An evaluation of techniques for monitoring rangeland health in semi-arid savannas in southern Africa

by

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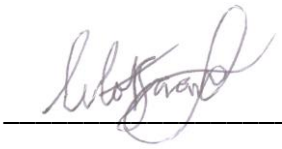
Declaration

I hereby declare that this dissertation, which I submit in fulfilment of the degree of Master of Science at the University of Pretoria, South Africa, is my own work. This work has not been previously submitted by me for a degree at another university or institution.



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Abstract

Rangeland health plays an important role in increasing the connectivity/contact between wildlife, livestock and people. Communal rangelands throughout the savanna areas of South Africa are experiencing increasing magnitudes of pressure as they are reduced in size mainly due to a rapidly expanding human population. These pressures emanate in greater levels of degradation, which results in greater competition between livestock and wildlife across the interface for ecosystem services (such as grazing). Ultimately, this brings wildlife and livestock closer together and enhances the probability of disease transmission. Long-term monitoring of rangeland health using survey methods that are comprehensive, rigorous (and accurate) and efficient in terms of time and cost are essential for the development of sustainable management approaches that aim to optimise the ecological, social and economic well-being of an area.

The primary aim of this study was to objectively assess a number of rangeland monitoring techniques [Multiple Indicator Monitoring (MIM) method, Adapted Point-centred Quarter (APCQ) method, Basal Cover (BC) method, Line-point Intercept (LPI) method and the Disc Pasture Meter (DPM) method that uses 100 recordings)]; taking into account efficiency (time and cost) and usefulness in recording indicators of rangeland health. The findings further provide a basis for the implementation of sound management practises across multiple land-use types.

The Mnisi Study Area (MSA) is situated in the Lowveld region of Mpumalanga, South Africa, and is comprised of communal rangelands and land zoned for conservation purposes. The rangelands under communal tenure belong to community members from various villages; and are the sole source of available grazing to the local livestock population, made up mainly of cattle and goats. Some of the communal rangelands exist at the wildlife-livestock interface of the Manyeleti Game Reserve. The Manyeleti is managed as an open system and contains a full suite of wild herbivores endemic to the area.

The MIM method was considered to be an efficient and rigorous survey method that can be used to detect comprehensive information of the health and status of rangelands across semi-arid savannas of southern Africa.

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List of Abbreviations

APCQ – Adapted Point-centered Quarter method

BC – Basal Cover method

BU/ha – Browser Unit per hectare

CDF - Community Development Forum

DARDLA - Department of Agriculture, Rural Development, and Land Administration

DPM – Disc pasture meter

FMD – Foot and mouth disease

HVUs – Homogenous vegetation units

KNP – Kruger National Park

LPI – Line-point Intercept method

MAP – Mean annual precipitation

MCP – Mnisi Community Programme

MCR – Mnisi communal rangelands

MGR – Manyeleti Game Reserve

MIM – Multiple Indicator Monitoring method

MSA – Mnisi Study Area

MTA – Mnisi Traditional Authority

MTPA – Mpumalanga Tourism and Parks Agency

PMB – Patch mosaic burning

RSA – Republic of South Africa

SAM – Strategic adaptive management

SANParks – South African National Parks

SRP – Species recording potential

T1 – Transect 1 of APCQ

T2 – Transect 2 of APCQ

TPCs – Thresholds of potential concern

1. INTRODUCTION

1.1 Literature review

1.1.1 Rangeland health and origins of rangeland ecology

Rangeland ecology emerged as a discipline within grassland science through the identification of various anomalies associated with the practise of pastoralism and the potential degradative effects which it imposes on the natural environment (Ellis and Swift 1988). As more knowledge has been gained over time, rangeland management approaches have adapted along with the development of various models and paradigms that aim to explain observed ecological trends (Peel et al. 1998).

It was initially perceived that vegetation trends were largely governed by animal numbers, however, perceptions continue to change as it is evident that there are a number of internal- and external-system factors which govern vegetation dynamics (Ellis and Swift 1988, Westoby et al. 1989, Fynn and O'connor 2000, Peel et al. 2005). Consequently, this has led to the development of various strategic adaptive management (SAM) regimes well suited for arid and semi-arid savanna systems (Westoby et al. 1989, Grant et al. 2011, Roux and Foxcroft 2011).

Pertinent to the establishment of sound SAM regimes is the assessment of health or functional integrity of rangelands (Briske et al. 2005). Rangelands that exist in a healthy state exhibit greater levels of functional integrity, which optimizes ecosystem productivity and increases provision of ecosystem services to humans and animals (Goldstein et al. 2011). Aiding our understanding of rangeland health in savanna systems requires the use of rapid and efficient techniques for detecting indicators of rangeland health associated with the herbaceous and woody vegetation components e.g. species composition, tree density, basal cover and biomass (Briske et al. 2005).

1.1.2 Semi-arid savannas

Semi-arid and arid savannas (savannas with a mean annual precipitation [MAP] of less than ~ 650 mm) are considered stable, but complex systems that function in various states of equilibrium and non-equilibrium (Peel et al. 1998, Scogings et al. 1999, Bond and Archibald 2003, Van den Berg and Kellner 2005, Sankaran et al. 2005). These systems are largely driven by stochastic events, which can present themselves in various natural and human-induced forms i.e. drought, inter-annual rainfall variability, fire, disease outbreak and over-utilization (Scogings et al. 1999, Bond and Archibald 2003, Synodinos et al. 2018).

In an unaltered state, semi-arid savannas are characterised by the co-occurrence of trees and grasses and are composed of vegetation that is water-limited due to the significant rainfall variability, which plays a primary role in governing system trends (Walker et al. 1981,

Ellis and Swift 1988, Fynn and O'connor 2000, Van Langevelde et al. 2003, Sankaran et al. 2005, Synodinos et al. 2018).

Global-scale effects such as fluctuating climatic conditions and increasing global CO_2 levels influence the tree-grass ratios of savannas across the world (Good and Caylor 2011, Bond and Midgley 2012, Buitenwerf et al. 2012). At a local scale, the effects of stochastic events, and anthropogenic disturbances driven by socioeconomic factors, play an important role in the dynamics and functioning of semi-arid savanna ecosystems (Scogings et al. 1999, Van Langevelde et al. 2003, Twine 2005, Govender et al. 2006, Bucini and Hanan 2007).

The semi-arid savanna of the central Eastern Lowveld of South Africa (RSA) plays host to an array of ecosystem services which supplement an economic/wealth spectrum (Twine 2005, Rogers et al. 2008). Servicing the upper end of the spectrum, ecotourism is practised throughout most protected areas and caters for members of the upper wealth classes, particularly international tourists (Lindsey et al. 2007). Ecotourism creates numerous job opportunities for local communities and, as cited in Lindsey et al. (2007), is believed to be "an effective means of redistributing wealth from developed to developing nations and for generating funds for conservation (Gössling 1999)".

Where poverty is common, many community members depend on the indigenous vegetation of semi-arid communal rangelands (Twine 2005). These rangelands provide an abundance of natural resources which supplement the livelihoods of people and their livestock, from subsistence- to commercial-level (Scogings et al. 1999, Twine 2005, Nyamukanza et al. 2008).

1.1.3 Top-down and bottom-up ecological disturbance agents that drive semi-arid savanna dynamics

Scale and management

Ecological processes interact with the abiotic template to produce vegetation trends that result in an array of abiotic and biotic dynamics or interactions across African savannas (Scogings et al. 1999, Van Langevelde et al. 2003, Sankaran et al. 2005). Changes in management regimes, along with changes in rainfall pattern and soil properties of an area, can result in short-medium term vegetation responses operating at a landscape scale (Peel et al. 1998, Van den Berg and Kellner 2005, Peel et al. 2005). Furthermore, operating at a large time scale, the top-down effects of changes in disturbance/management regimes can operate simultaneously with the climate and geomorphology to govern the composition and diversity of fauna and flora of an area (Peel et al. 1998, Van den Berg and Kellner 2005, Peel et al. 2005).

The functioning of the system at various spatial scales is influenced by management-related factors such as the distribution of water, presence of fences and burning of veld to create patch dynamics (Ellis and Swift 1988, Grant et al. 2011). Many of these management

interventions can be used as a tool that induces flux to a system, which aims to generate positive outcomes such as optimizing biodiversity (Van Wilgen et al. 2008). In many cases however, the effects are negative and the resulting outcomes can be detrimental to the system (Grant et al. 2011).

Fence boundaries influence the density and composition of herbivore species that an area can sustain, which further determines the scale of management intervention required for that area (Grant et al. 2011). Smaller fenced areas that depend on tourism, hunting or game ranching (game sales) for their operation implement policies that increases water provision and promotes increased densities of water-dependent herbivores (Grant et al. 2011). The even-distribution of artificial water is aimed at dispersing the intensity of resource utilization by herbivore species across the entire landscape (Grant et al. 2011). In extreme cases, however, this often results in resource over-utilization across an extended area (Grant et al. 2011). It has also been revealed that a high density of artificial water points may result in the progression of vegetation structure and composition towards a homogenous state, particularly due to the domination of competitive water-dependent herbivores such as impala *Aepyceros melampus* (Grant et al. 2011).

Herbivore-vegetation dynamics

Herbivores interact with a host of other top-down and bottom-up system drivers (Sankaran et al. 2005, Seydack et al. 2012). Their contribution to ecosystem dynamics renders them of significant importance to managers as indicators of system change, guiding intervention only when impacts become excessive (Grant et al. 2011).

Herbivores play an important role in redistributing nutrients and can increase the spatial heterogeneity of available plant nutrients in a landscape through faecal and urinary deposits (Augustine and Frank 2001). Grazer species that exhibit territorial behaviour; such as white rhinoceros *Ceratotherium simum*, often deposit dung in middens, which results in the localized deposition of nutrients in an area (Veldhuis et al. 2018). This affords an opportunity to species of specialized functional niches, such as dung beetles, which operate to disperse these nutrients as an indirect result of their activities (Veldhuis et al. 2018). See 'Herbivore-fire Interactions' for effects other grazers may have on vegetation dynamics.

Browsers feed predominantly on trees and shrubs, and their selective feeding behaviour is largely attributed to their nutritional needs and quality of available browse (Wigley et al. 2014). Less is known about the influence of browsers on woody plant communities in savanna systems, while their species-specific effects are better known (Wigley et al. 2014). Over time, feeding pressures from browsing herbivores have resulted in some woody species adapting chemical defences to deter herbivores, such as the increased production of secondary metabolites (Wigley et al. 2014, Scogings 2018). Fornara and Du Toit (2007) found that *Acacia nigrescens* individuals were able to alter their "normal" physiological functioning to show traits of tolerance and resistance. Individuals that experienced high levels of

browsing exhibited fast rates of regrowth, extensive branching and a reduction in spacing between thorns (Fornara and Du Toit 2007).

Elephants *Loxodonta africana* are mixed feeders that require large areas for their seasonally-governed movement patterns (Loarie et al. 2009). Foraging selectively at various height classes; elephants influence the woody structure and composition of an area at a species-level, resulting in direct and indirect effects on co-occurring species (Guldemond and Van Aarde 2007, Loarie et al. 2009). The development of fences and artificial water has contributed to the successful management and protection of this species; however, it has also resulted in unintentional impacts on elephant behaviour (Loarie et al. 2009). Both of these management interventions enhance the local impact on vegetation by generating unusual seasonal movement patterns of elephants, ultimately driving vegetation homogenisation (Bond and Archibald 2003, Loarie et al. 2009, Grant et al. 2011).

In communal rangelands, the harvesting of wood by community members results in anthropogenic impacts that closely mimic those caused by elephants (Twine 2005). Higgins et al. (1999) found a lower woody species richness in communal rangelands than in nearby private game reserves, suggesting that management and wood harvesting activities associated with communal rangelands can potentially have a negative impact on the species richness of savanna systems. Shackleton et al. (1994) also found a positive correlation between the increased intensity of disturbance and the reduction of woody species diversity in rangelands under communal management.

Woody savanna species have a high ability to resprout and exhibit coppicing growth, which enables them to withstand the effects of intense harvesting and increases the systems resilience to species loss (Higgins et al. 1999, Shackleton et al. 1994). Although the reduction of stem density is mitigated to some extent by coppiced regrowth, the effects of wood harvesting can be easily observed at tree-morphology level (Shackleton et al. 1994, Higgins et al. 1999, Twine 2005). The reduction of mature stems/trees (with increasing proximity to villages) has resulted in a greater number of coppicing adult trees which are functionally juvenile, influencing woody species demographics (Shackleton et al. 1994, Higgins et al. 1999, Twine 2005).

Fire

Fire frequency and intensity interact with the life history traits of plant species, and play an important role in determining the co-existence of grasses and trees in savanna systems (Higgins et al. 2000, Van Langevelde et al. 2003, Sankaran et al. 2005, Govender et al. 2006, Van Wilgen et al. 2008). While rainfall controls the likelihood of seedling establishment, fire contributes to the survivorship and recruitment of woody species to functional maturity (Higgins et al. 2000). Fire-related ecological research undertaken in the KNP was reviewed by Van Wilgen et al. (2007); resulting in conclusions that fires influence species composition less than their effect on biomass and vegetation structure (Van Wilgen et al. 2008).

Arguably, before human population levels were of the current magnitude, fires would have occurred primarily as a result of lightning ignitions in the late dry season (Van Wilgen et al. 2008). Management approaches that aim to distribute burning across the dry season create shifts in fire intensity, potentially resulting in increased localised tree mortality in certain areas (Van Wilgen et al. 2008).

The implementation of suitable fire management approaches are not universal to all areas of savanna systems, as they vary according to scale and vegetation response to disturbance across varying gradients of rainfall and soil fertility (Bond and Archibald 2003, Van Wilgen et al. 2008). Patch mosaic burning (PMB) is a fire management regime commonly used in savanna systems (Parr and Andersen 2006). It aims to create patches of varying fire histories, resulting in a spatially and temporally heterogeneous vegetation mosaic (Parr and Andersen 2006). There are; however, a number of potential shortcomings to the assumption that 'pyrodiversity begets biodiversity', which managers should consider during SAM planning (Parr and Andersen 2006, Van Wilgen et al. 2008, Roux and Foxcroft 2011).

Herbivore-fire interactions

MAP has an effect on the growth of herbaceous species and maximum woody cover in semi-arid savannas (Bond et al. 2003, Sankaran et al. 2005). In these systems, herbivore-fire interactions combined with soil properties, limits woody cover below the upper MAP-controlled margins (Sankaran et al. 2005). As with the case of greater-rainfall savannas (MAP of more than ~ 650 mm); they are considered to be 'unstable' systems, where woody canopy closure is possible in the absence of disturbance due to sufficient rainfall (Bond et al. 2003, Sankaran et al. 2005). In these systems, the disturbance created by fire and herbivory plays a role in driving ecosystem dynamics (Sankaran et al. 2005).

Apart from providing food for herbivore species, herbaceous biomass also serves as fuel to govern the intensity and frequency of fires (Van Langevelde et al. 2003). Continuous, heavy grazing reduces herbaceous biomass, resulting in less intense fires which lack sufficient heat to successfully suppress the establishment and growth of woody species (Eckhardt et al. 2000, Van Langevelde et al. 2003). This process can continue to function as a negative feedback loop (particularly in times of drought when the grass sward is under stress) as seen in Figure 1; where woody species are able to out-compete herbaceous species for water, light and soil resources (Higgins et al. 2000, Eckhardt et al. 2000, Van Langevelde et al. 2003). With regard to vegetation dynamics, the decline in biomass and diversity of herbaceous species has been shown to result in the progressive transformation of savanna systems towards woodlands (Higgins et al. 2000, Van Langevelde et al. 2003).

The enhanced effect of herbivore-fire interactions on woody vegetation trends has been strongly correlated with elephant-associated activities and bark-feeding vertebrates (such as porcupine *Hystrix africaeaustralis*) (Yeaton 1988, Grant et al. 2011). Both species, particularly elephants, play a significant role in large tree mortality by rendering woody

individuals more susceptible to damage during fires (Van Wilgen et al. 2008, Grant et al. 2011). Pellegrini et al. (2016) infer that tree mortality caused by elephants could surpass woody plant productivity rates, while mortality induced by fire alone could not. Preventing or reducing the rate of large-tree decline in an area cannot be addressed by simply manipulating elephant population dynamics or fire regimes, and the need for consideration of various factors (i.e. browsing of seedlings by other herbivores, patterns of seasonal rainfall and seed predation by insects and small mammals) is promoted (Owen-Smith et al. 2006, Van Wilgen et al. 2008).

Disease transmission

The presence of diseases of veterinary and medical importance such as foot and mouth disease (FMD), corridor disease, rabies, canine distemper and canine parvovirus further contribute to ecosystem dynamics (Norval et al. 1991, Bruckner et al. 2002, Van Schalkwyk et al. 2016). The proximity of humans, domestic animals and livestock to protected areas allows for increased frequencies of disease transmission across the wildlife-livestock interface; particularly when competition for resources between people and their livestock, and wildlife, is enhanced as a result of poor rangeland health (Thomson et al. 2013, Van Schalkwyk et al. 2016).

The integration of rangeland health information with that of other disciplines (climate change data, animal movement patterns and disease epidemiology) allows for the development and implementation of interdisciplinary mitigation measures for the effects of disease outbreak (Barrett et al. 2011, Grant et al. 2011). Such outbreaks affect communities and their livestock living adjacent to protected areas, and may also threaten meta-populations of sensitive species within protected areas (Van Heerden et al. 2002, Van Schalkwyk et al. 2016). The prevention and mitigation of the effects of disease ultimately reduces economic, social, medical and environmental setbacks to an area (Barrett et al. 2011).

Figure 1: Poor grass cover (0-10% cover) and strong woody presence in the communal rangelands of Utah B towards the end of the 2015/2016 drought (29 September 2016).

1.1.4 Long-term rangeland monitoring across the interface to inform management

The importance of long-term monitoring

A degree of heterogeneity exists across the wildlife-livestock interface due to varying management policies and objectives, different approaches to land utilization and the presence of various herbivore assemblages that exist across the interface (Twine 2005, Grant et al. 2011, Wolfaard 2013). Tracking vegetation trends and evaluating outcomes associated with management interventions requires monitoring of rangeland health indicators in a standardised, rigorous and comparative manner to inform sound management practises (Grant et al. 2011, Scholes and Kruger 2011, Roux and Foxcroft 2011).

Long-term monitoring of ecosystem health indicators allows for the development of hypotheses used to evaluate the progress of benchmark thresholds of potential concern (TPCs) (Grant et al. 2011, Roux and Foxcroft 2011, Scholes and Kruger 2011). These TPCs represent revisable boundaries of a desired state, which institute the outer limits of acceptable change for an area (Roux and Foxcroft 2011, Grant et al. 2011). Subsequently, gaining an understanding of ecological trends and vegetation dynamics through long-term monitoring provides a framework for the development of SAM approaches that aim to contribute to the socio-economic viability of the area in an ecologically sustainable manner (Grant et al. 2011, Roux and Foxcroft 2011, Scholes and Kruger 2011).

The integration of knowledge on long-term rangeland dynamics with other disciplines assists decision-making in the broader perspective of the 'One Health' concept (Barrett et al. 2011, Zinsstag et al. 2011). In Africa, transboundary zoological diseases pose numerous challenges for wildlife conservation and the future of livestock production (Scoones et al. 2010). There is a need for inter-disciplinary approaches to resolving issues associated with global change; sustainability of land use practises; and emerging diseases and their transmission between wildlife, livestock and humans across the interface (Barrett et al. 2011).

Long-term monitoring in the Mnisi Study Area

In 2009 various indicators of rangeland health were assessed at selected sites throughout the study area (Müller 2009). A comparative study was later conducted at the same sites; where the corresponding indicators of rangeland health were assessed using the same methodology, revealing changes in vegetation throughout the different land use types (Wolfaard 2013).

Assessment of changes at monitoring sites in the Mnisi Study Area (MSA) revealed varying degrees of resource utilization (Wolfaard 2013). It was inferred that the heterogeneity observed across the land uses could be largely attributed to: differences in localised rainfall and MAP; the difference in composition and foraging behaviour of herbivore species across land uses; as well as the implementation of a PMB fire regime across the Manyeleti Game Reserve (MGR) (every 3 to 4 years) and lack of fire regime in the communal rangelands (Wolfaard 2013).

Part of the research being undertaken in the Mnisi Community Programme (MCP) is the detection and surveillance of diseases of veterinary and medical importance. The MSA forms part of the FMD protection zone of South Africa in accordance with the 'Veterinary procedural notice for foot and mouth disease control in South Africa' by the Department of Agriculture, Forestry and Fisheries (VPN 2014). Foot and mouth disease is endemic in many protected areas in Africa, such as the Greater KNP, because the primary reservoir for the FMD virus is the African buffalo *Syncerus caffer* (Bruckner et al. 2002, Vosloo et al. 2002, Scoones et al. 2010, Van Schalkwyk et al. 2016). Vaccination of cattle against FMD is compulsory throughout the FMD protection zone along the western boundary of the KNP, and as a result, renders protection against the spread of the disease to the rest of South Africa which is free from FMD (Van Schalkwyk et al. 2016). Most importantly, FMD is controlled in the protection zone through movement control of all cloven hoofed species and their products (VPN 2014). As a result, livestock trade is severely constrained which contributes to a lack of market development and animal off-take in both wildlife and livestock management areas (Thomson et al. 2013). Stocking rates are therefore difficult to manipulate based on climate (resource) variability which results in severe and detrimental animal impact during droughts or dry seasons (Thomson et al. 2013, Van Rooyen 2017).

There is therefore the need to monitor the effect of climatic, anthropogenic, animal and management impacts on rangeland health and productivity across multiple land use systems in a consistent and cost-effective way.

1.2 Problem and hypotheses

A number of survey methods are used to assess various rangeland health indicators associated with both the herbaceous layer and the woody component (Herrick et al. 2005, Grant et al. 2011, Karl et al. 2011). Many of these monitoring techniques can be costly and/or time-consuming, and when used alone, may produce insufficient data on which broader management decisions can be based (Herrick et al. 2005, Godínez-Alvarez et al. 2009).

The survey method used by Peel et al. (2005) (referred to in this study as the 'Multiple Indicator Monitoring [MIM] method') has been applied throughout the Lowveld region of South Africa since 1989, and in Zimbabwe and Mozambique in recent years. This method integrates methodologies from a number of well documented and widely used survey techniques, which have been adapted in the MIM method to increase overall time effectiveness without an apparent compromise on scientific rigour and management efficacy (Peel et al. 2005). Observations during the use of the MIM method over the years suggest that the method can potentially be utilized as a cost-effective tool (both in terms of time and money) through simultaneous data collection of multiple related qualitative and quantitative indicators, which assist in understanding the complex changes associated with the health of savanna rangelands (Peel et al. 2005).

- **Null hypothesis 1 ($H_0$1):** There are no significant differences between the indicator values generated using the MIM method and:

  a. the Adapted Point-centred Quarter method for assessing

  - grass species composition, abundance and dominance;
  - basal cover;
  - woody species composition, abundance and dominance;
  - an index of tree height (structure); and
  - tree density.

  b. the Basal Cover method for assessing

  - grass species composition, abundance and dominance; and
  - basal cover.

  c. the Line-point Intercept method for assessing

- grass species composition, abundance and dominance;
- basal cover;
- woody species composition, abundance and dominance;
- an index of tree height (structure); and
- canopy cover.

d. the Disc Pasture Meter method for assessing grass biomass.

- **Null hypothesis 2 ($H_0$2):** There is no significant difference between the time it takes to complete the MIM method, and the:

  a. Adapted Point-centred Quarter method;

  b. Basal Cover method;

  c. Line-point Intercept method; and

  d. Disc Pasture Meter method.

1.3 Aims and objectives

The primary objective of this research is to assess the MIM method to determine its efficacy in capturing qualitative and quantitative spatial data in a time- and cost effective manner that is both scientifically sound (rigorous and accurate) and useful to policy makers and management. Achieving the desired objectives requires a direct and extensive comparison with other well-documented and widely used techniques, with evidence and support provided through statistical analysis.

The purpose of this study was not to determine whether the MIM method was the most objective or most efficient method in assessing rangeland health, but rather to determine how scientifically comparable it is to existing methods. Due to the different techniques of each of the methods, differences in results can be expected. The purpose; therefore, was to quantify the level of significance of any differences and what implications these differences may have on the use of methods for management decisions.

It is envisaged that the long-term data will serve as a framework for developing strategies aimed at improving range management and sustainable utilization while preventing habitat degradation through the use of a holistic, integrated conservation approach. Such an approach takes into account the health and status of numerous components of the entire social-ecological system and assists decision-makers with the planning of policies and interventions associated with sustainable rural development and mainstreaming biodiversity conservation within the communal farming landscape. Overall, the acquisition of such data will aid with future risk analysis, contributing to the greater interdisciplinary proactive

decision-making that is guided by the ‘One Health’ concept (Barrett et al. 2011, Zinsstag et al. 2011).

2. MATERIALS AND METHODS

2.1 Study region

2.1.1 Area location and size

The study was undertaken in the MSA in the Bushbuckridge Local Municipality, Ehlanzeni District Municipality in the Lowveld region of the province of Mpumalanga. The study area of the MCP of the University of Pretoria includes the Manyeleti and Andover provincial game reserves, as well as the communal living areas which lie between the two reserves. The communal areas consist of several villages belonging to the Mnisi Traditional Authority (MTA). The study area is bordered by Sandringham Game Reserve, Timbavati Private Nature Reserve and Hans Hoheisen private properties to the north, the KNP to the east, the Sabi Sand Wildtuin to the south, and the community of Acornhoek to the west (Figure 2).

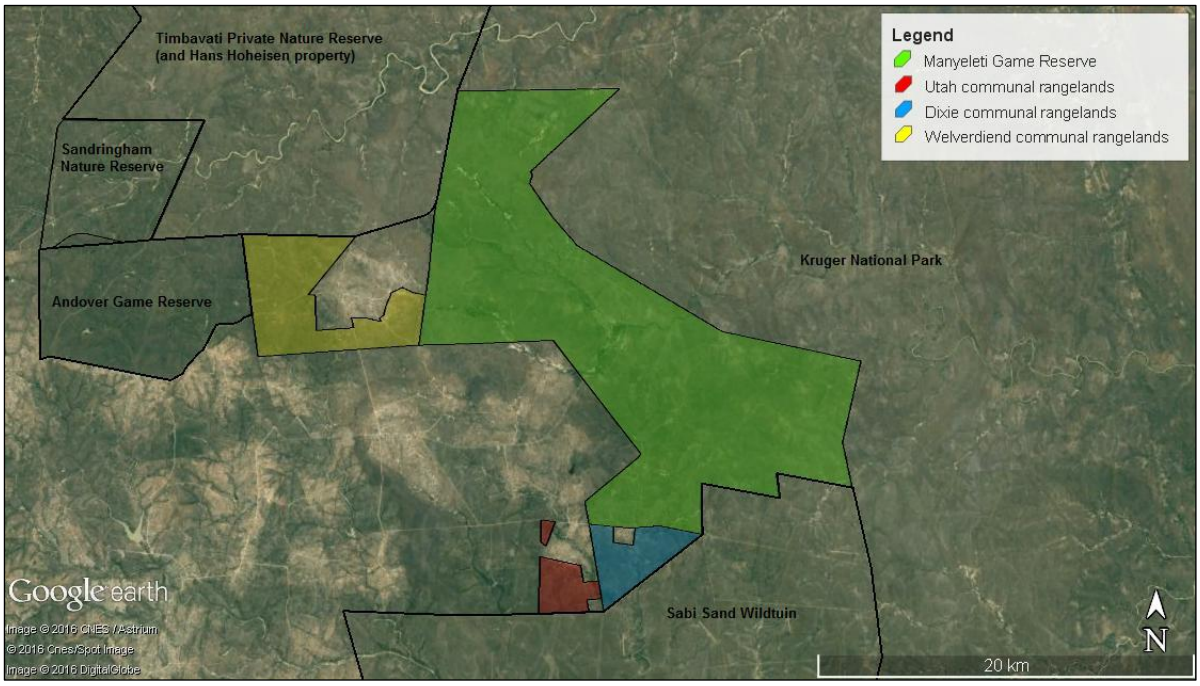

Figure 2: Location of the study area within the Mnisi Study Area (MSA), along with surrounding conservation (or protected) areas (Google Earth 2015).

The study covered two adjacent areas [approximately 28 169 hectares (ha) in total] with different land-use objectives. The MGR is approximately 22 618 ha in size, while the portion of the Mnisi communal area under investigation is approximately 5551 ha and consists of the villages of Utah (approximately 789 ha), Dixie (approximately 1287 ha) and Welverdiend (approximately 3475 ha).

Results have been displayed for the land uses combined as well as for the MGR and Mnisi communal rangelands (MCR) separately. It is important to make mention that no statistical analyses were performed on differences in findings between each of the land uses (see

section 2.3). The reason for including them separately was purely for observational purposes to see whether there are in fact differences across the land uses for the indicators of rangeland health recorded by each method. This is further discussed in section 5.

2.1.2 Ecological characteristics

Geology

The major geological formations within the study area are granites belonging to the Meinhardskraal and Drakensberg Groups (Müller 2009). Other minor geological formations include Timbavati Gabbro, which occupies a small section of the north-western region of the MGR (Müller 2009), as well as exposed sections of a large dolerite dyke which occurs throughout parts of the western portion of the reserve (Bredenkamp et al. 1983).

Soils

The western part of the study area is composed of regic sands, while the central and eastern regions of the study area are characterised by Glenrosa and/or Mispah soil forms (Müller 2009). The soils associated with the upland granitic areas are generally coarse in texture, sandy, acidic, leached and dystrophic; while the lowland areas contain soils which are fine in texture, neutral, calcareous and mesotrophic, and sometimes brackish (Bredenkamp et al. 1983).

Vegetation

The study area is characterised by vegetation associated with the Granite Lowveld (SVI 3) vegetation type (Mucina and Rutherford 2006). Vegetation in the MGR has previously been classified into seven major plant communities based on the ecological characteristics of the habitat in which they occur (Table 1) (Bredenkamp et al. 1983, Cronje et al. 2005).

Table 1: Classification of the major plant communities that occur in the MGR

Plant community	Habitat
Perotis patens / Terminalia sericea	Upland, sandy soils derived from granite
Euclea divinorum / Acacia nigrescens	Bottomland, black clayey soils derived from granite
Themeda triandra / Acacia gerrardii	Red clayey soils derived from granite and dolerite
Euclea divinorum / Albizia harveyi	Bottomland, black clayey, brackish granitic soils
Themeda triandra / Setaria incrassata	Black clayey soils derived from dolerite
Cardiospermum corindum / Acacia nigrescens	Rocky hills
Spirostachys africana / Diospyros mespiliformis	River banks

2.1.3 Background and management

The Manyeleti Game Reserve

The MGR is situated in the eastern region of the study area and is open to other surrounding protected areas to the north, east, south and north-west (Figure 2). It is a state-owned protected area, which was established in 1963 and by 1975 it had been expanded to its present size of 22 618 ha (Wolfaard 2013). In 1996 the reserve removed its fences with neighbouring reserves and the KNP.

Although the management objectives of protected areas can vary, most of the private and state-owned protected areas neighbouring the KNP adopt similar conservation policies as that of the KNP and South African National Parks (SANParks) (Peel et al. 1998, Rogers et al. 2008, Grant et al. 2011). The management objectives of many protected areas are developed largely on sound scientific evidence, and focus on the importance of conserving biodiversity and ecosystem function to promote ecosystem resilience (Walker 1995, Rogers et al. 2008, Grant et al. 2011, McGranahan and Kirkman 2013, Mori et al. 2013). Such outcomes are achieved by implementing SAM strategies that aim to increase spatial heterogeneity, which supports the functioning of important patterns and processes that drive ecological change and sustain ecosystem health (Walker 1995, Rogers et al. 2008, Grant et al. 2011, Roux and Foxcroft 2011, Mori et al. 2013).

The MGR contains a full complement of herbivore species and is managed as part of the open system, the Greater Kruger National Park and Kruger – Canyon Biosphere Reserve (MTPA 2015). The local Shangaan people belonging to the Mnisi chiefdom have, in recent years, laid claim to their ancestral land which includes the MGR and Andover Game Reserve.

The Mnisi Study Area (MSA)

The MCR area are wedged between the MGR and the Andover Game Reserve (Figure 2). Much of the land in this area is under communal tenure and used primarily for subsistence or small-scale livestock (cattle and goat) farming; however, many community members also practise small-scale agriculture (Twine 2005, Wolfaard 2013). Additionally, these rangelands provide a host of other natural resources that are used domestically or to generate income and supplement the livelihood of community members (Twine 2005).

The villages within this area are overseen by village Induna's (chiefs) who are responsible for most of the decision-making at village level. Each village has its own cattle committee (often referred to as a dip tank committee) who represent livestock farmers on the Community Development Forum (CDF). This committee is chaired by the village Induna, together with the local municipal ward councillor, and is largely responsible for decision-making with regards to grazing management.

Rangeland management interventions are either village-driven or driven by the Land Care division of the local Department of Agriculture, Rural Development, and Land Administration (DARDLA Ehlanzeni North). Furthermore, DARDLA have set up numerous land-care policies and support programmes which serve to aid community members in practising sustainable resource management.

The policies set up by SANParks and the Mpumalanga Tourism and Parks Agency (MTPA) include the establishment of buffer zones at the interface of the communal lands and conservation areas. Further policy has been established by dip tank committees from various villages, which bring about some form of control to disease transmission across the interface. The study area falls within the foot-and-mouth disease (FMD) protection with vaccination zone, which means that livestock movement within and out of the area is strictly regulated. Livestock movement and management, including dipping and all aspects of disease control, are managed by Mpumalanga Veterinary Services, more specifically by the State Veterinary Office (Orpen).

2.2 Experimental procedures

2.2.1 Desktop analysis

Long-term monitoring sites were established across the study area through a stratified sampling technique, where satellite imagery from Google Earth was used to identify homogenous vegetation units (HVUs) occurring across the different land-use types (Figure 3). HVUs occur throughout natural landscapes and possess particular ecological

characteristics based on their location in the landscape and the environmental conditions under which they exist. The sites were selected to represent both high-lying and low-lying areas, which differ in vegetation structure and composition.

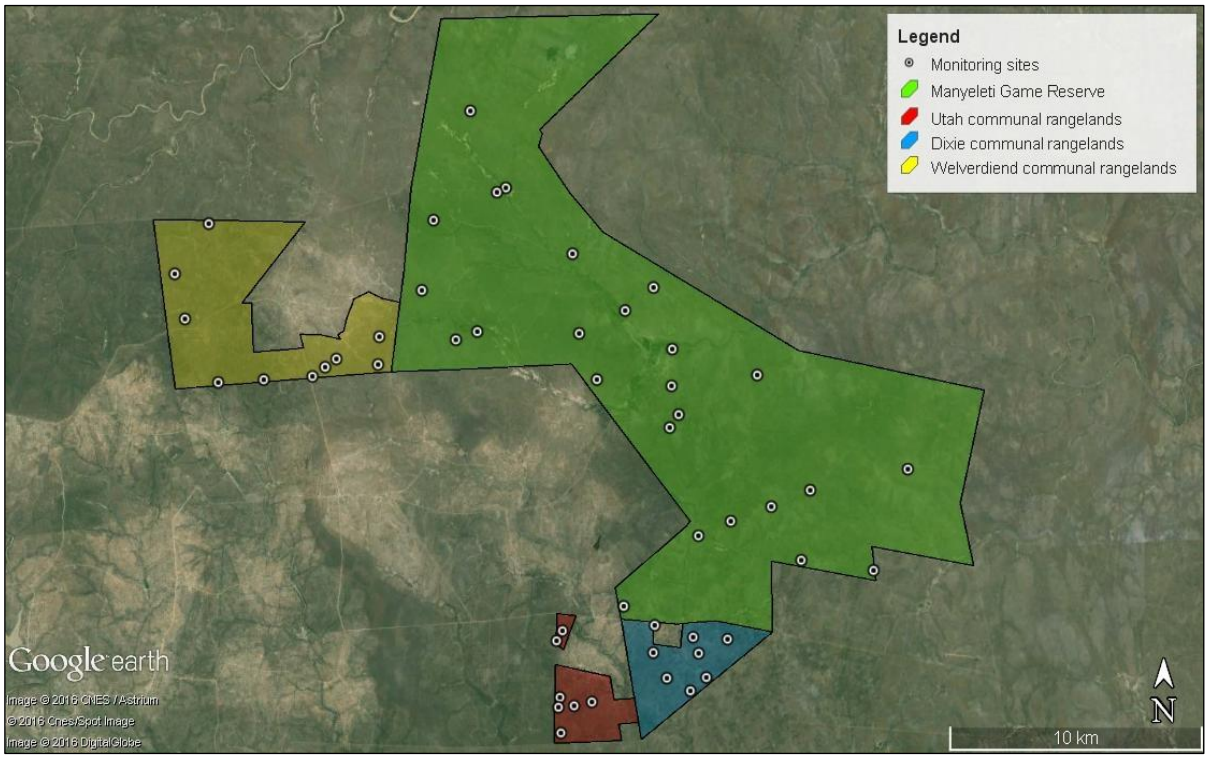

Figure 3: Location of the 50 fixed monitoring sites that were surveyed within the study area (Google Earth 2015).

2.2.2 Fieldwork phase

Data were collected from 50 monitoring sites within the study area (Figure 3). Twenty five of the monitoring sites occur in the MGR; and seven, eight and 10 monitoring sites occur in the communal rangelands of Utah, Dixie and Welverdiend, respectively. The primary objectives of the study required a site-by-site comparison of the indicators of rangeland health recorded using each of the methods, therefore site distribution did not require strict systematic selection across each of the land uses.

The site co-ordinates were uploaded to a Garmin Oregon 550 handheld GPS, and cardinal photographs of each site from the most recent years' monitoring period were uploaded to a Samsung S6 smartphone. The photographs provided a visual for comparison of any changes to a site with time, but also provide an indication of the direction of the tape and any significant elements which may assist in identification/recognition of the site (see Figure 4). This ensures that the same site location is monitored during each monitoring exercise.

Figure 4: Cardinal photographs illustrating the direction of the tape and any visible changes to the site 'Many 9' between 2011/2012 (left) and 2014/2015 (right).

2.2.3 The survey methods

The following section explains the origins and methodology associated with each of the survey methods that were assessed.

Multiple Indicator Monitoring Method

Background

The MIM method has been used to monitor rangelands across savanna and grassland biomes in the Highveld and Lowveld over the past 28 years and in Zimbabwe and Mozambique in the last decade (Sutherland and Peel 2011). The method combines a number of elements from widely used and well-documented survey methods to measure indicators associated with rangeland health, and assesses various parameters of both the herbaceous and woody component.

Methodology

Herbaceous survey

The herbaceous survey uses the nearest plant approach for recording species composition (Foran et al. 1978). The method also uses distance measures of Hardy and Tainton (1993) combined with tuft diameter measurements for an estimation of basal cover. Application of the nearest plant approach alone may provide limited information about the presence of perennial grasses, particularly during the wetter seasons when annual species are in their greatest abundance. Many annual grass species have opportunistic life cycles and tend to colonise bare patches, while perennial grasses are regarded as longer-lived species that contribute more sustainably to the herbaceous layer.

A 100m tape measure is used to establish a 25m x 25m transect (Figure 5). Measurements are recorded at each meter mark up until the 50m mark, thereafter measurements are

recorded at every odd meter mark (i.e. 51, 53, 55, etc.) to give a total minimum of 75 herbaceous meter-recordings per monitoring site.

A thin wire rod is dropped vertically to the ground at each of the meter marks, where the following herbaceous indicators of rangeland health are determined (Peel et al. 2005, Buitenwerf et al. 2011):

- The nearest rooted herbaceous individual:
- Should the closest individual be a perennial grass species, the 'annual' column in the datasheet is left blank and only the necessary measurements of the perennial species recorded.
- If the closest individual is an annual, it is recorded first. Thereafter the closest perennial grass species is recorded as a '2nd species'. Annual grass species are recorded by species name, herbaceous dicotyledons are recorded as 'forb', and species belonging to the family Cyperaceae are recorded as 'sedge'.
- Distance-to-tuft and tuft diameter measurements (mm) of the above-mentioned individuals are recorded to provide an estimation of herbaceous basal cover. Greater distance-to-tuft measurements and smaller tuft diameter measurements are indicative of poor basal cover.
- An estimate of percentage canopy cover is determined by extending a vertical projection above each meter mark. The growth of many palatable and productive grass species is associated with canopy cover.

Application of the DPM method (Bransby and Tainton 1977) in the MIM method entails recording grass biomass every 3m for the length of the 25m x 25m transect, giving a total of 33 measurements per monitoring site.

Woody survey

The woody survey component involves an area-based (plot-based) method that uses a 100m x 2m belt transect (using the same tape as the herbaceous survey). Area-based methods involve the establishment of one or more plots (belts or quadrats) of known area, which allows for the determination of plant density in terms of the number of plants per area (Cottam and Curtis 1956, Mitchell 2007). The following parameters associated with the woody component are measured and recorded:

- Species names of all woody species occurring within the 100m x 2m belt transect. (Note: recordings are taken of species occurring within each meter for the entire 100m transect, where species occurring on the inside of the transect from 0-25m and 50-75m are recorded and species occurring on the outside of the transect from 25-50m and 75-100m are recorded).
- Number of roots and stems per individual/s.

- The height class[1] of each recorded species.
- A scale to quantify elephant impact on the most dominant species at a site. The scale allows for the assessment of 10 individuals per height class and uses the Walker scale for percentage impact to the individuals (where 0 = 0%, 1 = 1-10%, 2 = 11-25%, 3 = 26-50%, 4 = 51-75%, 5 = 76-90%, 6 = 90-99%, and 7 = 100%) (Walker 1976). The impact can be described as having broken branch/es (BB), broken stem/s (BS), bent stem/s (BeS), stripped bark (B), being uprooted (U) or pushed over (PO).

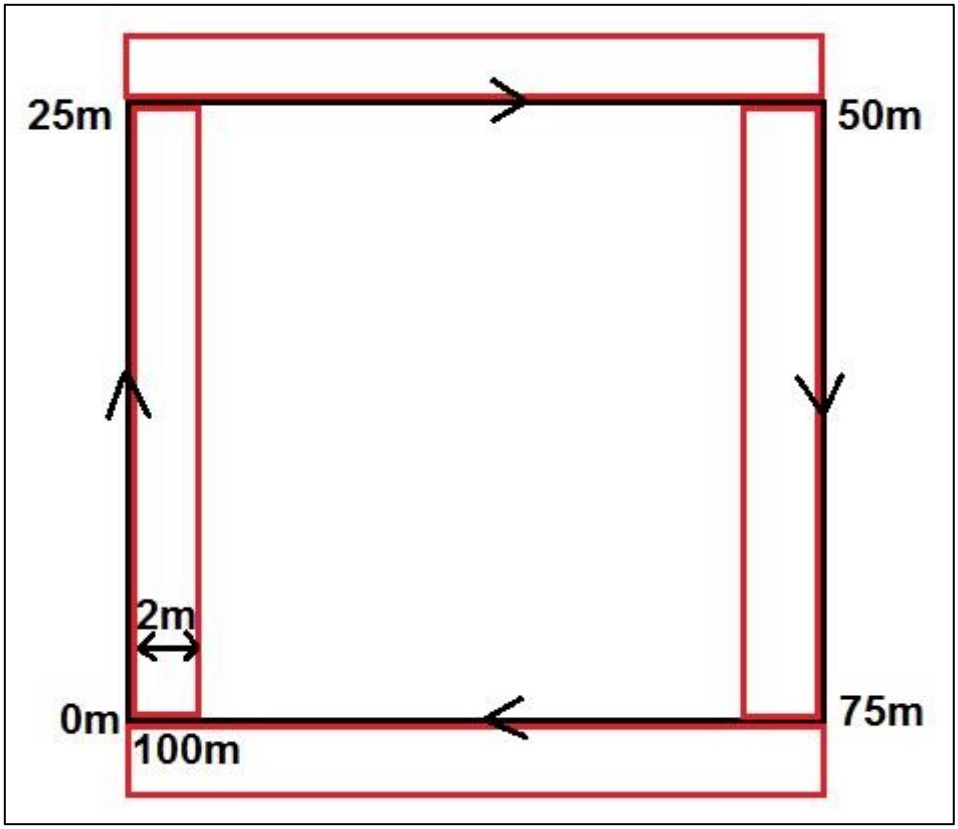

Figure 5: Layout of the MIM method.

Adapted Point-centred Quarter Method

Background

The traditional Point-centred Quarter method is a plotless method which has been refined since its early development, and is used for measuring parameters associated with

[1] Height class 1 = 0 – 1m

Height class 2 = 1.1 – 2m

Height class 3 = 2.1 – 5m

Height class 4 = > 5m

vegetation structure such as density, basal cover, biomass and canopy height (Cottam and Curtis 1956, Mitchell 2007). Plotless methods typically measure distances to trees along a transect within a random sample of vegetation, which allows for the determination of plant density in terms of the amount of area per plant (Mitchell 2007). Some believe plotless methods are more time efficient and require less equipment and man-power (Cottam and Curtis 1956, Mitchell 2007). What remains a question, however, is whether the trade-off of reduced sampling time compromises the rigour of results obtained.

More recently, the traditional form of the Point-centred Quarter method was adapted by Trollope et al. (2013) for conducting vegetation assessments to estimate the veld condition of shrub and tree communities in the thicket and savanna biomes in Africa. This technique is purported to overcome problems associated with area-based methods, which are considered to oversample small woody species and under sample taller trees or shrubs (Trollope et al. 2013).

Methodology

Herbaceous survey

The APCQ method consists of 8 circular quadrats (40m in diameter) which are arranged in two parallel and contiguous transects (Figure 6). The herbaceous component of the method uses the nearest plant approach to determine grass species composition (Foran et al. 1978), while also recording distance-to-tuft measurements to estimate basal cover of the grass sward (Hardy and Tainton 1993). The closest rooted grass species and distance-to-tuft measurements are recorded every 3m along the tape (at the centre of each of the two circular PCQ transects), giving a sub-total of 50 recordings per transect and a total sample size of 100 recordings at a monitoring site.

The APCQ method also measures grass biomass, where DPM measurements are recorded along both transects at 3m intervals for a total of 100 recordings (50 recordings per transect). Due to time constraints it was decided to omit the grass biomass component of the APCQ method.

Woody survey

Application of the woody survey allows for the determination of woody species composition, tree density (plants/ha), phytomass (tree equivalents/ha) and browsing potential ($BUha^{-1}$) (Trollope et al. 2013). Assessment of these parameters assists in determining optimal browser stocking rates, provides insight into the species composition of herbivores best suited for an area and plays an important role in predicting fire intensities for the development of fire regimes, which can be used to manage bush encroachment (Trollope et al. 2013).

For all quarters of each circular quadrat, various parameters of the nearest rooted tree or shrub '< 2m' in height, '> 2m' in height and the 'tallest tree or shrub' within 20m of the

central sampling point are recorded. These parameters include the species name, distance from central sampling point, overall canopy height and height of lowest browseable material (Trollope et al. 2013).

In cases where the tree or shrub species in the '> 2m' height class was also the 'tallest tree', that same individual was recorded for both the '> 2m' height category and the 'tallest tree' height category. In cases where there were no trees or shrubs in the '< 2m' and '> 2m' height categories within 20m of the sampling point, a dash was used on the datasheet to represent no plants.

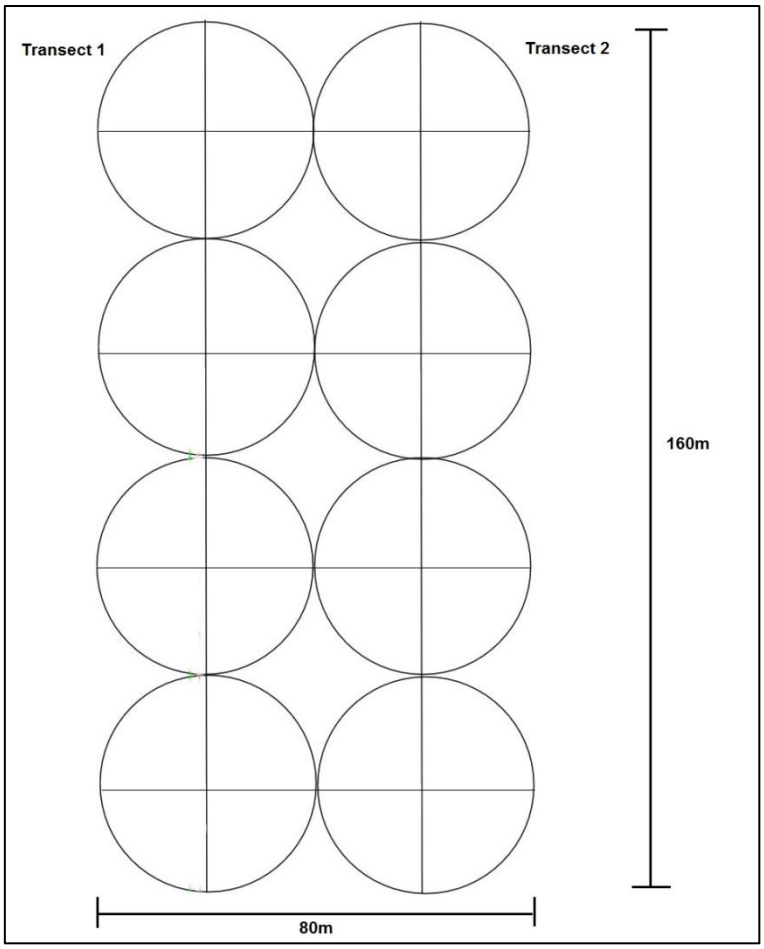

Figure 6: Layout of the APCQ method.

Basal Cover Method

Background

A primary indicator used to determine the herbaceous structure of an area is the estimation of basal cover (Hardy and Tainton 1993, Buitenwerf et al. 2011). A relationship exists between basal cover and ecosystem functions such as reduced rates of water evaporation and run-off and the facilitation of water infiltration (Buitenwerf et al. 2011). At a finer scale, high grass production and soil nitrogen concentrations are often associated with greater estimates of basal cover (Buitenwerf et al. 2011).

A traditional method of estimating basal cover is the strike approach, where the number of tuft strikes (direct tuft hits) are expressed proportionately against the total number of observations at a monitoring site (Hardy and Tainton 1993). This approach for estimating basal cover has two limitations, which potentially act as a source of error. The first being that large sample sizes (between 100 and 300 observations per monitoring site) are required for acceptable levels of confidence of the estimate, as the probability of recording a strike is relatively low; and the second limitation being the significant amount of variation that exists between samplers in the identification of a strike (Hardy and Tainton 1993). In order to avoid these limitations and provide a more elaborate estimation of basal cover, the method was modified to include recording the distance of the nearest rooted grass species from the sampling point, along with the tuft diameter (Hardy and Tainton 1993).

Methodology

The technique uses a 20m x 20m plot (Figure 7), where a thin wire rod is dropped vertically every 2m for a total of 100 recordings, and the species name of the nearest rooted grass tuft is recorded at each sampling point along with the distance-to-tuft (*D*) and tuft diameter (*d*) measurements of that species (Hardy and Tainton 1993). Due to time constraints it was decided to improvise the plot within the 25m x 25m quadrat (of the MIM method), using strides to estimate each 2m sampling point.

In order to eliminate observer bias of what is perceived as a strike, each time the rod struck a plant base the distance was recorded as 10mm as opposed to 0mm (Hardy and Tainton 1993). Tuft diameter was measured by calculating the mean diameter from measures of the longest and shortest axes of the grass tuft (Hardy and Tainton 1993). If the centre of the tuft was dead but the circumference appeared to be alive, the entire diameter of the tuft was recorded (Hardy and Tainton 1993). If tufts occurred in separate units (separated by a 10mm gap) then the diameter of the unit closest to the sampling point was measured and recorded (Hardy and Tainton 1993).

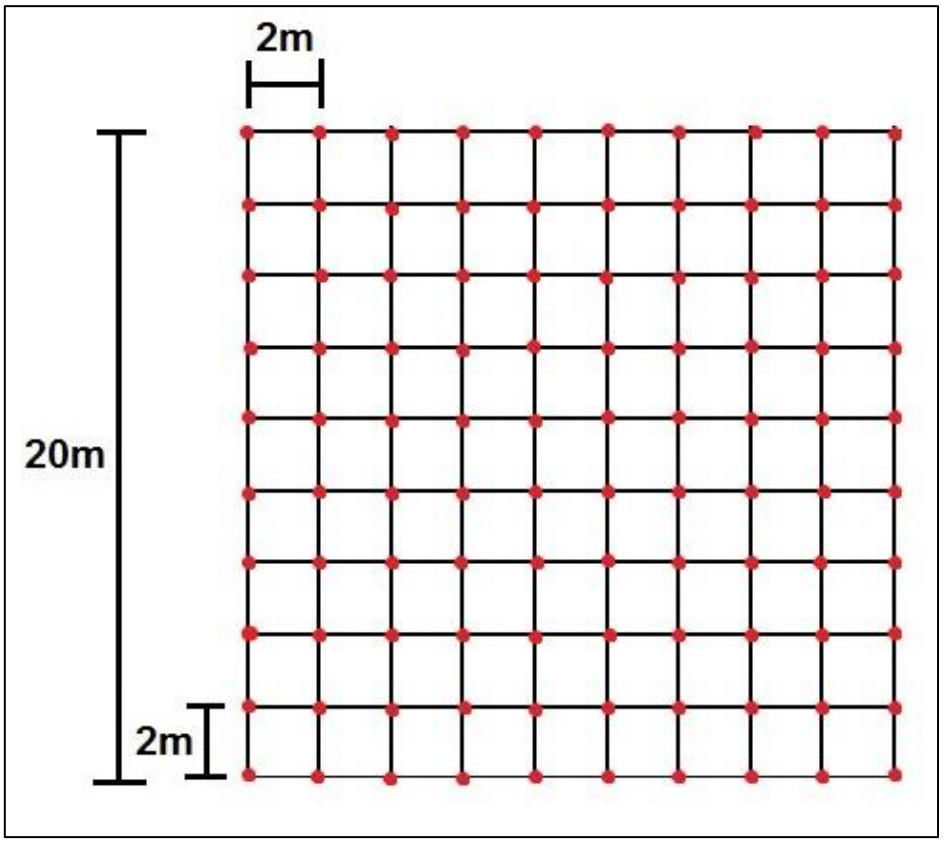

Figure 7: Layout of the BC method.

Line-point Intercept Method

Background

The LPI method is a survey technique used to detect changes in vegetation composition, structure, cover, density and browse availability (Ripley et al. 1960, Ripley et al. 1963, Muttlak and Sadooghi-Alvandi 1993, Herrick et al. 2005, Godínez-Alvarez et al. 2009, Toledo et al. 2010). Some versions of the method have been adapted to include the determination of soil surface properties; which provide an indication of soil stability, erosion rates, infiltration rates and the resilience of a particular site to degradation (Herrick et al. 2005).

The LPI method combines the assessment of some parameters from the similarly named point intercept method (Jonasson 1988) and the line intercept method (Brun and Box 1963). The point intercept method is a non-destructive technique used to estimate plant biomass, where a pin or straight thin rod is dropped numerous times throughout a stand of vegetation and recordings are taken each time the pin makes contact with a plant (Jonasson 1988).

The line intercept method has been modified throughout the years (Brun and Box 1963, Ripley et al. 1963) since its development in earlier days by Canfield (1941). There are numerous variations in the methodology of this technique, which can be applied in different vegetation types and in accordance with the desired outcomes of the particular research in

question (Ripley et al. 1963). For most part, however, the methodology remains relatively consistent.

Methodology

The technique used in this study followed the methodology of an alternative LPI method as described by Herrick *et al.* (2005), which includes a height component for woody species. The process entailed the establishment of a straight line transect as close as possible to the ground, using a 100m measuring tape which was anchored at both ends and passed under or through shrubs and trees which were intercepted by the tape (Herrick et al. 2005).

Beginning at 1m and progressing along the transect to 100m, a straight stick or thin wire rod (approximately 1mm in diameter) was dropped from a standard height (approximately 100mm) to create a vertical projection which passed perpendicularly through each meter mark to the ground. The first leaf, stem or plant base of a tree, shrub or herbaceous individual that intercepted the wire rod was recorded at each meter mark, giving a total sample size of 100 recordings per monitoring site. If no species of graminoid, shrub or tree within the lower or middle canopy intercepted the wire rod, then the vertical projection was extrapolated upwards by the observer to determine whether taller trees or shrubs were intercepted.

Species names of graminoids, shrubs and trees were recorded, while forbs and sedges were represented by ticking the 'Forb/sedge' column of the datasheet. Heights of all woody species that intercepted the vertical projection first were estimated in meters. Vegetation cover could be estimated using the strike approach, where the number of times a plant base is intercepted by the rod is expressed as a proportion of the total number of recordings at a site. If the rod was not intercepted by a stem, leaf or plant base of either of the plant forms, the column 'None' was marked with a tick for that particular meter mark.

Disc Pasture Meter Method (100 recordings)

Background

There are a number of methods that can be used to estimate grass biomass; however, a method which is widely used in various savanna rangelands across South Africa is that of the DPM method (Bransby and Tainton 1977, Trollope and Potgieter 1986, Govender et al. 2006, Zambatis et al. 2006). The estimation of grass biomass assists rangeland managers with important decision-making processes such as the determination of fire burning regimes and animal stocking rates (Bransby and Tainton 1977, Trollope and Potgieter 1986, Govender et al. 2006, Zambatis et al. 2006).

The DPM was developed by Phillips and Clarke (1971) as a rapid non-destructive sampling method for measuring the height of compressed grass in a particular area, wherefrom available grass standing crop can be estimated with the use of a calibrated regression equation (Bransby and Tainton 1977, Wentzel et al. 1991, Zambatis et al. 2006).

Trollope and Potgieter (1986) calibrated the DPM method and developed a regression equation for the estimation of grass biomass for use in the KNP, which was used for this study. A study undertaken in the south-eastern section of the KNP by Wentzel et al. (1991) reported using a total of 100 DPM recordings per monitoring site, where 25 points were sampled along each of four parallel transect lines (10m apart) at two-step intervals. An adequate sample size ensures a high level of confidence that the variation in estimated grass biomass is due to variation in disc height (Trollope and Potgieter 1986).

Methodology

Application of the DPM in the field entailed dropping the disc at each meter mark along the straight line tape (for a total of 100 recordings) and recording disc-height readings for the estimation of grass biomass (Zambatis et al. 2006). The principle behind the concept is that of a constant mass falling from a constant height, where the settling height of the disc represents the grass biomass (Bransby and Tainton 1977, Govender et al. 2006). In order to avoid error in estimating grass biomass, the DPM should be dropped on even ground and should avoid non-herbaceous components or lignified tall-grass tufts (Zambatis et al. 2006).

2.2.4 Sequence of data collection

1. Data were collected at each monitoring site using the various survey methods (as described in section 2.2.3) in accordance with the following sequence: Set tapes 1 & 2 – once the site had been located and identified, the 100m measuring tape (tape 1) and 50m measuring tape (tape 2) were set out to form transect 1 (T1) of the Adapted Point-centred Quarter (APCQ) method. The 100m tape alone provided the transect line for the Line-point Intercept (LPI) method and the Disc Pasture Meter (DPM) method.
2. Conduct LPI method - the LPI method was conducted starting from 1m and progressing to the end of the 100m tape.
3. Measure grass biomass (100 recordings) – DPM recordings were taken at each meter mark, working from 100m back to 1m.
4. Conduct herbaceous survey for APCQ (T1) method – the herbaceous parameters were recorded from 3m to the end of the 150m tape.
5. Conduct woody survey for APCQ (T1) method – the 140m mark was marked with a peg and tape 2 was rolled up and used to measure the distances to tree species within each quarter of the point-centres.
6. Set tape 3 – the 100m tape was set along the same transect line (for the first 25m) to form a 25m x 25m quadrat (tape 3), which was used for undertaking the MIM method as well as the Basal Cover (BC) method. The tape always proceeded to the right, unless it was not possible or illogical i.e. due to the presence of a road, riverbed or similar.
7. Conduct MIM method.
8. Conduct BC method.

9. Set tape 4 – the 100m tape and 50m tape (both combined represented tape 4) were used to set up T2 of the APCQ method.
10. Conduct herbaceous survey of APCQ (T2) method.
11. Conduct woody survey of APCQ (T2) method.

The layout of each of the methods in relation to one another is shown in **Figure 8**.

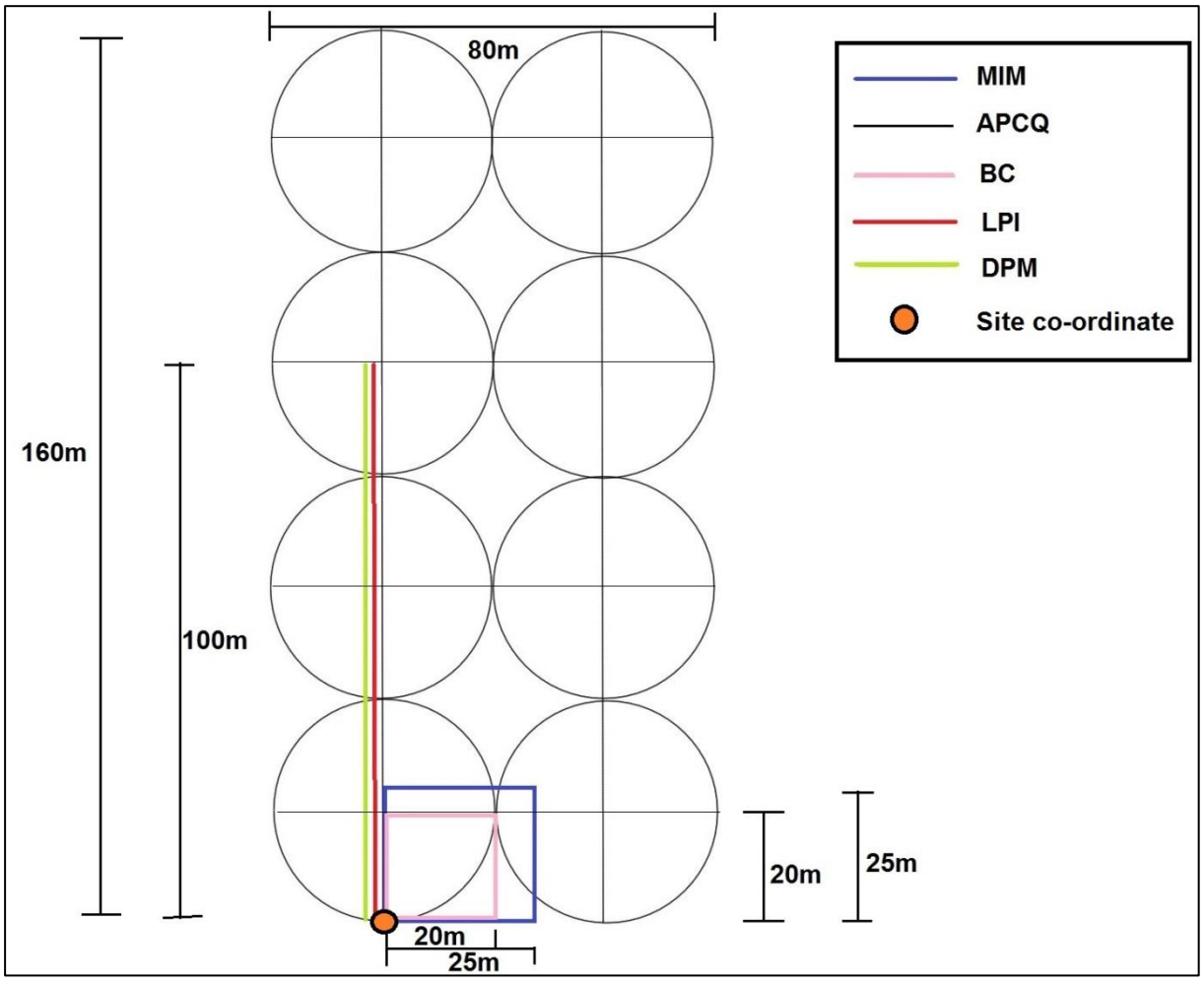

Figure 8: Layout of each of the methods in relation to one another.

2.2.5 Measurement of time

Fieldwork phase

The time to undertake each survey method was recorded using a Sportline 240 stopwatch. Separate times were taken for setting the tape for each of the survey methods, which were combined with the time taken to undertake each of the components of each method (i.e. herbaceous and/or woody), giving a mean time for the fieldwork phase of each method.

Data processing

The time taken to process data of each survey method was recorded for five monitoring sites from each locality, giving a total sample size of 20 monitoring sites for comparison of the mean time taken to enter data of each method.

All times, including those recorded during the fieldwork phase, were converted to decimal minutes for comparison.

2.3 Data processing and analysis

All data were processed using Microsoft Excel 2010. Data processing for each of the methods was divided into the following components where applicable:

i. Herbaceous
ii. Woody
iii. Time

All data were compared statistically and tested for significance using R software (R Core Team 2017). The analysis compared data from the APCQ method, BC method, LPI method and DPM method with data recorded by the MIM method for both land uses combined as well as the MGR and MCR separately. Note: no statistical analyses were performed that compared differences between the combined land uses, MGR and MCR for the herbaceous, woody or time data. The following tests were performed separately on the herbaceous data, the woody data and the time data: a Shapiro test was run on all variables to check for normality at a 95% confidence level. All P-values were <0.05, indicating that none of the distribution was normally distributed, and consequently the Wilcoxon Rank-Sum Test was used for comparison. All results were analysed and discussed using the mean and standard error, and differences were tested for significance at a 95% confidence level ($P < 0.05$).

2.4 Analytical procedures

2.4.1 Grass species dominance

The five dominant grass species recorded using each of the survey methods, both within the MGR and MCR, were determined and used to generate a list of dominant grass species (nine species) recorded across both land uses by all methods. The results of the five dominant grass species and remaining four “other dominant” grass species are shown in section 3.1.2 as they formed an important component of the hypotheses testing, however for the purposes of this study only the five dominant grass species recordings formed the basis of the discussion.

2.4.2 Woody species dominance

The five most dominant woody species recorded by each of the survey methods, both within the MGR and MCR, were determined and used to generate a list of dominant woody species (14 species) recorded across both land uses by all methods. The results of the five dominant

woody species and remaining nine “other dominant” woody species are shown section 3.2.2 as they formed an important component of the hypotheses testing, however for the purposes of this study only the five dominant woody species recordings formed the basis of the discussion.

2.4.3 Index of tree height (structure)

The height (m) of individual trees recorded using the APCQ and LPI methods were converted into the corresponding height classes of the MIM method and compared across both land uses, as well as for the MGR and MCR separately.

2.4.4 Tree density

The tree density estimated using the MIM method at each monitoring site was calculated by converting the number of roots recorded by 50m. The area of the belt transect is equal to 25m x 2m x 4 sides = 200m^2. In order to convert the area to 10 000m^2 (1 ha) it is necessary to multiply the area (and the number of roots) by a factor of 50, giving the following equation:

Tree density (plants/ha) = No. of roots x 50m

The tree density estimated using the APCQ method is calculated by dividing 10 000m^2 (1 ha) by the mean corrected distance squared (D^2). The mean corrected distance is the average of the distances of the closest trees in each height class to the sampling point. The calculation is performed using the following equation:

Tree density (plants/ha) = 10 000m^2/D^2

2.4.5 Other recorded parameters

The following calculations can be performed for each of the methods using the data collected from a monitoring site (Table 2):

Table 2: Indicators/ parameters of rangeland health recorded and analysed in each of the survey methods (Recorded = "X"; Not recorded = "-")

		Survey method					
		MIM	**APCQ**	**BC**	**LPI**	**DPM**	**Calculation**
Indicators/ parameters of rangeland health	**Species composition (herbaceous)**	X	X	X	X	-	Recordings of each herbaceous species
	Mean number of species recorded (herbaceous)	X	X	X	X	-	$\frac{\text{No. of recordings of each herbaceous species}}{\text{Total no. of species recorded}}$
	% herbaceous basal strikes	X	X	-	X	-	$\frac{\text{No. of strikes}}{\text{Total no. of herbaceous recordings}}\ x\ 100$
	Mean distance-to-tuft (mm)	X	X	X	-	-	$\frac{\sum \text{distance} - \text{to} - \text{tuft recordings}}{\text{Total no. of herbaceous recordings}}$
	Mean tuft diameter (mm)	X	-	X	-	-	$\frac{\sum \text{tuft diameter recordings}}{\text{Total no. of herbaceous recordings}}$
	Species composition (woody)	X	X	-	X	-	Recordings of each woody species
	Mean number of species recorded (woody)	X	X	-	X	-	$\frac{\text{No. of recordings of each woody species}}{\text{Total no. of species recorded}}$
	% Canopy cover	X	-	-	X	-	$\frac{\text{No. of canopy strikes}}{\text{Total no. of recordings}}$ x 100
	Grass biomass (kgha^{-1}); where *y* = disc height recordings	X	X[b]	-	-	X	$\left[\left(\sqrt{\frac{\Sigma(y)}{33}}\right)(2260)\right] - 3019$

[b] This study did not record grass biomass during application of the APCQ method, however the column has been checked with an 'X' as normal application of the method does record grass biomass

3. RESULTS

3.1 Herbaceous component

3.1.1 Species detection

The total number of grass species recorded across both land uses with the MIM, APCQ, BC and LPI methods was 53. The total number of grass species recorded using the various survey methods in both the MGR and MCR was 40 each (Table 3).

Table 3: Total number of grass species recorded using the various survey methods across both land uses and for the MGR and MCR separately

	MIM	APCQ	BC	LPI	Total
Number of species recorded across both land uses	39	47	43	41	53
Number of species recorded in the MGR	29	36	30	31	40
Number of species recorded in the MCR	30	35	34	31	40

Combined land uses

The MIM method recorded 74% of the total number of grass species that were recorded using all methods across both land uses (n = 50 sites) (Figure 9). The APCQ method recorded 89% of the total number of grass species (n = 50 sites), the BC method recorded 81% (n = 46 sites) and the LPI method recorded 77% (n = 50 sites) (Figure 9).

The mean number of grass species recorded per site by the MIM method across both land uses was 8.8 species (± 2.3 SD) (Figure 10). The APCQ method recorded significantly more grass species than the MIM method (mean = 9.6 species ± 2.4 SD, P = 0.02); there were no significant differences in the mean number of grass species recorded by the BC method (mean = 8.2 species ± 2.8 SD, P = 0.1) and the LPI method recorded significantly fewer grass species than the MIM method across both land uses (mean = 7.7 species, ± 2.0 SD, P = 0.002) (Figure 10).

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The MIM method recorded 73% of the total number of grass species that were recorded using all methods in the MGR (n = 25 sites) (Figure 9). The APCQ method recorded 90% (n = 25 sites), the BC method recorded 75% (n = 21 sites) and the LPI method recorded 78% (n = 25 sites) (Figure 9).

The mean number of grass species recorded per site using the MIM method in the MGR was 8.4 species (± 2.6 SD) (Figure 10). There were no significant differences in the mean number of grass species recorded by the APCQ method (mean = 9.0 species ± 2.8 SD, P = 0.4); the BC

method recorded significantly fewer grass species (mean = 7.2 species ± 2.8 SD, P = 0.01); and there were no significant differences in the mean number of grass species recorded by the LPI method (mean = 7.8 species ± 2.2 SD, P = 0.2) (Figure 10).

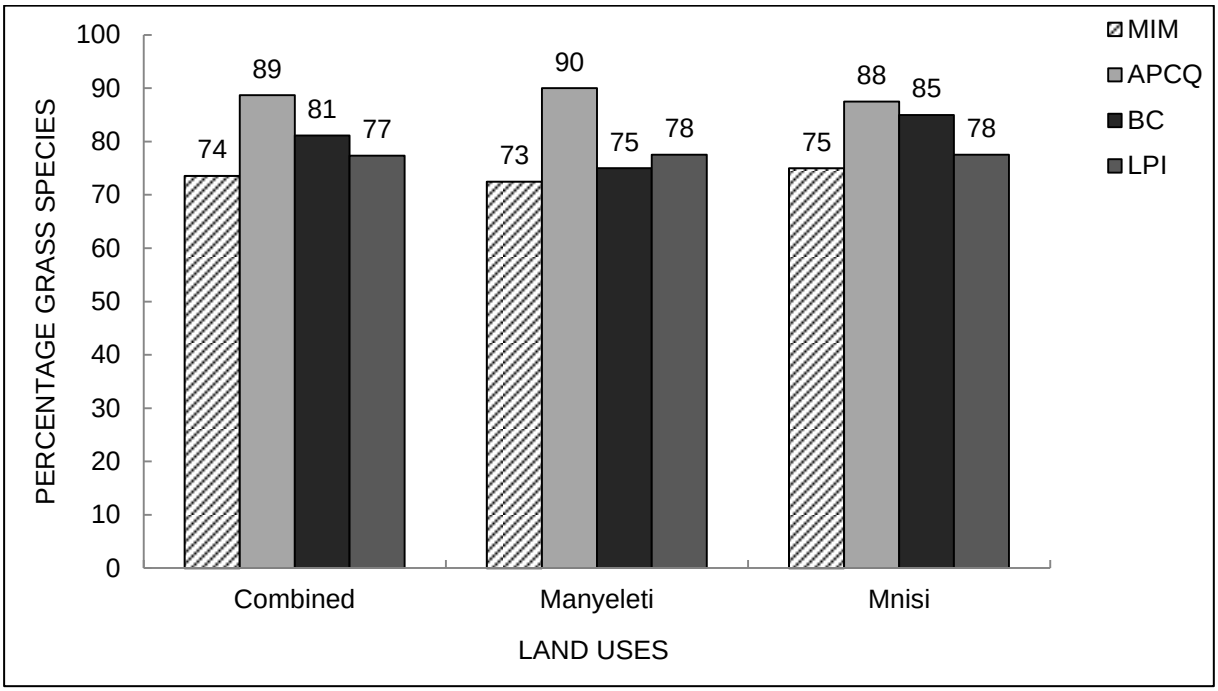

Figure 9: The percentage of grass species recorded using each survey method relative to the total number of grass species recorded using all methods across each of the land uses.

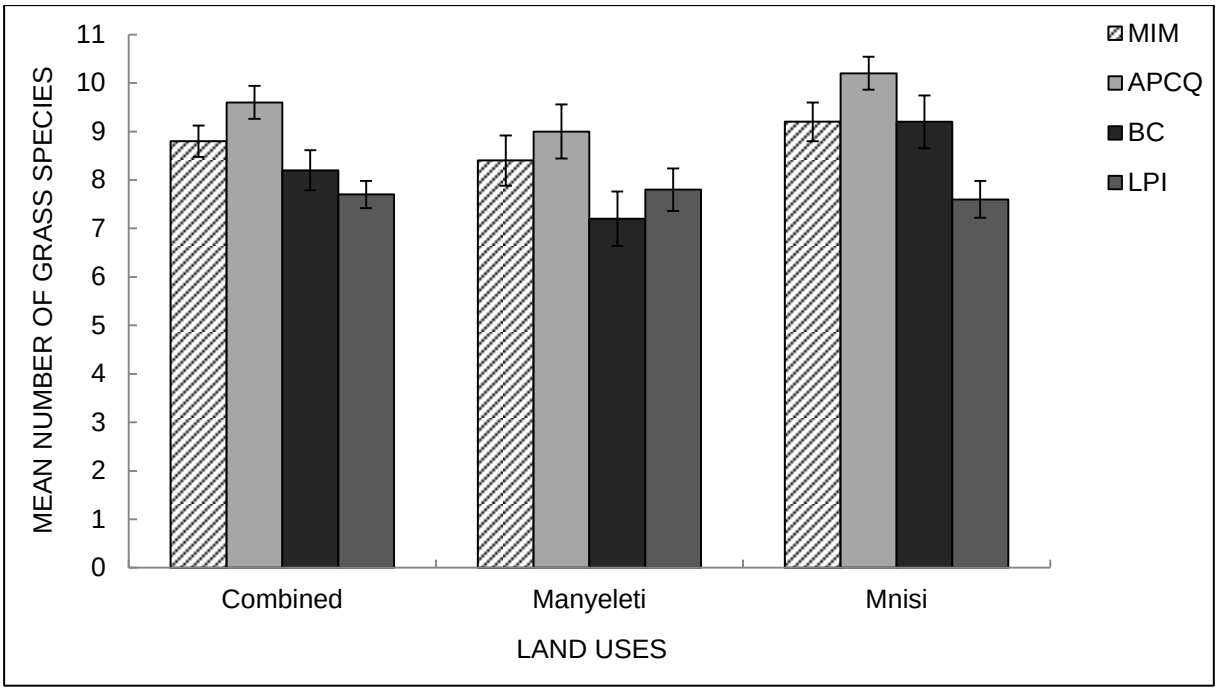

Figure 10: Mean number of grass species recorded using each of the survey methods across the land uses (where error bars reflect Standard Error).

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The MIM method recorded 75% of the total number of grass species that were recorded using all methods in the MCR (n = 25 sites) (Figure 9). The APCQ method recorded 88% of the total number of grass species (n = 25 sites), the BC method recorded 85% (n = 21 sites) and the LPI method recorded 78% (n = 25 sites) (Figure 9).

The mean number of grass species recorded per site using the MIM method in the MCR was 9.2 species (± 2.0 SD) (Figure 10). The APCQ method recorded significantly more grass species than the MIM method (mean = 10.2 species ± 1.7 SD, P = 0.01); there were no significant differences in the mean number of grass species recorded by the BC method (mean = 9.2 species ± 2.5; and the LPI method recorded significantly fewer grass species (mean = 7.6 species ± 1.9 SD, P = 0.03) (Figure 10).

3.1.2 Grass species composition, abundance and dominance

Combined land uses

Five dominant grass species

The five dominant grass species recorded per site using the MIM method across both land uses combined were *Panicum maximum* (mean = 22.3 individuals ± 13.6 SD), *Digitaria eriantha* (mean = 19.6 individuals ± 15.3 SD), *Urochloa mosambicensis* (mean = 8.7 individuals ± 8.1 SD), *Aristida adscensionis* (mean = 4.3 individuals ± 5.9 SD) and *Aristida congesta var. barbicollis* (mean = 3.8 individuals ± 5.5 SD) (Table 4 and Figure 11).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded using the APCQ method reveals that the latter method recorded significantly more *Panicum maximum* (mean = 28.6 individuals ± 15.1 SD, $P < 0.001$) (Table 4 and Figure 11). There were no significant differences in the mean number of *Digitaria eriantha* recorded by the APCQ method (mean = 23.7 individuals ± 16.7 SD, $P = 0.05$); there were no significant differences in the mean number of *Urochloa mosambicensis* recorded by the APCQ method (mean = 10.0 individuals ± 8.7 SD, $P = 0.1$); there were no significant differences in the mean number of *Aristida adscensionis* recorded by the APCQ method (mean = 3.6 individuals ± 4.7 SD, $P = 0.2$); and there were no significant differences in the mean number of *Aristida congesta var. barbicollis* recorded by the APCQ method (mean = 4.4 individuals ± 5.2 SD, $P = 0.1$) (Table 4 and Figure 11).

Table 4: The mean number of individuals of the nine dominant grass species recorded using the various survey methods across both land uses (n = 50 sites). The five dominant grass species recorded for each of the survey methods are represented by [D]; while significant differences in comparison to the MIM method at a 95% confidence interval (P < 0.05) are represented by [*] and significant differences in comparison to the MIM method at a 99% confidence interval (P < 0.01) are represented by [**]

	MIM		APCQ			BC			LPI		
	Mean	SD	Mean	SD	P-value	Mean	SD	P-value	Mean	SD	P-value
Panicum maximum	22.3[D]	13.6	28.6[D]	15.1	<0.001[**]	26.3[D]	17.5	0.005[**]	13.7[D]	9.9	<0.001[**]
Digitaria eriantha	19.6[D]	15.3	23.7[D]	16.7	0.05	24.7[D]	19.9	0.05	15.0[D]	14.3	0.001[**]
Urochloa mosambicensis	8.7[D]	8.1	10.0[D]	8.7	0.1	11.4[D]	11.4	0.004[**]	6.2[D]	6.4	0.01[*]
Aristida adscensionis	4.3[D]	5.9	3.6	4.7	0.2	4.4[D]	5.4	0.05	1.2	2.1	<0.001[**]
Aristida congesta var. barbicollis	3.8[D]	5.5	4.4[D]	5.2	0.1	4.3	5.5	0.08	1.8	3.3	<0.001[**]
Brachiaria deflexa	2.5	3.5	4.0	5.4	0.01[*]	4.7[D]	7.0	<0.001[**]	0.5	1.2	<0.001[**]
Perotis patens	2.2	4.1	4.7[D]	7.2	<0.001[**]	2.0	4.8	0.8	1.2	2.2	0.01[*]
Themeda triandra	2.1	4.2	4.1	6.6	<0.001[**]	3.9	7.9	0.004[**]	4.7[D]	9.6	0.003[**]
Heteropogon contortus	0.9	1.6	2.1	2.8	0.003[**]	1.1	2.2	0.8	2.0[D]	3.7	0.05

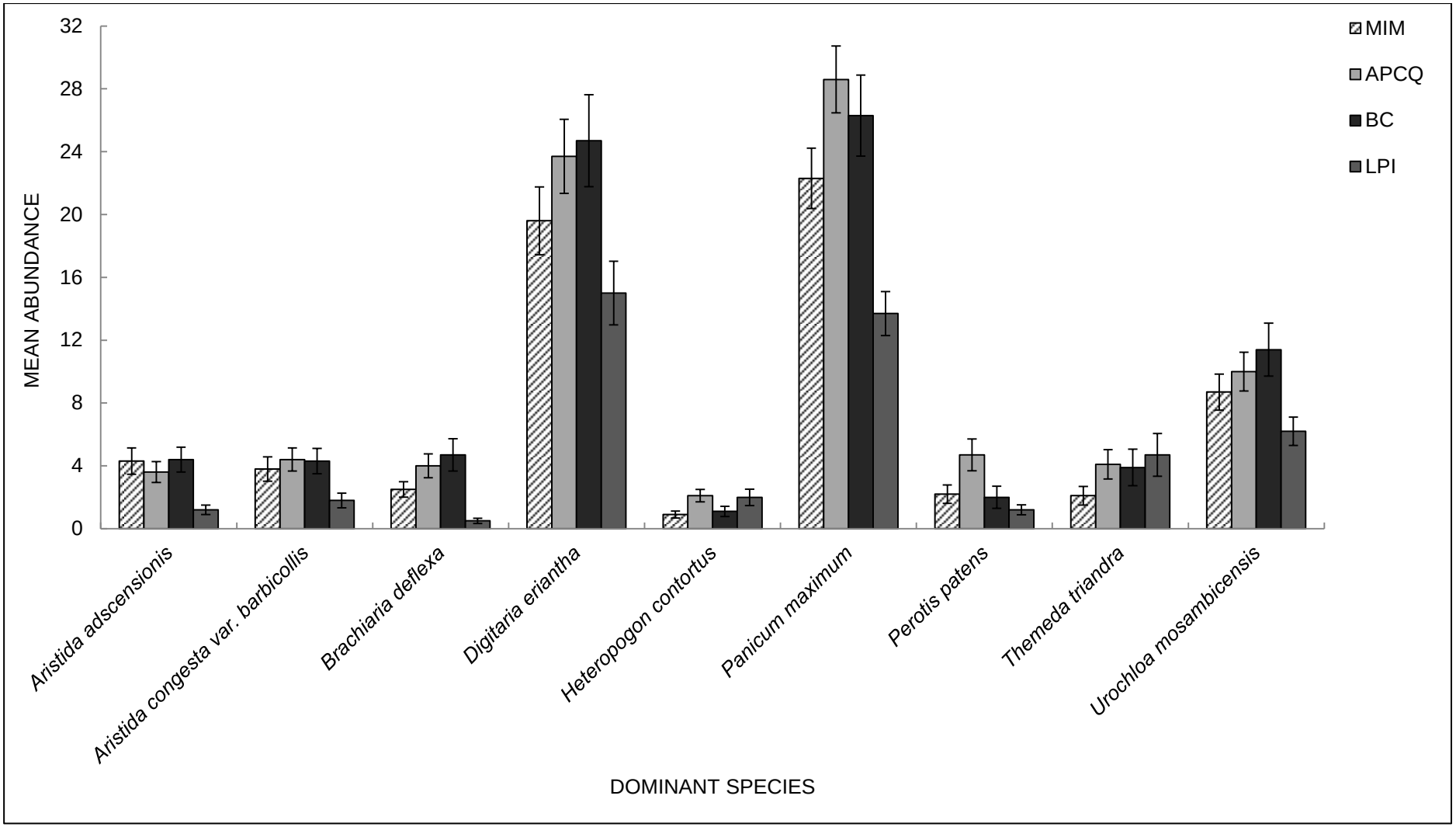

Figure 11: Mean abundance of dominant grass species recorded using each of the survey methods across both land uses (where error bars reflect Standard Error).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded with the BC method reveals that the latter method recorded significantly more *Panicum maximum* (mean = 26.3 individuals ± 17.5 SD, P = 0.005) (Table 4 and Figure 11). There were no significant differences in the mean number of *Digitaria eriantha* recorded by the BC method (mean =24.7 individuals ± 19.9 SD, P = 0.05); there were significantly more *Urochloa mosambicensis* (mean = 11.4 individuals ± 11.4 SD, P = 0.004); there were no significant differences in the mean number of *Aristida adscensionis* (mean = 4.4 individuals ± 5.4 SD, P = 0.05); and there were no significant differences in the mean number of *Aristida congesta var. barbicollis* recorded by the BC method (mean = 4.3 individuals ± 5.5 SD, P = 0.08) (Table 4 and Figure 11).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded with the LPI method reveals that the latter method recorded significantly fewer *Panicum maximum* (mean = 13.7 individuals ± 9.9 SD, P < 0.001) (Table 4 and Figure 11). There were significantly fewer *Digitaria eriantha* recorded than the MIM method (mean = 15.0 individuals ± 14.3 SD, P = 0.001); there were significantly fewer *Urochloa mosambicensis* (mean = 6.2 individuals ± 6.4 SD, P = 0.01); there were significantly fewer *Aristida adscensionis* (mean = 1.2 individuals ± 2.1 SD, P < 0.001); and there were significantly fewer *Aristida congesta var. barbicollis* (mean = 1.8 individuals ± 3.3 SD, P < 0.001) (Table 4 and Figure 11).

Other dominant grass species

The mean number of individuals of the remaining four "other dominant" grass species recorded using the MIM method across both land uses were *Brachiaria deflexa* (mean = 2.5 individuals ± 3.5 SD); *Perotis patens* (mean = 2.2 individuals ± 4.1 SD); *Themeda triandra* (mean = 2.1 individuals ± 4.2 SD); and *Heteropogon contortus* (mean = 0.9 individuals ± 1.6 SD) (Table 4 and Figure 11).

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site using the MIM method with the mean number of individuals recorded using the APCQ method yielded the following results (Table 4): there were significantly more *Brachiaria deflexa*, *Perotis patens*, *Themeda triandra* and *Heteropogon contortus* than the mean number of individuals recorded using the MIM method.

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the BC method yielded the following results (Table 4): there were no significant differences in the mean number of *Perotis patens* and *Heteropogon contortus*; and there were significantly more *Brachiaria deflexa* and *Themeda triandra* than the mean number of individuals recorded using the MIM method.

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the LPI method yielded the following results (Table 4): there were no significant differences in the mean number of *Heteropogon contortus*; there were significantly fewer *Brachiaria deflexa* and *Perotis patens*; and there were significantly more *Themeda triandra* than the mean number of individuals recorded using the MIM method.

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Five dominant grass species

The five most dominant grass species recorded per site using the MIM method in the MGR were *Digitaria eriantha* (mean = 28.4 individuals ± 13.5 SD), *Panicum maximum* (mean = 20.2 individuals ± 12.7 SD), *Urochloa mosambicensis* (mean = 8.4 individuals ± 8.8 SD), *Themeda triandra* (mean = 3.6 individuals ± 5.1 SD) and *Aristida congesta var. barbicollis* (mean = 1.8 individuals ± 2.7 SD) (Table 5 and Figure 12).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded using the APCQ method reveals that there were no significant differences in the mean number of *Digitaria eriantha* recorded by the APCQ method (mean = 33.5 individuals ± 15.1 SD, P = 0.2) (Table 5 and Figure 12). There were significantly more *Panicum maximum* recorded than the MIM method (mean = 27.4 individuals ± 17.0 SD, P = 0.008); there were significantly more *Urochloa mosambicensis* (mean = 10.9 individuals ± 8.7 SD, P = 0.04); there were significantly more *Themeda triandra* (mean = 7.4 individuals ± 7.8 SD, P < 0.001); and there were no significant differences in the mean number of *Aristida congesta var. barbicollis* recorded by the APCQ method (mean = 1.4 individuals ± 1.9 SD, P = 0.5) (Table 5 and Figure 12).

The comparison of the mean number of individuals of the five dominant grass species recorded per site by the MIM method with those of the means recorded using the BC method reveals that there were no significant differences in the mean number of *Digitaria eriantha* recorded by the BC method (mean = 32.4 individuals ± 19.7 SD, P = 0.07) (Table 5 and Figure 12). There were significantly more *Panicum maximum* recorded than by the MIM method (mean = 26.3 individuals ± 19.9 SD, P = 0.02); there were significantly more *Urochloa mosambicensis* (mean = 12.2 individuals ± 13.7 SD, P = 0.01); there were significantly more *Themeda triandra* (mean = 6.4 individuals ± 9.8 SD, P = 0.009); and there were no significant differences in the mean number of *Aristida congesta var. barbicollis* recorded by the BC method (mean = 2.0 individuals ± 3.5 SD, P = 0.5) (Table 5 and Figure 12).

Table 5: The mean number of individuals of the nine dominant grass species recorded using the various survey methods in the MGR (n = 25 sites). The five dominant grass species recorded for each of the survey methods are represented by ^{D}; while significant differences in comparison to the MIM method at a 95% confidence interval (P < 0.05) are represented by * and significant differences in comparison to the MIM method at a 99% confidence interval (P < 0.01) are represented by **

	MIM		APCQ			BC			LPI		
	Mean	SD	Mean	SD	P-value	Mean	SD	P-value	Mean	SD	P-value
Digitaria eriantha	28.4^{D}	13.5	33.5^{D}	15.1	0.2	32.4^{D}	19.7	0.07	23.7^{D}	13.2	0.04*
Panicum maximum	20.2^{D}	12.7	27.4^{D}	17.0	0.008**	26.3^{D}	19.9	0.02*	16.2^{D}	11.1	0.03*
Urochloa mosambicensis	8.4^{D}	8.8	10.9^{D}	8.7	0.04*	12.2^{D}	13.7	0.01*	8.4^{D}	7.4	0.6
Themeda triandra	3.6^{D}	5.1	7.4^{D}	7.8	<0.001**	6.4^{D}	9.8	0.009**	9.0^{D}	12.1	<0.001**
Aristida congesta var. barbicollis	1.8^{D}	2.7	1.4	1.9	0.5	2.0	3.5	0.5	0.6	0.8	0.01*
Heteropogon contortus	1.6	1.9	3.1^{D}	3.1	0.03*	1.7	2.4	0.6	3.4^{D}	4.7	0.08
Brachiaria deflexa	1.6	2.4	2.6	4.5	0.06	2.6^{D}	3.5	0.03*	0.1	0.4	0.002**
Aristida adscensionis	1.1	1.9	0.5	0.9	0.1	1.7	3.0	0.1	0.4	0.8	0.05
Perotis patens	0.3	0.6	0.9	2.4	0.1	0.1	0.3	0.09	0.3	1.4	0.6

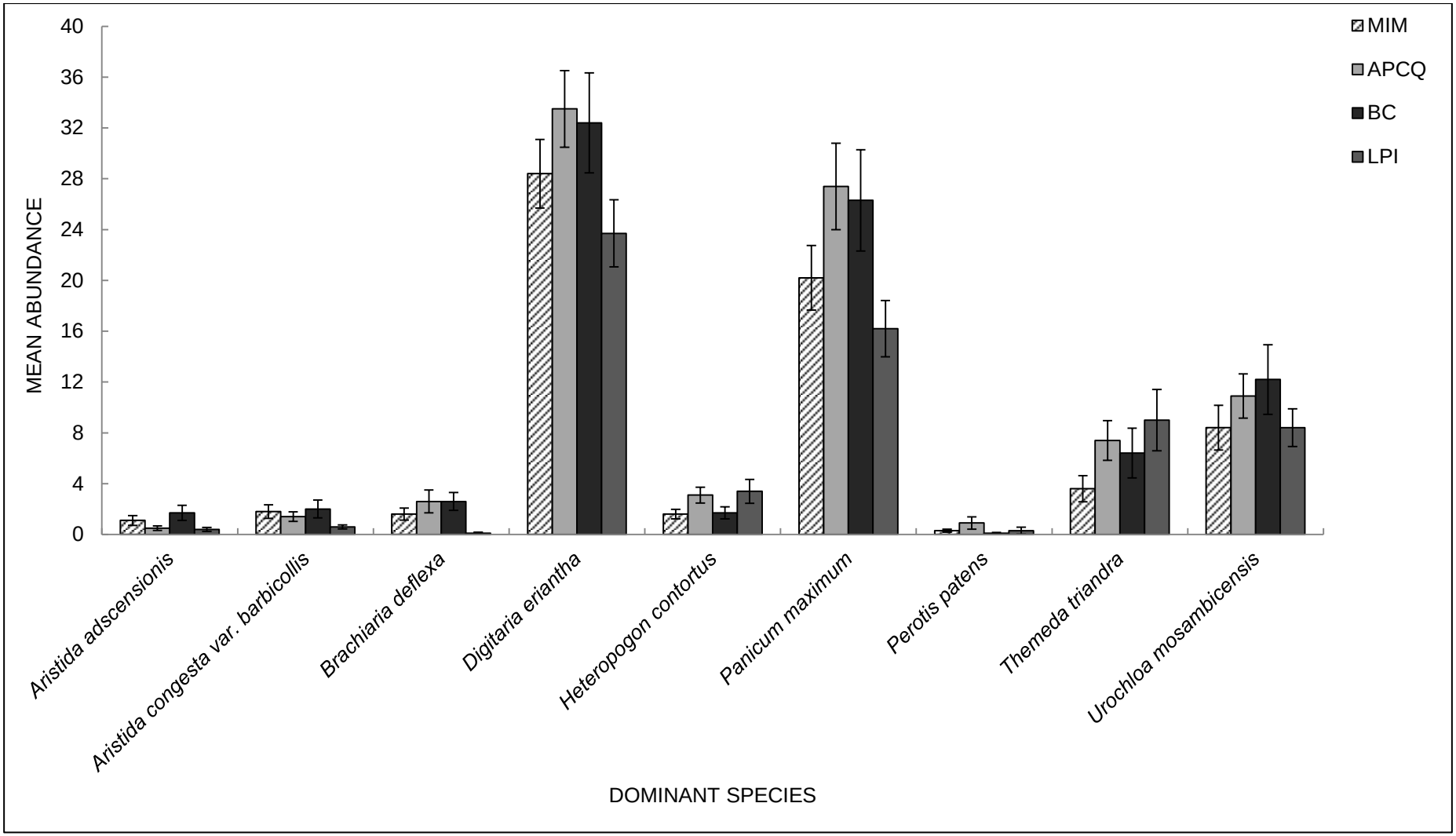

Figure 12: Mean abundance of dominant grass species recorded using each of the survey methods in the MGR (where error bars reflect Standard Error).

The comparison of the mean number of individuals of the five dominant grass species recorded per site by the MIM method with those of the means recorded using the LPI method reveals that the latter method recorded significantly fewer *Digitaria eriantha* (mean = 23.7 individuals ± 13.2 SD, P = 0.04) (Table 5 and Figure 12). There were significantly fewer *Panicum maximum* recorded than by the MIM method (mean = 16.2 individuals ± 11.1 SD, P = 0.03); there were no significant differences in the mean number of *Urochloa mosambicensis* recorded by the LPI method (mean = 8.4 individuals ± 7.4 SD); there were significantly more *Themeda triandra* (mean = 9.0 individuals ± 12.1 SD, P < 0.001); and there were significantly fewer *Aristida congesta var. barbicollis* recorded by the LPI method (mean = 0.6 individuals ± 0.8 SD, P = 0.01) (Table 5 and Figure 12).

Other dominant grass species

The mean number of individuals of the remaining four "other dominant" grass species recorded using the MIM method in the MGR were *Brachiaria deflexa* (mean = 1.6 individuals ± 2.4 SD); *Heteropogon contortus* (mean = 1.6 individuals ± 1.9 SD); *Aristida adscensionis* (mean = 1.1 individuals ± 1.9 SD); and *Perotis patens* (mean = 0.3 individuals ± 0.6 SD) (Table 5 and Figure 12).

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the APCQ method yielded the following results (Table 5): there were no significant differences in the mean number of *Aristida adscensionis*, *Brachiaria deflexa* and *Perotis patens*; and there were significantly more *Heteropogon contortus* than the mean number of individuals recorded per site by the MIM method.

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the BC method yielded the following results (Table 5): there were no significant differences in the mean number of *Perotis patens*, *Aristida adscensionis* and *Heteropogon contortus*; and there were significantly more *Brachiaria deflexa* recorded per site using the MIM method.

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the LPI method yielded the following results (Table 5): there were no significant differences in the mean number of *Perotis patens*, *Aristida adscensionis, Brachiaria deflexa* and *Heteropogon contortus* than the mean number of individuals recorded per site using the MIM method.

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Five dominant grass species

The five most dominant grass species recorded per site using the MIM method in the MCR were *Panicum maximum* (mean = 24.4 individuals ± 14.4 SD), *Digitaria eriantha* (mean = 10.8 individuals ± 11.6 SD), *Urochloa mosambicensis* (mean = 9.2 individuals ± 7.6 SD), *Aristida adscensionis* (mean = 7.5 individuals ± 6.7 SD) and *Aristida congesta var. barbicollis* (mean = 5.8 individuals ± 6.7 SD) (Table 6 and Figure 13).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded using the APCQ method reveals that the latter method recorded significantly fewer *Panicum maximum* (mean = 19.7 individuals ± 13.2 SD, P = 0.03) (Table 6 and Figure 13). There were no significant differences in the mean number of *Digitaria eriantha* recorded by the APCQ method (mean = 13.9 individuals ± 11.8 SD, P = 0.09); there were no significant differences in the mean number of *Urochloa mosambicensis* (mean = 9.1 individuals ± 8.7 SD, P = 0.8); there were no significant differences in the mean number of *Aristida adscensionis* (mean = 6.6 individuals ± 4.9 SD, P = 0.6); and there were no significant differences in the mean number of *Aristida congesta var. barbicollis* (mean = 7.3 individuals ± 5.8 SD, P = 0.05) (Table 6 and Figure 13).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded using the BC method reveals that there were no significant differences in the mean number of *Panicum maximum* recorded by the BC method (mean = 26.3 individuals ± 14.7 SD, P = 0.1) (Table 6 and Figure 13). There were no significant differences in the mean number of *Digitaria eriantha* (mean = 15.4 individuals ± 16.1 SD, P = 0.5); there were no significant differences in the mean number of *Urochloa mosambicensis* (mean = 10.5 individuals ± 8.2 SD, P = 0.1); there were no significant differences in the mean number of *Aristida adscensionis* (mean = 7.6 individuals ± 5.8 SD, P = 0.2); and there were no significant differences in the mean number of *Aristida congesta var. barbicollis* (mean = 6.9 individuals ± 6.3 SD, P = 0.1) (Table 6 and Figure 13).

The comparison of the mean number of individuals of the five dominant grass species recorded per site using the MIM method with those of the means recorded using the LPI method reveals that the latter method recorded significantly fewer *Panicum maximum* (mean = 11.3 individuals ± 8.1 SD, P < 0.001) (Table 6 and Figure 13). There were significantly fewer *Digitaria eriantha* recorded than by the MIM method (mean = 6.2 individuals ± 9.3 SD, P = 0.004); there were significantly fewer *Urochloa mosambicensis* (mean = 3.8 individuals ± 4.2 SD, P < 0.001); there were significantly fewer *Aristida adscensionis* (mean = 2.0 individuals ± 2.6 SD, P < 0.001); and there were significantly fewer *Aristida congesta var. barbicollis* (mean = 3.0 individuals ± 4.3 SD, P = 0.01) (Table 6 and Figure 13).

Table 6: The mean number of individuals of the nine dominant grass species recorded using the various survey methods in the MCR (n = 25 sites). The five dominant grass species recorded for each of the survey methods are represented by ^{D}; while significant differences in comparison to the MIM method at a 95% confidence interval (P < 0.05) are represented by * and significant differences in comparison to the MIM method at a 99% confidence interval (P < 0.01) are represented by **

	MIM		APCQ			BC			LPI		
	Mean	SD	Mean	SD	P-value	Mean	SD	P-value	Mean	SD	P-value
Panicum maximum	24.4^{D}	14.4	19.7^{D}	13.2	0.03*	26.3^{D}	14.7	0.1	11.3^{D}	8.1	<0.001**
Digitaria eriantha	10.8^{D}	11.6	13.9^{D}	11.8	0.09	15.4^{D}	16.1	0.5	6.2^{D}	9.3	0.004**
Urochloa mosambicensis	9.2^{D}	7.6	9.1^{D}	8.7	0.8	10.5^{D}	8.2	0.1	3.8^{D}	4.2	<0.001**
Aristida adscensionis	7.5^{D}	6.7	6.6	4.9	0.6	7.6^{D}	5.8	0.2	2.0	2.6	<0.001**
Aristida congesta var. barbicollis	5.8^{D}	6.7	7.3^{D}	5.8	0.05	6.9	6.3	0.1	3.0^{D}	4.3	0.01*
Perotis patens	4.1	5.2	8.4^{D}	8.5	<0.001**	4.2	6.5	0.5	2.1^{D}	2.5	0.007**
Brachiaria deflexa	3.5	4.1	5.5	5.8	0.05	7.0^{D}	9.1	0.008**	1.0	1.6	0.003**
Themeda triandra	0.6	2.3	0.7	2.3	0.9	1.0	3.3	0.4	0.5	1.9	0.6
Heteropogon contortus	0.3	0.7	1.0	2.1	0.02*	0.6	1.7	0.6	0.5	1.3	0.3

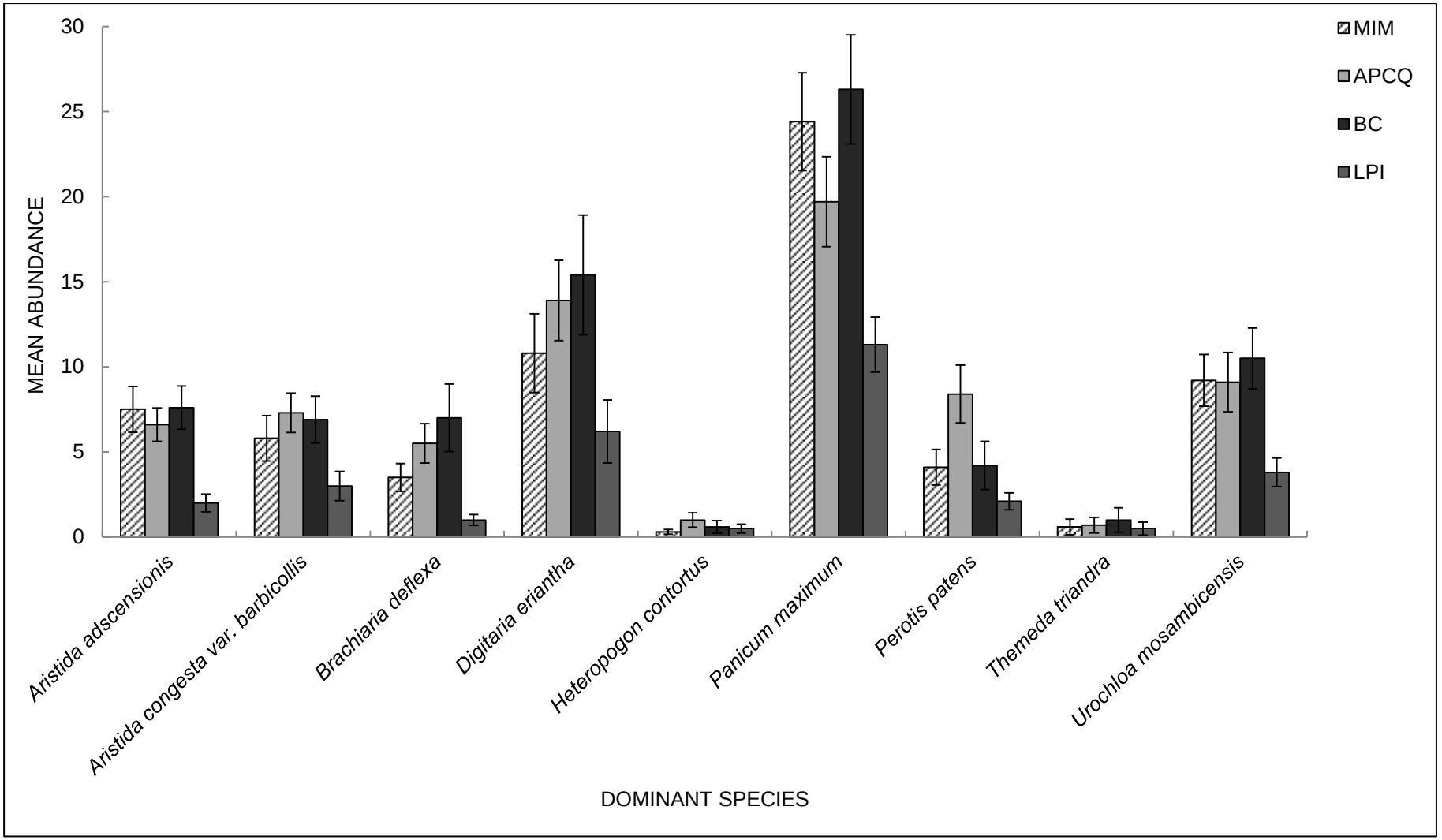

Figure 13: Mean abundance of dominant grass species recorded using each of the survey methods in the MCR (where error bars reflect Standard Error).

Other dominant grass species

The mean number of individuals of the remaining four "other dominant" grass species recorded using the MIM method in the MCR were *Perotis patens* (mean = 4.1 individuals ± 5.2 SD); *Brachiaria deflexa* (mean = 3.5 individuals ± 4.1 SD); *Themeda triandra* (mean = 0.6 individuals ± 2.3 SD); and *Heteropogon contortus* (mean = 0.3 individuals ± 0.7 SD) (Table 6 and Figure 13).

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded using the MIM method with the mean number of individuals recorded using the APCQ method yielded the following results (Table 6): there were no significant differences in the mean number of *Brachiaria deflexa* and *Themeda triandra*; and there were significantly more *Heteropogon contortus* and *Perotis patens* than the mean number of individuals recorded per site by the MIM method.

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the BC method yielded the following results (Table 6): there were no significant differences in the mean number of *Heteropogon contortus*, *Perotis patens* and *Themeda triandra*; and there were significantly more *Brachiaria deflexa* than the mean number of individuals recorded per site by the MIM method.

The comparison of the mean number of individuals of the remaining four "other dominant" grass species recorded per site by the MIM method with the mean number of individuals recorded using the LPI method yielded the following results (Table 6): there were no significant differences in the mean number of *Themeda triandra* and *Heteropogon contortus*; and there were significantly fewer *Brachiaria deflexa* and *Perotis patens* than the mean number of individuals recorded per site by the MIM method.

3.1.3 Estimates of basal cover

Basal strikes

Combined land uses

The mean number of grass basal strikes recorded per site by the MIM method was 2.8 (± 2.0 SD) (Figure 14). There were no significant differences in the mean number of grass basal strikes recorded per site by the APCQ method (mean = 3.0 ± 2.5 SD, P = 0.7) and by the LPI method (mean = 3.4 ± 2.6 SD, P = 0.1) (Figure 14).

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The mean number of grass basal strikes recorded per site by the MIM method was 3.0 (± 2.1 SD) (Figure 14). There were significantly more grass basal strikes recorded per site by the APCQ method (mean = 4.4 ± 2.3 SD, P = 0.01) (Figure 14). There were no significant

differences in the mean number of grass basal strikes recorded per site by the LPI method than by the MIM method (mean = 3.4 ± 2.8 SD, P = 0.4) (Figure 14).

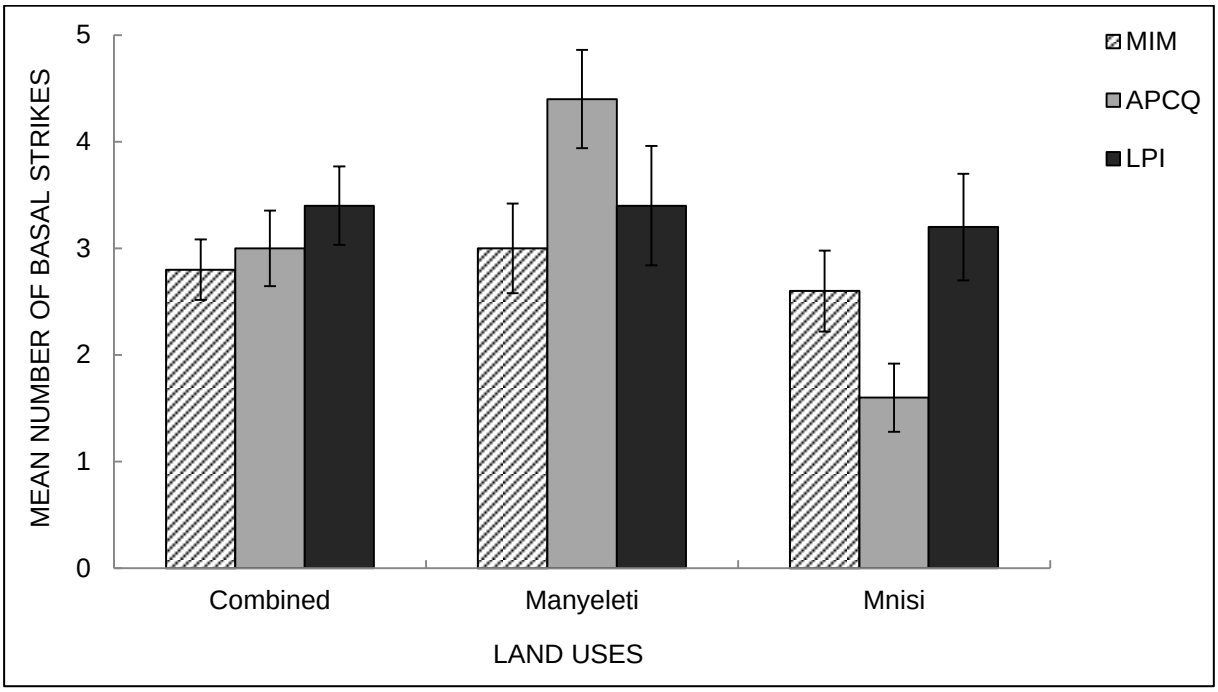

Figure 14: Mean number of grass basal strikes recorded using the MIM, APCQ and LPI methods across each of the land uses (where error bars reflect Standard Error).

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The mean number of grass basal strikes recorded per site by the MIM method was 2.6 (± 1.9 SD) (Figure 14). There were no significant differences in the mean number of grass basal strikes recorded per site by the APCQ method (mean = 1.6 ± 1.6 SD, P = 0.05) (Figure 14). There were no significant differences in the mean number of grass basal strikes recorded per site by the LPI method than by the MIM method (mean = 3.2 ± 2.5 SD, P = 0.2) (Figure 14).

Distance-to-herbaceous tuft

Combined land uses

The mean distance-to-herbaceous tuft recorded per site using the MIM method was 53.9 mm (± 16.7 SD) (Figure 15). There were no significant differences in the mean distance-to-herbaceous tuft recorded using the APCQ method than that recorded by the MIM method (mean = 57.9 mm ± 18.0 SD, P = 0.1) and there were no significant differences in the mean distance-to-herbaceous tuft recorded using the BC method (mean = 54.0 mm ± 16.2 SD, P = 0.7) (Figure 15).

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The mean distance-to-herbaceous tuft recorded per site using the MIM method was 51.0 mm (± 18.6 SD) (Figure 15). There were no significant differences in the mean distance-to-herbaceous tuft recorded using the APCQ method than that recorded by the MIM method

(mean = 54.8 mm ± 17.7 SD, P = 0.5) (Figure 15) and there were no significant differences in the mean distance-to-tuft recorded by the APCQ method (mean = 52.2 mm ± 15.4 SD, P = 0.4) (Figure 15).

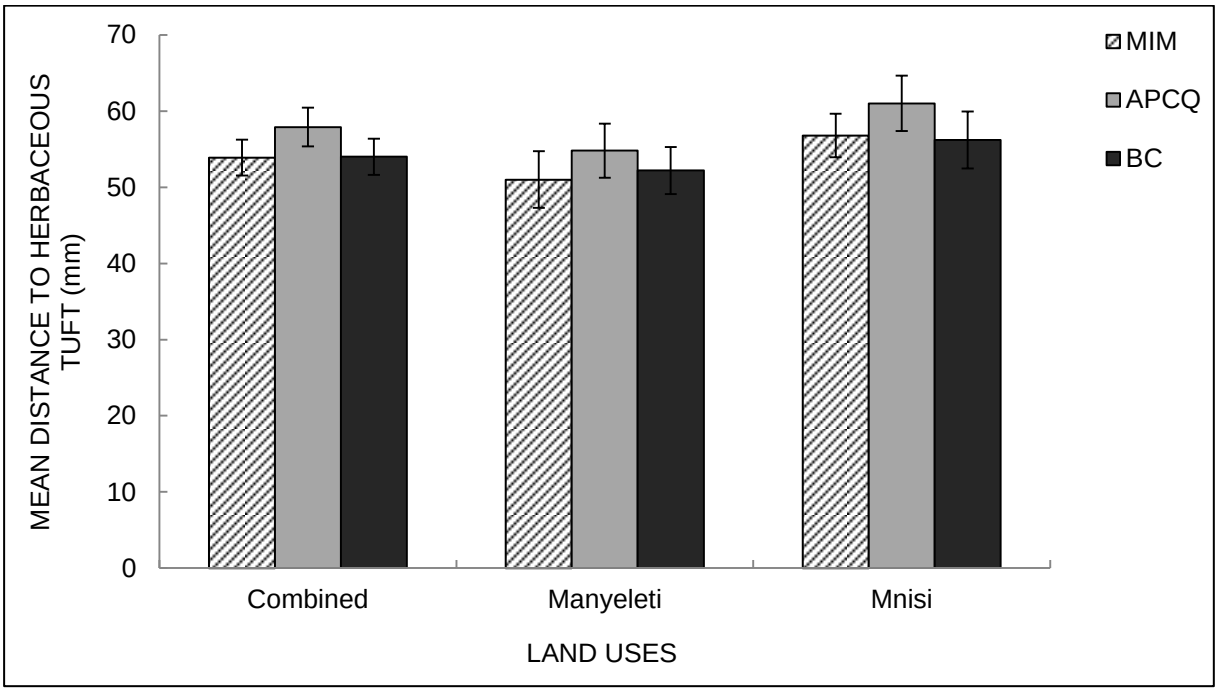

Figure 15: Mean distance-to-herbaceous tuft recorded using the MIM, APCQ and BC methods across each of the land uses (where error bars reflect Standard Error).

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The mean distance-to-herbaceous tuft recorded per site by the MIM method was 56.8 mm (± 14.3 SD) (Figure 15). There were no significant differences in the mean distance-to-herbaceous tuft recorded by the APCQ method (mean = 61.0 mm ± 18.2 SD, P = 0.1); and there were no significant differences in the mean distance-to-herbaceous tuft recorded by the BC method (mean = 56.2 mm ± 17.1 SD, P = 0.7) (Figure 15).

Herbaceous tuft diameter

Combined land uses

The mean tuft diameter recorded per site using the MIM method was 19.7 mm (± 5.1 SD) (Figure 16). The BC method recorded a significantly greater mean tuft diameter than the MIM method (mean = 22.7 mm ± 4.3 SD, P < 0.001) (Figure 16).

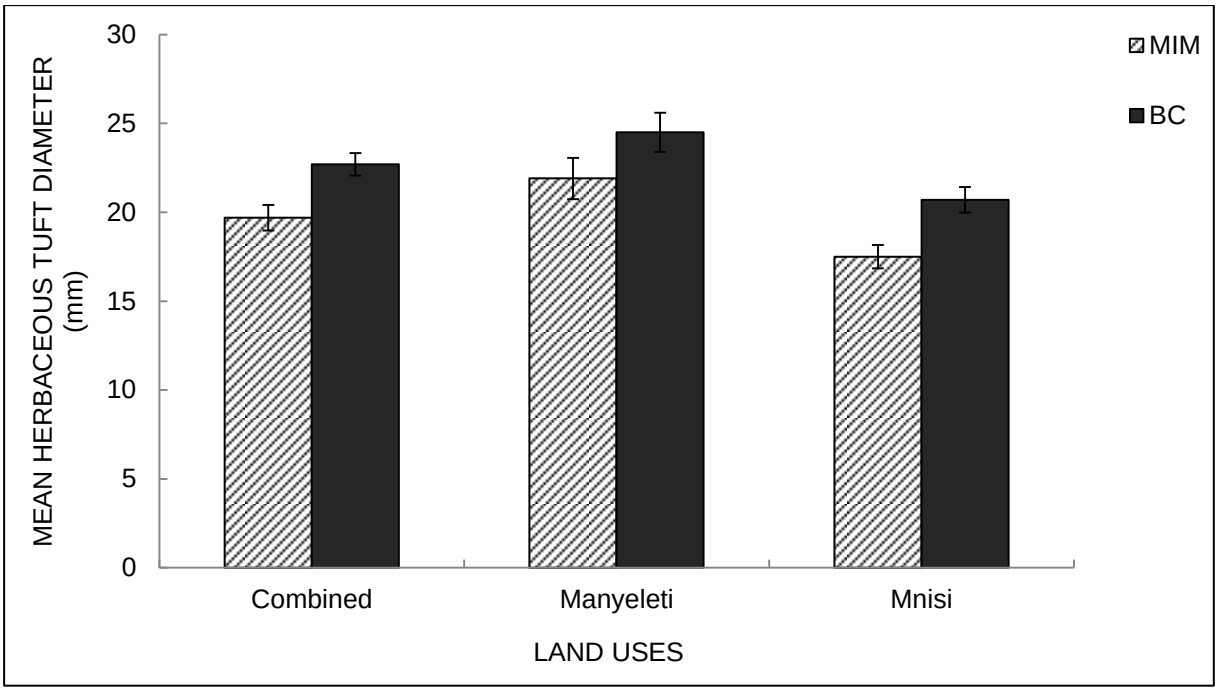

Figure 16: Mean tuft diameter recorded using the MIM and BC methods across each of the land uses (where error bars reflect Standard Error).

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The mean tuft diameter recorded per site using the MIM method was 21.9 mm (± 5.8 SD) (Figure 16). The BC method recorded a significantly greater mean tuft diameter than the MIM method (mean = 24.5 mm ± 5.5 SD, $P < 0.001$) (Figure 16).

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The mean tuft diameter recorded per site using the MIM method was 17.5 mm (± 3.3 SD) (Figure 16). The BC method recorded a significantly greater mean tuft diameter than the MIM method (mean = 20.7 mm ± 3.3 SD, $P = 0.001$) (Figure 16).

3.1.4 Grass biomass

The mean grass biomass estimated at sites across both land uses using 33 recordings (n = 47 sites) was 816.7 $kgha^{-1}$ (± 897.3 SD) (Figure 17). There were no significant differences in the mean grass biomass estimated using the DPM method that measures 100 recordings (n = 47 sites) than that estimated using 33 recordings with the MIM method (mean = 933.3 $kgha^{-1}$ ± 990.7 SD, $P = 0.1$) (Figure 17).

The mean grass biomass estimated at sites in the MGR using 33 recordings (n = 25 sites) was 1295.4 $kgha^{-1}$ (± 841.9 SD) (Figure 17). The DPM method using 100 recordings (n = 25 sites) estimated a significantly greater mean grass biomass than that estimated using 33 recordings with the MIM method (mean = 1558.0 $kgha^{-1}$ ± 845.7 SD, $P = 0.01$) (Figure 17).

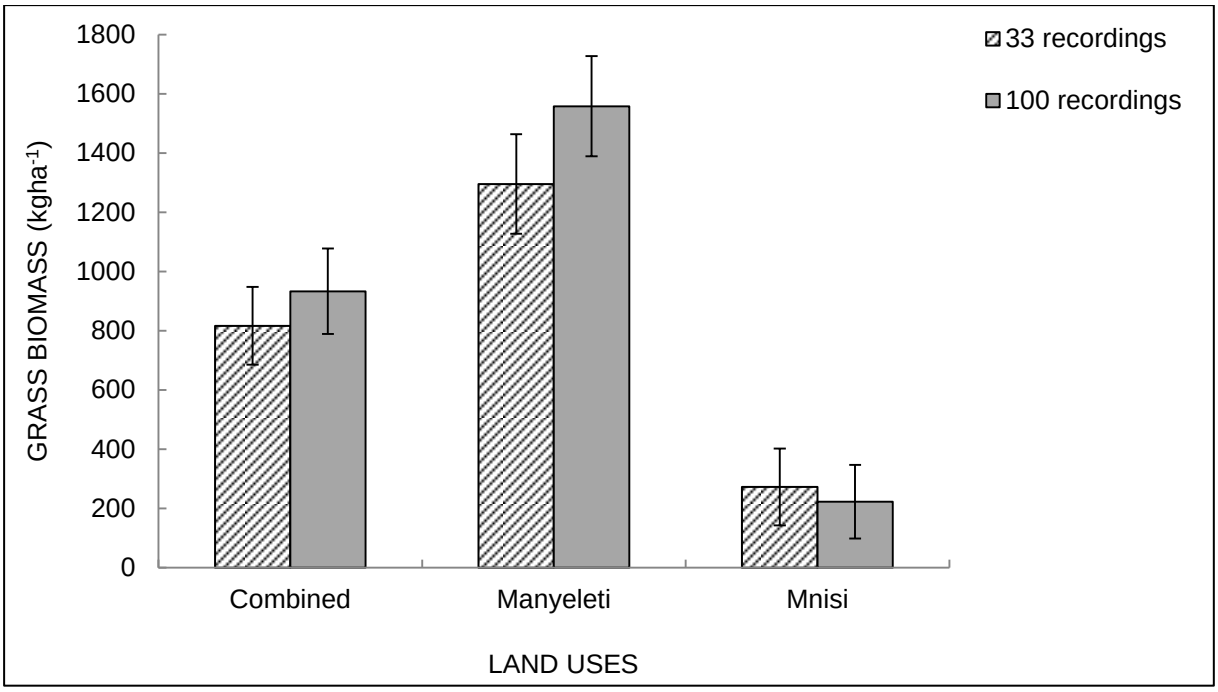

Figure 17: Mean grass biomass estimated using 33 recordings (MIM method) and 100 recordings across each of the land uses (where error bars reflect Standard Error).

The mean grass biomass estimated at sites in the MCR using 33 recordings (n = 22 sites) was 272.7 $kgha^{-1}$ (± 609 SD) (Figure 17). There were no significant differences in the mean grass biomass that was estimated using the DPM method that measures 100 recordings (n = 22 sites) than that estimated using 33 recordings with the MIM method (mean = 223.4 $kgha^{-1}$ ± 582.9 SD, P = 0.2) (Figure 17).

3.2 Woody component

3.2.1 Species detection

The total number of woody species recorded using the MIM, APCQ and LPI methods across both land uses was 71 (Table 7). The total number of woody species recorded using the various survey methods in the MGR was 57, while the total number of woody species recorded in the MCR was 63 (Table 7).

Table 7: Total number of woody species recorded using the various survey methods across both land uses and for the MGR and MCR separately

	MIM	APCQ	LPI	Total
Number of species recorded across both land uses	66	61	48	71
Number of species recorded in the MGR	49	49	28	57
Number of species recorded in the MCR	55	56	46	63

Combined land uses

The MIM method recorded 93% of the total number of woody species that were recorded using all methods across both land uses combined (n = 50 sites) (Figure 18). The APCQ method recorded 86% (n = 50 sites) and the LPI method recorded 68% (n = 50 sites) (Figure 18).

The mean number of woody species recorded per site using the MIM method across land uses was 14.3 species (± 4.0 SD) (Figure 19). The APCQ method recorded significantly more woody species than the MIM method (mean = 16.4 species ± 3.9 SD, P = 0.001) and the LPI method recorded significantly fewer woody species (mean = 5.9 species ± 4.1 SD, P < 0.001) (Figure 19).

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The MIM and APCQ methods both recorded 86% of the total number of woody species that were recorded using all methods in the MGR (n = 25 sites) and the LPI method recorded 49% (n = 25 sites) (Figure 18).

The mean number of woody species recorded per site using the MIM method in the MGR was 14.7 species (± 4.2 SD) (Figure 19). There were no significant differences in the mean number of woody species recorded using the APCQ method (mean = 16.4 species ± 4.7 SD, P = 0.08) and the LPI method recorded significantly fewer woody species (mean = 3.4 species ± 2.7 SD, P < 0.001) (Figure 19).

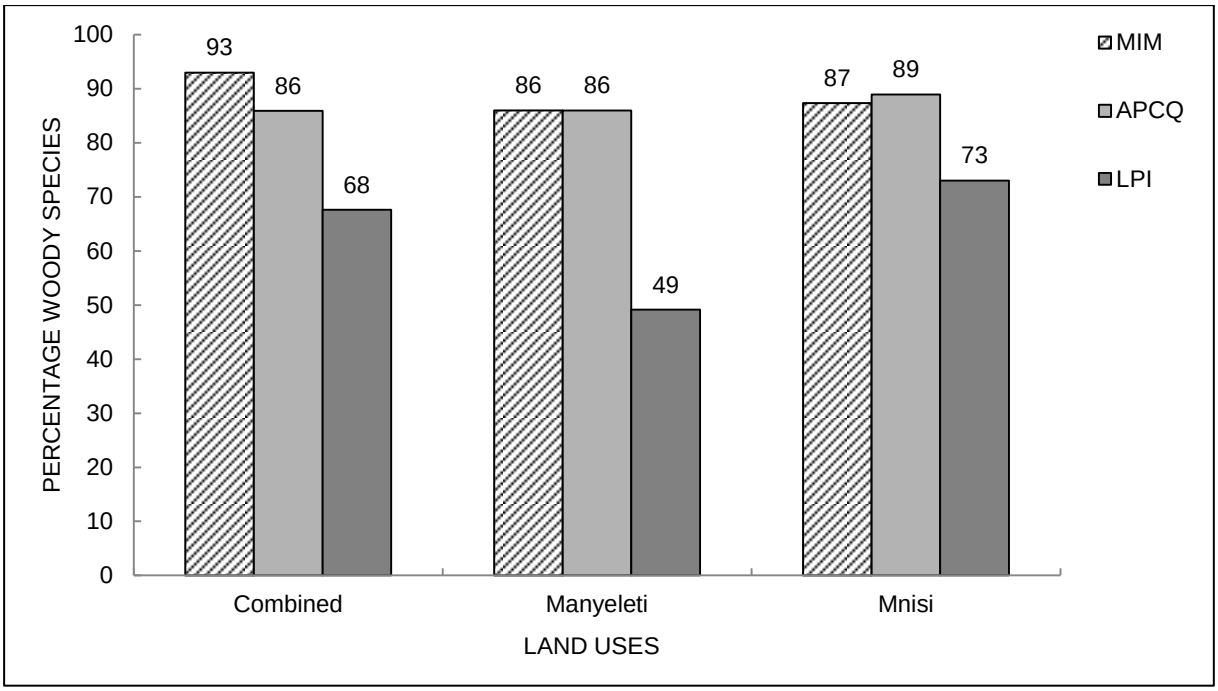

Figure 18: The percentage of woody species recorded using each survey method relative to the total number of woody species recorded using all methods across each of the land uses.

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The MIM method recorded 87% of the total number of woody species that were recorded using all methods in the MCR (n = 25 sites) (Figure 18). The APCQ method recorded 89% (n = 25 sites) and the LPI method recorded 73% (n = 25 sites) (Figure 18).

The mean number of woody species recorded per site using the MIM method in the MCR was 13.8 species (±3.9 SD) (Figure 19). The APCQ method recorded significantly more woody species than the MIM method (mean = 16.4 species ± 3.1 SD, P = 0.006) and the LPI method recorded significantly fewer woody species (mean = 8.3 species ± 3.8 SD, P < 0.001) (Figure 19).

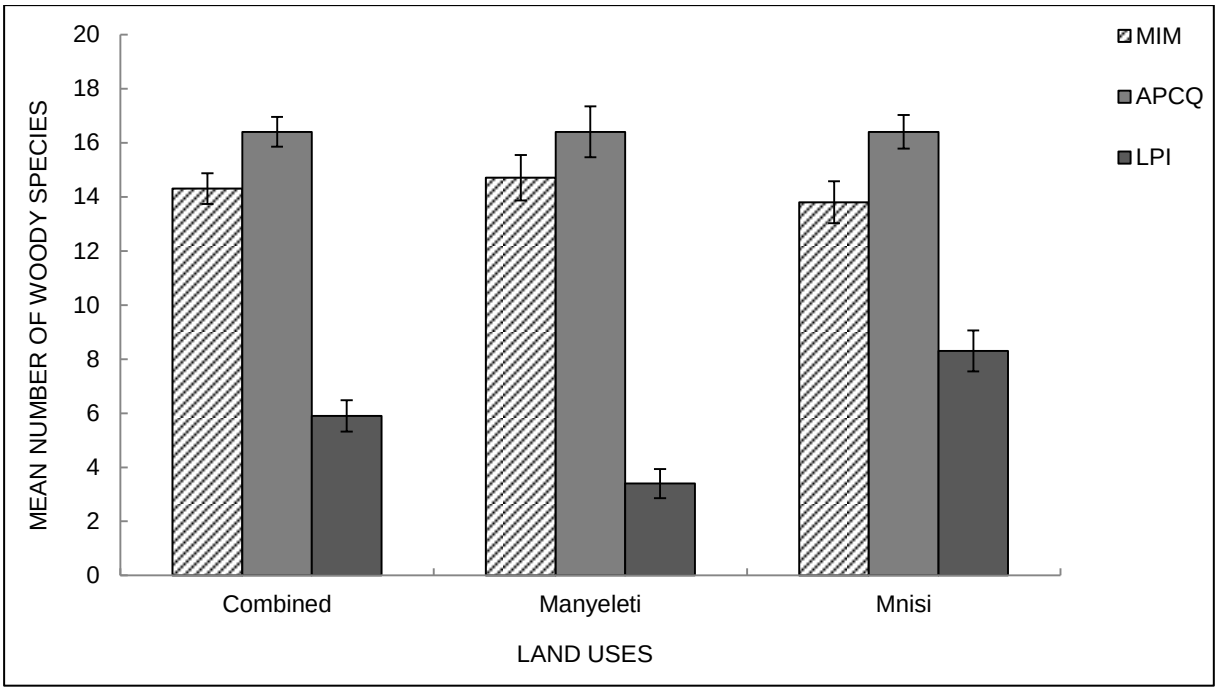

Figure 19: Mean number of woody species recorded using each of the survey methods across the land uses (where error bars reflect Standard Error).

3.2.2 Woody species composition, abundance and dominance

Combined land uses

Five dominant woody species

The five dominant woody species recorded per site using the MIM method across both land uses combined were *Strychnos madagascariensis* (mean = 17.9 individuals ± 57.5 SD), *Ehretia amoena* (mean = 16.9 individuals ± 81.1 SD), *Combretum apiculatum* (mean = 13.1 individuals ± 18.6 SD), *Acacia exuvialis* (mean = 9.5 individuals ± 13.0 SD) and *Ormocarpum trichocarpum* (mean = 8.4 individuals ± 18.3 SD) (Table 8 and Figure 20).

Table 8: The mean number of individuals of the 14 dominant woody species recorded using the various survey methods across both land uses (n = 50 sites). The five dominant woody species recorded for each of the survey methods are represented by [D]; while significant differences in comparison to the MIM method at a 95% confidence interval (P < 0.05) are represented by [*] and significant differences in comparison to the MIM method at a 99% confidence interval (P < 0.01) are represented by [**]

	MIM method		APCQ method			LPI method		
	Mean	SD	Mean	SD	P-value	Mean	SD	P-value
Strychnos madagascariensis	17.9[D]	57.5	3.3	7.7	0.04[*]	1.0[D]	2.9	0.01[*]
Ehretia amoena	16.9[D]	81.1	1.1	2.3	0.02[*]	0.1	0.3	<0.001[**]
Combretum apiculatum	13.1[D]	18.6	14.6[D]	15.8	0.2	2.8[D]	4.8	<0.001[**]
Acacia exuvialis	9.5[D]	13.0	4.9[D]	5.7	0.008[**]	0.5	1.2	<0.001[**]
Ormocarpum trichocarpum	8.4[D]	18.3	2.7	4.9	0.004[**]	0.1	0.4	<0.001[**]
Albizia harveyi	7.7	14.0	4.4	4.9	0.006[**]	0.5	1.1	<0.001[**]
Dichrostachys cinerea	7.6	13.9	5.0[D]	7.3	0.3	1.2[D]	2.8	<0.001[**]
Acacia nigrescens	6.5	17.3	3.7	6.0	0.80	0.4	0.9	<0.001[**]
Euclea divinorum	3.1	7.1	2.9	7.1	0.004[**]	0.9[D]	2.4	0.9
Combretum hereroense	2.6	5.8	6.5[D]	10.0	<0.001[**]	0.3	0.7	<0.001[**]
Ziziphus mucronata	1.9	3.0	3.2	4.8	0.02[*]	0.4	0.9	<0.001[**]
Combretum zeyheri	1.8	3.0	3.1	4.5	0.01[*]	1.0[D]	2.0	0.03[*]
Terminalia sericea	1.7	4.1	5.2[D]	9.1	0.006[**]	0.6	1.5	0.002[**]
Sclerocarya birrea	0.9	1.4	4.0	4.6	<0.001[**]	0.3	0.5	0.005[**]

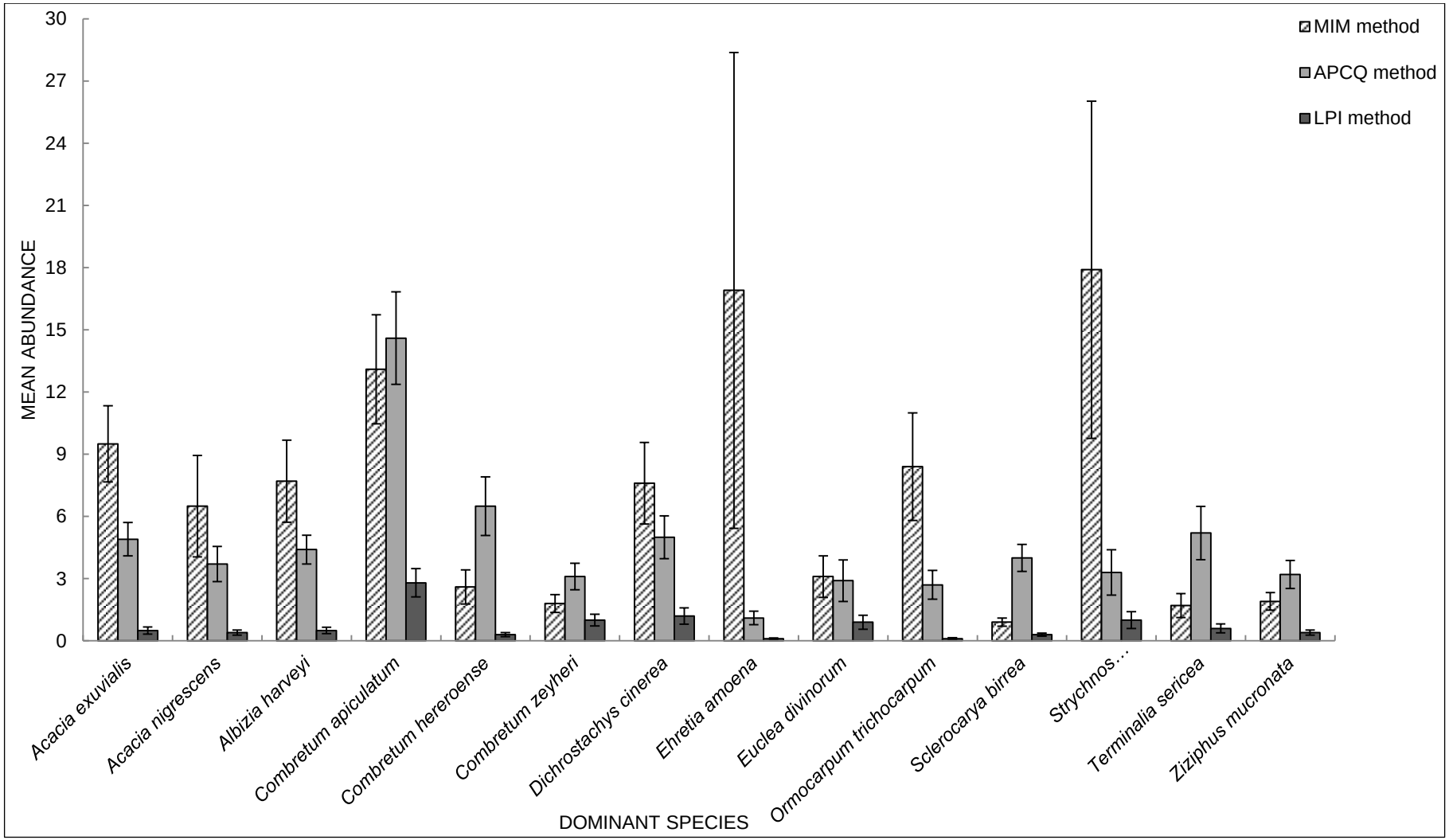

Figure 20: Mean abundance of dominant woody species recorded using each of the survey methods across both land uses (where error bars reflect Standard Error).

The comparison of the mean number of individuals of the five dominant woody species recorded per site using the MIM method with those of the means recorded using the APCQ method reveals that the latter method recorded significantly fewer *Strychnos madagascariensis* (mean = 3.3 individuals ± 7.7 SD, P = 0.04) (Table 8 and Figure 20). There were significantly fewer *Ehretia amoena* recorded than by the MIM method (mean = 1.1 individuals ± 2.3 SD, P = 0.02); there were no significant differences in the mean number of *Combretum apiculatum* (mean = 14.6 individuals ± 15.8 SD, P = 0.2); there were significantly fewer *Acacia exuvialis* (mean = 4.9 individuals ± 5.7 SD, P = 0.008); and there were significantly fewer *Ormocarpum trichocarpum* than the mean number recorded per site using the MIM method (mean = 2.7 individuals ± 4.9 SD, P = 0.004) (Table 8 and Figure 20).

The comparison of the mean number of individuals of the five dominant woody species recorded per site using the MIM method with those of the means recorded using the LPI method reveals that the latter method recorded significantly fewer *Strychnos madagascariensis* (mean = 1.0 individuals ± 2.9 SD, P = 0.01) (Table 8 and Figure 20). There were significantly fewer *Ehretia amoena* recorded than by the MIM method (mean = 0.1 individuals ± 0.3 SD, P < 0.001); there were significantly fewer *Combretum apiculatum* (mean = 2.8 individuals ± 4.8 SD, P < 0.001); there were significantly fewer *Acacia exuvialis* (mean = 0.5 individuals ± 1.2 SD, P < 0.001); and there were significantly fewer *Ormocarpum trichocarpum* than the mean number recorded using the MIM method (mean = 0.1 individuals ± 0.4 SD, P < 0.001) (Table 8 and Figure 20).

Other dominant woody species

The mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method across both land uses were *Albizia harveyi* (mean = 7.7 individuals ± 14.0 SD); *Dichrostachys cinerea* (mean = 7.6 individuals ± 13.9 SD); *Acacia nigrescens* (mean = 6.5 individuals ± 17.3 SD); *Euclea divinorum* (mean = 3.1 individuals ± 7.1 SD); *Combretum hereroense* (mean = 2.6 individuals ± 5.8 SD); *Ziziphus mucronata* (mean = 1.9 individuals ± 3.0 SD); *Combretum zeyheri* (mean = 1.8 individuals ± 3.0 SD); *Terminalia sericea* (mean = 1.7 individuals ± 4.1 SD); and *Sclerocarya birrea* (mean = 0.9 individuals ± 1.4 SD) (Table 8 and Figure 20).

The comparison of the mean number of individuals of the remaining nine "other dominant" woody species recorded per site by the MIM method with the mean number of individuals recorded using the APCQ method yielded the following results (Table 8): there were no significant differences in the mean number of *Acacia nigrescens* and *Dichrostachys cinerea*; there were significantly fewer *Albizia harveyi* and *Euclea divinorum*; and there were significantly more *Combretum hereroense*, *Combretum zeyheri*, *Sclerocarya birrea*, *Terminalia sericea* and *Ziziphus mucronata* than the mean number of individuals recorded using the MIM method.

The comparison of the mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method with the mean number of individuals recorded using the LPI method yielded the following results (Table 8): there were no significant

differences in the mean number of *Euclea divinorum*; and there were significantly fewer *Acacia nigrescens*, *Albizia harveyi*, *Combretum hereroense*, *Combretum zeyheri*, *Dichrostachys cinerea*, *Sclerocarya birrea*, *Terminalia sericea* and *Ziziphus mucronata* than the mean number of individuals recorded using the MIM method.

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Five dominant woody species

The five most dominant woody species recorded per site using the MIM method in the MGR were *Ehretia amoena* (mean = 32.1 individuals ± 113.6 SD), *Ormocarpum trichocarpum* (mean = 12.8 individuals ± 24.3 SD), *Combretum apiculatum* (mean = 11.8 individuals ± 16.3 SD), *Albizia harveyi* (mean = 8.8 individuals ± 17.9 SD) and *Acacia exuvialis* (mean = 7.0 individuals ± 9.0 SD) (Table 9 and Figure 21).

The comparison of the mean number of individuals of the five dominant woody species recorded per site using the MIM method with those of the means recorded using the APCQ method reveals that there were no significant differences in the mean number of *Ehretia amoena* individuals recorded by the APCQ method (mean = 1.9 individuals ± 3.0 SD, P = 0.07) (Table 9 and Figure 21). There were significantly fewer *Ormocarpum trichocarpum* recorded than the MIM method (mean = 2.7 individuals ± 4.6 SD, P = 0.003); there were no significant differences in the mean number of *Combretum apiculatum* (mean = 13.5 individuals ± 15.1 SD, P = 0.3); there were no significant differences in the mean number of *Albizia harveyi* (mean = 4.5 individuals ± 4.6 SD, P = 0.2); and there were significantly fewer *Acacia exuvialis* (mean = 2.8 individuals ± 3.8 SD, P = 0.004) (Table 9 and Figure 21).

The comparison of the mean number of individuals of the five dominant woody species recorded per site using the MIM method with those of the means recorded using the LPI method reveals that the latter method recorded significantly fewer *Ehretia amoena* individuals than the MIM method (mean = 0.1 individuals ± 0.3 SD, P < 0.001) (Table 9 and Figure 21). There were significantly fewer *Ormocarpum trichocarpum* recorded than by the MIM method (mean = 0 individuals ± 0.2 SD, P < 0.001); there were significantly fewer *Combretum apiculatum* (mean = 1.1 individuals ± 1.7 SD, P = 0.002); there were significantly fewer *Albizia harveyi* (mean = 0.2 individuals ± 0.5 SD, P < 0.001); and there were significantly fewer *Acacia exuvialis* (mean = 0.0 individuals ± 0.2 SD, P < 0.001) (Table 9 and Figure 21).

Table 9: The mean number of individuals of the 14 dominant woody species recorded using the various survey methods in the MGR (n = 25 sites). The five dominant woody species recorded for each of the survey methods are represented by [D]; while significant differences in comparison to the MIM method at a 95% confidence interval (P < 0.05) are represented by [*] and significant differences in comparison to the MIM method at a 99% confidence interval (P < 0.01) are represented by [**]

	MIM method		APCQ method			LPI method		
	Mean	SD	Mean	SD	P-value	Mean	SD	P-value
Ehretia amoena	32.1[D]	113.6	1.9	3.0	0.07	0.1	0.3	<0.001[**]
Ormocarpum trichocarpum	12.8[D]	24.3	2.7	4.6	0.003[**]	0.0	0.2	<0.001[**]
Combretum apiculatum	11.8[D]	16.3	13.5[D]	15.1	0.3	1.1[D]	1.7	0.002[**]
Albizia harveyi	8.8[D]	17.9	4.5[D]	4.6	0.2	0.2	0.5	<0.001[**]
Acacia exuvialis	7.0[D]	9.0	2.8	3.8	0.004[**]	0.0	0.2	<0.001[**]
Acacia nigrescens	6.4	14.4	4.6[D]	6.9	0.8	0.3[D]	0.7	0.002[**]
Combretum hereroense	4.4	7.6	11.5[D]	12.1	0.003[**]	0.4[D]	0.9	0.002[**]
Euclea divinorum	4.4	9.3	4.4[D]	9.3	0.9	1.0[D]	2.4	0.02[*]
Dichrostachys cinerea	4.2	5.0	2.8	2.6	0.2	0.2	0.6	<0.001[**]
Ziziphus mucronata	2.6	3.5	4.2	6.0	0.1	0.4[D]	0.9	<0.001[**]
Sclerocarya birrea	1.0	1.4	1.8	2.5	0.1	0.1	0.3	0.008[**]
Combretum zeyheri	0.9	1.7	2.1	2.6	0.01[*]	0.1	0.4	0.02[*]
Terminalia sericea	0.7	1.2	3.2	8.4	0.2	0.1	0.6	0.02[*]
Strychnos madagascariensis	0.0	0.0	0.1	0.0	1	0.0	0.0	-

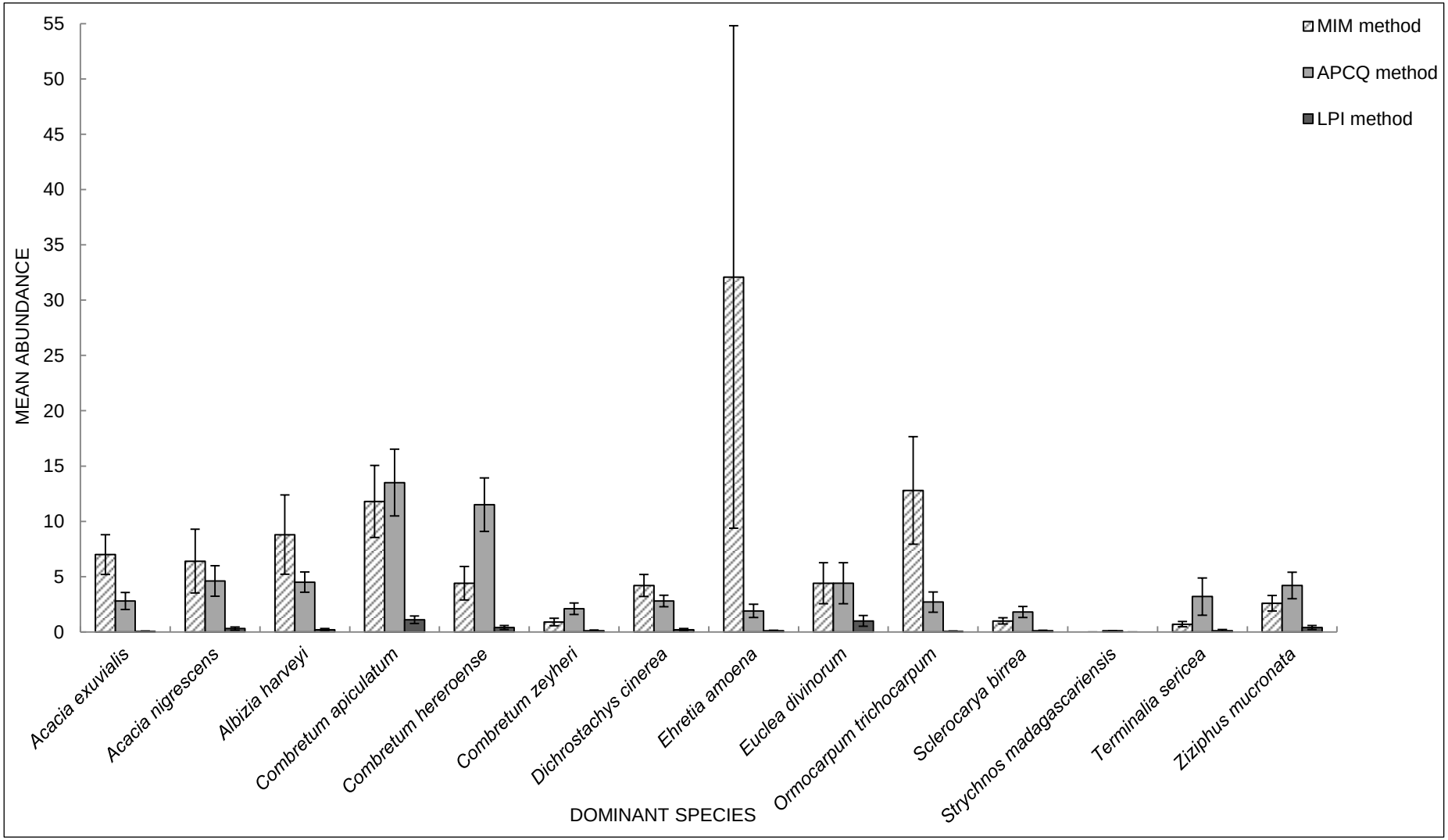

Figure 21: Mean abundance of dominant woody species recorded using each of the survey methods in the MGR (where error bars reflect Standard Error).

Other dominant woody species

The mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method in the MGR were *Acacia nigrescens* (mean = 6.4 individuals ± 14.4 SD); *Combretum hereroense* (mean = 4.4 individuals ± 7.6 SD); *Euclea divinorum* (mean = 4.4 individuals ± 9.3 SD); *Dichrostachys cinerea* (mean = 4.2 individuals ± 5.0 SD); *Ziziphus mucronata* (mean = 2.6 individuals ± 3.5 SD); *Sclerocarya birrea* (mean = 1.0 individuals ± 1.4 SD); *Combretum zeyheri* (mean = 0.9 individuals ± 1.7 SD); *Terminalia sericea* (mean = 0.7 individuals ± 1.2 SD); and *Strychnos madagascariensis* (mean = 0.0 individuals ± 0.0 SD) (Table 9 and Figure 21).

The comparison of the mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method with the mean number of individuals recorded using the APCQ method yielded the following results (Table 9): there were no significant differences in the mean number of *Euclea divinorum*, *Acacia nigrescens*, *Dichrostachys cinerea*, *Sclerocarya birrea*, *Strychnos madagascariensis*, *Terminalia sericea* and *Ziziphus mucronata*; and there were significantly more *Combretum hereroense* and *Combretum zeyheri* than the mean number of individuals recorded using the MIM method.

The comparison of the mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method with the mean number of individuals recorded using the LPI method yielded the following results (Table 9): there were no significant differences in the mean number of *Strychnos madagascariensis*; and there were significantly fewer *Acacia nigrescens*, *Combretum hereroense*, *Combretum zeyheri*, *Dichrostachys cinerea*, *Euclea divinorum*, *Sclerocarya birrea*, *Terminalia sericea* and *Ziziphus mucronata* than the mean number of individuals recorded per site using the MIM method.

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Five dominant woody species

The five most dominant woody species recorded per site using the MIM method in the MCR were *Strychnos madagascariensis* (mean = 35.8 individuals ± 78.1 SD), *Combretum apiculatum* (mean = 14.4 individuals ± 20.9 SD), *Acacia exuvialis* (mean = 12.0 individuals ± 15.7 SD), *Dichrostachys cinerea* (mean = 10.9 individuals ± 18.6 SD) and *Acacia nigrescens* (mean = 6.6 individuals ± 20.1 SD) (Table 10 and Figure 22).

The comparison of the mean number of individuals of the five dominant woody species recorded per site using the MIM method with those of the means recorded using the APCQ method reveals that the latter method recorded significantly fewer *Strychnos madagascariensis* individuals than the MIM method (mean = 6.6 individuals ± 1.0 SD, P = 0.03) (Table 10 and Figure 22). There were no significant differences in the mean number of *Combretum apiculatum* recorded than by the MIM method (mean = 15.6 individuals ± 16.8 SD, P = 0.5); there were no significant differences in the mean number of *Acacia exuvialis* (mean = 6.7 individuals ± 6.5 SD, P = 0.2); there were no

significant differences in the mean number of *Dichrostachys cinerea* (mean = 7.3 individuals ± 9.6 SD, P = 0.7); and there were no significant differences in the mean number of *Acacia nigrescens* (mean = 2.8 individuals ± 4.9 SD, P = 0.9) (Table 10 and Figure 22).

The comparison of the mean number of individuals of the five dominant woody species recorded per site using the MIM method with those of the means recorded using the LPI method reveals that the latter method recorded significantly fewer *Strychnos madagascariensis* individuals than the MIM method (mean = 2.1 individuals ± 3.8 SD, P = 0.01) (Table 10 and Figure 22). There were significantly fewer *Combretum apiculatum* recorded than by the MIM method (mean = 4.6 individuals ± 6.1 SD, P = 0.003); there were significantly fewer *Acacia exuvialis* (mean = 1.0 individuals ± 1.5 SD, P < 0.001); there were significantly fewer *Dichrostachys cinerea* (mean = 2.2 individuals ± 3.6 SD, P < 0.001); and there were significantly fewer *Acacia nigrescens* (mean = 0.4 individuals ± 1.1 SD, P = 0.04) (Table 10 and Figure 22).

Other dominant woody species

The mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method in the MCR were *Albizia harveyi* (mean = 6.5 individuals ± 9.0 SD); *Ormocarpum trichocarpum* (mean = 4.0 individuals ± 7.3 SD); *Combretum zeyheri* (mean = 2.7 individuals ± 3.8 SD); *Terminalia sericea* (mean = 2.7 individuals ± 5.5 SD); *Euclea divinorum* (mean = 1.9 individuals ± 3.9 SD); *Ehretia amoena* (mean = 1.8 individuals ± 4.5 SD); *Ziziphus mucronata* (mean = 1.2 individuals ± 2.3 SD); *Combretum hereroense* (mean = 0.9 individuals ± 2.3 SD); and *Sclerocarya birrea* (mean = 0.8 individuals ± 1.4 SD) (Table 10 and Figure 22).

The comparison of the mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method with the mean number of individuals recorded using the APCQ method yielded the following results (Table 10): there were no significant differences in the mean number of *Albizia harveyi*, *Ehretia amoena*, *Euclea divinorum*, *Ormocarpum trichocarpum*, *Combretum hereroense*, *Combretum zeyheri* and *Ziziphus mucronata*; and there were significantly more *Sclerocarya birrea* and *Terminalia sericea* than the means recorded using the MIM method.

The comparison of the mean number of individuals of the remaining nine "other dominant" woody species recorded per site using the MIM method with the mean number of individuals recorded using the LPI method yielded the following results (Table 10): there were no significant differences in the mean number of *Combretum hereroense*, *Combretum zeyheri*, *Euclea divinorum* and *Sclerocarya birrea*; and there were significantly fewer *Albizia harveyi*, *Ehretia amoena*, *Ormocarpum trichocarpum*, *Terminalia sericea* and *Ziziphus mucronata* than the means recorded using the MIM method.

Table 10: The mean number of individuals of the 14 dominant woody species recorded using the various survey methods in the MCR (n = 25 sites). The five dominant woody species recorded for each of the survey methods are represented by [D]; while significant differences in comparison to the MIM method at a 95% confidence interval (P < 0.05) are represented by [*] and significant differences in comparison to the MIM method at a 99% confidence interval (P < 0.01) are represented by [**]

	MIM method		APCQ method			LPI method		
	Mean	SD	Mean	SD	P-value	Mean	SD	P-value
Strychnos madagascariensis	35.8[D]	78.1	6.6[D]	1.0	0.03[*]	2.1[D]	3.8	0.01[*]
Combretum apiculatum	14.4[D]	20.9	15.6[D]	16.8	0.5	4.6[D]	6.1	0.003[**]
Acacia exuvialis	12.0[D]	15.7	6.7[D]	6.5	0.2	1.0	1.5	<0.001[**]
Dichrostachys cinerea	10.9[D]	18.6	7.3[D]	9.6	0.7	2.2[D]	3.6	<0.001[**]
Acacia nigrescens	6.6[D]	20.1	2.8	4.9	0.9	0.4	1.1	0.04[*]
Albizia harveyi	6.5	9.0	4.4	5.2	0.2	0.9	1.4	<0.001[**]
Ormocarpum trichocarpum	4.0	7.3	2.8	5.3	0.5	0.2	0.5	0.005[**]
Combretum zeyheri	2.7	3.8	4.0	5.7	0.2	1.9[D]	2.5	0.2
Terminalia sericea	2.7	5.5	7.2[D]	9.4	0.03[*]	1.2[D]	1.8	0.03[*]
Euclea divinorum	1.9	3.9	1.4	3.4	0.6	0.8	2.5	0.07
Ehretia amoena	1.8	4.5	0.4	0.8	0.2	0.1	0.3	0.01[*]
Ziziphus mucronata	1.2	2.3	2.2	3.0	0.08	0.4	1.0	0.04[*]
Combretum hereroense	0.9	2.3	1.4	1.0	0.2	0.3	0.5	0.1
Sclerocarya birrea	0.8	1.4	6.1	5.3	<0.001[**]	0.5	0.6	0.3

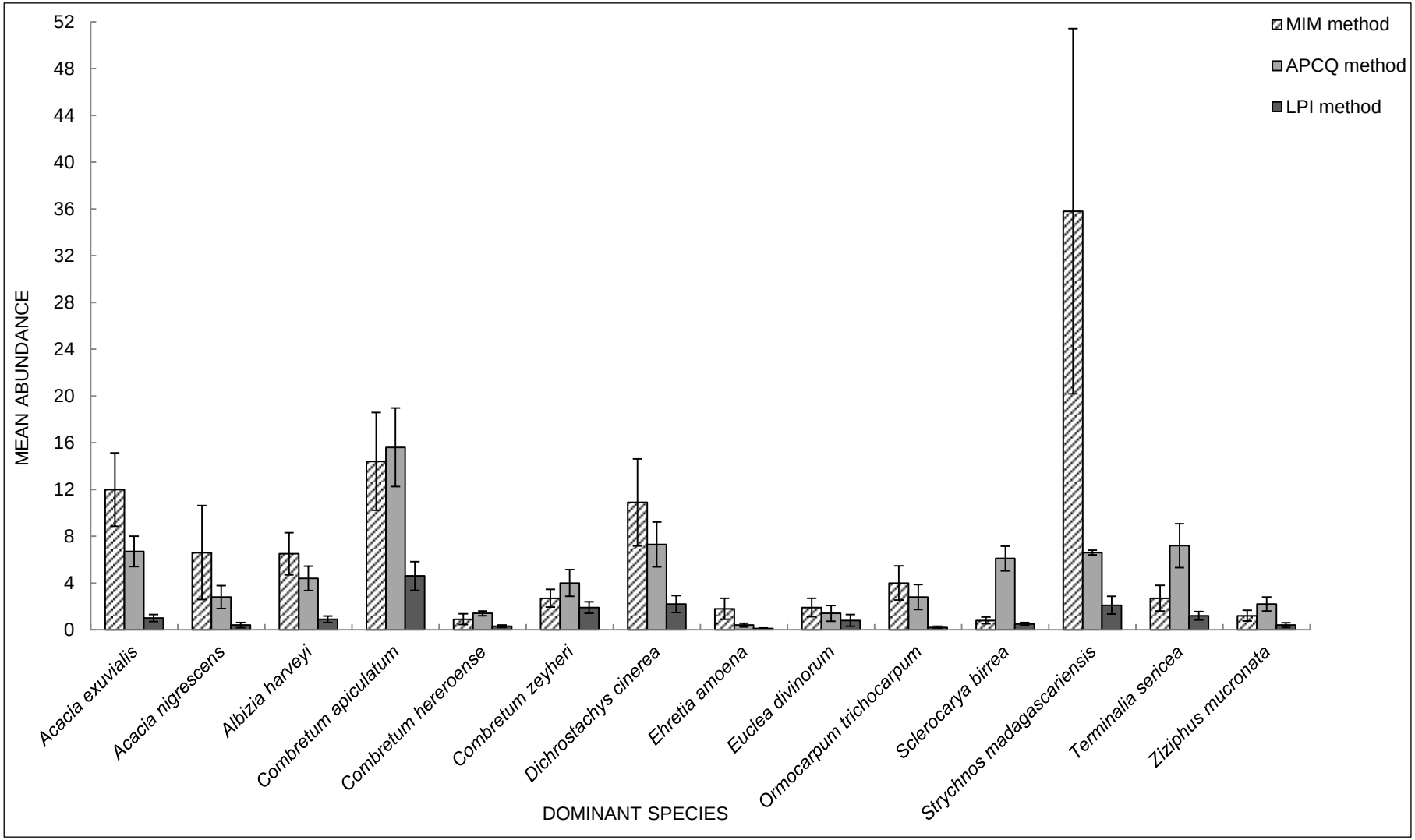

Figure 22: Mean abundance of dominant woody species recorded using each of the survey methods in the MCR (where error bars reflect Standard Error).

3.2.3 Index of tree height (structure)

Combined land uses

The mean number of woody individuals recorded using the MIM method in height class 1 was 84.8 individuals (± 92 SD); height class 2 was 25.3 individuals (± 21.3 SD), height class 3 was 15.4 individuals (± 13.3 SD) and height class 4 was 3.8 individuals per site (± 5.2 SD) (Figure 23).

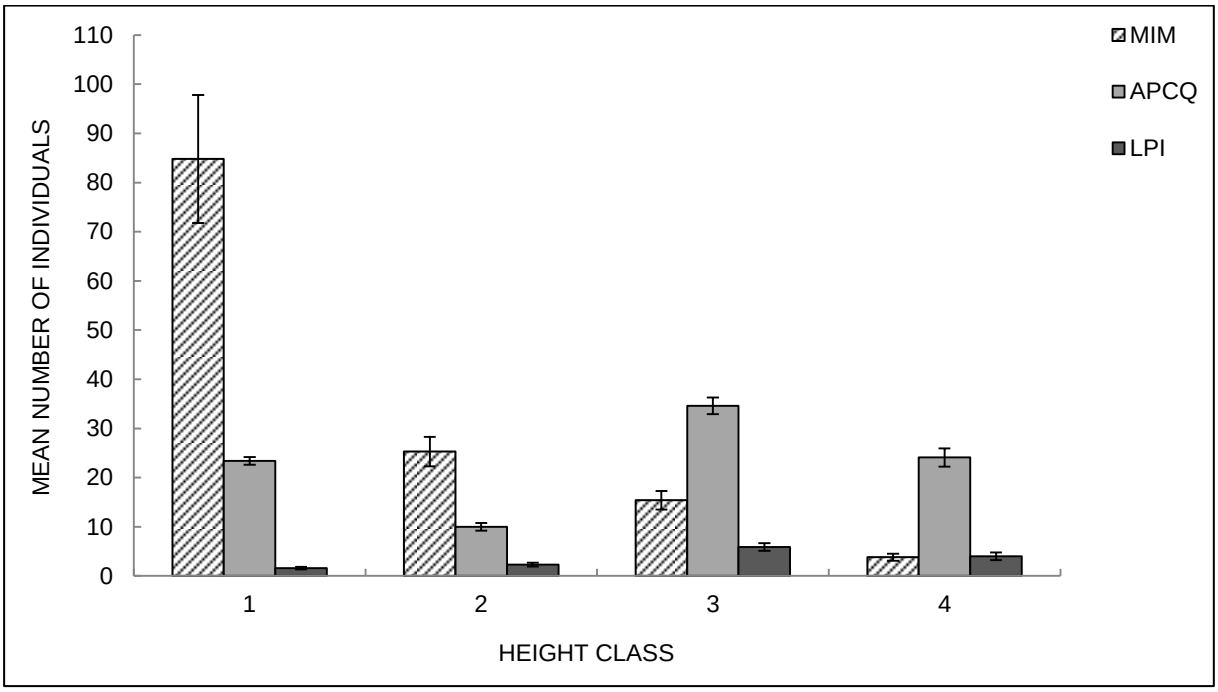

Figure 23: Mean number of individuals in each height class recorded using the various survey methods across both land uses (where error bars reflect Standard Error).

The APCQ method recorded a significantly lower mean number of woody individuals in height class 1 (mean = 23.4 individuals ± 5.7 SD, $P < 0.001$) (Figure 23). There were significantly fewer individuals recorded in height class 2 by the APCQ method (mean = 10.0 individuals ± 5.4 SD, $P < 0.001$); there were significantly more individuals recorded in height class 3 (mean = 34.6 individuals ± 11.9 SD, $P < 0.001$); and there were significantly more individuals recorded in height class 4 (mean = 24.1 individuals ± 13.2 SD, $P < 0.001$) (Figure 23).

The LPI method recorded significantly fewer woody individuals in height class 1 (mean = 1.6 individuals ± 1.9 SD, $P < 0.001$) (Figure 23). There were significantly fewer individuals recorded in height class 2 by the LPI method (mean = 2.3 individuals ± 2.6 SD, $P < 0.001$); there were significantly fewer individuals recorded in height class 3 (mean = 5.9 individuals ± 5.6 SD, $P < 0.001$); and there were no significant differences in the mean number of individuals recorded in height class 4 (mean = 4.0 individuals ± 5.5 SD, $P = 0.9$) (Figure 23).

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The mean number of woody individuals recorded in height class 1 by the MIM method was 100.4 individuals (± 121.6 SD), height class 2 was 17.7 individuals (± 15.2 SD), height class 3 was 7.0 individuals (± 6.8 SD) and height class 4 was 1.0 individuals per site (± 1.8 SD) (Figure 24).

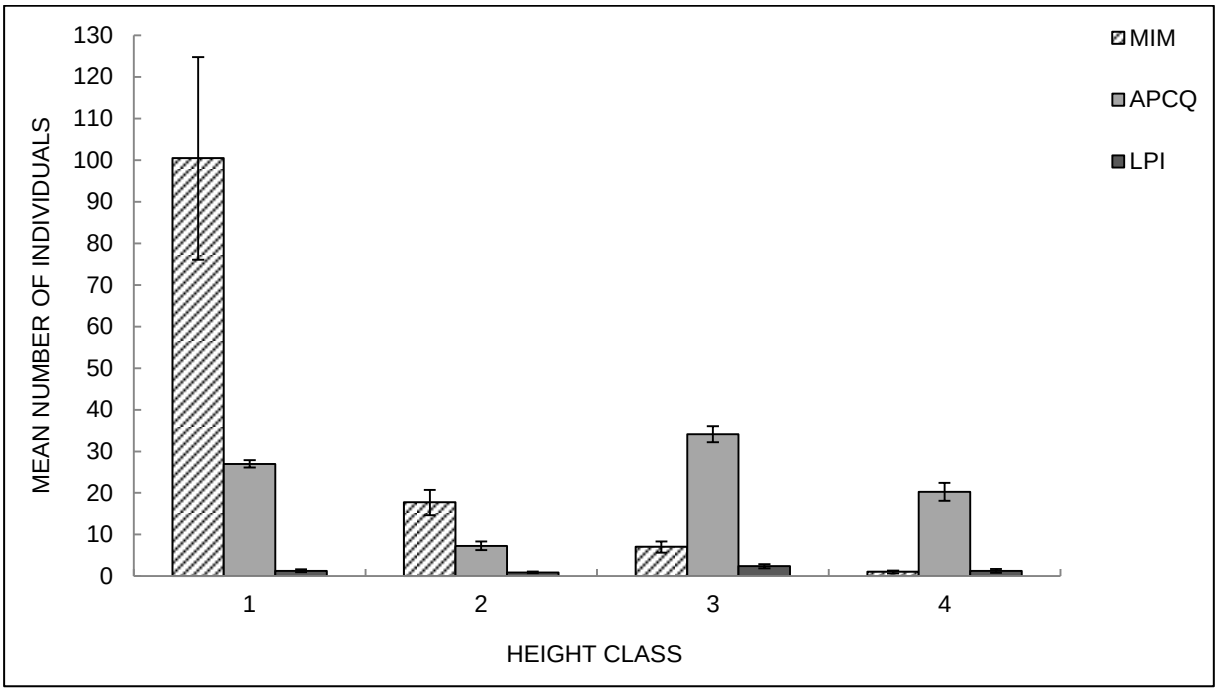

Figure 24: Mean number of individuals in each height class recorded using the various survey methods in the MGR (where error bars reflect Standard Error).

The APCQ method recorded a significantly lower mean number of woody individuals in height class 1 (mean = 27.0 individuals ± 4.3 SD, $P < 0.001$) (Figure 24). There were significantly fewer individuals recorded in height class 2 by the APCQ method (mean = 7.3 individuals ± 5.2 SD, $P = 0.001$); there were significantly more individuals recorded in height class 3 (mean = 34.1 individuals ± 9.6 SD, $P < 0.001$); and there were significantly more individuals recorded in height class 4 (mean = 20.3 individuals ± 10.8 SD, $P < 0.001$) (Figure 24).

The LPI method recorded significantly fewer woody individuals in height class 1 (mean = 1.3 individuals ± 1.7 SD, $P < 0.001$) (Figure 24). There were significantly fewer individuals recorded in height class 2 by the LPI method (mean = 0.9 individuals ± 1.0 SD, $P < 0.001$); there were significantly fewer individuals recorded in height class 3 (mean = 2.4 individuals ± 2.5 SD, $P = 0.001$); and there were no significant differences in the mean number of individuals recorded in height class 4 (mean = 1.3 individuals ± 2.3 SD, $P = 1.0$) (Figure 24).

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The mean number of woody individuals recorded in height class 1 using the MIM method was 69.2 (± 44.6 SD), height class 2 was 33.0 individuals (± 24.0 SD), height class 3 was 23.7 individuals (± 13.2 SD) and height class 4 was 6.6 individuals per site (± 6.0 SD) (Figure 25).

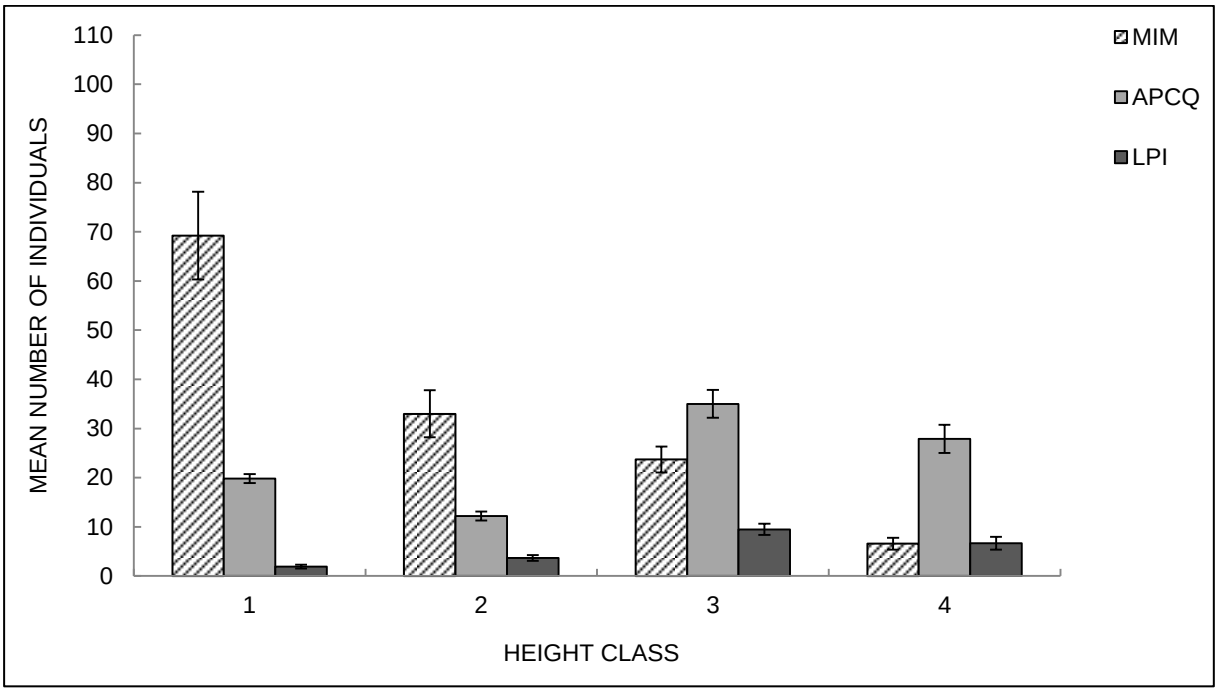

Figure 25: Mean number of individuals in each height class recorded using the various survey methods in the MCR (where error bars reflect Standard Error).

The APCQ method recorded a significantly lower mean number of woody individuals in height class 1 (mean = 19.8 individuals ± 4.6 SD, $P < 0.001$) (Figure 25). There were significantly fewer individuals recorded in height class 2 by the APCQ method (mean = 12.2 individuals ± 4.5 SD, $P < 0.001$); there were significantly more individuals recorded in height class 3 (mean = 35.0 individuals ± 14.1 SD, $P = 0.01$); and there were significantly more individuals recorded in height class 4 (mean = 27.9 individuals ± 14.4 SD, $P < 0.001$) (Figure 25).

The LPI method recorded significantly fewer woody individuals in height class 1 (mean = 1.9 individuals ± 2.0 SD, $P < 0.001$) (Figure 25). There were significantly fewer individuals recorded in height class 2 by the LPI method (mean = 3.7 individuals ± 2.9 SD, $P < 0.001$); there were significantly fewer individuals recorded in height class 3 (mean = 9.5 individuals ± 5.7 SD, $P < 0.001$); and there were no significant differences in the mean number of individuals recorded in height class 4 (mean = 6.7 individuals ± 6.5 SD, $P = 1.0$) (Figure 25).

3.2.4 Tree density

The mean tree density per site recorded using the MIM method across both land uses was 6465 plantsha^{-1} (± 4875 SD) (Figure 26). The mean tree density recorded per site using the

MIM method in the MGR was 6306 plantsha^{-1} (± 6128 SD), while the mean tree density recorded in the MCR was 6624 plantsha^{-1} (± 3303 SD) (Figure 26).

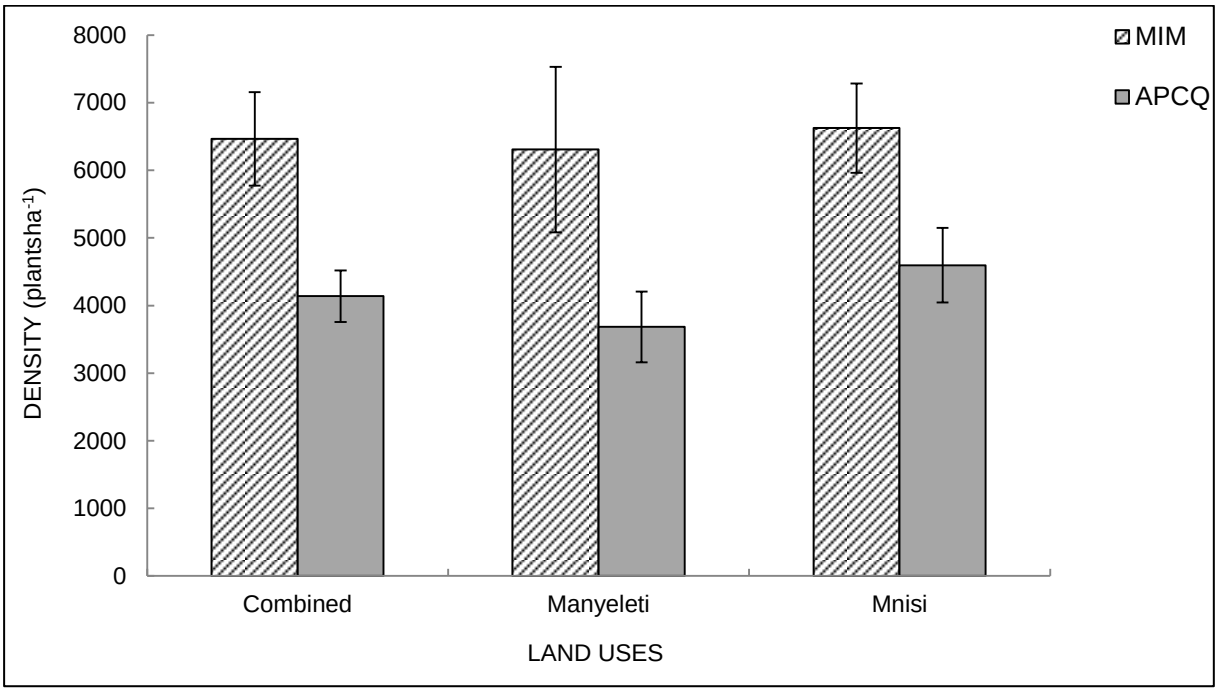

Figure 26: Mean tree density recorded using the MIM and APCQ methods across each of the land uses (where error bars reflect Standard Error).

The APCQ method estimated a significantly lower mean tree density than the MIM method across both land uses (mean = 4139 plantsha^{-1} ± 2700 SD, P < 0.001) (Figure 26). The APCQ method estimated a significantly lower mean tree density than the MIM method in the MGR (mean = 3684 plantsha^{-1} ± 2614 SD, P < 0.001); and the APCQ method estimated a significantly lower mean tree density than the MIM method in the MCR (mean = 4595 plantsha^{-1} ± 2760 SD, P < 0.001) (Figure 26).

3.2.5 Canopy cover

The mean percentage canopy cover per site recorded using the MIM method across both land uses was 41.5% (± 23.5 SD) (Figure 27). The mean percentage canopy cover recorded using the MIM method in the MGR was 25.7% (± 16.0 SD), while the mean percentage canopy cover recorded in the MCR was 57.2% (± 18.9 SD) (Figure 27).

The LPI method recorded a significantly lower mean percentage canopy cover than the MIM method across both land uses (mean = 13.8% ± 11.8 SD, P < 0.001) (Figure 27). The LPI method recorded a significantly lower mean percentage canopy cover than the MIM method in the MGR (mean = 5.8% ± 5.4 SD, P < 0.001), while in the MCR the LPI method recorded a significantly lower mean percentage canopy cover (mean = 21.8% ± 11.0 SD, P < 0.001) (Figure 27).

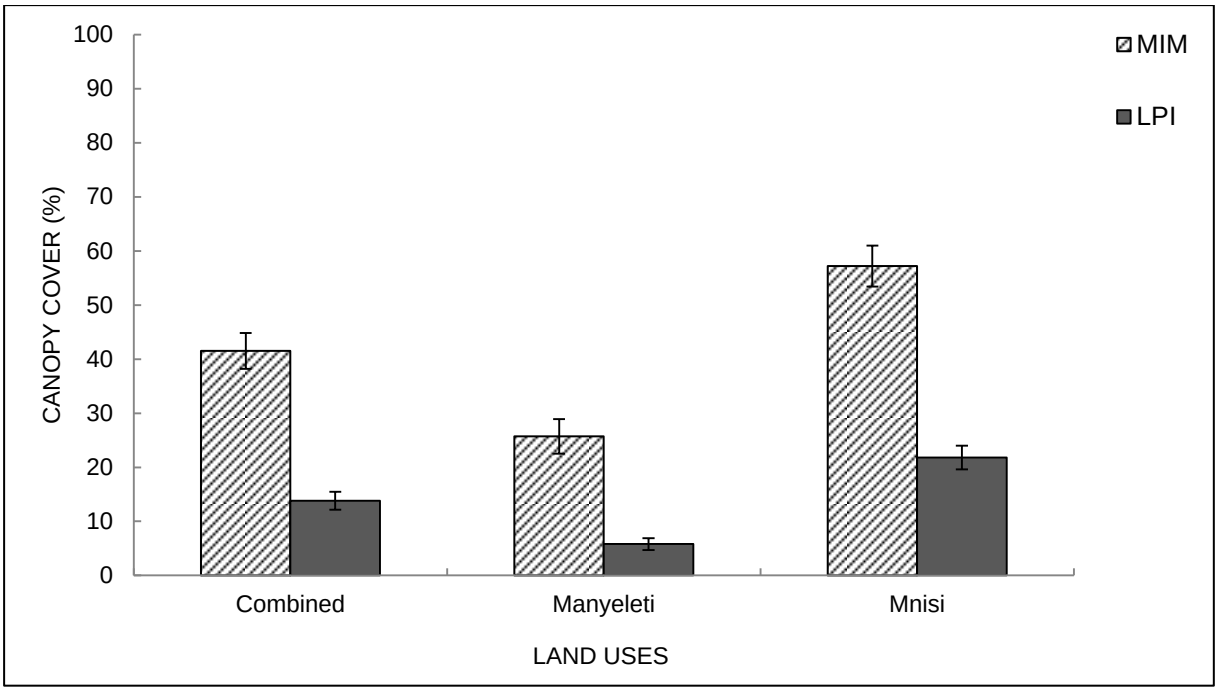

Figure 27: The mean percentage canopy cover estimated using the various survey methods across each of the land uses (where error bars reflect Standard Error).

3.3 Time for fieldwork and data processing

3.3.1 Fieldwork

Combined land uses

The mean time taken to undertake the fieldwork phase of the MIM method (n = 50 sites) across the combined land uses was 80.0 decimal minutes per site (± 19.4 SD) (Figure 28). The mean time taken to undertake the fieldwork phase of the APCQ method (n = 50 sites) was significantly longer than the MIM method (mean = 132.1 decimal minutes per site ± 29.2 SD, $P < 0.001$); the BC method (n = 46 sites) was significantly shorter (mean = 49.2 decimal minutes per site ± 10.7 SD, $P < 0.001$); and the time taken to undertake the fieldwork phase of the LPI method (n = 50 sites) was significantly shorter (mean = 32.9 decimal minutes per site ± 7.3 SD, $P < 0.001$) (Figure 28).

The mean time taken to do 33 recordings of the DPM method (n = 46 sites) was 6.1 decimal minutes per site (± 1.8 SD) (Figure 28). The mean time taken to do the DPM method using 100 recordings (n = 46 sites) was significantly longer than the former method (mean = 19.0 decimal minutes ± 5.3 SD, $P < 0.001$) (Figure 28).

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The mean time taken to undertake the fieldwork phase of the MIM method (n = 25 sites) in the MGR was 76.0 decimal minutes per site (± 21.7 SD) (Figure 28). The mean time taken to undertake the fieldwork phase of the APCQ method (n = 25 sites) was significantly longer than the MIM method (mean = 108.4 decimal minutes per site ± 15.9 SD, $P < 0.001$); the BC

method (n = 25 sites) was significantly shorter (mean = 45.5 decimal minutes per site ± 8.8 SD, P <0.001); and the time taken to undertake the fieldwork phase of the LPI method (n = 25 sites) was significantly shorter (mean = 32.2 decimal minutes per site ± 6.5 SD, P <0.001) (Figure 28).

The mean time taken to do 33 recordings of the DPM method (n = 24 sites) was 6.3 decimal minutes per site (± 1.6 SD) (Figure 28). The mean time taken to do the DPM method using 100 recordings (n = 24 sites) was significantly longer than the former method (mean = 19.0 decimal minutes ± 4.9 SD, P < 0.001) (Figure 28).

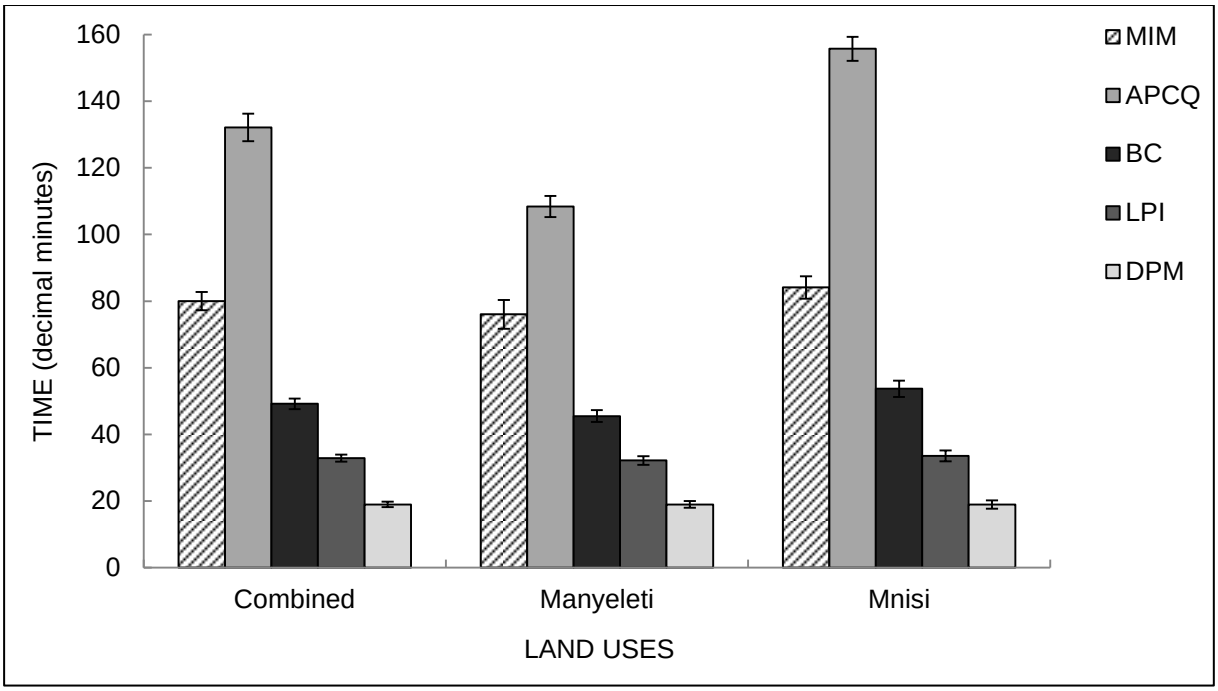

Figure 28: Mean number of decimal minutes for the fieldwork phase of each survey method across the land uses (where error bars reflect Standard Error).

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The mean time taken to do the fieldwork phase of the MIM method in the MCR (n = 25 sites) was 84.1 decimal minutes per site (± 16.7 SD) (Figure 28). The mean time taken to undertake the fieldwork phase of the APCQ method (n = 25 sites) was significantly longer than the MIM method (mean = 155.7 decimal minutes per site ± 18.0 SD, P < 0.001); the BC method (n = 21 sites) was significantly shorter (mean = 53.7 decimal minutes per site ± 11.2 SD, P <0.001); and the mean time taken to undertake the fieldwork phase of the LPI method (n = 25 sites) was significantly shorter (mean = 33.6 decimal minutes per site ± 8.2 SD, P <0.001) (Figure 28).

The mean time taken to do 33 recordings of the DPM method (n = 22 sites) was 5.8 decimal minutes per site (± 2.0 SD) (Figure 28). The mean time taken to do the DPM method using

100 recordings (n = 22 sites) was significantly longer than the former method (mean = 19.0 decimal minutes ± 5.9 SD, P < 0.001) (Figure 28).

3.3.2 Data processing

The mean time taken to enter the data for the MIM method (n = 20 sites) was 29.5 decimal minutes per site (± 6.9 SD) (Figure 29). The mean time taken to enter the data of the APCQ method (n = 20 sites) was significantly shorter than the MIM method (mean = 24.7 decimal minutes per site ± 2.1 SD, P = 0.006) (Figure 29). The mean time taken to enter the data for the BC method (n = 20 sites) was significantly shorter (mean = 10.4 decimal minutes per site ± 2.0 SD, P < 0.001); and the mean time taken to enter the data for the LPI method (n = 20 sites) was significantly shorter than the MIM method (mean = 9.8 decimal minutes per site ± 2.2 SD, P < 0.001) (Figure 29).

The mean time taken to enter the data for the 33 recordings of the DPM method (n = 20 sites) was 0.9 decimal minutes per site (± 0.07 SD). The mean time taken to enter the data for the DPM method using 100 recordings was significantly longer than the former method (mean = 3.3 decimal minutes per site ± 0.9 SD, P < 0.001) (Figure 29).

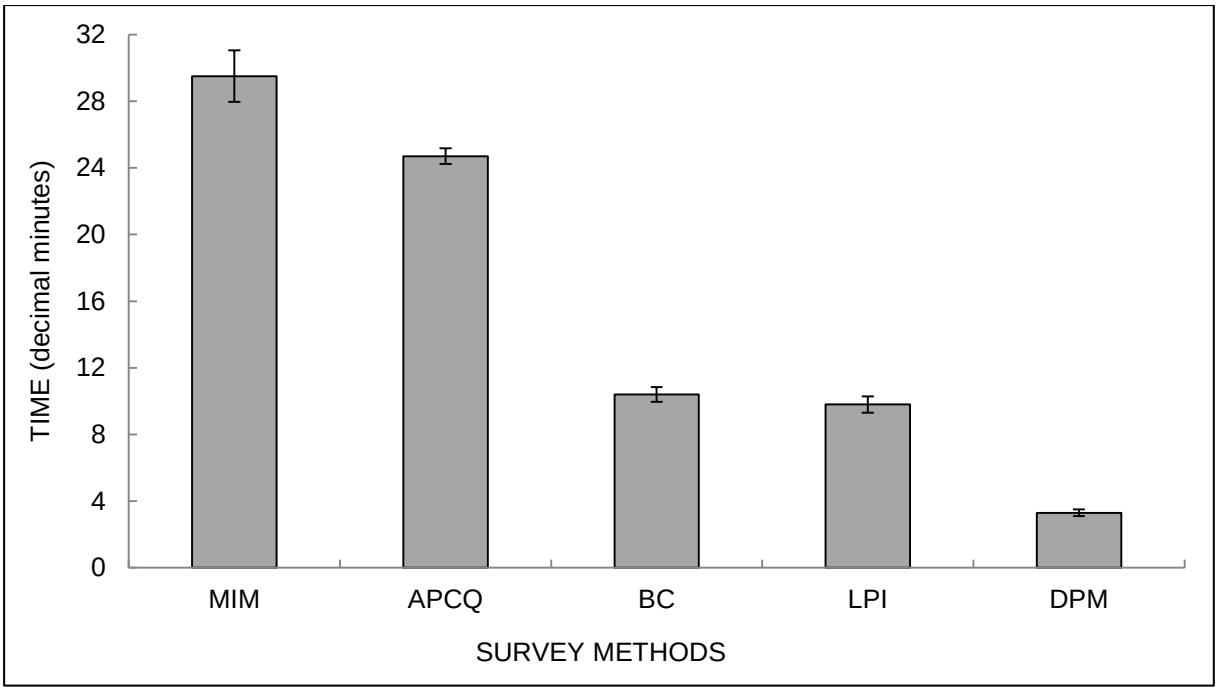

Figure 29: Mean number of decimal minutes taken for the data processing phase of each survey method across the land uses (where error bars reflect Standard Error).

3.4 Hypotheses testing

Table 11: Testing of significant differences between the parameters of rangeland health recorded by each survey method across the land uses. "+" indicates significance and "–" indicates no significance

Hypotheses	Rangeland Health Indicator	Parameter under assessment	Combined land uses	MGR	MCR
$H_0$1a (significant differences between the MIM and APCQ methods)	**Grass species composition, abundance and dominance**	Grass species recorded	+	-	+
		Dominant grass species recorded	+ - - - - + + + +	- + + + - - - - +	+ - - - - - - + +
		Hypothesis result	**Reject**	**Fail to reject**	**Fail to reject**
	Basal cover	Basal strikes	-	+	-
		Distance-to-herbaceous tuft	-	-	-
		Tuft diameter	N/A	N/A	N/A
		Hypothesis result	**Fail to reject**	**Fail to reject**	**Fail to reject**
	Woody species composition, abundance and dominance	Woody species recorded	+	-	+
		Dominant woody species recorded	+ + - + + - - + + + + + + +	- + - - + - - - - - - - + +	+ - - - - - - - - - - + +
		Hypothesis result	**Reject**	**Fail to reject**	**Fail to reject**
	Index of tree height (structure)	Height classes 1, 2, 3 and 4	+ + + +	+ + + +	+ + + +
		Hypothesis result	**Reject**	**Reject**	**Reject**
	Tree density	Tree density	+	+	+
		Hypothesis result	**Reject**	**Reject**	**Reject**

Hypotheses	Rangeland Health Indicator	Parameter under assessment	Combined land uses	MGR	MCR
$H_0$1b (significant differences between the MIM and BC methods)	**Grass species composition, abundance and dominance**	Grass species recorded	-	+	-
		Dominant grass species recorded	+ - + - - - - + +	- + + + - - - - +	- - - - - - - - +
		Hypothesis result	**Fail to reject**	**Fail to reject**	**Fail to reject**
	Basal cover	Basal strikes	N/A	N/A	N/A
		Distance-to-herbaceous tuft	-	-	-
		Tuft diameter	+	+	+
		Hypothesis result	**Fail to reject**	**Fail to reject**	**Fail to reject**
$H_0$1c (significant differences between the MIM and LPI methods)	**Grass species composition, abundance and dominance**	Grass species recorded	+	-	+
		Dominant grass species recorded	+ + + + + - + + +	+ + - + + - - - -	+ + + + + - - + +
		Hypothesis result	**Reject**	**Fail to reject**	**Reject**
	Basal cover	Basal strikes	-	-	-
		Distance-to-herbaceous tuft	N/A	N/A	N/A
		Tuft diameter	N/A	N/A	N/A
		Hypothesis result	**Fail to reject**	**Fail to reject**	**Fail to reject**
	Woody species composition, abundance and dominance	Woody species recorded	+	+	+
		Dominant woody species recorded	+ + + + + - + + + + + + + +	+ + + + + - + + + + + + + +	+ + + + + - - - - + + + + +
		Hypothesis result	**Reject**	**Reject**	**Reject**

Hypotheses	Rangeland Health Indicator	Parameter under assessment	Combined land uses	MGR	MCR
	Index of tree height (structure)	Height classes 1, 2, 3 and 4	+ + + -	+ + + -	+ + + -
		Hypothesis result	**Reject**	**Reject**	**Reject**
	Canopy cover	Percentage canopy cover	+	+	+
		Hypothesis result	**Reject**	**Reject**	**Reject**
H_o1d (significant differences between 33 recordings and 100 recordings)	**Grass biomass**	**Grass biomass**	-	+	-
		Hypothesis result	**Fail to reject**	**Reject**	**Fail to reject**
H_o2 (significant differences in time to undertake the MIM method with the APCQ, BC, LPI and DPM methods)	**APCQ method**	**Time (fieldwork)**	+	+	+
		Time (data processing)	+	N/A	N/A
		Hypothesis result	**Reject**	**Reject**	**Reject**
	BC method	**Time (fieldwork)**	+	+	+
		Time (data processing)	+	N/A	N/A
		Hypothesis result	**Reject**	**Reject**	**Reject**
	LPI method	**Time (fieldwork)**	+	+	+
		Time (data processing)	+	N/A	N/A
		Hypothesis result	**Reject**	**Reject**	**Reject**

Hypotheses	Rangeland Health Indicator	Parameter under assessment	Combined land uses	MGR	MCR
	DPM (100 recordings)	**Time (fieldwork)**	+	+	+
		Time (data processing)	+	N/A	N/A
		Hypothesis result	**Reject**	**Reject**	**Reject**

4. DISCUSSION

4.1 Herbaceous component

4.1.1 Species detection

Comparing the methods with each other across the land uses with regards to the mean percentage of grass species recordings, revealed a consistency in the "species recording potential" (SRP) of each method. The SRP of a survey method can be referred to as the potential of the method to record all species occurring within the homogenous vegetation unit in which the monitoring site occurs. The similar range of accuracy for each method in recording a percentage of species provides strong evidence to be used as a baseline indication of the SRP of each method across the land uses. Important mention should be made that these are percentages relative to the total number of species recorded by all of the survey methods across the land uses. Ultimately, methods that contribute more to the overall suite of species recorded in an area will display a stronger SRP, as was the case with the APCQ method.

The ability of the APCQ method to detect a greater mean percentage and mean number of grass species can be largely accredited to the size of the transect. The greater area covered by the APCQ method increases the probability for detecting species that are characteristic of different vegetation communities and ecotones, or zones of ecological transition (Araújo 2002, Solaimani and Shokrian 2011). Similar to the APCQ method, the greater percentage species recorded by the LPI method (in comparison to the MIM method) across the MGR and MCR could be accredited to the longer length of the transect. These landscapes are heterogeneous at relatively small scales and this finding confirms that survey methods which potentially cover larger areas (longer straight-line transects), may cover more than one vegetation community and result in a greater total number of species being recorded.

Noteworthy is that due to the nature of the study in contributing to a long-term vegetation monitoring programme (Wolfaard 2013), previous site selection did not entirely account for the use of methods that cover larger areas. Different results could be expected if the monitoring sites were established such that all considerations of the survey methods are accounted for i.e. the SRP of all survey methods will likely be similar when sites are selected within the same vegetation community. Evidence for this can be seen in the similar mean number of species recorded by the MIM and BC methods, which cover a similar sized area.

The BC method recorded a greater percentage of species than the LPI method (except in the MGR), which suggests that a greater standardised sample size for the herbaceous component of each method affects the probability of recording a greater number and percentage of grass species. When comparing the MIM method with the BC method in this regard, one would expect the MIM method to potentially detect a lower number of grass species than the BC method, as the standardised number of grass recordings per monitoring

site is lower for the MIM method (given no annuals are recorded). This held true for the percentage of the total number of grass species recorded, where the total number recorded using the MIM method remained below the standardised number of recordings of the BC method (100 recordings) for all monitoring sites. When comparing the mean number of grass species recorded however, overall the MIM method recorded a greater mean number of grass species than the BC method. Further investigation is required to determine possible explanation/s for this finding. It is also important to recognise the degree of biological importance of observations in relation to statistical significance, where observations that show significant statistical evidence may have less significant ecological/biological importance (Lovell 2013).

The LPI method combines measured parameters of the herbaceous and woody component for a standardised sample size of 100 recordings per site. The combination of the herbaceous and woody component in the same survey (along the same transect line in this case) means there is an increased probability to record fewer grass species compared to methods that focus solely on measuring the herbaceous layer (BC method) or which have separate survey components for the herbaceous and woody layers (MIM and APCQ methods). Moreover, the standardised sample size of 100 recordings for the herbaceous component each for the APCQ and BC methods, and the standard minimum sample size of 75 recordings for the MIM method (given no annuals are recorded), means there is greater potential for these methods to detect a greater number of grass species than the LPI method. The mean number of grass species recorded using the LPI method is therefore somewhat 'diluted', as it is dependent on the number of times a grass individual intercepts the tape (if at all) before other herbaceous or woody individuals. This could be viewed as a limitation of the LPI method, depending on the perspective of the desired outcomes for monitoring.

4.1.2 Grass species composition, abundance and dominance

Analysis of the statistical differences in the "grass species composition, abundance and dominance" component of hypothesis 1 for the APCQ ($H_0$1a), BC ($H_0$1b) and LPI methods ($H_0$1c) yielded the following conclusions:

- Significant differences in the grass species composition, abundance and dominance recordings between the MIM method and APCQ method result in the rejection of hypothesis $H_0$1a across both land uses (combined). The lack of significant differences when comparing land uses separately however, results in failure to reject hypothesis $H_0$1a for both the MGR and MCR.
- The lack of significant differences in grass species composition, abundance and dominance recorded with the BC method results in failure to reject hypothesis $H_0$1b for the combined land uses, and the MGR and MCR separately.
- Significant differences in the grass species composition, abundance and dominance recorded with the LPI method across both land uses (combined) and for the MCR

separately result in the rejection of hypothesis $H_0$1c for the land use scales. The lack of significant differences between the MIM and LPI methods in the MGR results in failure to reject hypothesis $H_0$1c for the MGR.

All five dominant grass species recorded by the MIM method across both land uses were perennials. The only method that recorded an annual grass species (*Brachiaria deflexa*) as one of the dominant species in both the MGR and MCR was the BC method. *Brachiaria deflexa* is an important pioneer species during the time of drought and is considered to have an average grazing value (Van Oudtshoorn 2012). This species germinates after rainfall (particularly towards the end of the wet season), and is almost always found in the shade of other vegetation. From our results it was evident that there was a greater abundance of *Brachiaria deflexa* in the MCR than the MGR. Analysis of the tree density data illustrates that there were more trees recorded in the MCR than the MGR, which would assist in explaining the greater abundance of this species in the MCR along with its affinity for shade.

The strong SRP of *Brachiaria deflexa* by the BC method shifts attention to the remaining dominant species, and it is evident that the BC method recorded a greater mean abundance of each dominant species than the other methods. These findings, along with the nature of the BC method, provides evidence that a compact transect that records points uniformly in a vegetation community is best suited when more accurate measures of dominant grass species abundance are required.

The lower mean abundance of most of the dominant grass species recorded using the LPI method can be largely accredited to the nature of the method and the 'dilution' effect (as described in the previous section), in that the method does not have a standard sample size for the herbaceous component, but rather the sample size is dependent on the number of times individual grasses intercept the line before other recorded components (if at all).

Of the five dominant grasses recorded by the MIM method, *Panicum maximum* and *Digitaria eriantha* are regarded as sub-climax/climax species with a high grazing value (Van Oudtshoorn 2012). *Urochloa mosambicensis* is regarded a pioneer or sub-climax species with an average grazing value, while *Aristida adscensionis* and *Aristida congesta var. barbicollis* are considered pioneer species that are of poor grazing value (Van Oudtshoorn 2012).

Interestingly, *Panicum maximum* and *Digitaria eriantha* were the top two dominant species recorded by all methods in both the MGR and MCR, meaning that majority of the sites were characterised by grasses in a sub-climax/climax state. The remaining dominant grass species, being indicative of pioneer and sub-climax successional states, assist in our understanding of the health of the rangelands, which appear to exist in a flux of successional states. Such fluxes are likely a result of different management approaches and land use practises and the presence of different herbivore assemblages across the interface over time (Twine 2005, Grant et al. 2011, Wolfaard 2013).

When viewing both land uses combined, the APCQ and BC methods each recorded four of the five dominant grass species recorded by the MIM method, while the LPI method recorded three. This is not the case when viewing the MGR and MCR separately, where all of the methods each recorded four of the five dominant grass species of the MIM method. This similarity illustrates the effectiveness of each of the methods in recording the dominant grass species in an area.

4.1.3 Estimates of basal cover

Analysis of the statistical differences in the "basal cover" component of hypothesis 1 for the APCQ ($H_0$1a), BC ($H_0$1b) and LPI methods ($H_0$1c) yielded the following conclusions:

- The lack of significant differences in the basal cover recordings between the MIM and APCQ method results in failure to reject hypothesis $H_0$1a for the combined land uses, and the MGR and MCR separately.
- The lack of significant differences in basal cover recorded using the BC method results in failure to reject hypothesis $H_0$1b for the combined land uses, and the MGR and MCR separately.
- The lack of significant differences in basal cover recorded by the LPI method results in failure to reject hypothesis $H_0$1c for the combined land uses, and the MGR and MCR separately.

A higher number of basal strikes are indicative of greater basal cover in an area (Hardy and Tainton 1993), which for this study indicates a greater basal cover in the MGR than the MCR recorded by each method. Further evidence for the greater basal cover in the MGR is illustrated by the greater inter-tuft distances recorded by all methods in the MCR, as well as the smaller tuft diameters that were recorded at the latter land use. A direct relationship between greater basal cover, smaller inter-tuft distance and larger tuft size has been acknowledged by Buitenwerf et al. (2011).

Buitenwerf et al. (2011) illustrated the lag effect of previous rainfall seasons on the tuft diameter and inter-tuft distances between grasses. Often as a consequence of over-utilization and trampling, low basal cover is associated with degradation and poor rangeland condition, which results in further repercussions (Smet and Ward 2005, Snyman and du Preez 2005, Vetter and Bond 2012). Such repercussions include greater levels of soil compaction, increased water runoff and rates of soil erosion, reduced organic matter content and reduced water infiltration rates; which all affect ecosystem function and productivity (Smet and Ward 2005, Snyman and du Preez 2005, Vetter and Bond 2012).

The significantly lower mean tuft diameter recorded using the MIM method in comparison to the BC method is likely due to the MIM method recording forbs and sedges (in addition to annual and perennial grasses). In most cases the tuft diameter of forbs and sedges are smaller than that of annual and perennial grasses, which consequently would have reduced

the mean tuft diameter recorded using the MIM method. The BC method records only annual and perennial grasses (which generally have a larger tuft diameter than forbs and sedges), resulting in a greater mean tuft diameter recorded by the method. Although larger tufts are associated with greater basal cover, a common observation is that numerous large tufts can create greater inter-tuft distances, likely a result of greater shade production and the establishment-inhibiting effects it has on other grass individuals/species (Dugmore 2012).

4.1.4 Grass biomass

Analysis of the statistical differences in the "grass biomass" component of hypothesis 1 for 33 recordings (of the MIM method) and the DPM method that uses 100 recordings ($H_0$1d) yielded the following conclusions:

- The lack of significant differences in grass biomass estimated using 33 recordings and 100 recordings across combined land uses and in the MCR results in failure to reject hypothesis $H_0$1d for the land use scales.
- Significant differences in grass biomass estimated using 33 recordings and 100 recordings results in rejection of hypothesis $H_0$1d for the MGR.

There were no significant differences in the mean grass biomass recorded using both methods across combined land uses. Regarding the time taken to undertake the fieldwork phase of both methods (as later discussed in section 4.3.1), the time taken to undertake 33 recordings was significantly less than the method using 100 recordings. For management related purposes, this illustrates the advantage of using 33 recordings for time efficiency when estimating grass biomass across an extended area.

The mean grass biomass estimated using both 33 and 100 recordings was significantly greater in the MGR than the MCR, which hints at higher levels of veld utilization within the MCR or lower grazing pressures associated with the MGR. This could be a result of the different herbivore assemblages and numbers present within each land use, along with differences in the scale of utilization that exists between fenced communal rangelands and the rangelands of an open protected area (Wolfaard 2013).

Trollope and Potgieter (1986) found that increasing the number of DPM recordings taken at a monitoring site above 100 recordings did not increase the statistical accuracy for estimating grass fuel loads. The mean grass biomass recorded using 100 recordings in the MGR was just above 1500 $kgha^{-1}$, which is regarded as the lower limit for the involuntary spread of fire (Trollope and Potgieter 1986); while the method which uses 33 recordings estimated a mean grass biomass below this limit. In the MCR, both methods yielded grass biomass estimates significantly lower than those estimated in the MGR. The use of burning to remove moribund and unpalatable grasses is recommended when grass biomass reaches or exceeds 4000 $kgha^{-1}$ (Dugmore 2012). Burning of moribund material not only 'resets' the

system to allow for the establishment of various grass species, but also assists by suppressing smaller woody species establishment (mitigation of bush encroachment) and reduces the tick load in an area (Buitenwerf et al. 2011, Dugmore 2012). It should also be mentioned that burning contributes to the increase in global CO_2 levels, which results in woody densification of savannas (Good and Caylor 2011, Bond and Midgley 2012, Buitenwerf et al. 2012, Dugmore 2012). In cases where grass biomass is ≥ 4000 $kgha^{-1}$, stocking rates of animals should be reassessed as the likelihood of rangeland under-utilization as a result of understocking is greater (Dugmore 2012).

4.2 Woody component

4.2.1 Species detection

In section 4.1.1 it was observed that herbaceous SRP of survey methods had a direct relationship with the area covered by a method. Applying this observation to the woody component reveals alternative findings; where the similar percentage of woody species recorded by the MIM and APCQ methods (particularly when the MGR and MCR are viewed separately) illustrates a weaker relationship between transect size and woody SRP. Given that the relationship between SRP and transect size remained true for the herbaceous component, it illustrates that herbaceous ecotone transitions are more discrete than those of their woody counterparts in semi-arid savannas. Mapping of herbaceous and woody information separately will provide further information of ecotone distributions and dynamics, and can be included as sub-divisions of each vegetation type presented by Mucina and Rutherford (2006).

The LPI method recorded fewer species than both the MIM and APCQ methods across the land uses, which can largely be accredited to the dilution effect and dynamic sample size of the woody and herbaceous components (as explained in section 4.1.1). The greater woody SRP of the MIM method in comparison to the LPI method is likely a result of the area-based nature of the MIM method, which records all woody species within a belt transect rather than limited to those which intercept a line.

The greater total number of woody species recorded in the MCR than the MGR suggests findings that contradict those of Higgins et al. (1999) and Shackleton et al. (1994), who found a greater total number of woody species in protected areas than in communal rangelands. A possible explanation for the greater number of woody species recorded in the MCR could be that the increased grazing pressure reduces competition from herbaceous species, which facilitates woody plant vigour and ensures a high reproductive potential for woody species (Teague and Smit 1992, Dean et al. 1999, Higgins et al. 1999). This is further enhanced by the ability of woody savanna species to exhibit coppicing growth and withstand the effects of intense harvesting, given that individuals are provided the opportunity to reach/maintain reproductive maturity (Shackleton et al. 1994). Knowledge of the woody species

composition and density of an area will assist managers in identifying areas for concern of potential future bush encroachment.

4.2.2 Woody species composition, abundance and dominance

Analysis of the statistical differences for the "woody species composition, abundance and dominance" component of hypothesis 1 for the APCQ ($H_0$1a) and LPI methods ($H_0$1c) yielded the following conclusions:

- Significant differences in the woody species composition, abundance and dominance recordings between the MIM method and APCQ method result in the rejection of hypothesis $H_0$1a across both land uses (combined). The lack of significant differences when comparing land uses separately however, results in failure to reject hypothesis $H_0$1a for both the MGR and MCR.
- Significant differences in woody species composition, abundance and dominance recorded with the LPI method across the land uses combined, and for the MGR and MCR separately, results in the rejection of hypothesis $H_0$1c across the land uses.

Of the five woody species detected by the MIM method across both land uses (combined), four (*Strychnos madagascariensis*, *Ehretia amoena, Acacia exuvialis* and *Ormocarpum trichocarpum*) are considered small shrubs/trees (Van Wyk and Van Wyk 1997 and Schmidt et al. 2002), which is relevant to the discussion in section 4.2.3. All of these species provide browse for various game species and stock (primarily goats, but also cattle in the dry years). The only consistent species recorded by all three methods across both land uses (and the MGR and MCR separately) was *Combretum apiculatum*. This small to medium-sized tree occurs in well drained soils and also provides browse for various species of game and stock (Van Wyk and Van Wyk 1997).

Comparison of the remaining five dominant woody species recorded by the MIM method across both land uses (combined) with those of the APCQ and LPI methods revealed differences, where the only other dominant species recorded by the APCQ method, consistent with the MIM method, was *Acacia exuvialis*. The species *Strychnos madagascariensis* was the only other dominant species recorded by the LPI method that was consistent with the MIM method. This illustrates a degree of variability in the SRP of dominant woody species by each method which, similar to the discussion in 4.1.2, could be a result of transect size/length or the area-based nature of the MIM method.

The greater mean abundance of the five dominant woody species recorded by the MIM method (across both land uses combined) was greater than those of the same species recorded by the APCQ and LPI methods. A possible explanation for this could be that the area-based nature of the MIM method allows for recording of a greater mean abundance as the sample size is dynamic and largely dependent on the number of individuals occurring with the belt-transect. Consequently, survey methods that use area-based methods will yield

greater mean species abundances (particularly in areas of greater tree density or encroached areas) compared to those with a standardised sample size (APCQ), and even more so, methods that combine the herbaceous and woody component in a standard sample size (LPI).

4.2.3 Index of tree height (structure)

Analysis of the statistical differences for the "index of tree height (structure)" component of hypothesis 1 for the APCQ ($H_0$1a) and LPI methods ($H_0$1c) yielded the following conclusions:

- Significant differences in the tree height (structure) recorded by the MIM method in comparison to the APCQ method result in the rejection of hypothesis $H_0$1a across both land uses (combined), and for the MGR and MCR separately.
- Significant differences in the tree height (structure) recorded by the LPI method results in the rejection of hypothesis $H_0$1c across both land uses (combined), and for the MGR and MCR separately.

It has been postulated that area-based methods often oversample smaller woody trees and shrubs and under sample taller ones (Trollope et al. 2013). Our results agree with this, where it is evident that the MIM method (which uses an area-based survey) recorded significantly more woody individuals in height classes 1 and 2 (particularly height class 1) across both land uses (combined) compared to the APCQ and LPI methods. Although it is considered oversampling, the detection of smaller woody individuals is necessary for pre-emptive management of woody plant encroachment. Various environmental repercussions have been associated with bush encroachment i.e. the reduction or loss of biodiversity and the loss of various ecosystem goods and services (Parr et al. 2012, Coetsee et al. 2013). An increase in woody biomass can also negatively affect animal stocking rates; as animals are directly impacted on by 1. the reduction in grass cover and replacement of palatable grass species with unpalatable woody species, and 2. the reduced movement of animals and access of available forage within rangelands (Smet and Ward 2005, Gray and Bond 2013).

Comparing the number of taller individuals recorded across the land uses (combined) reveals that the APCQ method recorded significantly more individuals in height classes 3 and 4 than the MIM and LPI methods (which both recorded similar). This evidence raises controversy whether the APCQ method could oversample taller trees and provide a skewed representation of the woody structure of an area. Analysis of tree density is important for both the MIM and APCQ methods in this regard.

Large trees play an important ecological role as they provide food and ideal habitat sites for nesting birds and smaller mammals, and also supply shade and food for larger mammals (Belsky 1994, Dean et al. 1999). Furthermore, Dean et al. (1999) found increased soil concentrations of macronutrients beneath larger trees compared to the surrounding shrubland due to faeces, remains of carcasses and fallen nest material left below larger

trees; while Campbell et al. (1988) showed the ability of some larger trees to draw nutrients from deeper soil layers and distribute them as plant material on the ground. These nutrient hotspots facilitate the establishment and growth of other plant species, which contribute greatly to the biodiversity of an area.

Of particular interest regarding tree structure when comparing the MGR and MCR separately was the greater mean number of smaller individuals recorded using both the MIM and APCQ methods in the MGR. This goes against expectations that there would be more small woody species in communal rangelands than nearby protected areas, due to the reduced competition from grasses as a result of increased grazing pressures (Teague and Smit 1992, Dean et al. 1999). Also noteworthy is the lower mean number of taller individuals recorded using both methods in the MGR compared to the MCR. This would suggest that there was a greater impact on larger trees in the MGR by animals such as elephants than the impact of wood harvesting by community members in the MCR. Pellegrini et al. (2014) found that the complex interactions between herbivory and fire can result in greater mortality of larger trees, which has a negative effect on woody cover. In order to promote structural heterogeneity of the woody layer in protected areas, it is important that management give regard to the various factors that affect vegetation dynamics at various height classes (Van Wilgen et al. 2014).

4.2.4 Tree density

Analysis of the statistical differences for the "tree density" component of hypothesis 1 for the APCQ method ($H_0$1a) yielded the following conclusions:

- Significant differences in tree densities recorded by the MIM method in comparison to the APCQ method result in the rejection of hypothesis $H_0$1a across both land uses (combined), and for the MGR and MCR separately.

A large proportion of the significantly greater tree density recorded across the land uses with the MIM method in comparison to the APCQ method can be accredited to a high contribution of trees in height classes 1 and 2. Although this could be viewed as oversampling on behalf of the MIM method, here we see the importance of recording elaborate information of smaller height classes, so as to avoid potential future risks of woody bush encroachment (explained in section 4.2.3).

The tree density recorded using both the MIM and APCQ methods were slightly greater in the MCR than in the MGR, which illustrates the relationship between increased woody densities as a result of reduced competition from grass species due to grazing induced pressures (Teague and Smit 1992, Dean et al. 1999). Furthermore, the lower tree density in the MGR could be explained by the presence of species such as elephants, which increase the susceptibility of trees to mortality during the use of periodic fires as a management tool (Van Wilgen et al. 2008, Grant et al. 2011). The greater impact exerted on the woody

component in the MGR is better illustrated by the differences observed in the percentage canopy cover recorded across both land uses; however, the observed results of the LPI method cannot solely be accredited to elephants, but rather the nature of the method (as is explained in section 4.2.5).

With regards to the MIM method, in most cases a single dominant species contributed greatly towards the greater tree density i.e. *Strychnos madagascariensis* contributed largely to the number of individuals recorded at various sites in the Dixie communal rangelands, which ultimately increased the tree density recorded within the MCR. A similar effect can however be seen in the MGR with consideration to *Ehretia amoena*.

4.2.5 Canopy cover

Analysis of the statistical differences for the "canopy cover" component of hypothesis 1 for the LPI method ($H_0$1c) yielded the following conclusions:

- Significant differences in canopy cover recorded by the MIM method in comparison to the LPI method result in the rejection of hypothesis $H_0$1c across both land uses (combined), and for the MGR and MCR separately.

The significantly greater percentage canopy cover recorded using the MIM method compared to the LPI method across both land uses can be accredited to the methodology associated with both methods. The MIM method is designed to detect canopy cover at each meter mark, while the percentage canopy cover recorded by the LPI method is dependent on a woody individual intercepting the tape before a herbaceous individual (if at all). This can be seen as a set-back of the LPI method because there may be no record of tree canopy (even when a canopy is present) if an herbaceous individual intercepts the tape first.

The greater mean percentage canopy cover recorded using both the MIM and LPI methods in the MCR is consistent with the greater tree density recorded in the MCR compared to the MGR. Again, the greater percentage canopy cover recorded in the MCR can be accredited to the difference in land use practises, where the effect of long-term overgrazing promotes the establishment and growth of woody species (Teague and Smit 1992, Dean et al. 1999). The mechanisms which facilitate encroachment of woody species range from the ability of cattle to disperse woody seeds (Tietema et al. 1991 cited in Dean et al. 1999) to the increased rate of establishment and vigour of woody species as a result of the absence of fire and competition from herbaceous species (Bond and Van Wilgen 1996). From the perspective of the MGR, our findings are consistent with the findings of Scholtz et al. (2016), where woody cover was lower in areas which had been subject to greater elephant densities over a long period of time.

4.3 Time for fieldwork and data processing

Analysis of the statistical differences in the time taken to complete the MIM method (fieldwork and data processing) in comparison to the other methods yielded the following conclusions for hypothesis $H_0$2:

- Significant differences between the time taken to complete the MIM method in comparison to the APCQ, LPI, BC and DPM (100 recordings) methods result in the rejection of hypothesis $H_0$2 for all methods across both land uses (combined), and for the MGR and MCR separately.

4.3.1 Fieldwork

The APCQ method took the longest time to complete. This was followed by the MIM method, the BC method, the LPI method and finally the DPM method. A large portion of the time taken for the APCQ method related to the woody survey component, specifically the distance to each woody individual and particularly the closest individuals '> 2m' and the closest 'tallest' individuals. It follows therefore, that the greater the tree density, the longer the time taken to complete the site (e.g. MCR compared to the MGR). This can also be seen with the MIM method (area-based woody component), which took longer in the more dense MCR. Further, it took significantly more time to set out the tape for the transect for the APCQ method (larger area), particularly in areas where the bush was dense and where spine-armed species such as *Dalbergia melanoxylon* and *Dichrostachys cinerea* dominated.

The DPM component of the MIM method (33 recordings) took significantly less time than the method of 100 recordings, however this comes with a trade-off in accuracy of predicting the grass biomass at smaller spatial scales, which can be observed in results of the MGR. Across a larger spatial scales (combined land uses in this case), the time efficiency using 33 recordings provides an accurate estimation of grass biomass to determine both animal stocking rates and fire management strategies relating to the removal of moribund material and mitigation of bush encroachment.

The greater number of indicators of rangeland health recorded by the MIM and APCQ methods in comparison to the BC and LPI methods also contributed to the longer time taken to carry out the former mentioned methods. The MIM and APCQ methods however, provide more elaborate information on the herbaceous and woody components, which can be used to inform management strategies.

4.3.2 Data processing

The longest time was taken to enter data for the MIM method; which was followed by the APCQ, BC, LPI and DPM methods. Comparison of the time taken to enter herbaceous data for the MIM method revealed a large degree of consistency between each monitoring site. Comparing the time taken to enter data of the woody component for the MIM method however, revealed a large degree of variation, where monitoring sites with a greater tree density also took longer for data processing (MCR). Furthermore, the greater number of

indicators of rangeland health recorded by the MIM method contributes to the longer time taken to process data from the method.

The longer time taken to enter data of the APCQ method in comparison to the BC method can also be accredited to the greater number of indicators of rangeland health recorded using the former mentioned method. The same cannot be said when comparing the APCQ method with the LPI method however, which recorded the same number of indicators of rangeland health. In this case the differences observed can be accredited to the standardised sample size of the separate herbaceous and woody component for the APCQ method, while the LPI method combines the herbaceous and woody component in a standardised sample size (dilution effect).

5. CONCLUSIONS

The similar range of accuracy in recording a particular percentage of grass species when comparing the methods with themselves across land uses provides strong evidence to be used as a baseline indication of the SRP of each method. It is apparent that objectives which require a more accurate representation of species composition in homogenous vegetation communities necessitates the use of compact methods that encompass smaller sized areas, such as the MIM or BC method. The BC method utilizes a smaller area and also measures herbaceous indicators in a uniform matrix, which resulted in its efficiency for measuring dominant grass species abundance. When management objectives require knowledge of grass species potentially occurring at a greater scale however, the use of methods that encompass larger areas are better suited (such as the APCQ and LPI methods). It was found that larger or longer transects are more likely to transcend different vegetation communities and possibly ecotones, and as a result have more chance of recording a greater number of species. Furthermore, methods that record parameters of the herbaceous and woody component separately (MIM, APCQ and BC methods) have greater potential of recording more species than methods which experience a "dilution effect" as a result of combining measurements of the two components in a standardised sample size (such as the LPI method).

The similarity in dominant grass species recorded by each method reveals the efficiency of each method for determining species dominance in an area. The two most dominant grass species recorded by all methods in the MGR and MCR were *Panicum maximum* and *Digitaria eriantha* (both considered subclimax/climax grasses), indicating that most of the sites monitored existed in later successional stages of the succession spectrum. The three remaining dominant grass species, being indicative of pioneer and sub-climax states, provides further insight to assist our understanding of the successional flux in which these rangelands exist. This can likely be accredited to the disturbance-effects associated with various management approaches, different herbivore assemblages and different practises of land utilization across the interface.

There was a greater basal cover recorded in the MGR, which was illustrated by the increased number of basal strikes, smaller distance-to-herbaceous tuft and larger tuft sizes recorded by all methods in comparison to the MCR. The lower basal cover in the MCR is most likely the effect of higher veld utilisation by cattle and the resultant greater ability for woody plant establishment due to reduced competition from grasses, which operates as a negative feedback loop.

Estimation of grass biomass using 33 recordings showed no significant differences when compared with that estimated using 100 DPM recordings across both land uses (combined). The grass biomass estimated for the MGR using both methods was greater than that of the MCR, again hinting at the higher levels of veld utilisation in the MCR or lower grazing

pressures associated with the MGR. This is again likely a result of the different herbivore assemblages and numbers occurring within each land use, along with the difference in scale of utilization that exists across the interface. It was found that in the MGR, 100 DPM recordings estimated a mean grass biomass just above 1500 $kgha^{-1}$ (considered the minimum fuel load for the spread of fire), while 33 DPM recordings estimated a grass fuel load below this guideline figure. This would imply that there is a degree of uncertainty when using 33 DPM recordings at a smaller land use scale, which could affect management decisions for stocking rates and fire regimes. Review of both methods in the MCR, however, showed similar results that would not have a considerable effect on management decisions when using one or the other method.

With regards to the woody component, the similar percentages of woody species recorded by the MIM and APCQ methods (particularly when viewing the MGR and MCR separately) reveals a weaker relationship between SRP and transect size than was shown for the herbaceous component. Should this observation hold true for the herbaceous component however, this would illustrate that herbaceous ecotone transitions are more discrete than woody ecotones in semi-arid savannas. Given this however, comparing the suite of dominant woody species recorded by each method across both land uses revealed inconsistencies in the similarity of dominant species recorded, which can likely be accredited to the nature of the MIM, APCQ and LPI methods i.e. the MIM method being an area-based method, the APCQ method covering a large area and the dilution effect of the LPI method (as previously discussed).

There was a greater number of woody species recorded in the MCR than the MGR, possibly due to the reduction in competition from grasses (as a result of greater grazing pressures). A reduction in herbaceous competition facilitates woody plant vigour and ensures a high reproductive potential for woody species (given they reach/maintain reproductive maturity). In part, this also provides explanation for the greater tree density (recorded by the MIM and APCQ methods) and canopy cover (MIM and LPI methods) in the MCR than in the MGR.

A greater mean abundance of the five dominant species was recorded by the MIM method (across both land uses combined) in comparison to the APCQ and LPI methods. Area-based methods (such as the MIM) will yield greater mean species abundances (particularly in denser or encroached areas) compared to methods that use a standardised sample size (APCQ) or combine measurement of the herbaceous and woody component in a single sample size (LPI). Interestingly, four of the five dominant species detected by the MIM method across both land uses (combined) are regarded as small shrubs/trees, which has relevance to the discussion of height classes (structure) of trees recorded by each method.

A review of the methodology of the MIM method is necessary in order to improve the efficacy for detecting larger trees, which play an important ecological role in a system. The high rate of detection of smaller trees by the MIM method (as an area-based method) is

regarded as oversampling by some, however it is necessary to obtain an accurate measure of small-tree densities in order to predict potential cases of bush encroachment and plan for the necessary management intervention/s (in terms of time and cost), to address the problem and avoid ecological repercussions. A review of the APCQ method, on the other hand, is necessary to increase the efficiency of sampling within the shorter height classes, as these classes are where the potential for bush encroachment are first identified.

The fewer large trees recorded in the MGR than in the MCR suggests that animal species such as elephant had a greater effect on larger trees (with contribution by fire) in comparison to rates of mortality induced on larger trees by wood harvesting activities carried out in the communal rangelands.

Table 12 provides a visual comparison of the indicators of rangeland health recorded by each method along with the time taken to complete each method. Assessment of each survey method in terms of efficiency, rigour in application and comprehensiveness of rangeland health indicators recorded for management decision-making purposes, revealed that the MIM method proved to be the most efficient for recording the greatest number of ecological indicators for the time spent when monitoring the health and status of semi-arid savanna rangelands in southern Africa. This was followed by the APCQ method. Contributing to the comprehensiveness of indicators of rangeland health recorded using both of these 'adapted' methods (MIM and APCQ methods) in comparison to the other survey methods is that both methods are comprised of a separate herbaceous and woody survey component, which incorporates the methodology from more traditional survey methods.

The greater time spent undertaking the MIM method compared to the BC and LPI methods presented a clear trade-off between time and the number of rangeland health indicators recorded to inform management-related decisions. The MIM method took less time to undertake the fieldwork phase compared to the APCQ method, while the data processing phase took longer than the APCQ method. Overall, however, even if the data processing phase of the APCQ method was ignored in this study, the MIM method would still show better time efficiency than the APCQ method. There is opportunity to further reduce the time for data processing of the MIM method in the future with the development of digital data entry during the fieldwork phase.

There is difficulty in distinguishing the most efficient method between the BC and LPI methods, as overall the BC method records fewer indicators of rangeland health than the LPI method; however it records more elaborate information on the herbaceous component (which is the only component measured by the BC method). The greater number of indicators recorded by the LPI method (which combines measurement of the herbaceous and woody component) gives it potential to be regarded as more efficient (also because of the lower time taken to complete). This however, is not an accurate assumption considering the significant differences of numerous rangeland health indicators recorded when

compared to the MIM method. The efficiency of the BC and LPI methods would vary according to the monitoring and management objectives.

Table 12: Summary of the indicators of rangeland health recorded by each method and the time to complete each method. The grey-filled gradient provides visual representation of the mean time taken to complete each method (per site) as a percentage relative to the method that took the longest to complete (APCQ). The length of the shading down each column coincides with the mean time taken to complete each method (as a percentage of the method that took the longest)

		Survey method				
		MIM	**APCQ**	**BC**	**LPI**	**DPM**
Indicators/ parameters of rangeland health	**Species composition (herbaceous)**	X	X	X	X	-
	Mean number of species recorded (herbaceous)	X	X	X	X	-
	% herbaceous basal strikes	X	X	-	X	-
	Mean distance-to-tuft (mm)	X	X	X	-	-
	Mean tuft diameter (mm)	X	-	X	-	-
	Species composition (woody)	X	X	-	X	-
	Mean number of species recorded (woody)	X	X	-	X	-
	% Canopy cover	X	-	-	X	-
	Density (plants/ha)	X	X	-	-	-
	Grass biomass (kgha^{-1}); where *y* = disc height recordings	X	-[3]	-	-	X
	Mean time (decimal minutes)	109.5	156.8	59.6	42.7	22.3
	Time (as a percentage of the method that took longest)	70%	100%	38%	27%	14%

It is worthwhile to make mention again that although differences in the indicators of rangeland health detected by each method have not been statistically compared across the land uses; they have been included in this dissertation for observational purposes. It is clear that for some methods, there are differences between the land uses in the indicators of rangeland health recorded. This provides an opportunity for a follow-up study that

[3] This study did not record grass biomass during application of the APCQ method, however normal application of the method does record grass biomass

statistically tests the level of significance of these differences and what drivers may be causing the observed differences.

Rangeland health plays an important role in increasing connectivity/contact between wildlife, livestock and people. Rangelands that are more degraded results in greater competition of livestock with wildlife for ecosystem services (such as grazing), which brings wildlife and livestock closer together with an enhanced probability of disease transmission. It is therefore apparent that communal systems comprised of healthy rangelands will experience reduced competition for resources with conservation areas, or between people and their livestock, and wildlife.

The MIM method, currently applied to some 400 000 ha in the eastern Lowveld as well as in Zimbabwe and Gorongosa National Park in Mozambique, is considered an efficient and rigorous vegetation survey technique that can be used to detect comprehensive information of the health and status of rangelands across semi-arid savannas of southern Africa. Continuous future long-term monitoring using the MIM method (with a possible adjustment in the detection of taller trees) and collation of such data with information from other disciplines; is vital for informing management decisions that aim to optimise the ecological, social and economic well-being of protected areas such as the MGR and communal rangelands, such as those of the MTA.

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7. APPENDIX A: HERBACEOUS SPECIES PRESENCE/ABSENCE

Table 13: Grass species presence/absence based on detections by each method within the different land uses (where 1 = present, 0 = absent)

	MIM		APCQ		BC		LPI	
Grass species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Aristida adscensionis	1	1	1	1	1	1	1	1
Aristida congesta var. barbicollis	1	1	1	1	1	1	1	1
Aristida congesta var. congesta	1	1	1	1	1	1	1	1
Aristida stipitata	0	1	0	1	0	1	0	1
Bothriochloa insculpta	1	0	1	0	1	0	1	0
Bothriochloa radicans	1	0	1	0	1	0	1	0
Brachiaria deflexa	1	1	1	1	1	1	1	1
Brachiaria eruciformis	0	1	0	1	0	1	0	1
Brachiaria nigropedata	0	1	1	1	0	1	0	1
Brachiaria serrata	0	1	0	0	0	0	0	0
Cenchrus ciliaris	0	0	1	0	0	0	1	0
Chloris virgata	0	0	1	0	1	0	0	0
Cymbopogon caesius	0	0	1	0	0	0	0	0
Cymbopogon pospischilii	1	0	1	0	1	0	1	0
Cynodon dactylon	1	1	1	1	1	1	1	1
Dactyloctenium aegyptium	1	0	0	0	0	1	0	0
Dactyloctenium australe	1	1	1	1	1	1	1	1
Dactyloctenium giganteum	0	1	0	1	0	1	0	1

	MIM		APCQ		BC		LPI	
Grass species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Digitaria diagonalis	1	0	0	1	0	0	0	0
Digitaria eriantha	1	1	1	1	1	1	1	1
Digitaria longiflora	1	1	0	1	0	1	0	1
Diheteropogon amplectens	0	0	0	0	1	0	0	0
Enneapogon cenchroides	0	0	0	1	0	1	0	1
Eragrostis chloromelas	0	1	0	1	1	1	0	1
Eragrostis curvula	0	0	1	0	0	0	0	0
Eragrostis cilianensis	0	0	0	0	0	0	0	1
Eragrostis gummiflua	1	1	1	1	1	1	1	1
Eragrostis heteromera	0	1	0	1	0	1	0	0
Eragrostis rigidior	1	1	1	1	1	1	1	1
Eragrostis superba	1	1	1	1	1	1	1	1
Fingerhuthia africana	0	0	0	0	0	1	0	0
Heteropogon contortus	1	1	1	1	1	1	1	1
Hyparrhenia hirta	0	0	0	1	0	0	0	0
Melinis repens	1	1	1	1	1	1	1	1
Oropetium capense	1	0	1	1	1	0	1	0
Panicum coloratum	1	0	1	0	1	0	1	0
Panicum maximum	1	1	1	1	1	1	1	1
Panicum schinzii	0	0	1	0	0	0	0	0
Perotis patens	1	1	1	1	1	1	1	1

	MIM		APCQ		BC		LPI	
Grass species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Pogonarthria squarrosa	1	1	1	1	1	1	1	1
Schmidtia pappophoroides	0	1	1	1	0	0	1	1
Setaria sphacelata var. sericea	0	0	1	0	0	0	1	0
Setaria sphacelata var. sphacelata	1	0	1	1	1	1	1	0
Sporobolus africanus	0	1	0	0	0	1	0	0
Sporobolus fimbriatus	1	1	1	1	1	1	1	1
Sporobolus ioclados	1	0	1	0	1	0	1	0
Sporobolus nitens	0	0	0	1	0	1	0	1
Themeda triandra	1	1	1	1	1	1	1	1
Tragus berteronianus	0	1	1	1	0	1	1	1
Tricholaena monachne	1	1	1	1	1	1	1	1
Trichoneura grandiglumis	1	0	0	1	0	1	0	1
Urochloa mosambicensis	1	1	1	1	1	1	1	1
Urochloa oligotricha	1	0	1	0	1	0	1	0

8. APPENDIX B: HERBACEOUS DATA STATISTICS

Table 14: Statistics for all herbaceous parameters recorded by each of the methods across both land uses combined

	MIM				APCQTot					BC					LPI				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Total herbaceous individuals detected (annuals+perrenials+forbs)	50	88.8	9.4	1.3	50	100.0	0.1	0.0	<0.001	46	100.0	0.0	0.0	<0.001	50	60.9	21.3	3.0	<0.001
Total grass species detected	50	8.8	2.3	0.3	50	9.6	2.4	0.3	0.02	46	8.2	2.8	0.4	0.1	50	7.7	2.0	0.3	0.002
Total individual grasses detected (annuals+perrenials)	50	77.6	3.4	0.5	50	100.0	0.1	0.0	<0.001	46	100.0	0.0	0.0	<0.001	50	54.9	24.3	3.4	<0.001
Annuals detected - absolute	50	2.6	3.7	0.5	50	4.3	5.5	0.8	0.006	46	4.8	7.4	1.1	<0.001	50	0.6	1.3	0.2	<0.001
Perennials detected - absolute	50	75.0	0.0	0.0	50	95.7	5.5	0.8	<0.001	46	95.2	7.4	1.1	<0.001	50	54.3	24.9	3.5	<0.001
Forbs detected - absolute	50	11.2	7.5	1.1											50	5.9	4.5	0.6	<0.001
Annuals detected - % of total herbaceous recordings detected by each method	50	2.7	3.6	0.5	50	4.3	5.5	0.8	0.007	46	4.8	7.4	1.1	0.002	50	1.6	3.6	0.5	0.01
Perrenials detected - % of total herbaceous recordings detected by each method	50	85.2	8.7	1.2	50	95.7	5.5	0.8	<0.001	46	95.2	7.4	1.1	<0.001	50	85.0	16.1	2.3	0.9
Forbs detected -% of total herbaceous recordings detected by each method	50	12.0	7.2	1.0											50	13.4	13.7	1.9	0.7
Grass basal strikes - absolute	50	2.8	2.0	0.3	50	3.0	2.5	0.4	0.7						50	3.4	2.6	0.4	0.1
Grass basal strikes - % of total herbaceous recordings detected by each method	50	3.2	2.4	0.3	50	3.0	2.5	0.4	0.6						50	5.9	4.2	0.6	<0.001
Forb basal strikes - absolute	50	0.2	0.5	0.1											50	0.2	0.5	0.1	0.5
Forb basal strikes - % of total herbaceous recordings detected by each method	50	0.2	0.5	0.1											50	0.4	1.5	0.2	0.7
Distance forb - absolute	50	496.7	375.9	53.2															
Distance annual - absolute	50	132.4	200.9	28.4	50	275.2	394.9	55.8	0.004	46	281.5	504.3	74.4	<0.001					
Distance perennial - absolute	50	4219.6	1373.5	194.2	50	5505.2	1707.7	241.5	<0.001	46	5117.4	1468.1	216.5	<0.001					
Mean distance to herbaceous tuft/number of herbaceous recordings by each method	50	53.9	16.7	2.4	50	57.9	18.0	2.5	0.1	46	54.0	16.2	2.4	0.7					
Tuft forb - absolute	50	104.4	81.4	11.5															
Tuft ann. - absolute	50	26.8	35.6	5.0						46	50.3	78.6	11.6	0.003					
Tuft perren. - absolute	50	1594.4	385.4	54.5						46	2220.4	475.3	70.1	<0.001					
Mean herbaceous tuft diameter/number of herbaceous recordings by each method	50	19.7	5.1	0.7						46	22.7	4.3	0.6	<0.001					

Table 15: Statistics for all herbaceous parameters recorded by each of the methods in the MGR

	MIM				APCQTot					BC					LPI				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Total herbaceous individuals detected (annuals+perrenials+forbs)	25	84.0	5.8	1.2	25	100.0	0.2	0.0	<0.001	25	100.0	0.0	0.0	<0.001	25	75.9	14.1	2.8	0.04
Total grass species detected	25	8.4	2.6	0.5	25	9.0	2.8	0.6	0.4	25	7.2	2.8	0.6	0.01	25	7.8	2.2	0.4	0.2
Total individual grasses detected (annuals+perrenials)	25	76.6	2.4	0.5	25	100.0	0.2	0.0	<0.001	25	100.0	0.0	0.0	<0.001	25	72.8	14.7	2.9	0.3
Annuals detected - absolute	25	1.6	2.4	0.5	25	2.8	4.5	0.9	0.03	25	2.6	3.5	0.7	0.03	25	0.2	0.5	0.1	0.003
Perennials detected - absolute	25	75.0	0.0	0.0	25	97.1	4.5	0.9	<0.001	25	97.4	3.5	0.7	<0.001	25	72.7	14.7	2.9	0.5
Forbs detected - absolute	25	7.5	5.6	1.1											25	3.1	2.2	0.4	<0.001
Annuals detected - % of total herbaceous recordings detected by each method	25	1.7	2.7	0.5	25	2.8	4.5	0.9	0.004	25	2.6	3.5	0.7	0.07	25	0.2	0.5	0.1	0.003
Perrenials detected - % of total herbaceous recordings detected by each method	25	89.5	6.4	1.3	25	97.2	4.5	0.9	<0.001	25	97.4	3.5	0.7	<0.001	25	95.5	3.2	0.6	<0.001
Forbs detected -% of total herbaceous recordings detected by each method	25	8.6	6.2	1.2											25	4.3	3.3	0.7	0.002
Grass basal strikes - absolute	25	3.0	2.1	0.4	25	4.4	2.3	0.5	0.01						25	3.4	2.8	0.6	0.4
Grass basal strikes - % of total herbaceous recordings detected by each method	25	3.6	2.5	0.5	25	3.3	2.3	0.5	0.1						25	4.9	3.8	0.8	0.08
Forb basal strikes - absolute	25	0.1	0.3	0.1											25	0.1	0.3	0.1	0.8
Forb basal strikes - % of total herbaceous recordings detected by each method	25	0.1	0.3	0.1											25	0.1	0.3	0.1	0.8
Distance forb - absolute	25	329.6	314.4	62.9															
Distance annual - absolute	25	76.4	125.3	25.1	25	164.0	275.4	55.1	0.09	25	135.6	211.6	42.3	0.07					
Distance perennial - absolute	25	3893.5	1426.3	285.3	25	5307.2	1805.9	361.2	<0.001	25	5081.2	1486.2	297.2	<0.001					
Mean distance to herbaceous tuft/number of herbaceous recordings by each method	25	51.0	18.6	3.7	25	54.8	17.7	3.5	0.5	25	52.2	15.4	3.1	0.4					
Tuft forb - absolute	25	65.4	58.1	11.6															
Tuft ann. - absolute	25	17.4	28.0	5.6						25	27.6	38.8	7.8	0.08					
Tuft perren. - absolute	25	1736.7	449.7	89.9						25	2415.4	453.1	90.6	<0.001					
Mean herbaceous tuft diameter/number of herbaceous recordings by each method	25	21.9	5.8	1.2						25	24.5	5.5	1.1	<0.001					

Table 16: Statistics for all herbaceous parameters recorded by each of the methods in the MCR

	MIM				APCQTot					BC					LPI				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Total herbaceous individuals detected (annuals+perrenials+forbs)	25	93.6	9.9	2.0	25	100.0	0.0	0.0	0.006	21	100.0	0.0	0.0	0.02	25	45.8	15.9	3.2	<0.001
Total grass species detected	25	9.2	2.0	0.4	25	10.2	1.7	0.3	0.01	21	9.2	2.5	0.5	0.5	25	7.6	1.9	0.4	0.03
Total individual grasses detected (annuals+perrenials)	25	78.6	4.4	0.9	25	100.0	0.0	0.0	<0.001	21	100.0	0.0	0.0	<0.001	25	37.0	17.9	3.6	<0.001
Annuals detected - absolute	25	3.6	4.4	0.9	25	5.7	6.1	1.2	0.05	21	7.4	9.7	2.1	0.01	25	1.1	1.7	0.3	0.003
Perennials detected - absolute	25	75.0	0.0	0.0	25	94.3	6.1	1.2	<0.001	21	92.6	9.7	2.1	<0.001	25	35.9	18.5	3.7	<0.001
Forbs detected - absolute	25	15.0	7.4	1.5											25	8.8	4.5	0.9	<0.001
Annuals detected - % of total herbaceous recordings detected by each method	25	3.6	4.1	0.8	25	5.7	6.1	1.2	0.04	21	7.4	9.7	2.1	0.02	25	3.0	4.8	1.0	0.3
Perrenials detected - % of total herbaceous recordings detected by each method	25	80.8	8.7	1.7	25	94.3	6.1	1.2	<0.001	21	92.6	9.7	2.1	<0.001	25	74.4	17.0	3.4	0.02
Forbs detected -% of total herbaceous recordings detected by each method	25	15.5	6.6	1.3											25	22.5	14.1	2.8	0.01
Grass basal strikes - absolute	25	2.6	1.9	0.4	25	1.6	1.6	0.3	0.05						25	3.2	2.5	0.5	0.2
Grass basal strikes - % of total herbaceous recordings detected by each method	25	2.8	2.2	0.4	25	1.6	1.6	0.3	0.03						25	6.8	4.5	0.9	<0.001
Forb basal strikes - absolute	25	0.4	0.6	0.1											25	0.3	0.6	0.1	0.7
Forb basal strikes - % of total herbaceous recordings detected by each method	25	0.4	0.6	0.1											25	0.8	2.0	0.4	0.5
Distance forb - absolute	25	663.8	362.5	72.5															
Distance annual - absolute	25	188.4	245.2	49.0	25	386.4	465.6	93.1	0.01	21	455.2	678.4	148.0	0.002					
Distance perennial - absolute	25	4545.6	1263.3	252.7	25	5793.2	1616.0	323.2	<0.001	21	5160.5	1481.7	323.3	0.02					
Mean distance to herbaceous tuft/number of herbaceous recordings by each method	25	56.8	14.3	2.9	25	61.0	18.2	3.6	0.1	21	56.2	17.1	3.7	0.7					
Tuft forb - absolute	25	143.4	83.6	16.7															
Tuft ann. - absolute	25	36.2	40.2	8.0						21	77.4	100.3	21.9	0.02					
Tuft perren. - absolute	25	1452.2	242.6	48.5						21	1988.3	397.1	86.7	<0.001					
Mean herbaceous tuft diameter/number of herbaceous recordings by each method	25	17.5	3.3	0.7						21	20.7	3.3	0.7	0.001					

Table 17: Statistics for grass biomass recorded by the DPM methods using 33 and 100 recordings across the land uses

	33 recordings				100 recordings				
	n	mean	SD	SE	n	mean	SD	SE	p-value
Combined	47	816.7	897.3	130.8847	47	933.3	990.7	144.5084	0.1
Manyeleti	25	1295.4	841.9	168.38	25	1558	845.7	169.14	0.01
Mnisi	22	272.7	609	129.8392	22	223.4	582.9	124.2747	0.2

9. APPENDIX C: WOODY SPECIES PRESENCE/ABSENCE

Table 18: Woody species presence/absence based on detections by each method within the different land uses (where 1 = present, 0 = absent)

	MIM method		APCQ method		LPI method	
Woody species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Acacia ataxacantha	0	1	0	0	0	0
Acacia burkei	0	0	0	1	0	0
Acacia exuvialis	1	1	1	1	1	1
Acacia gerrardii	1	1	1	1	1	1
Acacia mellifera	1	1	1	1	1	0
Acacia nigrescens	1	1	1	1	1	1
Acacia nilotica	0	1	0	1	0	1
Albizia harveyi	1	1	1	1	1	1
Balanites maughamii	0	1	1	1	0	1
Berchemia discolor	1	0	1	1	0	0
Bolusanthus speciosus	1	1	1	1	0	0
Boscia albitrunca	0	1	1	0	0	0
Carissa edulis	1	0	0	0	0	0
Cassia abbreviata	1	0	1	0	1	0
Cassine transvaalensis	1	1	1	1	0	0
Catunaregam spinosa	0	1	0	1	0	1
Cissus cornifolia	1	1	1	1	0	1

	MIM method		APCQ method		LPI method	
Woody species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Coddia rudis	0	1	1	0	0	1
Combretum apiculatum	1	1	1	1	1	1
Combretum collinum	1	0	1	1	1	1
Combretum hereroense	1	1	1	1	1	1
Combretum imberbe	1	1	1	1	0	1
Combretum molle	1	0	1	1	1	1
Combretum zeyheri	1	1	1	1	1	1
Commiphora africana	1	1	1	1	1	1
Commiphora harveyi	0	1	1	1	0	1
Commiphora mollis	1	1	1	1	0	0
Dalbergia melanoxylon	1	1	1	1	0	1
Dichrostachys cinerea	1	1	1	1	1	1
Diospyros mespiliformis	1	1	1	1	0	1
Dombeya rotundifolia	0	1	0	1	0	1
Ehretia amoena	1	1	1	1	1	1
Euclea divinorum	1	1	1	1	1	1
Euclea natalensis	1	1	1	1	0	0
Flueggea virosa	1	1	1	1	1	1
Gardenia volkensii	1	1	1	1	0	0
Gossypium herbacium	1	1	0	0	0	0
Grewia bicolor	1	1	1	1	1	1

	MIM method		APCQ method		LPI method	
Woody species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Grewia flavescens	1	1	1	1	1	1
Grewia monticola	1	1	1	1	0	1
Gymnosporia buxifolia	1	1	1	1	1	1
Gymnosporia maranguensis	0	1	0	1	0	0
Gymnosporia senegalensis	1	1	1	1	1	1
Lannea discolor	0	1	1	1	0	1
Lannea schweinfurthii	1	1	1	1	0	1
Lippia rehmannii	1	0	0	0	0	0
Manilkara mochisia	1	0	0	0	0	0
Mundulea sericea	1	0	1	0	0	0
Ormocarpum trichocarpum	1	1	1	1	1	1
Ozoroa engleri	0	1	0	0	0	0
Ozoroa paniculosa	0	0	0	1	0	0
Pappea capensis	1	0	0	0	0	0
Pavetta schumanniana	0	1	1	0	0	0
Peltophorum africanum	1	1	1	1	1	1
Philenoptera violacea	1	1	1	1	1	1
Phyllanthus reticulatus	0	0	0	1	0	1
Pterocarpus angolensis	0	0	0	1	0	1
Pterocarpus rotundifolius	1	1	1	1	1	1

	MIM method		APCQ method		LPI method	
Woody species	**MGR**	**MCR**	**MGR**	**MCR**	**MGR**	**MCR**
Rhoicissus tridentata	1	0	0	0	0	0
Schotia brachypetala	1	1	0	1	0	0
Sclerocarya birrea	1	1	1	1	1	1
Searsia gueinzii	0	0	0	1	0	1
Searsia pyroides	1	0	0	0	0	0
Senna petersiana	0	1	0	1	0	1
Spirostachys africana	0	1	1	1	0	1
Strychnos madagascariensis	0	1	1	1	0	1
Strychnos spinosa	0	0	0	1	0	0
Terminalia sericea	1	1	1	1	1	1
Turraea nilotica	0	1	0	0	0	0
Vangueria infausta	0	1	0	1	0	1
Ximenia caffra	1	1	1	1	1	1
Ziziphus mucronata	1	1	1	1	1	1

10. APPENDIX D: WOODY DATA STATISTICS

Table 19: Statistics for all woody parameters recorded by each of the methods across both land uses combined

	MIM				APCQT1					APCQT2					APCQTot					LPI				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Total species detected	50	14.3	4.0	0.6	50	12.3	2.8	0.4	0.004	50	12.4	2.4	0.3	<0.001	25	16.4	3.9	0.6	0.001	50	5.9	4.1	0.6	<0.001
Trees in height class 1 - absolute	50	84.8	92.0	13.0	50	11.9	3	0.4	<0.001	50	11.5	3.4	0.5	<0.001	50	23.4	5.7	0.8	<0.001	50	1.6	1.9	0.3	<0.001
Trees in height class 2 - absolute	50	25.3	21.3	3.0	50	4.7	3.1	0.4	<0.001	50	5.1	3.2	0.5	<0.001	50	10	5.4	0.8	<0.001	50	2.3	2.6	0.4	<0.001
Trees in height class 3 - absolute	50	15.4	13.3	1.9	50	17.2	6.5	0.9	0.1	50	17.4	6.3	0.9	0.1	50	34.6	11.9	1.7	<0.001	50	5.9	5.6	0.8	<0.001
Trees in height class 4 - absolute	50	3.8	5.2	0.7	50	11.8	6.8	1.0	<0.001	50	12.3	6.8	1.0	<0.001	50	24.1	13.2	1.9	<0.001	50	4	5.5	0.8	0.9
Total individual trees detected	50	129.3	97.5	13.8	50	45.6	3.5	0.5	<0.001	50	46.2	3.4	0.5	<0.001	50	91.8	6.6	0.9	0.01	50	13.8	11.8	1.7	<0.001
Tree density (plants/ha)	50	6465	4874.7	689.4											50	4139	2699.8	381.8	<0.001					
% canopy cover	50	41.5	23.5	3.3																50	13.8	11.8	1.7	<0.001

Table 20: Statistics for all woody parameters recorded by each of the methods in the MGR

	MIM				APCQT1					APCQT2					APCQTot					LPI				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Total species detected	25	14.7	4.2	0.8	25	12.3	2.8	0.6	0.01	25	12.9	2.5	0.5	0.01	25	16.4	4.7	0.9	0.08	25	3.4	2.7	0.5	<0.001
Trees in height class 1 - absolute	25	100.4	121.6	24.3	25	13.8	2.6	0.5	<0.001	25	13.2	2.8	0.6	<0.001	25	27	4.3	0.9	<0.001	25	1.3	1.7	0.3	<0.001
Trees in height class 2 - absolute	25	17.7	15.2	3.0	25	3.4	3.4	0.7	<0.001	25	3.9	3	0.6	<0.001	25	7.3	5.2	1.0	0.001	25	0.9	1	0.2	<0.001
Trees in height class 3 - absolute	25	7	6.8	1.4	25	16.7	5.7	1.1	<0.001	25	17.4	5.3	1.1	<0.001	25	34.1	9.6	1.9	<0.001	25	2.4	2.5	0.5	0.001
Trees in height class 4 - absolute	25	1	1.8	0.4	25	9.9	5.8	1.2	<0.001	25	10.4	5.7	1.1	<0.001	25	20.3	10.8	2.2	<0.001	25	1.3	2.3	0.5	1
Total individual trees detected	25	126.1	122.6	24.5	25	43.8	4.2	0.8	<0.001	25	44.9	4.4	0.9	<0.001	25	88.7	8.3	1.7	0.4	25	5.8	5.4	1.1	<0.001
Tree density (plants/ha)	25	6306	6128	1225.6											25	3683.5	2613.5	522.7	<0.001					
% canopy cover	25	25.7	16	3.2																25	5.8	5.4	1.1	<0.001

Table 21: Statistics for all woody parameters recorded by each of the methods in the MCR

	MIM				APCQT1					APCQT2					APCQTot					LPI				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Total species detected	25	13.8	3.9	0.8	25	12.4	2.8	0.6	0.1	25	11.8	2.2	0.4	0.02	25	16.4	3.1	0.6	0.006	25	8.3	3.8	0.8	<0.001
Trees in height class 1 - absolute	25	69.2	44.6	8.9	25	10	2.2	0.4	<0.001	25	9.8	3.1	0.6	<0.001	25	19.8	4.6	0.9	<0.001	25	1.9	2	0.4	<0.001
Trees in height class 2 - absolute	25	33	24	4.8	25	6	2.2	0.4	<0.001	25	6.2	3.1	0.6	<0.001	25	12.2	4.5	0.9	<0.001	25	3.7	2.9	0.6	<0.001
Trees in height class 3 - absolute	25	23.7	13.2	2.6	25	17.6	7.3	1.5	0.05	25	17.4	7.2	1.4	0.04	25	35	14.1	2.8	0.01	25	9.5	5.7	1.1	<0.001
Trees in height class 4 - absolute	25	6.6	6	1.2	25	13.8	7.3	1.5	<0.001	25	14.2	7.5	1.5	<0.001	25	27.9	14.4	2.9	<0.001	25	6.7	6.5	1.3	1
Total individual trees detected	25	132.5	66.1	13.2	25	47.4	0.9	0.2	<0.001	25	47.6	0.7	0.1	<0.001	25	95	1.1	0.2	0.006	25	21.8	11	2.2	<0.001
Tree density (plants/ha)	25	6624	3303	660.6	25										25	4595	2760	552.0	<0.001					
% canopy cover	25	57.2	18.9	3.8	25															25	21.8	11	2.2	<0.001

11. APPENDIX E: TIME DATA STATISTICS

Table 22: Statistics for the time taken to complete each method across the land uses

	MIM				APCQ					BC					LPI					DPM				
	n	mean	SD	SE	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value	n	mean	SD	SE	p-value
Time for Fieldwork	50	80	19.4	2.743574	50	132.1	29.2	4.129504	<0.001	46	49.2	10.7	1.577629	<0.001	50	32.9	7.3	1.032376	<0.001	46	19	5.3	0.781442	<0.001
Time for Data entry	20	29.5	6.9	1.542887	20	24.7	2.1	0.469574	0.006	20	10.4	2	0.447214	<0.001	20	9.8	2.2	0.491935	<0.001	19	3.3	0.9	0.206474	<0.001
Time for Fieldwork																								
Manyeleti	25	76.0	21.7	4.34	25	108.4	15.9	3.18	<0.001	25	45.5	8.8	1.76	<0.001	25	32.2	6.5	1.3	<0.001	24	19.0	4.9	1.000208	<0.001
Mnisi	25	84.1	16.7	3.34	25	155.7	18.0	3.6	<0.001	21	53.7	11.2	2.44404	<0.001	25	33.6	8.2	1.64	<0.001	22	19.0	5.9	1.257884	<0.001
p-value (comparison Mnis-Manyeleti)	0.1				<0.001					0.008					0.6					0.8				

12. APPENDIX F: CARDINAL SITE PHOTOGRAPHS

Table 23: Cardinal site photographs

Site	Cardinal photograph	Date & direction of tape	Comments
DixA1		26 March 2015 Southwest	• No brush packing. • No BC done – site was too thick with *S. madagascariensis* for accurate distribution of points within 20 x 20m quad.

Site	Cardinal photograph	Date & direction of tape	Comments
DixA2		24 March 2015 South	• No brush packing. • Thick with *S. madagascariensis*
DixA3		25 March 2015 Southwest	• No brush packing. • Open site with numerous tall *C. imberbe* on eastern side of drainage line.

Site	Cardinal photograph	Date & direction of tape	Comments
DixA4	No photographs taken.	South	• No brush packing. • Lots of leaf litter. • No photographs taken.
DixA5		01 April 2015 South	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
DixA6		07 April 2015 Southeast	• None.
DixA7		18 May 2015 Southeast	• T2 of APCQ was done to the left due to the road.

Site	Cardinal photograph	Date & direction of tape	Comments
FMS21		23 March 2015 Southwest	• No brush packing.
UtA1		04 March 2015 South	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
UtA2		05 March 2015 South	• No BC done. • No DPM done – more leaf litter present than herbaceous biomass, resulting in over estimation of biomass.
UtB1		06 March 2015 South	• No BC done. • No DPM done.

Site	Cardinal photograph	Date & direction of tape	Comments
UtB2		31 March 2015 South	• None.
UtB3		11 March 2015 South	• T2 of APCQ was done to the left due to drainage line.

Site	Cardinal photograph	Date & direction of tape	Comments
UtB4		30 March 2015 Southwest	• No DPM done – more leaf litter present than herbaceous biomass, resulting in over estimation of biomass.
UtB5		31 March 2015 Southwest	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
WelA1		16 March 2015 East	• Heavy presence of brush packing. • No BC done – too encroached and too much brush packing for accurate distribution of points within 20 x 20m quad.
WelA2		17 March 2015 Southeast	• Moderate brush packing present.

Site	Cardinal photograph	Date & direction of tape	Comments
WelB1		12 March 2015 Southeast	• Minimal brush packing present.
WelB2		13 March 2015 East	• Moderate brush packing present.

Site	Cardinal photograph	Date & direction of tape	Comments
WelD1		07 May 2015 Northwest	• Minimal brush packing present.
WelD2		06 May 2015 Northeast	• Site heavily encroached with *D. cinerea*.

Site	Cardinal photograph	Date & direction of tape	Comments
WelE1		19 March 2015 South	• Minimal brush packing present.
WelE3		20 March 2015 Southeast	• No brush packing.

Site	Cardinal photograph	Date & direction of tape	Comments
WelE4	No photographs taken.	Southwest	• None.
FMS19		10 March 2015 Northeast	• None.
Man2		25 May 2015 Northwest	• Heavy presence of pepper ticks.

Site	Cardinal photograph	Date & direction of tape	Comments
Man3		28 April 2015 Southwest	• Lots of moribund material present (mostly *D. eriantha* and *P. maximum*) – this added to time as this material needed to be cleared at the relevant sampling points in order to determine distance-to-tuft measurements of the closest rooted grass species.
Man5		09 June 2015 South	• None.

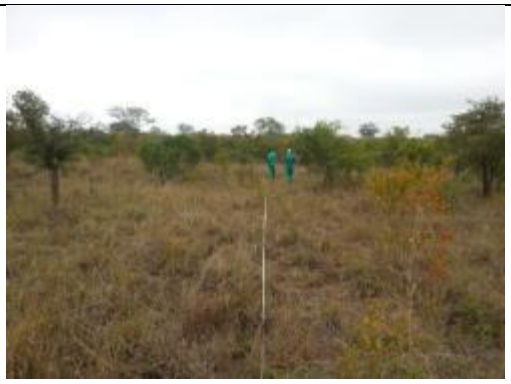
Site	Cardinal photograph	Date & direction of tape	Comments
Man6		28 May 2015 Northwest	• None.
Man7		21 May 2015 West	• None.
Man8		27 May 2015 Northeast	• Tape for MIM and BC methods ran to the left due to the road.

Site	Cardinal photograph	Date & direction of tape	Comments
Man9		07 April 2015 North	• None.
Man10		22 May 2015 East	• None.
Man12		13 April 2015 South	• None.

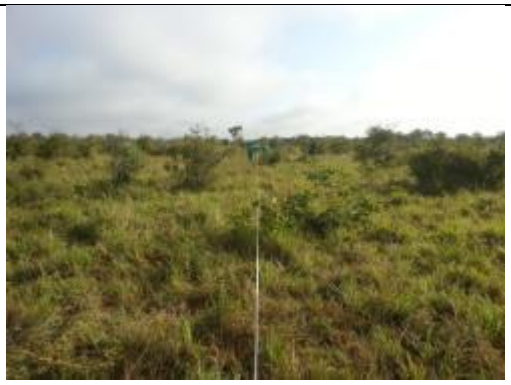
Site	Cardinal photograph	Date & direction of tape	Comments
Man19		11 June 2015 North	• None.
Man21		10 June 2015 Southwest	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
Man23		10 April 2015 Northeast	• None.
Man25		08 April 2015 Southwest	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
Man27		21 April 2015 East	• None.
Man33		14 April 2015 West	• None.
Man35		22 April 2015 South	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
Man40		20 April 2015 Southwest	• None.
FMS1		23 April 2015 South	• None.
FMS2		20 May 2015 Southeast	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
FMS3		30 April 2015 South	• None.
FMS4		15 April 2015 North	• None.
FMS5		18 May 2015 South	• None.

Site	Cardinal photograph	Date & direction of tape	Comments
FMS6		19 May 2015 West	• None.
FMS8		28 April 2015 East	• None.
FMS9		29 April 2015 North	• Lots of moribund material to clear in order to determine distance-to-tuft measurements of closest rooted grass species in the applicable

Site	Cardinal photograph	Date & direction of tape	Comments
			methods.

13. APPENDIX G: APPROVAL DOCUMENTS

UNIVERSITEIT VAN PRETORIA
UNIVERSITY OF PRETORIA
YUNIBESITHI YA PRETORIA

Animal Ethics Research

18-Feb-2015

Approval Certificate

New Application

Ethics Reference No.: V008-15

Title: An evaluation of techniques for monitoring rangeland health in semi-arid savannas in southern Africa.

Dear Mr Graeme Wolfaard

The **New Application** as supported by documents specified in your application or your research received, was approved by the Animal Ethics Committee on the .

Please note the following about your ethics approval:

- Ethics Approval is valid from *18-Feb-2015* to *17-Feb-2016*.
- Please remember to use your protocol number (V008-15) on any documents or correspondence with the Animal Ethics Committee regarding your research.
- Please note that the Animal Ethics Committee may ask further questions, seek additional information, require further modification, or monitor the conduct of your research.

Ethics approval is subject to the following:

- The ethics approval is conditional on the receipt of an annual (after 12 months) written Progress Reports, and
- The ethics approval is conditional on the research being conducted as stipulated by the details of all documents submitted to the Committee. In the event that a further need arises to change who the investigators are, the methods or any other aspect, such changes must be submitted as an Amendment for approval by the Committee.

We wish you the best with your research.

Yours sincerely

With regards

Dr Daan Verwoerd

Chair Animals Ethics Committee, University of Pretoria

UNIVERSITY OF PRETORIA
FACULTY OF VETERINARY SCIENCE

RESEARCH COMMITTEE

PROTOCOL COVER PAGE 2015

PROJECT (please type)		
Title	An evaluation of techniques for monitoring rangeland health in semi-arid savannas in southern Africa	NUMBER

APPLICANTS	NAME	QUALIFICATIONS	EMAIL ADDRESS	SIGNATURE
RESEARCHER	G Wolfaard	BSc (Hons) Wildlife Management	graeme.wolfaard@outlook.com	
SUPERVISOR	J van Rooyen	MSc(Agric) Animal Science, PhD cand	Jacques.vanrooyen@up.ac.za	
CO-SUPERVISOR	M Peel	PhD Wildlife Management – Rangeland Science	MikeP@arc.agric.za	

TYPE OF RESEARCH				
MSc X	MMedVet	PhD	Contract	Other

RESEARCH THEMES		Tick ✓
FOOD	Veterinary aspects of food safety and food security	
WEH	Wildlife and Environmental Health	X
MIPD	Molecular studies on infectious and parasitic diseases of animals	
PEVM	Phytomedicine and ethno-veterinary medicine	
ECA	Equine and companion animal health and welfare	
ANPH	Anatomical and physiological studies on animals	

PROPOSED REFEREES		
Name and Title	Postal Address	E-mail Address
1.		
2.		

FACILITY	
Name of Facility to be used	N / A

Signature of Head	

CONSENT TO PROVIDE SERVICES

Type of Service	Personnel	Signature
1.		
2.		
3.		
4.		

AUTHORISATION

NAME OF HOST DEPARTMENT

The Head of Department certifies that:

1. The design of the research is in line with accepted standards for this type of research
2. The necessary steps for quality control have been taken

SIGNED: HEAD OF DEPARTMENT	DATE 12/2/15
SIGNED: DEPARTMENTAL RESEARCH CO-ORDINATOR	DATE 12/2/2015

RESCOM APPROVAL

PROTOCOL APPROVED	
SIGNED: RESCOM CHAIRPERSON	DATE