterised by strong catenas with sandy gravely crests and clay bottomlands (Venter et al., 2003). The granite crests have low nutrients and do not hold water well whereas the bottomlands have a moderate level of nutrients and high water holding capacity (Venter et al., 2003). The calcaric sedimentary areas are overlain by sandy soils and form deep nutrient rich clays in drainage lines (Stalmans et al., 2004).

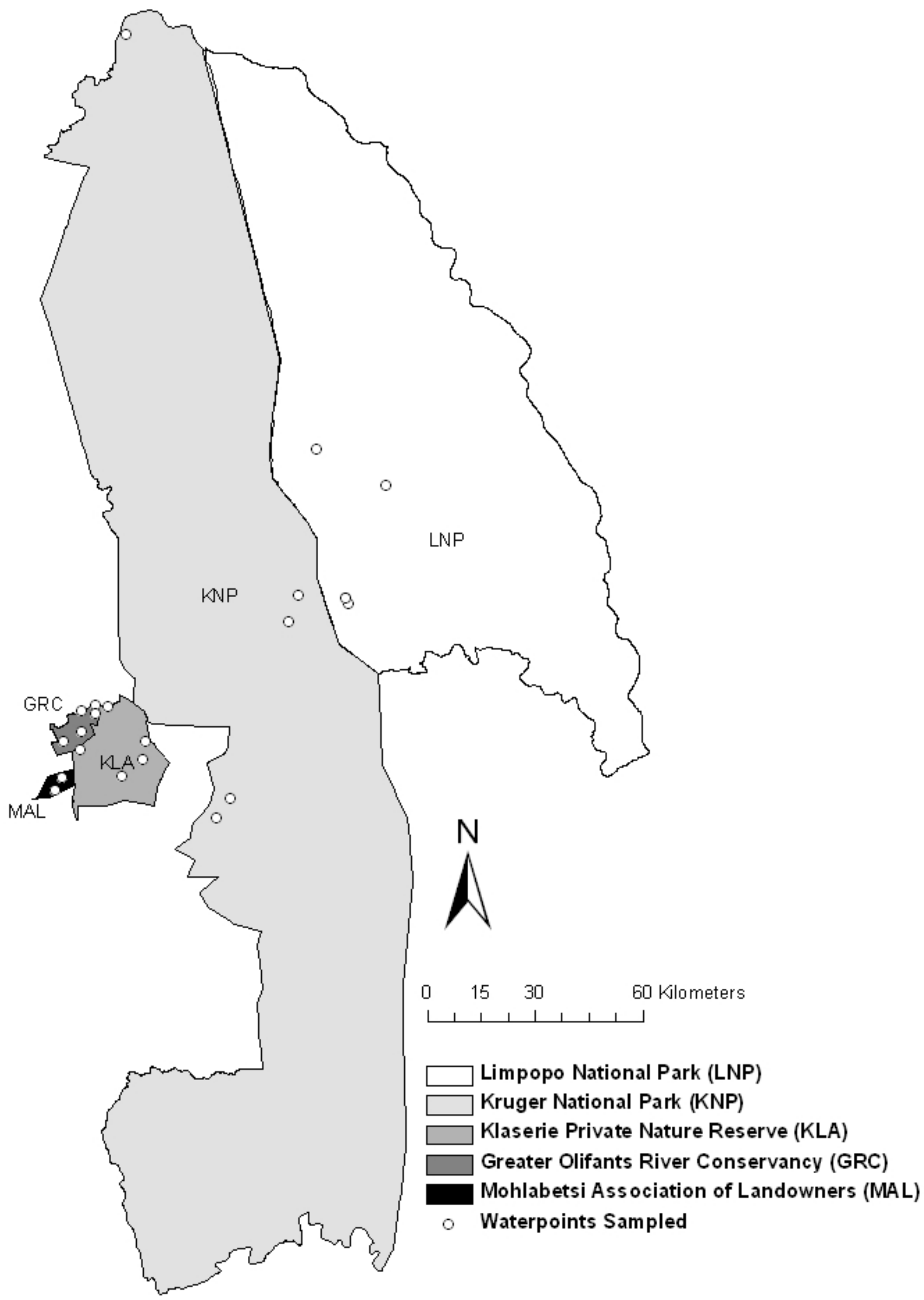

Figure 1: Map of the study area with darker shading indicating more intensive management and points indicating locations of waterpoints sampled.

Table 1: Artificial waterpoint density and management description of properties from the Great Limpopo Transfrontier Park included in the study.

Name	Abbreviation	Size (km^2)	Management intensity	Management emphasis	Artificial waterpoint density ($points/km^2$)
Private Reserves					
Mohlabetsi Association of Landowners	MAL	48	Very high	Tourism	0.357
Greater Olifants River Conservancy	GRC	105	High	Biodiversity/ Tourism	0.143
Klaserie Private Nature Reserve	KLA	578	Medium High	Biodiversity/ Tourism	0.118
National Parks					
Kruger National Park	KNP	18956	Medium	Biodiversity	0.008
Limpopo National Park	LNP	9845	Low	Wilderness	0

Data collection

Sampling time was divided between the three management areas and types of waterpoint (Table 2). A limited selection of the KNP landscape types are found in the private reserves and LNP (Gertenbach, 1983; Stalmans *et al.*, 2004; Peel *et al.*, 2007). Waterpoints were first selected in the private reserves and LNP and then waterpoints from KNP were selected in order to occur in comparable landscape types (Table 3). Waterpoints were selected without input from reserve management to avoid personal bias towards unusual or interesting waterpoints.

Table 2: Distribution of waterpoints sampled between properties and waterpoint types.

	Natural	Artificial		*Total*
		Open	Closed	
Private Reserves	3	5	4	*12*
Kruger National Park	0	1	4	*5*
Limpopo National Park	5	None available to sample	None available to sample	*5*
Total	*8*	*6*	*8*	*22*

Table 3: Landscape type occurrence of waterpoints sampled. Some transects in Limpopo National Park crossed landscape type boundaries because of their length.

Landscape Type	6	22	26	30	31	35
Private Reserves	12					
Kruger National Park	2	2	1			
Limpopo National Park		1	2	1	2	1

Landscapes sampled were 6, 22, 26, 30, 31 and 35. Landscape 6 is *Combretum*/*Colophospermum mopane* Woodland of Timbavati which consists of slightly irregular granitic plains with Colophospermum mopane bush savanna, or irregular granitic hills with *C. mopane* tree savanna (Gertenbach, 1983). Landscape 22 is the *Combretum/Colophosperumum mopane* Rugged Veld which is relatively variable, consisting of irregular basaltic plains with *Acacia nigrescens* bush savanna or *C. mopane* bush savanna, or basaltic plains with *C. mopane* shrub savanna; or slightly undulating basaltic plains with *C. mopane* shrub savanna (Gertenbach, 1983).

Landscape 26 is *Colophospermum mopane* Shrubveld on Calcrete which consists of moderately undulating gabbroic plains with *C. mopane* shrub savanna or irregular calcrete plains with *C. mopane* shrub savanna (Gertenbach, 1983). Landscape 30 is the Pumbe Sandveld consisting of recent sand plains with *Terminalia sericea* bush savanna or *Baphia massaiensis* bush savanna (Gertenbach, 1983). Landscape 31 is the Lebombo North, slightly undulating basaltic plains or low rhyolitic mountains with *Combretum apiculatum* bush savanna, or low rhyolitic mountains with *C. mopane* bush savanna (Gertenbach, 1983). Landscape 35 is the *Salvadora angustifolia* Floodplains which consists of alluvial plains with *Salvadora australis* tree savanna (Gertenbach, 1983).

Following methods of previous piosphere studies, a single transect was sampled from each waterpoint (Lange, 1969; Thrash, 1998b; Brits *et al.*, 2000). If the piosphere theory is valid, the same pattern will be visible on any transect direction. Using a Thiessen polygon approach, transects were oriented so as to cover the greatest distance whilst remaining within the area influenced primarily by the waterpoint of interest (Parker & Witkowski, 1999; Ryan & Getz, 2005). Maximum transect length varied from 1km to 7km depending on inter-waterpoint distances (Table 4). In order to avoid bias caused by human perception of spatial patterns (Cramer & Hobbs, 2005) transects were not truncated at predetermined lengths. Sampling time was limited on each property in order that all properties could be sampled within one growing season (see below). The maximum possible time was spent sampling on each property but in some cases transects had to be truncated before the distances were complete (Table 4). Despite this, all transects exceeded the maximum length used in previous piosphere studies (Thrash, 1998a).

There is a trade-off between robustness of results and efficiency of sampling (Philippi *et al.*, 1998). Previous piosphere assessments have used interval sampling to increase sampling efficiency (Thrash, 2000; Heshmatti *et al.*, 2002; Riginos & Hoffman, 2003; Nangula & Oba, 2004). Varied interval length enables detection of the rapid change expected near the waterpoint whilst conserving sampling effort far from the waterpoint when changes are expected to be low (Lange, 1969; Graetz & Ludwig, 1978; Jeltsch *et al.*, 1997; Adler & Hall, 2005). Interval length for this study was determined using results from published studies and analysis of data from the

Agricultural Research Council (ARC) private reserve monitoring (M. Peel, J. Peel & A. Jacobs, unpublished data) and preliminary continuous sampling (Appendix 3). Interval lengths (the distance between two consecutive sampling sites along the waterpoint transect) varied from 50m close to the waterpoint to 1km far from the waterpoint (Figure 2). The location of each sampling site was recorded with a GPS (Appendix 4).

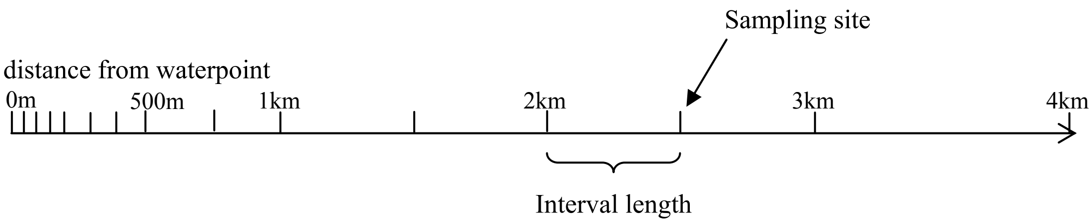

Figure 2: Scaled diagrammatic representation of interval length along a waterpoint transect.

Fieldwork was performed between November 2006 and June 2007, during the grass flowering season in order to ease species identifications. Water is relatively abundant at this time of year and herbivore impacts are widespread (Redfern *et al.*, 2003; Redfern *et al.*, 2005; Ryan & Getz, 2005) but long-term effects of piospheres are still visible (Adler & Hall, 2005). Inter-annual rainfall variation (Schulze, 1997) can affect the short-term response of piosphere vegetation so fieldwork was restricted to one growing season. At each sampling site, data on vegetation and soil were collected to test for the piosphere effect.

Table 4: Lengths of transects from waterpoints in the Great Limpopo Transfrontier Conservation Area

Property	Transect	Final Length (km)	Maximum Length (km)	Reason for early truncation	Missed Intervals	Reason for Missing
Limpopo National Park	Bona Kaya (BNK)	4	4	n/a	none	n/a
	Long Hippo Pool (LHP)	0.2	7	Maximum use of sampling time	none	n/a
	Machampane - camp side (MCC)	0.15	5	Close proximity to tourist camp	none	n/a
	Machampane Pool (MCP)	7	7	n/a	none	n/a
	Ngwenya (NGW)	1	6	Maximum use of sampling time	none	n/a
Kruger National Park	Bull Frog (BFG)	1.5	6	Maximum use of sampling time	50m	Drainage line
	Bvumanyun (BVU)	0.15	5	Maximum use of sampling time	none	n/a
	Engelhard Dam (EGH)	3	6	Maximum use of sampling time	none	n/a
	Eileen (ELN)	0.2	5	Maximum use of sampling time	50m	Drainage line
	Red Gorton (RGN)	5	5	n/a	none	n/a

Klaserie Private Nature Reserve	C20 (full name not given)	1	1	n/a	none	n/a
	C65 (full name not given)	1	1	n/a	none	n/a
	P26 (full name not given)	1.5	1.5	n/a	none	n/a
	D20 (full name not given)	1.5	1.5	n/a	none	n/a
	R01 (Olifants River)	1.5	1.5	n/a	none	n/a
Greater Olifants River Conservancy	Double Dam (DBD)	0.2	1.5	Maximum use of sampling time	none	n/a
	Ian's Pan (IAN)	1	1	n/a	100, 150, 200m	Railway line
	Olifants River near Rusermi (RVR)	1.5	1.5	n/a	none	n/a
	Olifants River near Seekoeigat (RVS)	1.5	1.5	n/a	none	n/a
	Seekoeigat 1 (SEE)	1	1	n/a	none	n/a
Mohlabetsi association of Landowners	Jejane (JEJ)	1	1	n/a	none	n/a
	Pusa Manzi (PUM)	1	1	n/a	none	n/a

For the herbaceous vegetation, data from the ARC private reserve monitoring program (M. Peel, J. Peel & A. Jacobs, unpublished data) were analysed to determine the most efficient sample size required to represent each interval (Appendix 5). It was concluded that a sample size of 80 points was required to represent the herbaceous vegetation at an interval. Herbaceous vegetation data were collected on a 4m by 15m grid of 1m by 1m cells oriented with its longest axis along the waterpoint transect (Figure 3). At each grid cell corner, the distance to nearest grass and basal diameter of tuft were measured and the species identified. Perennial grasses give a good indication of veld condition as they form longer-lived obstructions to resource flow than annual vegetation (Ludwig *et al.*, 2000) and degradation leads to their loss (Scholes, 1997; Thrash & Derry, 1999; Parker & Witkowski, 1999). Annual vegetation is often not recorded in vegetation assessments (Thrash, 1998a; Zambatis, 2005) but plays an important role in providing soil cover in the central part of the piosphere (Nangula & Oba, 2004). If there was no grass within 50cm of the point, the nearest forb was recorded. If there was no forb, the point was recorded as bare ground. Bare ground can be considered as dysfunctional parts of the ecosystem (Ludwig *et al.*, 2000).

A literature review of sampling areas for woody vegetation was used to determine an appropriate sampling area (Appendix 6). It was concluded that a 240m^2 belt plot would be appropriate. Woody vegetation data were collected in an 8m by 30m belt plot with the first half centred on the herbaceous grid (Figure 3). All woody plants within the belt were counted and species identified. Woody species composition can be used to indicate changes in vegetation type (Witkowski & O'Connor, 1996) and can be affected by distance to water (Tolsma *et al.*, 1987). Height class of each individual was recorded (<1m, 1-2m, 2-3m, 3-5m, >5m) as habitat structural variation is important for biodiversity conservation (Noss, 1990; Tews *et al.*, 2004; Ruiz-Jaen & Aide, 2005; Lunt *et al.*, 2007; Oliver *et al.*, 2007). Seedlings were not separated from the <1m category. Canopy cover of woody vegetation was recorded on the herbaceous sampling grid as presence/absence at each sampling point.

Ecological assessment of soil includes measurements of infiltration (Thrash, 1997), biotic crust (Belnap & Gillette, 1998), nitrogen fixation (Chen *et al.*, 2003), microbial activity (Belnap *et al.*, 2005) and surface crusting (Mills & Fey, 2004). Landscape

Function Analysis (LFA) is an integrated, indicator based approach which incorporates all of these variables within a spatial context (Tongway & Hindley, 2004). Eleven soil surface indicators, verified through laboratory and field experiments, are used to evaluate soil stability, infiltration and nutrient cycling (Tongway & Hindley, 1995; Tongway & Hindley, 2004). Six indicators from the LFA data collection are combined to give an infiltration index, three indicators for the nutrient cycling index and eight for the stability index (Tongway & Hindley, 2004). Functional landscapes (high stability, infiltration and nutrient cycling) are considered to be in good condition (Ludwig *et al.*, 2000).

LFA gradsects of 20m were assessed at each sampling site, down the centre-line of the herbaceous grid (Figure 3). If the centre-line of the herbaceous grid did not follow the resource flow gradient of the sampling area, the LFA gradsect was swivelled on the 10m point so that it was oriented in the direction of resource flow. Patches and interpatches were mapped to generate an index of landscape organisation (see Appendix 7 for more detailed LFA method information). Each patch and interpatch type was assessed using eleven indicators to characterise function. The indicators were combined into indices for stability, infiltration and nutrient cycling using the LFA data analysis spreadsheet.

Data analysis

In order to test the piosphere model, waterpoint transects were assessed individually. Data collected in this study were used to address variables from previous piosphere studies in the same area and outside the GLTFCA (Table 5).

Soil infiltration, soil nutrient cycling, soil stability, percentage of bare ground, grass tuft size and density, and woody plant density, canopy cover, species richness, proportion per height class, density per height class and height diversity were assessed in relation to distance from water using linear regression (y = m*x + c) and logistic regression (y = m*logx + c) in Statistica. Analyses were restricted to those waterpoints that had at least five intervals of data so three waterpoints (ELN, MCC and BVU) were excluded. LFA data were not collected on KLA and sampling sites with missing indicators had to be removed from the LFA analysis. This led to removal of LHP, DBD and IAN in addition to the KLA waterpoints from the LFA analyses.

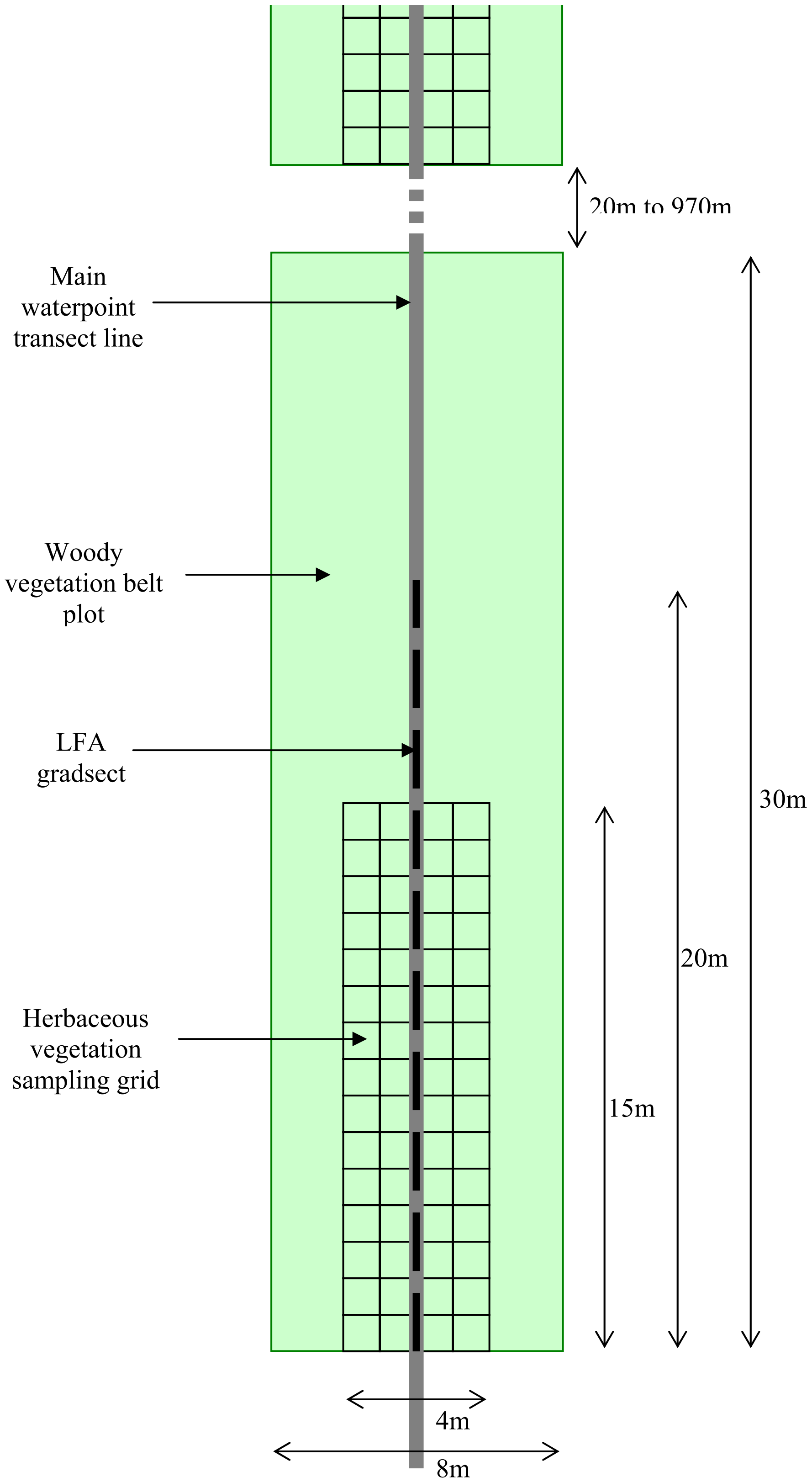

Figure 3: Sampling design showing layout of woody vegetation, herbaceous vegetation and soil sampling areas at each sampling site.

Table 5: Variables investigated in previous piosphere studies and the data used in this study for comparison.

Variable considered	Reference(s)	Current study
Previous studies from the study area		
Density of woody plants	(Thrash *et al.*, 1991b; Brits *et al.*, 2002)	Number of stems per belt plot
Density of woody plants in height classes	(Thrash *et al.*, 1991b)	Number/Proportion of stems in that height class per belt plot
Canopy cover of woody plants	(Thrash *et al.*, 1991b; Brits *et al.*, 2000)	Proportion of herbaceous grid points under canopies
Height of woody plants	(Brits *et al.*, 2000; Brits *et al.*, 2002)	Height classes of woody plants
Woody species richness	(Thrash *et al.*, 1991b)	Number of species per belt plot
Woody community composition	(Thrash *et al.*, 1991b)	Species frequencies
Density of grasses	(Thrash, 1998a; Thrash, 1998b; Thrash, 2000)	Average distance to tuft
Basal cover of grasses	(Thrash *et al.*, 1991a; Thrash, 1998a; Thrash, 1998b; Thrash, 2000)	Average tuft longest axis
Herbaceous community composition	(Thrash, 1998a; Thrash, 1998b; Parker & Witkowski, 1999; Thrash, 2000)	Species frequencies
Grass cover	(Parker & Witkowski, 1999)	Data of cover of bare ground used for inverse variable
Soil infiltration rate	(Thrash, 1997)	Indicators from LFA used to generate infiltration index
Previous studies from other areas		
Diversity of structural forms	(Todd, 2006)	Structural variation from height class data
Soil stability	(Hodgins & Rogers, 1997; Nash *et al.*, 2003)	Indicators from LFA used to generate stability index
Soil nutrient cycling	(Tolsma *et al.*, 1987; Turner, 1998)	Indicators from LFA used to generate nutrient cycling index

Woody plant density (plants/m^2) was expected to increase with increasing distance from water (Brits *et al.*, 2002). Two alternative effects of distance on height classes were expected, (1) all woody vegetation height classes would increase with increasing distance from water (Brits *et al.*, 2002), or (2) shrubs would decrease and trees would increase with increasing distance from water because of the effect of bush encroachment near waterpoints (Tolsma *et al.*, 1987; Thrash *et al.*, 1991b). Woody vegetation canopy cover was expected to increase with increasing distance from water (Thrash *et al.*, 1991b; Brits *et al.*, 2002). Species richness, the number of woody vegetation species per belt plot (240m^2), was expected to increase with distance from water (Tolsma *et al.*, 1987). Height diversity (structural variation) was calculated using Simpson's Diversity, D (Magurran, 1988) and was expected to increase with distance from water (Todd, 2006).

Herbaceous tuft size, average tuft diameter, was expected to increase with distance from water because of the shift from dominance by annual plants to greater dominance by perennial plants (Thrash *et al.*, 1991a). High intensity grazing, as found in the central piosphere zones, can also lead to smaller tuft size of perennials (Parsons *et al.*, 1997). Grass density, plants/m^2, was calculated by 1 / (average dist)^2 (Causton, 1988), and was expected to increase with distance from water (Thrash, 1998a).

Soil infiltration, nutrient cycling and stability indices were calculated using the LFA data analysis spreadsheet. Infiltration (Thrash, 1997; McIntyre & Tongway, 2005) and stability (Hogins & Rogers, 1997; Nash *et al*., 2003) were expected to increase with increasing distance from water. Nutrients were expected to decrease with increasing distance from water (Tolsma *et al.*, 1987; Turner, 1998). Bare ground cover, the percentage of herbaceous grid points recorded as bare ground, was expected to decrease with increasing distance from water (Thrash, 1998a; Smet & Ward, 2005).

The relationship between vegetation community composition and distance from water was investigated using ordination methods in CANOCO. Woody and herbaceous vegetation data were analysed separately and as a total vegetation dataset for each waterpoint. DCAs (Detrended Correspondence Analysis, for unconstrained tests) and DCCAs (Detrended Canonical Correspondence Analysis, for constrained tests) were used to test whether data should be subjected to unimodal or linear ordination.

Gradient length in the detrended analyses indicates heterogeneity in community composition (Leps & Smilauer, 2003). When the maximum gradient is long, over 4, unimodal methods should be used as community composition is heterogeneous (Leps & Smilauer, 2003). When the maximum gradient is short, less than 3, linear methods should be used as community composition is homogeneous (Leps & Smilauer, 2003). When the maximum gradient falls between 3 and 4, either unimodal or linear methods can be used (Leps & Smilauer, 2003). For these waterpoints, the unconstrained method chosen was the one that gave the highest percentage species variance explained by the first axis. For constrained tests, the method that gave the lowest p-value in the Monte Carlo test was chosen.

PCA (Principal Components Analysis, linear) or CA (Correspondence Analysis, unimodal) tests were used to determine whether there were any unknown or unexpected patterns relating the sample sites. To test the effect of distance to water, data were subjected to RDA (Redundancy Analysis, linear) or CCA (Canonical Correspondence Analysis, unimodal). The first ordination axis of the RDAs and CCAs was constrained (using distance to water), subsequent axes were unconstrained. All constrained models were tested using Monte Carlo permutation tests on the distance to water axis. Influence of spatial autocorrelation was avoided by correcting tests for the collection of data along a linear transect (Leps & Smilauer, 2003). Tests were corrected by restricting the sample reassignment during shuffling when calculating the Monte Carlo test statistic (Leps & Smilauer, 2003). Rare species were not downweighted because piospheres can have important impacts on rarity (Todd, 2006).

RESULTS

Soil

Bare ground cover varied from 0% to 44% (Table 6). The lowest bare ground cover was found in the private reserves and the highest in LNP. The most intensely managed property, MAL had the lowest average bare ground cover and the highest occurrence of 0% bare ground. Occurrence of 0% bare ground decreased with increasing management intensity (Table 6). Maximum coverage of bare ground tended to be found close to waterpoints and minima were found along the entire waterpoint transect (Figure 4).

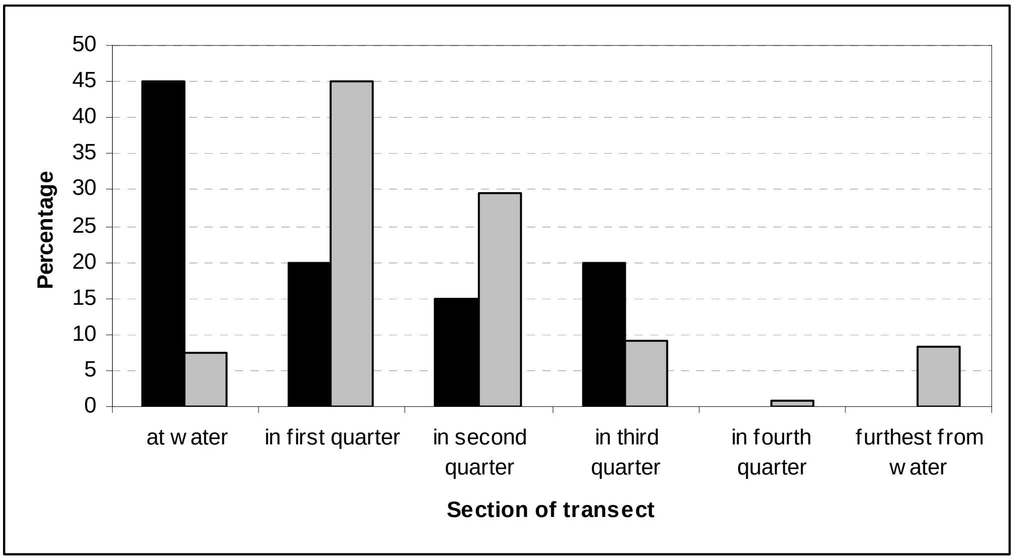

Figure 4: Locations of maxima (black bars) and minima (grey bars) of bare ground cover based on herbaceous grid data along waterpoint transects in the study area. Percentage of total maximum and minimum count shown as total count of minima is much greater than total count of maxima.

Table 6: Descriptive statistics of variables from sampling sites along waterpoint transects.

Bare ground cover (%)

Property	Minimum	Maximum	Average	SD	n	% min
LNP	0	43.8	8.1	11.7	48	40
KNP	0	32.5	3.3	6.8	40	50
KLA	0	18.8	1.5	4.0	53	75
GRC	0	8.8	1.5	2.3	44	57
MAL	0	13.8	0.8	3.1	20	90

Soil Function Indices (LFA)

Property	Variable	Minimum	Maximum	Average	SD	n
LNP	Nutrient cycling	11.4	57.9	24.3	9.8	47
KNP		7.9	53.7	22.1	9.3	44
GRC		12.1	40.1	20.9	6.3	33
MAL		11.1	26.1	16.8	4.3	20
LNP	Stability	36.5	72.9	57.1	7.8	47
KNP		38.2	71.9	55.9	7.7	44
GRC		39.1	68.6	54.2	6.3	33
MAL		43.2	61.1	52.9	4.2	20
LNP	Infiltration	23.2	61.8	32.5	7.8	47
KNP		12.7	51.6	26.0	8.0	44
GRC		12.6	42.3	27.3	6.2	33
MAL		17.7	32.2	23.7	4.4	20

Table 6 continued

Herbaceous vegetation density (plants/m^2)

Property	Minimum	Maximum	Average	SD	n
LNP	15	368	88	89	48
KNP	22	965	180	203	40
KLA	33	925	284	183	53
GRC	29	6944	292	1046	44
MAL	44	721	281	193	20

Herbaceous plant basal diameter (mm)

Property	Minimum	Maximum	Average	SD	n
LNP	11.70	54.11	33.29	9.13	48
KNP	6.53	65.69	30.32	14.33	40
KLA	5.35	32.33	16.68	6.39	53
GRC	4.19	33.26	14.56	6.73	44
MAL	9.61	25.21	15.85	4.47	20

Woody vegetation density (trees/m^2)

Property	Minimum	Maximum	Average	SD	n
LNP	0.01	4.46	0.64	0.88	48
KNP	0	2.62	0.39	0.46	40
KLA	0.05	0.40	0.17	0.08	53
GRC	0	0.88	0.24	0.16	44
MAL	0.01	0.54	0.20	0.13	20

Woody vegetation structural diversity (Simpson’s Diversity, D)

Property	Minimum	Maximum	Average	SD	n
LNP	0.05	1.00	0.47	0.24	48
KNP	0	0.79	0.45	0.23	36
KLA	0.36	0.85	0.69	0.01	53
GRC	0	0.78	0.59	0.17	44
MAL	0.40	0.76	0.61	0.11	20

Woody vegetation canopy cover (%)

Property	Minimum	Maximum	Average	SD	n
LNP	4	100	48	29	48
KNP	0	80	30	22	40
KLA	4	65	22	16	53
GRC	0	84	40	23	44
MAL	8	70	39	20	20

Woody vegetation species richness

Property	Minimum	Maximum	Average	SD	n
LNP	1	19	9.1	4.0	48
KNP	0	15	6.5	4.4	38
KLA	5	17	9.9	2.9	53
GRC	0	24	11.5	5.3	44
MAL	3	17	11.1	3.6	20

Table 6 continued

Woody vegetation density in height classes (trees/m^2)

Property	Height	Minimum	Maximum	Average	SD	n
LNP	< 1m	0	4.188	0.506	0.845	48
KNP		0	2.554	0.313	0.479	36
KLA		0	0.300	0.067	0.062	53
GRC		0	0.713	0.123	0.116	43
MAL		0.008	0.375	0.111	0.095	20
LNP	1 – 2m	0	0.713	0.089	0.127	48
KNP		0.013	0.229	0.079	0.059	36
KLA		0.004	0.092	0.041	0.020	53
GRC		0	0.204	0.082	0.056	43
MAL		0	0.108	0.043	0.034	20
LNP	2 – 3m	0	0.108	0.016	0.021	48
KNP		0	0.167	0.023	0.033	36
KLA		0	0.108	0.041	0.022	53
GRC		0	0.079	0.023	0.018	43
MAL		0	0.071	0.023	0.017	20
LNP	3 – 5m	0	0.046	0.013	0.012	48
KNP		0	0.046	0.013	0.012	36
KLA		0	0.071	0.018	0.013	53
GRC		0	0.033	0.011	0.010	43
MAL		0	0.046	0.012	0.011	20
LNP	> 5m	0	0.092	0.014	0.018	48
KNP		0	0.021	0.003	0.005	36
KLA		0	0.021	0.004	0.005	53
GRC		0	0.029	0.008	0.008	43
MAL		0	0.025	0.006	0.008	20

The soil nutrient cycling index varied from 9.8 to 57.9 (Table 6). LNP had the highest nutrient cycling and the private reserves the lowest. The most intensely managed property, MAL, had the lowest average nutrient cycling. Maxima and minima for soil nutrient cycling followed the same pattern with occurrences decreasing with increasing distance from water (Figure 5). The soil stability index varied from 36.5 to 72.9 (Table 6). There was little difference in average soil stability between the four properties (range of 52.9 to 57.1), and variability increased with average stability. Minimum values showed a peak close to water and maximum values showed a peak in the first quarter of the transect (Figure 5). Otherwise, distribution of maxima and minima were the same. The soil infiltration index varied from 12.6 to 61.8 (Table 6). MAL had the lowest infiltration index and LNP had the highest. Maximum and minimum values showed similar distributions along the transect (Figure 5).

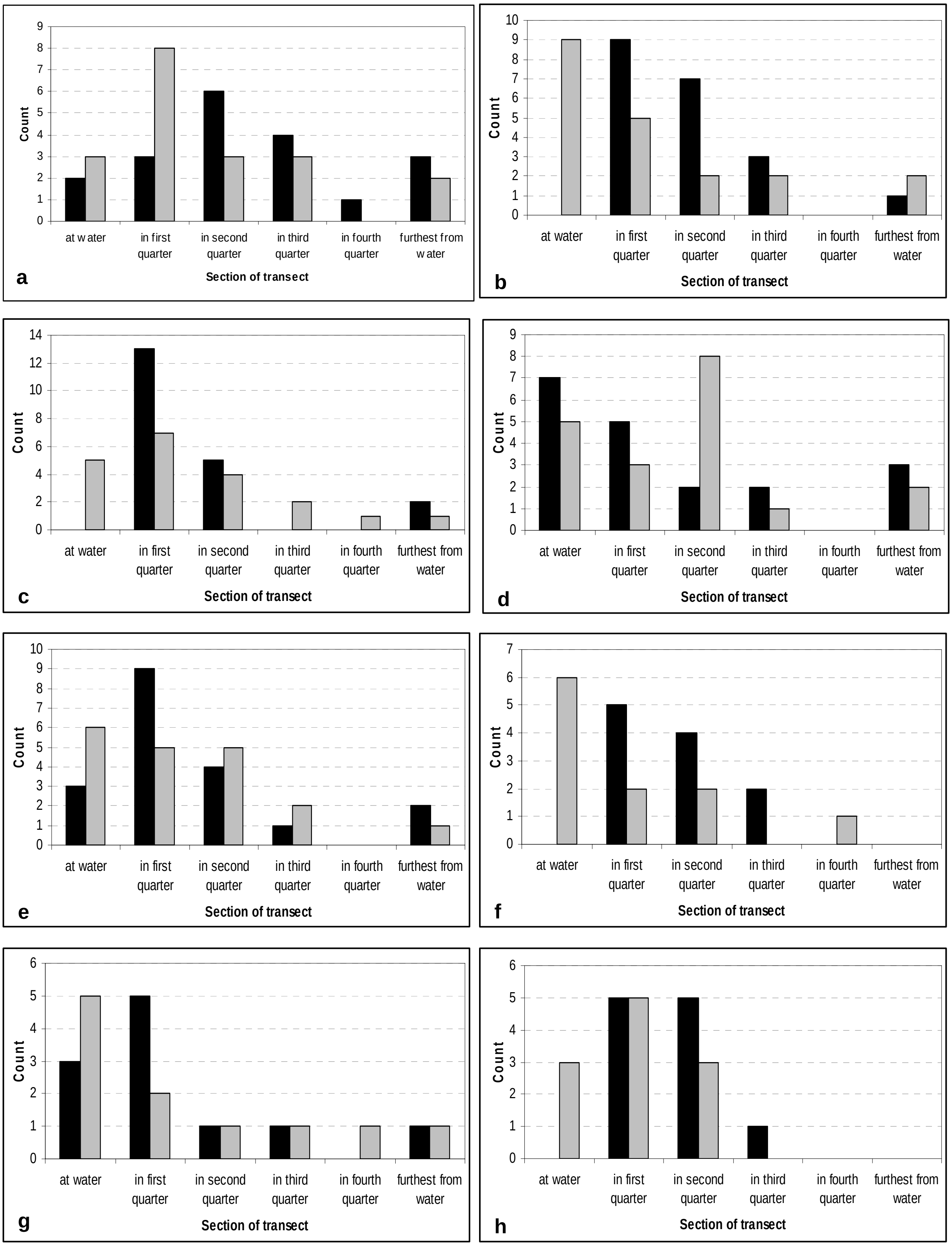

Figure 5: Frequency distributions of maxima (black bars) and minima (grey bars) occurrences for variables along waterpoint transects. a = structural variation, b = woody canopy, c = woody species richness, d = herbaceous density, e = herbaceous basal diameter, f = soil nutrient cycling index , g = soil stability index and h = soil infiltration index.

Herbaceous

The most relatively abundant herbaceous species were *Urochloa mossambicensis* (17%), *Panicum maximum* (15%) and *Digitaria eriantha* (11%). A total of 78 species were sampled with three occurring only once (Appendix 8). KNP was dominated by *Urochloa mosambicensis* (15%), *Bothriochloa radicans* (13%), *Digitaria eriantha* and *Panicum maximum* (11% each). LNP was dominated by *Panicum maximum* (30%) and *Digitaria eriantha* (22%). The private reserves were dominated by *Urochloa mossambicensis* (23% of records) and *Panicum maximum* (10%). The private reserves had the highest diversity of species (9.85/plot) followed by KNP (8.12/plot) and then LNP (7.44/plot).

Herbaceous vegetation density varied from 15 to 6944 plants/m^2 (Table 6). The maximum value, 6944 plants/m^2 was from a riverbank in GRC covered by a *Cynodon dactylon* lawn where average distance to tuft was 12mm (SD = 8) and average tuft size was 4mm (SD = 3). The private reserves had the highest and LNP had the lowest average density of herbaceous vegetation. Maximum and minimum values of distance to tuft were distributed along the entire waterpoint transect with a peak in maximum densities closer to waterpoints (Figure 5). Average basal diameter of grass tufts at a sampling site varied from 4.2mm to 65.7mm (Table 6). Average diameters on the private reserves were about half those in the national parks. Maximum and minimum tuft sizes tended to be found in the first half of the transects (Figure 5).

Using the DCAs to test for unimodal or linear ordinations led to the use of seventeen PCAs and five CAs. There were no species that consistently caused spreading of sampling sites in unconstrained ordinations. 26 outliers were identified, 50% of which occurred at 0m from water. Of these, 62% were located at dams (open and closed) and 38% were located at natural waterpoints. 11% of outliers at waterpoints were caused by high occurrence of *Sporobolus ioclados*, 9% by *Cynodon dactylon* and 9% by *Dactyloctenium aegyptium*. Variance explained by the first axis varied from 17.6% to 84.4% (average = 47.8, SD = 18.1).

Woody

The most relatively abundant woody species of the study area were *Colophospermum mopane* (38%), *Acacia nigrescens* (12%) and *Grewia bicolor* (8%). A total of 152 species were sampled with 30 occurring only once (Appendix 9). KNP was dominated by *Colophospermum mopane* (62%). LNP was dominated by *Colophospermum mopane* (51%) and *Acacia nigrescens* (19%). The private reserves were dominated by *Grewia bicolor* (22%), *Acacia nigrescens* (12%) and *Combretum apiculatum* (7%). The private reserves had the highest overall woody species diversity (10.82 species/plot), followed by LNP (9.52 species/plot) and then KNP (7.64 species per plot). Plot level woody plant species richness varied from 0 to 24 (Table 6). There was little difference in the distribution of species richness maximum and minimum scores with most occurring in the first half of the waterpoint transect (Figure 5).

Density of woody plants varied from 0 to 4.45 trees/m^2 (Table 6). Overall, the private reserves had the lowest woody plant densities and LNP had the highest. Minimum values for density of woody plants were distributed across all sections of the transect while over 50% of the maximum values were found in the first quarter of the transect (Figure 6). Canopy cover varied from a minimum of 0% to a maximum of 100% (Table 6). Full canopy cover was only found in LNP. KLA had the lowest overall canopy cover. The occurrence of both maximum and minimum values of canopy cover declined with increasing distance from water (Figure 5). Woody vegetation structural diversity varied from 0 to 1 (Table 6). The private reserves had a higher diversity than the national parks which were similar to each other. Maxima and minima of structural diversity had similar distributions with the majority of both being found near to waterpoints (Figure 5).

The distribution of woody vegetation height classes varied between management intensities (Figure 6). The low management intensity property, LNP, had the most variation and the highest proportion of >5m trees. Medium management intensity KNP had the most regular distribution of height classes within and between sampling sites. The shortest height class (which includes seedlings) and the tallest height class were at their maximum density in LNP (Table 6). The lowest density of the shortest height class was found on MAL and the lowest density of the tallest height class was found on KNP. Height class maxima and minima did not separate along the length of the transect (Figure 7).

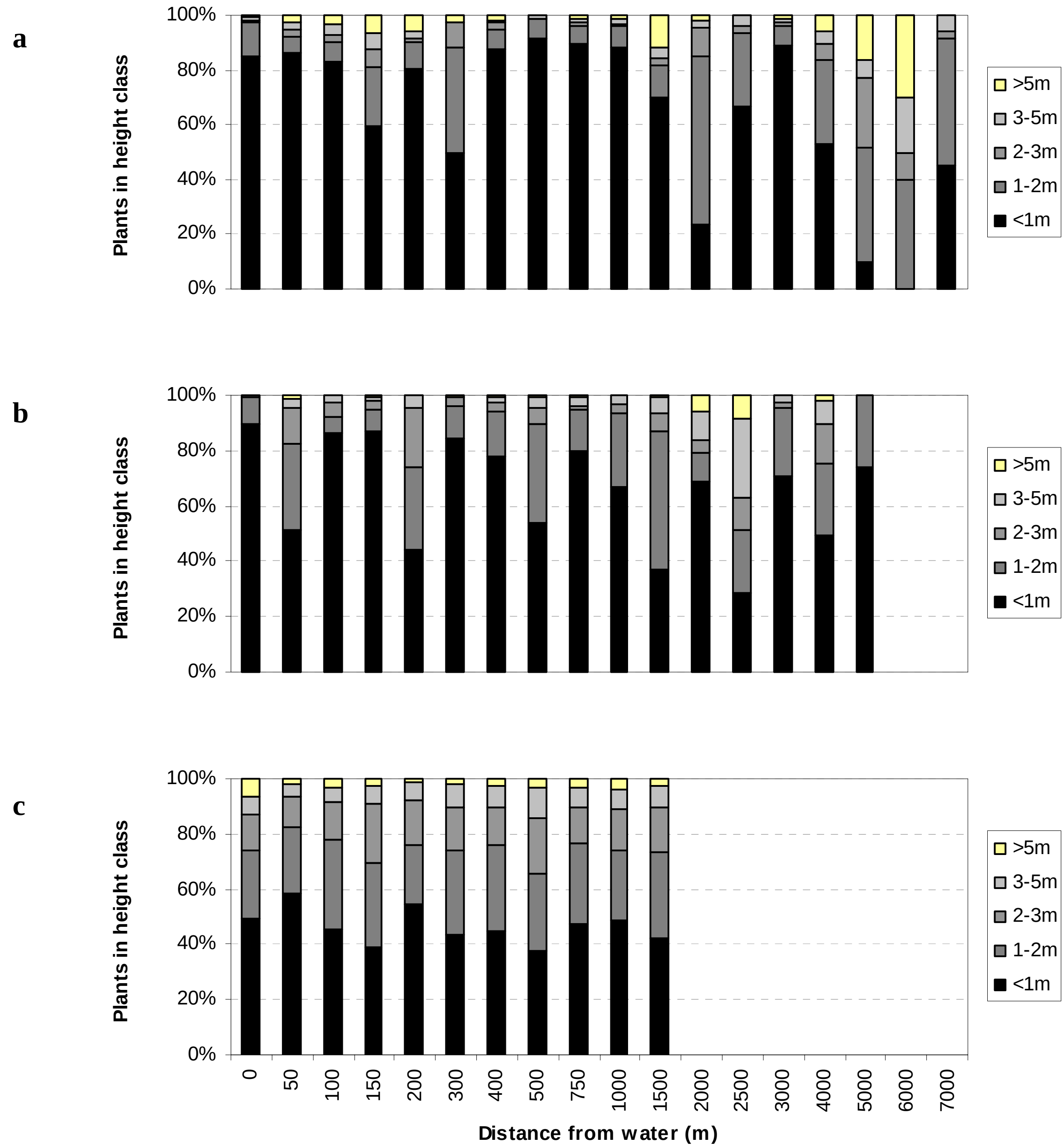

Figure 6: Distribution of height classes along distance from water transects for **(a)** Limpopo National Park, **(b)** Kruger National Park, and **(c)** the private reserves.

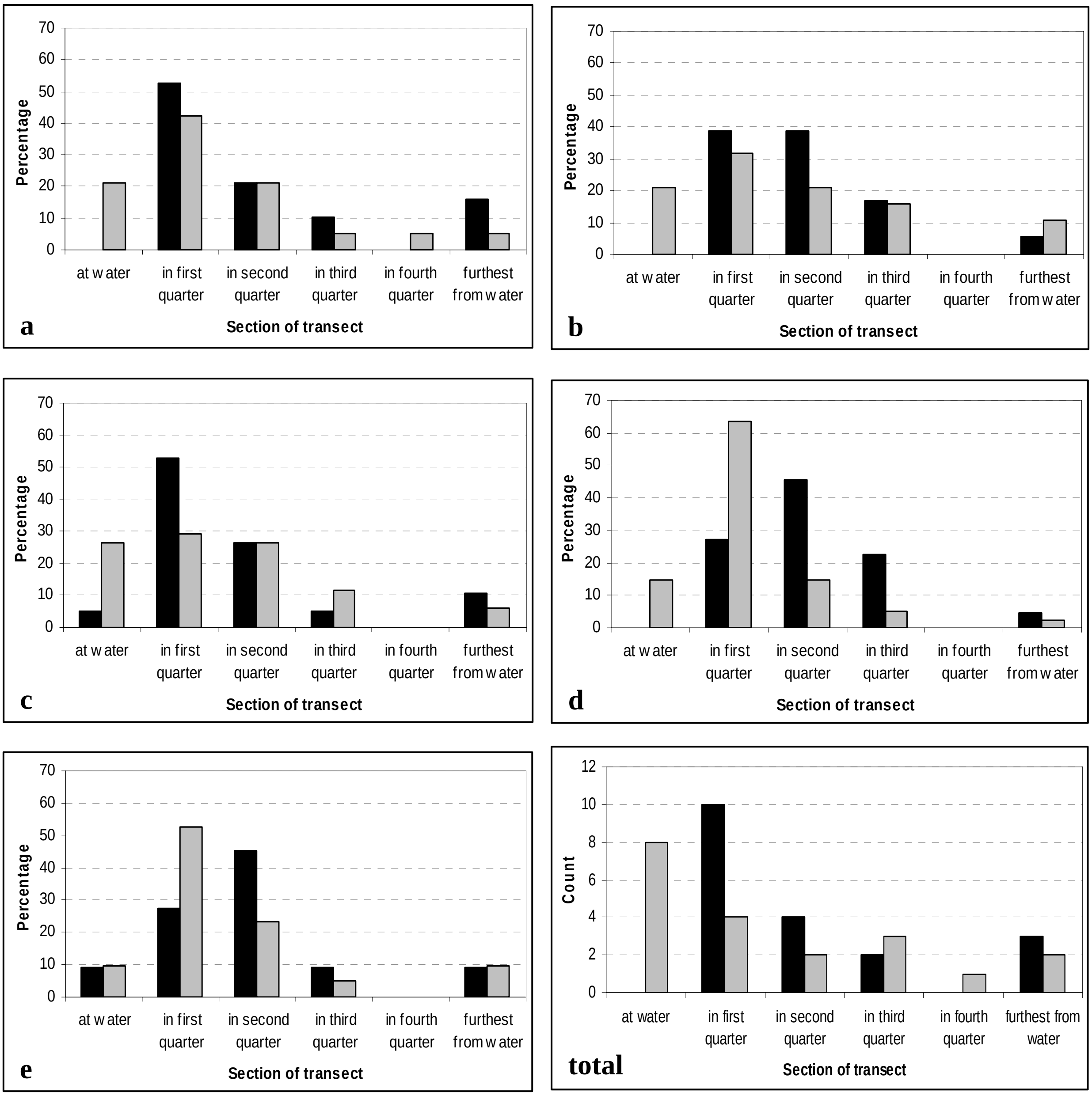

Figure 7: Frequency distributions of maxima (black bars) and minima (grey bars) occurrences for woody plant densities (height classes and total woody vegetation) along waterpoint transects. Height class a = < 1m, b = 1 – 2m, c = 2 – 3m, d = 3 – 5m and e = > 5m.

Using the DCAs to test for unimodal or linear ordinations led to the use of twenty PCAs and two CAs for woody vegetation. Sampling sites were often separated based on the occurrence of *Colophospermum mopane* in KNP and LNP and on occurrence of *Grewia bicolor* and *Acacia nigrescens* in the private reserves. 20 outliers were identified but no clear pattern in relation to distance from waterpoint. Variance explained by the first axis varied from 17.6% to 99.3% (average = 66.1, SD = 24.6).

Testing distance to water using regression

A total of 356 linear regressions were done on 20 variables across 19 waterpoints. 37 of the 356 tests (10%) revealed significant relationships with distance to water (Table 7). For each of the variables, the maximum percentage of significant tests (i.e. the percentage of transects which tested significant for a relationship with water for this variable) was 27% for LFA soil stability index. The minimum percentage of significant tests was 0% for woody species richness, proportion in woody height class a, proportion in woody height class b and density of height class e. 40% of the variables had significant results for less than 10% of their tests (Figure 8). No variable could be found that had a consistent relationship with distance to water.

For each of the transects, the maximum percentage of significant tests (i.e. the percentage of variables from a transect which tested significant for a relationship with water) was 24% for both C20 and D20. Five waterpoints, representing all waterpoint types over all management intensities, had no significant tests. 53% of transects had significant results for less than 10% of their tests (Figure 8). No transect could be found to illustrate a classical piosphere pattern for more than 4 out of a total of 20 variables which were expected to show relationships as based on results of previous studies.

The logistic regressions (356 tests) performed better than the linear regressions with 68 significant tests (19%) (Table 8). For each of the variables, the maximum percentage of significant tests was 42% for grass tuft size and herbaceous density. The minimum percentage of significant tests remained at 0% but only for proportion in height class b. The percentage of significant tests increased with 90% of variables having more than 10% of tests significantly related to distance to water (Figure 8). Herbaceous density, one of the strongest variables, showed contradictory effects of distance to water: JEJ and PUM are waterpoints of the same type from the same property yet they have opposite relationships between herbaceous density and distance to water (Figure 9).

Table 7: Overall significance results of linear regressions for variables along distance from water transects. ** = significant at $p < 0.05$, ns = not significant at $p < 0.05$, - = not tested. Transect full names given in Table 4.

Transect	% bare ground	LFA stability index	LFA nutrient cycling index	LFA infiltration index	Grass tuft size	Grass density	Species richness	Structural variation	Proportion Height Class a	Proportion Height Class b	Proportion Height Class c	Proportion Height Class d	Proportion Height Class e	Woody density	Density Height Class a	Density Height Class b	Density Height Class c	Density height Class d	Density Height Class e	Woody canopies
BNK	**	**	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	ns	ns
LHP	ns	-	-	-	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
MCP	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns
NGW	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
BFG	ns	**	ns	ns	ns	ns	ns	**	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns
EGH	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns
RGN	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
DBD	ns	-	-	-	ns	ns	ns	ns	ns	ns	**	**	ns	ns	ns	ns	ns	ns	ns	ns
IAN	ns	-	-	-	ns	ns	ns	**	ns	ns	ns	**	**	ns	ns	ns	ns	ns	ns	ns
JEJ	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	**	**	**	ns	ns	ns
PUM	ns	**	**	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
RVR	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns
RVS	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
SEE	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	**	ns	**
C20	ns	-	-	-	**	ns	ns	ns	ns	ns	ns	ns	ns	**	ns	**	ns	**	ns	ns
C65	ns	-	-	-	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
D20	ns	-	-	-	**	ns	ns	ns	ns	ns	ns	ns	ns	**	**	ns	**	ns	ns	ns
P26	ns	-	-	-	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
R01	ns	-	-	-	ns	ns	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns

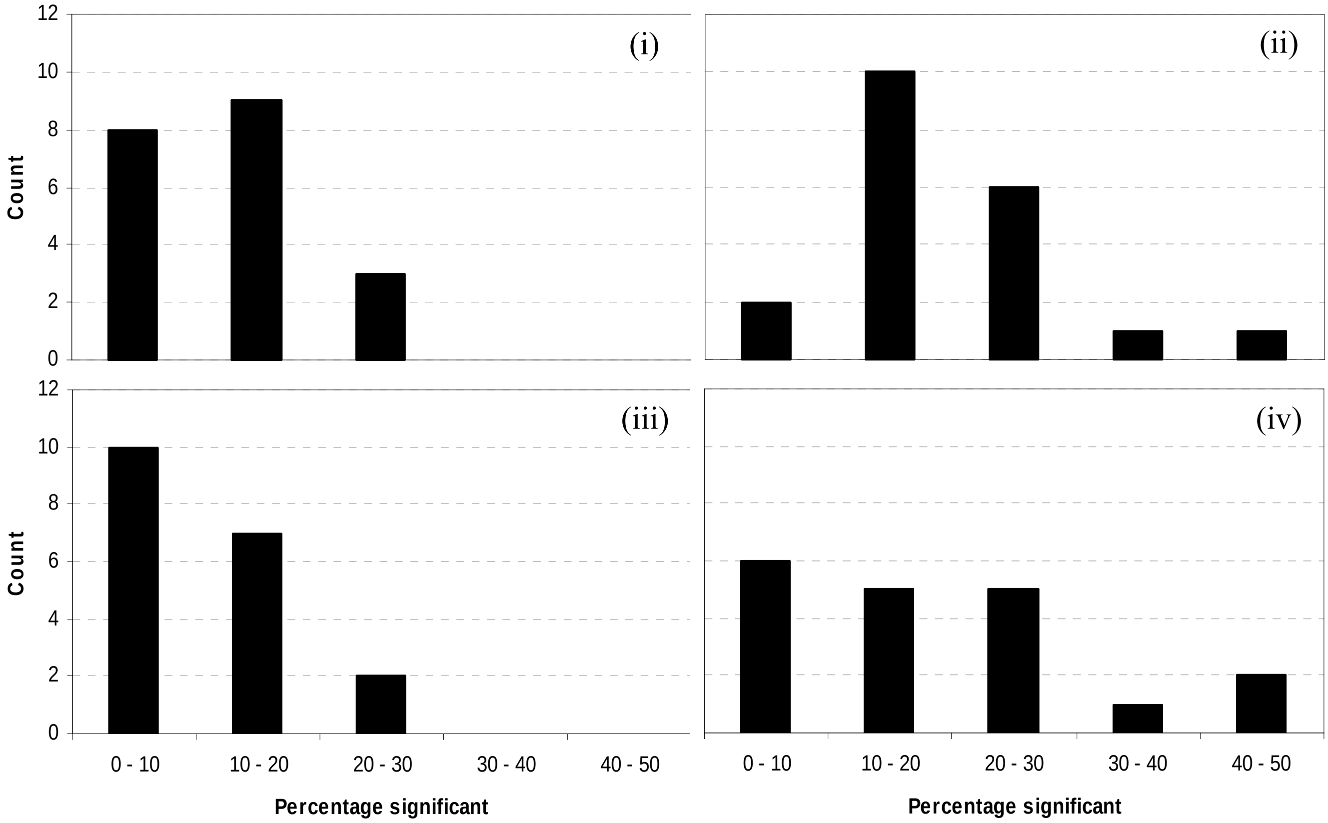

Figure 8: Frequency distribution of the percentage of statistically significant tests for (a) variables and distance to water by (i) linear regression and (ii) logistic regression and (b) transects and distance to water by (iii) linear regression and (iv) logistic regression.

Table 8: Overall significance results of logistic regressions for variables along distance from water transects. ** = significant at $p < 0.05$, ns = not significant at $p < 0.05$, - = not tested. Transect full names given in Table 4.

Transect	% bare ground	LFA stability index	LFA nutrient cycling index	LFA infiltration index	Grass tuft size	Grass density	Species richness	Structural variation	Proportion Height Class a	Proportion Height Class b	Proportion Height Class c	Proportion Height Class d	Proportion Height Class e	Woody density	Density Height Class a	Density Height Class b	Density Height Class c	Density height Class d	Density Height Class e	Woody canopies
BNK	**	**	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	ns	ns
LHP	ns	-	-	-	**	ns	ns	ns	**	ns	**	ns	ns	**	**	ns	ns	ns	ns	ns
MCP	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	**	ns	ns	**	ns	ns	ns	ns
NGW	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns
BFG	ns	**	ns	ns	ns	ns	**	**	**	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns
EGH	ns	ns	ns	ns	**	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	ns
RGN	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
DBD	ns	-	-	-	**	ns	ns	**	**	ns	**	**	**	ns	ns	ns	ns	ns	**	ns
IAN	ns	-	-	-	**	**	**	**	**	ns	ns	**	**	ns	**	ns	ns	ns	ns	ns
JEJ	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	**	**	**	**	ns	ns	ns
PUM	-	**	**	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
RVR	ns	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	**	**	ns	ns	ns	ns	ns	ns	**
RVS	**	ns	ns	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
SEE	ns	ns	**	ns	**	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	**	ns	**
C20	-	-	-	-	**	ns	**	ns	ns	ns	ns	ns	ns	**	ns	**	**	**	ns	ns
C65	ns	-	-	-	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
D20	**	-	-	-	**	ns	ns	ns	ns	ns	ns	ns	ns	**	**	ns	**	ns	ns	ns
P26	ns	-	-	-	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
R01	ns	-	-	-	ns	ns	ns	ns	**	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns

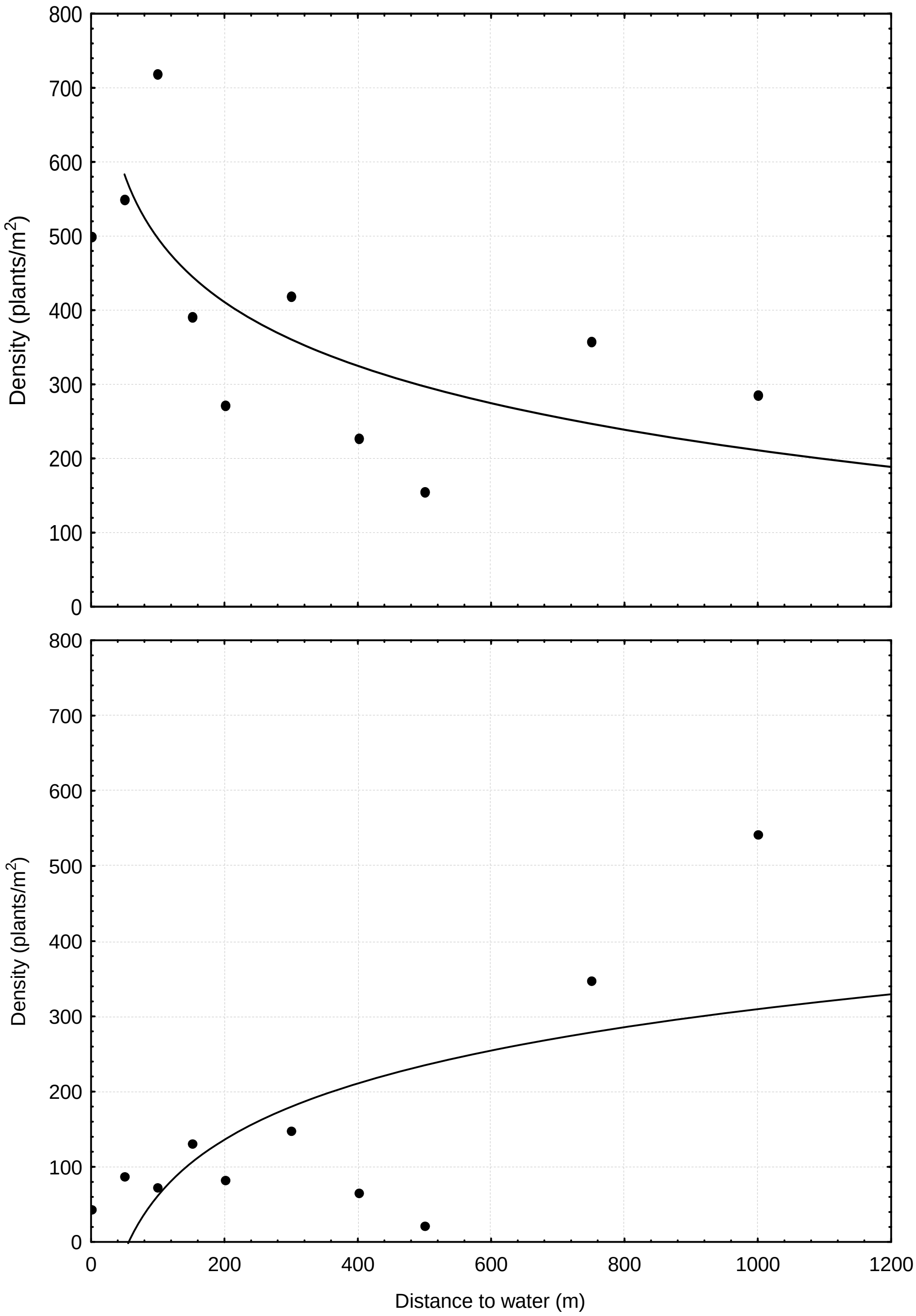

Figure 9: Comparison of the relationship between distance to water and density of herbaceous species from two open artificial dams on one property. Both relationships are significant when tested with logistic regression.

For the transects, the maximum percentage of significant tests was 47% for IAN. Three waterpoints, representing all waterpoint types across KNP and the private reserves, had no significant tests. The distribution of significant tests improved with 68% of transects having over 10% of their tests significantly related to distance to water (Figure 8).

Testing distance to water using ordination

A total of 66 ordinations were run with only one giving a significant result. There were no significant ordinations (seventeen RDAs and five CCAs) for woody vegetation. Eighteen RDAs and four CCAs were run on herbaceous vegetation with one giving a significant result. There were no significant ordinations (fourteen RDAs and eight CCAs) for total vegetation. Overall, much less variation was explained by distance to water than was explained in the unconstrained ordinations (Figure 10).

For woody vegetation, eigenvalues for distance to water ranged from 0.007 to 0.647. 82% of the tests revealed that the second eigenvalue (unconstrained) was a better predictor than distance to water. Variance explained by distance to water ranged from 0.7% to 64.7% (average = 21.7, SD = 18.7). This average is only 33% of the average variance explained by the first axis of the unconstrained ordinations.

There was one significant ordination (RDA) for herbaceous vegetation (Monte Carlo Test: Trace = 0.368, F-ratio = 7.579, $p < 0.05$), from BNK in LNP (Figure 11). Eigenvalues for herbaceous vegetation ranged from 0.033 to 0.648. 91% of the tests revealed that the second eigenvalue (unconstrained) was a better predictor than distance to water. Variance explained by distance to water varied from 3.3% to 50.9% (average = 18.9, SD = 12.3). This average is only 40% of the average variance explained by the first axis of the unconstrained ordinations.

For total vegetation, eigenvalues ranged from 0.008 to 0.541. 82% of the tests revealed that the second eigenvalue (unconstrained) was a better predictor than distance to water. Variance explained by distance to water varied from 0.8% to 47.1% (average = 18.0, SD = 11.8). This average is only 36% of the average variance explained by the first axis of the unconstrained ordinations.

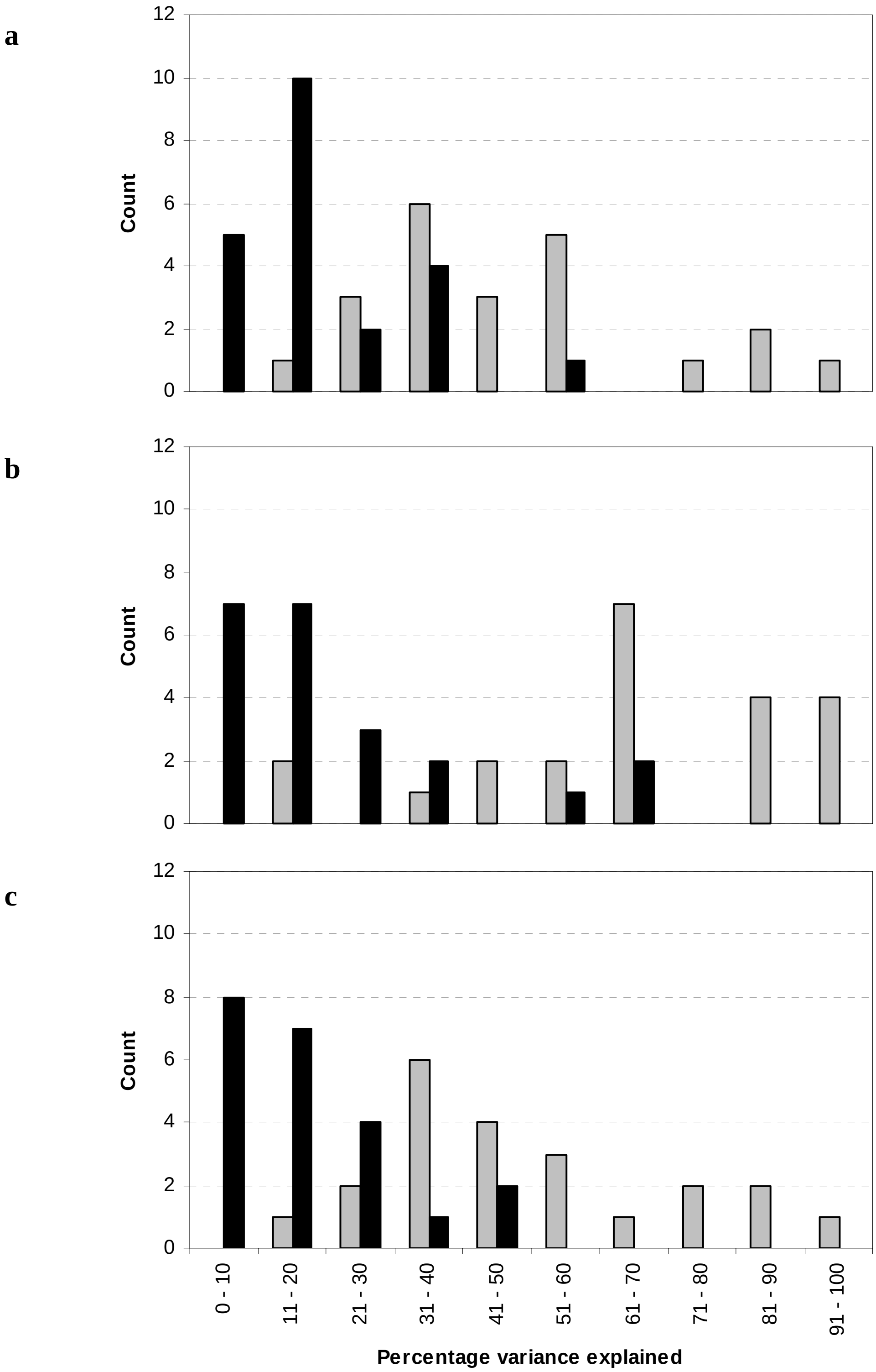

Figure 10: Comparison of distribution of percentage variance explained between ordinations of (a) herbaceous vegetation, (b) woody vegetation, and (c) total vegetation. Black bars indicate percentage variance explained by distance to water (constrained ordinations). Grey bars indicate percentage variance explained by the first axis of the unconstrained ordinations.

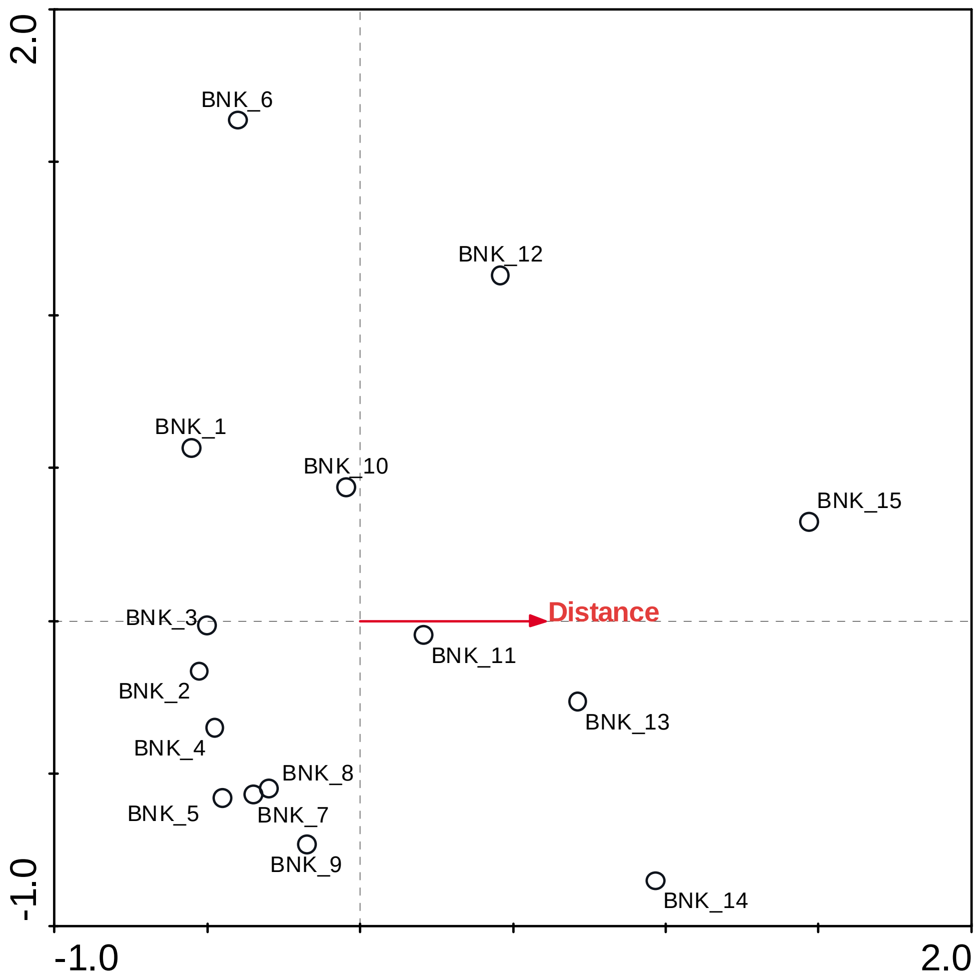

Figure 11: RDA ordination plot for waterpoint BNK from Limpopo National Park. Distance to water (Distance) had a significant effect on species composition.

DISCUSSION

Understanding the impact of herbivores around waterpoints is critical for understanding the impacts of water supplementation on properties and therefore for making informed management decisions about supplementation levels. To date, understanding of impacts of water supplementation in the southern African savanna is based on the piosphere model developed in Australia (Owen-Smith, 1996; Gaylard *et al.*, 2003). The high level of heterogeneity found in southern African savannas (Pickett *et al.*, 2003) has led to questioning of the application of a homogeneity based model in management (Chapter 3). This study aimed to determine the general applicability of the piosphere model in heterogeneous southern African savannas. A total of 23 variables (soil functionality and herbaceous and woody vegetation) were investigated over 22 waterpoints from five properties. A total of 778 tests were performed to test the effect of distance to water and only 106 tests (14%) were significant. This very low general explanatory power of the piosphere model leads to the recommendation that it is not a good basis for understanding the impacts of water provision in heterogeneous southern African savannas. Because of the lack of consistent pattern between variables and waterpoints, results from small scale homogeneous studies cannot be transferred to unsampled waterpoints or scaled up across properties.

Previous piosphere studies in the same area have concluded in favour of the piosphere approach (Thrash *et al.*, 1991a; Thrash, 1997; Brits *et al.*, 2002). However, these studies were often limited to small sample sizes and distances. Significant relationships between degradation variables and distance to water were found in specifically selected homogeneous areas (Thrash, 1997, 1998a,b; Brits *et al.*, 2002; Smet & Ward, 2005). Studies at broader scales in other areas have found problems with applying the piosphere approach in heterogeneous landscapes (Nangula & Oba, 2004; Washington-Allen *et al.*, 2004; Chamaillé-Jammes *et al.*, 2009). The length of transects used in this study, and the variety of waterpoints sampled enabled the effects of landscape heterogeneity to be seen.

All variables chosen relate to ecosystem health as this is important for ecosystem function and resilience and therefore for reserve management. Soil is the basis of ecosystem health as loss of soil can lead to loss of vegetation (Rietkerk & van de Koppel, 1997; MacGregor & O'Connor, 2002). Soil is an ultimate determinant of vegetation patterns and intact soil is essential for proper ecosystem function (Bell, 1982; Swanson *et al.*, 1988; Venter *et al.*, 2003). Vegetation cover and quality are important for functional integrity of a landscape and the biodiversity that it retains (Ludwig *et al.*, 2004). Vegetation is a proximate determinant of herbivore population size and is therefore particularly important when objectives are production related (Grossman *et al.*, 1999).

The private reserves had the worst quality soil and therefore the lowest potential ecosystem health even though they have the lowest coverage of bare soil. Herbaceous vegetation density and species per plot was highest in the private reserves. The smaller tuft sizes found in the private reserves could be due to a high occurrence of annual plants. However, the grasses were dominated by *Urochloa mossambicensis* and *Panicum maximum*. Woody vegetation of the private reserves had the highest species richness per interval and overall species per plot. Density and cover of woody vegetation was lower than in the national parks with less variation between intervals. There have been questions raised about the sustainability of the high grazing impact on the private reserves (Craine et al. 2009). Previous work has found smaller tufts in areas with more intensive grazing (Parsons *et al.*, 1997) and a greater sensitivity of large tussock grasses to grazing (McIntyre & Tongway, 2005). These results suggest that although the ecosystem is currently supporting relatively high densities of herbivores, its resilience is low (MacGregor & O'Connor, 2002).

KNP are trying to enhance biodiversity and natural processes with minimal management intervention (Mabunda *et al.*, 2003). However, they are also dealing with a history of management impact (Mabunda *et al.*, 2003). In terms of soil function, KNP is similar to LNP for nutrient cycling and stability though the infiltration is worse and there is less bare ground. Herbaceous vegetation in KNP is similar to LNP in terms of basal diameter of tufts but KNP have a higher overall herbaceous species diversity (species per plot) and density. Overall woody vegetation species per plot in KNP is lower than LNP, as is their woody density, canopies and species richness. In general, KNP tended to fall between LNP and the private reserves.

The healthiest soil was found in LNP with the highest values for nutrient cycling, stability and infiltration. However, average bare ground cover in LNP is highest and the proportion of sampling sites with no bare ground is lowest. A medium overall number of grass species per plot was found in LNP with the lowest density of herbaceous plants. The highest basal diameter of grass tufts was found here indicating low levels of utilisation (Parsons *et al.*, 1997; McIntyre & Tongway, 2005). A medium level of woody vegetation species richness and overall species per plot were found although the highest density and canopy cover of woody vegetation were found in LNP.

The first indication that the piosphere model may not be generally applicable was the lack of consistent pattern in the maxima and minima graphs. If the piosphere model held, variables should show separation of their maximum and minimum values along the distance from water transect. When looking at formal testing of the variables, the piosphere model (distance from water) explained variations in variable measurements in less than 50% of each kind of test. Previous studies have successfully used linear regression (Thrash *et al.*, 1991a; Riginos & Hoffman, 2003), logistic regression (Thrash *et al.*, 1991a; Thrash, 1997, 1998a,b; Brits *et al.*, 2002; Riginos & Hoffman, 2003) and ordinations (Fernandez-Gimenez & Allen-Diaz, 2001; Heshmatti *et al.*, 2002) to find relationships between variables and distance to water. However, these studies were often either based on homogeneous study sites or were small scale (Thrash, 1997, 1998b). If the piosphere model was a suitable basis for understanding and management of water supplementation, a larger proportion of significant tests should have been found.

The soil stability index showed the highest percentage of significant relationships with distance to water under linear regression. Woody densities increased with distance from water, as expected from previous studies (Thrash *et al.*, 1991b; Brits *et al.*, 2002). Grass tuft size also increased with distance from water, as expected (Parsons *et al.*, 1997; McIntyre & Tongway, 2005). Half of the transects with no significant relationships were at natural waterpoints. Logistic regression performed better than linear regression as expected as it has been stated in previous studies that logistic regression is a better intuitive fit to piosphere data (Thrash *et al.*, 1991a; Thrash, 1997, 1998a,b). Grass tuft size showed the highest percentage of significant relationships with distance to water under logistic regression. Five transects showed an increase in grass tuft size with distance from water whilst three showed a decrease in grass tuft size. It was expected that grass tuft size would increase with distance from water (Parsons *et al.*, 1997; McIntyre & Tongway, 2005) and this was found for 63% of the significant relationships. For the variables, a high proportion of significant relationships did not always have the same directional relationship with distance to water.

Ordination of species revealed only one significant relationship with distance from water, a natural waterpoint in LNP. On this transect, *Digitaria eriantha* and *Bothriochloa radicans* increased with distance from water while *Panicum maximum* decreased. This species composition does not fit a piosphere utilisation gradient as *B. radicans* increases in disturbed areas (van Oudtshoorn, 2004). This distribution may be reflecting habitat preferences as *P. maximum* often occurs in shady, fertile areas near rivers (van Oudtshoorn, 2004). In all constrained ordinations, the second eigenvalue was often stronger than the first, illustrating the weakness of the distance to water effect. Similar work in other areas has found that environmental variables can have stronger effects than distance to water (Makhabu *et al.* 2002).

The low number of significant piosphere tests is likely to be due to violation of an assumption of the piosphere model. The piosphere model was developed in homogeneous vegetation (Lange, 1969) and the fit of the logistic equation assumes underlying environmental homogeneity (Graetz & Ludwig, 1978). The standard piosphere relationship is obtained when the water-forage trade-off is the only factor affecting herbivore movement patterns (Graetz & Ludwig, 1978; Adler & Hall, 2005). Savannas are heterogeneous at multiple scales (Pickett *et al.*, 2003) creating a range of mosaics (Law & Dickman, 1998; Bowyer & Kie, 2006) which foraging herbivores respond to (Arditi & Dacorogna, 1988; Bailey *et al.*, 1996).

Spatial heterogeneity refers to the degree of difference between parts of a landscape including the amount and spatially explicit configuration of the environmental resources and constraints (Pickett *et al.*, 2003). Savanna heterogeneity arises at broad scales from geology and climate and at finer scales from fire and herbivore (Scholes, 1990; Skarpe, 1992; Wiegand *et al.*, 2006). Vegetation is then further modified by invertebrate (Mobaek *et al.*, 2005) and vertebrate (Verweij *et al.*, 2006; Anderson *et al.*, 2007; Waldram *et al.*, 2008) herbivores, rainfall (Wiegand *et al.*, 2006) and fire (Parr & Anderson, 2006; Govender *et al.*, 2006; Higgins *et al.*, 2007). The interaction of multiple drivers gives rise to a savanna ecosystem mosaic at various scales (Scoones, 1995; Augustine, 2003).

This study focused on five of the 35 landscapes of KNP in order that results could be compared between properties. Landscape types are defined by a combination of geomorphology, soil, climate, vegetation and faunal characteristics (Gertenbach, 1983) and are therefore potentially homogeneous blocks. However, heterogeneity within the transects illustrates that even within these ‘homogeneous’ landscape types, vegetation and soil of different areas interact with herbivores. Extreme changes in vegetation between intervals in close proximity shows patchiness, for example the

Cynodon dactylon lawns next to some waterpoints. Species composition results indicate a higher level of homogeneity in woody species than in grasses.

In a heterogeneous area, the landscape template interacts with herbivores (Bailey *et al.*, 1996; de Knegt *et al.*, 2008) meaning that the water-forage trade-off is not the only factor affecting herbivore movement, as is required for the piosphere model (Graetz & Ludwig, 1978; Adler & Hall, 2005). Other studies have found that heterogeneity can disrupt the piosphere pattern. Vegetation type (Hanan *et al.*, 1991; Cridland & Stafford Smith, 1993; van Rooyen *et al.*, 1994; Friedel, 1997; Nangula & Oba, 2004), soil type (Hanan *et al.*, 1991; Nash *et al.*, 1999; Turner, 1999) and rainfall (Hanan *et al.*, 1991; Turner, 1999) have been found to affect herbivore utilisation leading to disruption of the piosphere pattern. Whilst utilisation gradients do exist for some variables around some waterpoints in southern African savannas, there are no consistent relationships that can be relied on for management. The concentric circular pattern implied by previous piosphere work (e.g. Owen-Smith, 1996) does not hold for the southern African savannas.

CONCLUSION

After investigation of waterpoints from a range of properties and landscape types, this study rejects the piosphere model as a basis for waterpoint understanding and management in heterogeneous southern African savannas. The lack of consistency in piosphere patterns between variables and waterpoints illustrates the unsuitability of this model. The heterogeneity found in southern African savannas violates the assumptions of the piosphere model.

Understanding and management of water provision in these savanna conservation areas needs to continue. Therefore, we need to develop a new approach to understanding and managing the impact of artificial water supplementation in heterogeneous southern African savannas. This new approach needs to take into account the underlying heterogeneity of the area and the effect this has on herbivore utilisation patterns.

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Chapter 5

Factors affecting herbivore impact around waterpoints in heterogeneous savanna: a new approach to understanding impact across properties

Helen Farmer

ABSTRACT

Water supplementation is a contentious management issue in the southern African savanna. Current management and understanding is based on the piosphere model which has been shown to not be applicable in heterogeneous savanna. This study aimed to contribute towards development of an alternative approach to understanding herbivore impact across a landscape by determining how position and type of waterpoint affect extent and intensity of impact and how environmental and management factors affect the function of a particular area. Data collected on soil functionality and herbaceous and woody vegetation were used to compare degradation within 200m of the waterpoint between different types of waterpoint and location. The effect of environmental and management variables on degradation variables (e.g. annual:perennial grasses) and species composition was tested using ordination methods (PCA and RDA). Waterpoint type significantly affected impact with artificial waterpoints being more degraded than natural waterpoints. Position of waterpoint did not significantly affect impact Ordination analyses revealed that environmental variables were stronger than management variables in explaining variation in degradation variables and species composition. The best explanation of variation was a combination of environmental and management variables. Using classifications of function based on environmental and management variables it was possible to develop a basic approach to characterising the functionality of properties.

KEY WORDS

Conservation management; grazing gradients; Landscape Function Analysis; piospheres; spatial variation

INTRODUCTION

Water management is a contentious issue in southern African savannas with many properties having a high degradation risk approach to water supplementation (Chapter 2). Current water management and understanding of impacts is based on the piosphere approach where variables of interest are simply the number of waterpoints and the distance between them (Redfern *et al.*, 2003; Smit *et al.*, 2007). Concentric circular impact patterns are used to understand projected impact of herbivores from waterpoints across the landscape (Gaylard *et al.*, 2003). This is then translated to ecosystem function and degradation with high degradation and low function near waterpoints (Thrash & Derry, 1999). Degradation is therefore taken as increasing directly and linearly with the density of waterpoints, and properties with higher waterpoint densities are said to be less resilient to disturbance. There is evidence that properties with high waterpoint densities support larger water-dependent herbivore populations and that this causes reduced resilience of the system (Walker *et al.*, 1987; Craine *et al.*, 2009). However, the spatial patterning of this variation in resilience has not been shown.

The lack of a consistent piosphere pattern (Chapter 4) indicates that an alternative understanding is required for the spatial effect of water supplementation on ecosystem function and therefore on property resilience. Intrinsic system processes such as soil quality (Rietkerk *et al.*, 1997; Harrison & Shackleton, 1999) and the impact of management actions such as increased herbivory (Walker *et al.*, 1981; Mworia *et al.*, 1997) both contribute to ecosystem resilience. Piosphere studies in savanna conservation areas have focused on homogeneous landscape units (Thrash *et al.*, 1991b; Thrash, 1997, 1998; Brits *et al.*, 2002) and found that the piosphere approach works. These studies also tend to investigate the effects of herbivores with no reference to the influence of intrinsic system processes (Thrash *et al.*, 1991a,b).

Environmental variables can be considered as representing intrinsic system processes. Soil is the basis of ecosystem health (Rietkerk & van de Koppel, 1997; MacGregor & O'Connor, 2002) and vegetation patterning at small to medium scales (Bell, 1982; Venter *et al.*, 2003). Landscape types are determined by a combination of geomorphology, soil, climate and vegetation (Gertenbach, 1983) and therefore represent intrinsic system properties. Within landscapes, nutrients and water availability vary through the catena with subsequent effects on vegetation (Scholes, 1990; Witkowski & O'Connor, 1996; Venter *et al.*, 2003), for example there is a lower herbaceous and woody vegetation biomass on upper slopes and crests (Augustine, 2003; Wu & Archer, 2005). Photoperiod, the radiation that vegetation receives, influences vegetation growth (Schulze, 1997) and is therefore potentially important in vegetation resilience to herbivore impact. Aspect and slope affect photoperiod (Schulze, 1997). Slope can have additional effects on soil susceptibility to erosion as steeper slopes have a faster resource flow (Tongway & Hindley, 2004).

Natural water availability is an important intrinsic system property when considering the effect of water supplementation. Different landscape types have different levels of natural water availability (Chapter 2) and will therefore respond differently to increased herbivore pressure. Geomorphology, rainfall and soil contribute significantly to natural water retention (Ayeni, 1977; Gaylard *et al.*, 2003). Differences in soil type and higher levels of soil moisture mean that drainage lines have vegetation which differs from the rest of the landscape (Venter *et al.*, 2003; Wu & Archer, 2005). These differences affect the attractiveness of drainage lines and therefore could influence herbivore utilisation and movement across the surrounding landscape. Additionally, perennial rivers offer a permanent source of water.

The impact of management actions are manipulations of the ecosystem and therefore the differ from, and potentially effect, intrinsic system properties and processes. Property size is important as smaller properties are less natural because management actions are intensified (Baker, 1992; Peel *et al.*, 1999) and constrained by available area (Chapter 2). There is less area available for a natural disturbance regime (Baker, 1992). Fencing causes further disruption of natural processes as it results in the isolation of a property (Forman & Godron, 1981). Management decisions can differ across fence-lines and intensification of management on one side of the fence can lead to increased differentiation of the property from the surrounding landscape.

Artificial water availability is also an impact of management actions and not an intrinsic system property. Supplementation of permanent water increases herbivore-available water but not plant-available water. Artificial waterpoint density affects the stocking rate of a property as an increase in water availability leads to a higher density of water-dependent herbivores (Walker *et al.*, 1987). Differences have been found between impact patterns of natural and artificial waterpoints (Washington-Allen *et al.*, 2004). It is also important to differentiate between open and closed waterpoints. Open waterpoints affect herbivore numbers and distribution at the time of the study. Closed waterpoints should not be ignored because they can have long lasting legacy effects on soils and vegetation (K. Matchett, PhD Submitted 2010).

To inform a property that their basis of management is unsuitable (Chapter 4) is futile unless an alternative approach to management can be provided. This study therefore aimed to determine (1) how position and type of waterpoint affect surrounding impact, (2) how environmental and management factors affect degradation in a particular area, and (3) an alternative way to begin characterising the functionality of properties. Ultimately, this study is contributing towards development of an alternative approach to piospheres for understanding the impact of water supplementation on property function and resilience.

METHODS

During fieldwork performed for Chapter 4, data collected were more extensive than those which were analysed to test the general applicability of the piosphere approach. This chapter therefore uses data from the same sampling sites. Please refer to the methods section of Chapter 4 (pages 85 to 93) for detailed information on the study area, site selection and sampling approach.

Importance of position and type of waterpoint

This analysis was limited to the first 200m in order that waterpoints could be directly compared. This distance has been highlighted as the maximum important distance when considering waterpoint impact in southern African savannas (Thrash, 2000). Impact scores were calculated for bare ground (% cover), soil stability, infiltration and nutrient cycling (LFA indices); herbaceous density (plants/m^2), grass tuft size (mm), annual:perennial grass ratio and grazing value (an index calculated from production, palatability, nutritional value, growth vigour, digestibility and habitat preference (van Oudtshoorn, 2004)); woody vegetation density (plants/m^2), species richness, canopy cover (%) and height diversity (Simpson's D). Minimum, Q1, median, Q3 and maximum values were calculated for each variable and used to create four classes.

Using the median and interquartile range to create classes is more appropriate for skewed data.

Scores of 1 to 4 were assigned to each class with 1 being the least degraded and 4 the most degraded. Degradation was taken as a lower functionality (e.g. bush encroachment, low soil stability and high annual herbaceous vegetation). The overall score of a sampling site was calculated by averaging the scores of its variables. Impact scores were compared between distances, properties, transects, waterpoint type and waterpoint locations using One-Way ANOVA and Tukey's Pairwise Comparisons in Statistica. IAN, BVU and BVA were excluded from the transect analysis as they had less than four sampling sites in the 0 – 200m distance.

Factors affecting impact – Variables Considered

Ordination analysis was used to determine the influence of environmental and management variables on soil and vegetation. Four different types of variables were investigated: species composition, degradation variables, environmental variables and management variables. Species composition was taken as the woody and herbaceous vegetation of an area. Degradation variables were characteristics of vegetation and soils that respond to herbivore impact (e.g. the ratio of annual to perennial herbaceous vegetation (Parker & Witkowski, 1999) or soil infiltration (Milchunas & Lauenroth, 1993)). Environmental variables are variables that naturally characterise the condition of the landscape (e.g. catenal position (Augustine, 2003)). Management variables are variables that humans change that alter the condition of the landscape (e.g. artificial waterpoint density or fencing). Each sampling site was considered as individual and independent. Spatial auto-correlation was not considered to have a significant effect because of the lack of a strong gradient with distance to water for the variables under consideration (Chapter 4).

Species Composition

Herbaceous and woody vegetation were combined into a total vegetation data set for each sampling site. Herbaceous species composition is sensitive to grazing impacts as species vary in their ability to compensate for losses due to grazing (Milchunas & Lauenroth, 1993). Grazing can lead to spatial limitation of species occurrence and rarity in species frequency (Landsberg *et al.*, 2003). Woody species composition is sensitive to herbivore utilisation intensity. Direct utilisation of trees can lead to changes in composition (Levick & Rogers, 2008), as can indirect effects through alteration of soil quality (MacGregor & O'Connor, 2002). Species composition was analysed using the frequency of each species at each sampling site.

Degradation Variables

Bare ground can be considered as dysfunctional parts of the ecosystem (Ludwig *et al.*, 2000). Soil-vegetation relationships are important for maintaining healthy ecosystem states (Rietkerk & van de Koppel, 1997). Bare ground was calculated as the percentage of herbaceous grid points which were recorded as bare ground for each sampling site. Soil stability indicates the level and susceptibility of an area to erosion. High trampling and grazing can lead to loss of the upper protective layers of the soil (Belnap & Gillette, 1998) and reduction in surface variation (Nash *et al.*, 2003) which leads to increased soil loss through erosion (McIntyre & Tongway, 2005). Soil infiltration indicates the level of water which is available to plants for growth and reproduction (Castellano & Valone, 2007). Infiltration and available soil water are lower in areas with higher grazing (Pandey & Singh, 1991; Milchunas & Lauenroth, 1993; McIntyre & Tongway, 2005). Soil nutrients have been shown to have varied relationships with intensity of grazing (Milchunas & Lauenroth, 1993). Areas with high intensity grazing can be characterised by high nutrients because of deposition by herbivores (Fernandez-Gimenez & Allen-Diaz, 2001; McIntyre & Tongway, 2005; Craine *et al.*, 2009) or low nutrients because of loss through erosion (Mlambo *et al.*, 2005). Soil stability, infiltration and nutrient cycling indices were obtained from LFA calculations. LFA was not performed at 34% of sampling sites so nutrient cycling, stability and infiltration indices were given a reduced weight of 0.66 in ordination analyses.

Herbaceous quality consists of the ratio of annual to perennial grasses (annual:perennial), tuft size and grazing value. Perennial grasses give a good indication of ecosystem condition as they form longer-lived obstructions to resource flow than annual vegetation (Ludwig *et al.*, 2000) and degradation leads to their loss (Scholes, 1997; Thrash & Derry, 1999; Parker & Witkowski, 1999). Annual vegetation plays an important role in providing soil cover in the central part of the piosphere where perennial species have been lost (Nangula & Oba, 2004). High intensity grazing can lead to the dominance of smaller grass tufts (Parsons *et al.*, 1997). Large tussock grasses are more sensitive to grazing and therefore will be lost from high intensity grazing areas (McIntyre & Tongway, 2005). Grazing value indicates the value of the grass as a forage plant (van Oudtshoorn, 2004). Grass species were classified as annual, perennial or weakly perennial using van Oudtshoorn (2004) and the abundance of each class used to calculate the annual:perennial ratio. Weak perennials (34% of plant records) were split to contribute half to the perennial and half to the annual occurrence. A lower value (more perennial vegetation) was expected in less degraded sites (McIntyre & Lavorel, 2001). Average tuft size was calculated for each interval from the tuft longest axis measurements. Grazing value for each species was taken from van Oudtshoorn (2004). Unidentified grasses (2% of records) had to be excluded from the annual:perennial ratio and grazing value analyses. High levels of grazing and trampling can lead to a lower density of grasses (Pandey & Singh, 1991; Fernandez-Gimenez & Allen-Diaz, 2001). Herbivore impacts on soil can lead to subsequent losses of herbaceous vegetation (Fernandez-Gimenez & Allen-Diaz, 2001). Grass density, plants/m^2, was calculated by 1 / (average distance)^2 (Causton, 1988).

In areas with large elephant populations, density of trees increases with distance from water (Thrash & Derry, 1999). Other studies have found that high impact levels are associated with bush encroachment, an increase in density of woody vegetation (Moleele & Perkins, 1998; Britz & Ward, 2007). Heavy browsing can lead to decreases in canopy cover (Legget *et al.*, 2003; Dharani *et al.*, 2008) and areas protected from herbivores have much higher canopy cover (Levick *et al.*, 2009). Woody vegetation density was calculated as plants/m^2. Woody vegetation canopy cover was calculated as the percentage of herbaceous grid points which were covered by a woody canopy. Habitat structural variation is important for biodiversity conservation (Noss, 1990; Tews *et al.*, 2004; Ruiz-Jaen & Aide, 2005; Oliver *et al.*, 2007). Height diversity was calculated using Simpson's Diversity Index (D) (Magurran, 1988), and was expected to increase in areas with lower impact (Todd, 2006).

Environmental Variables

Sampling site aspect was recorded during data sampling. Aspect was coded using dummy variables: east, west, north and south. Slope was recorded at each sampling site using a clinometer. Catenal position of each sampling site was recorded during fieldwork and was coded using dummy variables: crest, upper, mid, lower, flat and drainage. Landscapes for KNP were taken from Gertenbach (1983), for the private reserves from Peel *et al*. (2007) and for LNP from Stalmans *et al*. (2004). The private reserve and LNP maps are based on the same classifications as the KNP map (Stalmans *et al.*, 2004; Peel *et al.*, 2007). Landscape type was coded for using dummy variables as shown in Table 1. In order to represent the natural water availability, ephemeral and permanent natural water availability scores for landscape types (Chapter 2) were included in the analysis. Distances of each sampling site to the nearest non-perennial drainage line and perennial river were measured in Arc GIS using the 'point distance' function in ET GeoWizards.

Table 1: Dummy variables for landscape types

Dummy variable	Full Name
LS6	*Combretum/Colophospermum mopane* Woodland of Timbavati
LS22	*Combretum/Colophospermum mopane* Rugged Veld
LS26	*Colophospermum mopane* Shrubveld on Calcrete
LS30	Pumbe Sandveld
LS31	Lebombo North
LS35	*Salvadora angustifolia* Floodplains

Management Variables

Total property size was included in analyses. Fencing was included as a dummy variable: fencing and no fencing. The density of open artificial waterpoints (Chapter 2) was used to represent the current potential stocking density. Type of waterpoint was included as a dummy variable: natural, artificial open or artificial closed. Distance to (a) the nearest permanent waterpoint (artificial open or closed or natural), and (b) the nearest open artificial waterpoint or permanent natural water were measured exactly in Arc GIS using the 'point distance' function in ET GeoWizards.

Factors affecting impact – Data analysis

The influence of environmental and management variables on species composition and degradation variables were assessed using ordinations. Ordination approaches enable simultaneous elucidation of the effects of a range of factors on a range of species or response variables (Leps & Smilauer, 2003). Unconstrained ordinations are used to summarise community patterns with no prior knowledge of the environment (Leps & Smilauer, 2003). Constrained ordinations use measured environmental variables to find the best explanation of the community variation (Leps & Smilauer, 2003). Constrained axes are also known as canonical axes.

Species composition and degradation variables were analysed separately. Data were first subjected to a PCA (Principal Components Analysis) to enable determination of the strength of the best axis of explanation of the variation (Leps & Smilauer, 2003). The degradation variables have different units so linear analysis methods had to be used. After the PCAs, data were subjected to a series of RDAs (Redundancy Analysis). All analyses had no transformation of species scores and were standardised by species because measurement scales differ (Leps & Smilauer, 2003). Monte Carlo Permutation Tests were performed on the first axis alone and on all the canonical axes combined. Rare species were not down-weighted because the grid based method used for herbaceous vegetation sampling is less likely to detect rare species than a quadrat based approach.

RDAs tested the effect of environmental and management variables on species composition and degradation variables. Covariables are used to remove variability caused by underlying factors (Leps & Smilauer, 2003). In this study, environmental variables characterise the template over which the herbivore impact is superimposed, the intrinsic system properties. The effect of management variables on species composition and degradation variables was therefore tested with environmental variables as covariables. The combined effect of environment and management was also tested by assessing the variables simultaneously. This represents the situation of long-term impact of management actions leading to a disruption of the natural intrinsic system properties.

For each model, Marginal Effect Variance Explained was investigated in order to determine the important environmental and management variables. Marginal Effect Variance Explained provides the proportion of variance explained by that variable when it is the only explanatory variable considered. As such, it indicates the explanatory strength of each variable.

Characterisation of properties

In order to characterise the condition of portions of landscapes and/or properties, the link between environmental and management variables and degradation needs to be understood. General Linear Model (GLM) species response curves in CanoDraw were used to test for significant relationships between environmental and management variables and degradation variables. The results of these tests were used to categorise functionality levels of environmental and management variables from low function (a score of 1) to high function (a score of 6). More functional areas were those with higher quality soils and vegetation. With measurement of environmental and degradation variables within or between properties a more realistic conception of functionality can be obtained than by using the piosphere concentric circle approach. The top five most important variables from the overall ranking of Marginal Effect Variance Explained were used to generate a basic characterisation of the properties in the study area.

RESULTS

Importance of position and type of waterpoint

Impact scores varied with distance to water (Figure 1) with a significantly higher impact found closest to water ($F_{(4,\ 99)} = 6.487$, $p<0.05$). Impact scores varied significantly between properties (Figure 1) with the two most intensely managed private reserves having the highest scores and LNP the lowest ($F_{(4,\ 99)} = 3.406$, $p<0.05$). Within properties, impact scores varied between transects (Figure 2). The differences between transects were statistically significant ($F_{(19,77)} = 1.979$, $p<0.05$).

Type of waterpoint influences the impact level around it. Impact scores revealed that open artificial waterpoints had the highest impact (the most degradation), followed by closed artificial waterpoints with natural waterpoints being the least impacted (Figure 1). This analysis pools waterpoint types from all the properties but the difference between artificial and natural waterpoints was still statistically significant ($F_{(2, 101)}$ = 3.377, $p<0.05$). Catenal position of waterpoint had no significant effect on impact score (Figure 1, $F_{(5,98)}$ = 0.501, $p>0.05$).

Landscape factors affecting impact

A total of 218 sampling sites were analysed. There was wide variation in variable scores between sampling sites with standard deviation often being larger than average values (Table 2). PCAs revealed that 4% of the variation in species composition and 27% of the variation in degradation variables is explained by the best possible theoretical predictor (Table 3). RDAs revealed good explanations of patterns of species composition and degradation variables (Table 3). All ordinations were highly significant for the first canonical axis and the combined canonical axes.

Environment was a better predictor of species composition patterns than management with an explanatory power of 13% over all the canonical axes (trace = 0.131, F = 1.779, $p<0.01$). Canonical eigenvalues for management explained only 6% (trace = 0.063, F = 2.033, $p<0.01$). The variance explained by management when environment was accounted for had a lower explanatory power, 1% (trace = 0.048, F = 1.626, $p<0.01$), than the PCA first axis (Table 3). The highest explanatory power, 18% of variance explained, was found when environmental and management variables were combined (trace = 0.180, F = 1.762, $p<0.01$).

For species composition, the strongest correlations with the first canonical axis were Landscape 6 (r = -0.73), ephemeral natural water availability (r = -0.61) and permanent natural water availability (r = 0.61) for the environmental variables. Strongest correlations for management variables were artificial waterpoint density (r = -0.58) and property size (r = 0.53). When environmental and management variables were combined, the strongest correlations with the first canonical axis were Landscape 6 (r = -0.59), artificial waterpoint density (r = -0.55), property size (r = 0.53) and ephemeral natural water availability (r = -0.50).

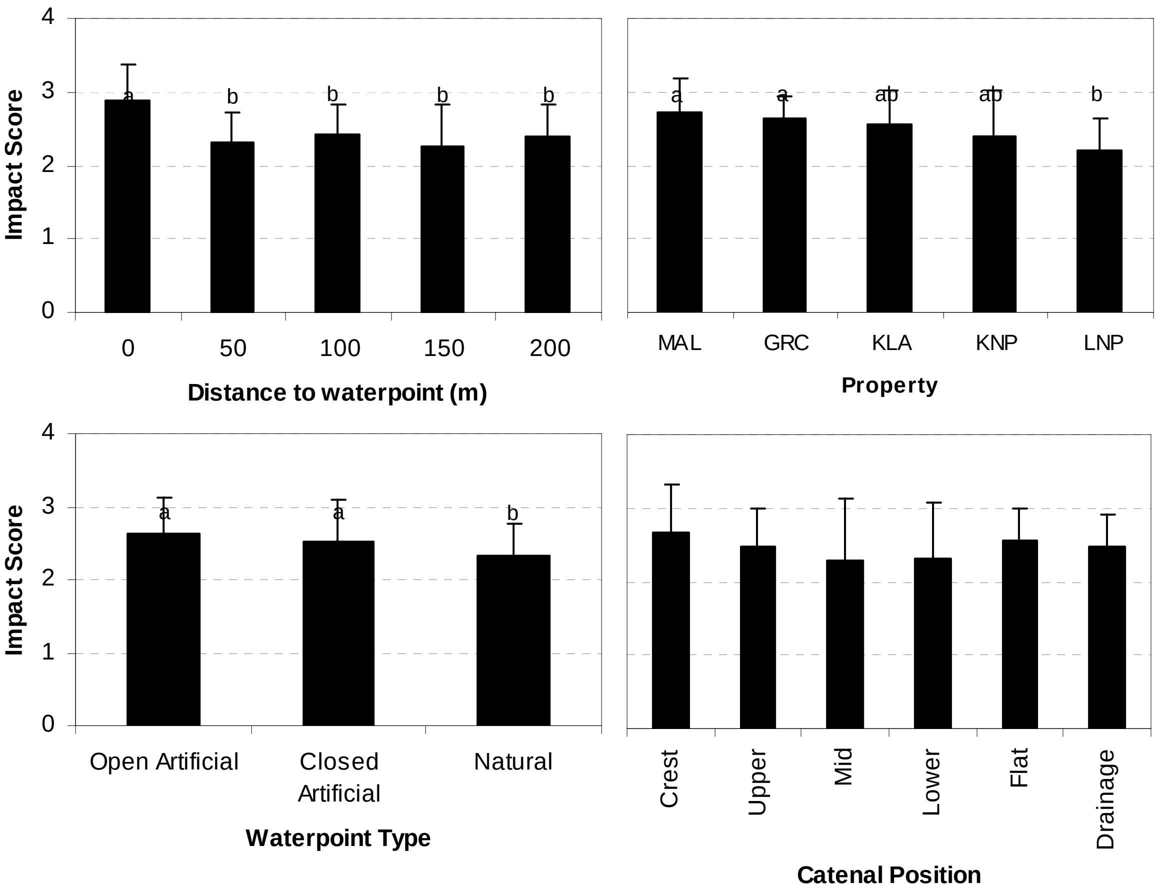

Figure 1: Comparison of impact scores for (a) distance to waterpoint, (b) property (see Table 1 for full names), (c) waterpoint type and (d) catenal position. Bars with different letters per sub-figure are significantly different (Tukey, $p<0.05$).

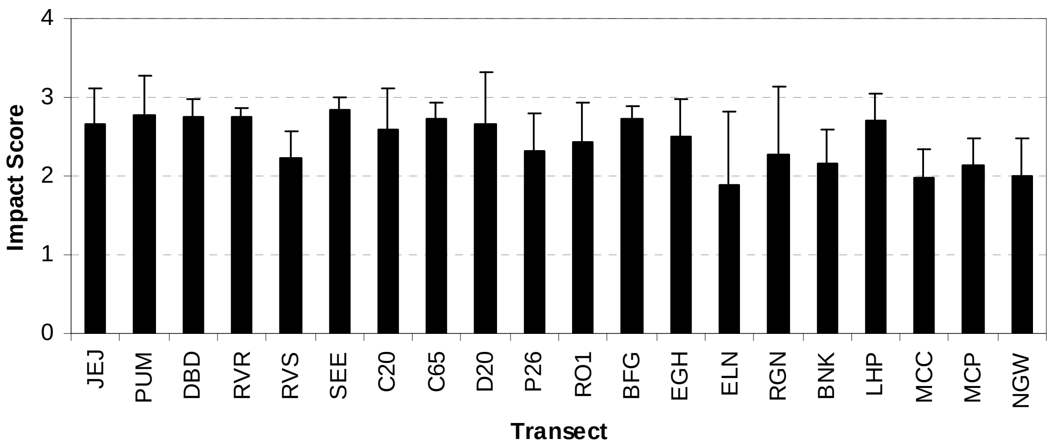

Figure 2: Impact scores for transects on properties (no Tukey's tests were significant although overall ANOVA test was).

Table 2: Descriptive statistics of continuous environmental, management and degradation variables.

Variable		Minimum	Maximum	Average	SD
Bare ground	%	0	44	3	7
Stability index (LFA)		37	104	56	8
Infiltration index (LFA)		13	62	28	8
Nutrient cycling index (LFA)		8	58	22	9
Grazing value score		1	3	2.3	0.5
Annual:Perennial ratio		0	15.2	0.8	1.2
Tuft size	mm	4	66	23	12
Herbaceous plant density	/m^2	15	6944	211	495
Woody plant density	/m^2	0	4.5	0.4	0.5
Canopy cover	%	0	100	36	25
Structural diversity	D	0	1	0.55	0.21
Ephemeral score (Chapter 2)		0.007	0.034	0.027	0.001
Permanent score (Chapter 2)		0.002	0.206	0.032	0.046
Nearest drainage line	m	3	632	161	103
Nearest perennial river	m	0	13584	4271	4105
Distance to open	m	0	10173	1905	2523
Distance to historic	m	0	6979	655	1029
Slope	°	0	4.76	1.08	1.17

Table 3: Eigenvalue results for PCA and RDA testing of the effects of environmental and management variables on species composition and degradation variables.

	Axis 1	Axis 2	Axis 3	Axis 4	All canonical axes
Species Composition					
PCA	0.037	0.029	0.025	0.021	n/a
RDA: Environmental	0.017	0.016	0.011	0.010	0.131
RDA: Management	0.019	0.012	0.009	0.008	0.063
RDA: Management with Environment as a covariable	0.012	0.009	0.007	0.006	0.048
RDA: Management and Environmental combined	0.020	0.017	0.014	0.012	0.180
Degradation Variables					
PCA	0.272	0.170	0.156	0.115	n/a
RDA: Environmental	0.130	0.059	0.032	0.024	0.280
RDA: Management	0.129	0.032	0.020	0.012	0.203
RDA: Management with Environment as a covariable	0.038	0.017	0.014	0.005	0.080
RDA: Management and Environmental combined	0.154	0.069	0.041	0.034	0.360

Environment was a better predictor of degradation variable patterns than management with an explanatory power of 28% for all canonical axes combined (trace = 0.280, F = 4.568, $p<0.01$). Canonical axes for management explained 20% of the variance (trace = 0.203, F = 7.663, $p<0.01$). The variance explained by management when environment was accounted for was 5% (trace = 0.080, F = 3.457, $p<0.01$). The overall best explanation of variation in the degradation variables was the combination of environmental and management factors which explained 36% of the variation (trace = 0.360, F = 4.522, $p<0.01$).

For degradation variables, the strongest correlations with the first canonical axis were Landscape 6 (r = -0.60) and permanent natural water availability (r = 0.50) for environmental variables. Strongest correlations for disturbance variables were artificial waterpoint density (r = -0.55) and property size (r = 0.50). When environmental and management variables were combined, strongest correlations with the first canonical axis were Landscape 6 (r = -0.63) and artificial waterpoint density (r = -0.53).

Marginal Effect Variance Explained in the models varied between tests. Four variables never made a significant contribution to the model creation: southerly aspect, westerly aspect, mid position on the catena and crest position on the catena. Variable groups were spread over the ranking so dummy variables were combined to simplify ranking (Table 4) and determine the effect of environmental and management variables.

Ranking of simplified variables based on their Marginal Effect Variance Explained in the RDAs indicated that for the environmental variables, natural water availability was the most important feature explaining variation between sampling sites (Table 5). Artificial water availability was most important for management variables (Table 5). When management and environmental variables were combined the most important variables explaining variation were natural and artificial water availability, followed by property size (Table 5). Distance to current available water ranked at 5^{th} position and distance to historic available water ranked at 6^{th} position.

When environmental and management variables are combined, the proportional importance in explanation of variability is split about 50% between management and environmental variables (Figure 3). Degradation variables had a large proportion of their variability explained by artificial water availability, property size, natural water availability, landscape type and distance to perennial river. The variability of species was more evenly explained by all environmental and management variables (explanatory variable average importance for species composition = 0.078, SD = 0.019; explanatory variable average importance for degradation variables = 0.077, SD = 0.047).

Table 4: Simplification of dummy variables used in ranking to determine the strength of effects of environmental and management variables.

Simplified variable	Dummy variables included
Artificial water availability	none – single variable
Aspect	North, South, East, West
Catenal position	Crest, Upper, Mid, Lower, Flat, Drainage
Distance to current water	none – single variable
Distance to drainage line	none – single variable
Distance to historical water	none – single variable
Distance to perennial river	none – single variable
Fencing	Present, Absent
Landscape type	LS6, LS22, LS30, LS31, LS35
Natural water availability	Ephemeral score, Permanent score
Property size	none – single variable
Slope	none – single variable
Waterpoint type	Open artificial, Closed artificial, Natural

Table 5: Ranking of importance of environmental and management variables based on strength in Marginal Effect Variance Explained from RDAs of species composition and degradation variables

Variables	Rank
Environmental alone	
Natural Water Availability	1
Distance to Perennial River	2
Landscape Type	2
Slope	3
Catenal Position	4
Aspect	5
Distance to Drainage Line	6
Management alone	
Artificial Water Availability	1
Property Size	2
Waterpoint Type	3
Distance to Current Water	4
Distance to Historical Water	4
Fencing	5
Management and Environmental Combined	
Natural Water Availability	1
Artificial Water Availability	1
Property Size	2
Distance to Perennial River	3
Landscape Type	3
Slope	4
Waterpoint Type	4
Distance to Current Water	5
Distance to Historical Water	6
Catenal Position	7
Aspect	8
Fencing	8
Distance to Drainage Line	9

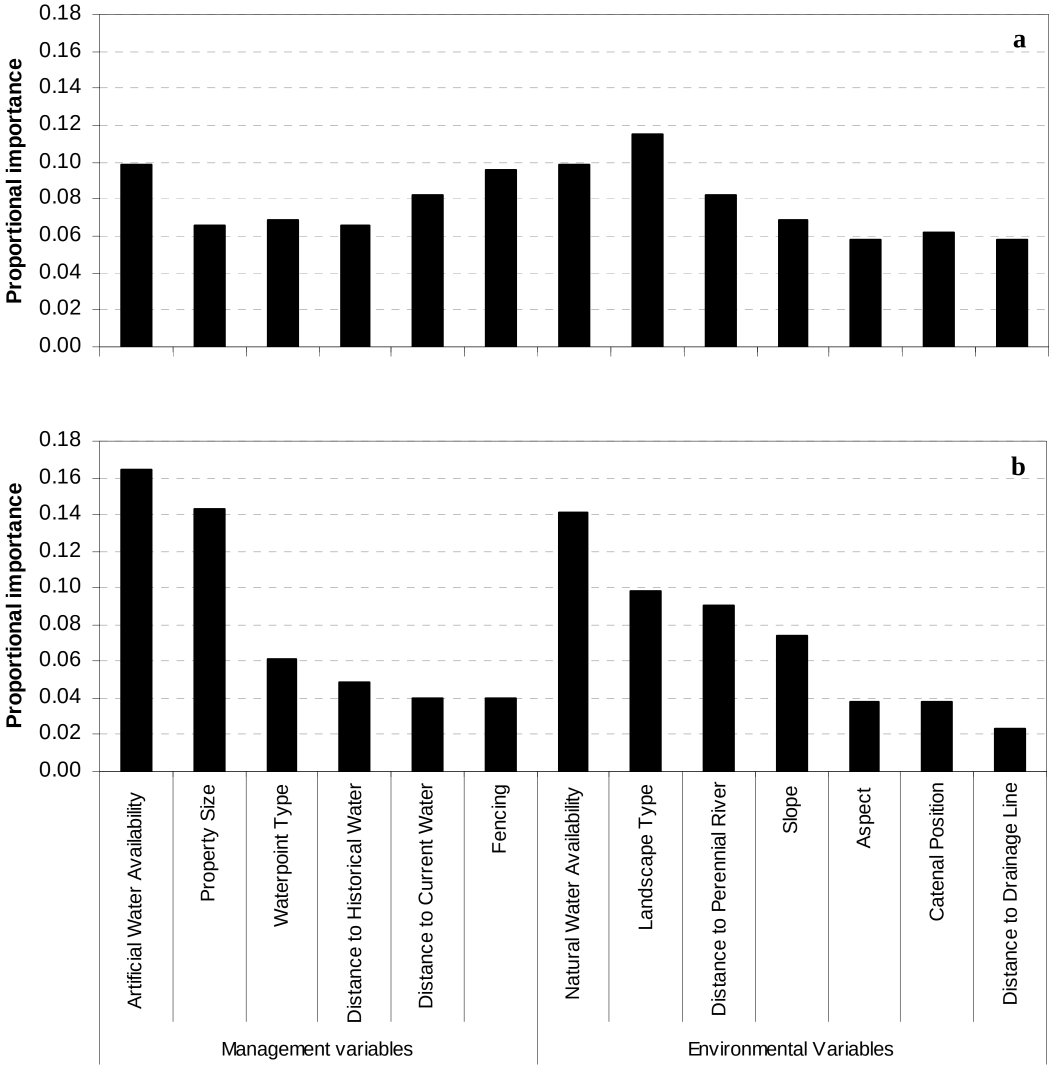

Figure 3: Proportional importance of management and environmental variables in explaining variability between sampling sites for (a) species composition and (b) degradation variables. Proportional importance calculated from Marginal Effect Variance Explained from RDAs of species composition and degradation variables.

Characterisation of properties

Functionality scores were assigned to environmental and management variables as shown in Table 6. Class boundaries were limited to data collected in this study and therefore do not reflect the full potential variation of the southern African savanna. A basic characterisation of the properties was done using the overall top five important variables from the Marginal Effect Variance Explained when management and environmental variables were combined – natural water availability, artificial water availability, property size, distance to a perennial river and landscape type. Scoring of landscape type was limited to those landscape types which were sampled during this study. Functionality scores were calculated by multiplying the proportion of the property in the functionality class by the score for that class.

Based on the top five important variables KNP was the most functional property, followed by LNP and then the private reserves (Figure 4). Functionality scores enable the differentiation between factors that reserve management can and cannot alter. Factors that management can alter includes variables like artificial waterpoint density. Factors that management cannot alter includes variables like landscape type. When functionality is scored using only the factors that management cannot effect, it is found that the environmental situation of the private reserves means that their base functionality is lower than the national parks (Figure 4). When looking at the scores in more detail, GRC and KLA both have higher functionality scores (4.29 and 5.22 respectively) than KNP and LNP (3.9 and 2.35 respectively) when based only on distance to a perennial river.

Table 6: Functionality score classes for environmental and management variables. Values relate to data gathered only in this study.

Functionality Score		**1 (low)**	**2**	**3**	**4**	**5**	**6 (high)**
Landscape Type		6	22	26	31	35	30
Catenal position		Mid	Flat	Upper	Drainage	Crest	Lower
Distance to historical	m	0 - 1163	1163 - 2326	2326 - 3489	3489 - 4652	4652 - 5815	5815 - 6979
Distance to current	m	0 - 1696	1696 - 3392	3392 - 5088	5088 - 6784	6784 - 8480	8480 - 10173
Property size	km^2	48 - 3199	3199 - 6350	6350 - 9501	9501 - 12652	12652 - 15803	15803 - 18956
Waterpoint density	points/km^2	0.297 - 0.357	0.237 - 0.297	0.177 - 0.237	0.117 - 0.177	0.057 - 0.117	0 - 0.057
Distance to non-perennial	m	3 - 108	108 - 213	213 - 318	318 - 423	423 - 528	528 - 632
Distance to perennial	m	11320 - 13584	9056 - 11320	6792 - 9056	4528 - 6792	2264 - 4528	0 - 2264
Slope	°	3.97 - 4.76	3.18 - 3.97	2.39 - 3.18	1.6 - 2.39	0.81 - 1.6	0 - 0.81
Permanent natural availability score		0.002 - 0.036	0.036 - 0.07	0.07 - 0.104	0.104 - 0.138	0.138 - 0.172	0.172 - 0.206
Ephemeral natural availability score		0.03 - 0.034	0.026 - 0.03	0.022 - 0.026	0.018 - 0.022	0.014 - 0.018	0.007 - 0.014

Functionality Score	**1**	**2.67**	**4.34**	**6**
Aspect	North	South	West	East

Functionality Score	**1**	**3.5**	**6**
Waterpoint type	Open	Closed	Natural

Functionality Score	**1**	**6**
Fencing	Present	Absent

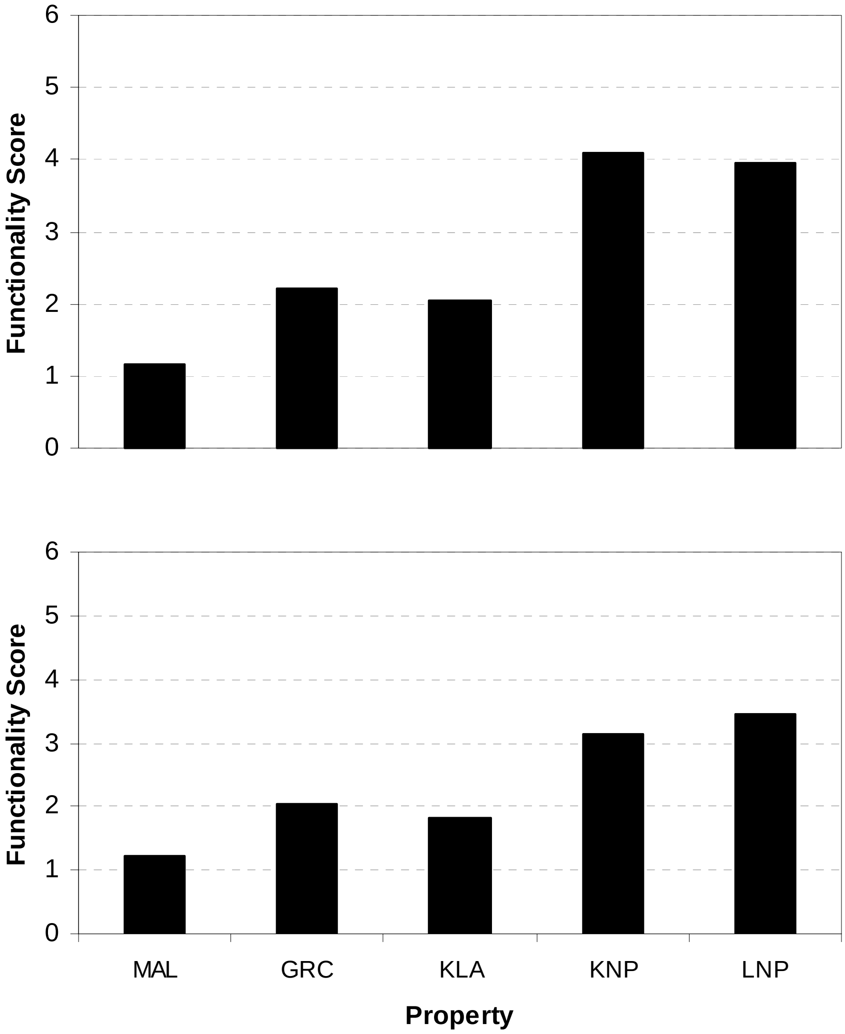

Figure 4: Functional characterisation of properties of the study area based on (a) the top five most important, and (b) variables that management cannot affect from the top five most important, from Marginal Effect Variance Explained from RDAs of species composition and degradation variables.

DISCUSSION

Understanding the impact of water supplementation is important for conservation management in the southern African savannas. Installation of permanent artificial waterpoints for herbivores is a contentious issue as it can lead to degrading impacts on soils and vegetation (Thrash & Derry, 1999; Parker & Witkowski, 1999; James *et al.*, 1999). To date, the impacts of artificial water supplementation have been understood in terms of concentric circular patterns, the piosphere, focused on waterpoints (Owen-Smith, 1996; Thrash, 2000; Gaylard *et al.*, 2003). It has been shown that this approach is too simplistic for use in the heterogeneous southern African savannas (Chapter 4). Position and type of waterpoint can affect the surrounding impact level and both environmental and management variables have important impacts on ecosystem function.

Higher levels of herbivore impact are associated with higher levels of degradation and a loss of ecosystem function (McIntyre & Tongway, 2005; Levick *et al.*, 2009). Water provision was again highlighted as a potentially problematic management action as artificial waterpoints were shown to be more degraded than natural waterpoints, with no significant differences between open and closed artificial waterpoints. This is in agreement with other work which has found no effect of closure on reducing degradation (K. Matchett, PhD submitted 2010). At broad scales, more intensely managed properties (Chapter 2) had higher impact levels. At finer scales, the highest levels of impact were restricted to the immediate surroundings of the waterpoints. This has previously been termed the sacrifice zone (van der Schijff, 1959; Graetz & Ludwig, 1978; Thrash & Derry, 1999). It is important that the aesthetics of the immediate waterpoint surroundings are not confused with impacts that the waterpoint has across the property.

When considered alone, environmental variables explained a greater degree of variation in species composition and degradation variables than management variables. This highlights the importance of the landscape template (Scoones, 1995; Augustine, 2003) in influencing resilience of an area. The landscape template affects how animals move (Coughenour, 1991; Bailey *et al.*, 1996) and how vegetation responds to herbivores (Milchunas *et al.*, 1988; Prins & van der Jeugd, 1992; Mushove *et al.*, 1995). The best explanation of species composition and degradation variables was a combination of environmental and management variables. Under this scenario, environmental and management variables had approximately equal importance. The piosphere effect was again shown as non-dominant, distance to current waterpoint and distance to historic waterpoint ranked at 5th and 6th positions.

As water provision is associated with an increased degradation risk it is essential that management is based on sound ecological theory (Chapter 2). As our ecological understanding of the southern African savannas is increasing e.g. (du Toit *et al.*, 2003;

Craine *et al.*, 2009) it is important to make concomitant advances in management (Friedel, 1991; Roux *et al.*, 2006). In order to cover the important factors in determining function in an area, a combination of environmental and management variables is needed. This will enable a better understanding of the ecological impacts of water supplementation across properties. Moving away from the piosphere patterning of herbivore impact towards an understanding which incorporates the influence of the heterogeneity of the landscape template will enable a more realistic estimation of property resilience.

One of the management variables highlighted as important was property size. This gives an indication of the scope of the property for natural management (Baker, 1992; Owen-Smith, 1996; Peel *et al.*, 1999). When looking at characterisation of property functionality, smaller properties were found to be less functional. In smaller properties, management actions are intensified (Baker, 1992; Peel *et al.*, 1999). The ecosystem becomes compressed and more highly managed which can lead to a loss of resilience (Owen-Smith, 1996; Holling & Meffe, 1996). Fencing further decreases the functionality of these properties as it creates an unnatural boundary in the ecosystem (Forman & Godron, 1981). As expected, fenced properties were less functional than unfenced properties.

The level of artificial supplementation was also an important management variable with areas with a higher level of supplementation being less functional. Stabilising water availability has two major effects on herbivore populations, (1) an alteration of movement and utilisation patterns (Owen-Smith, 1996; Chamaillé-Jammes *et al.*, 2007), and (2) release of the natural limiting factor and therefore an increase in population size (Cronje *et al.*, 2005; Chamaillé-Jammes *et al.*, 2007). Alteration of movement and utilisation patterns results in herbivores using areas of the landscape that they would not have access to under a natural water distribution. Additionally, the pressure they exert in these areas is sustained for a longer time period. It has been stated that after a certain level of supplementation, increases in water availability no longer alter the condition of vegetation and soils (Thrash, 2000). Herbivore populations reach such a size that they are controlled by forage availability rather than water availability (Walker *et al.*, 1987). It is therefore important to understand the multi-scale effects of water supplementation.

The difference in impact between artificial and natural waterpoints should be highlighted. Artificial waterpoints, whether open or closed, have a higher impact level than natural waterpoints. Associated with this, areas around open waterpoints were found to be the most degraded whilst areas around natural waterpoints were the least degraded. Supplementation of permanent waterpoints creates degradation nodes and increases the general degradation level across the property. This multi-level impact on resilience should be a cause for concern for property management. Resilience is an important aspect of conservation management and the sustainability of property management approaches (Holling & Meffe, 1996; Gunderson, 2000; du Toit *et al.*, 2003).

Natural water availability has a strong influence on the baseline functionality of a property. An overall pattern can be found between the results from the natural water availability and the distance to natural water sources: where there is unreliable water, the resilience of the soils and vegetation to sustained herbivore impact is lower. Areas with high permanent natural water availability (Chapter 2) and/or closer to a perennial river have a higher functionality. Areas with a high ephemeral natural water availability (Chapter 2) and/or closer to a non-perennial drainage line have a lower functionality. Under natural conditions, areas of the landscape with permanent water would have received year-round pressure from herbivores. Areas of the landscape with ephemeral water would have received short periods of pressure during the wet season only. This patterning of water availability and herbivore use of vegetation has occurred over evolutionary time scales (Derry & Dougill, 2008; Bouchenak-Khelladi *et al.*, 2009). Changing the herbivore pressure in areas by providing artificial water therefore leads to unusual pressures vegetation and soil.

Resilience of soil and vegetation is also affected by other factors. Catenal processes have a strong influence on vegetation (Witkowski & O'Connor, 1996; Venter *et al.*, 2003) so catenal position of a sampling site should be important. Granitic landscapes in the study area are characterised by strong catenas with shallow, sandy/gravely and low nutrient soil with unpalatable vegetation on crests and deep, clayey soil with moderate nutrients and palatable vegetation on bottomlands (Witkowski & O'Connor, 1996; Venter *et al.*, 2003). It was therefore expected that functionality would be higher nearer to the drainage line. However, no regular pattern of change in function along the catena was found.

Aspect and slope contribute to the solar radiation that vegetation receives, with a higher level of radiation resulting in a higher growth rate (Schulze, 1997). Greater growth rates can increase vegetation resilience to herbivory (Skarpe *et al*., 2000; Holdo, 2006). On gentle slopes, north facing areas have low radiation in the winter months (Schulze, 1997). The low function found on north facing slopes could be a result of the combination of low radiation and low water availability as these would decrease vegetation resilience to herbivory. Steepness of slope also contributes to the resilience of the ecosystem as it affects the speed with which nutrients are lost from the system (Tongway & Hindley, 2004). A steeper slope leads to faster water flows and this results in a greater loss of soil and nutrients from the system (Tongway & Hindley, 2004; Ludwig *et al*., 2005). Areas with gentler slopes were found to be more functional. This could by caused by the slower movement of resources through this area leading to a greater uptake by vegetation (Ludwig *et al*., 2005). A greater cover of vegetation and slower movement of surface water will also reduce the erosion of soil from the area (Tongway & Hindley, 2004).

Functionality scores for management and environmental variables were used to generate property scores for functionality. This is an alternative to understanding properties simply in terms of their artificial waterpoint density. However, it is also important to understand the variation in scales over which the variables are changing. For example, artificial waterpoint density is taken as a constant over the property whilst distance to drainage line has a maximum value of 632m. The variation in scale at which the different management and environmental variables affect function is important in the determination of property resilience. Calculating a single score over a large landscape area does not indicate the variability in the resilience across the property. Spatial heterogeneity has important impacts on resilience (Eriksson, 1996; Nystrom & Folke, 2001; Suding *et al*., 2004) so it is important to understand how spatial heterogeneity varies within properties. As our understanding of the influence of the landscape template and the scales at which it varies improves, the approach to understanding property resilience presented here can be further developed.

An important factor to note is the variation in property baseline potential function. This refers to environmental variables which management cannot alter, for example landscape type or distance to a perennial river. These variables have important implications for the response of the ecosystem to management actions. Supplementation of permanent water will have less of an effect in KNP or LNP where there is a naturally higher level of permanent water availability than in the private reserves where there is a high level of ephemeral water availability. Previous studies have raised concerns about the high level of herbivore impact in the private reserves and its subsequent effect on system resilience (Walker *et al.*, 1987; Craine *et al.*, 2009) and this study found that the national parks were more functional than the private reserves. It is important, however, that discussions comparing properties take into account the underlying environmental constraints to resilience.

Analysis of property functionality and the impact of management and environmental variables was limited to their effects on degradation variables, effects on species composition were not considered in this study. Analysis of species composition needs to be more detailed before it can be used to characterise landscapes and properties. A sensitivity analysis is needed to determine how each species responds to utilisation by herbivores within a heterogeneous environment. Work in the Karoo has shown that a simple Increaser/Decreaser approach is not applicable to vegetation along distance from water transects (Todd, 2006). The Increaser/Decreaser approach does not take into account the effects of factors such as the interactions between species and therefore is not straightforwardly applicable in a heterogeneous environment. Further intensive study of the data collected here would be required to determine species patterns, their general applicability and their sensitivity to change before species data are used for characterisation or monitoring.

CONCLUSION

The supplementation of permanent water sources for herbivores is associated with a high degradation risk and detrimental impacts on property resilience which can lead to jeopardising of sustainable management of conservation properties. It is essential that management therefore have a sound ecological basis for understanding the impacts of water provision. The piosphere model currently used in management is an over-simplification of the system as it does not take into account heterogeneity in the landscape template. Environmental and management factors both contribute to the functionality of an area and therefore both need to be considered in assessment of the impact of artificial water supplementation.

Our understanding of the landscape is continuously improving and heterogeneity is now an important factor in many aspects of conservation management and ecosystem understanding (du Toit *et al.*, 2003). However, the incorporation of spatial heterogeneity into the understanding of the impacts of waterpoints has been long delayed and is only recently becoming highlighted as important (Chamaillé-Jammes *et al.*, 2009). There is an urgent need to move away from the piosphere approach to understanding the impacts of water provision. Whilst the functionality assessment presented in this paper is relatively basic, it highlights the importance of factors other than waterpoint density and distance between waterpoints. There is a wide range of studies being performed in KNP and understanding of the ecosystem is continually improving. It is likely that there is data in existence that could be used to characterise a much greater proportion of the landscape variation and understand the link this has with functionality. This would enable the approach to become a working approach and to be continuously updated as the understanding of the ecosystem increases.

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Chapter 6

Synthesis: heterogeneity, management objectives and understanding waterpoints

Helen Farmer

ABSTRACT

Artificial water supplementation has impacts on soils, vegetation, herbivore populations and ecosystem resilience. This paper addresses two important questions with regards to understanding the ecological impact of water supplementation: (1) how does spatial heterogeneity affect impact patterns around waterpoints and what are the implications of this for landscape resilience, and (2) how could an understanding of spatial heterogeneity and ecosystem resilience be used to improve conservation management with regards to waterpoints? Water management is currently based on piospheres, an approach developed in homogeneous systems in Australia. Heterogeneity of the southern African savannas led to questioning of the applicability of a piosphere based approach to management. The piosphere model was tested over 22 waterpoints on five properties with differing conservation management approaches. Data were collected on herbaceous vegetation, woody vegetation and soil functionality along transects extending from waterpoints. Variables from published studies were tested for a relationship with distance to water using linear regression, logistic regression and ordinations. Out of 782 tests, only 109 (14%) showed significant relationships leading to the rejection of the piosphere model as a basis for understanding and managing impact around waterpoints in the southern African savanna. Data were then investigated to begin development of an alternative approach to understanding the effect of water supplementation on resilience of properties. Using ordination methods, it was found that environmental variables were stronger than management variables. The best explanation of species composition and degradation variable variation was a combination of management and environmental variables. The new approach highlights the importance of the baseline potential functionality of properties. A more complex approach to understanding the impact of water supplementation on ecological resilience of properties is required.

KEY WORDS

Conservation; grazing gradients; herbivore impact; piospheres; resilience; savanna

INTRODUCTION

Water provision is an important aspect of conservation management in southern African savannas. When properties were fenced to create private and government conservation areas, migration routes were broken and waterpoints had to be installed for water-dependent herbivores (Walker *et al.*, 1987; Mabunda *et al.*, 2003). Large numbers of waterpoints were installed to buffer herbivore populations against drought (Walker *et al.*, 1987) and to ensure full use of property resources (Aucamp *et al.*, 1992). Subsequent research found detrimental long-term effects of water supplementation on vegetation (Thrash, 1998a; Thrash & Derry, 1999; Parker & Witkowski, 1999; Thrash, 2000) and herbivore populations (Walker *et al.*, 1987; Owen-Smith, 1996).

In order to manage the impact of water supplementation, an approach to understanding the links between water supplementation, herbivore utilisation and the degradation of vegetation and soils was needed. When conservation managers began dealing with these problems, the only available theory was agricultural in origin (Aucamp *et al.*, 1992; Mabunda *et al.*, 2003). The piosphere model, developed in Australia, offered an efficient way to understand the interaction between water supplementation and herbivore impact (Lange, 1969; Thrash & Derry, 1999; Adler & Hall, 2005). A piosphere is an ecological and management unit describing the utilisation of vegetation by water-dependent herbivores (Lange, 1969). Piospheres are characterised by a logistic equation (Figure 1) that links herbivore impact with distance from water (Graetz & Ludwig, 1978).

Piospheres are based on an assumption of soil and vegetation homogeneity where the forage-water trade-off is the only factor affecting herbivore movement (Graetz & Ludwig, 1978; Adler & Hall, 2005). Homogeneity implies a lack of environmental or topographical influences on herbivore movement, for example as is found in the Australian arid zone (Andrew, 1988). In contrast, southern African savannas are characterised by high levels of heterogeneity at multiple scales (Skarpe, 1992; Pickett *et al.*, 2003; Sankaran *et al.*, 2004). This heterogeneity occurs within the piosphere ecological unit; the water-forage trade-off is not the only factor affecting herbivore movement. Due to the high level of heterogeneity, this study investigated the applicability of the piosphere model in southern African savannas (Chapter 4).

Investigation into the general applicability of the piosphere model in the heterogeneous southern African savannas led to rejection of piospheres as a suitable basis for conservation management (Chapter 4). The high level of environmental heterogeneity led to a lack of consistency between variables and between waterpoints in exhibiting the piosphere effect. Data gathered were therefore used to develop a new approach to understanding waterpoint impact in heterogeneous savanna conservation areas that takes spatial heterogeneity into account (Chapter 5). This included differentiation of environmental variables and management variables and investigation of their impact on degradation variables and species composition at various points across the landscape. It was found that environmental variables have a stronger impact on species composition and degradation than managment variables (Chapter 5). Because of the extensive management history in the area (Peel *et al.*, 2005), the best explanation of species and degradation variability was a combination of disturbance and environmental variables (Chapter 5).

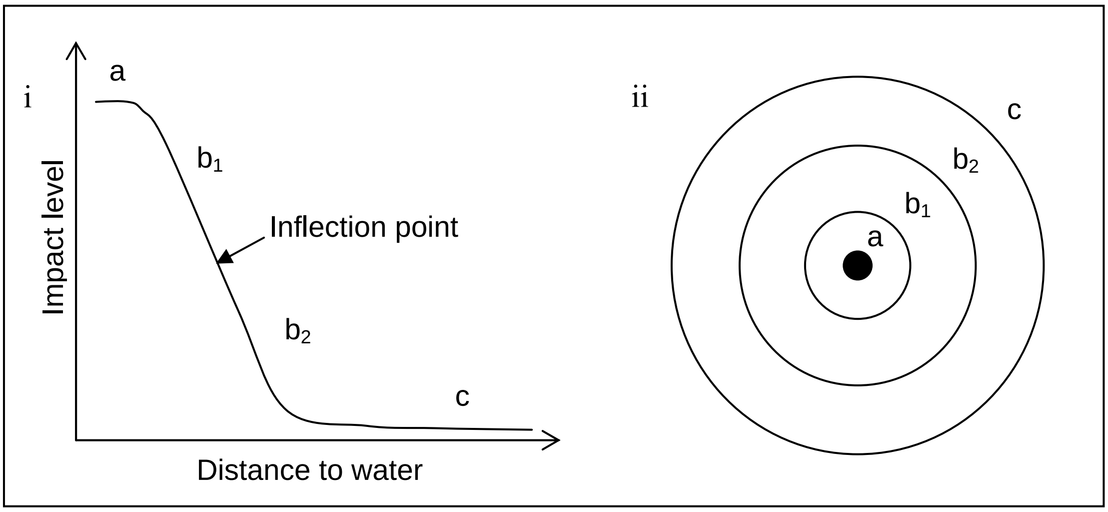

Figure 1: (i) The logistic curve of Graetz & Ludwig (1978) used to describe the zones of a piosphere, and **(ii)** the piosphere as concentric rings of different impact levels around a waterpoint (black circle). Zones are labelled as (a) sacrifice zone, poor condition, (b_1) changing impact, fair condition, (b_2) changing impact, good condition, and (c) very little impact, excellent condition.

This synthesis chapter assesses the findings from the previous five chapters and deals with two important general questions addressed in this study. (1) How does spatial heterogeneity affect the impact pattern around waterpoints and what are the implications of this for landscape resilience? (2) How could an understanding of spatial heterogeneity and ecosystem resilience be used to improve conservation management with regards to waterpoints?

WATERPOINT MANAGEMENT HISTORY AND THEORY

Our understanding of waterpoints has changed over the last 60 years (Figure 2). The first key study was Lange (1969) with the development of the piosphere model for understanding the impact around and managing waterpoints. Graetz & Ludwig (1978) developed an operational approach for investigating range condition based on piospheres. At the end of the 1980s, two key papers set the stage for waterpoint studies over the next 20 years. Andrew (1988) published a piosphere review and Pickup & Chewings (1988) reported a study on modelling grazing and cattle distribution in large paddocks. These two studies led to the distinction between piospheres and grazing gradients with piosphere studies often implying a concentric circle pattern (Tolsma *et al.*, 1987; Adler & Hall, 2005) and grazing gradient studies tending to incorporate the importance of heterogeneity (Bastin *et al.*, 1993; Pickup & Bastin, 1997). Acknowledgement of homogeneity during sampling increased in the 2000s leading to a greater acceptance of grazing gradients and piospheres (Chapter 1).

Current conservation management is moving towards landscape conservation with explicit recognition of ecosystem processes and the importance of spatial heterogeneity and resilience (Baker, 1992; Holling & Meffe, 1996; Rogers, 2003; Boyd *et al.*, 2008). Spatial heterogeneity has been recognised as important for the conservation of biodiversity (Fuhlendorf & Engle, 2001) and the sustainability of ecosystem processes (Elmqvist *et al.*, 2003; Tylianakis *et al.*, 2008). Spatial heterogeneity and ecosystem resilience are also tightly linked (Forman & Godron, 1981; Eriksson, 1996; Suding *et al.*, 2004; Chapter 3). With conservation management objectives taking explicit cognisance of spatial heterogeneity, it is important that our understanding and management of ecosystems are in line with spatial heterogeneity. Models such as Owen-Smith (1996) do not include the influence of spatial heterogeneity on herbivore movement and impact patterns.

Due to the basis of piospheres on homogeneity (Lange, 1969; Graetz & Ludwig, 1978), their application in the southern African savannas was theoretically questionable. Averaging values into zones surrounding waterpoints removes heterogeneity which is of ecological and management importance (Chamaillé-Jammes *et al.*, 2009). Southern African savannas are highly heterogeneous at multiple scales (Skarpe, 1992; Pickett *et al.*, 2003). At broad scales, heterogeneity arises from geology and climate, and at finer scales from fire and herbivory (Scholes, 1990; Skarpe, 1992; Venter *et al.*, 2003; Wiegand *et al.*, 2006). Interactions between top-down and bottom-up drivers result in spatial patterning (Bestelmeyer *et al.*, 2006). Soil nutrients are heterogeneous both spatially and temporally (Scholes, 1990; Pärtel & Helm, 2007) and subtle differences in soil can cause vegetation patterns (Ben-Shahar, 1991; Bestelmeyer *et al.*, 2006).

The understanding of spatial heterogeneity and herbivore impact surrounding waterpoints has implications for the understanding of ecosystem resilience across properties. Ecosystem resilience refers to the ability of a system to withstand the pressures of disturbance and maintain the state it is in (Holling, 1973; Holling & Meffe, 1996; Gunderson, 2000). Areas that are under high disturbance pressure, for example areas immediately surrounding waterpoints, are likely to be resilient in a degraded state. Areas that are far from water, and therefore outside the regular impact zone of herbivores, are likely to be resilient in a non-degraded, highly functional state. Understanding how spatial heterogeneity affects the impact patterns around waterpoints is therefore important when estimating property resilience.

The ecological resilience of a property is important when considering conservation across broad landscapes as resilience of the landscape as a whole is dependent on resilience of its constituent parts (Carpenter *et al.*, 2001; Cumming *et al.*, 2005). This is important when a property is considering its density of waterpoints and when transboundary conservation areas are considering their sustainability. Waterpoint densities are relatively high on privately owned nature reserves (Chapter 2). Questions have been raised about the sustainability of the intensity of management on these properties (Craine *et al.*, 2009). To understand in more detail how spatial heterogeneity and water provision affect resilience of the property is of key importance to these properties.

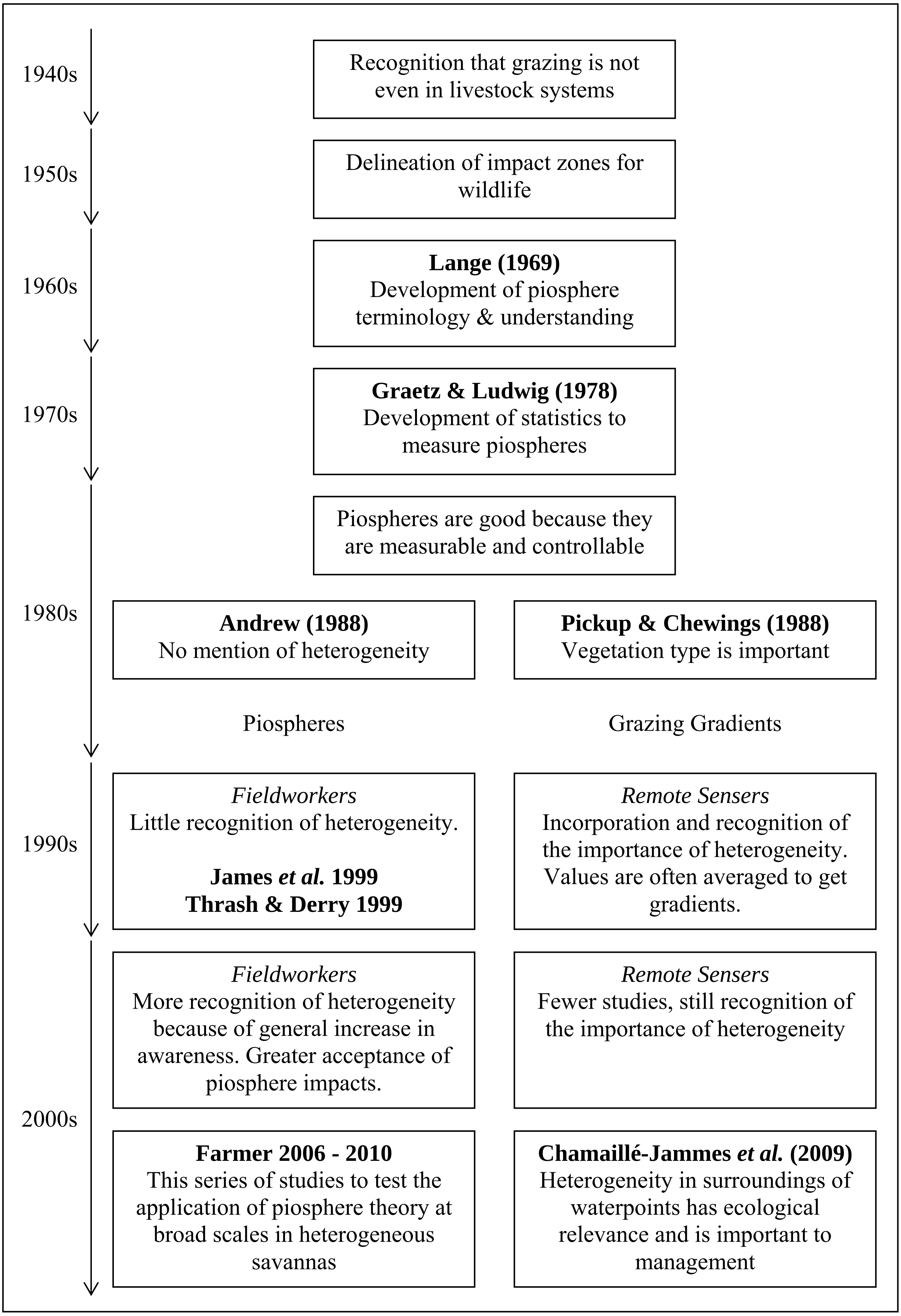

Figure 2: Timeline of the development of understanding regarding the impacts of water supplementation. Key studies highlighted in bold.

TESTING THE PIOSPHERE MODEL

If the piosphere model is generally applicable then the distance to water relationship should hold between waterpoints and across variables. Considerable work has been done in Australia specifically to determine the applicability of piospheres (Graetz & Ludwig, 1978; Andrew, 1988; Pickup & Chewings, 1988; James *et al.*, 1999; Landsberg *et al.*, 2003) but this has not been done in the southern African savannas. Studies in the southern African savannas have been focused on relatively small, homogeneous sections of the landscape (Thrash, 1997, 1998a,b) but their results have been scaled up to property management (Thrash, 2000). It needed to be determined whether such an approach is applicable across these heterogeneous landscapes.

Previous piosphere studies in the southern African savannas have found that basal cover of grasses (Thrash *et al.*, 1991a), woody vegetation density (Brits *et al.*, 2002) and herbaceous species composition (Parker & Witkowski, 1999) respond to distance from water gradients as expected from piosphere theory. Eleven variables from studies conducted in the same area were tested, along with three variables from other piosphere studies. Soil variables considered were cover of bare ground, infiltration, stability and nutrient cycling indices; herbaceous vegetation variables considered were species composition, tuft size and plant density; woody vegetation variables considered were species composition, species richness, proportion in different height classes, total plant density, density of different height classes and structural variation.

The study was performed over five properties with differing management approaches and intensities. Following previous studies, a single transect was sampled from each waterpoint (Lange, 1969; Thrash, 1998a; Brits *et al.*, 2002). Transect length was determined by maximum distance within the zone of influence of a waterpoint (Parker & Witkowski, 1999; Ryan & Getz, 2005) and sampling time available. Transect length varied from 150m to 7km with more shorter transects (Figure 3). Interval sampling was used to increase efficiency with shorter intervals closer to the waterpoints in order to resolve the rapid utilisation changes predicted in this zone (Graetz & Ludwig, 1978). Sampling was performed during summer and autumn when herbivore impact was widely spread (Redfern *et al.*, 2003, 2005; Ryan & Getz, 2005) but long-term patterns of utilisation were still visible (Adler & Hall, 2005). Fieldwork was restricted to one season to avoid the effects of inter-annual rainfall variability (Schulze, 1997).

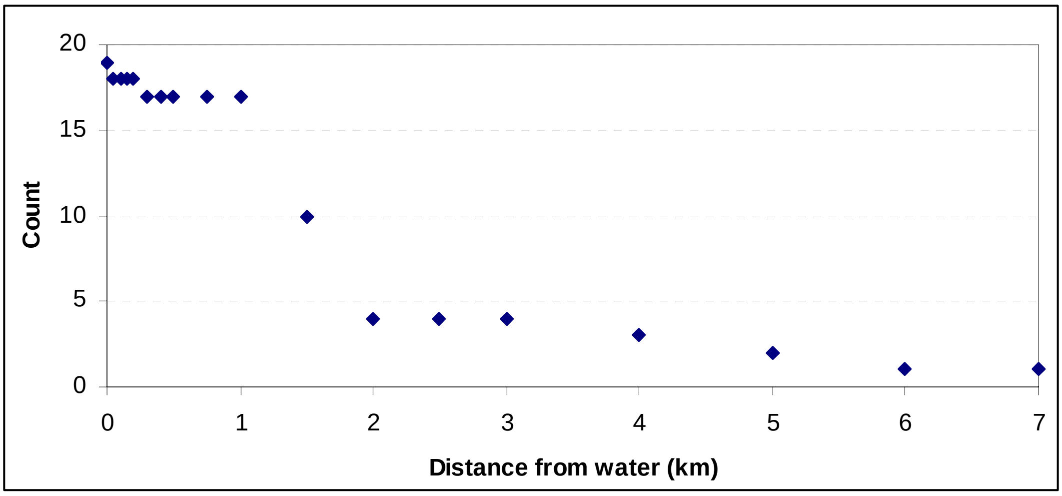

Figure 3: Frequency of sampling distances at different distances from water

Like previous studies, data were analysed through linear regression (Thrash *et al.*, 1991a; Riginos & Hoffman, 2003), logistic regression (Thrash *et al.*, 1991a; Thrash, 1997, 1998a,b; Brits *et al.*, 2002; Riginos & Hoffman, 2003) and ordination (Fernandez-Gimenez & Allen-Diaz, 2001; Heshmatti *et al.*, 2002). Waterpoint transects were each tested separately for significant relationships between variables and distance to water. Not every waterpoint could be tested for every variable due to sample size and data constraints. A total of 358 linear regressions, 358 logistic regressions, and 66 ordinations were performed.

Of the total 782 statistical tests of the effect of distance to water on the herbaceous vegetation, woody vegetation and soil variables, only 109 (14%) showed significant relationships. Logistic regressions (20% of tests significant) performed better than linear regressions (11% of tests significant). No variable or transect had significant regression results for more than 50% of it's transects/variables (Figure 4). Only one ordination (herbaceous vegetation at a natural waterpoint in a national park with very low management intensity) was significant. The very low explanatory power of the piosphere model led to the recommendation that it be rejected as a basis for managing water provision in the heterogeneous southern African savanna.

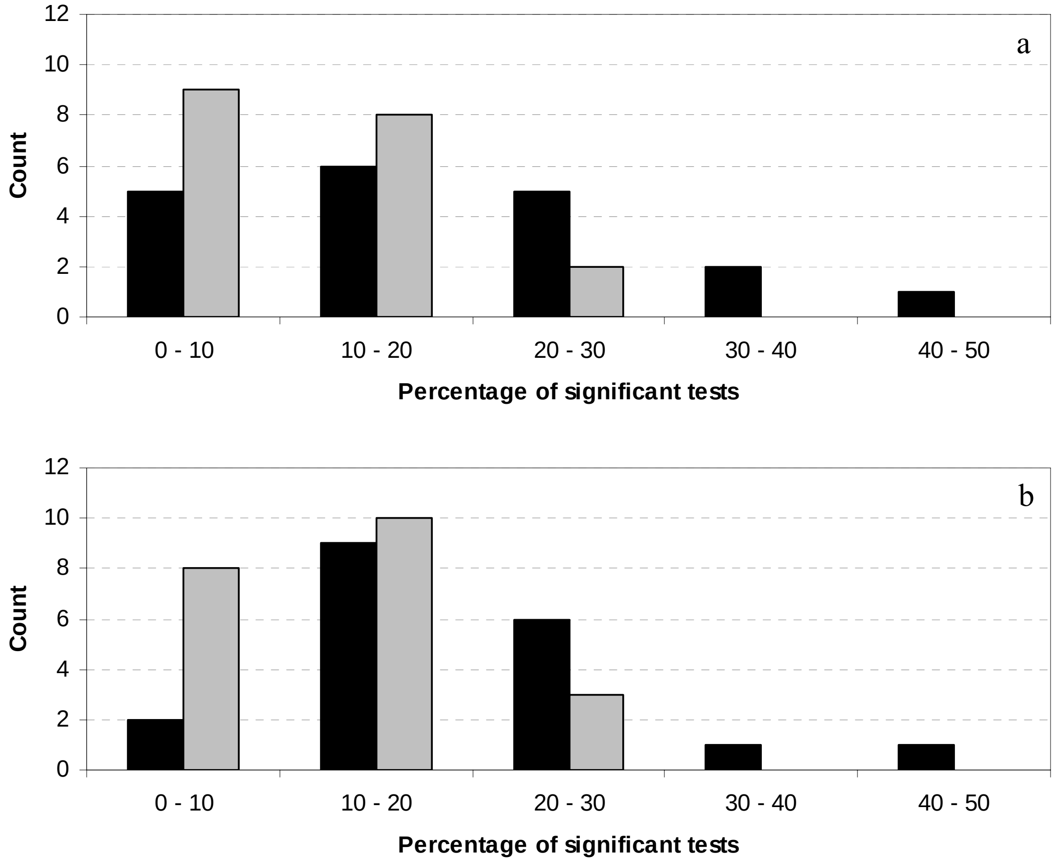

Figure 4: Percentage of significant relationships with distance to water for (a) transects and (b) variables. Results of linear regression tests indicated by grey bars. Results of logistic regression tests indicated by black bars.

The low proportion of significant relationships between variables and distance to water is likely to be a direct or indirect result of heterogeneity. Savannas are heterogeneous at multiple scales (Pickett *et al.*, 2003) creating a range of mosaics (Law & Dickman, 1998; Bowyer & Kie, 2006) which foraging herbivores respond to (Arditi & Dacorogna, 1988; Bailey *et al.*, 1996). This means that the forage-water trade-off is not the dominant factor influencing herbivore utilisation, as is required for the piosphere model (Lange, 1969; Graetz & Ludwig, 1978). Previous studies from other areas have found disruptive effects of heterogeneity. Vegetation type (Hanan *et al.*, 1991; van Rooyen *et al.*, 1994; Nangula & Oba, 2004), soil type (Nash *et al.*, 1999; Turner, 1999) and rainfall (Hanan *et al.*, 1991; Turner, 1999) have all been found to disrupt the detection of grazing gradients. Environmental factors have been found to be more important than distance to water when determining species composition (Makhabu *et al.*, 2002).

Previous studies in the same area as this study have concluded in favour of the piosphere effect (Thrash *et al.*, 1991a; Thrash, 1997, 1998a; Parker & Witkowski, 1999; Brits *et al.*, 2002). However, these studies were all performed in specifically selected homogeneous sections of the landscape. The lack of consistency of the piosphere effect when heterogeneity is included in sampling shows that the results from studies in homogeneous areas should not be generalised between waterpoints or be scaled up to property management levels.

The major implication of this study for understanding resilience of southern African savanna conservation areas is that resilience is not linked simply to distance to water. The use of concentric circular patterns to understand herbivore impact and therefore degradation and resilience is not appropriate. An alternative approach to understanding the spatial patterns of degradation and resilience is required.

DEVELOPMENT OF A NEW APPROACH

With the rejection of the piosphere model as a basis for understanding and managing impact around waterpoints in the heterogeneous southern African savanna, an alternative approach needs to be developed. The new approach has to take into account spatial heterogeneity as this is important in ecological resilience. Conservation management is increasingly recognising the importance of heterogeneity (Rogers, 2003) so this new approach will also bring conservation management more closely in line with ecological theory (Friedel, 1991; Christensen *et al.*, 1996). Factors such as vegetation heterogeneity (Nangula & Oba, 2004), soil differences (Makhabu *et al.*, 2002), herbivore foraging area preferences (Bailey *et al.*, 1996), artificial waterpoint placement (Ayeni, 1977; Owen-Smith, 1996) and waterpoint type (Washington-Allen *et al.*, 2004) needed to be considered.

In addition to heterogeneity, another factor missing from previous waterpoint studies in the Great Limpopo Transfrontier Conservation Area was the effect of intrinsic system properties and processes. Studies have investigated the effects of herbivores on soils and vegetation with no reference to the effects of intrinsic system properties such as geology on soils and soils on vegetation (Thrash *et al.*, 1991a,b; Thrash, 1997; Brits *et al.*, 2002). A study in Botswana found that environment was more important than disturbance when defining vegetation composition (Makhabu *et al.*, 2002) and a study from Australia found that environment and disturbance can have different effects on species composition (McIntyre & Lavorel, 1994). In the southern African savannas, geology and soils have important effects on vegetation (Venter *et al.*, 2003).

This study used data collected during the piosphere testing phase to begin the development of a new approach to understanding the impact around waterpoints. Because water provision is associated with a high risk of degradation (Chapter 2), it is essential that management is based on sound ecological theory. The effect of waterpoint type (open artificial, closed artificial or natural) and catenal position of the waterpoint on impact level within 200m of the waterpoint was investigated to determine whether artificial waterpoints were substantially different to natural waterpoints. Impact level was a score calculated from soil functionality and herbaceous and woody vegetation characteristics. Impact scores were tested over 22 waterpoints from five properties.

Waterpoint type affected the impact level within 200m of the waterpoint with natural waterpoints having a significantly lower level of degradation than artificial waterpoints. It is important to note that this analysis combines waterpoints from different management areas which have previously been subjectively labelled as very different in terms of impact levels. Despite this subjective separation, the effect of natural vs. artificial waterpoint is still statistically significant. Catenal location of waterpoint had no significant effect on impact level.

In order to determine factors that have important influences on degradation and species composition, variables were split between species composition, degradation variables, environmental variables and management variables. Species composition was taken as the frequency of herbaceous and woody species. Degradation variables were characteristics of vegetation and soils that respond to herbivore impact, for example soil infiltration (Thrash, 1997) or the proportion of perennial plants (McIntyre & Tongway, 2005). Environmental variables were variables that naturally characterise the condition of the landscape, for example landscape type (Gertenbach, 1983) or catenal position (Augustine, 2003). Management variables were actions that management have taken which could change the condition of the landscape, for example artificial water supplementation (James *et al.*, 1999). Ordinations were performed with data from 218 sampling sites.

Ordination analyses revealed that environmental variables had a stronger effect on explaining variation in species composition and degradation variables than management variables. The best predictor of variation in species composition and degradation variables was the combination of management and environmental variables. Under this scenario, grouped environmental and management variables had approximately equal importance for explaining variation in both species and degradation variables (Figure 5). Natural and artificial water availability levels were the most important variables overall for explaining variation between sampling sites. Property size and distance to a perennial river were the next most important. Distance to waterpoint (the piosphere effect) was much less important.

Now that the important factors in determining degradation and species composition in the study area had been identified, it was necessary to transfer this into an approach applicable to management. Management currently understands the landscape in terms of concentric circles of impact (and therefore function and resilience) focused on waterpoints. In order to characterise landscapes and properties, the relationship between each management and environmental variable and each degradation variable was investigated. This enabled development of functionality scores for each management and environmental variable which ranged from 1 (low function) to 6 (high function). Properties could then be scored for management and environmental variables.

Characterisation of properties was performed based on the top five most important variables: natural water availability, artificial water availability, property size, distance to a perennial river and landscape type. This analysis revealed that the national parks had a similar functionality and were more functional than the private reserves. However, it also highlighted that the private reserves have a lower baseline potential functionality than the national parks (Figure 6). This is primarily due to their smaller size which results in a lower variation in landscape types. The landscape types found in the private reserves are drier overall than those in the national parks (Chapter 2).

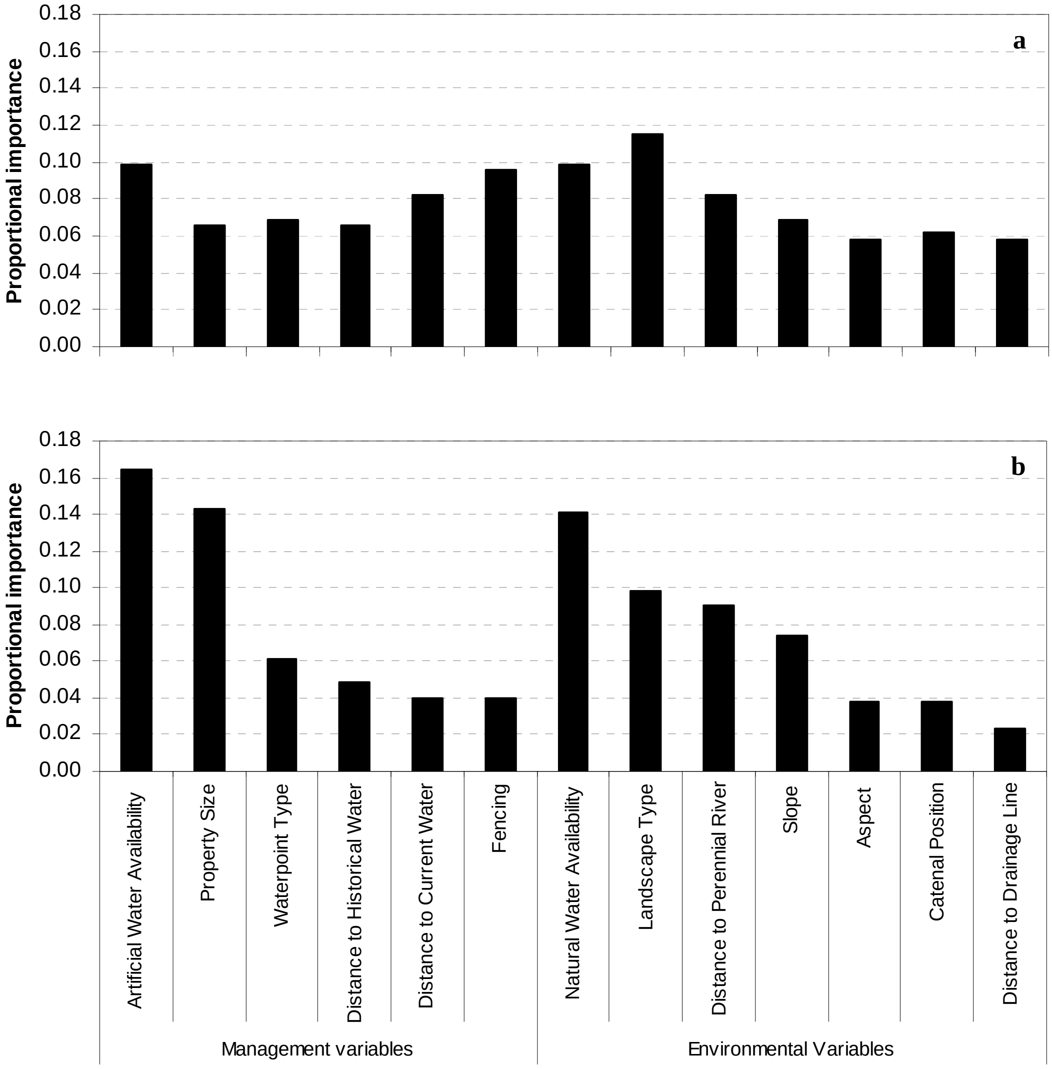

Figure 5: Proportional importance of management and environmental variables in explaining variability between sampling sites for (a) species composition and (b) degradation variables. Proportional importance calculated from Marginal Effect Variance Explained from RDAs of species and degradation variables.

Characterisation of multiple variables from one property highlights the variation in the scale of impact between the different environmental and management variables (Figure 7). The management and environmental variables that were highlighted as particularly important in explaining variation are broad-scale variables. This has important implications for monitoring the effects of water supplementation on property resilience – it is likely that broad scale monitoring can be used and that waterpoints will not need to be specifically targeted.

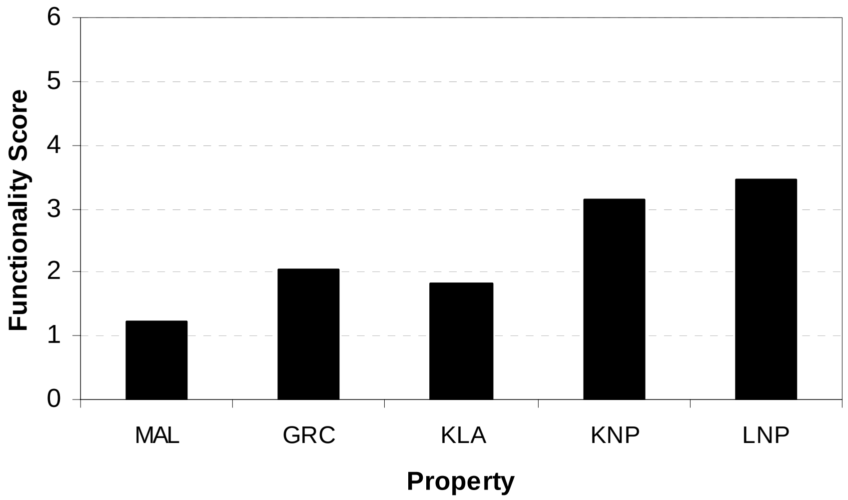

Figure 6: Functional characterisation of properties of the study area based on variables that management cannot affect from the top five most important from Marginal Effect Variance Explained: natural water availability, distance to perennial river and landscape type.

The most important variables that were highlighted during analysis were artificial and natural water availability. Artificial water availability is linked to the property stocking rate (herbivore pressure) as areas with more water can support (in non-drought years) a greater water-dependent herbivore stocking density (Cronje *et al.*, 2005; Peel *et al.*, 2005; Chamaillé-Jammes *et al.*, 2007). Natural water availability reveals the resilience of the vegetation to herbivore impact as areas that are naturally wetter will have vegetation which is adapted to tolerate high herbivore pressure (Milchunas *et al.*, 1988; Prins & van der Jeugd, 1992; Mushove *et al.*, 1995). Fire also plays an important role in vegetation change and dynamics (Conedera *et al.*, 2009). Patterns of grazing can have important effects on fire patterns and *vice versa* (Fuhlendorf *et al.*, 2009).

MOVING FORWARD

Spatial heterogeneity and ecosystem resilience are important in conservation (du Toit *et al.*, 2003). This group of studies has shown that the ecosystem impacts of artificial water supplementation are a potential cause for concern. Artificial waterpoints are found in very high densities on some properties and the distribution of available water, and therefore herbivore impact, does not always follow natural patterns. This leads to a potentially high risk of degradation of soils and vegetation across properties.

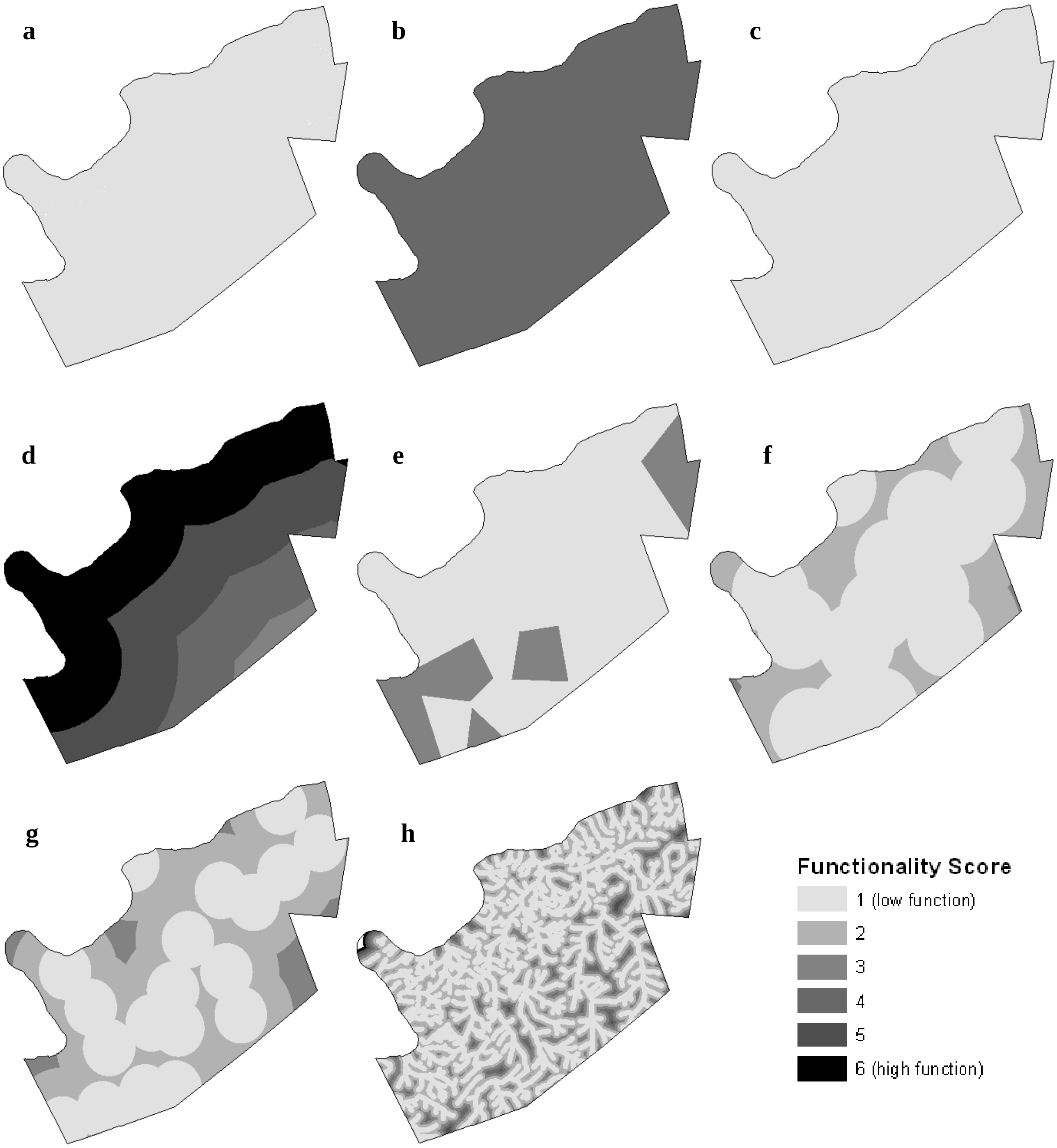

Figure 7: Functionality scores for environmental and management variables as found on Greater Olifants River Conservancy. Variables are (a) natural water availability, (b) artificial waterpoint density, (c) property size, (d) distance to a perennial river, (e) waterpoint type, (f) distance to a currently open waterpoint, (g) distance to a historical waterpoint, and (h) distance to non-perennial drainage line.

To date, the impacts of water supplementation have been understood in terms of concentric circular patterns focused on waterpoints (Owen-Smith, 1996). Under this approach, properties with higher density water have a higher proportion of their property at risk of degradation (areas closer to waterpoints have a higher degradation risk). This group of studies has shown that distance to water is not a dominant variable for indicating the degradation risk of a particular area. Broad scale environmental and management factors such as total artificial water availability and natural water availability are both more important. Interestingly, distance to a perennial river is more important than distance to an artificial waterpoint. This has important implications, particularly for the private reserves. These smaller properties can have a very low potential function as their smaller size means that they can be located far from perennial rivers.

The results of the studies have important implications for understanding and managing spatial heterogeneity and ecosystem resilience of southern African savanna conservation areas. A more detailed approach to understanding the effects of water management decisions on property resilience is required. This approach needs to encompass a wider range of variables, for example variables describing baseline potential function as well as management decisions. At the same time, monitoring does not need to be focused on the immediate surroundings of waterpoints because of the generalised impact that water supplementation has across a property.

In order to create a method and understanding which is transferable between properties, biomes and continents, it is important that the fundamental expression of, and reasons for, heterogeneity are understood. The understanding of the impact of supplementing water needs to move more into the field of landscape ecology and combine the understanding of heterogeneity with the biophysical properties of waterpoint surroundings and the subsequent impacts of disturbance on the biophysical properties.

The functionality scores developed as part of this study are based on simple data which has been collected as part of many studies across Kruger National Park. There is probably sufficient data available to characterise the full set of environmental and management variables found in the southern African savannas and to roll this approach out as a working management approach.

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Chapter 7

Waterpoints: management issue and management tool

Helen Farmer

ABSTRACT

Generation of results from conservation results has value only if the results are made available to management. This chapter presents a shortened synthesis to increase the information available for management with regards to water supplementation decisions and monitoring of the effects of water supplementation. Supplementation of water is an important tool in the southern African savannas as it can be used to achieve production and biodiversity objectives. However, artificial water supplementation leads to greater and more extensive herbivore pressure on vegetation and soils which can lead to a decrease in resilience of the property. This can result in ecological consequences which are contradictory to the objectives of the property. The effect of water supplementation on the ecological resilience of properties is currently understood using the piosphere model which has been shown to be an oversimplification of the heterogeneous southern African savannas. A new approach to understanding the effect of water supplementation on ecological resilience needs to incorporate a variety of environmental and management variables. Important variables relate to the herbivore pressure exerted on a property and the natural resilience of the vegetation and soils to herbivore pressure. Spatial heterogeneity and the scale of variation of the variables are also important aspects to consider. It is likely that current monitoring approaches can be adapted to monitor the impact of water supplementation rather than a completely new approach developed. However, it is essential that monitoring is performed with recognition of the importance of the biophysical template.

KEY WORDS

Conservation; grazing gradients; herbivore impact; monitoring; piospheres; savanna

INTRODUCTION

Generation of results from conservation research has value only if the results are made available to management. Many scientific studies lead to increased understanding of the ecology of an area, but the translation of this into management is often ineffective (With, 1997; Underwood, 1998; Maclean & Macintosh, 2002). The level of water provision is an important aspect of southern African savanna conservation management (Owen-Smith, 1996; Gaylard *et al.*, 2003). Increasing permanent water availability for herbivores has subsequent effects on soil and vegetation function and quality (Andrew, 1988; Thrash & Derry, 1999; James *et al.*, 1999). To date, the understanding of the impacts of water supplementation in southern African savannas has been based on an oversimplified view of the ecosystem (Chapter 4). Throughout savanna ecology, the importance of heterogeneity is being increasingly recognised (du Toit *et al.*, 2003). It is imperative that this is translated into all aspects of management, including water provision.

Decisions on water supplementation are ultimately made by the owners and managers of a property and these decisions are affected by human and ecological aspects (Farmer, 2009). It is therefore essential that the ecological understanding presented to management is continuously updated with advances in research (Roux *et al.*, 2006). If the links between water supplementation and ecological changes can be demonstrated, the understanding of owners and managers increases. The human aspect of management decisions cannot be downplayed or regarded as unimportant as it is human demands that set property objectives and human perception of the ecosystem which determines management actions.

This chapter provides a shortened synthesis to specifically address the management aspects of the first five chapters of this study. The aim is to increase the information available for management with regards to water supplementation decisions and monitoring of the impacts of water supplementation. It is important that decisions regarding water supplementation are defensible and logical (Farmer, 2009) and in order to achieve this it is important that management and owners understand the ecological impacts of water supplementation. In contrast to the previous synthesis, this chapter is aimed specifically at a management audience.

WATERPOINTS – MANAGEMENT TOOL

Supplementation of water is an important management tool in the southern African savannas. The establishment of national and private conservation areas from 1926 to the 1960s was associated with the fencing of properties (Peel *et al.*, 2005). Fencing of properties led to greater pressure on existing water sources and the disruption of migration routes, so permanent water sources were supplemented (Walker *et al.*, 1987; Mabunda *et al.*, 2003). The objective for water provision was to stabilise water availability in the dry season and to fully use forage resources on properties (Walker *et al.*, 1987; Aucamp *et al.*, 1992; Grossman *et al.*, 1999). These objectives were tightly linked to agricultural approaches (Aucamp *et al.*, 1992), the only available management theory at the time.

After more research, water provision became seen as an opportunity to increase the biodiversity of a property (Owen-Smith, 1996; Thrash, 1998). Specific location of waterpoints and careful distancing between them, was seen as an opportunity to increase the habitat heterogeneity with associated subsequent increases in biodiversity (Collinson, 1983; Owen-Smith, 1996; Thrash, 1998; Smit *et al.*, 2007). With increased understanding about the effects of water supplementation, a split appeared between properties where differing objectives led to differing levels of water provision. Higher profit requirements have been linked to higher levels of water provision which leads to maximum utilisation of forage resources (Aucamp *et al.*, 1992; Grossman *et al.*, 1999). Closure of waterpoints has been applied in properties which emphasise biodiversity conservation (Gaylard *et al.*, 2003). In areas where the objective is to conserve the natural wilderness, there is no artificial supplementation of water (Grossman & Holden, 2003).

WATERPOINTS – MANAGEMENT ISSUE

Artificial water supplementation uncouples herbivore populations from their natural limitations leading to greater and more extensive herbivore pressure across properties (Illius & O'Connor, 1999; Cronje *et al.*, 2005; Chamaillé-Jammes *et al.*, 2007). The higher stocking rate of properties associated with increase water provision has led to questions about the sustainability of these systems (Craine *et al.*, 2009). When the limitation caused by water availability is removed, herbivore populations become limited by forage availability which can lead to mass mortality during droughts

(Walker *et al.*, 1987). The underlying vegetation functional potential of a property (Peel *et al.*, 2005) becomes more important in determining stocking rate as water provision increases (Thrash, 2000).

Repetitive grazing, browsing and trampling around permanent waterpoints leads to degradation of soil and vegetation (Lange, 1969; Graetz & Ludwig, 1978; Adler & Hall, 2005). Soil health is compromised by high herbivore impact, leading to a reduction in vegetation biomass (Ludwig *et al.*, 2005). Water infiltration (Thrash, 1997) and seedling establishment (Bassett *et al.*, 2005) decline and erosion increases (Belnap & Gillette, 1998). Herbaceous plant density decreases (Bestelmeyer *et al.*, 2006) and there is a shift towards unpalatable (O'Connor, 1994) and annual (Nangula & Oba, 2004) vegetation. The size and growth rate of woody vegetation is decreased (Levick & Rogers, 2008) and susceptibility to fire increases (Mills & Fey, 2005).

The increase in general herbivore pressure across a property and the creation of extreme degradation nodes around waterpoints can lead to property level impacts such as a reduction of resilience (Carpenter *et al.*, 2001; Cumming *et al.*, 2005) and therefore a greater sensitivity to negative effects of disturbance (Gunderson, 2000). The reduction in the resilience and greater sensitivity to disturbance can result in ecological consequences which are contradictory to the objectives of the property (e.g. Walker *et al.*, 1987).

UNDERSTANDING THE IMPACTS OF WATER SUPPLEMENTATION

Currently, the impact of water supplementation on the ecological resilience of a property is understood based on the piosphere model of concentric circles of differing impact levels focused on waterpoints (Owen-Smith, 1996; Gaylard *et al.*, 2003; Ryan & Getz, 2005). The piosphere model was developed in Australia as an approach to understand and manage water provision in livestock systems (Lange, 1969; Graetz & Ludwig, 1978). The pattern of impact around waterpoints was termed a *piosphere* and was described by a zone of high impact near the waterpoint, then a zone of decreasing utilisation followed by a zone of negligible utilisation far from the waterpoint (Lange, 1969; Graetz & Ludwig, 1978). The logistic curve (Figure 1) was put forward as the best way to describe the relationship between herbivore impact and distance from water (Graetz & Ludwig, 1978).

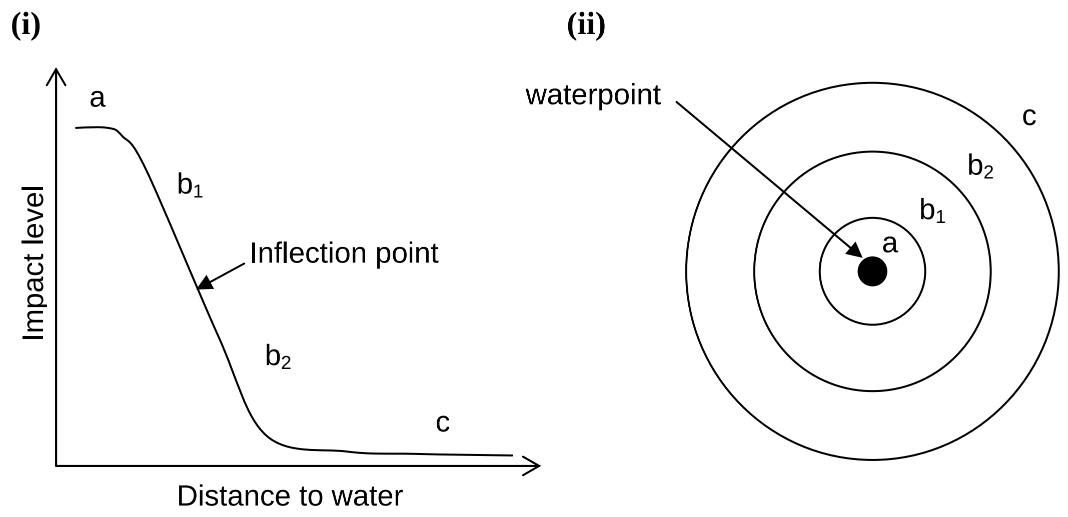

Figure 1: (i) The logistic curve of Graetz & Ludwig (1978) used to describe the zones of a piosphere and, **(ii)** the piosphere as concentric rings of different impact levels with rings corresponding to the logistic curve around a waterpoint indicated by a black circle. Zones are labelled as (a) sacrifice zone, poor condition, (b_1) changing impact, fair condition, (b_2) changing impact, good condition and (c) very little impact, excellent condition.

The piosphere model provides a simple and controllable view of utilisation of the landscape (Foran, 1980; Pickup, 1994) and is therefore attractive to management. Using the piosphere model, the impact of water supplementation can easily be scaled up from individual waterpoints to across proeprties. The landscape is seen as a series of circular impact zones, each centred on a waterpoint (Graetz & Ludwig, 1978; Gaylard *et al.*, 2003; Adler & Hall, 2005). The zones are separated when waterpoints are far apart, and merge when waterpoints are close together. Using this approach, understanding the effects of water supplementation on resilience is simple as function of an area is related simply to its distance from water. Unfortunately, this is an oversimplification of the southern African savannas. There is a high level of heterogeneity in the southern African savannas (Pickett *et al.*, 2003) and this has important impacts on function (Suding *et al.*, 2004).

The piosphere based approach to understanding and managing impact around waterpoints does not acknowledge or incorporate spatial heterogeneity. Conversely, it averages broad areas of the landscape into homogeneous zones (Owen-Smith, 1996;

Redfern *et al.*, 2003; Ryan & Getz, 2005). Even though the idea of concentric circles has been labelled as simplistic (Rietkerk *et al.*, 2000) and data has to be heavily manipulated (Getzin, 2005), the approach is still applied. As heterogeneity within the piosphere has both ecological and management importance, removing it through data analysis does not make sense (Chamaillé-Jammes *et al.*, 2009).

Sampling of transects from waterpoints without the removal of spatial heterogeneity resulted in the rejection of the piosphere model as an appropriate base for conservation management and waterpoint impact understanding (Chapter 4). The biophysical template has a strong influence on herbivore movement (Bailey *et al.*, 1996) and vegetation adaptation to herbivore pressure (Anderson *et al.*, 2007). This means that grazing and browsing impacts are superimposed on a heterogeneous template, rather than the homogeneous template assumed in the piosphere model (Graetz & Ludwig, 1978). Even in the close vicinity of water where water is the single most important attractor in the landscape, the biophysical template affects herbivores and vegetation to an extent where it is possible to get no consistent pattern in vegetation change within 50m of a single waterpoint (Figure 2).

In order to develop a new approach to understanding herbivore impact across properties, the effects of management and environmental variables on degradation variables and species composition were investigated (Chapter 5). Management variables considered were current and historical waterpoint locations, type of nearest waterpoint, artificial waterpoint density, property size and fencing. Environmental variables considered were aspect, slope, catenal position, landscape type, natural water availability, distance to drainage line and distance to perennial river. Among other things, these variables relate to intensity of management, property herbivore pressure and vegetation resilience to herbivore impact.

Variables that consistently emerged as important were natural water availability, level of artificial supplementation, property size, distance to perennial river and landscape type. The important environmental variables relate to the ability of the area to withstand herbivore impacts. Under a natural system, areas of the landscape with permanent water would have received higher utilisation pressure from herbivores during the dry season (Chamaillé-Jammes *et al.*, 2007). Vegetation of drainage lines is characterised by adaptations to handle consistent herbivore pressure (Milchunas *et al.*, 1988). Important management variables relate to stocking rate and intensity of management. Stabilising water availability reduces variability in access to forage resource and therefore reduces the likely natural cause of fluctuations in herbivore abundance (Cronje *et al.*, 2005; Chamaillé-Jammes *et al.*, 2007). As properties get larger, they have a greater scope for more natural and broader scale management (Peel *et al.*, 1999).

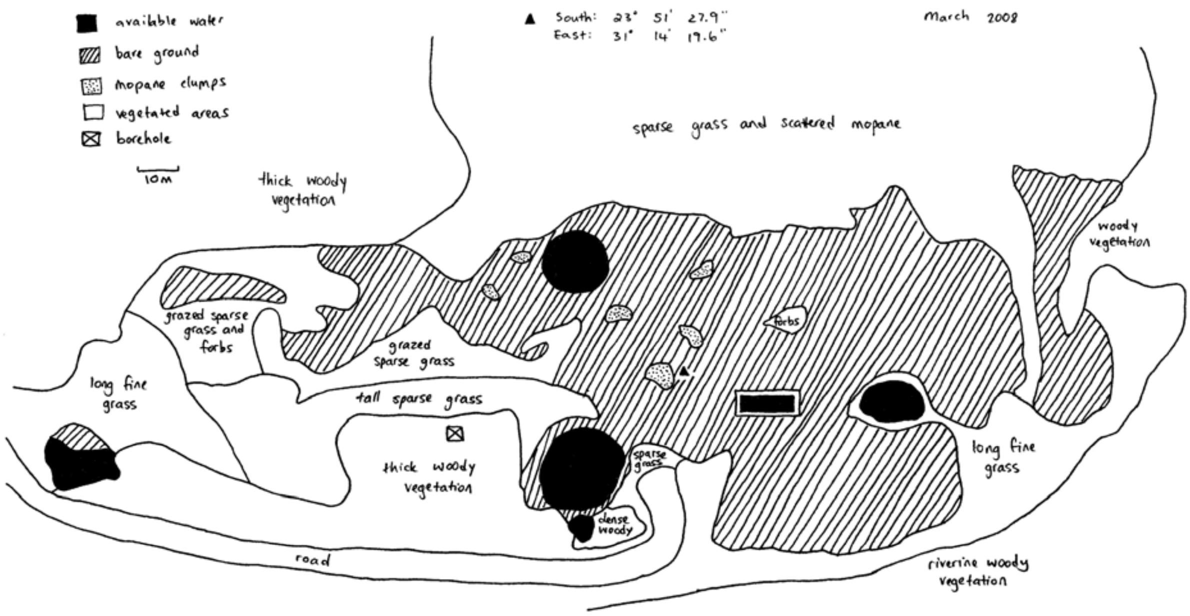

Figure 2: Scale drawing of a waterpoint in Kruger National Park (Ngwenyeni) showing the lack of consistent pattern of impact on vegetation within close proximity to water.

When considering the variation in management and environmental variables across properties, the impact of heterogeneity on the scale of change in variables is obvious (Figure 3). The variation in scale of change between variables highlights the importance of the inclusion of heterogeneity in understanding the impacts of herbivores across properties and the effects of water supplementation. In order to give a more accurate representation of the impacts of water supplementation on ecological resilience of a property, it is essential to include a greater number of variables than simply distance to water. The complexity of the southern African savanna ecosystem means that a simplified model such as the piosphere model is not applicable for management (Chapter 4, 5). Heterogeneity is important for biodiversity and ecosystem function (Fuhlendorf & Engle, 2001; Elmqvist *et al.*, 2003; Tylianakis *et al.*, 2008) and therefore for conservation management objectives. Heterogeneity therefore should not be trivialised in any aspect of management. As our understanding of the ecosystem increases, management of conservation properties needs to be continuously updated (Roux *et al.*, 2006).

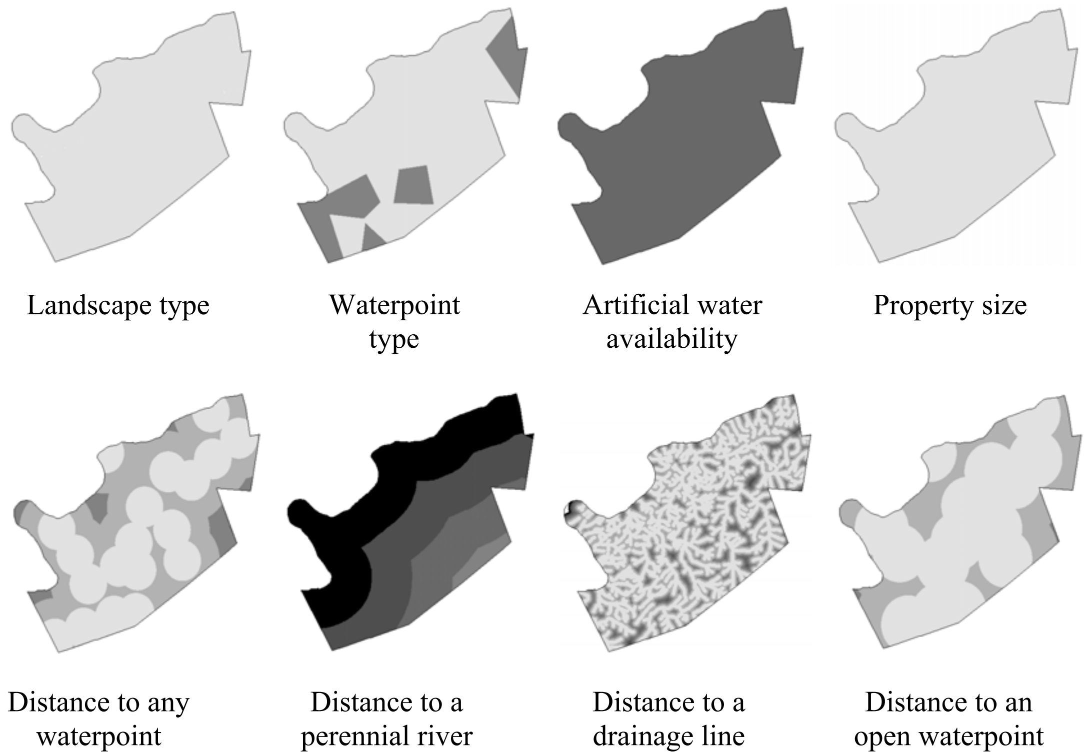

Figure 3: The scale of variation of selected environmental and management variables over the Greater Olifants River Conservancy. Darker shading indicates a higher level of functionality.

RESILIENCE AND WATER SUPPLEMENTATION

Variations in herbivore utilisation across a property and the resilience of the soils and vegetation to herbivore impact are important when considering the ecological resilience of a property. An important aspect of sustainability of the management approach of a property is the property's resilience. At broader scales, this is also important in transboundary conservation areas as the relative resilience of properties contributes to the resilience of the region as a whole (Carpenter *et al.*, 2001; Cumming *et al.*, 2005). A management approach which decreases resilience will ultimately lead to breakdown of the system and undesired ecological impacts (e.g. Walker, 1987). Declines in ecosystem resilience of functional states are linked to herbivore impact and the biophysical template (Westoby *et al.*, 1989; Friedel, 1991; Suding *et al.*, 2004). Artificial waterpoints do form degradation nodes with very high localised impact (Chapter 5). However, beyond the immediate surroundings of the waterpoint, the heterogeneity in the biophysical template has important mitigating or exacerbating effects on degradation (Chapter 4, 5).

When discussing variations in property resilience, management approaches and ecological impacts of management decisions, it is important to recognise and consider the management constraints that differ between properties. Property size and location have important impacts on management decisions and the effects of management actions (Chapter 2, 5). Variations in the biophysical template between properties can affect the responses of the herbivores and vegetation resulting in potentially different responses to the same management action. Small properties located in areas with lower natural resilience to herbivore impact will be more highly degraded and impacted. However, the size of these properties means that at the broader scale of the transfrontier conservation area, their relative impact on the resilience of the region will be low.

The variation in constraints and resilience between properties and differing responses to the same management action can help properties learn about the effects of management strategies. For example, the high level of water provision on the private reserves may enable detection of other factors that play an important role in herbivore distribution and impact patterns. Increased communication within the transfrontier conservation area offers the opportunity for increased information exchange. Understanding the importance of the biophysical template contributes to improved communication between properties as management constraints are more clearly understood. The impacts of differing management approaches can be more easily understood when the underlying variation that they are effected upon is understood. It is important that information exchange between properties is scale and management constraint appropriate.

APPROACHING WATER PROVISION

When discussing the issue of how to approach water provision, it is important to remember that there is no "recipe for success". How a property approaches water provision should be directly and explicitly driven by their objectives. This study does provide a few general rules which can help in reducing the risks associated with artificial water supplementation. Most importantly, areas of the landscape with higher natural levels of water are more resilient to herbivore impacts.

Questions of water provision levels become more important as property size decreases. Smaller properties have a lower chance of being able to minimise detrimental impacts of water provision, for example if they are fenced and located completely in a naturally drier area. In these properties, water provision levels have to be decided based on a risk assessment of the property functionality and water provision locations have to be decided based on a set of explicit and logical rules (Farmer, 2009).

THE IMPORTANCE OF MONITORING

An important element in understanding the effects of artificial water manipulation is monitoring. Monitoring enables management to assess the current alignment of their property's ecological state with their conservation objectives and to track changes in the ecological state with changes in management (Rogers & Biggs, 1999). Monitoring therefore needs to address factors which are important to management objectives (Biggs & Rogers, 2003). Factors that emerged as important when understanding the impact of water supplementation were (1) the natural resilience of the vegetation and soils to herbivore impact, and (2) the herbivore pressure exerted on the ecosystem (Chapter 5). Therefore, it is likely that current monitoring approaches can be adapted rather than a completely new approach developed. Specifically, the data needs to be looked at from a different perspective and the importance of the biophysical template needs to be recognised.

The importance of the biophysical template can be easily incorporated in vegetation and soil monitoring. The inclusion of soil functional health would improve the monitoring in terms of property resilience. Soil forms the basis of ecosystem health (Ludwig *et al.*, 2000) and is therefore important in understanding the ecological resilience of an area. Vegetation monitoring plots should be arranged according to the biophysical template. Variations in landscape type and slope were more important in determining degradation level than distance to waterpoints so factors such as these should be included in monitoring. It is important that a dataset of environmental and management variables be collected to fully describe each vegetation monitoring plot (Table 1). This dataset must be actively used in analysis and interpretation of monitoring data.

Monitoring of herbivore pressure should be performed at broad and fine scales. Stocking rate of a property gives the overall herbivore pressure exerted on its resources. The spatial distribution of pressure is also important. The pressure at each vegetation monitoring site should be recorded as it is important in interpretation of monitoring data. Interpreting herbivore monitoring data with regard to variables such as distance to perennial river and not just distance to artificial waterpoint will help to start to understand the complex relationships between management and environmental variables and herbivore distribution and impact patterns.

Table 1: Important variables that need to be collected at monitoring sites and actively used in analysis and interpretation of monitoring data.

Management Variables	**Environmental Variables**
Artificial water availability - property density of artificial waterpoints - distance to the nearest artificial waterpoint - type of nearest waterpoint - distance to the nearest open artificial waterpoint	Natural water availability - landscape type ephemeral and permanent water availability scores - distance to the nearest perennial river - distance to the nearest drainage line
Stocking rate	Catenal position Aspect Slope Landscape type

CONCLUDING REMARKS

The current approach to understanding property impacts of artificial water supplementation is based on an oversimplification of the ecosystem. A range of environmental and management variables are important in determining degradation impact in an area. In order to understand the impact of water supplementation, the resilience of vegetation and soils to herbivore utilisation and the degree of herbivore pressure need to be considered. With the inclusion of a more complex model of the ecosystem in the understanding of the impacts of artificial water supplementation, it will be possible to more finely attune management decisions to management objectives.

Based on this new approach to understanding the impacts of water supplementation, a more holistic system can be designed for the management and monitoring of water supplementation in the heterogeneous southern African savannas. This system will fit within the Thresholds of Potential Concern approach currently applied in the Kruger National Park (Biggs & Rogers, 2003) and offers the opportunity to extend this approach to neighbouring properties and across the transfrontier region. The system will need to be tested over a period of at least a decade in order to experience the temporally variability as well as the spatial variability in the region.

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Appendices

A1 WATERPOINT LITERATURE DATABASE

Details of publications used during the waterpoint literature analysis. Full reference details for each paper can be found in the reference list of Chapter 1. 'Pios' refers to whether the paper used the piosphere terminology as opposed to 'GG' for grazing gradients.

#	Author	Year	Pios	GG	Country	Variable class	Animal Type
1	Valentine	1947		1	United States	Vegetation	Livestock
2	van der Schijff	1959			South Africa	Vegetation	Wild
3	Lange	1969	1		Australia	Vegetation	Livestock
4	Graetz & Ludwig	1978	1		Australia	Vegetation	Livestock
5	Foran	1980	1		Australia	Vegetation	Livestock
7	Collinson	1983			South Africa	Vegetation	Wild
8	Lange	1985			Australia	Animals	Livestock
9	Andrew & Lange	1986	1		Australia	Soil	Livestock
10	Andrew & Lange	1986	1		Australia	Vegetation	Livestock
11	Tolsma et al.	1987			Botswana	Soil	Livestock
12	Andrew	1988	1		Review		Both
13	Pickup & Chewings	1988		1	Australia	remote	Livestock
14	StaffordSmith	1990	1		Australia	Model	Livestock
15	Van Rooyen et al.	1990		1	South Africa	Vegetation	Wild
16	Kalikawa	1990		1	Botswana	Vegetation	Wild
17	Thrash et al.	1991		1	South Africa	Vegetation	Wild
18	Thrash et al.	1991		1	South Africa	Vegetation	Wild
19	StaffordSmith	1991		1	Australia	Model	Livestock
20	Hanan et al	1991	1		Senegal	remote	Livestock
21	Bosch & Gauch	1991		1	South Africa	Vegetation	Livestock
22	Thrash et al.	1993		1	South Africa	Animals	Wild
23	Bastin et al	1993	1	1	Australia	remote	Livestock
24	Bastin et al	1993		1	Australia	remote	Livestock
25	Cridland & StaffordSmith	1993			Australia	remote	Livestock
26	Van Rooyen et al.	1994		1	South Africa	Vegetation	Wild
27	Pickup	1994		1	Australia	remote	Livestock
28	Thrash et al.	1995		1	South Africa	Animals	Wild
29	Fusco et al.	1995		1	United States	Vegetation	Livestock
30	OwenSmith	1996	1		Southern Africa	Model	Wild
31	Navie et al.	1996		1	Australia	Vegetation	Livestock
32	Thrash	1997		1	South Africa	Soil	Wild
33	Jeltsch et al.	1997	1		South Africa	Model	Livestock
34	Friedel	1997		1	Australia	Vegetation	Livestock
35	Pickup & Bastin	1997		1	Australia	remote	Livestock
36	Hodgins & Rogers	1997	1		Australia	Soil	Livestock
37	Thrash	1998		1	South Africa	Vegetation	Wild
38	Pickup et al.	1998		1	Australia	remote	Livestock
39	Verlinden et al.	1998		1	Botswana	Vegetation	Both
40	Moleele & Perkins	1998	1		Botswana	Soil	Livestock
41	du Plessis et al.	1998		1	Namibia	Vegetation	Wild
42	Thrash	1998		1	South Africa	Vegetation	Wild
43	Turner	1998		1	Mali	Soil	Livestock
44	Weber et al.	1998		1	South Africa	Model	Livestock

45	Thrash & Derry	1999	1		South Africa		Wild
46	Parker & Witkowski	1999		1	South Africa	Vegetation	Wild
47	Nash et al.	1999		1	United States	Vegetation	Livestock
48	Turner	1999		1	Mali	Vegetation	Livestock
49	James et al.	1999	1		Australia		Both
50	Brits et al.	2000	1		South Africa	Vegetation	Wild
51	Thrash	2000		1	South Africa	Vegetation	Wild
52	Rietkerk et al.	2000		1	Burkina Faso	Vegetation	Livestock
53	Hunt	2001	1		Australia	Vegetation	Livestock
54	Fernandez-Gimenez & Allen-Diaz	2001		1	Mongolia	Vegetation	Livestock
55	Heshmatti et al.	2002	1		Australia	Vegetation	Livestock
56	Brits et al.	2002	1		South Africa	Vegetation	Wild
57	Landsberg et al.	2002		1	Australia	Vegetation	Livestock
58	Makhabu et al.	2002		1	Botswana	Vegetation	Wild
59	Redfern et al.	2003		1	South Africa	Animals	Wild
60	Riginos & Hoffman	2003		1	South Africa	Vegetation	Livestock
61	Beukes & Ellis	2003	1		South Africa	Soil	Livestock
62	Harris & Asner	2003		1	United States	remote	Livestock
63	Landsberg et al.	2003		1	Australia	Vegetation	Livestock
64	Nash et al.	2003	1		United States	Soil	Livestock
65	Tobler et al.	2003		1	Tanzania	Vegetation	Both
66	Leggett et al.	2003	1		Namibia	Vegetation	Both
67	Nangula & Oba	2004	1		Namibia	Vegetation	Livestock
68	Todd	2004	1		South Africa	Vegetation	Livestock
69	WashingtonAllen et al.	2004	1		United States	remote	Livestock
70	Adler & Hall	2005	1		United States	Model	Livestock
71	Getzin	2005		1	Namibia	Vegetation	Wild
72	Smet & Ward	2005	1		South Africa	Vegetation	Both
73	Ryan & Getz	2005	1		South Africa		Wild
74	Cronje et al.	2005	1		South Africa		Wild
75	Todd	2006	1		South Africa	Vegetation	Livestock
76	Smet & Ward	2006	1		South Africa	Soil	Both
77	Brooks et al.	2006	1		United States	Vegetation	Livestock
78	ChamailleJammes et al.	2007	1		Zimbabwe		Wild
79	Derry & Dougill	2008	1	1	Africa		Wild
80	ChamailleJammes et al.	2009	1		Zimbabwe	remote	Wild

A2 NATURAL WATER AVAILABILITY OF KRUGER NATIONAL PARK LANDSCAPE TYPES

Ephemeral and permanent water availability scores for landscapes of Kruger National Park. Landscapes are ordered by number. Detail on how water availability scores were calculated is given in Chapter 3.

Landscape number	Landscape type name	Ephemeral Score	Permanent Score
1	Lowveld Sour Bushveld of Pretoriuskop	0.011	0.136
2	Malelane Mountain Bushveld	0.003	0.045
3	*Combretum collinum*/*Combretum zeyheri* Woodland	0.025	0.059
4	Thickets of the Sabie and Crocodile River	0.020	0.047
5	Mixed *Combretum*/*Terminalia sericea* Woodland	0.033	0.024
6	*Combretum*/*Colophospermum mopane* Woodland of Timbavati	0.034	0.010
7	Olifants River Rugged Veld	0.005	0.055
8	Phalaborwa Sandveld	0.025	0.021
9	*Colophospermum mopane* Woodland/Savanna on Basic Soil	0.023	0.022
10	Letaba River Rugged Veld	0.019	0.035
11	Tsende Sandveld	0.038	0.030
12	*Colophospermum mopane*/*Acacia nigrescens* Savanna	0.025	0.042
13	*Acacia welwitschii* Thickets on Karoo Sediments	0.062	0.026
14	Kumana Sandveld	0.023	0.014
15	*Colophospermum mopane* Forest	0.022	0.035
16	Punda Maria Sandveld on Cave Sandstone	0.015	0.046
17	*Sclerocarya birrea* subspecies *caffra*/*Acacia nigrescens* Savanna	0.016	0.062
18	Dwarf *Acacia nigrescens* Savanna	0.008	0.037
19	Thornveld on Gabbro	0.027	0.034
20	Bangu Rugged Veld	0.005	0.033
21	*Combretum*/*Acacia nigrescens* Rugged Veld	0.006	0.088
22	*Combretum*/*Colophospermum mopane* Rugged Veld	0.019	0.077
23	*Colophospermum mopane* Shrubveld on Basalt	0.006	0.015
24	*Colophospermum mopane* Shrubveld on Gabbro	0.017	0.024
25	*Adansonia digitata*/*Colophospermum mopane* Rugged Veld	0.002	0.020
26	*Colophospermum mopane* Shrubveld on Calcrete	0.007	0.002
27	Mixed *Combretum*/*Colophospermum mopane* Woodland	0.009	0.007
28	Limpopo/Luvuvhu Floodplains	0.009	0.040
29	Lebombo South	0.013	0.071
30	Pumbe Sandveld	0.009	0.005
31	Lebombo North	0.013	0.056
32	Nwambiya Sandveld	0.019	0.000
33	*Pterocarpus rotundifolius*/*Combretum collinum* Woodland	0.015	0.011
34	Punda Maria Sandveld on Waterberg Sandstone	0.006	0.022
35	*Salvadora angustifolia* Floodplains	0.029	0.206

A3: DETERMINATION OF APPROPRIATE DISTANCES BETWEEN SAMPLING POINTS ALONG WATERPOINT TRANSECTS

The first 250m of herbaceous vegetation and 3.5km of woody vegetation around waterpoints is said to have been well characterised. This study did not set out to describe the piosphere effect for the first time but to investigate applicability of piospheres across a management gradient. It was therefore decided that sampling efficiency would be increased by the use of interval sampling. Published results from previous piosphere studies and analysis of Agricultural Research Council (ARC) monitoring data and preliminary field data were used to determine appropriate interval lengths.

Piosphere Zones from the Literature

Zones of impact of herbivore utilisation with increasing distance from water have been identified in the literature (Table 1, Figure 1). Zones vary between variables measured as each is impacted differently by herbivores. A generalised model of change in herbivore impact (piosphere intensity) is often modelled using a logistic equation (Graetz & Ludwig 1978). It is expected that herbivore impact will be higher closer to the waterpoint. Using impact levels on variables given in Table 1, piosphere intensity is higher when there are more variables undergoing high impact (Figure 2).

Table 1: Zones of impact caused by herbivore utilisation for different ecological variables.

Zone	High impact	Medium impact	Low impact	No impact	References
Herbaceous vegetation	<50m	50 – 200m	200m – 10km	>10km	(Thrash et al. 1991a; Thrash 1998a; Thrash & Derry 1999)
Woody vegetation	<1.5km	1.5 – 3km	3 – 7km	>7km	(Thrash et al. 1991b; Brits et al. 2000)
Soil infiltration	<30m	30 – 100m	100 – 150m	>150m	(Thrash 1997)
Herbivore density	<500m	500m – 2.5km	2.5km – 5km	>5km	(Thrash et al. 1995; Owen-Smith 1996; Redfern et al. 2003)

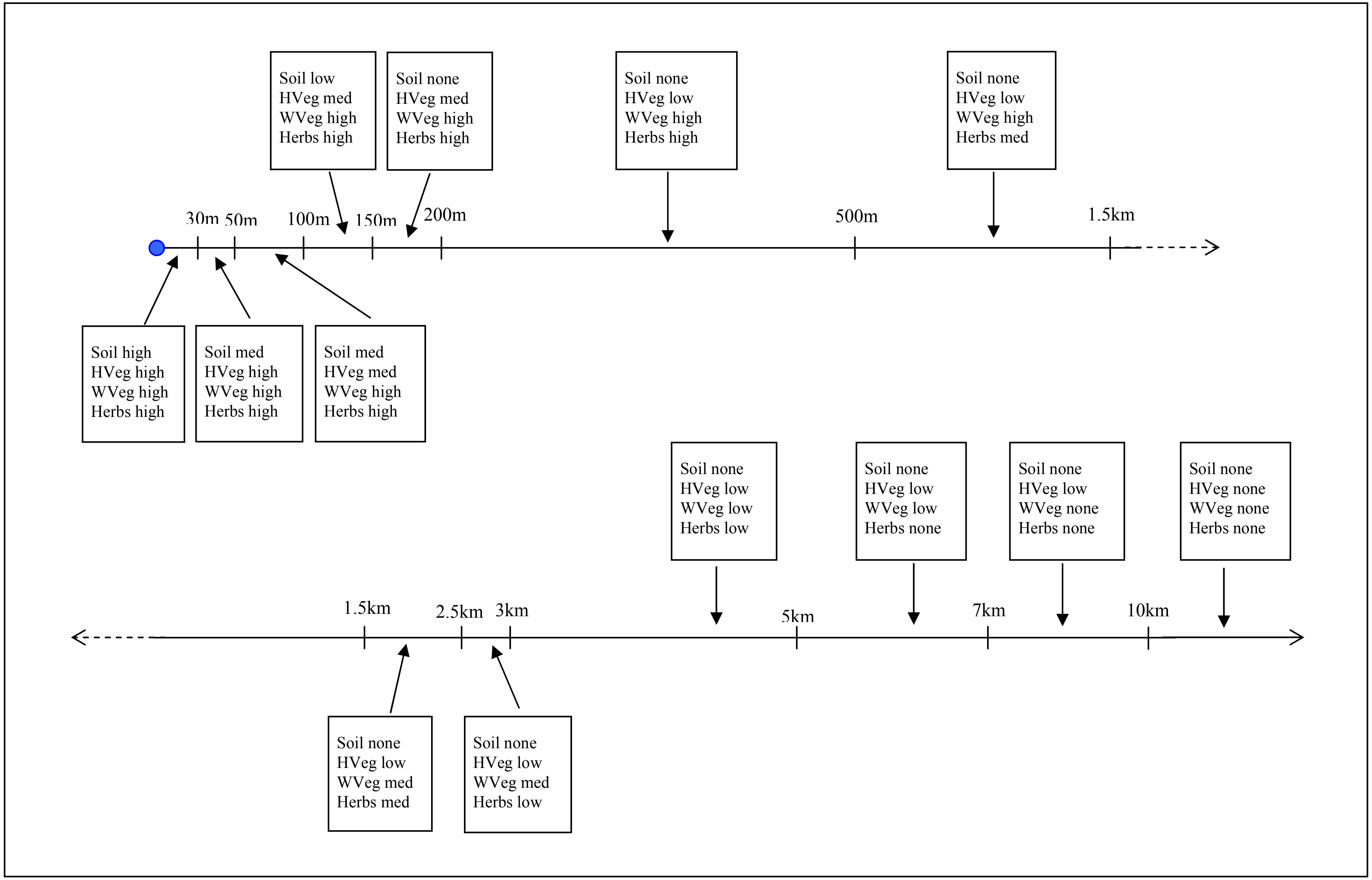

Figure 1: Illustration of the extent of effects on piosphere variables. Soil = soil infiltration (Thrash 1997); HVeg = herbaceous vegetation (Thrash et al. 1991a; Thrash 1998a; Thrash & Derry 1999); WVeg = woody vegetation (Thrash et al. 1991b; Brits et al. 2000); Herbs = herbivore density (Thrash et al. 1995; Owen-Smith 1996; Redfern et al. 2003). High to none describes the impact level.

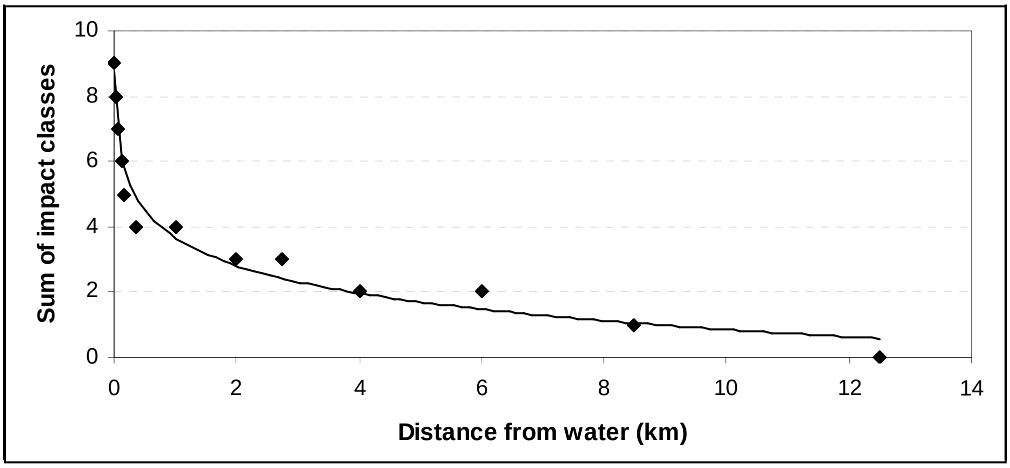

Figure 2: Change in impact intensity with increasing distance from water based on impact classes of Figure 1. High impact = 3, medium impact = 2, low impact = 1, no impact = 0. Impact scores for all variables (Table 1) summed at centre of each interval.

Interval Lengths from the Literature

Piosphere sampling tends to be based on interval sampling with intervals varying between 5m and 400m (Thrash et al. 1991a; Thrash 2000; Riginos & Hoffman 2003; Nangula & Oba 2004). Some studies use constant interval lengths (Thrash et al. 1991a; Thrash et al. 1991b; Brits et al. 2000; Heshmatti et al. 2002) and others use intervals of increasing size (Thrash 2000; Riginos & Hoffman 2003; Nangula & Oba 2004; Adler & Hall 2005). One study was found with continuous sampling (Thrash 1998a) but this study had a maximum transect length of approximately 250m and other studies by the same author have used interval sampling (Thrash et al. 1991a; Thrash et al. 1991b; Thrash et al. 1993; Thrash et al. 1995; Thrash 1997; Thrash 1998b; Thrash 2000).

In studies with a constant interval length, interval length is larger in studies with longer transects (Figure 3) to increase efficiency. However, this approach risks losing the fine detail of change near the waterpoint. In studies with a varying interval length, interval length increases with distance from water (Figure 4). This approach uses higher intensity sampling when changes in piosphere intensity over distance are expected to be rapid (near the waterpoint) and low intensity sampling when changes are expected to be much more gradual (far from the waterpoint).

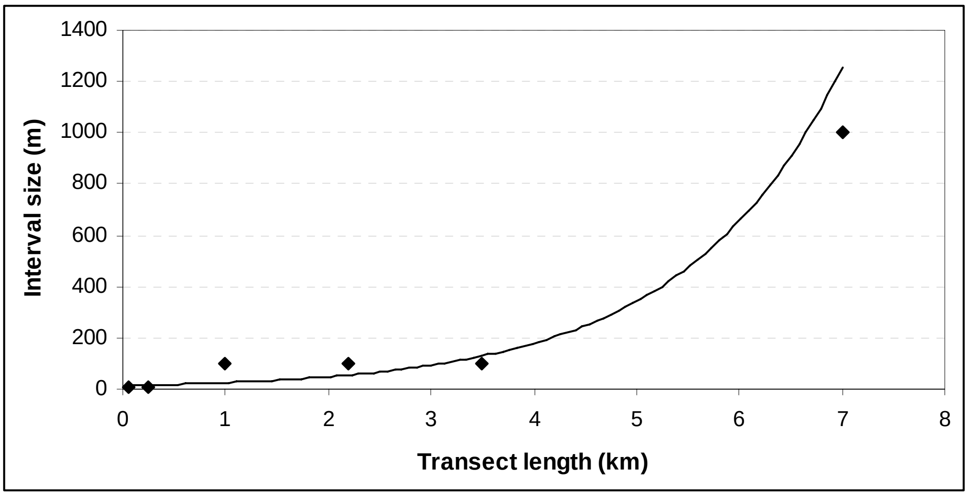

Figure 3: Variation in interval size with increasing transect length from studies published in the scientific literature.

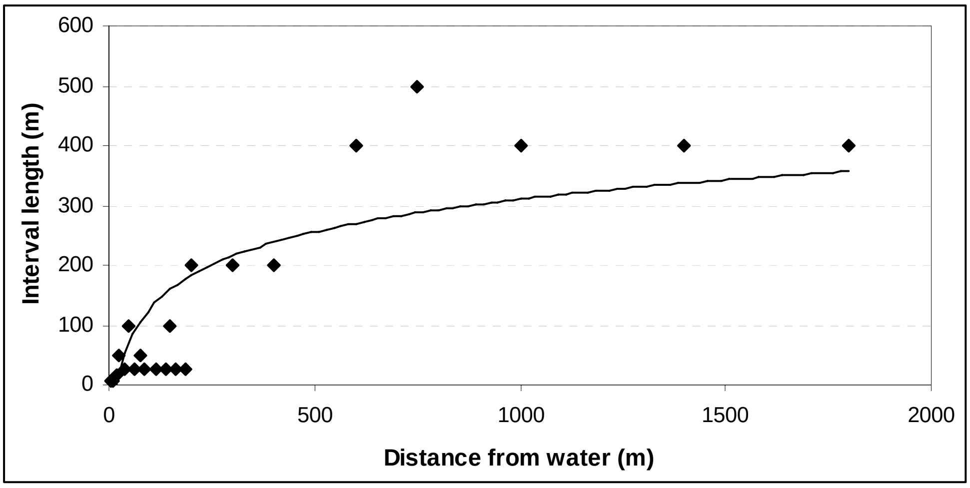

Figure 4: Variation in interval length with increasing distance from water for published studies using variable interval lengths.

ARC Monitoring Data

As a second source of information, data from the ARC Savanna Ecosystem Dynamics' annual vegetation monitoring in the private reserves was used to investigate the piosphere effect. The majority of published studies were performed in Kruger where there is a lower density of waterpoints than in the private reserves. ARC monitoring plots are at varying distances from permanent waterpoints in the private reserves. Herbaceous vegetation data (species composition, distance to tuft and tuft diameter measurements) from 47 monitoring plots was used to investigate the piosphere effect.

Distances between monitoring plots and the nearest permanent waterpoint were measured using a Nearest Neighbour extension in ArcView 3.3. 63% of the waterpoints were between 500m and 1.5km from another waterpoint (Figure 5). Monitoring plots measure 25m x 25m and are therefore small enough to fall within a particular piosphere zone. In order to investigate the piosphere relationship, point to tuft distances and tuft longest axis were plotted against distance from water. Variations in landscape position, vegetation type and property management may mask the piosphere effect and these variables were not controlled in this analysis.

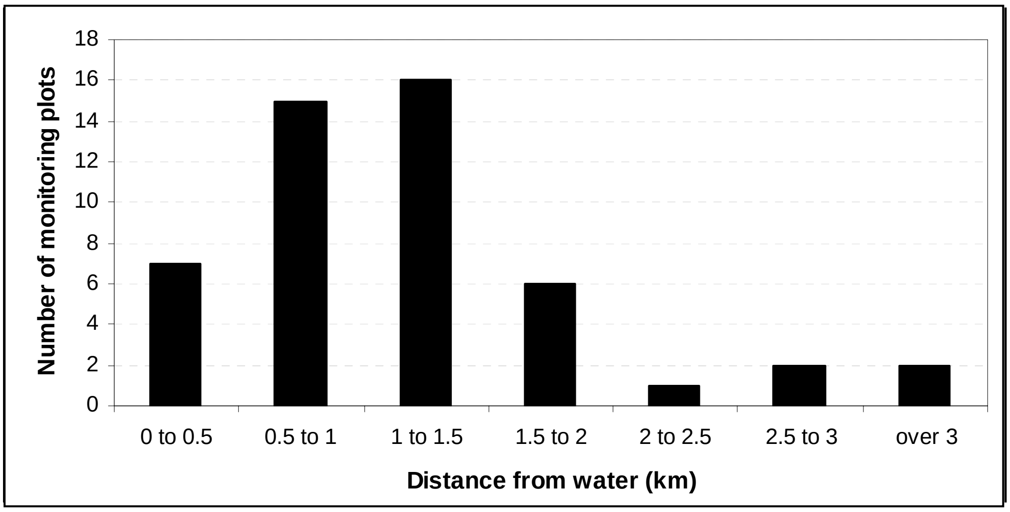

Figure 5: Frequency distribution of distance to water of Agricultural Research Council monitoring plots

ARC Data – Piosphere Effect

As part of annual monitoring, data were collected for annual and perennial grass species. The nearest plant to the sampling point was identified. If the closest plant was an annual grass, sedge or forb, the nearest perennial grass was also recorded (termed second species perennial). Point to tuft distances for perennials and second species perennials were analysed separately.

Annual plant data showed more of a piosphere effect for distance to tuft (distance decreases with increasing distance from water) than for tuft diameter (Figure 6). For the perennial plants, distance to tuft and tuft diameter showed little effect of distance to water (Figure 7)

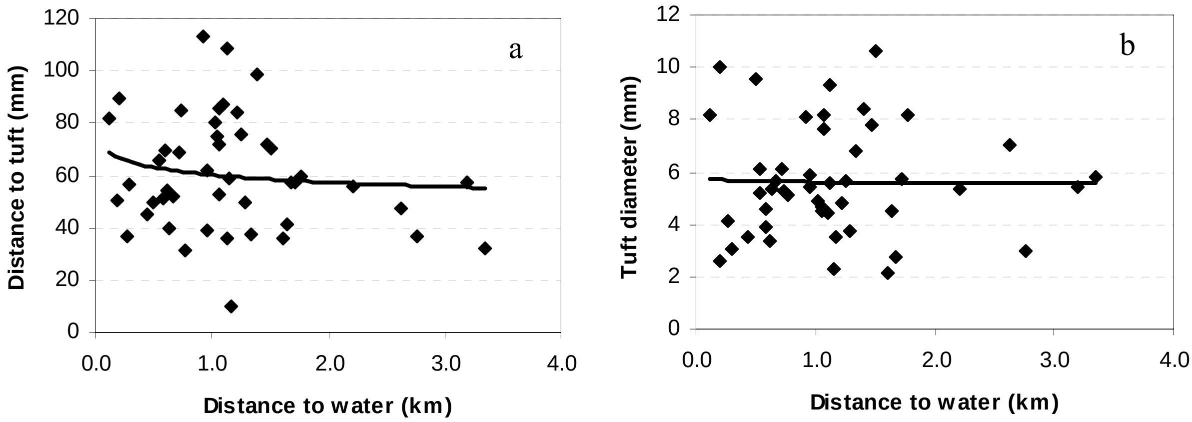

Figure 6: ARC measurements for annual plants (a) point to tuft distance, (b) tuft longest axis.

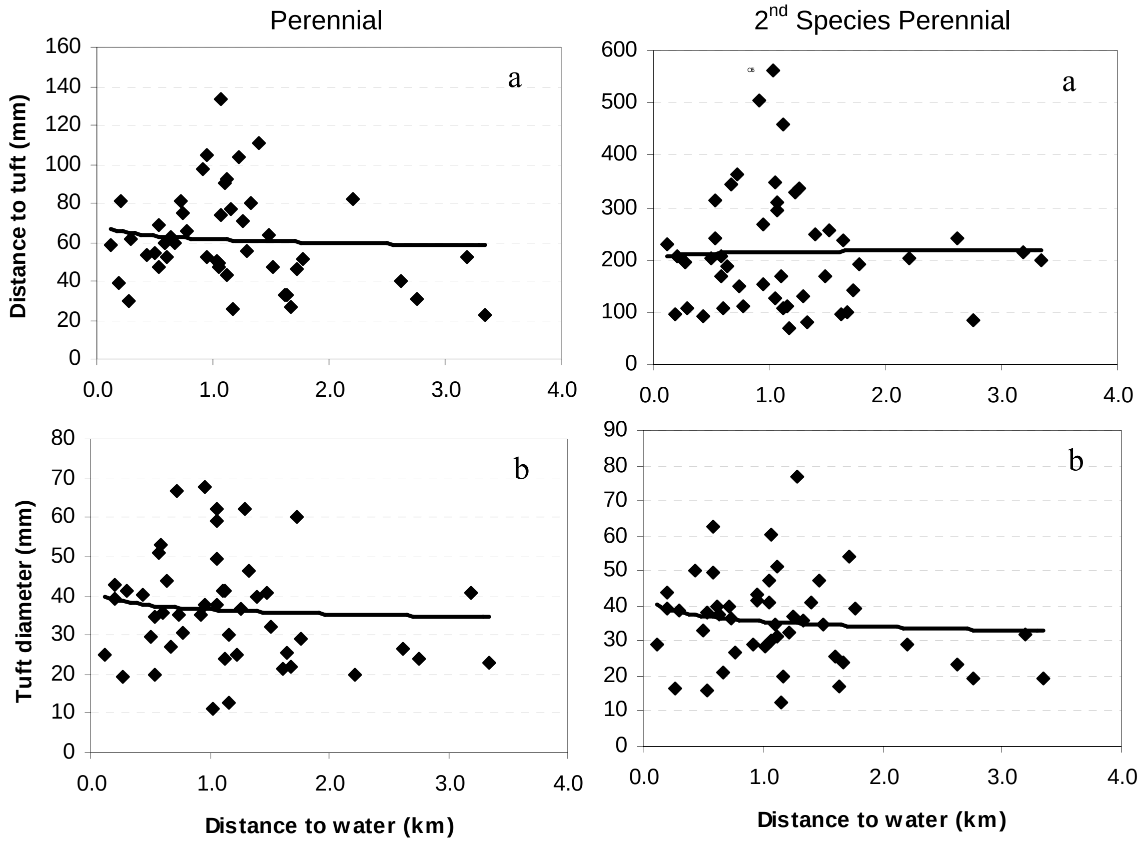

Figure 7: ARC measurements for perennial and second species perennial plants (a) point to tuft distance, (b) tuft longest axis.

ARC Data – Split by Property

In order to decrease variation in vegetation type and soil type and remove variation caused by differences in stocking rate, ARC data was separated by reserves. Tuft diameter tended to decrease with distance from water except in UMB where it rapidly increased with distance from water and GRC where there was a slight and slower increase in tuft diameter with increasing distance from water (Figure 8). Distance to tuft measurements also showed two groups with UMB, GRC and MAL having a different trend to KLA and THB (Figures 9 & 10). This may be a reflection of waterpoint densities and arrangements, UMB and MAL have the highest waterpoint densities of the five reserves analysed and GRC and MAL have the most regular arrangements of waterpoints.

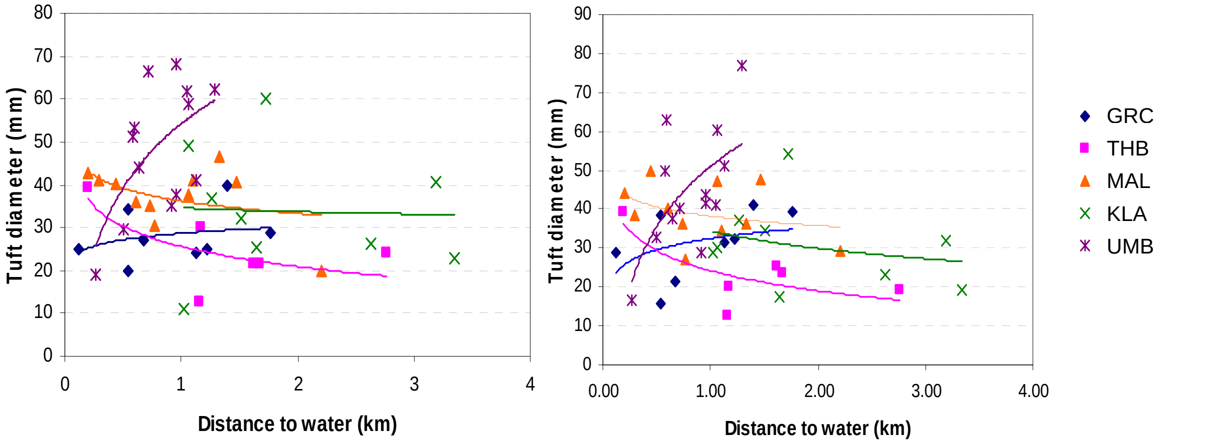

Figure 8: Change in tuft diameter of perennial plants (left 1st perennial, right 2nd perennial) with increasing distance from water.

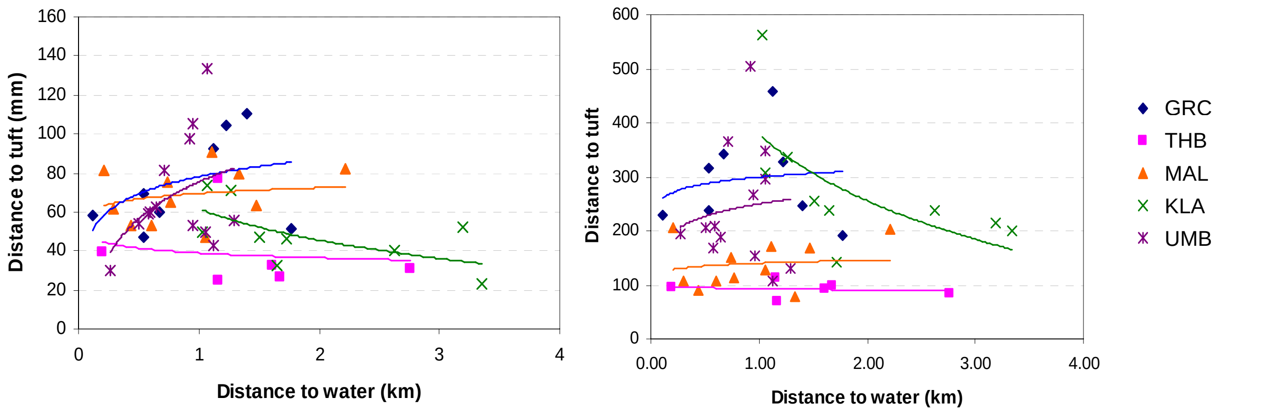

Figure 9: Change in distance to tuft of perennial species with increasing distance from water.

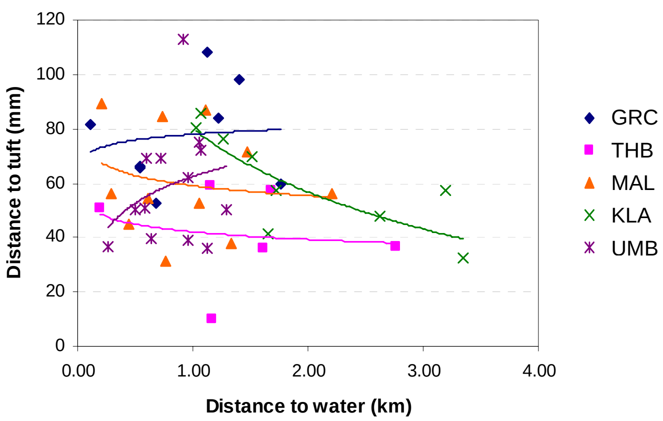

Figure 10: Change in distance to tuft of annual species with increasing distance from water.

Splitting the ARC data by properties showed that the piosphere effect is highly variable between properties. It must be born in mind that the effects of variation in landscape position have not been isolated, and effects of vegetation type and soil type have not been completely removed by analysing the data by property.

Preliminary Data – Piosphere Effect

In May 2006 data was collected at a dam on THB in order to test sampling methods. The first 400m from the waterpoint were sampled using a continuous approach (points were taken every 1m, searching for grasses within 50cm of each point). The point data was averaged over 50m blocks as this is the smallest distance of change expected for herbaceous vegetation according to the literature (Figure 1; Thrash et al. 1991a; Thrash 1998a; Thrash & Derry 1999). The data were then plotted against the distance of the centre of the interval in order to determine whether a piosphere effect could be detected. Point to tuft distances showed a slight decrease with increasing distance from water (Figure 11), though variation within each interval was high. Tuft size was larger and more variable nearer to water (Figure 12). With increasing distance from water tuft size decreases and becomes less variable (Figure 12).

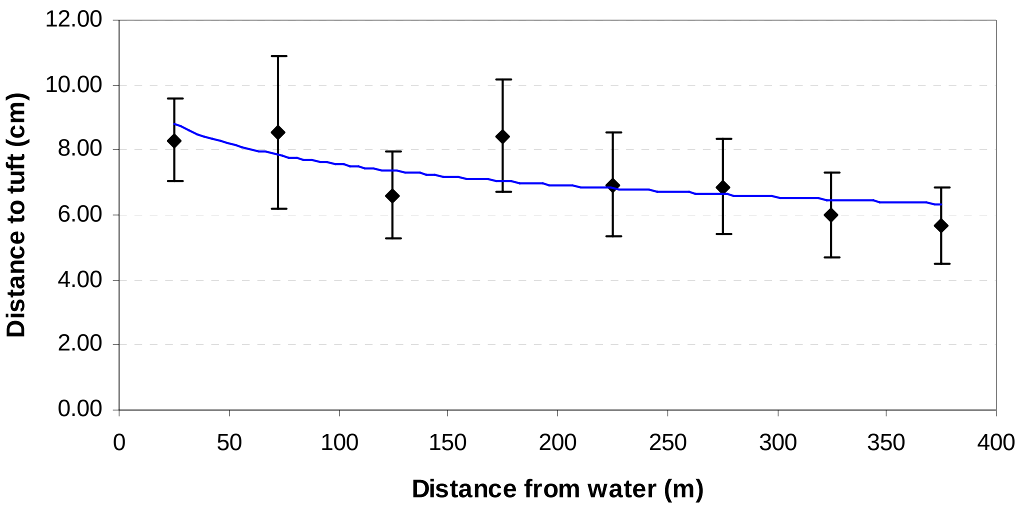

Figure 11: Change in average (±1 SE) sampling point to tuft measurements in the first 400m of a piosphere. Data points were taken every 1m and averaged over 50m. (THB: May 2006)

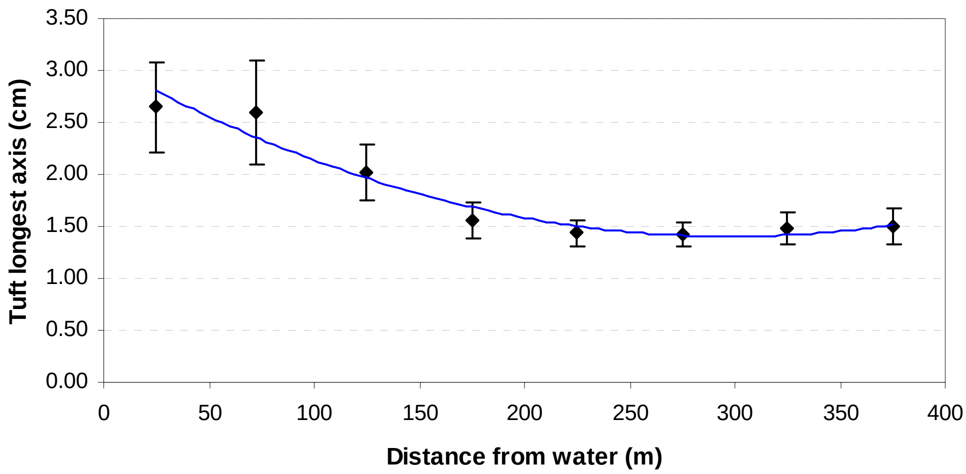

Figure 12: Change in average (±1 SE) measurement of tuft longest axis in the first 400m of a piosphere. Data points taken every 1m and averaged over 50m. (THB: May 2006)

Study Interval Length Determination

According to published studies, piosphere intensity is greatest nearer to the waterpoint, decreases rapidly to approximately 1km after which the change slows down, until after 4km the change in intensity becomes very small (Figure 2). The unit of change for variables changes with distance from water (Figure 1, Table 2). In order to have maximum detection of change, intervals must be smaller in areas with a greater rate of change. Using small intervals in areas with slow rates of change is inefficient. Varying interval lengths will therefore be used.

Table 2: Variation in smallest unit of change of piosphere intensity (impact level of variables) with increasing distance from water. See also Figure 1.

Distance from water	Unit of change
0 – 200m	50m
200m – 3km	500m
over 3km	1000m

The piosphere variables (soil, herbaceous vegetation and woody vegetation) change at different rates. It is inefficient to have separate interval lengths for each of the variables. Woody vegetation change is slower than herbaceous vegetation change so intervals that are sufficiently fine-scale to detect change in herbaceous vegetation should be able to detect change in woody vegetation. The use of one interval distance for all variables will enable easier comparison and calculation of a general piosphere intensity index for a particular distance from water.

A suggested layout of sampling points (and interval lengths) based on the units of change given in Table 2 is shown in Figure 13. Sampling intensity is higher nearer the water and decreases with increasing distance from water. After 3km (when all piosphere variables are at low or no impact) sampling is reduced to a constant interval length of 1km. This 1km interval will be repeated for the whole transect length.

The change in interval length with increasing distance from water from the sampling design suggested in Figure 13 is shown in Figure 14. On comparison with interval lengths used in the literature (Figure 4), interval lengths of up to 500m along the transect are of comparable length. At 1km other studies have slightly shorter intervals (more intensive sampling). However their total transect length is usually shorter. The 1000m intervals after 3km cannot be compared with the variable interval length in the literature as other transects do not go beyond 2km.

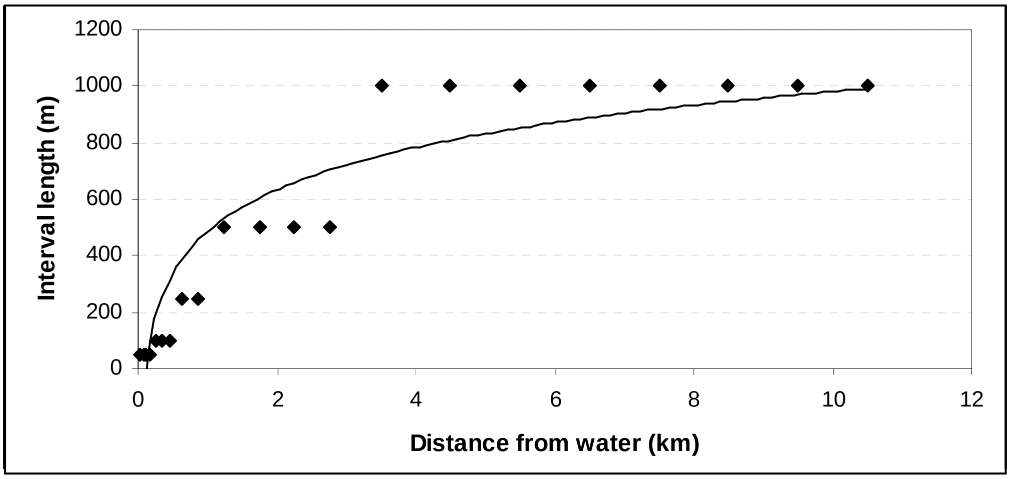

Figure 14: Change of interval length with increasing distance from water for sampling design suggested for this study.

Conclusions

This study will use varying interval lengths between sampling sites starting with 50m at the waterpoint and ending with 1km interval lengths far from the waterpoint (as suggested in Figure 13). Sampling at each sampling site will be oriented in the direction of the waterpoint transect.

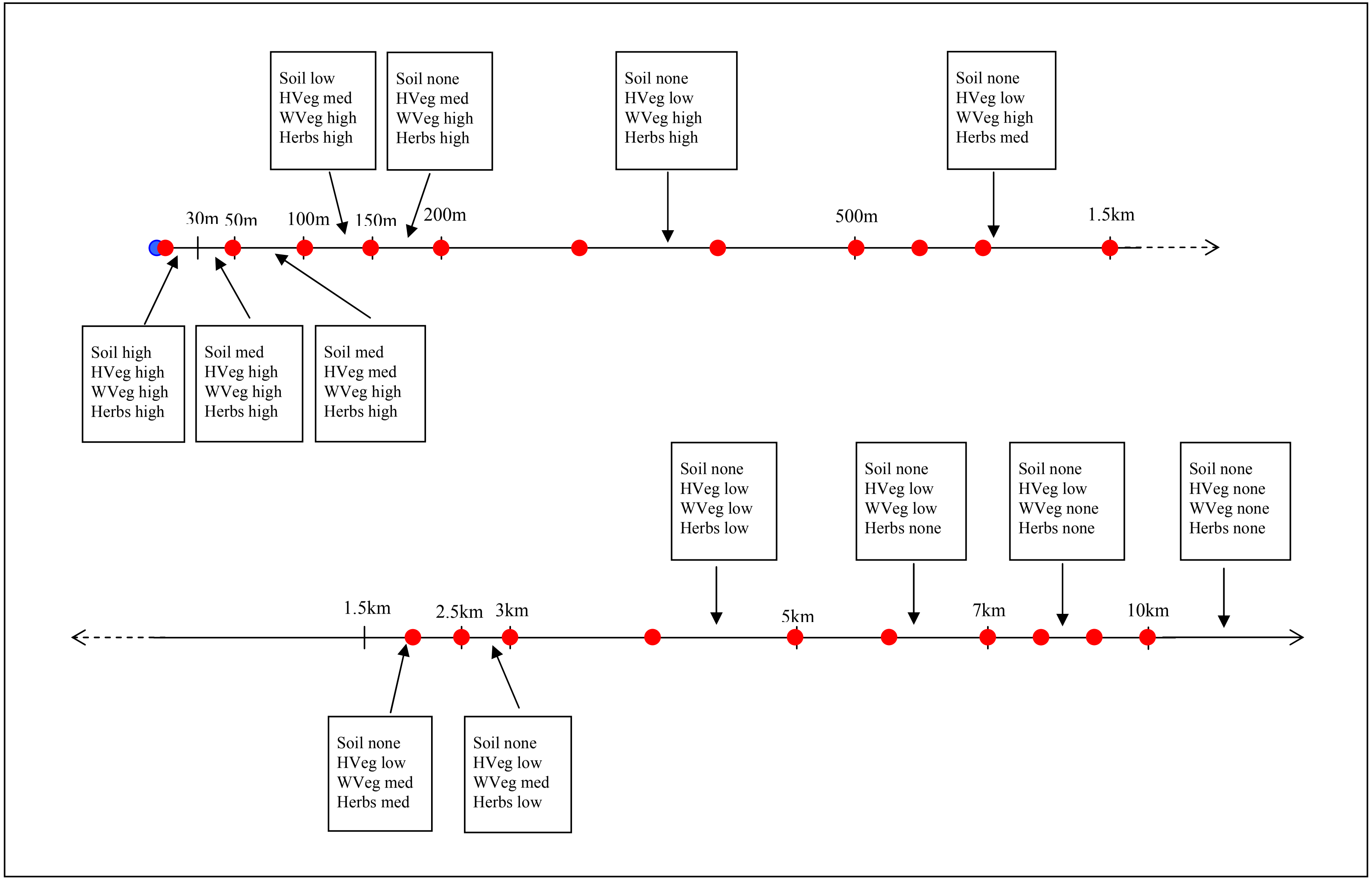

Figure 13: Illustration of layout of sampling points. (Sampling at 0m, 50m, 100m, 150m, 200m, 300m, 400m, 500m, 750m, 1km, 1.5km, 2km, 2.5km, 3km, 4km, 5km, 6km, 7km, and thereafter at 1km increments if transect length permits.). Soil = soil infiltration (Thrash 1997); HVeg = herbaceous vegetation (Thrash et al. 1991a; Thrash 1998a; Thrash & Derry 1999); WVeg = woody vegetation (Thrash et al. 1991b; Brits et al. 2000); Herbs = herbivore density (Thrash et al. 1995; Owen-Smith 1996; Redfern et al. 2003). High to none describes the impact level.

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A4 LOCATIONS OF SAMPLING SITES

Waterpoints sampled (name supplied by property and abbreviation used in this study) with distances of sampling sites from the waterpoint and GPS positions of the start of herbaceous vegetation and woody vegetation sampling zones. Reserves listed alphabetically by name

Greater Olifants River Conservancy (GRC)

Waterpoint	Abbreviation	Distance	South	East
Double Dam	DBD	0	24˚ 8' 19.9"	31˚ 4' 19.3"
Double Dam	DBD	50	24˚ 8' 20.5"	31˚ 4' 21.2"
Double Dam	DBD	100	24˚ 8' 21.3"	31˚ 4' 22.9"
Double Dam	DBD	150	24˚ 8' 21.8"	31˚ 4' 24.5"
Double Dam	DBD	200	24˚ 8' 22.4"	31˚ 4' 26.2"
Ian’s Pan	IAN	0	24˚ 9' 57.2"	31˚ 1' 28.3"
Ian’s Pan	IAN	50	24˚ 9' 55.5"	31˚ 1' 30.7"
Ian’s Pan	IAN	300	24˚ 9' 56.2"	31˚ 1' 38"
Ian’s Pan	IAN	400	24˚ 9' 56.4"	31˚ 1' 42"
Ian’s Pan	IAN	500	24˚ 9' 56.5"	31˚ 1' 45.3"
Ian’s Pan	IAN	750	24˚ 9' 56.6"	31˚ 1' 54"
Ian’s Pan	IAN	1000	24˚ 9' 57.4"	31˚ 2' 1.4"
River - near Rusermi	RVR	0	24˚ 5' 9.5"	31˚ 4' 14.7"
River - near Rusermi	RVR	50	24˚ 5' 10.5"	31˚ 4' 16.2"
River - near Rusermi	RVR	100	24˚ 5' 11.5"	31˚ 4' 17.3"
River - near Rusermi	RVR	150	24˚ 5' 12.6"	31˚ 4' 18.8"
River - near Rusermi	RVR	200	24˚ 5' 13.4"	31˚ 4' 20.4"
River - near Rusermi	RVR	300	24˚ 5' 15.5"	31˚ 4' 23.5"
River - near Rusermi	RVR	400	24˚ 5' 17.6"	31˚ 4' 25.9"
River - near Rusermi	RVR	500	24˚ 5' 19.4"	31˚ 4' 30.7"
River - near Rusermi	RVR	750	24˚ 5' 24.4"	31˚ 4' 35.7"
River - near Rusermi	RVR	1000	24˚ 5' 29.4"	31˚ 4' 42.6"
River - near Rusermi	RVR	1500	24˚ 5' 39.8"	31˚ 4' 57.5"
River - near Seekoeigat	RVS	0	24˚ 4' 15.7"	31˚ 6' 22.3"
River - near Seekoeigat	RVS	50	24˚ 4' 17.1"	31˚ 6' 22.1"
River - near Seekoeigat	RVS	100	24˚ 4' 18.9"	31˚ 6' 21.9"
River - near Seekoeigat	RVS	150	24˚ 4' 20.8"	31˚ 6' 22.1"
River - near Seekoeigat	RVS	200	24˚ 4' 22.4"	31˚ 6' 22"
River - near Seekoeigat	RVS	300	24˚ 4' 25.7"	31˚ 6' 21.8"
River - near Seekoeigat	RVS	400	24˚ 4' 29"	31˚ 6' 21.8"
River - near Seekoeigat	RVS	500	24˚ 4' 32.4"	31˚ 6' 21.7"
River - near Seekoeigat	RVS	750	24˚ 4' 40.3"	31˚ 6' 21.2"
River - near Seekoeigat	RVS	1000	24˚ 4' 48.3"	31˚ 6' 21.3"
River - near Seekoeigat	RVS	1500	24˚ 5' 4.3"	31˚ 6' 20.2"
Seekoeigat 1	SEE	0	24˚ 5' 31.7"	31˚ 6' 33.1"
Seekoeigat 1	SEE	50	24˚ 5' 33.9"	31˚ 6' 32.5"
Seekoeigat 1	SEE	100	24˚ 5' 35.3"	31˚ 6' 32.3"
Seekoeigat 1	SEE	150	24˚ 5' 36.8"	31˚ 6' 32.3"
Seekoeigat 1	SEE	200	24˚ 5' 38.6"	31˚ 6' 32.1"
Seekoeigat 1	SEE	300	24˚ 5' 41.8"	31˚ 6' 32.1"
Seekoeigat 1	SEE	400	24˚ 5' 45"	31˚ 6' 31.5"

Seekoiegat 1	SEE	500	24° 5' 48.1"	31° 6' 31.3"
Seekoeigat 1	SEE	750	24° 5' 56.1"	31° 6' 30.3"
Seekoeigat 1	SEE	1000	24° 6' 4"	31° 6' 29.5"

Klaserie Private Nature Reserve (KLA)

(Property names for waterpoints were not available.)

Waterpoint	Abbreviation	Distance	South	East
Closed pumped dam	C20	0	24° 14' 44.1"	31° 11' 33.9"
Closed pumped dam	C20	50	24° 14' 43.2"	31° 11' 33"
Closed pumped dam	C20	100	24° 14' 42"	31° 11' 31.6"
Closed pumped dam	C20	150	24° 14' 40.7"	31° 11' 29.9"
Closed pumped dam	C20	200	24° 14' 40.1"	31° 11' 28.5"
Closed pumped dam	C20	300	24° 14' 38.2"	31° 11' 25.4"
Closed pumped dam	C20	400	24° 14' 34.3"	31° 11' 20"
Closed pumped dam	C20	500	24° 14' 29.7"	31° 11' 12.9"
Closed pumped dam	C20	750	24° 14' 24.5"	31° 11' 5.5"
Closed pumped dam	C20	1000	24° 14' 14.8"	31° 10' 50.9"
Closed dam	C65	0	24° 11' 7.8"	31° 4' 26.8"
Closed dam	C65	50	24° 11' 8.3"	31° 4' 29.8"
Closed dam	C65	100	24° 11' 6.2"	31° 4' 30.4"
Closed dam	C65	150	24° 11' 5.6"	31° 4' 32.3"
Closed dam	C65	200	24° 11' 4.7"	31° 4' 34"
Closed dam	C65	300	24° 11' 3.1"	31° 4' 36.9"
Closed dam	C65	400	24° 11' 1.5"	31° 4' 40.1"
Closed dam	C65	500	24° 11' 0.1"	31° 4' 43.1"
Closed dam	C65	750	24° 10' 56.2"	31° 4' 51.1"
Closed dam	C65	1000	24° 10' 52.7"	31° 4' 58.9"
Open dam	D20	0	24° 9' 16.8"	31° 15' 7.2"
Open dam	D20	50	24° 9' 18.1"	31° 15' 7"
Open dam	D20	100	24° 9' 19.6"	31° 15' 6.9"
Open dam	D20	150	24° 9' 21.3"	31° 15' 6.9"
Open dam	D20	200	24° 9' 23"	31° 15' 6.7"
Open dam	D20	300	24° 9' 26.4"	31° 15' 6.7"
Open dam	D20	400	24° 9' 29.9"	31° 15' 6.6"
Open dam	D20	500	24° 9' 32.5"	31° 15' 7.42"
Open dam	D20	750	24° 9' 41"	31° 15' 6.7"
Open dam	D20	1000	24° 9' 49.2"	31° 15' 6.4"
Open dam	D20	1500	24° 10' 4.9"	31° 15' 7.4"
Open pumped dam	P26	0	24° 12' 5.2"	31° 14' 40.6"
Open pumped dam	P26	50	24° 12' 5"	31° 14' 39.4"
Open pumped dam	P26	100	24° 12' 5.1"	31° 14' 37.7"
Open pumped dam	P26	150	24° 12' 5.7"	31° 14' 36.1"
Open pumped dam	P26	200	24° 12' 6.5"	31° 14' 34.2"
Open pumped dam	P26	300	24° 12' 7"	31° 14' 30.6"
Open pumped dam	P26	400	24° 12' 7.8"	31° 14' 27.2"
Open pumped dam	P26	500	24° 12' 8.6"	31° 14' 23.8"
Open pumped dam	P26	750	24° 12' 10.7"	31° 14' 15"
Open pumped dam	P26	1000	24° 12' 12.7"	31° 14' 6.6"
Open pumped dam	P26	1500	24° 12' 16.9"	31° 13' 49.2"
River	R01	0	24° 4' 21.7"	31° 8' 32.5"
River	R01	50	24° 4' 23.1"	31° 8' 31.1"

River	R01	100	24° 4' 25"	31° 8' 33.7"
River	R01	150	24° 4' 27"	31° 8' 33.6"
River	R01	200	24° 4' 29.2"	31° 8' 34.2"
River	R01	300	24° 4' 32.1"	31° 8' 36.1"
River	R01	400	24° 4' 36.2"	31° 8' 37.8"
River	R01	500	24° 4' 38.9"	31° 8' 38.1"
River	R01	750	24° 4' 46.5"	31° 8' 39.6"
River	R01	1000	24° 4' 54.4"	31° 8' 31.8"
River	R01	1500	24° 5' 9.8"	31° 8' 46.8"

Kruger National Park (KNP)

Waterpoint	Abbreviation	Distance	South	East
Bull Frog	BFG	0	23° 46' 11.5"	31° 39' 0.4"
Bull Frog	BFG	100	23° 46' 9.8"	31° 38' 57.3"
Bull Frog	BFG	150	23° 46' 10"	31° 38' 55.4"
Bull Frog	BFG	200	23° 46' 9.9"	31° 38' 53.7"
Bull Frog	BFG	300	23° 46' 9.5"	31° 38' 50.2"
Bull Frog	BFG	400	23° 46' 7.4"	31° 38' 46.9"
Bull Frog	BFG	500	23° 46' 6.4"	31° 38' 43.6"
Bull Frog	BFG	750	23° 46' 3.8"	31° 38' 34.9"
Bull Frog	BFG	1000	23° 46' 1.3"	31° 38' 27"
Bull Frog	BFG	1500	23° 45' 56.3"	31° 38' 9.8"
Bvumanyun	BVU	0	22° 23' 44.5"	31° 5' 51.9"
Bvumanyun	BVU	50	22° 23' 44.3"	31° 5' 54.4"
Bvumanyun	BVU	50	22° 23' 44.7"	31° 5' 56.3"
Bvumanyun	BVU	100	22° 23' 45.1"	31° 5' 57.8"
Bvumanyun	BVU	150	22° 23' 45.6"	31° 5' 59.6"
Engelhard Dam	EGH	0	23° 50' 13.4"	31° 37' 47"
Engelhard Dam	EGH	50	23° 50' 14.8"	31° 37' 46.7"
Engelhard Dam	EGH	100	23° 50' 16.4"	31° 37' 46.2"
Engelhard Dam	EGH	150	23° 50' 17.9"	31° 37' 45.9"
Engelhard Dam	EGH	200	23° 50' 19.5"	31° 37' 45.1"
Engelhard Dam	EGH	300	23° 50' 22.8"	31° 37' 44.4"
Engelhard Dam	EGH	400	23° 50' 25.9"	31° 37' 43.3"
Engelhard Dam	EGH	500	23° 50' 28.9"	31° 37' 42.4"
Engelhard Dam	EGH	750	23° 50' 36.8"	31° 37' 40.2"
Engelhard Dam	EGH	1000	23° 50' 44.6"	31° 37' 37.6"
Engelhard Dam	EGH	1500	23° 51' 0.5"	31° 37' 33"
Engelhard Dam	EGH	2000	23° 51' 16"	31° 37' 28"
Engelhard Dam	EGH	2500	23° 51' 32"	31° 37' 23.7"
Engelhard Dam	EGH	3000	23° 51' 47.6"	31° 37' 18.5"
Eileen	ELN	0	24° 17' 8.5"	31° 29' 35.5"
Eileen	ELN	100	24° 17' 11.2"	31° 29' 37.1"
Eileen	ELN	150	24° 17' 12.8"	31° 29' 37.5"
Eileen	ELN	200	24° 17' 14.4"	31° 29' 37.9"
Red Gorton	RGN	0	24° 20' 11.8"	31° 27' 32.6"
Red Gorton	RGN	50	24° 20' 11.7"	31° 27' 30.1"
Red Gorton	RGN	100	24° 20' 11.8"	31° 27' 28.2"
Red Gorton	RGN	150	24° 20' 11.3"	31° 27' 25.6"
Red Gorton	RGN	200	24° 20' 11.4"	31° 27' 23.9"
Red Gorton	RGN	300	24° 20' 11.8"	31° 27' 20.4"

Red Gorton	RGN	400	24° 20' 12.1"	31° 27' 16.9"
Red Gorton	RGN	500	24° 20' 12.4"	31° 27' 13.2"
Red Gorton	RGN	750	24° 20' 13.3"	31° 27' 4.4"
Red Gorton	RGN	1000	24° 20' 14.1"	31° 26' 55.5"
Red Gorton	RGN	1500	24° 20' 15.5"	31° 26' 37.9"
Red Gorton	RGN	2000	24° 20' 17.5"	31° 26' 20.2"
Red Gorton	RGN	2500	24° 20' 19.1"	31° 26' 2.5"
Red Gorton	RGN	3000	24° 20' 20.9"	31° 25' 44.5"
Red Gorton	RGN	4000	24° 20' 24.7"	31° 25' 9.5"
Red Gorton	RGN	5000	24° 20' 32.1"	31° 24' 44.3"

Limpopo National Park (LNP)

Waterpoint	Abbreviation	Distance	South	East
Bona Kaya	BNK	0	23° 29' 6.6"	31° 52' 29.9"
Bona Kaya	BNK	50	23° 29' 5.2"	31° 52' 29.6"
Bona Kaya	BNK	100	23° 29' 3.6"	31° 52' 30.2"
Bona Kaya	BNK	150	23° 29' 1.8"	31° 52' 30.7"
Bona Kaya	BNK	200	23° 29' 0.2"	31° 52' 30.5"
Bona Kaya	BNK	300	23° 28' 56.8"	31° 52' 30"
Bona Kaya	BNK	400	23° 28' 53.5"	31° 52' 30"
Bona Kaya	BNK	500	23° 28' 49.9"	31° 52' 30.8"
Bona Kaya	BNK	750	23° 28' 41.2"	31° 52' 29.9"
Bona Kaya	BNK	1000	23° 28' 32.6"	31° 52' 30.1"
Bona Kaya	BNK	1500	23° 28' 15.7"	31° 52' 29.9"
Bona Kaya	BNK	2000	23° 27' 58.5"	31° 52' 29.9"
Bona Kaya	BNK	2500	23° 27' 41.4"	31° 52' 29.8"
Bona Kaya	BNK	3000	23° 27' 24.3"	31° 52' 29.6"
Bona Kaya	BNK	4000	23° 26' 50.5"	31° 52' 29.8"
Long Hippo Pool	LHP	0	23° 47' 7"	31° 47' 20.4"
Long Hippo Pool	LHP	50	23° 47' 6.4"	31° 47' 21.9"
Long Hippo Pool	LHP	100	23° 47' 5.3"	31° 47' 23.1"
Long Hippo Pool	LHP	150	23° 47' 4.9"	31° 47' 24.8"
Long Hippo Pool	LHP	200	23° 47' 4"	31° 47' 26.3"
Machampane - Camp side	MCC	0	23° 46' 9.8"	31° 46' 50.5"
Machampane - Camp side	MCC	50	23° 46' 10"	31° 46' 48.6"
Machampane - Camp side	MCC	100	23° 46' 10.2"	31° 46' 46.6"
Machampane - Camp side	MCC	150	23° 46' 10.5"	31° 46' 44.8"
Machampane	MCP	0	23° 46' 8.6"	31° 46' 50.8"
Machampane	MCP	50	23° 46' 7.7"	31° 46' 52.1"
Machampane	MCP	100	23° 46' 6.9"	31° 46' 53.2"
Machampane	MCP	150	23° 46' 5.7"	31° 46' 54.5"
Machampane	MCP	200	23° 46' 4.7"	31° 46' 55.9"
Machampane	MCP	300	23° 46' 2.7"	31° 46' 58.8"
Machampane	MCP	400	23° 46' 0.6"	31° 47' 1.4"
Machampane	MCP	500	23° 45' 58.6"	31° 47' 4.3"
Machampane	MCP	750	23° 45' 53.5"	31° 47' 11"
Machampane	MCP	1000	23° 45' 48.5"	31° 47' 18"
Machampane	MCP	1500	23° 45' 38.2"	31° 47' 31.8"
Machampane	MCP	2000	23° 45' 28.2"	31° 47' 45.5"
Machampane	MCP	2500	23° 45' 17.9"	31° 47' 59.3"
Machampane	MCP	3000	23° 45' 8"	31° 48' 13.2"

Machampane	MCP	4000	23˚ 44' 47.5"	31˚ 48' 40.7"
Machampane	MCP	5000	23˚ 44' 27.1"	31˚ 49' 8.4"
Machampane	MCP	6000	23˚ 44' 7"	31˚ 49' 35.8"
Machampane	MCP	7000	23˚ 43' 46.1"	31˚ 50' 3"
Ngwenya	NGW	0	23˚ 24' 16.3"	31˚ 40' 46"
Ngwenya	NGW	50	23˚ 24' 14.7"	31˚ 40' 46.3"
Ngwenya	NGW	100	23˚ 24' 13.2"	31˚ 40' 46.5"
Ngwenya	NGW	150	23˚ 24' 11.1"	31˚ 40' 47.2"
Ngwenya	NGW	200	23˚ 24' 9.8"	31˚ 40' 47.6"
Ngwenya	NGW	300	23˚ 24' 6.5"	31˚ 40' 48.5"
Ngwenya	NGW	400	23˚ 24' 3.3"	31˚ 40' 49.4"
Ngwenya	NGW	500	23˚ 23' 59.7"	31˚ 40' 50.3"
Ngwenya	NGW	750	23˚ 23' 51.5"	31˚ 40' 52.9"
Ngwenya	NGW	1000	23˚ 23' 43.2"	31˚ 40' 54.8"

Mohlabetsi Association of Landowners (MAL)

Waterpoint	Abbreviation	Distance	South	East
Jejane	JEJ	0	24˚ 17' 22.2"	31˚ 0' 37.3"
Jejane	JEJ	50	24˚ 17' 21.5"	31˚ 0' 35.5"
Jejane	JEJ	100	24˚ 17' 22.8"	31˚ 0' 34.4"
Jejane	JEJ	150	24˚ 17' 23.1"	31˚ 0' 31.8"
Jejane	JEJ	200	24˚ 17' 23.3"	31˚ 0' 31.1"
Jejane	JEJ	300	24˚ 17' 24"	31˚ 0' 26.6"
Jejane	JEJ	400	24˚ 17' 24.8"	31˚ 0' 23.1"
Jejane	JEJ	500	24˚ 17' 25.1"	31˚ 0' 19.5"
Jejane	JEJ	750	24˚ 17' 26.6"	31˚ 0' 10.8"
Jejane	JEJ	1000	24˚ 17' 27.9"	31˚ 0' 2.2"
Pusa Manzi	PUM	0	24˚ 15' 17.1"	31˚ 1' 38.5"
Pusa Manzi	PUM	50	24˚ 15' 18.5"	31˚ 1' 40"
Pusa Manzi	PUM	100	24˚ 15' 19.5"	31˚ 1' 41"
Pusa Manzi	PUM	150	24˚ 15' 20.8"	31˚ 1' 42.1"
Pusa Manzi	PUM	200	24˚ 15' 22.5"	31˚ 1' 42.9"
Pusa Manzi	PUM	300	24˚ 15' 26.1"	31˚ 1' 43.6"
Pusa Manzi	PUM	400	24˚ 15' 29.4"	31˚ 1' 44.8"
Pusa Manzi	PUM	500	24˚ 15' 33.1"	31˚ 1' 45.8"
Pusa Manzi	PUM	750	24˚ 15' 41"	31˚ 1' 49.2"
Pusa Manzi	PUM	1000	24˚ 15' 49.4"	31˚ 1' 52.6"

A5 DETERMINATION OF AN APPROPRIATE SAMPLE SIZE FOR HERBACEOUS VEGETATION

Piosphere studies typically use point sampling to assess herbaceous vegetation. There is great discrepancy between the sample sizes used for grass surveys in previous piosphere assessments and those used in vegetation monitoring. Piosphere assessment work uses 12 to 50 points whilst vegetation monitoring tends to use 100 points. In order to detect differences between zones of a piosphere it is important that vegetation at each sampling site is correctly represented by the sampling technique. Monitoring data from the ARC Savanna Ecosystem Dynamics' annual vegetation monitoring in the private reserves was analysed to determine an appropriate sample size of points.

Checking 100 Point Sampling

Analyses were restricted to one reserve over multiple years to provide a variation in species composition similar to what would be encountered during a piosphere assessment. In order to check that 100 point sampling was sufficient, summary plant species data spanning the years 1992 to 2002 from 23 monitoring plots within one reserve were analysed using species richness estimators in EstimateS (Colwell, 2001). The species accumulation curve (Figure 1) reaches an asymptote indicating that sampling intensity is sufficient. Species richness estimators (Figure 2) generated figures that were close to the actual number of species found (49) with a range of 48.21 (MMRuns) to 53.98 (Jack 2). Best estimators were found to be the abundance based estimators ACE (49.83 species) and Chao 1 (50 species).

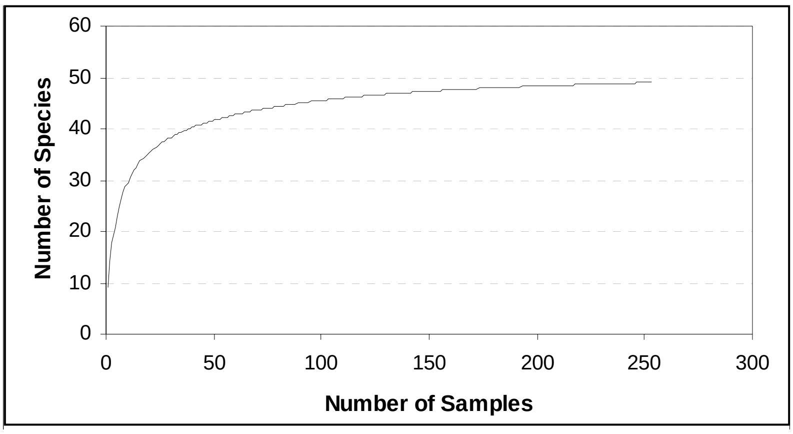

Figure 1: Species accumulation curve based on 253 monitoring plots with 100 sampling points each from one reserve. Total sampled species was 49.

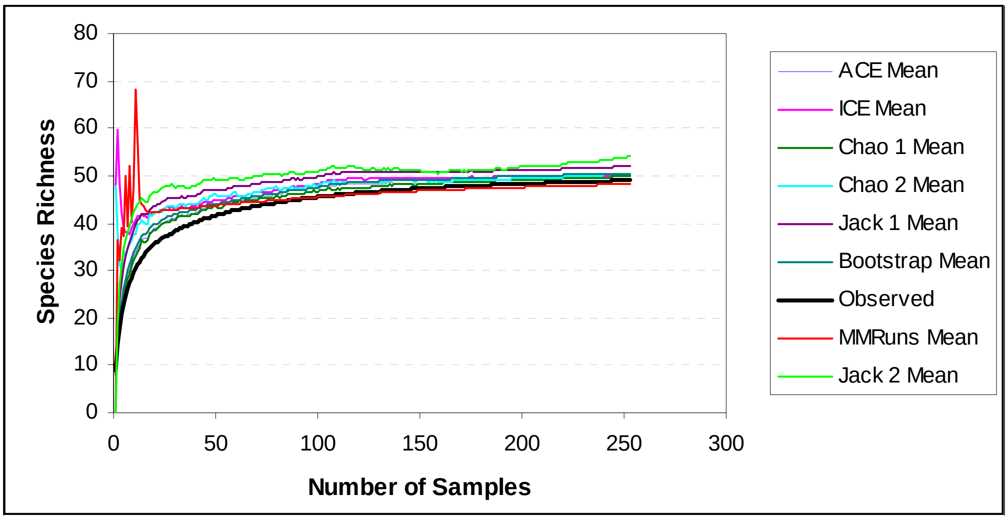

Figure 2: Species richness estimators based on 253 monitoring plots with 100 sampling points each from one reserve. Total species found was 49. Best estimators were shown to be ACE and Chao 1.

Investigation of Different Sample Sizes

In order to test different sample sizes, detailed 100 point data from 13 monitoring plots from one reserve in one year were analysed. Different sample sizes were created using random numbers to choose individual records. Sample sizes of 50, 75 and 80 points were assessed. 50 points represents the largest sample size previously used in piosphere assessments. 75 points represents a 25% reduction from the 100 point sampling used in annual vegetation monitoring. 80 points represents the smallest reduction of sampling points that would be considered. Species accumulation (Figure 3), species richness (Figure 4), species diversity (Figure 5) and evenness (Figure 6) were calculated in EstimateS and graphed in Excel. These measures indicate how well the particular level of sampling effort represents the plant species in an area.

When the species accumulation curve becomes asymptotic it indicates that no further new species are expected. The species accumulation curve did not show an asymptotic value for any sample size (Figure 3). The number of species sampled was 35. The lack of asymptote for even the 100 point sample is likely to be due to the smaller sample size (13 monitoring plots) than was used when checking the 100 point data (Figure 1).

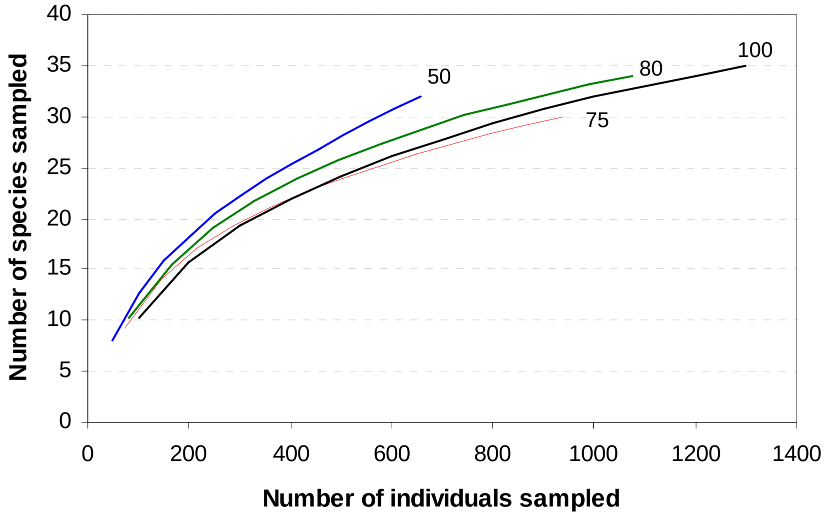

Figure 3: Species accumulation curves for varying sample sizes. Curves calculated from 13 monitoring plots with a maximum of 100 sampling points from one reserve. Total species found was 35.

Species richness estimators are based on the presence of rare species, areas with more rare species will have more of an increase in species encountered with increasing sample size. Only curves for ACE and Chao 1 are shown as these were determined to be the most informative (Figure 4). Species richness curves based on ACE show sample sizes of 75 and larger approach asymptotes. 50 points was shown to be insufficient. A sample size of 50 has more singletons which is why ACE predicts more species overall. Species richness curves based on Chao 1 appear to be approaching asymptotes for sample sizes of 75 and larger (Figure 4). 50 points was again shown to be insufficient. This estimator suggests that 75 points would be a sufficient sampling size for grasses.

Species diversity curves suggest smaller sample sizes are sufficient as diversity values for the area are low. Graphs of diversity based on the Shannon Index show all sample sizes approach asymptotic values (Figure 5). Diversity curves based on Simpson's index also show tendencies towards asymptotic values for sample sizes greater than 75 while 50 points were shown to be insufficient (Figure 5).

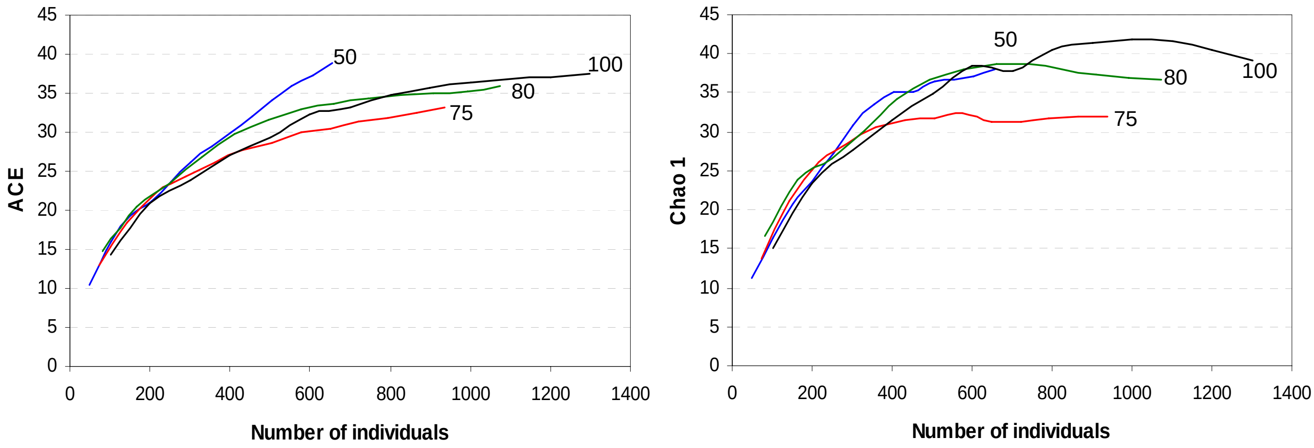

Figure 4: Species richness curves based on ACE and Chao 1 for varying sample sizes. Curves calculated from 13 monitoring plots with a maximum of 100 sampling points. Total species found was 35.

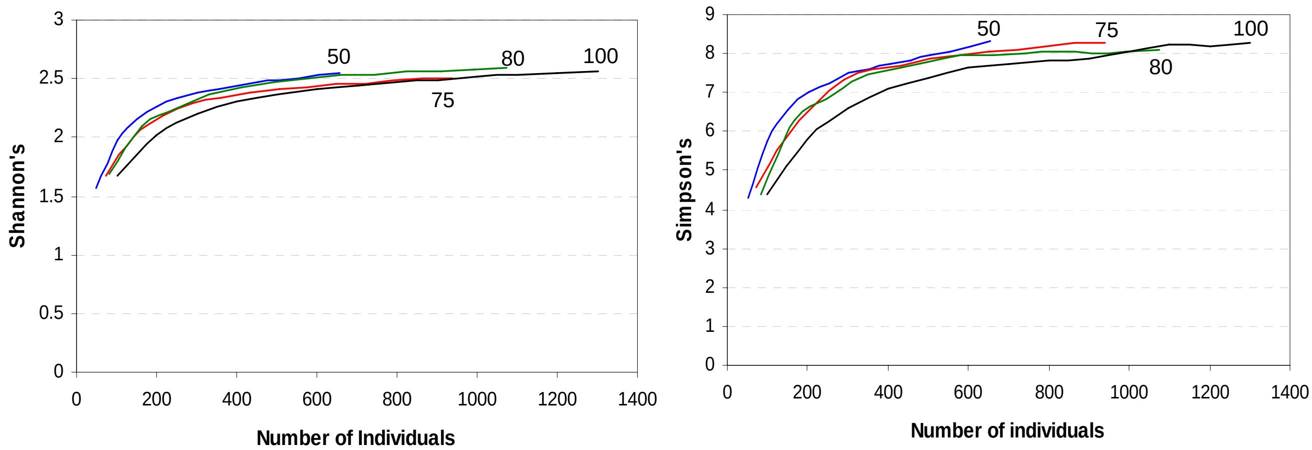

Figure 5: Species diversity curves based on the Shannon and Simpson's Indices for varying sample sizes. Curves calculated from 13 monitoring plots with a maximum of 100 sampling points from one reserve.

Evenness reaches asymptotic values, indicating a good trend (Figure 6). There is some dominance, particularly in the smaller sample sizes. This is because the 100 point data was sub-sampled to get the smaller sample sizes. Based on evenness, all sample sizes are sufficient to pick up dominance in the vegetation. However, representation of reality improves with larger sample sizes. There is an improvement between sample sizes of 75 and 80.

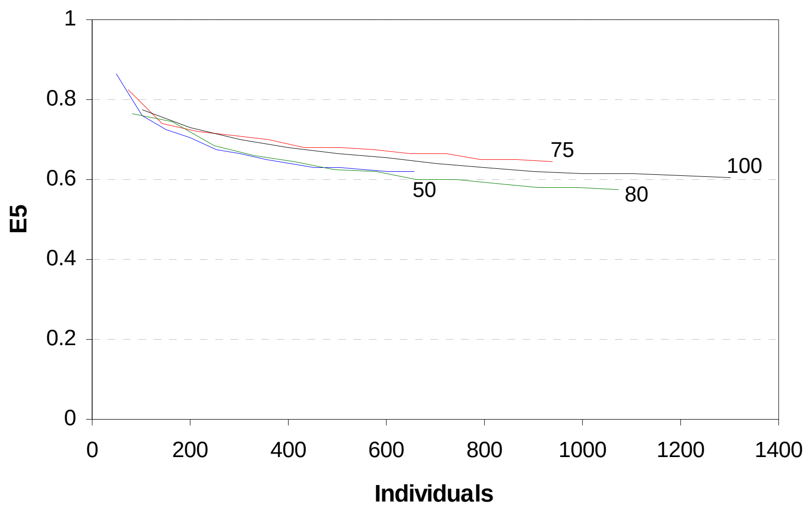

Figure 6: Evenness values (based on Shannon and Simpson's diversity indices) for varying sample sizes. Curves calculated from 13 monitoring plots with a maximum of 100 sampling points from one reserve.

Final Decision

A sample size of 80 points is to be used for each sampling site. Although 50 points would be faster, this does not give a reliable representation of the vegetation. The difference in effort between 75 and 80 sampling points is insubstantial when compared with the increase in accuracy.

The method used for point sampling of grasses will be a distance-to-tuft approach. At each point, the distance to the nearest grass tuft within 50cm will be measured. No differentiation will be made between annual and perennial grasses. If there is no grass tuft within a 50cm radius of the point, either "forb", "sedge" or "bare ground" will be recorded depending on whether there is other herbaceous vegetation present. Use of a 50cm radius prevents re-sampling of plants as data will be recorded on a 15m by 4m grid with a cell size of 1m by 1m.

References

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A6 DETERMINATION OF AN APPROPRIATE PLOT SIZE FOR WOODY VEGETATION

There is great variation between sizes of sampling areas of woody vegetation studies. Previous piosphere studies of woody vegetation have used areas of over $400m^2$ whereas current monitoring approaches in the study area use much smaller areas ($200m^2$ in the private reserves and 24 to $632m^2$ in Kruger National Park). Data was not available for analysis to determine the appropriate sampling area so a decision was made based on sampling areas used in published studies and current monitoring.

Published Sampling Areas

Sampling areas for woody vegetation were extracted from literature collected as part of the literature review for the project. Published sample areas varied from $1m^2$ for seedling studies (Mlambo & Nyathi, 2004) to 10 $000m^2$ for a resprouting study on a single species (Mlambo & Mapaure, 2006). A general trend was that a smaller sample area was required for smaller individuals. 50% of the studies had sample sizes between 200 and $600m^2$ with a slightly higher representation in the 200 to $400m^2$ class (Figure 1). Three studies had sample sizes of $1000m^2$ and higher including one which used a grid approach within the sampling area (Thrash *et al.*, 1991).

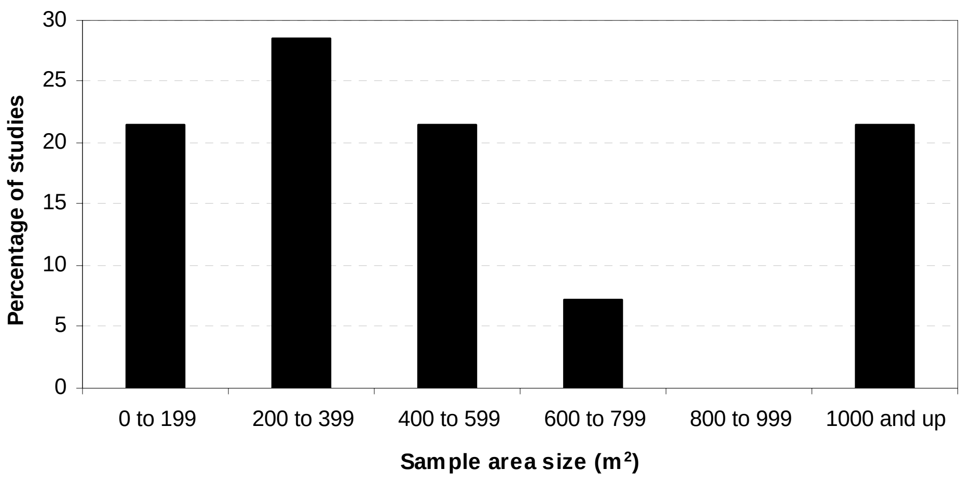

Figure 1: Woody plant sampling areas used in published studies

Limitations to Sampling Area

This project did not aim to characterise the savanna piosphere for the first time, but rather to assess the applicability of piospheres under different conservation management regimes. A belt plot approach was used and sampling efficiency was important. Interval sampling was used to increase efficiency with the smallest interval size equal to 50m. The belt plot can therefore not exceed 40m in length to ensure that sampling is within a zone.

Suggested Sampling Areas

Herbaceous sampling is based on a 4 x 15m grid with the long axis oriented along the direction of the main transect. It makes most sense to sample woody vegetation in a similar area. The herbaceous grid only covers $60m^2$ which in comparison to published areas could be considered insufficient for woody vegetation. It was most practical to add 1m or 2m to either side of the grid and to extend the area sampled for woody vegetation. Table 1 gives possible areas that could be covered using this approach.

Table 1: Potential sampling areas based on the herbaceous sampling grid.

Description	**Woody transect dimensions**	**Woody sampling area (m^2)**
Original herbaceous grid	4m x 20m	80
(1) Add 1m either side of grid	6m x 20m	120
(2) Add 2m either side of grid	8m x 20m	160
(3) Add 15m to end of grid	4m x 30m	120
(4) Add 25m to end of grid	4m x 40m	160
(4) Combine (1) and (3)	6m x 30m	180
(5) Combine (1) and (4)	6m x 40m	240
(6) Combine (2) and (3)	8m x 30m	240
(7) Combine (2) and (4)	8m x 40m	320

Final Decision

At each sampling site along the waterpoint transect, woody plant individuals greater than 1m in height will be sampled in a belt plot of 8m x 30m ($240m^2$). This gives a sampling design as shown in Figure 2. A 6m x 40m belt plot would give the same sampling area but has a higher risk of entering the next piosphere zone.

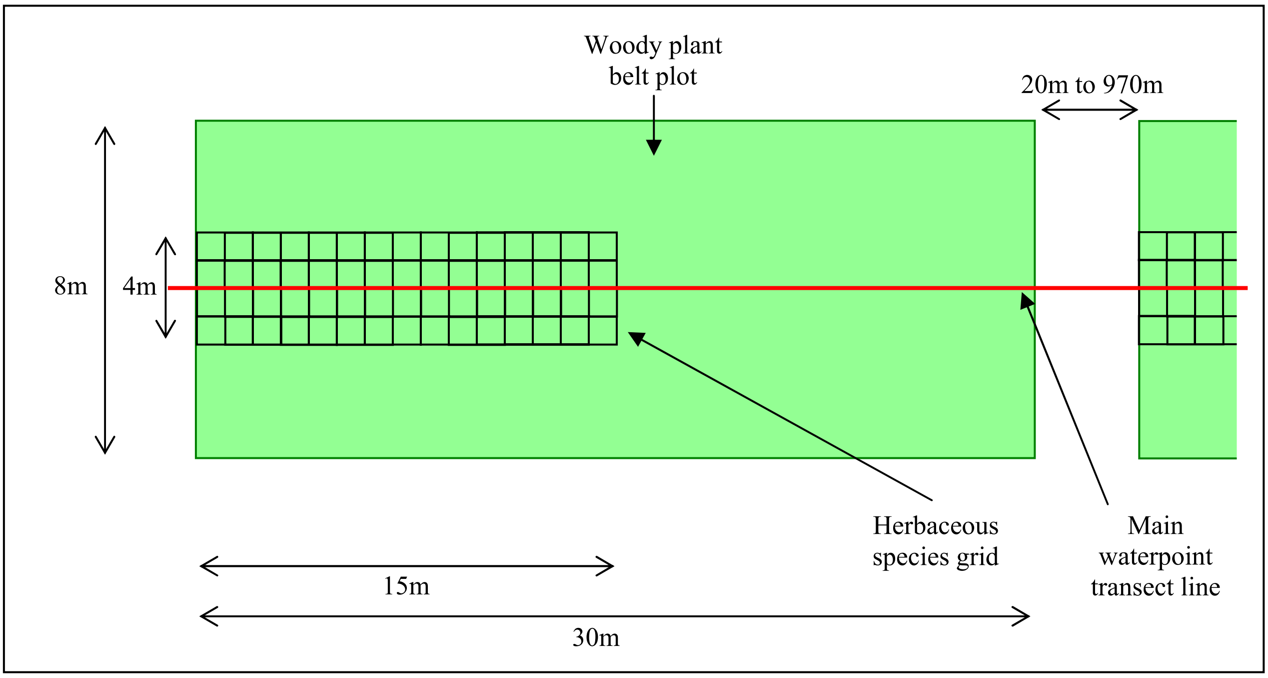

Figure 2: Diagrammatic representation of the woody plant sampling design for one sampling site along the piosphere transect.

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A7 LANDSCAPE FUNCTION ANALYSIS (LFA) METHOD

This appendix is intended to be a summarised description of the LFA process, for full details on how to perform an LFA, please see Tongway, D. & Hindley, N. (2004) Landscape Function Analysis: Procedures for Monitoring and Assessing Landscapes. CSIRO Sustainable Ecosystems, Canberra.

LFA assesses landscape function through a series of measurements of landscape organisation and indicators that relate to stability, infiltration and nutrient cycling. LFA transects must always run in the direction of resource flow. They are therefore termed *gradsects* as they are transects following a gradient and as such have a particular set of constraints (Gillison & Brewer, 1985). After preliminary work with D. Tongway it was determined that a standardised 20m length gradsect would be appropriate for this study.

Landscape organisation

Landscape organisation is the relative amounts of patches and inter-patches (in this document, collectively called 'zones'). Patches are areas of the landscape where nutrients are accumulated and retained and inter-patches are areas of the landscape where nutrients are lost. Examples of patches are: grass tufts, litter, shrubs. Inter-patches tend to be formed of bare soil. Five replicates of each patch/inter-patch type along a gradsect are recommended for statistical reliability.

Once the gradsect line has been established with a tape measure, the patches and inter-patches are measured as per a line intercept method. Additionally, the width of patches perpendicular to the gradsect line (up to a maximum of 5m either side of the gradsect line) is recorded. Measurements along the tape measure should be made to the nearest cm. Patches or interpatches that constitute less than 5% of the gradsect are excluded from the next stage.

An important discussion point in patch description is the presence of 'grass swards'. A grass sward can be defined as an area where grass plants are so dense that there is no longer resource movement between them. This study distinguished between 'sparse grass' and 'grass' patches. 'Grass' patches were either individual tussocks or very dense swards of grass, with generally less than 5cm between plants. 'Sparse grass' patches had more than 5cm between plants but still operated as swards with very little to no movement of litter or soil between plants.

Zone characterisation

Eleven indicators are assessed for each zone type (all patch and inter-patch types). The LFA manual refers to this stage as Soil Surface Assessment. Five query zones are located in each zone type. Ideally, query zones are 1m in length and each in a different piece of the zone type along the full length of the gradsect. If the extent of the zone type does not allow for this, query zones can be shortened and/or repeated in a section of the zone type.

Indicators were carefully chosen and tested for reliable relationships with infiltration, stability and nutrient cycling. It is important to follow the instructions precisely, especially when it comes to variables that should be excluded/included from measurement of particular indicators.

1. Rainsplash protection

This assesses the degree to which physical factors (e.g. rocks) and perennial plants protect the soil surface from rain drops. Plant material has to be below 50cm in order to protect the soil. Above this height, gravity drops made up of combined raindrops cause more impact than raindrops themselves. Bear in mind that this protection is from 2.5mm diameter raindrops.

2. Perennial vegetation cover

This indicator infers the belowground biomass of plants and their contribution to nutrient cycling and infiltration. Grass belowground biomass is taken as equal to the basal cover on the surface. Tree and shrub belowground biomass is taken as equal to their canopy cover.

3. Litter

Litter contributes to the nutrient pool and also protects the soil surface from rainfall. Litter consists of detached plant material and annual grasses and forbs (even if they are still growing). The first step in assessing litter is to determine the level of cover. If there is more than 10% cover of litter, where it is from and the degree to which it is incorporated into the soil are also assessed.

4. Crust brokenness

This investigates the physical crust on the surface of soil as it tells what soil material is loose and available for erosion. If the soil type or location is such that a physical crust would not form, this indicator is recorded as zero (this removes the variable from analysis in the spreadsheet). Otherwise, the degree to which the crust is broken and curling is recorded.

5. Cryptogram Cover

This indicator assesses visible cryptograms (algae, fungi, lichens, mosses and liverworts) on the soil surface. If there is no habitat for cryptograms (e.g. the soil surface is very unstable) this indicator is recorded as zero. Otherwise, the cover of cryptograms is estimated. If crust brokenness is recorded as zero, cryptogram cover is automatically also zero.

6. Soil Erosion Type and Severity

Soil erosion looks at the loss of soil material from the zone types, indicating both the extent and the severity of erosion. Full descriptions and photos of different types of erosion are given in the LFA manual.

7. Deposited Materials

Deposited materials determines the amount of alluvium (silt, sands and gravel) that has been transported onto the zone. It is important to consider the volume of the deposited materials, not just the surface cover.

8. Soil Surface Roughness

This measures the microtopography of the zone to indicate how easily resources are lost by movement over the soil surface. Consider both physical roughness (e.g. rocks sticking up and dents into the soil surface) and biological roughness (e.g. perennial grass butts). To help with assessing the height of roughness, look at the depth of resource accumulation such as littler and silt deposition around perennial grass tufts.

9. Surface Nature

Surface nature assesses how easily the soil surface can be broken to yield soil for erosion. It is very important that this is assessed on a dry crust.

10. Slake Test

This tests the stability of the soil when it is rapidly wetted. A 1cm cube of soil is used from the soil surface and it is very important that this soil is dry. If the soil does not form cubes (e.g. very sandy) then this indicator is recorded as zero. Wet soil can be collected, air dried and tested later.

11. Texture

Assessing the texture of the soil indicates its permeability. This only needs to be tested once for each zone type. A pedologist's moist bolus step is used with the results simplified into a four point scale.

Data analysis

Once data has been collected it is entered into an analysis spreadsheet available from David Tongway (david.tongway@csiro.au). One spreadsheet is filled for each gradsect performed. Calculations are built into the spreadsheet with results generated immediately. Results are split into landscape organisation and landscape function. Landscape organisation investigates the patch and inter-patch data. Landscape function uses the indicators to assess stability, infiltration and nutrient cycling. The relative contribution of each patch and inter-patch type to the three functions across the landscape (represented by the gradsect) is also assessed.

References

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A8 HERBACEOUS SPECIES

Herbaceous species identified in the study area and the properties on which they occurred. LNP = Limpopo National Park, KNP = Kruger National Park, KLA = Klaserie Private Nature Reserve, GRC = Greater Olifants River Conservancy, MAL = Mohlabetsi Association of Landowners. Species are ordered alphabetically by name. Abbreviation refers to the code for the species used in this project during data collection and analysis. Perenniality and grazing value were taken from van Wyk, E. & van Oudtshoorn, F. 2004. *Guide to Grasses of Southern Africa*. Briza Publications, Pretoria.

Species	Abbreviation	Perenniality	Grazing Value	MAL	GRC	KLA	KNP	LNP
Andropogon chinensis	andchi	Perennial	Medium			1		1
Andropogon gayanus	andgay	Perennial	High			1		1
Aristida adscensionis	aads	Annual	Low	1	1	1	1	1
Aristida congesta subsp *barbicollis*	abarb	Weak Perennial	Low	1	1	1	1	1
Aristida congesta subsp *congesta*	acon	Weak Perennial	Low	1	1	1	1	1
Aristida junciformis	ajunc	Perennial	Low	1	1	1		1
Aristida stipitata	astip	Weak Perennial	Low	1	1	1	1	1
Bothriochloa insculpta	botins	Weak Perennial	Medium		1		1	1
Bothriochloa radicans	botrad	Perennial	Low	1	1	1	1	1
Brachiaria deflexa	bdef	Annual	Medium	1	1	1	1	1
Brachiaria nigropedata	bracnig	Perennial	High			1	1	1
Cenchrus ciliaris	cencil	Perennial	High		1	1	1	1
Chloris roxburgiana	chlrox	Weak Perennial	Medium					1
Chloris virgata	chlvir	Weak Perennial	Medium	1		1	1	1
Cymbopogon excavatus	cymexc	Perennial	Low					1
Cymbopogon plurinodis	cymplu	Perennial	Low		1		1	
Cynodon dactylon	cyndac	Perennial	High		1	1	1	
Dactyloctenium aegyptium	dacaeg	Annual	Medium	1	1		1	
Dactyloctenium giganteum	dacgig	Annual	High			1	1	1
Digitaria eriantha	digi	Perennial	High	1	1	1	1	1
Echinochloa colona	echcol	Annual	High		1		1	
Echinochloa pyramidalis	echpyr	Perennial	High					1
Enneapogon cenchroides	ecen	Weak Perennial	Medium	1	1	1	1	1
Enneapogon scoparius	esco	Perennial	Low	1	1		1	
Enteropogon macrostachyus	entmac	Perennial	High		1			
Eragrostis aspera	easp	Annual	Low		1			1
Eragrostis heteromera	ehet	Perennial	Medium			1		1
Eragrostis inamoena	eina	Perennial	Low					1

Species	**Abbreviation**	**Perenniality**	**Grazing Value**	**MAL**	**GRC**	**KLA**	**KNP**	**LNP**
Eragrostis lehmanniana	elehm	Perennial	Medium	1	1	1	1	1
Eragrostis rigidior	erig	Weak Perennial	Medium	1	1	1	1	1
Eragrostis rotifer	erot	Perennial	Medium					1
Eragrostis superba	esup	Weak Perennial	Medium	1	1	1	1	1
Eragrostis trichophora	etri	Weak Perennial	Medium		1	1	1	1
Eriochloa meyeriana	erimey	Perennial	High				1	
Fingerhutia africana	finafr	Perennial	Medium				1	1
Forbs (all)	forb	n/a	n/a	1	1	1	1	1
Heteropogon contortus	hetcon	Perennial	Medium	1	1	1	1	1
Melinis repens	melrep	Weak Perennial	Low	1	1	1	1	1
Michrochloa caffra	miccaf	Perennial	Low	1	1		1	
Miscanthus junceus	misjun	Perennial	Low			1		
Oropetium capense	orocap	Perennial	Low	1	1	1		1
Panicum coloratum	pcol	Perennial	High			1	1	1
Panicum deustum	pdeus	Perennial	High				1	1
Panicum maximum	pmax	Perennial	High	1	1	1	1	1
Paspalum distichum	pasdis	Perennial	High			1		
Paspalum notatum	pasnot	Perennial	Medium			1		
Perotis patens	ppat	Weak Perennial	Low					1
Pogonarthria squarrosa	pogsq	Weak Perennial	Low	1	1	1	1	1
Schmidtia pappophoroides	schm	Perennial	High	1	1	1	1	1
Sedges (all)	sedge	n/a	n/a		1	1	1	1
Setaria sagittifolia	setsag	Annual	Medium		1			
Setaria spacelata var. *sericea*	setser	Perennial	High	1	1			
Sporobolus fimbriatus	spfim	Perennial	High		1		1	1
Sporobolus ioclados	spioc	Weak Perennial	Medium			1	1	1
Sporobolus nitens	spnit	Weak Perennial	Low	1	1	1	1	
Themeda triandra	ttri	Perennial	High		1	1	1	1

Species	**Abbreviation**	**Perenniality**	**Grazing Value**	**MAL**	**GRC**	**KLA**	**KNP**	**LNP**
Tragus berteronianus	traber	Annual	Low	1	1	1	1	
Tricholaena monachne	trimon	Weak Perennial	Medium	1	1	1	1	1
Unidentified a	gun_a	n/a	n/a	1				1
Unidentified b	gun_b	n/a	n/a	1				
Unidentified c	gun_c	n/a	n/a	1				
Unidentified d	gun_d	n/a	n/a		1			1
Unidentified e	gun_e	n/a	n/a		1			
Unidentified f	gun_f	n/a	n/a				1	
Unidentified g	gun_g	n/a	n/a				1	
Unidentified h	gun_h	n/a	n/a				1	
Unidentified i	gun_i	n/a	n/a				1	
Unidentified j	gun_j	n/a	n/a				1	
Unidentified k	gun_k	n/a	n/a				1	
Unidentified l	gun_l	n/a	n/a					1
Unidentified m	gun_m	n/a	n/a					1
Unidentified n	gun_n	n/a	n/a					1
Unidentified q	gun_q	n/a	n/a					1
Unidentified r	gun_r	n/a	n/a		1			
Unidentified s	gun_s	n/a	n/a		1			
Urochloa mosambicensis	umos	Weak Perennial	High	1	1	1	1	1
Urochloa oligotricha	uoli	Perennial	High					1
Urochloa panicoides	upan	Annual	Low	1		1		

A9 WOODY SPECIES

Woody species identified in the study area and the properties on which they occurred. LNP = Limpopo National Park, KNP = Kruger National Park, KLA = Klaserie Private Nature Reserve, GRC = Greater Olifants River Conservancy, MAL = Mohlabetsi Association of Landowners. Species are ordered alphabetically by name. Abbreviation refers to the code for the species used in this project during data collection and analysis.

Species	Abbreviation	LNP	KNP	KLA	GRC	MAL
Acacia burkei	aburk		1	1		1
Acacia erubescens	aerub	1	1	1	1	1
Acacia exuvialis	aexuv	1	1	1		1
Acacia gerrardii	agerr		1	1		
Acacia grandicornuta	agrand		1	1		
Acacia karroo	akar		1	1	1	
Acacia nigrescens	anig	1	1	1	1	1
Acacia nilotica	anilo		1			
Acacia robusta	arob	1		1	1	1
Acacia schweinfurthii	asch	1				
Acacia senegal	asen				1	
Acacia tortilis	atort	1	1			1
Acacia xanthophloea	axan	1				
Albizia anthelmintica	albant	1				
Albizia harveyi	albhar	1	1	1	1	1
Albizia petersiana	albpet	1				
Anisotes formosissimus	aniform	1				
Anisotes rogersii	anirog	1				
Azima tetracantha	azima	1				
Berchemia zeyheri	berzey			1	1	
Bolusanthus speciosus	bolspec		1			1
Boscia albitrunca	bosalb	1	1	1	1	1
Boscia angustifolia	bosang	1	1			
Boscia foetida	bosfoet	1	1		1	1
Boscia mossambicensis	bosmos	1				
Bridelia cathartica	bricat	1				
Bridelia mollis	brimol	1	1			
Capparis separia	capsep	1				
Capparis tomentosa	captom	1				
Carissa bispinosa	caribi		1			
Cassia abbreviata	casabb	1	1	1	1	
Cissus cactiformis	ciscac		1		1	
Cissus cornifolia	ciscor	1	1	1	1	1
Colophospermum mopane	colmop	1	1	1	1	
Combretum apiculatum	capic	1	1	1	1	1
Combretum hereroense	cher	1	1	1	1	
Combretum imberbe	cimb	1	1	1	1	1
Combretum microphyllum	cmic	1				
Combretum mossambicense	cmoss	1	1		1	1
Combretum zeyheri	czey	1				
Commiphora africana	comafr	1	1	1	1	1
Commiphora glandulosa	comgla	1				

Species	Abbreviation	LNP	KNP	KLA	GRC	MAL
Commiphora harveyi	comhar					1
Commiphora mollis	commol	1	1	1	1	1
Commiphora neglecta	comneg					1
Commiphora pyracanthoides	compyr		1	1	1	1
Commiphora schimperi	comsch	1				
Cordia ovalis	corova	1			1	1
Crotalaria monteiroi	cromon	1				
Croton megalobotrys	cromeg	1			1	
Dalbergia melanoxylon	dalbmel	1	1	1	1	1
Dalbergia nitidula	dalnit	1				
Dichrostachys cinerea	diccin	1	1	1	1	1
Diospyros lycioides	diolyc					1
Diospyros mespiliformis	diomes	1	1			
Dovyalis caffra	dovcaf		1			
Ehretia amoena	ehram	1	1		1	1
Ehretia obtusifolia	ehrobt	1	1		1	
Elaeodendron transvaalense	elatra		1			
Elephantorrhiza goetzei	elegoe	1				
Erythrococca menyharthii	erymen		1			
Euclea crispa	ecrisp			1	1	1
Euclea divinorum	ediv	1	1	1	1	1
Euclea natalensis	enat		1		1	
Euclea schimperi	eschi	1				
Faidherbia albida	faidalb	1				
Ficus abutilifolia	fabut				1	
Flueggea virosa	fleuvir	1	1	1	1	
Galpinia transvaalensis	galtra	1				
Gardenia resiniflua	garres	1				
Gardenia volkensii	garvol	1			1	1
Gossypium herbaceum	gosher	1			1	1
Grewia bicolor	gbic	1	1	1	1	1
Grewia flava	gflava	1	1	1	1	1
Grewia flavescens	gflaves	1	1	1	1	
Grewia hexamita	ghex	1		1	1	1
Grewia inaequilatera	gina	1				
Grewia monitcola	gmonti	1	1	1	1	1
Grewia occidentalis	gocc	1				
Grewia sulcata	gsul	1			1	
Grewia villosa	gvill		1	1	1	1
Gymnosporia buxifolia	gymbux	1	1	1	1	1
Gymnosporia glaucophylla	gymglau				1	
Gymnosporia senegalensis	gymsen	1	1	1	1	
Hippobromus pauciflorus	hippau		1			
Hippocratea crenata	hipcre	1				
Hippocratea longipetiolata	hiplon	1			1	
Hymenodictyon parvifolium	hympar	1				
Jasminum fluminense	jasflu	1				
Jasminum multipartitum	jasmul			1		
Jasminum stenolobum	jasste	1	1		1	
Kirkia acuminata	kirkac			1		
Lannea schweinfurthii	lanschw	1	1	1	1	1

Species	Abbreviation	LNP	KNP	KLA	GRC	MAL
Maclura africana	macafr	1		1		
Maerua decumbens	maedec	1				
Manilkara mochisia	manmoc				1	
Mundulea sericea	munser	1	1			1
Ochna inermis	ochine				1	
Ochna pulchra	ochpul			1		
Ormocarpum trichocarpum	ormtri	1	1	1	1	1
Ozoroa species A	ospa	1	1	1	1	1
Pappea capensis	papcap	1			1	
Peltophorum africanum	pelafr		1	1	1	
Philonoptera violacea	phivio	1	1	1	1	1
Phyllanthus reticulatus	phyret		1			
Plectroniella armata	plearm	1				
Pouzolzia mixta	poumix	1				
Rhigozum zambesiacum	rhizam	1			1	
Rhoicissus revoilii	rhorev	1				
Rhus gueinzii	rhugue	1				
Rhus transvaalensis	rhutra		1			
Ricinis communis	riccom	1				
Salvadora australis	salaus	1				
Schotia brachypetala	schotia	1	1		1	
Schotia capitata	schocap	1	1			
Sclerocarya birrea	sclbir	1	1	1	1	1
Spirostachys africana	spiafr	1		1	1	
Sterculia rogersii	sterog				1	
Strychnos decussata	strydec	1				
Strychnos madagascariensis	strymad	1	1			1
Strychnos spinosa	stryspi		1			
Teclea pilosa	tecpil	1				
Terminalia prunioides	terpru		1	1	1	1
Terminalia sericea	terser	1				1
Thilachium africanum	thiafr	1	1			
Triapsis glaucophylla	trigla				1	
Turraea obtusifolia	turobt	1	1			
Unidentified A	wun_a				1	
Unidentified B	wun_b				1	
Unidentified C	wun_c				1	
Unidentified D	wun_d				1	
Unidentified E	wun_e				1	
Unidentified F	wun_f			1		
Unidentified G	wun_g			1		
Unidentified H	wun_h		1			
Unidentified I	wun_i		1			
Unidentified J	wun_j	1				
Unidentified K	wun_k	1				
Unidentified L	wun_l	1				
Unidentified M	wun_m	1				
Unidentified N	wun_n	1				
Unidentified O	wun_o	1				
Unidentified P	wun_p	1				
Unidentified Q	wun_q	1				

Species	Abbreviation	LNP	KNP	KLA	GRC	MAL
Unidentified S	wun_s					1
Unidentified T	wun_t					1
Vernonia aurantiaca	veraur	1				
Ximenia americana	ximamer	1	1	1	1	
Ximenia caffra	ximcaff				1	
Xylopia parvifolia	xylpar	1				
Zanthoxylum humile	zanhum	1				
Ziziphus mucronata	zizmuc	1	1	1	1	1

A10 FARMER, H. RAPID ARTIFICIAL WATERPOINT ASSESSMENT TO INFORM MANAGEMENT DECISIONS

Rapid artificial waterpoint assessment to inform management decisions

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Abstract

The history of artificial water provision in southern African savannas has led to management dilemmas today. This study presents a rapid waterpoint assessment technique to inform reserve management of defensible suggestions for waterpoint closures. A five-point scoring system is used to combine ecological and human variables. The method was tested in York Private Nature Reserve (South Africa) where current water provision is perceived as excessive. Ecological variables (erosion, utilisation and vegetation cover) and human variables (ease to break/remove waterpoint, desirability and game viewing value) were assessed for the 52 artificial waterpoints (seasonal and permanent) on the property. Suggestion of waterpoint closures led to future predictions of considerably lower densities (closer to those of neighbouring properties) and a more even distribution of waterpoints between constituent properties. The rapid assessment approach was shown to generate management applicable information and is suitable for wider use and development.

Keywords

Conservation; herbivores; degradation; people; ecological awareness

Introduction

Provision of artificial waterpoints for herbivores has a long history in the southern African savanna. Construction of fences around protected areas in the mid-1900s prevented migrations to natural seasonal water sources and/or removed access to permanent natural water sources, so installation of artificial waterpoints was considered essential (Joubert, 1986; Walker et al., 1987; Chamaillé-Jammes et al., 2007). Artificial water provision stabilises water availability and can increase herbivore population size by increasing dry season foraging area (Aucamp et al., 1992; Grossman et al., 1999; Chamaillé-Jammes et al., 2007). Subsequent research has shown that long-term supplementation of water can cause environmental impacts which reduce ecosystem sustainability (Owen-Smith, 1996; Thrash, 1998; James et al., 1999; Thrash and Derry, 1999).

Ecosystem sustainability is reduced when vegetation is utilised to a point where it can no longer spontaneously recover: resilience of the system is lost (Holling, 1973; Walker et al., 1981; van de Koppel et al., 1997). With installation of permanent waterpoints, areas previously unused or only lightly used under a natural water distribution are opened for grazing and browsing and the period of utilisation is prolonged (Chamaillé-Jammes et al., 2007). Persistent high grazing and browsing leads to vegetation changes such as the replacement of perennial species by annuals and an associated decline in system productivity (Illius and O'Connor, 1999; Parker and Witkowski, 1999). High levels of trampling lead to increased erosion (Beukes and Ellis, 2003) and soil compaction (Bassett et al., 2005; Snyman and du Preez, 2005). The loss of ecosystem function leads to a reduction of system resilience and therefore a decline in sustainability (Holling and Meffe, 1996; Ludwig et al., 2001; Tongway and Hindley, 2004).

The current understanding of herbivore impact around waterpoints led to the suggestion that inter-waterpoint distances should be double the maximum distance travelled by foraging herbivores, leaving vegetation between permanent waterpoints that is unutilised in the dry season (Owen-Smith, 1996; Thrash, 1998; Smit et al., 2007). This vegetation provides forage during the wet season (Owen-Smith, 1996) and could be an important source of propagules for regeneration of dry season grazing areas (Eriksson, 1996; van Nes et al., 2002). On the basis of this understanding, some properties made decisions to reduce the density of waterpoints (Gaylard et al., 2003). On other properties, high waterpoint densities were maintained because of the increased revenue generated by larger herbivore populations (Aucamp et al., 1992; Grossman et al., 1999).

Many of the private savanna conservation areas in South Africa are too small to apply the spacing rule. In order to make management decisions, management applicable information is required (Pullin et al., 2004). Alternative guidelines need to be created when available approaches are not applicable. Information gathered needs to be directly applicable to the management question and the objective of the property (Underwood, 1998; Tranfield, 2002). It is important that private properties, often small, adjacent and unfenced, manage waterpoints jointly as herbivores do not respect property boundaries (Coughenour, 1991). Particularly in these smaller and privately owned reserves, management also needs to take into account the human element of the ecosystem such as profit requirement and tourism objectives (Brussard et al., 1998).

This study presents a rapid waterpoint assessment approach with the objective of informing reserve management waterpoint closure decisions. The approach was tested in a South African private reserve where management considers the current water provision level to be excessive.

Method

At the time of the study (July 2007), no rapid, simple assessment technique was available to assess waterpoints for closure potential, enabling defensible decision making. To use available techniques would have taken a minimum of 3 hours 10 minutes per waterpoint for a full assessment of soil functionality, grasses and woody vegetation or a minimum of 1 hour 26 minutes per waterpoint for a soil functionality assessment (H. Farmer, unpublished data). Additionally, there were no available approaches incorporating the human aspect of waterpoint closure.

This study developed a simple, rapid assessment technique that included both ecological and human factors. The method combined waterpoint description with five-point scoring, requiring 18 minutes per waterpoint. Five-point scoring is a rapid, simple and repeatable assessment technique that enables combination of ecological and human variables within calculations. Categories were kept broad to minimise interpretation differences between observers and boundaries were kept clear to minimise subjectivity. The method works on a comparative basis so as long as there is consistency across the study area, subjectivity problems are minimised. Multiple waterpoint variables (e.g. vegetation cover and human desirability) were scored on a scale of 1 to 5 and the final scores combined. High scores (closer to 5) were assigned for high ecological intactness and high human desirability. The decision process is then to close waterpoints with lower scores. See Figure 1 for a flow chart of the approach.

Study area

The method was tested at York Private Nature Reserve (YPNR), a 4 875ha reserve in the lowveld region of South Africa (30˚52’46” to 31˚1’41” East, 24˚11’38” to 24˚16’34” South). The reserve consists of six privately owned properties varying between 175ha and 1400ha in size. There is no internal fencing and animals move freely between properties. Use of the properties is private, with two properties having tourist lodges. Owners and management of YPNR consider the reserve to have an ecologically unsound waterpoint density and therefore have a desire to close waterpoints.

YPNR falls within the savanna biome. Approximately 75% of the reserve is dominated by *Combretum apiculatum*, *Grewia bicolor*, *Grewia monticola* and *Cissus cornifolia* (Peel et al., 2007). The lowest lying area of the reserve (380-440m asl), forming approximately one quarter of the reserve in the eastern half, is dominated by *Cissus cornifolia*, *Grewia bicolor* and *Dichrostachys cinerea* (Peel et al., 2007). The geology is dominated in the west by Harmony granite (Mashishimale suite) and in the east by Makhutswi gneiss with the rock type distinguished in the gneisses of the Swazian basement complex in this area being biotite gneiss (Walraven, 1989). The soils are generally of the Mispah and/or Glenrosa forms although other types may also occur (Land Type Series Map Pilgrims Rest, 1986). The climate is subtropical and semi-arid with cool, dry winters and hot, wet summers. An average of 365mm of rain (based on 12 years data) falls primarily between November and February. There are no permanent natural water sources on YPNR.

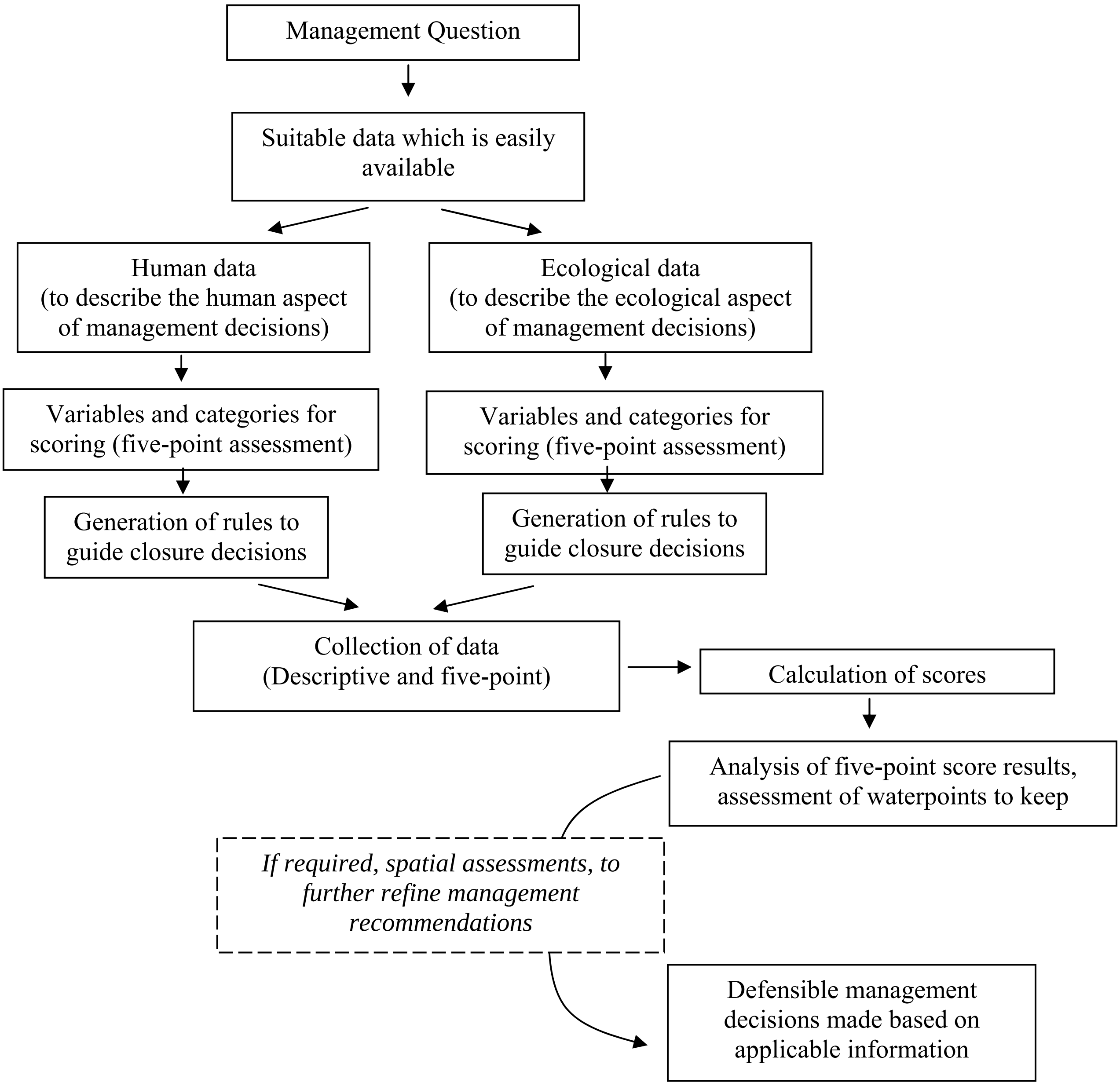

Figure 1: Flow chart of the approach to method development used in this study. This study used waterpoints, the method could be adapted for other management questions.

Waterpoint description

Location of each waterpoint was recorded with a GPS to enable generation of maps showing current water availability and to create a baseline on which to determine future possible waterpoint configurations. These data were also used to calculate the distance from each waterpoint to its nearest neighbouring waterpoint. Perimeters of waterpoints were walked with a GPS in order to calculate the area covered by each at average wet season size (determined by the vegetation line).

Type of waterpoint was recorded as trough (filled by a borehole pump), catchment dam (filled only by rainfall and runoff) or pumped dam (filled by rainfall, runoff and a borehole pump). Troughs and pumped dams are permanent whilst catchment dams are seasonal and dry out during the winter.

Ecological data

The highest degree of impact around a waterpoint is found in the first few hundred meters (Thrash, 1997; Thrash, 1998; Thrash and Derry, 1999), so ecological assessment was limited to the area immediately surrounding the waterpoint. Three variables were covered: waterpoint utilisation, erosion, and vegetation cover (Table 1).

Table 1: Waterpoint five-point score classification for ecological variables.

Score	Utilisation	Erosion	Vegetation Cover
1	Very busy waterpoint with lots of animals	Lots of active erosion, evidence of gullies cutting into soil	No or very little vegetation
2	Often used by herds	Lots of movement but not much gulley formation	Very poor grass cover but some woody plants present
3	Average use	Average	Average cover of grasses and some woody plants present
4	Sometimes used	Minor movement of soil	Average cover of grasses and structural variation in woody plants
5	Rarely used	Hardly any erosion to none at all	High grass cover and structural variation in woody plants

Waterpoint utilisation was determined through the quality of game viewing usually found at the waterpoint as this suggests the relative pressure of herbivore utilisation.

Waterpoints with higher pressure will degrade faster and to a greater distance (Adler and Hall, 2005). High scores were therefore given to waterpoints with bad game viewing, and therefore relatively low utilisation.

Erosion was scored separately to utilisation as factors such as road placement and soil type can affect the degree of erosion at a waterpoint. Erosion suggests the degradation of the area and the risk of soil loss – ultimately the loss of ecosystem function (MacGregor and O'Connor, 2002; Tongway and Hindley, 2004). Waterpoints with more advanced erosion are at a higher degradation risk and therefore received lower scores.

Vegetation cover was assessed for each waterpoint to gain a broad idea of the resilience of the area immediately surrounding the waterpoint. It is expected that areas with more vegetation, and more diverse vegetation, will recover better than those with large expanses of bare ground (Primack and Miao, 1992; Caylor and Shugart, 2004). Waterpoints with little, and less structurally complex and diverse, vegetation scored low. Vegetation description was limited to structure in order to remove the need for expert knowledge on plant species identification.

Human data

Human data were collected through expert knowledge of the properties, waterpoints and landowners. Variables considered were related to closure of the waterpoint: the ease of breaking or removing infrastructure, the human desirability of the waterpoint and the value of the waterpoint for game sightings (Table 2). Variables and categories were based on expert management advice, the aim of this section was to consider the human data only.

The ease with which a waterpoint can be broken will contribute greatly to whether or not a suggested closure is carried out. This reason was included because it is a realistic aspect of waterpoint management although not necessarily an ecologically preferred one. Breaking a waterpoint was considered to be removing its ability to hold water, not complete removal of all infrastructure. The score for this variable was a combination of monetary cost and work effort: waterpoints which would be cheap and easy to break got low scores. In some cases it may be necessary or desirable to completely remove a waterpoint and all associated infrastructure. Scoring for removal followed the same categories as breaking.

Table 2: Waterpoint five-point score classification for human variables.

Score	Ease to break/remove	Human desirability	Game sightings
1	Cheap and easy	No-one would care	No-one ever sees anything
2	Easy/cheap for one factor	Only a few people would care	Check it once in a while and maybe get lucky
3	Average for both factors	Average	Average
4	Expensive/difficult for one factor	Would be nice to keep it	Often lucky with sightings
5	Expensive and difficult	Have to keep it	Well known as a good spot

The human desirability of a waterpoint was assessed in terms of how much opposition would be expected in response to a decision to close the waterpoint. Waterpoints with sentimental value or located in front of camps were given high scores.

Game sightings at a waterpoint indicate the economic value of the waterpoint. Waterpoints with a higher incidence of game sightings were presumed to be preferred so higher scores went to more frequently used waterpoints. Seasonal waterpoints were assessed based on their attraction during the wet season.

Description analysis

In order to determine water availability on YPNR and to compare to neighbouring properties, waterpoint densities were calculated in Excel. Densities were calculated (1) overall, (2) by waterpoint type, and (3) for each property within YPNR. A Mann-Whitney U Test was used to compare the permanent waterpoint density with other APNR properties (limited to permanent waterpoints because of data availability from other properties).

The distance from a waterpoint to its nearest neighbour indicates the probability of degradation stretching between neighbouring waterpoints (Adler and Hall, 2005). To determine current spacing and arrangement of waterpoints, ArcView 3.2 was used to measure the distance from each waterpoint to its nearest neighbour. Due to the very short distances between some waterpoints and the large variation in size between waterpoint types, edge-edge distances were measured rather than centroid distances. The standard deviation of nearest neighbour distances was calculated to determine variability of waterpoint distribution.

Calculation of five-point scores

Five-point scores were calculated by averaging scores for individual factors. Ecological five-point scores were the average of three factors (Table 1) and human five-point scores were the average of four factors (Table 2). The overall five-point score of a waterpoint was the average of the human and ecological scores. In this study an even average was used though the weighting of the ecological and human factors could be changed.

Exploration of five-point scores

Scientific literature suggests that seasonal waterpoints should be less impacted than permanent waterpoints (Harris and Asner, 2003; Chamaillé-Jammes et al., 2007). Catchment dams are seasonal whilst pumped dams and troughs are permanent. Ecological five-point scores were compared between waterpoint types using a Mann-Whitney U Test to investigate whether seasonal waterpoints had a lower impact than permanent waterpoints.

Areas with a higher waterpoint density should have a higher degree of impact as they support a greater number of herbivores (Chamaillé-Jammes et al., 2007). The relationship between waterpoint density and degree of impact (ecological score) was tested using Regression.

In order to investigate the dependence of the reserve on animal populations, the relationship between human desirability and aesthetics was tested by comparing the desirability and game sighting scores with vegetation cover and erosion using a

Wilcoxon Matched Pairs test. Areas with higher vegetation cover and less erosion were considered to have a greater aesthetic value.

Data analysis: Closure suggestions and the future scenario

In order to determine which waterpoints should be closed, a series of rules were applied to the five-point scores (Table 3, rules 1 to 9). Waterpoints which came out as inconclusive based on five-point score analysis were assessed visually using ArcGIS, again using a rule based approach to determine which waterpoints should be closed (Table 3, rules 10 to 12). When waterpoints were equal in all factors in the spatial assessment, human desirability dominated the decision on which waterpoint should be closed.

Table 3: Rules used for decisions to keep or close waterpoints. Rules 1-9 used on five-point scores, rules 10-12 used on spatial maps.

	Decision name	**Further notes**
Keep	1. Camp	Waterpoint situated in front of a camp
	2. High ecological	High ecological score, average human score
	3. High human	High opposition to and cost of closure
	4. High total	High ecological and human score
	5. Impossible break	One seasonal dam is impossible to break due to a road
Close	6. Low ecological	Ecological score very low
	7. Break planned	Decision has already been made to close the waterpoint
	8. Low human	No need to keep the waterpoint open as no-one would care
	9. Low total	Low ecological and human score
	10. Near permanent	Close to a permanent waterpoint
	11. Gap creation	Removal creates a large gap between waterpoints
	12. Too close	Waterpoint within the boundary of another waterpoint

Ecological warnings were given to waterpoints which were kept for human reasons but had low ecological scores. A warning highlights a waterpoint which is ecologically unsound (e.g. heavily eroded) and may therefore require restoration or rehabilitation even though it is kept open.

The future scenario (if all suggested closures are carried out) was assessed using the same statistical descriptions as the current YPNR waterpoint situation. The property water density equality of the current and future scenarios was investigated using χ^2 tests. A regression was used to test the effect of current waterpoint density on the level of reduction required by individual properties. The future permanent waterpoint density was compared to neighbouring APNR properties using a Mann Whitney U Test.

Results

A total of 52 waterpoints were sampled in 16 hours of fieldwork. Analysis was simple and rapid enabling a short turnaround time between data collection and management information (Farmer, 2007).

Total waterpoint density of YPNR is 1.07 points/km^2 (54% seasonal dams, 31% permanent dams and 15% permanent troughs). Nearest neighbour calculations including all current waterpoints on YPNR show very short distances between waterpoints and wide variation of waterpoint distribution (Table 4). YPNR and its constituent properties have a significantly higher density of waterpoints than neighbouring properties (Table 5; $Z = -2.56$, $n = 4,6$, $p = 0.010$).

Table 4: Descriptive statistics of current waterpoints on YPNR properties.

Property	Total Density (points/km^2)	Nearest Neighbour (m) mean (sd)
1	1.00	551 (395)
2	0.43	970 (454)
3	0.73	43 (2)
4	1.64	501 (248)
5	1.71	399 (107)
6	2.80	160 (111)
YPNR	1.07	481 (376)

Final five-point scores considering all factors varied from 2 to 4 whilst those for ecological and human factors separately varied from 1 to 5. The average score for all factors considered was 3.1 (sd = 0.529), for ecological it was 3.0 (sd = 1.039) and human factors it was 3.2 (sd = 1.155).

Table 5: Permanent waterpoint densities on YPNR and neighbouring, similarly managed, properties.

Property	Permanent waterpoint density (points/km^2)
YPNR	1.03
Neighbour 1	0.14
Neighbour 2	0.12
Neighbour 3	0.13
Neighbour 4	0.10

There was no significant difference between the ecological five-point scores of seasonal and permanent waterpoints across YPNR ($Z = 1.45$; $n = 28,24$; $p = 0.151$). When all waterpoints were grouped together, there was no relationship between property total waterpoint density and ecological five-point score ($n = 52$, $F_{(1,50)} = 1.479$, $R = 0.170$, $p = 0.230$). When waterpoint types were split, the lack of significant result was maintained for permanent waterpoints: permanent dams ($n = 16$, $F_{(1,14)} = 0.010$, $R = 0.027$, $p = 0.921$) and troughs ($n = 8$, $F_{(1,6)} = 0.547$, $R = 0.289$, $p = 0.487$). However, for seasonal (catchment) dams, the ecological score was significantly lower when the property waterpoint density was higher ($n = 28$, $F_{(1,26)} = 6.687$, $R = 0.452$, $p = 0.016$): seasonal waterpoints were more degraded on properties with higher waterpoint densities.

There was no significant relationship between the aesthetic value of a waterpoint and its value for game sightings ($n = 52$, $Z = 0.974$, $p = 0.330$) or its human desirability ($n = 52$, $Z = 0.169$, $p = 0.866$).

YPNR closures and the future scenario

The closure rules (Table 3, rules 1 to 9) were applied to the five-point scores leading to a recommendation to keep 20 waterpoints and close 16. A further 16 waterpoints could not be classified using the five-point rules so they were assessed using spatial rules (Table 3, rules 10 to 12). Most of the waterpoints to be kept were permanent dams whilst those to be lost were troughs and seasonal dams.

Reasons for keeping waterpoints varied between properties. The reasons 'camp' and 'high human' kept high numbers of waterpoints and also had high numbers of waterpoints with ecological warnings: 44% of waterpoints kept because of 'camp' and 83% of waterpoints kept because of 'high human' had ecological warnings. Seasonal waterpoints demonstrated a greater number of ecological problems.

Decisions to break permanent waterpoints were only made when the overall score (human and ecological combined) was low. This means that the waterpoint is neither ecologically sound nor desired by humans and is easy and cheap to remove. Decisions to break seasonal waterpoints were primarily made when human factor scores were low.

After application of the spatial rules, an additional 10 waterpoints were categoriesd for closure. The use of spatial decision rules varied between properties with the most commonly used reason being 'gap creation' (used on 80% of properties). After all assessments, it was recommended that 26 waterpoints be kept and 26 closed.

Current waterpoint densities are significantly different between properties within the YPNR (χ^2 = 17.073, df = 5, p = 0.004). Properties with high original waterpoint densities keep a lower percentage of this original density (R = 0.922, $F_{(1,4)}$ = 22.57, p = 0.009). If the closure suggestions are carried through, the future density of waterpoints on YPNR would be 0.53 points/km^2, a 50% reduction from the current density. Future densities of waterpoints are not significantly different between properties (χ^2 = 7.514, df = 5, p = 0.185).

For permanent waterpoints, reduction per property varies between 0 and 50%. Future suggested densities are still higher than neighbouring properties but this difference is not statistically significant (Z = -1.706, n = 4,6, p = 0.114). There is a large increase in average nearest neighbour distance across all properties bringing each of them closer to the average of their neighbours when all waterpoints are included in the analysis (Table 6).

Table 6: Descriptive statistics of future waterpoints on YPNR properties.

Property	Total Density (points/km^2)	Nearest Neighbour (m) mean (sd)
1	0.43	787 (234)
2	0.36	1275 (721)
3	0.36	2060 (-)
4	0.73	994 (166)
5	0.57	-
6	1.60	252 (278)
YPNR	0.53	905 (512)

Discussion

The level of water supplementation in South African conservation areas is hotly debated. Previous research has highlighted long-term negative impacts of water provision (Illius and O'Connor, 1999; Parker and Witkowski, 1999) and this has led to suggestions of waterpoint closure in areas with high densities of artificial waterpoints (Owen-Smith, 1996; Thrash, 2000). Approaches have been developed to assist closure decisions but these are limited to spatial analysis and only work over very large areas (Owen-Smith, 1996; Smit et al., 2007), many of the South African private reserves are too small to use these techniques. There is a need for management applicable information that can be rapidly gained, incorporate comparisons and include expert knowledge in areas where data is lacking (Underwood, 1998; Tranfield, 2002; Huntington et al., 2004).

This study designed a method that can be used to rapidly gain management applicable information with regards to waterpoint closure decisions. The technique developed was tested in York Private Nature Reserve, South Africa, where the waterpoint density is higher than on neighbouring properties with similar management. There is a landowner and management perception of excessive waterpoints but decisions on which waterpoints to close need to be defensible and logical to maintain relationships between landowners within the reserve.

In order to ensure inclusion of all important variables, factors considered were split into human and ecological data. Ecological factors reveal a skew towards lower scores, indicating more waterpoints with ecological problems. This corroborates the perception of excessive water provision leading to degradation (R. Ahlers pers. comm.). The distribution of human scores tends towards higher scores indicating resistance to waterpoint closure.

Current densities and nearest neighbour distances suggest that waterpoint numbers on YPNR should be reduced, in line with owner perceptions. High densities of waterpoints potentially reduce the resilience of vegetation and increase the probability of degradation (Walker et al., 1981; van de Koppel et al., 1997). Degradation subsequently reduces sustainability of ecosystem use (Ludwig et al., 2001; Tongway and Hindley, 2004).

Assessment of ecological five-point scores revealed no significant difference in impact around seasonal and permanent waterpoints. This is in contradiction to Harris and Asner (2003) who found no impact at seasonal artificial waterpoints. The scale of investigation of this study was much smaller and the waterpoint density was higher; Harris and Asner (2003) worked in a 21 000 ha area with only three seasonal and one permanent waterpoint (seasonal waterpoint density of 0.01 points/km^2 and a permanent waterpoint density of 0.005 points/km^2). This study found that property permanent waterpoint density affects seasonal waterpoint impact, properties with higher permanent waterpoint densities have higher degradation of seasonal waterpoints. The presence of permanent water prolongs the utilisation of vegetation into the dry season and close proximity of waterpoints results in no seasonal movement into new foraging areas during the wet season (Aucamp et al., 1992; Chamaillé-Jammes et al., 2007).

When looking at human preference and aesthetic value as represented by vegetation cover and soil stability, there was no significant relationship. However, there was a slight trend towards preference of degraded and ecologically unsound waterpoints due to better game viewing opportunities. This highlights the economic dependence of privately owned properties on animal populations rather than scenery. Conservation management needs to strive towards balancing ecological requirements for sustainability and an often opposing human preference.

If closures are performed, the future YPNR waterpoint scenario is more ecologically sound than the current. Due to the small size of the reserve, commonly used spacing guidelines for waterpoint distribution could not be followed. A spacing approach of 'as far as possible' was applied and closing rules were also determined based on alternative guidelines. These guidelines were based on ecological concepts such as the ability of vegetation and soils to withstand herbivore impact. Natural positioning of waterpoints leads to higher herbivore utilisation in areas with vegetation and soils which are more evolved to handle the impact (Milton, 1991; Washington-Allen et al., 2004; February et al., 2007). Dams are therefore preferred over troughs as they keep water within the drainage line rather than on highly sensitive soils which are often adjacent to the drainage line. However, construction of dams leads to disruption of the riverine system so it is important not to construct too many dams on a particular drainage system or river (O'Connor, 2001).

Due to human preferences and resistance to closure, some waterpoints were kept in spite of low ecological scores. These waterpoints should be highlighted as targets for potential restoration or rehabilitation work. Over time, the utilisation zone around a waterpoint expands (Adler and Hall, 2005), so care must be taken that these areas do not become more extensively degraded. The importance of human opinion was highlighted in this study through the human variable scores and their impact on waterpoint closure decisions. Currently, financial and recreational needs dominate over ecological issues, increasing ecological awareness amongst landowners may lead to a shift in this focus.

Concluding remarks

Water provision is a hotly debated topic in southern African savanna reserves and management related information is required in order to substantiate decisions. Water provision debates and subsequent decisions are often restricted to ecological impacts with little or no attention paid to the human side of water provision. Water provision, however, has large economic impacts through profit generation and it is therefore important to note the role that human factors can play in closure decisions. Starting this study with a management question lead to development of a method directly applicable to answering the question. Due to its simplicity, the method can be easily modified, developed and fine-tuned for more intensive surveys or for use in other areas or on other questions. The five-point scores could be based on a weighted average which could be combined with different management options to perform a risk assessment by illustrating potential impacts of differing management decisions.

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