People, parks and rangelands: an analysis of three-dimensional woody vegetation structure in a semi-arid savanna

Jolene Tichauer Fisher

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Declaration

I declare that this thesis is my own, unaided work, unless otherwise noted within the text. It is being submitted for the Degree of Doctor of Philosophy at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any other degree or examination in any other university.

Jolene Tichauer Fisher

29th day of July 2013 in Johannesburg

Abstract

Effective management of protected areas and communal rangelands, which are often juxtaposed in developing countries, is essential to prevent biodiversity decline and ensure a sustainable resource base for rural communities. However, in human-modified landscapes, there are complex interactions between factors that determine woody vegetation structural patterns. While the underlying biophysical template continues to influence vegetation patterns in a predictable manner; the intensity and type of disturbances that are the result of resource extraction, fire and herbivory can have an overriding impact. In order to effectively conserve biodiversity and plan for sustainable resource use, an understanding of land-use and land management is required. A case study of adjacent protected areas (Kruger National Park (KNP), a national protected area and Sabi Sand Wildtuin (SSW), a private game reserve) and communal rangelands (in Bushbuckridge Municipality (BBR) with varying intensities of use) in north-eastern South African savannas was used to study the spatio-temporal patterns of three-dimensional (3D) woody vegetation structure as a result of natural resource management and abiotic drivers.

The aim of this PhD thesis is to advance our understanding of the effects of management of natural resources on spatio-temporal patterns of 3D woody vegetation structure across land uses in a heterogeneous semi-arid savanna system. Vegetation structure was measured using small-footprint, discrete-return LiDAR (Light Detection and Ranging) collected by the CAO (Carnegie Airborne Observatory) Alpha System over 35 000 ha across the study area. 3D woody vegetation structure was compared both within land uses (KNP versus northern SSW, and within BBR) and between land-uses (southern SSW versus BBR) to address two objectives, namely 1. Can LiDAR be used as a monitoring tool for management of woody vegetation structure and biodiversity in semi-arid savannas and 2. What is the impact of land use and the corresponding management of resources on woody vegetation structure in semi-arid savannas?

Different land-use legacy timelines and current management objectives at sites in KNP and northern SSW has resulted in an average of 2.5 times higher vegetation density <3 m and >6 m in SSW. These differences in vegetation structure are exacerbated by current management practices, with implications for faunal biodiversity conservation across all scales. Not all reserves are equal in their ability to conserve biodiversity and such knowledge should be considered in conservation planning and management. In the communal rangelands, intense fuelwood harvesting has resulted in coppiced trees <3 m in height, and the only trees >5 m are preserved for cultural reasons, producing similar vegetation patterns to Sabi Sand Wildtuin. Disturbance (extraction and grazing) gradients occur with distance from settlements, with utilization intensity affecting vegetation cover within the size class distributions, but not the shape. Gradients diminish under heavier utilization resulting in a more structurally homogenous landscape, which may be used as an early warning sign of woodland degradation. The increase in >3 m tall trees was twice as high in low intensity use CRs adjacent to SSW compared to those in southern SSW from 2008 to 2010, indicating the impacts of treefall from megaherbivores and fire management reducing plant recruitment/regeneration in the protected area. Knowledge from investigation of socio-ecological drivers in the two land-uses were used to construct an ecologically relevant 3D woody vegetation structural classification which can be used by land managers to plan for sustainable resource use and effective conservation of biodiversity.

The management of natural resources, including direct use of fuelwood and the management of herbivory and fire affects woody structural dynamics; however, a lack of knowledge exists around the social and ecological context of natural resource management. The use of remote sensing, the knowledge of savanna ecology and an understanding of community-based natural resource management is integrated in this thesis to contribute to the context specific understanding of drivers of woody vegetation structure in two socio-ecological systems (protected areas and communal rangelands) which can be used in sustainable natural resource management plans.

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Disclaimer

This thesis consists of a series of chapters that have been prepared for submission to a range of scientific journals. As a result styles may vary between chapters in the thesis and overlap may occur to secure publishable entities. Author contributions are specified in Chapter 1.

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List of abbreviations

AACL	Altitude Above Closest Channel
ABRL	Altitude Below Ridge Line
AToMS	Airborne Taxonomic Mapping Systems
BBR	Bushbuckridge Municipality
CAO	Carnegie Airborne Observatory
CC	Canopy Cover
CHM	Canopy Height Model
CL	Canopy Layers
CLICK	Centre for LiDAR Information Coordination and Knowledge
CSIR	Council for Scientific and Industrial Research
DEM	Digital Elevation Model
DSM	Digital Surface Model
GIS	Geographic Information Systems
GLAS	Geosciences Laser Altimeter System
GLCC	Global Land Cover Classification
GPS-IMU	Geographic Positioning System-Inertial Measuring Unit
ICESat	Ice, Cloud and Land Elevation Satellite
IGBP	International Geosphere-Biosphere Project
K2C	Kruger to Canyons Biosphere Reserve
KNP	Kruger National Park
LCCS	Land Cover Classificaiton System
LiDAR	Light Detection and Ranging
MalaMala	MalaMala Private Game Reserve, a concession within Sabi Sands Wildtuin
NLC	National Land Classification
SAC	Satellite Applications Centre
SCC	Sub-Canopy Cover
SCD	Size Class Distribution
SDI	Simpson's Diversity Index
SES	Socio-ecological systems
SSW	Sabi Sand Wildtuin
USGS	United States Geological Survey
Voxel	Volumetric pixel

Chapter 1 : Woody vegetation structure in human-modified landscapes

1.1 Rationale

Savannas have been considered to be complex adaptive systems, and although this is widely accepted, it is far from being understood. Patterns reflect processes occurring at multiple spatio-temporal scales (Gillson 2004; Scholes & Walker 1993), with geology and climate being broad scale determinants of savanna structure and function, while fire, herbivory and people influence woody vegetation structural dynamics at fine scales (Gillson 2004; Sankaran et al. 2008). However, the interactions between these drivers are non-linear, confounding our ability to define cause and effect (Cumming 2011). The challenge of managing complex systems effectively and sustainably is in understanding them at scales relevant to the processes occurring.

Fire and herbivory are integral disturbances controlling the dynamic tree:grass relationship in savannas (Bond et al. 2003; Sankaran et al. 2005; Sankaran et al. 2008). However, since people have been living in savannas for millennia, their effect on, and management of, natural resources could also be regarded as a 'natural' disturbance. Not only do people interact with and affect the environment through natural resource harvesting in rural areas and communal rangelands, but they are also integral components of maintaining ecosystem function in protected areas through management of fire, decisions on stocking densities of animals, artificial water provision, tourism, fences and disease management (Freitag-Ronaldson & Foxcroft 2003). The relationship between people and savannas is as complex and adaptive as the savanna system itself. The human-environment association can be defined as a socio-ecological system (SES). Ostrom (2009) provides a useful framework with which to study SES, dividing the system into four units: resource system, resource units, users and governance system. Multiple second-level variables can affect the sustainability or resilience of the system such as the size of the resource system, predictability of the resource system, number of users, leadership, history of use, importance of resources to users and knowledge of SES among others (Fig. 1.1; Ostrom 2009).

SES refers to both rural areas where people rely primarily on the natural resource base for their income, and conservation areas which are often located in close proximity to rural areas. Therefore negative (overharvesting and land degradation) and positive (biodiversity corridor) attributes of rural areas can have 'spillover' effects into surrounding protected areas. In the Lowveld region of South Africa protected areas and communal rangelands occur adjacent to one another within the Kruger to Canyons (K2C) Biosphere Reserve (http://www.kruger2canyons.com/learningcentre/kruger_to_canyons_biosphere.php). Biosphere reserves address biodiversity conservation while at the same time providing ecosystem services to expanding populations (UNESCO 1996). Research within biosphere reserves is encouraged to increase our understanding of how people influence and interact with the environment (UNESCO 1996). A better understanding of the human dimension of global environmental change, in particular land-use and land management, is becoming increasingly important if we are to ensure long-term sustainability (Kangalawe 2009; Vitousek et al. 1997). Within K2C, the ideal opportunity is presented to study the effects of natural resources management and use on woody vegetation structure in two types of conservation areas (statutory national park and private game reserve) and adjacent communal rangelands.

The three study areas were initially chosen to study factors contributing to land degradation as part of a collaboration between The University of the Witwatersrand, Council for Scientific and Industrial Research (CSIR), South African National Parks (SANParks) and the Carnegie Airborne Observatory (CAO) of Carnegie Institution for Science. The areas also provide a natural experiment to study the effects of land management as well as the underlying abiotic template on biodiversity. The collaboration was initiated by the CAO who launched CAO-Alpha in 2007, a remote sensing system providing in-flight fusion of LiDAR (Light Detection and Ranging) and hyperspectral imagery (Asner et al. 2007). The CAO's mission is to use these remote sensing products to "understand how changes in land-use, climate and natural disturbances affect the structure, composition and functioning of ecosystems, and how these changes alter services provided by ecosystems to people" (http://cao.stanford.edu/). This study forms part of a larger project ultimately aimed at measuring, monitoring and mapping species composition, vegetation structure and finally function in this South African semi-arid

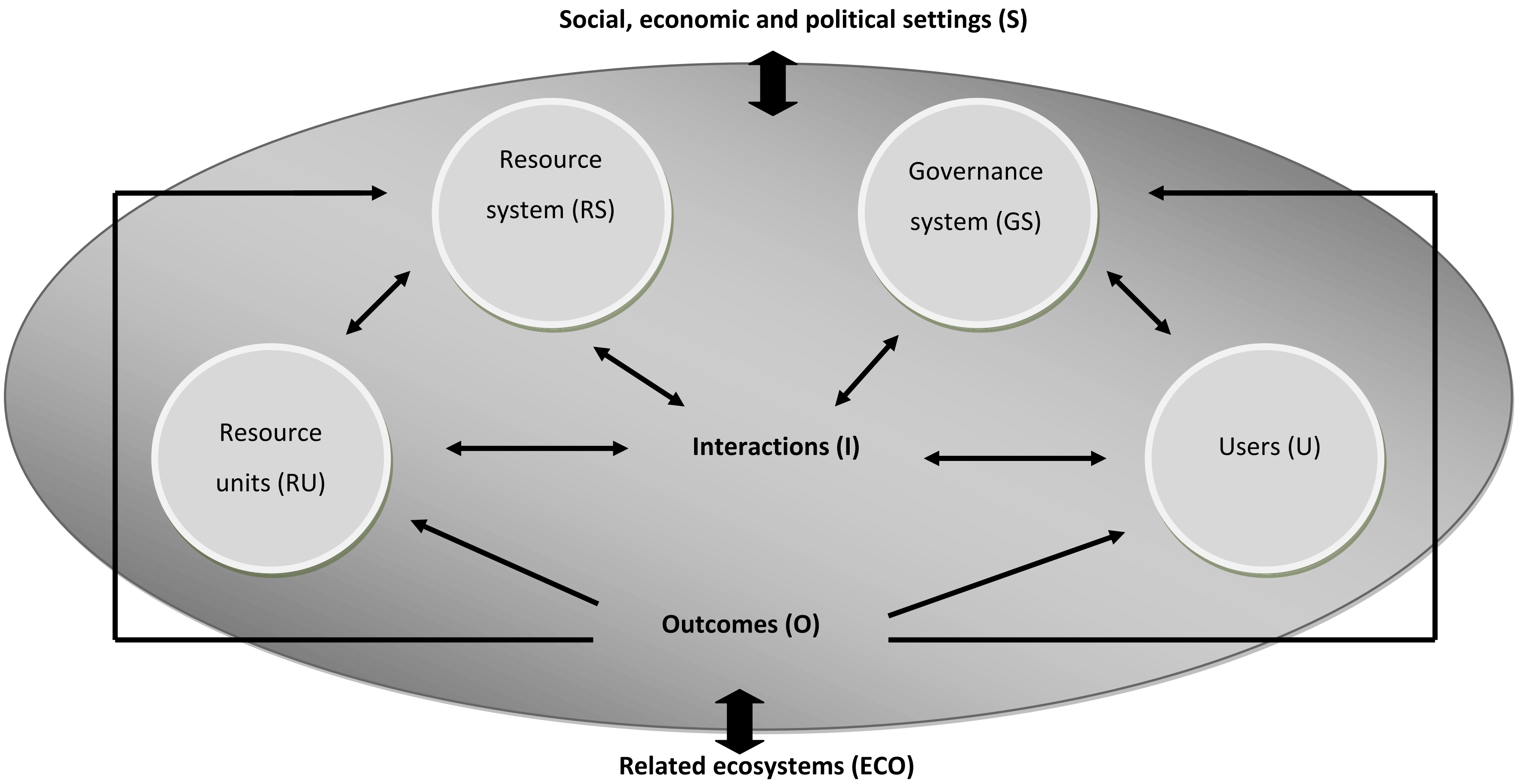

Figure 1.1: The core subsystems in a framework for analysing socio-ecological systems (Ostrom 2009).

savanna. With global change a reality, the results from the collaboration will aid in effective conservation planning and sustainable use of resources in the K2C biosphere reserve.

Woody vegetation structure, both at a broad scale (patches in the landscape) and a fine scale (patch and tree architecture), defines savannas. Recent remote sensing technologies such as LiDAR are making it possible to study woody vegetation structure at these fine as well as broad scales. LiDAR provides a three-dimensional (3D) image of woody vegetation, making it possible to study the effect of land use on biodiversity through quantifying 3D woody vegetation structure and structural patterns across the landscape at scales relevant to management (Lefsky et al. 2002a; Turner et al. 2003). This thesis uses LiDAR to measure woody vegetation structure across land use types and intensities to separate the effects of land use and management of natural resources from the abiotic template. Furthermore, the advantages and disadvantages of using LiDAR as a monitoring tool for management of woody structural diversity in savannas are assessed.

1.2 Background literature

1.2.1 Drivers of savanna structure and pattern

Situated in Limpopo and Mpumalanga Provinces in north-eastern South Africa, the lowveld (low altitude) savannas occur on a mixture of geologies, but predominantly basalt and granite. Geology influences vegetation structure and composition at a broad scale (Fig. 1.2). Soils are a product of the parent material, and soil type will also affect water flow through the area, thus controlling which types of vegetation are able to withstand the prevailing conditions. On the basalt soils, which usually have a higher clay content, fine, compound-leaved, thorny vegetation is common; while broad-leaved vegetation that employs chemical defence over physical defences are found on the infertile granitic soils (Scholes & Walker 1993). At a finer scale, vegetation structure is known to shift across hill-slope/catena position as a result of changes in soil and hydrological conditions (Venter et al. 2003). The broad differences in vegetation structure observed between basalt and granite substrates are reflected across the catena, with broad-leaf species such as those belonging to the Combretaceae family (e.g. species in the genera *Combretum* and *Terminalia*) and the Caesalpinioideae subfamily of Fabaceae (e.g. *Bauhinia, Colophospermum, Cassia* and *Peltophorum*) occurring on the less fertile, well drained, sandy soils on the crest and fine-leaf vegetation, including species from the subfamily Mimosideae (Fabaceae, e.g. *Acacia* and *Dichrostachys*) on the more fertile, alluvial rich, soils towards the valley (Mueller-Dombois & Ellenberg 1974). Rainfall, another key determinant of savanna structure, interacts with geology/soils by influencing moisture and nutrient availability. For example, high rainfall on sandy soils will increase the leaching of nutrients, creating infertile soils on which broad-leaf vegetation will occur. Higher soil moisture content will also allow for a greater density of woody plants to be supported by the land. In addition to the bottom-up effects of rainfall determining the savanna structure, woody plant species have also adopted different life-history traits in order to deal with intense rainfall and periods of drought (Coughenour & Ellis 1993; Scholes & Walker 1993). While geology, topography and rainfall are primary determinants of vegetation structure controlling the distribution of functional types (broad and fine leaf vegetation), fire, herbivory and land-use/management affects fine scale properties of vegetation structure such as canopy cover, connectivity, population structure and morphology (Coughenour & Ellis 1993).

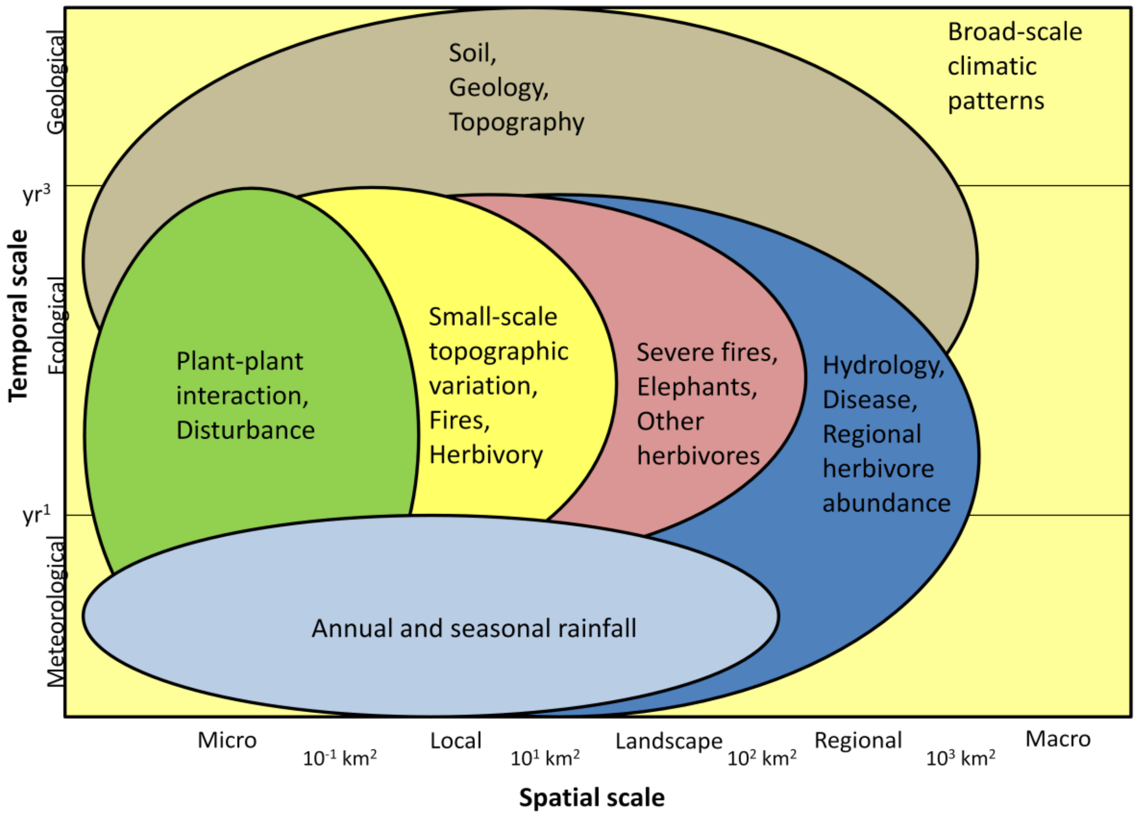

Figure 1.2: A spatio-temporal framework for the scale of processes influencing tree abundance in savannas (Gillson 2004).

1.2.2 Fire and herbivory as disturbances in savannas

The dynamic tree: grass distribution in savannas is, albeit not exclusively, spatially and temporally governed by disturbances such as fire, herbivory and people (Sankaran et al. 2008; Scholes & Archer 1997; Scholes & Walker 1993). Frost has only recently been recognised as a disturbance in savannas, impacting *Colophospermum mopane* trees <4 m which do not regain their former canopy height if 100% freeze-damage occurred in the previous season (Whitecross et al. 2013). The intensity and frequency of fires vary with soil fertility, rainfall and herbivory; which affect fuel load and are also dependant on prevailing weather conditions (Van Wilgen et al. 2008; Archibald et al. 2009). The role of fire in maintaining the structure of savannas, both at a fine scale (tree: grass interactions) and a landscape scale (patch heterogeneity), is still not clearly understood (Van Wilgen et al. 2008). At high fire frequencies (e.g. tri-annual), fire can lead to an increase in woody biomass by removing the herbaceous layer and thus competition, as well as resulting in less intense fires as a result of the fuel load reduction (Smit et al. 2010). A similar scenario can be created at low fire frequencies (e.g. annual burns) as more woody plants are able to escape the fire trap thereby increasing woody biomass (Smit et al. 2010; van Wilgen et al. 2003). While fire frequency does play a role in how fires affect vegetation structure, the underlying physical template (geological and topographical) affects how the vegetation will respond (Levick et al. 2012). The effect of fire as a disturbance in savannas has been likened to that of herbivory, with fire being likened to a herbivore unconstrained by food quality (Bond & Keeley 2005). However, long term studies have shown the effects of fire to be height specific (Levick et al. 2009; Smit et al. 2010), with vegetation height showing a more heterogeneous response than canopy cover to fire frequency (Levick et al. 2012).

Herbivory affects fires by reducing fuel loads in heavily grazed areas, but also alters savanna dynamics in its own right (Scholes & Walker 1993). Grazers reduce herbaceous biomass and hence competition with woody plants, but at the same time browsers can maintain woody vegetation in a ‘herbivore trap’ which limits vertical growth of trees and can lead to increased bushiness (Neke 2005; Owen-Smith 1988; Witkowski & O'Connor 1996). However, studies of vegetation structure in long term herbivore exclusion sites in Kruger National Park (KNP) have identified vegetation cover is higher in the treatments excluding herbivores (Asner et al. 2009; Levick et al. 2009). Smaller herbivores, such as goats and impala, can impact heavily on tree seedling recruitment (Prins & van der Jeugd 1993); and high impala utilisation of seedlings or rodent seed predation can result in an adult dominated population with no juveniles or seedlings (Helm & Witkowski 2012a). Megaherbivores such as elephant (*Loxodonta africana*) and giraffe (*Giraffa camelopardalis*) browse from tall vegetation up to 5.5 m in height. Elephants also act as ecosystem engineers in African savannas by felling or ringbarking trees (Whyte et al. 2003; Asner & Levick 2012; Helm & Witkowski 2012a), although they are selective agents of disturbance targeting preferred species such as marula (*Sclerocarya birrea*) (Van De Vijver et al. 1999; Helm & Witkowski 2012b). Helm et al. (2009) investigated the mortality and utilization of marulas in the Kruger National Park (KNP) and found that the prevalent fire regime in the area was responsible for a lack of recruitment into the adult size classes. Additionally, elephants have been found to be responsible for the majority of mortalities of the adult trees in Kruger National Park (Asner & Levick 2012), up to 25% of the population from 2001 to 2010 (Helm & Witkowski 2012b).

1.2.3 People in savannas – management and disturbance

People have been living in African savannas for at least 250 000 years, shaping patterns and processes through resource utilization and land management (Freitag-Ronaldson & Foxcroft 2003; Scholes & Walker 1993). Human impact on landscapes is often only considered in areas outside of reserves, but even in protected areas humans influence savanna dynamics by altering fire frequencies, introducing and removing animals and especially in smaller private reserves, bush clearing may be practiced.

Fire regimes in protected areas have evolved since the 1900's ranging from block burning, to only allowing lightening induced fires, to supplementing natural fires, to burn an annual target area in KNP (van Wilgen et al. 2008). In private reserves burning may involve controlled annual burns or controlling natural fires but the decision is usually left to the land owner/manager (http:\\www.malamala.com\conservation.htm, accessed January 2013). In a large reserve such as KNP animal populations are not intensively managed (culling of elephant was discontinued in 1995 (van Aarde et al. 1999)); however, in private, or at least smaller, reserves stocking densities of animals might be managed either for tourism reasons (improve game viewing) or ecological reasons (reduce impact on vegetation). Some conservation areas are exposed to greater management such as bush clearing and the introduction of artificial water points, which changes animals' usage of the landscape and surrounding vegetation (Parker & Witkowski 1999). In a perfect world we would be able to leave natural areas ungoverned, but with only 12% of the earth's surface formally protected (Chape et al. 2005; WWF 2006), people need to manage these ecosystems to ensure maximum biodiversity is conserved. Many reserves are managed to protect key species that are threatened with extinction (Mills et al. 1993; WWF 2006) but there is a shift towards protecting ecosystems as arenas for biodiversity rather than the conventional species-centric approach, especially in the face of climate change (Beier & Brost 2010; Malcolm et al. 2002).

One approach to conserving landscapes rather than species is the biosphere reserves initiative formulated by UNESCO as part of the Man and the Biosphere Programme in 1995 (UNESCO 1996). Biosphere reserves are intended to reconcile conservation, economic and social development while maintaining cultural values. Central to each biosphere reserve are core zones (secure protected areas), a buffer zone around the core zones (used for ecotourism, learning and research) and a transition zone occupied by settlements and agriculture (UNESCO 1996). Kruger National Park, Sabi Sand Wildtuin and the communal rangelands of Bushbuckridge form part of one such reserve in South Africa, the Kruger to Canyons (K2C) Biosphere Reserve (Coetzer et al. 2010). KNP and SSW fall into the core zone of the biosphere reserve, and the communal rangelands are part of the transition zone. Human activities in the rangelands such as cattle grazing, resource extraction, agriculture, the introduction of invasive species among other activities are at odds with conservation, affecting protected areas both directly and indirectly (Defries et al. 2005; Hansen & Defries 2007). However, they are also a reality of the 21st century and expanding human populations. The value of these communal rangelands within K2C is in the form of areas of intact woodland which act as corridors between core conservation zones (Chazdon et al. 2009); but unsustainable resource extraction within these areas is cause for concern.

Banks et al. (1996) predicted complete woodland deforestation around a rural settlement in Bushbuckridge within 15 years. A follow on study in 2009 (17 years after the initial data collection and prediction) (Matsika et al. 2012) showed this has not occurred; however, people have begun harvesting from an adjacent communal rangeland and buy fuelwood to supplement their needs. Reliance on natural resources, especially fuelwood, is high in rural areas (Kirkland & Hunter 2007). Although people realise the negative effect of their harvesting on the natural resource base (Shackleton et al. 2007), the direct use value of fuelwood alone was 44% of the total gross direct-use value of all resources per household per year (Dovie et al. 2002). Natural resources provide a safety net for people living in rural areas, where work is hard to obtain and often poorly paid, human population densities are high (Shackleton & Shackleton 2004) and HIV/AIDS is prevalent, often affecting the breadwinner (Hunter et al. 2011).

The increased demand for fuelwood and timber now means that historic cultural values which prevented live wood harvesting of culturally important, medicinal and fruiting trees are being outweighed (Higgins et al. 1999). In particular limbs of marula trees are being harvested (personal observation), altering the structure of these ecologically important large trees, and harvesting of trees <3 m in height such as young silver cluster leaf (*Terminalia sericea*) is resulting in thickening of bush due to coppicing. Coppicing occurs when a woody plant is damaged and resprouts. Often strong resprouters are poor recruiters favouring rebuilding biomass to producing seed. This results in persistence rather than replacement. The size class distribution thus contains few reproductive adults and subsequently fewer juveniles, affecting local and landscape scale structural diversity (Bond & Midgley 2001; Neke 2005). Structural diversity is further affected by a harvesting gradient. Shackleton et al. (1994) showed a disturbance gradient radiating from four villages in Bushbuckridge with high disturbance close to the village where it is easier to harvest fuelwood and disturbance decreasing further away. People also show preferences for certain height classes and species, affecting community structure (Shackleton et al. 1994; Neke 2005).

1.2.4 The importance of structural diversity in savannas

Biodiversity has been recognised to comprise of the structure, composition and function of living organisms within a system (Noss 1990). The majority of biodiversity research has focussed on compositional diversity, yet structural diversity is what gives rise to landscape heterogeneity. Woody vegetation is an integral component of savannas, a system which is defined by the mixture of trees and grasses (Scholes & Walker 1993). Vegetation structure is the spatial and temporal organisation of quantity, extent, type, connectivity and regeneration of the aboveground components of vegetation (Lefsky et al. 2002a). At a fine scale, structural heterogeneity is defined by the distribution of plants, population structure (age and sex ratios) and morphological variability, while at a landscape scale the connectivity, patch size and shape determine heterogeneity (Noss 1990). Structural diversity at all levels of organisation has functional implications for fauna and flora, as well as ecological processes, affecting primary productivity, shade, water flow, local nutrient concentrations, habitat niches and regeneration nuclei (Manning et al. 2006).

Large trees are especially important in a savanna landscape, providing corridors for fauna such as birds (Fischer & Lindenmeyer 2007; Smart et al. 2012), and form the patches that define savannas (Manning et al. 2006). At the local scale they provide shade, decrease evapo-transpiration of the below-canopy herbaceous layer and increase local nutrients which accumulate close to the root system (Belsky et al. 1993; Belsky 1994; Manning et al. 2006; Treydte et al. 2009). Increased nutrients also promotes higher ungulate densities creating a positive feedback of nutrients being returned to the soils in the area, as well as being positive for game viewing in reserves, enhancing ecotourism. While large trees are of utmost importance in savannas, small trees and shrubs have a great influence on landscape pattern and process as well. An increasing high density of this bushy layer, termed as bush encroachment, is commonly caused by prolonged heavy grazing and/or absence of fire which allows increased recruitment by encroaching species. This vegetation state results in a decline in palatable grass species (Oba et al. 2000) as well as prevents large ungulates from using the space due to decreased predator visibility (Ripple & Beschta 2004). In human-modified landscapes excessive harvesting of wood and subsequent coppicing can also lead to bush encroachment. Coppicing is a natural state for savanna trees, however, excessive harvesting leads to a skewed population structure with adult trees prevented from becoming reproductively active. This state leads to fewer recruits, thus reducing long-term resilience of the population (Lykke 1998). Fine-scale spatial niches created by complex vertical architecture provide additional niches for smaller fauna such as bats, spiders, arthropods and reptiles (Lumsden & Bennett 2005; MacArthur & MacArthur 1961; Means et al. 1999), as well as increasing vertical heterogeneity. Vertical heterogeneity is often overlooked as it is difficult to measure over large extents, although its importance in savanna functioning is widely recognised (Ishii et al. 2004).

1.2.5 Methods for measuring woody vegetation structure

In order to measure biodiversity, or the chosen aspect of biodiversity, in this case structural diversity, three steps should be taken. Firstly, the aspect or entity to be measured needs to be defined in as quantifiable a way as possible. Secondly, it needs to be quantified in a statistically reliable number of cases, and finally relationships in a set of indicators should be tested (Duelli & Obrist 2003). Traditionally plant ecologists have used field-based methods to measure vegetation; however, due to cost and time restraints, it is not possible to measure all vegetation over large areas (e.g. >5 ha), rather samples are measured (Mueller-Dombois & Ellenberg 1974). Vegetation is sampled within plots or transects; the arrangement, number and location of which can be chosen in a variety of ways either subjectively or objectively (Goldsmith & Harrison, 1976). Studies within the Kruger to Canyons Biosphere Reserve, specifically in Bushbuckridge municipality, have successfully used field-based methods to establish the effects of land-use (Higgins et al. 1999;

Shackleton 2000; Shackleton et al. 1994), and wood harvesting (Neke et al. 2006; Matsika et al. 2012) on vegetation structure, but in order to capture the heterogeneity inherent in savannas (Scholes & Archer 1997) alternative methods to measure vegetation structure at a broad landscape scale are required. One such method is remote sensing, which is the process of obtaining information about an object without being in physical contact with it (Lillesand et al. 2004).

Remote sensing techniques have proven useful for monitoring biodiversity at various spatial scales (Muchoney 2008; Turner et al. 2003), both directly (quantifying individual organisms, species assemblages or communities) and indirectly using environmental parameters as proxies. Mapping vegetation types (Fuller et al. 1998) and land use and land cover (Di Gregorio & Jansen 2000; Coetzer et al. 2010; Fensham et al. 2005; Fuller et al. 1998; Giannecchini et al. 2007) are common broad scale applications using multispectral imagery of 5-30 m spatial resolution. Land cover classifications such as the Land Cover Classification System (LCCS; Di Gregorio & Jansen 2000) and the National Land Cover classification of South Africa (NLC; Thompson 1996) define vegetation classes based on woody cover. Often the classifications are for a wide range of land covers at a global or regional scale; therefore, the cover classes need to be broad to encompass all possible land cover types. In contrast, when a classification is specific to mapping perturbations within one land cover class such as savannas, these broad percent cover classes may not adequately represent the situation on the ground. For example, a detailed bottom-up assessment of the classifier rule set with *in-situ* field validation is necessary when applying LCCS to southern African savannas (Hüttich et al. 2011). Except for Edwards' (1983) structural classification, the LCCS and the National Vegetation Classification System (NVCS) for North America (FDGC 1997), which are acknowledged for being structural classifications as opposed to a 2D land cover classification, many other classifications such as the NLC (Thompson 1996), the Global Land Cover Classification (GLCC; Hansen et al. 2000) and the International Geosphere-Biosphere Programme Data and Information System (IGBP; Loveland and Belward 1997) do not account for the shrub layer interspersed within the tree layer. In addition, the spatial arrangement of the woody layer as a whole, and of each cover type (tree/shrub), is not taken into consideration using these classifications. This is because vegetation structure is not easily mapped using standard 2D passive remote sensing products; however, it is possible.

Ingram et al. (2005) successfully mapped forest stem density and basal area by relating field measurements to NDVI (Normalized Difference Vegetation Index, calculated from Landsat Thematic Mapper at 30 m spatial resolution) and using artificial neural networks to predict values with r=0.69 and r=0.79 for stem density and basal area respectively. Maselli et al. (2005) used Landsat Enhanced Thematic Mapper Plus imagery to estimate basal area with relatively acceptable accuracy (Root Mean Square Error (RMSE) = 4.02 m^2/ha), however, the relationship between vegetation height and satellite measured spectral reflectance is non-linear, and thus conventional least squares regressions should not be used (Donoghue & Watt 2006). The relationship between spectral signature and height can be predicted more accurately in less dense areas (i.e. gaps of approximately 10 m between canopies) (Donoghue & Watt 2006), which limits the applicability of using passive remotely sensed data, to measure vegetation structure. Active remote sensing provides a useful alternative to passive remote sensing as the sensor emits energy and measures the return signal or signals. Active sensors, such as LiDAR (Light Detection and Ranging) or RADAR (Radio Detection and Ranging) are often used to map vegetation structure and create digital elevation models (DEM) based on the laser's ability to penetrate through vegetation (Wehr & Lohr 1999). Active remote sensing methods are more robust for measuring vegetation structure due to the linear relationship between LiDAR measured height and field-measured height (Donoghue & Watt 2006; Wessels et al. 2011).

1.2.6 LiDAR (Light Detection and Ranging): Remote measurements of 3D structure

LiDAR uses an active sensor emitting laser pulses to measure the distance between the sensor and the target. Distance is calculated by measuring the time elapsed between the emission of the laser pulse from the sensor and the return of the reflection of the pulse, dividing that time by two, and multiplying the figure by the speed of light (Wehr & Lohr 1999). The nature of this laser pulse allows for two different LiDAR systems, discrete return and waveform systems. Discrete return systems operate either on single- or multiple- return systems. Discrete laser pulses are emitted, and measure either one or multiple heights (depending on the system used) by identifying the peaks that represent discrete objects

(Fig. 1.3). Multiple-return systems are necessary for mapping vegetation structure and digital terrain models (if vegetation is present) as a single-return system would only measure the height of the first object the pulse comes into contact with (first return) and not penetrate through the canopy to the ground. The waveform system can distinguish the entire vegetation profile by recording the time varying intensity of laser return signal as a continuous signal and not as discrete returns (Fig. 1.3; Lefsky et al. 2002a). Because of the three-dimensional nature of LiDAR data it is possible to measure the three-dimensional attributes of vegetation structure, and depending on the extent of the data acquired, structure at various spatial scales can be reliably quantified.

LiDAR has primarily been applied to forestry environments to measure fuel loads, and forest structural attributes such as basal area, stand volume, mean tree height, canopy cover and biomass (Hudak et al. 2008; Lefsky et al. 1999; Lefsky et al. 2002b; Pascual et al. 2008; Skowronski et al. 2007). Certain structural attributes are not directly measurable with LiDAR but instead can be inferred using other measurements such as canopy height. Lefsky et al. (1999) found the quadratic mean canopy height was a better predictor of basal area and aboveground biomass than the maximum, median and mean canopy heights.

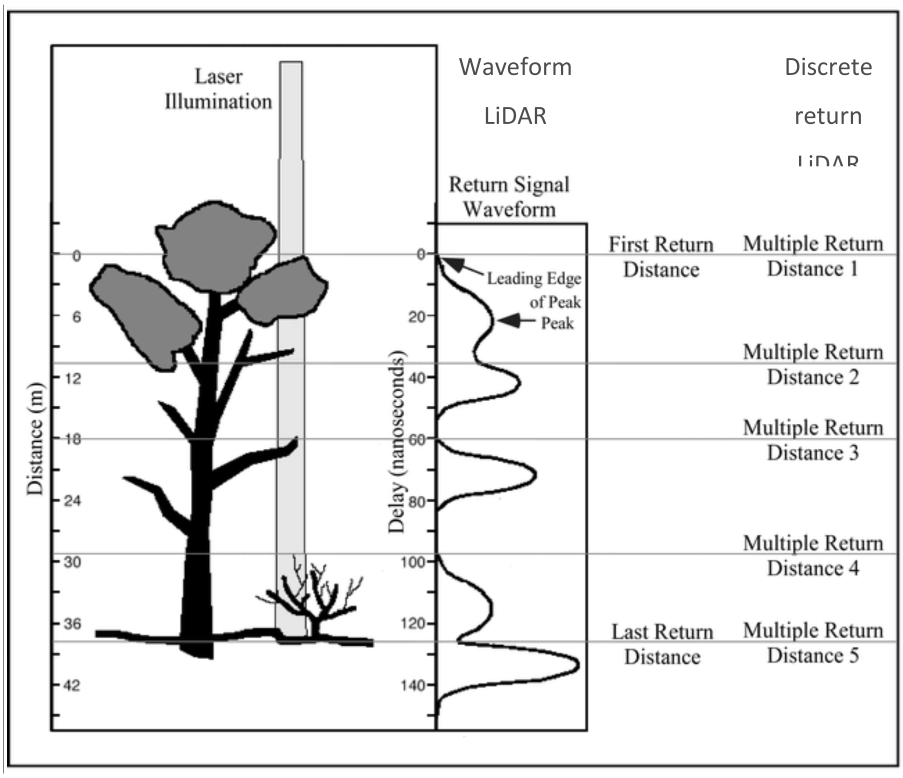

Figure 1.3: LiDAR measurements of vegetation canopies (after Lefsky et al. 2002a).

Tree height has often been underestimated in the forest environment for two reasons. Either dense forest-floor covering leads to an overestimation of the DEM, and thus an underestimation of the DSM (Digital Surface Model) (Clark et al. 2004; Lefsky et al. 1999; Lefsky et al. 2002b), or top-canopy LiDAR hits may not always penetrate the crown space. Used in a semi-arid sagebrush steppe environment, LiDAR underestimated vegetation height as well, though in this case it was possibly due to the first LiDAR return signal being reflected from within the shrub canopy rather than from the top of it (Streutker & Glenn 2006). A similar problem may occur in savannas, especially if the canopies are not dense, e.g. in autumn as the majority of trees are semi-deciduous or deciduous. The heterogeneous and often discontinuous canopy structure in savannas may present problems for measurement of woody structural attributes due to interpolation artefacts (Levick & Rogers 2011). Nevertheless, small-footprint discrete-return and waveform LiDAR present novel opportunities in savannas due to their higher resolutions which will improve the ability to measure sparse canopies.

Alternative uses of LiDAR data apart from commercial applications are for ecological applications such as characterising faunal habitats (e.g. Smart et al. 2012; Palminteri et al. 2012), carbon storage (e.g. Asner et al. 2012a) and identifying different plant species (Lefsky et al. 2002b). Avian species diversity is often highly correlated to vegetation structure. LiDAR derived canopy volume has been found to be a good predictor of bird density, whilst foliage height diversity is a good predictor of bird species richness (Clawges et al. 2008; Goetz et al. 2007). Estimating carbon stocks has become necessary because of the current rates of anthropogenic global change and the trading of carbon futures. LiDAR can be used to assess carbon storage in woody vegetation although field based carbon estimates are still required. Through destructive sampling and allometric equations (both species specific and general equations incorporating tree height and stem diameter) the above ground carbon content can be estimated for individual trees and stands of trees (Patenaude et al. 2004; Shackleton & Scholes 2011). Using the same allometric equations applied to LiDAR-derived canopy height models, the amount of carbon storage can be remotely determined (Asner et al. 2012a; Colgan et al. 2012; Patenaude et al. 2004). With regard to distinguishing between plant species, if large differences occur in their heights, LiDAR can be used to map species without the use of additional spectral information (Donoghue et al. 2007), but spectral information often improves the results (Anderson et al. 2008). Anderson et al. (2008) used 24 noise-limited hyperspectral bands from AVIRIS (Airborne Visible/Infrared Imaging Spectrometer) in conjunction with LiDAR data from NASA's Laser Vegetation Imaging Sensor (LVIS) and each separately to measure basal area, above-ground biomass and quadratic mean stem diameter of a northern temperate forest. A combination of the two sensors explained 8-9% more of the variation in estimates compared to when only one type of sensor was used.

1.2.7 CAO (Carnegie Airborne Observatory)

The Carnegie Airborne Observatory (CAO) was launched to study ecosystems around the world in order to understand how land use, climate change and natural disturbances affect the structure, function and composition of ecosystems. From 2007 the CAO operated the Alpha system (Table 1.1) which was decommissioned in 2011 and replaced by the AToMS (Airborne Taxonomic Mapping System; Asner et al. 2012b) platform. Up to 2013 the CAO has been operated in Colombia, Hawaii, Madagascar, Panama, Peru and South Africa (http://cao.stanford.edu). The CAO Alpha system was operated in South Africa in 2008, 2010 and CAO AToMs in 2012. CAO Alpha combines advanced integrated imaging spectroscopy (IS) and LiDAR remote sensing through in-flight fusion, as well as automated algorithms for precise co-location and geo-orthorectification (Asner et al. 2007). The CAO specifications can vary depending on the type of information required related to the spatial and spectral resolution, spectral range and laser spot spacing (Table 1.1). The LiDAR sensor is operated at 1064 nm, a wavelength where vegetation reflectance is high but this wavelength is absorbed by clouds. Flights are therefore restricted to cloud free days. When data from the CAO is processed a series of core and synthetic products can be derived, depending on both the scientific requirements of the project and the quality of published algorithms (Asner et al. 2007). Core products produced include upper-canopy pigment concentrations and indices, canopy water content and indices, canopy height and architecture, and ground topography. Synthetic products are aboveground biomass (carbon) stocks in vegetation, canopy light use efficiency, gross primary production (GPP) and species dominance and diversity (Asner et al. 2007). Most recently, the data generated by the CAO has been used successfully to measure the effect of elephant on treefall in South Africa (Asner & Levick 2012; Levick & Asner 2013), predict the spatial distribution of a forest primate in the Peruvian Amazon (Palminteri et al. 2012) and to map carbon stocks in the Colombian Amazon (Asner et al. 2012a) among other applications (see http://cao.stanford.edu/?page=publications&pag=0 for a comprehensive list of CAO publications).

Table 1.1: Requirements of Carnegie Airborne Observatory Alpha system imaging spectrometer and LiDAR for measurement and monitoring of ecosystem biochemistry, physiology and three-dimensional structure (after Asner et al. 2007).

Specification – CAO Alpha	**Requirement**	
	Minimum	**Optimal**
General		
Range of flying altitudes	500-3500 m	
Imaging spectrometer		
Spatial resolution	≤ 5 m	≤ 1 m
Spectral range	400-1050 nm	400-2500 nm
Spectral resolution	≤ 10 nm	≤ 5 nm
Signal to Noise (SNR) 400-1050 nm	≥ 500 @ 550 nm for live vegetation targets	
	≥ 400 @ 850 nm	
Signal to Noise (SNR) 1050-2500 nm	≥ 100 @ 2100 nm for live vegetation targets	
Spectral uniformity	≥ 95% cross-track	
LiDAR		
Laser pulse repetition frequency	≥ 50 kHz	≥ 100 kHz
Discrete laser return measurement mode	≥ 4 laser ranges/4 laser intensities	
Waveform laser return measurement mode	≥ 200 elevations per laser pulse	
Effective spatial resolution/laser spot spacing	≤ 2 m	≤ 1 m
Laser point distribution	Evenly spaced across swath, across- and down-track	

1.3 Research aims, objectives and key questions of the study

The aim of this PhD thesis is to advance our understanding of the effects of management of natural resources on spatio-temporal patterns of 3D woody vegetation structure across land uses in a heterogeneous semi-arid savanna system. Central to this aim is the development of methods and evaluating the use of LiDAR as a monitoring tool to aid management in semi-arid savannas. The broad objectives of the research were divided into 2 categories:

1.3.1 Objective 1: Assess if LiDAR can be used as a monitoring tool for management of woody vegetation structure and biodiversity in semi-arid savannas

- How does the type of data collected using LiDAR compare to field surveys and passive remote sensing methods in measuring woody structural diversity at fine scales and large extents in a semi-arid savanna (addressed in Chapter 2 and 3)?
- What are the advantages of a 2D over a 3D vegetation structural classification? (addressed in Chapter 4)
- Create an ecologically meaningful classification of savanna woody vegetation structure that can be used for both snapshot and change analyses (addressed in Chapter 4)

1.3.2 Objective 2: Investigate the impact of land use and the corresponding management of resources on woody vegetation structure in semi-arid savannas

- How does woody vegetation structure reflect differences in conservation management objectives, in a statutory national park and a private game reserve (addressed in Chapter 2)?
- How does woody vegetation structure change with increased distance from settlements (addressed in Chapter 3)?
- How is woody vegetation structure influenced by topographic position relative to distance from settlement in communal rangelands (addressed in Chapter 3)?
- How do geophysical factors and resource utilization in communal rangelands affect vegetation structure (addressed in Chapter 3)?
- How does human use and management of the landscape in a protected area and communal rangeland affect woody vegetation structural dynamics (addressed in Chapter 5)?

1.4 Structure of thesis

The chapters of this thesis, excluding the introductory and concluding chapters, have been written in a free-standing format for submission or publication to a scientific journal. While care has been taken to avoid overlap between the introduction and methods sections of each chapter, this was at times unavoidable, especially when describing the study region and LiDAR data collection and processing. The rationale for the study, aims and objectives as well as a general literature review are provided in Chapter 1. The results of the study are presented in chapters 2 - 6. Chapter 2 has been prepared and formatted for submission to *South African Journal of Botany*. Chapter 3 has been published in *Environmental Conservation* (Fisher et al. 2012) and Chapter 4 has been published in *Applied Vegetation Science*. The introduction of each chapter links back to the literature reviewed in Chapter 1.

Chapter 1 provides an introduction to the thesis, establishing the research aim and objectives as well as providing context within the international and local literature. In Chapter 2 management objectives in two protected areas were investigated taking into account the different land-use succession timelines and assess their effects on woody vegetation structure. In Chapter 3 the focus was on the effects of natural resource extraction on woody vegetation structure, investigating structural gradients around settlements and the contribution various factors (distance to rivers, roads, settlements, topography, slope and aspect) have on structural diversity. The background and lessons learnt from Chapters 2 and 3 allowed for the creation of an ecologically meaningful classification of savanna woody vegetation related to human use of the landscape and natural disturbances such as fire, herbivory and frost, presented in Chapter 4. In Chapter the structural classification is used for change detection of savanna woodland patterns and dynamics in one of the protected areas and the adjacent communal rangeland, drawing on the knowledge gained in Chapters 2 and 3. The findings of each chapter are then discussed within the broader context of the study in Chapter 6, addressing the implications of land management on vegetation structure for conservation planning and sustainable resource use.

Due to the collaborative nature of the project, I have written or contributed to the following additional publications:

Erasmus, B.F.N., Coetzer, K.L., Mambo, J., Archer, E. R.M., Fisher, J.T. and Asner, G.P. (2011) Environmental change in Bushbuckridge. In: Earth observations on environmental change in South Africa (eds. Pauw, J., Zietsman, H.L., van Jaarsveld, A.S. and Wessels, K.J.) pp 20-26

Fisher, JT, Erasmus, BFN, Witkowski, ETF, van Aardt, J, Asner, GP, Kennedy-Bowdoin, T, Knapp, DE, Mathieu, R and Wessels, K (2009). Three-dimensional woody vegetation structure across different land-use types and land-use intensities in a semi-arid savanna. *Proceedings of the International Geoscience and Remote Sensing Symposium*, IEEE Catalog number CFP09IGA-CDR.

Wessels, KJ, Mathieu, R, Erasmus, BFN, Asner, GP, Smit, IPJ, van Aardt, J, Main, R, Fisher, J, Marais, W, Kennedy-Bowdoin, T, Knapp, DE, Emerson, R and Jacobson, J (2011) Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* **261**: 19-29

Wessels, K.J., Colgan, M.S., Erasmus, B.F.N., Asner, G.P., Twine, W.C., Mathieu, R., van Aardt, J.A.N., Fisher, J.T and Smit, I.P.J. (2013) Unsustainable fuelwood extraction from South African Savannas. *Environmental Research Letters* **8**: 10pp. doi:10.1088/1748-9326/8/1/014007

1.4.1 Author contributions

Due to the collaborative nature of this study, the papers on which chapters 2, 3, 4 and 5 are based, have multiple co-authors. All authors and their contributions are the same for each chapter. The following list contains a description of contributions for each co-author:

- J.T. Fisher: Primary author, conceived and developed research questions, conducted all data extraction, analysis of data and write up.
- B.F.N. Erasmus: PhD Co-supervisor, provided guidance and input with regard to theoretical ideas for the paper and analysis of data and commented on various drafts
- E.T.F Witkowski: PhD Co-supervisor, provided guidance and input with regard to theoretical ideas for the paper and analysis of data and commented on various drafts
- Jan van Aardt: PhD Co-supervisor and commented on various drafts
- K.J. Wessels: PhD Co-supervisor and commented on various drafts
- G.P Asner: CAO collaborator, developed and implemented CAO, provided LiDAR data and commented on various drafts

- <u>R. Mathieu:</u> CSIR collaborator, commented on final drafts, provided funding for Chapter 2 and 3

1.5 Study area

The Bushbuckridge region (BBR) (communal rangelands), Sabi Sand Wildtuin (SSW) (private conservation area) and Kruger National Park (KNP) (state-owned conservation area) form a west to east land-use intensity gradient which is similarly characterised by a west to east gradient in climate, topography and land-use intensity. The southern part of KNP and the SSW fall into the north-eastern section of Mpumalanga, and the Bushbuckridge region is situated in the southernmost section of the Limpopo Province, South Africa (Fig. 1.4). Mean annual precipitation over the study area ranges from > 1 200 mm in the west, falling to an average of 550 mm in the east, with mean annual temperature of 22 ˚C. The geology of the region is dominated by granite, with Timbavati gabbro intrusions. All study sites fall within three vegetation groups of the savanna biome: granite lowveld (dominant), gabbro grassy bushveld and legogote sour bushveld (Mucina & Rutherford 2006). Typical vegetation species in the granite lowveld include: *Terminalia sericea*, *Combretum zeyheri* and *C. apiculatum* occurring on the deep sandy toplands, and *Acacia nigrescens*, *Dichrostachys cinerea* and *Grewia bicolor* growing in the more clay soils of the bottomlands. In the two other vegetation groups additional common species include *Sclerocarya birrea*, *Lannea schweinfurthii*, *Ziziphus mucronata*, *Dalbergia melanoxylon*, *Peltoforum africanum* and *Pterocarpus rotundifolius* (Mucina & Rutherford 2006).

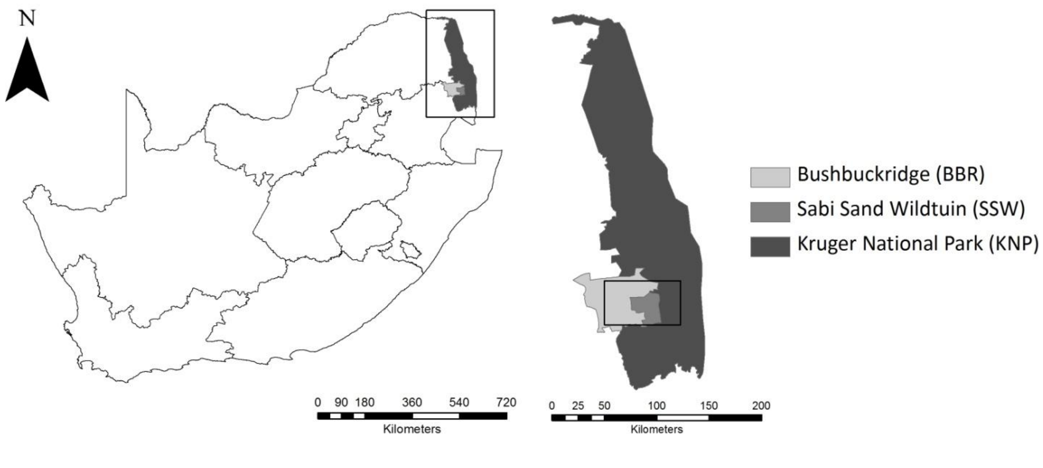

Figure 1.4: Study location – Bushbuckridge region, Sabi Sand Wildtuin and the southern section of Kruger National Park in the Limpopo and Mpumalanga Provinces, South Africa.

1.5.1 Bushbuckridge region (BBR)

Bushbuckridge (BBR) is a densely populated region as a result of its history. Two Apartheid homelands, Gazankulu and Lebowa (Thornton 2002), were formed with the Natives Land Act (No. 27) of 1913. Between 1972 and 2012 human population density increased in the area to 209 people / km^2 (Stats SA 2012), with resulting increase in land utilization intensity and economic impoverishment (Pollard et al. 2003). In 1994 the region was divided into Tribal Trust Lands and ruled by Tribal Authorities. Subsistence livelihoods are practiced, and land utilization tends to be higher closer to the villages (Shackleton et al. 1994). Historically, cultural values of the people in the area meant harvesting of medicinal, fruit and culturally important trees was prevented, however the demand for fuel wood and timber now overrides these values (Higgins et al. 1999).

1.5.2 Sabi Sand Wildtuin (SSW)

Sabi Sand Wildtuin (SSW) is 65 000 ha and was only formally proclaimed in 1965. From 1922 until 1934 the area now known as SSW was known as the Sabi Ranch, owned by the Transvaal Consolidated Lands (TCL), and was used for cattle farming. Additional areas in the current SSW were bought and used as game reserves around the same time, and in 1938 all cattle were shot as a result of a foot-and-mouth disease outbreak (Joubert 2007). Currently, SSW functions as a conservancy comprised of a mosaic of private ownership for personal use, and commercial ecotourism lodges. The mission statement of SSW tells how the reserve is “a contained proclaimed Protected Area in terms of the law with numerous inter-dependant owners with a vision to ensure the long-term preservation of the biodiversity of the region while cognisant that ecosystems are dynamic and ever changing. The SSW will endeavour to maintain the faunal and floral assemblages, ecological processes and cultural heritage characteristics representative of the area, and to foster co-operation with the bordering Kruger National Park in turn offering a long term benefit to the greater KNP region and society as a whole.” (J. Swart, pers. comm.).

One of the field sites falls within the southern portion of SSW, however the other site is in MalaMala, a commercial high-end ecotourism lodge which is recently separated region of SSW. The mission statement of MalaMala private game reserve is “through the sustainable and profitable management of the environment that forms MalaMala Game Reserve, and through practices of responsible and ethical tourism, this vast wilderness will remain untouched for the enjoyment of all inhabitants of this country (South Africa), and the planet." (www.malamala.com), showing a stronger tourism-based approach than SSW.

1.5.3 Kruger National Park (KNP)

Present day KNP was proclaimed a national protected area in 1926 with the passing of the National Parks Act. The original KNP spans an impressive 22 000 km^2, with an additional 10 000 km^2 increase from the recent establishment of the Greater Limpopo Transfrontier Park (Mabunda et al. 2003). Initial management of the reserve was aimed at the recovery of animal populations from the previous excessive meat and ivory hunting, and the Rinderpest pandemic. Current management has clear research and management objectives, intending "to maintain biodiversity in all its natural facets and fluxes and to provide human benefits in keeping with the mission of SANParks in a manner which detracts as little as possible from the wilderness qualities of the Kruger National Park" (Mabunda et al. 2003). Thresholds of Potential Concern (TPC's) form the backbone of KNP's Strategic Adaptive Management framework. Upper and lower limits of a continuum of change are defined for selected environmental variables, representing the amount of variation in attributes of the ecosystem that is acceptable to management and scientists (Biggs & Rogers 2003). In addition, several management objectives have been set including one on biodiversity. The biodiversity objective is to maintain biodiversity in all its natural facets and fluxes with regard to ecosystem functioning, and to take a pro-active role in legal and statutory issues affecting biodiversity. The balancing and people objectives comprise awareness about factors affecting people both inside and outside the park and understanding the socio-ecological system in order to maintain resilience in the area (www.sanparks.org).

1.6 References

Anderson, J.E., Plourde, L.C., Martin, M.E., Braswell, B.H., Smith, M-L., Dubayah, R.O., Hofton, M.A. and Blair, J.B. (2008) Integrating waveform lidar with hyperspectral imagery for inventory of a northern temperate forest. *Remote Sensing of Environment* **112**: 1856-1870

Archibald, S., Roy, D.P., van Wilgen, B.W. and Scholes, R.J. (2009) What limits fire? An examination of drivers of burnt area in Southern Africa. *Global Change Biology* **15**: 613-630

Asner, G.P., Knapp, D.E., Kennedy-Bowdoin, T., Jones, M.O., Martin, R.E., Boardman, J. and Field, C.B. (2007) Carnegie Airborne Observatory: in-flight fusion of hyperspectral imaging and waveform light detection and ranging (wLiDAR) for three-dimensional studies of ecosystems. *Journal of Applied Remote Sensing* **1, 013536:** pp 1-21

Asner, G.P, Levick, S.R., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R., Jacobson, J., Colgan, M.S. and Martin, R.E. (2009) Large-scale impacts of herbivores on the structural diversity of African savannas. *Proceedings of the National Academy of Science* **12**: 261-278

Asner, G.P. and Levick, S.R. (2012) Landscape-scale effects of herbivores on treefall in African savannas. *Ecology Letters* **15:** 1211-1217

Asner, G. P., Clark, J. K., Mascaro, J., Galindo García, G. A., Chadwick, K. D., Navarrete Encinales, D. A., Paez-Acosta, G., Cabrera Montenegro, E., Kennedy-Bowdoin, T., Duque, Á., Balaji, A., von Hildebrand, P., Maatoug, L., Phillips Bernal, J. F., Knapp, D. E., García Dávila, M. C., Jacobson, J., and Ordóñez, M. F. (2012a) High-resolution Mapping of Forest Carbon Stocks in the Colombian Amazon. *Biogeosciences* **9**: 2445-2479

Asner, G.P., Knapp, D.E., Boardman, J., Green, R.O., Kennedy-Bowdoin, T., Eastwood, M., Martin, R.E., Anderson, C. And Field, C.B. (2012b) Carnegie Airborne Observatory-2: Increasing science data dimensionality via high-fidelity multi-sensor fusion. *Remote Sensing of Environment* **124**: 454-465

Banks, D.I., Griffin, N.J., Shackleton, C.M., Shackleton, S.E. and Mavrandonis, J.M. (1996) Wood supply and demand around two rural settlements in a semi-arid savanna, South Colgan et al 2012Africa. *Biomass Bioenergy* **11**: 319-331

Beier, P. and Brost, B. (2010) Use of land facets to plan for climate change: Conserving the Arenas, not the Actors. *Conservation Biology* **24**: 701-710

Belsky, A.J., Mwonga, S.M., Amundson, R.G., Duxbury, J.M. and Ali, A.R. (1993) Comparative effects of isolated trees on their undercanopy environments in high- and low- rainfall savannas. *Journal of Applied Ecology* **30**: 143-155

Belsky, A.J. (1994) Influences of trees on savanna productivity: tests of shade, nutrients, and tree-grass competition. *Ecology* **75**: 922-932

Biggs, H.C. and Rogers, K.H. (2003) An adaptive system to link science, monitoring, and management in practice. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press**:** 59-80

Bond, W.J. and Midgley, J.J. (2001) Ecology of sprouting in woody plants: the persistence niche. *Trends in Ecology and Evolution* **16**: 45-51

Bond, W.J., Midgley, G.F. and Woodward, F.I. (2003) What controls South African vegetation – climate or fire? *South African Journal of Botany* **69**: 79-91

Bond, W.J. and Keeley, J.E. (2005) Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution* **20**: 387-394

Chape, S., Harrison, J., Spalding, M. and Lysenko, I. (2005) Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society Biological Sciences* **360**: 443-455

Chazdon, R.L., Harvey, C.A., Komar, O., Griffith, D.M., Ferguson, B.G., Martinez-Ramos, M., Morales, H., Nigh, R., Soto-Pinto, L., van Breugel, M. and Philpott, S.M. (2009) Beyond Reserves: A Research Agenda for Conserving Biodiversity in Human-modified Tropical Landscapes. *Biotropica* **41**: 142-153

Clark, M.L., Clark, D.B. and Roberts, D.A. (2004) Small-footprint lidar estimation of sub-canopy elevation and tree height in a tropical rain forest landscape. *Remote Sensing of Environment* **91**: 68-89

Clawges, R., Vierling, K., Vierling, L. and Rowell, E. (2008) The use of airborne lidar to assess avian species diversity, density, and occurrence in a pine/aspen forest. *Remote Sensing of Environment* **112**: 2064-2073

Coetzer, K.L., Erasmus, B.F.N., Witkowski, E.T.F. and Bachoo, A.K. (2010) Land cover change in the Kruger to Canyons Biosphere Reserve (1993-2006): A first step towards creating a conservation plan for the subregion. *South African Journal of Science* **106**: 26-35

Colgan, M.S., Asner, G.P., Levick, S.R., Martin, R.E. and Chadwick, O.A. (2012) Topo-edaphic controls over woody plant biomass in South African savannas. *Biogeosciences Discussions* **9**: 957-987

Coughenour, M.B. and Ellis, J.E. (1993) Landscape and climatic control of woody vegetation in a dry tropical ecosystem: Turkana District, Kenya. *Journal of Biogeography* **20**: 383-398

Cumming, G. S. (2011) Spatial resilience in socio-ecological systems (pp. 7-66). Springer, ISBN 978-94-007-0307-0 pp: 254

Defries, R, Hansen, A., Newton, A.C. and Hansen, M.C. (2005) Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecological Applications* **15**: 19-26

Di Gregorio, A. and Jansen, L.J.M. (2000) Land Cover Classification System (LCCS): Classification concepts and user manual LCCS version 1.0. GCP/RAF/287/ITA Africover, Food and Agriculture Organization of the United Nations Publishing Service, Rome (http://www.fao.org/docrep/003/x0596e/x0596e00.htm, accessed February 2013)

Donoghue, D.N.M. and Watt, P.J. (2006) Using LiDAR to compare forest height estimates from IKONOS and Landsat ETM+ data in Sitka spruce plantation forests. *International Journal of Remote Sensing* **27**: 2161-2175

Donoghue, D.N.M., Watt, P.J., Cox, N.J. and Wilson, J. (2007) Remote sensing of species mixtures in conifer plantations using LiDAR height and intensity data. *Remote Sensing of Environment* **110**: 509-522

Dovie, D.B.K., Shackleton, C.M. and Witkowski, E.T.F. (2002) Direct-use values of woodland resources consumed and traded in a South African village. *International Journal of Sustainable Development and World Ecology* **9**: 269-283

Duelli, P. and Obrist, M.K. (2003) Biodiversity indicators: the choice of values and measures. *Agriculture, Ecosystems and Environment* **98**: 87-98

Edwards, D. (1983) A broad-scale structural classification of vegetation for practical purposes. *Bothalia* **14**: 705-712

Fensham, R.J., Fairfax, R.J. and Archer, S.R. (2005) Rainfall, land use and woody vegetation cover change in semi-arid Australian savanna. *Journal of Ecology* **93**: 596-606

Fischer, J. and Lindenmayer, D.B. (2007) Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* **16**: 265-280

Fisher, J.T., Witkowski, E.T.F., Erasmus, B.F.N., van Aardt, J., Asner, G.P., Wessels, K.J. and Mathieu, R. (2012) Human-modified landscapes: patterns of fine-scale woody vegetation structure in communal savannah rangelands. *Environmental Conservation* **39**: 72-82

FGDC (Federal Geographic Data Committee, Vegetation Subcommittee) (1997) Vegetation Classification Standard. FGDC-STD-005

Freitag-Ronaldson, S. and Foxcroft, L.C. (2003) Anthropogenic influences at the ecosystem level. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press**:** 391-421

Fuller, R.M., Groom, G.B., Mugisha, S., Ipulet, P., Pomeroy, D., Katende, A., Bailey, R. and Ogutu-Ohwayo, R. (1998) The integration of field survey and remote sensing for biodiversity assessment: a case study in the tropical forests and wetlands of Sango Bay, Uganda. *Biological Conservation* **86**: 379-391

Giannecchini, M., Twine, W. and Vogel, C. (2007) Land-cover change and human-environment interactions in a rural cultural landscape in South Africa. *The Geographical Journal* **173**: 26-42

Gillson, L. (2004) Evidence of hierarchical patch dynamics in an east African savanna? *Landscape Ecology* **19**: 883-894

Goetz, S., Steinberg, D., Dubayah, R. and Blair, B. (2007) Laser remote sensing of canopy habitat heterogeneity as a predictor of bird species richness in an eastern temperate forest, USA. *Remote Sensing of Environment* **108**: 254-263

Goldsmith, F.B. and Harrison, C.M. (1976) Description and analysis of vegetation. *Methods in plant ecology*. Oxford, Blackwell Scientific Publications**:** 85-156

Hansen, M.C., DeFries, R.S., Townshend, J.R.G. and Sohlberg, R. (2000) Global land cover classification at 1 km spatial resolution using a classification tree approach. *International Journal of Remote Sensing* **21**: 1331-1364

Hansen, A.J. and Defries, R. (2007) Ecological mechanisms linking protected areas to surrounding lands. *Ecological Applications* **17**: 974-988

Helm, C.V., Witkowski, E.T.F., Kruger, L., Hofmeyr, M. and Owen-Smith, N. (2009) Mortality and utilization of *Sclerocarya birrea* subsp. *caffra* between 2001 and 2008 in the Kruger National Park, South Africa. *South African Journal of Botany* **75**: 475-484

Helm, C.V. and Witkowski, E.T.F. (2012a) Characterising wide spatial variation in population size structure of a key African savanna tree. *Forest Ecology and Management* **263**: 175-188

Helm, C.V. and Witkowski, E.T.F. (2012b) Continuing decline of a keystone tree species in the Kruger National Park, South Africa. *African Journal of Ecology* pp1-10, DOI: 10.1111/aje.12032

Higgins, S.I., Shackleton, C.M. and Robinson, E.R. (1999) Changes in woody community structure and composition under constrasting landuse systems in a semi-arid savanna, South Africa. *Journal of Biogeography* **26**: 619-627

Hudak, A.T., Crookston, N.L., Evans, J.S., Hall, D.E. and Falkowski, M.J. (2008) Nearest neighbor imputation of species-level, plot-scale forest structure attributes from LiDAR data. *Remote Sensing of Environment* **112**: 2232-2245

Hunter, L.M., Twine, W. And Johnson, A. (2011) Adult mortality and natural resource use in rural South Africa: Evidence from the Agincourt Health and Demographic Surveillance Site. *Society and Natural Resources* **24**: 256-275

Hüttich, C., Herold, M., Strohbach, B.J. and Dech, S. (2011) Integrating in-situ, Landsat, and MODIS data for mapping in southern African savannas: experiences of LCCS-based land-cover mapping in the Kalahari in Namibia. *Environmental Monitoring and Assessment* **176**: 531-547

Ingram, J.C., Dawson, T.P. and Whittaker, R.J. (2005) Mapping tropical forest structure in southeastern Madagascar using remote sensing and artificial neural networks. *Remote Sensing of Environment* **94**: 491-507

Ishii, H. T., Tanabe, S. and Hiura, T. (2004) Exploring the relationships among canopy structure, stand productivity, and biodiversity of temperate forest ecosystems. *Forest Science* **50**: 342–355

Joubert, S. (2007) *The Kruger National Park: A History, Volume 1*. Johannesburg, High Branching Pty(Ltd)

Kangalawe, R.Y.M. (2009) Ecosystems changes and implications on livelihoods of rural communities in Africa. *Science and Technology, African Journal of Ecology* **47**: 1-2

Kirkland, T. and Hunter, L.M. (2007) "The Bush is No More": Insights on Institutional Change and Natural Resource Availability in Rural South Africa. *Society and Natural Resources* **20**: 337-350

Lefsky, M.A., Cohen, W.B., Acker, S.A., Parker, G.G., Spies, T.A. and Harding, D. (1999) Lidar Remote Sensing of the Canopy Structure and Biophysical Properties of Douglas-Fir Western Hemlock Forests. *Remote Sensing of Environment* **70**: 339-361

Lefsky, M.A., Cohen, W.B., Parker, G.G. and Harding, D.J. (2002a) Lidar remote sensing for ecosystem studies. *BioScience* **52**: 19-30

Lefsky, M.A., Cohen, W.B., Harding, D.J., Parker, G.G., Acker, S.A. and Gower, S.T. (2002b) Lidar remote sensing of above-ground biomass in three biomes. *Global Ecology and Biogeography* **11**: 393-399

Levick, S.R., Asner, G.P., Kennedy-Bowdoin, T. and Knapp, D.E. (2009) The relative influence of fire and herbivory on savanna three-dimensional vegetation structure. *Biological Conservation* **142**: 1693-1700

Levick, S.R. and Roger, K.H. (2011) Context-dependant vegetation dynamics in an African savanna. *Landscape Ecology* **26**:515-528

Levick, S.R., Asner, G.P. and Smit, I.P.J. (2012) Spatial patterns in the effects of fire on savanna vegetation three-dimensional structure. *Ecological Applications* **22**: 2110-2121

Levick, S.R. and Asner, G.P. (2013) The rate and spatial pattern of treefall in a savanna landscape. *Biological Conservation* **157**: 121-127

Lillesand, T.M., Kiefer, R.W. and Chipman, J.W. (2004) *Remote sensing and image interpretation*. New York, John Wiley & Sons

Loveland, T.R. and Belward, A.S. (1997) The International Geosphere Biosphere Programme Data and Information System Global Land Cover Data Set (DISCover). *Acta Astronautica* **41**: 681-689

Lumsden, L.F. and Bennett, A.F. (2005) Scattered trees in rural landscapes: foraging habitat for insectivorous bats in south-eastern Australia. *Biological Conservation* **122**: 205-222

Lykke, A.M. (1998) Assessment of species composition change in savanna vegetation by means of woody plants' size class distributions and local information. *Biodiversity and Conservation* **7**: 1261-1275

Mabunda, D., Pienaar, D.J. and Verhoef, J. (2003) The Kruger National Park: A century of management and research. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press**:** 3-21

MacArthur, R.H. and MacArthur, J.W. (1961) On bird species diversity. *Ecology* **42**: 594-598

Malcolm, J.R., Liu, C., Miller, L.B., Allnutt, T. and Hansen, L. (2002) Habitats at Risk: Global warming and species loss in globally significant terrestrial ecosystems. WWF Climate Change Campaign, Gland, Switzerland, 39 pp

Manning, A.D., Fischer, J. and Lindenmayer, D.B. (2006) Scattered trees are keystone structures - Implications for conservation. *Biological Conservation* **132**: 311-321

Maselli, F., Chirici, G., Bottai, L., Corona, P. and Marchetti, M. (2005) Estimation of Mediterranean forest attributes by the application of k-NN procedures to multitemporal Landsat ETM+ images. *International Journal of Remote Sensing* **26**: 3781-3796

Matsika, R., Erasmus, B.F.N. and Twine, W. (2012b) A tale of two villages: assessing the dynamics of fuelwood supply in communal landscapes in South Africa. *Environmental Conservation.* DOI: http://dx.doi.org/10.1017/S0376892912000264

Means, J.E., Acker, S.A., Harding, D.J., Blair, J.B., Lefsky, M.A., Cohen, W.B., Harmon, M.E. and McKee, W.A. (1999) Use of Large-Footprint Scanning Airborne Lidar To Estimate Forest Stand Characteristics in the Western Cascades of Oregon. *Remote Sensing of Environment* **67**: 298-308

Mills, L.S., Soulé, M.E. and Doak, D.F. (1993) The keystone-species concept in ecology and conservation. *BioScience* **43**: 219-224

Muchoney, D.M. (2008) Earth observations for terrestrial biodiversity and ecosystems. *Remote Sensing of Environment* **112**: 1909-1911

Mucina, L. and Rutherford, M.C (Eds.) (2006) The vegetation of South Africa, Lesotho and Swaziland, Pretoria, *Strelitzia 19. South African National Biodiversity Institute*, pp. 439-539

Mueller-Dombois, D. and Ellenberg, H. (1974) *Aims and methods in vegetation ecology*. New York, Wiley and Sons

Neke, K. (2005) 'Woody plant allometry and demography in a semi-arid Africa savanna.' In The regeneration ecology of savanna woodlands in relation to human utilization. PhD thesis. University of the Witwatersrand, South Africa

Neke, K.S., Owen-Smith, N. and Witkowski, E.T.F. (2006) Comparative resprouting response of Savanna woody plant species following harvesting: the value of persistence. *Forest Ecology and Management* **232**: 114-123

Noss, R.F. (1990) Indicators for monitoring biodiversity: A hierarchical approach. *Conservation Biology* **4**: 355-364

Oba, G., Post, E., Syvertsen, P. O. and Stenseth, N. C. (2000) Bush cover and range condition assessments in relation to landscape and grazing in southern Ethiopia. *Landscape Ecology,* 15: 535–546

Ostrom, E. (2009) A general framework for analyzing sustainability of social-ecological systems. *Science* **325**: 419-422

Owen-Smith, N. (1988) *Megaherbivores: The influence of very large body size on ecology*, Cambridge University Press

Palminteri, S., Powell, G.V.N., Asner, G.P. and Peres, C.A. (2012) LiDAR measurements of canopy structure predict spatial distribution of a tropical mature forest primate. *Remote Sensing of Environment* **127**: 98-105

Parker, A.H. and Witkowski, E.T.F. (1999) Long-term impacts of abundant perennial water provision for game on herbaceous vegetation in a semi-arid African savanna woodland. *Journal of Arid Environments* **41**: 309-321

Pascual, C., Garcia-Abril, A., Garcia-Montero, L.G., Martin-Fernandez, S. and Cohen, W.B. (2008) Object-based semi-automatic approach for forest structure characterization using lidar data in heterogeneous *Pinus sylvestris* stands. *Forest Ecology and Management* **255**: 3677-3685

Patenaude, G., Hill, R.A., Milne, R., Gaveau, D.L.A., Briggs, B.B.J. and Dawson, T.P. (2004) Quantifying forest above ground carbon content using LiDAR remote sensing. *Remote Sensing of Environment* **93**: 368-380

Pollard, S., Shackleton, C. and Curruthers, J. (2003) Beyond the fence: people and the lowveld landscape. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*. Washington, DC, USA: Island Press: 422-446

Prins, H.H.T. and van der Jeugd, H.P. (1993) Herbivore population crashes and woodland structure in East Africa. *Journal of Ecology* **81**: 305-314

Ripple, W.J. and Beschta, R.L. (2004) Wolves and the ecology of fear: Can predation risk structure ecosystems? *BioScience* **54**: 755-766

Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux, J., Higgins, S.I., Le Roux, X., Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G., Caylor, K.K., Coughenour, M.B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C., Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K.L., Prins, H.H.T., Ringrose, S., Sea, W., Tews, J., Worden, J. and Zambatis, N. (2005) Determinants of woody cover in African Savannas. *Nature* **438**: 846–849

Sankaran, M., Ratnam, J. and Hanan, N. (2008) Woody cover in African savannas: the role of resources, fire and herbivory. *Global Ecology and Biogeography* **17**: 236-245

Scholes, R.J. and Walker, B.H. (1993) *An African savanna - Synthesis of the Nylsvley study*. Cambridge, Cambridge University Press

Scholes, R.J. and Archer, S.R. (1997) Tree-grass interactions in savannas. *Annual Review of ecology and Systematics* **28**: 517-544

Shackleton, C.M., Griffin, N.J., Banks, D.I., Mavrandonis, J.M. and Shackleton, S.E. (1994) Community structure and species composition along a disturbance gradient in a communally managed South African savanna. *Vegetatio* **115**: 157-167

Shackleton, C.M. (2000) Comparison of plant diversity in protected and communal lands in the Bushbuckridge lowveld savanna, South Africa. *Biological Conservation* **94**: 273-285

Shackleton, C. and Shackleton, S. (2004) The importance of non-timber forest products in rural livelihood security and as safety nets: a review of evidence from South Africa. *South African Journal of Science* **100**: 658-664

Shackleton, C.M., Buiten, E., Annecke, W., Banks, D., Bester, J., Everson, T., Fabricus, C., Ham, C., Kees, M., Modise, M., Phago, M., Prasad, G., Smit, W., Twine, W., Underwood, M., von Maltitz, G. and Wenzel, P. (2007) Exploring the options for fuelwood policies to support poverty alleviation policies: evolving dimensions in South Africa. *Forests, Trees and Livelihoods* **17**: 269-292

Shackleton, C.M. and Scholes, R.J. (2011) Above ground woody community attributes, biomass and carbon stocks along a rainfall gradient in the savannas of the central lowveld, South Africa. *South African Journal of Botany* **77**: 184-192

Skowronski, N., Clark, K., Nelson, R., Hom, J. and Patterson, M. (2007) Remotely sensed measurements of forest structure and fuel loads in the Pinelands of New Jersey. *Remote Sensing of Environment* **108**: 123-129

Smart, L.S., Swenson, J.J., Christensen, N.L. and Sexton, J.O. (2012) Three-dimensional characterization of pine forest type and red-cockaded woodpecker habitat by small-footprint, discrete-return LiDAR. *Forest Ecology and Management* **281**: 100-110

Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E. and Jacobson, J. (2010) Effects of fire on woody vegetation structure in African savanna. *Ecological Applications* **20:** 1865–1875

Stats SA (2012) http://interactive.statssa.gov.za/superweb/loadDatabase.do;jsessionid=3038C87155 1C2B4317A15FC54B4BFFBC?db=Descriptive11_wd, accessed January 2013

Streutker, D.R. and Glenn, N.F. (2006) LiDAR measurement of sagebrush steppe vegetation heights. *Remote Sensing of Environment* **102**: 135-145

Thompson, M. (1996) A standard land-cover classification scheme for remote sensing applications in South Africa. *South African Journal of Science* **92**: 34-42

Thornton, R. (2002) Environment and Land in Bushbuckridge, South Africa. Sciences-New York**:** 1-31

Treydte, A.C., Grant, C.C. and Jeltsch, F. (2009) Tree size and herbivory determine below-canopy grass quality and species composition in savannahs. *Biodiversity and Conservation* **18**: 3989-4002

Turner, W., Spector, S., Gardiner, N., Fladeland, M., Sterling, E. and Steininger, M. (2003) Remote sensing for biodiversity science and conservation. *Trends in Ecology & Evolution* **18**: 306-314

UNESCO (1996) Biosphere reserves: The Seville Strategy and the Statutory Framework of the World Network. Paris, UNESCO**:** 19-19

Van Aarde, R., Whyte, I. And Pimm, S. (1999) Culling and the dynamics of the Kruger National Park African elephant population. *Animal Conservation* **2**: 287-294

Van De Vijver, C.A.D., Folby, C.A. and Olff, H. (1999) Changes in the woody component of an east African savanna during 25 years. *Journal of Tropical Ecology* **15**: 545-564

van Wilgen, B.W., Trollope, W.S.W., Biggs, H.C., Potgeiter, A.L.F. and Brockett, B.H. (2003) Fire as a driver of ecosystem variability. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press**:** 149-170

van Wilgen, B.W., Govender, N. and Macfadyen, S. (2008) An assessment of the implementation and outcomes of recent changes to fire management in the Kruger National Park. *Koedoe* **50**: 22-31

Venter, F.J., Scholes, R.J. and Eckhardt, H.C. (2003) The abiotic template and its associted vegetation pattern. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger* *experience: ecology and management of savanna heterogeneity*. London, Island Press**:** 83-129

Vitousek, P., Mooney, H.A., Lubchenco, J. and Melillo, J.M. (1997) Human domination of earth's ecosystems. *Science* **277**: 494-499

Wehr, A. and Lohr, U. (1999) Airborne laser scanning - an introduction and overview. *ISPRS Journal of Photogrammetry and Remote Sensing* **54**: 68-82

Wessels, K.J., Mathieu, R., Erasmus, B.F.N., Asner, G.P., Smit, I.P.J., van Aardt, J., Main, R., Fisher, J., Marais, W., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R. and Jacobson, J. (2011) Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* **261**: 19–29

Whitecross, M.A., Archibald, S. And Witkowski, E.T.F. (2013) Do freeze events create a demographic bottleneck of *Colophospermum mopane*? *South African Journal of Botany* **83**: 9-18

Whyte, I.J., van Aarde, R. and Pimm, S.L. (2003) Kruger's elephant populations: Its size and consequences for ecosystem heterogeneity. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press**:** 332-348

Witkowski, E.T.F. and O'Connor, T.G. (1996) Topo-edaphic, floristic and physiognomic gradients of woody plants in a semi-arid African savanna woodland. *Vegetatio* **124**: 9-23

WWF (2006) Joining the dots: Species and protected areas - A contribution to the implementation of the CBD Programme of Work on Protected Areas. WWF report at CoP8, Curitiba, Brazil. http://assets.worldwildlife.org/publications/421/files/original/Connecting_the_Dots_-_Species_and_Protected_Areas.pdf?1345756580

Chapter 2 : From cattle ranching to conservation: land use succession effects on woody vegetation structure in a private and national reserve

J.T. Fisher[a,b,c], B.F.N. Erasmus[b], E.T.F. Witkowski[a], J. van Aardt[d], G.P. Asner[e], K.J. Wessels[f] and R. Mathieu[b]

[a]Restoration and Conservation Biology Research Group, School of Animal, Plant & Environmental Sciences, University of the Witwatersrand, WITS, Johannesburg, 2050, South Africa

[b] Centre for African Ecology, School of Animal, Plant and Environmental Sciences, University of the Witwatersrand, Private Bag 3, WITS, 2050, Johannesburg, South Africa

[c] Ecosystems Earth Observation, Natural Resource & Environment, Council for Scientific and Industrial Research (CSIR), P.O. Box 395, Pretoria, 0001, South Africa

[d] Chester F. Carlson Center for Imaging Science, Rochester Institute of Technology: 54 Lomb Memorial Drive, Rochester, New York, 14623, USA

[e] Department of Global Ecology, Carnegie Institution for Science, 260 Panama Street, Stanford, CA 94305, USA

[f] Remote Sensing Research Unit, Council for Scientific and Industrial Research (CSIR)-Meraka Institute, P.O. Box 395, Pretoria, 0001, South Africa

Corresponding Author:

J.T. Fisher

(Tel: +27 (11) 717 6408; Fax: +27 (11) 717 6494; jolenefisher@gmail.com)

2.1 Abstract

Land-use legacies and management policies and objectives will affect the effectiveness of protected areas to conserve biodiversity. MalaMala (a concession within Sabi Sand Wildtuin), a private game reserve, and the adjacent area in the Kruger National Park (Kruger) in South Africa provide a natural comparison of the efficacy of different types of conservation management. We measured three-dimensional woody vegetation structure, as an integral component of biodiversity, across 6 200 ha in the two reserves using a LiDAR (Light-Detection-and-Ranging) sensor. We compared the affect of land-use succession timelines and conservation management objectives in the two reserves on woody structural diversity. Vertical canopy diversity was measured using: i) percent cover of woody vegetation extracted from LIDAR canopy height models, ii) a volumetric pixel (voxel) approach to extract 3D vertical canopy-height profiles; and horizontal diversity using landscape metrics. MalaMala had higher vegetation density than Kruger in the <3 m (2.5 times) and >6 m (2.7 times) height classes. This vegetation was in the form of larger, more cohesive patches as a result of the legacy of previous land-use (cattle ranching) and current management practices (bush clearing, and the presence of megaherbivores). Length of exposure to, and recent densities of, megaherbivores (particularly elephants) has altered the density of tall trees in the two reserves, thus affecting structural heterogeneity and associated habitat options for small-bodied vertebrates. These differences in vegetation structure are exacerbated by current management practices (e.g. bush-clearing and fire regime), with potential implications for faunal biodiversity conservation across a wide range of scales.

Keywords: Carnegie Airborne Observatory, land-use succession, LiDAR, management, megaherbivores, structural heterogeneity

Abbreviations

CAO: Carnegie Airborne Observatory; Kruger: Kruger National Park; LiDAR: Light-Detection-and-Ranging; MalaMala: MalaMala Private Game Reserve; SSW: Sabi Sand Wildtuin; Voxel: Volumetric pixel

2.2 Introduction

The structure and composition of woody vegetation is a defining feature of savannas (Scholes and Archer 1997). Savanna trees, and large trees in particular, are vital to maintain ecosystem functioning. The contribution of large single standing trees to structural diversity and savanna function is widely recognized (Belsky et al. 1993; Manning et al. 2006; Treydte et al. 2009). Specifically, large trees provide shade, reduce evapo-transpiration of the below-canopy herbaceous layer, and increase local nutrients which are accumulated close to root systems (Belsky et al. 1993; Belsky 1994; Manning et al. 2006). The higher quality graze and shaded micro-habitat located around large trees therefore attracts fauna and so increases local biodiversity. In addition, the complex vertical architecture in savannas creates fine scale niches upon which birds (Bergen et al. 2007; Seymour & Dean 2009), arthropods (Halaj et al. 2000), mammals (Williams et al. 2002; Lumsden & Bennet 2007), reptiles (Smart et al. 2005) and parasitic plants (Dzerefos et al. 2003) are reliant. All of these aspects stress the need for conservation of structural diversity in savannas.

Compared to the areas surrounding reserves, different protected areas can easily be considered by the public to be equal in their ability to conserve biodiversity, thus discounting the effects of previous land-use and current management practices. Statutory protected areas only cover a little over 12% of the earth's land surface (Chape et al. 2005); therefore, evaluating their effectiveness in conserving biodiversity is of fundamental importance (Chape et al. 2005; Gaston et al. 2006; Latif Khan et al. 1997; Parrish et al. 2010). The lowveld (low altitude) region of South Africa has a rich history of different land uses (Joubert 2007), most recently seeing a change from cattle ranching to conservation / ecotourism in the 20th century. In 1918 the lowveld was a mosaic of government land, company-, and private-farms. In 1898 the Sabi Game Reserve was proclaimed, which ultimately led to the formation of the Kruger National Park (Kruger) in 1926. The inception of Kruger, the largest national protected area in South Africa, was a catalyst for the formation of many private game reserves that took advantage of the success of Kruger and the local and foreign tourists it attracted (Joubert 2007). The reserves in the lowveld therefore provide a natural template to study the potentially confounding effects of land use legacy with the effectiveness of different management objectives (private game reserve and statutory national park) for the conservation of biodiversity.

Plant ecologists traditionally have used field-based methods to measure vegetation structure, e.g., sampling vegetation using transects or plots. While these studies are effective at measuring a relatively large number of trees, in the order of 10^2-10^5, they typically cover small areas (<5 ha) (Higgins et al. 1999; Shackleton 2000; Witkowski and O'Connor 1996). However, the inherent heterogeneity and patchiness in savannas (Scholes and Archer 1997) require alternative methods to measure vegetation structure over larger extents and at various spatial scales to ensure heterogeneity at all scales is captured. Light Detection and Ranging (LiDAR), which is based on accurate measurement of the return trip distance of emitted laser pulses, is now widely used in terrestrial environments to assess woody vegetation structure and map landscape topography (e.g., Lefsky et al. 2002). With small-footprint (<1 m), discrete-return LiDAR (which collects point-based x, y, z data of all terrestrial structures), we are able to measure large areas at fine resolutions, obtaining fine scale results similar to field studies (Lefsky et al. 2002; Turner et al. 2003). LiDAR data are costly, but it is still more cost-effective per unit area compared to field studies when large tracts of land need to be analyzed (Kirton et al. 2009). Such data can be used to assess structural variation across landscapes (e.g. Wessels et al. 2011).

Land-use legacy effects and management practices will affect vegetation structural diversity in protected areas, thus altering their ability to effectively conserve biodiversity. We measured woody structural diversity in Kruger and an adjacent private game reserve (MalaMala, within SSW) using small-footprint, discrete-return LiDAR collected with the Carnegie Airborne Observatory Alpha sensor package (CAO, Asner et al. 2007) to evaluate the effectiveness of these protected areas. The aim of our investigation was to assess if the effect of different land use succession timelines, from cattle farming to conservation, and the different conservation management practices is reflected in the current patterns of woody structural diversity. We discuss implications for conservation and assess the usefulness of small-footprint, discrete-return LiDAR to measure woody vegetation structure at the landscape scale in semi-arid savannas.

2.3 Methods

2.3.1 Study site

The two study sites border one another on the boundary between Kruger and MalaMala in Mpumalanga Province, north-eastern South Africa (Fig. 2.1), spanning a total of 6 200 ha (2 900 ha in Kruger and 3 300 ha in MalaMala). The sites have the same landtype and vegetation types, and similar geologies, altitudinal range and mean annual precipitation and temperature, but different management objectives (Table 2.1). Vegetation structure comprises tall shrubland with a few trees and relatively dense low woodland. Dominant woody species include the trees *Terminalia sericea*, *Combretum zeyheri*, *C. apiculatum*, *Acacia nigrescens*, and the shrubs *Dichrostachys cinerea* and *Grewia bicolor*. Common grass species include *Pogonarthria squarrosa*, *Tricholaena monachne*, *Eragrostis rigidior*, *Panicum maximum*, *Aristida congesta*, *Digitaria eriantha*, and *Urochloa mossambicensis* (Mucina & Rutherford 2006).

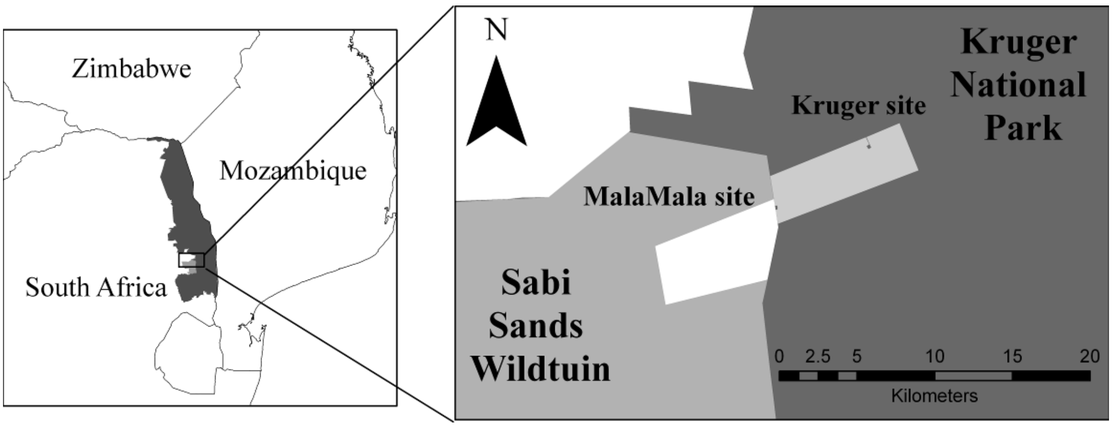

Figure 2.1: Location of study sites within the Kruger National Park and Sabi Sand Wildtuin (MalaMala Private Game Reserve), South Africa.

Kruger and MalaMala were managed as private cattle ranches from the early 1900's until 1926, when Kruger was established. MalaMala was operated as a private farm for an additional 12 years until an outbreak of rinderpest required the removal of all cattle (Joubert 2007). In 1934 MalaMala and surrounding farms formed a conservancy now known as Sabi Sand Wildtuin (SSW), but each farm/concession maintained separate management within SSW. A later outbreak of foot-and-mouth disease and continued hunting on the

Table 2.1: Similarities in abiotic attributes and differences in current management of the MalaMala Private Game Reserve and the Kruger National Park, South Africa.

	MalaMala Private Game Reserve	**Kruger National Park**
Geology	Granite (Potassic granite/gneiss, Makatswi gneiss, Nelspruit)	Granite (Potassic granite/gneiss)
Landtype (Soils)	Fersiallitic	Fersiallitic
Altitude	315 – 439 m above sea level	381 – 439 m above sea level
Mean annual temperature	21 ˚ C	21 ˚ C
Mean annual precipitation	Approx. 620 mm	Approx. 650 mm
Vegetation type	Granite Lowveld[1]	Granite Lowveld[1]
Fire regime	Natural fires are controlled	1954 – 1992: Triennial burns, block plots, in late winter
	Annual burns	1994-2002: *Laissez faire* burn policy (only lightening induced fires)
	No holistic fire policy exists for the Sabi Sand Wildtuin conservancy, each concessionaire manages land separately[2]	2002 – present: Combination of point ignitions, lightning fires and unplanned fires to burn an annual target area determined by rain and fuel load [3]
Land management	Land management of MalaMala by cattle ranchers in the early 1900's led to bush encroachment (roads were developed in seep zones, resulting in soil compaction influencing stream flows, natural fires were extinguished, elephant were hunted for ivory). Current management (since 1965) includes bush-clearing and mowing to combat bush encroachment, each concessionaire manages land to promote responsible and ethical tourism and preserve biodiversity[2]	The initial management was aimed at the recovery of ungulate populations from the rinderpest pandemic at the end of the 19th century and from previous excessive meat and ivory hunting. Current management has clear research-driven management objectives, culminating in a strategic adaptive management approach utilizing Thresholds of Potential Concern (TPC) derived by scientists, managers and stakeholders[4]
Area	3 300 ha (Study site) 13 500 ha (MalaMala) 65 000 ha (Sabi Sand Wildtuin)	2 900 ha (Study site) 2 200 000 ha (Kruger National Park) 3 200 000 ha (Great Limpopo Transfrontier Park)
Elephant density	0.014 elephant/ha[5] in SSW in 2008	0.039 elephant/ha (approximate elephant densities near Nkuhlu herbivore exclosure in 2008)[6]
Current mission statement / management objectives	"through the sustainable and profitable management of the environment that forms MalaMala Game Reserve, and through practices of responsible and ethical tourism, this vast wilderness will remain untouched for the enjoyment of all inhabitants of this country (South Africa), and the planet"	"to maintain biodiversity in all its natural facets and fluxes with regard to ecosystem functioning, and to take a pro-active role in legal and statutory issues affecting biodiversity"

[1] Mucina & Rutherford (2006); [2] http:\\www.malamala.com\conservation.htm, accessed January 2013; [3] van Wilgen et al. (2008); [4] Biggs & Rogers (2003); [5]M.Grover Pers. Comm.; [6] Asner & Levick 2012

private lands, led to the construction of a boundary fence separating Kruger and SSW in 1961 (http://www.sabisand.co.za/ssw-history.html, accessed December 2012), drawing an official line between the two different management approaches until the removal of the fence in 1993. In 1962 MalaMala was one of the first private reserves in South Africa to replace hunting safaris with photographic safaris. The present day MalaMala is still contained and unfenced within the SSW boundary, however, it is no longer part of the conservancy.

2.3.2 Light Detection and Ranging (LiDAR) data

LiDAR data were collected in April 2008 using the Carnegie Airborne Observatory (CAO) Alpha system for approximately 35 000 ha that spans a comprehensive land-use gradient from the Kruger, through the Sabi Sand reserve, to the intensively utilized communal lands. The work reported here used a subset (6 200 ha) and focused on the conservation land use, as opposed to communal rangelands to the west. The CAO combines both imaging spectroscopy (hyperspectral imaging) and LiDAR technologies to study ecosystems at the regional scale (Asner et al. 2007). The CAO was operated in Alpha mode, which is intended for high-resolution mapping of up to 20 000 ha/day at a 0.5-1.5 m spatial resolution. The spectrometer can acquire imagery in up to 288 channels of 1.8 nm bandwidth in the 400-1050 nm wavelength range and has a swath of 1,500 pixels. The spectrometer is co-mounted with the LiDAR sensor which can acquire both waveform and discrete-return data; however, only discrete-return data were used for this study. The integrated GPS-IMU sub-system in the CAO provides the position and orientation of the sensors in 3D, while the CAO algorithms ensure that data inputs from both the spectrometer and the LiDAR system are co-located and precisely projected to ensure geographically aligned output (Asner et al. 2007). The CAO Alpha LiDAR sub-system provides 3D vegetation structural information, as well as high resolution digital elevation models. For this study, the discrete-return LiDAR data were collected at 2000 m above ground level with a laser pulse repetition frequency of 50 kHz, laser spot spacing of 1.12 m, and four returns per pulse. The first LiDAR return typically indicates the top of canopy, or the sole return in the case of a ground hit, while the last return is often associated with the ground, unless dense vegetation hindered signal penetration to the ground level. Algorithms, based on between-return vertical angles, are used in pre-processing steps to classify ground versus non-ground returns.

LiDAR data are collected as a three-dimensional point cloud (Fig. 2.2a). A digital surface model (DSM) and digital elevation model (DEM) were derived through linear interpolation of the first and ground returns, respectively. A canopy height model (CHM) was subsequently constructed (1.12 m resolution) by subtracting the DEM from the DSM to be used for top-of-canopy vegetation structural analysis (Fig. 2.2b); this CHM represents the highest value for each pixel. In addition, we used a volumetric pixel-based (voxel) approach, which decreases sensitivity to local variations in leaf and branch characteristics (Asner et al. 2008; Lefsky et al. 2002; Popescu & Zhao 2008), in order to quantify vertical vegetation structure. The 3D point cloud was divided into voxels of 5 x 5 x 1 m (length, width, height), with each voxel value representing the frequency of LiDAR returns for that voxel relative to the number of returns in the entire 5 x 5 m horizontal cell. This also helps to normalize and account for different point densities in areas of overlapping flight lines. Individual 1 m height binned images (the horizontal image resolution is 5 x 5 m) were stacked to create multiband images (Fig. 2.2c). The frequency values of the stacked voxels over each 5 x 5 m horizontal cell in the multiband image can be represented in two-dimensions as a vertically distributed vegetation density profile, representing the frequency of LiDAR returns in one meter height increments. Vertical profiles can be extracted for either a single 5 x 5 m ground cell or a group of 5 x 5 m ground cells. In the latter case, the mean number of laser returns at each height for the given area was represented, along with the standard deviation of the mean number of returns. These values were converted to a percent value, thereby normalizing for area sampled. The resulting vertical profile is an indication of the mean density of vegetation at a particular height in one meter vertical increments. Both CHM and voxel measurements are useful for identifying differences in woody vegetation, with voxel measurements representing complexity within the canopy that CHM measurements would miss.

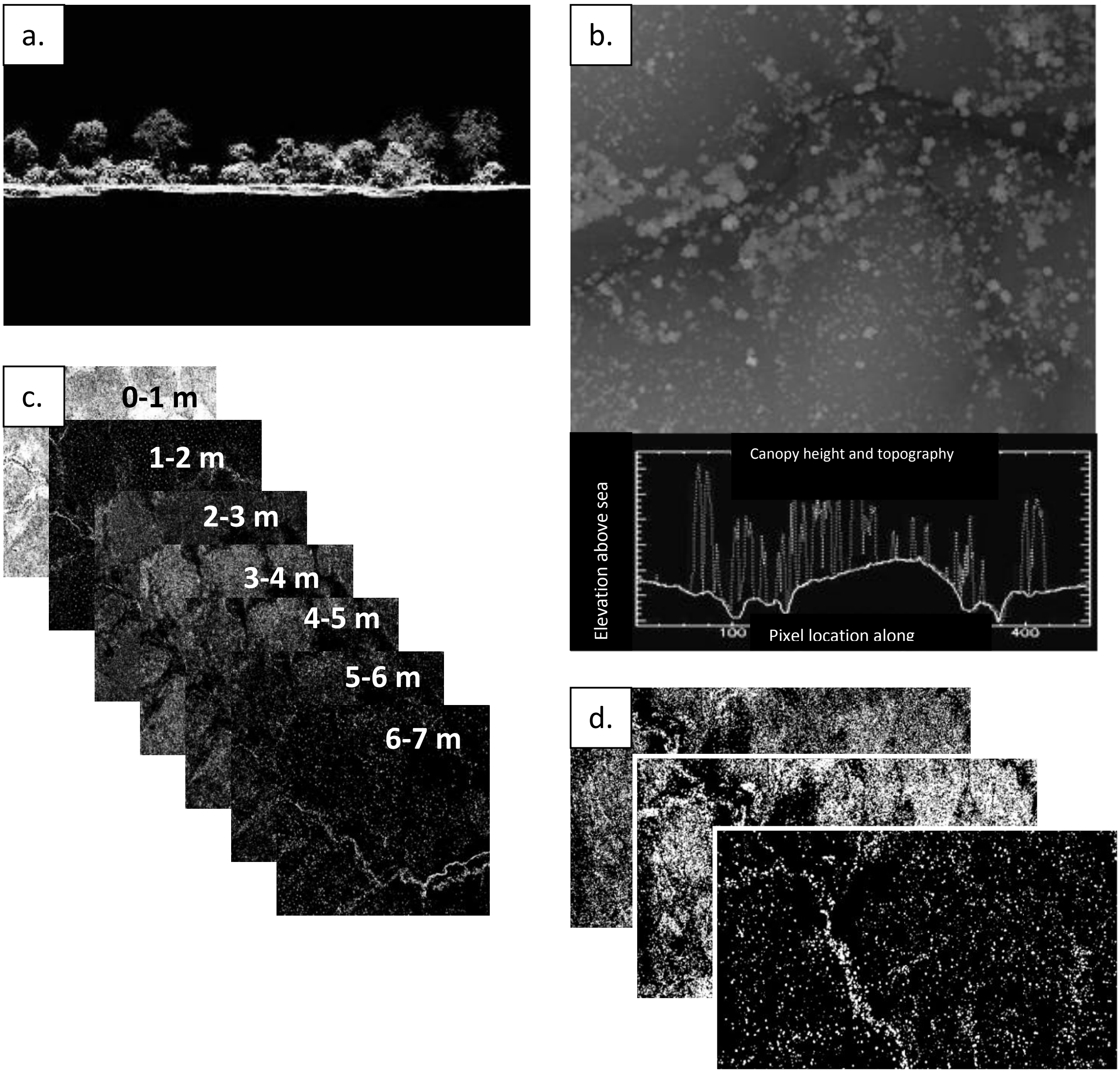

Figure 2.2: The various ways LiDAR (Light Detection and Ranging) data can be represented: (a) transect cross section through a point cloud, (b) overhead view of canopy height model, with a cross section indicating the digital elevation model (DEM, solid line) and the digital surface model (DSM, dotted line), (c) images of slices taken through a LiDAR point cloud in one metre increments (white indicates frequency of LiDAR returns which represents vegetation) and (d) sliced images as before binned into 1-3 m, 3-6 m and 6-15 m height classes.

2.3.3 Ground validation of vegetation height

We conducted concurrent field surveys during the flight campaign in April 2008 to assess the accuracy of the CAO LiDAR 1.12 m top-of-canopy height estimates. Thirty-six plots of 50 m x 50 m were sampled across the entire study site (35 000 ha) where a range of field data (e.g., grass biomass, woody biomass, tree species, fractional cover) were collected for various studies. A total of 883 trees (1-15 m in height) from a wide range of common lowveld species were sampled either in the sampling plots or in close proximity to the plots for LiDAR validation purposes and their exact GPS location recorded. Maximum tree heights were measured using either a graduated range pole, a laser rangefinder (TruPulse™ 360 ˚ B), or a Vertex hypsometer. A Trimble (Trimble® Recon® Handheld with aerial backpack) or Leica (GS20 Professional Data Mapper with handheld aerial) differential GPS was used to collect accurate geographic coordinates, which were differentially corrected to sub-meter accuracy using the Nelspruit trigonometric base station one second data (http://www.trignet.co.za/, accessed May 2008). We plotted the coordinates of each tree on the LiDAR derived vegetation canopy height model, extracted the maximum canopy height pixel for each tree, and verified the LiDAR-field data relationship using linear regression.

2.3.4 Vegetation structural analysis

Riparian corridors, water bodies, and roads were digitized using the hyperspectral data and masked out from the LiDAR-derived rasters prior to data extraction. We measured vertical vegetation structure from both the canopy height model (percent woody canopy cover, Fig. 2.2b) and using the voxel method (vertical vegetation density, Fig. 2.2c). We extracted vegetation height values and converted them to percent cover of woody vegetation at one meter increments for Kruger (2 900 ha) and MalaMala (3 300 ha), capturing both height and canopy cover variability. Differences in overall woody cover between the two sites were compared using a 2 x 2 contingency table χ^2 test. Height class distributions of percent woody cover (horizontal cover) were compared using the Kolmogorov-Smirnov (K-S) test. The voxel-based LiDAR approach was used to extract the three dimensional (3D) vertical vegetation density profiles for the two sites (Fig. 2.2c). Our sample sizes for each site was the number of voxels (n_{Kruger} = 1 160 022 voxels, $n_{MalaMala}$ = 1 318 506 voxels). We compared the two vertical profiles using descriptive statistics, e.g., kurtosis and variance, as well as Simpson's Index of Dominance (Wiegand et al. 2000) and the K-S test. Simpson's Index of Dominance values represented for each site are relative to one another, with '1' indicative of higher diversity and '0' zero diversity.

We furthermore calculated three landscape metrics, patch density, patch cohesion and largest patch index, using Fragstats 3.3 (McGarigal & Marks 1995) in order to assess the horizontal heterogeneity of woody vegetation in the two landscapes. The multi-band 'structural' images containing height bins of 1 m increments (Fig. 2.2c) were reclassified into three layers: 1-3 m, 3-6 m, and 6-15 m (Fig. 2.2d). These were heights at which the majority of changes were observed in the vertical height profiles. The selected height categories are also ecologically meaningful and relate to fire and herbivory: vegetation <3 m are affected by fire (Govender et al. 2006) and frost (Whitecross et al. 2012) and heavily browsed by small- to medium-size herbivores (Birkett and Stevens-Wood 2005, Scholes and Walker 1993, Witkowski 1983), vegetation in the 3-6 m height class are targeted by megaherbivores (elephant (*Loxodonta africana*) and giraffe (*Giraffa camelopardalis*) (Owen-Smith 1988, Asner & Levick 2012, Levick & Asner 2013), while the final height class is less influenced by fire and herbivory. The landscape metrics were calculated for woody vegetation within each height layer. Patch density is a measure of how many patches are present in the landscape. A higher patch density in the higher height classes implies increased 'bushiness' in the context of this study. Patch cohesion is indicative of how aggregated the patches are within the landscape and values range between zero and 100, with 100 representing greater aggregation or clumping (McGarigal & Marks 1995). Finally, largest patch index was included as a measure of how the woody vegetation patch sizes differ between the two sites.

2.4 Results

2.4.1 Ground validation of vegetation height

LiDAR derived vegetation heights showed a strong positive relationship with field measured vegetation heights ($p<0.001$, SE=0.73 m, $r = 0.96$). However, trees <2 m in height were underestimated by the LiDAR sensor (Fig. 2.3).

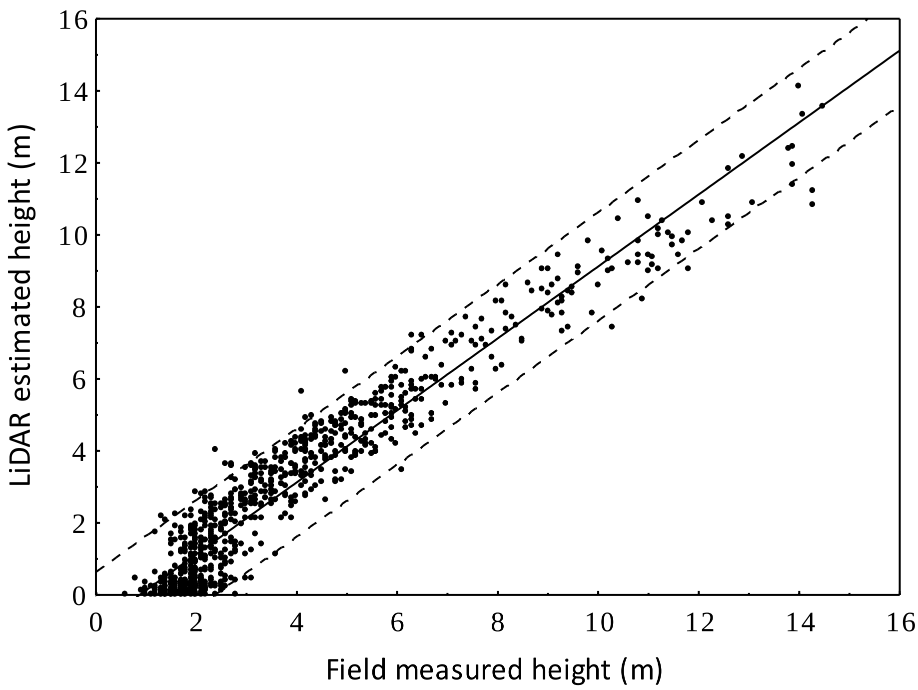

Figure 2.3: Field validation of LiDAR measured tree heights, calculated from 883 trees measured during the April 2008 Carnegie Airborne Observatory (CAO) Alpha aerial flight campaign (the dotted lines represent the 95% predicted confidence interval) (published in Wessels et al. 2011).

2.4.2 Vertical vegetation structure

MalaMala exhibited significantly greater woody cover (22.4%) than Kruger (19.6%) (χ^2_1=3.2x10^6; p<0.0001), a relative difference of 14.3%. In addition, the height class distribution of woody cover in the two reserves was significantly different (K-S, p<0.05). Both height class distributions are inverse J-shaped, typically considered indicative of a reproductive population structure (Mori et al. 1989); however, MalaMala had two times greater percent cover of woody vegetation above 5 m than Kruger (5.8 % versus 2.7 %, Fig. 2.4a).

Kruger had greater vertical heterogeneity than MalaMala, with higher vegetation density in the 4-6 m vertical profile, but lower density below (2.5 times less than MalaMala) and above (2.7 times less than MalaMala) these heights (Fig. 2.4b). This resulted in a profile with higher kurtosis, variance, and diversity (7.98, 1.49, and 1 respectively) than MalaMala (4.96, 1.16, and 0.77 respectively). The two vertical vegetation profiles are significantly different (K-S, p<0.05).

2.4.3 Horizontal vegetation structure

The Kruger site had many small patches (canopies) in the 1-3 m height class that occur close to one another (Fig. 2.5). This height class, on the other hand, was denser in MalaMala, with a similar number of patches and cohesion compared to Kruger (Fig. 2.5a and 2.5b), although these patches were four times larger in MalaMala compared to Kruger (Fig. 2.5c). These patterns were repeated in the 3-6 m height class, where we observed fewer but larger canopies occurring in close proximity to one another in MalaMala (Fig. 2.5). Similarly, the 3-6 m height class for both sites contained the largest, most cohesive (and thus fewer) patches compared to all other height classes. The greatest difference in horizontal canopy distribution between the sites occurred in the 6-15 m height class. Kruger had very few tall trees (1.8 times lower patch density than MalaMala, Fig. 2.5a) and these occurred far apart from one another (low cohesion, 1.3 times lower, Fig. 2.5b), in comparison to MalaMala, where there was a greater number of tall tree canopies that are more cohesive within the landscape.

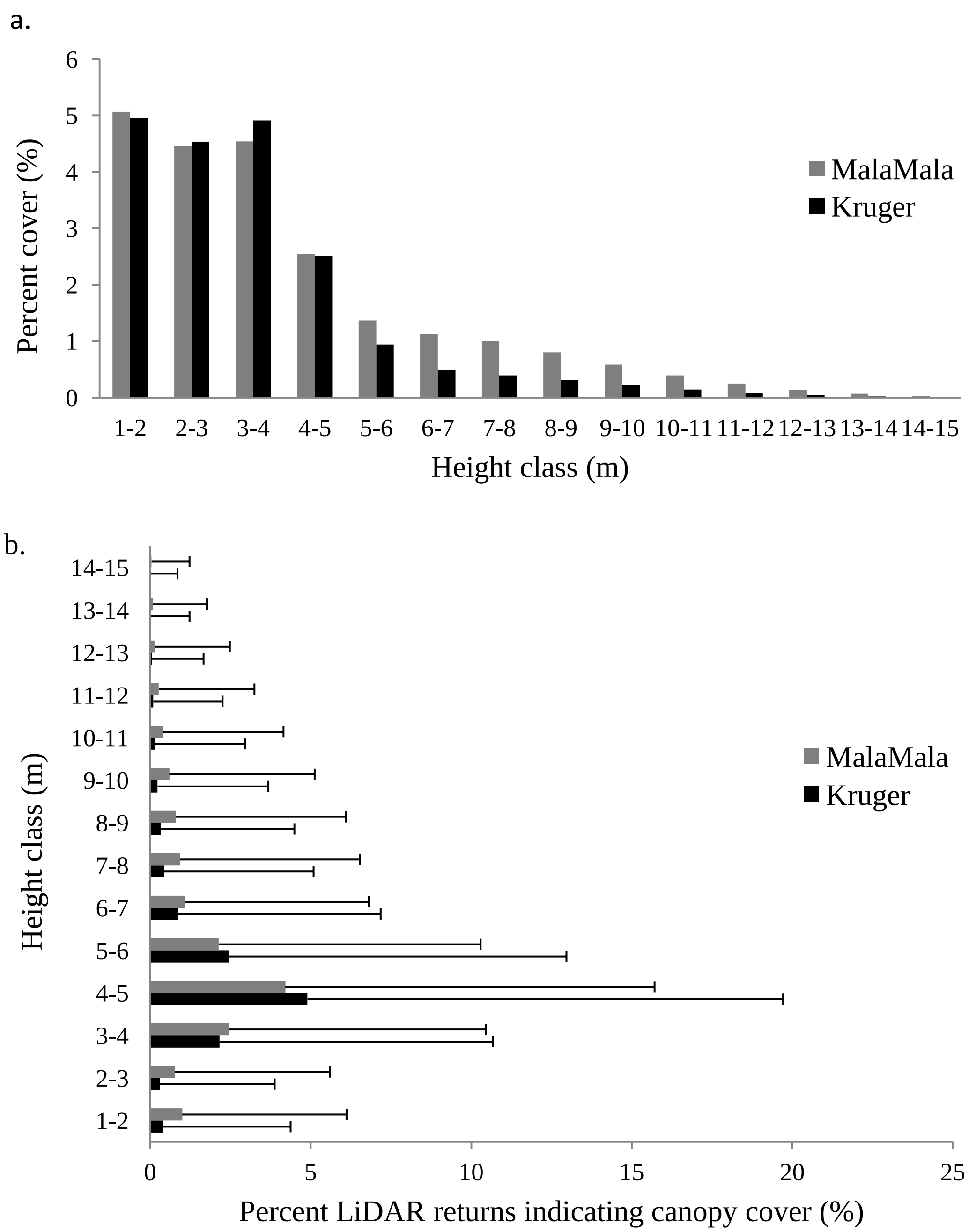

Figure 2.4: Measures of woody vegetation structure for 2 900 ha in Kruger National Park (Kruger) and 3 300 ha in MalaMala Private Game Reserve, South Africa. (a) Percent cover of woody vegetation, histograms represent area-normalised woody percentage cover (frequency/ha) in one metre increments, derived from the LiDAR canopy height model and (b) area-weighted mean three-dimensional vertical distribution of vegetation density (percent canopy cover) (error bars denote one standard deviation). Vegetation height classes should be interpreted as follows: 1-2 m includes vegetation from 1-1.9 m, 2-3 includes vegetation from 2-2.9, etc.

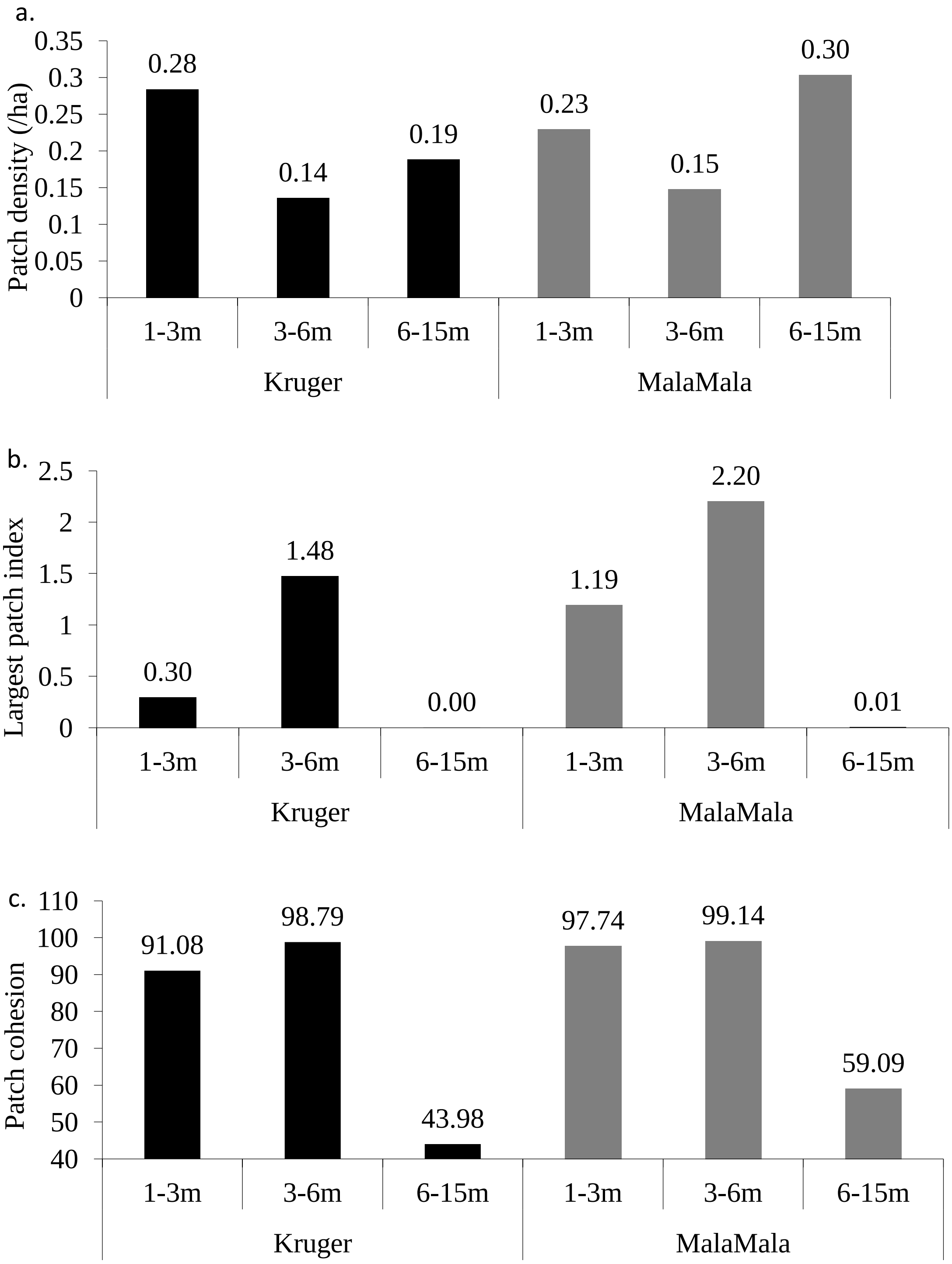

Figure 2.5: Horizontal vegetation structure measured for the 1-3 m, 3-6 m, and 6-15 m height bins for 2 900 ha in Kruger National Park (Kruger) and 3 300 ha in MalaMala Private Game Reserve, South Africa, using landscape metrics (a) patch density, (b) patch cohesion, and (c) largest patch index. Vegetation height classes should be interpreted as follows: 1-3 m includes vegetation from 1-2.9 m, 3-6 includes vegetation from 3-5.9 and 6-15 m is an inclusive category.

2.5 Discussion

The vegetation structural differences, due to land-use succession timelines and current management objectives, are relatively subtle in Kruger and MalaMala. MalaMala was only operated as a cattle ranch for an additional 12 years compared to Kruger, and fences between the reserves were only present for 32 years. Yet the differences reflected in both overall percent woody cover, and the vertical composition of that cover, are significant both statistically and ecologically. The most marked differences in vertical vegetation structure between the two reserves are below 3 m (Fig. 2.4b and 2.5c) and above 6 m (Fig. 2.4 and 2.5). Previous heavy grazing by cattle renders land more prone to bush encroachment (Papanastasis 2009; Tobler et al. 2010). Although we expect the two reserves to have similar vegetation density <3 m as both were previously used for cattle grazing, we see a higher vegetation density in this height category in MalaMala (with greater patch density and larger patches, Fig. 2.5). Differences were attributed to both historic land-use and management, and current bush clearing techniques. A longer period of cattle ranching in MalaMala relative to Kruger, combined with altered fire frequencies by land owners in the 1920's, contributed to increased woody biomass in the shorter height classes (Levick et al. 2009). Current MalaMala management attempted to rectify the bush encroachment problem by practicing bush clearing, mowing, and burning annually along seeplines (http://www.malamala.com/conservation.htm, accessed January 2013). However, the persistence of higher vegetation density below three metres indicates that management interventions, such as bush clearing at a local scale, may have been largely unsuccessful at the landscape scale because of woody encroachment occurring in non-cleared areas. Annual burns and bush clearing will inevitably maintain vegetation in a fire and 'herbivore' trap (bush clearing has similar effects on vegetation structure as browsing), leading to coppicing (Govender et al. 2006; Neke 2005; Owen-Smith 1988; Witkowski and O'Connor 1996) and a subsequent persistence of dense bush. Bush-encroached land is unfavourable to grazing herbivores in particular, with low predator visibility increasing stress levels in ungulate populations (Ripple and Beschta 2004), as well as reduced grazing quality (Treydte et al. 2009).

The large differences in tall tree abundances between Kruger and MalaMala (2.7% versus 5.8%; Fig. 2.4) might be due to each area's length of exposure to megaherbivores. The combination of an extra 12 years of cattle ranching, prior to proclamation, and subsequent hunting as a form of ecotourism for 24 years in MalaMala, as well as a fence separating the two reserves for 32 years, meant lower densities of megaherbivores such as elephant and giraffe in the pre-1993 MalaMala. Lower densities of megaherbivores would allow for greater recruitment and persistence of tall trees (Birkett and Stevens-Wood 2005; Helm et al. 2009; Hiscocks 1999; Levick et al. 2009; Owen-Smith 1988). For example, Helm et al. (2009) found elephants predominantly utilized marulas (*Sclerocarya birrea*), a dominant tall tree species in the South African Lowveld, in the 5-8 m height class. Elephant have also been shown to be responsible for treefall in savannas at an average rate of 2.6 trees ha^{-1} year^{-1} (Levick & Asner 2013), six times higher than in areas inaccessible to elephant (Asner & Levick 2012). Within the two reserves, fire and bush clearing (or lack thereof in Kruger) has the most notable effect on woody vegetation cover <3 m, while the 2.5 times difference in tall vegetation (>5 m) is attributed to the tenure length of megaherbivores. We have shown that even though the length of exposure to the same densities of megaherbivores in the two reserves has not been that different (densities were lower in MalaMala for 34 years while the perimeter fence was present), the effect on large tree densities is highly significant. The low abundances of tall trees and thus upper canopy patch cohesion could result in a reduction of faunal diversity, as well as reduced connectivity at the landscape level (Manning *et al.* 2006).

Semi-arid savannas are complex, heterogeneous systems making it a challenge to measure biodiversity and establish the causes affecting it. While field studies and passive remote sensing techniques are useful for detecting differences in woody vegetation structure as a function of management regimes (Fensham et al. 2005; Franklin et al. 2008; Higgins et al. 1999; Shackleton 2000; Witkowski and O'Connor 1996), they often lack vital 3D information over large areas. Field methods are effective at assessing differences in height class distributions, but results may be subject to observer and site selection biases, leading to a poor representation of the large-scale patterns and processes in heterogeneous environments. Alternatively, passive remote sensing (typically using multi-spectral sensors) is useful to monitor large-scale vegetation variation, especially woody cover; but inferences on the impacts of management practices on functional biodiversity are limited without any vertical information. However, with small-footprint, discrete-return LiDAR we found significant differences in vegetation structural heterogeneity between Kruger and MalaMala (Fig. 2.4) at the broad landscape level relevant to conservation planning and management. The relationship between LiDAR estimated heights and field measured heights deteriorates for trees below 2 m (Fig. 2.3). This weaker relationship is due to the sensor's horizontal and vertical resolution (which is dependent on the flight altitude) and collection settings, e.g., where a finite number of returns/pulse is collected, followed by the final return for a given pulse. For example, young or small trees grow vertically to escape the fire and herbivore trap before expanding their canopies horizontally. Low leaf area index cross-sections would therefore result in the tree being undetected by the laser beam. Since conditions were constant between the two sites (flying procedure, weather, phenology and species suite), we inferred that the underestimation of vegetation height below 2 m by the LiDAR was equal between study sites. Therefore, although values below 2 m may not be absolute, only relative values were necessary for comparisons between the two study sites. This phenomenon has since been corrected through the use of a different processing technique and the replacement of the CAO Alpha system with CAO AToMS (Airborne Taxonomic Mapping System) in 2011 (Asner et al. 2012). This is due to the sensor configuration, whereby per-pulse returns are measured for the first finite set of laser interactions, while always recording the last return for that pulse.

The succession of land-use and current management practices is reflected in the woody structural heterogeneity. Not all protected areas are equal in their ability to conserve biodiversity, with differences in vegetation structure likely to increase under current management practices (bush clearing and the presence of megaherbivores). From a broad conservation perspective, conversion of land-use from agriculture to conservation can be successful; although the success of conversion may depend on management practices such as fire regime, stocking density of megaherbivores and techniques to combat bush-encroachment which alter fine scale vertical heterogeneity. Particularly in southern African savannas, the impact of megaherbivores needs to be included in management objectives. The consequences of elephant behaviour on vegetation structure over a short-period of time can have a significant impact on woody structural diversity (Asner & Levick 2012; Levick

& Asner 2013) and thus the effectiveness of protected areas in semi-arid southern African savannas. We have provided a large-scale measurement of woody structural diversity, an integral component of savanna biodiversity, in two reserves. Small-footprint, discrete-return LiDAR is a more cost-effective, objective assessment tool of structural heterogeneity than field methods across the broad extents relevant for reserve managers.

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2.7 References

Asner, GP, Knapp, DE, Kennedy-Bowdoin, T, Jones, MO, Martin, RE, Boardman, J and Field, CB (2007). Carnegie Airborne Observatory: in-flight fusion of hyperspectral imaging and waveform light detection and ranging (wLiDAR) for three-dimensional studies of ecosystems. *Journal of Applied Remote Sensing* **1, 013536:** 1-21.

Asner, GP, Hughes, RF, Vitousek, PM, Knapp, DE, Kennedy-Bowdoin, T, Boardman, J, Martin, RE, Eastwood, M and Green, RO (2008). aInvasive plants transform the three-dimensional structure of rain forests. *Proceedings of the National Academy of Science* **105**: 4519-4523.

Asner, GP and Levick, SR (2012). Landscape-scale effects of herbivores on treefall in African savannas. *Ecology Letters* **15**: 1211-1217.

Asner, GP, Knapp, DE, Boardman, J, Green, RO, Kennedy-Bowdoin, T, Eastwood, M, Martin, RE, Anderson, C and Field, CB (2012). Carnegie Airborne Observatory-2: Increasing science data dimensionality via high-fidelity multi-sensor fusion. *Remote Sensing of Environment* **124**: 454-465.

Belsky, AJ, Mwonga, SM, Amundson, RG, Duxbury, JM and Ali, AR (1993). Comparative effects of isolated trees on their undercanopy environments in high- and low- rainfall savannas. *Journal of Applied Ecology* **30**: 143-155.

Belsky, AJ (1994). Influences of trees on savanna productivity: tests of shade, nutrients, and tree-grass competition. *Ecology* **75**: 922-932.

Bergen, K.M., Gilboy, A.M. & Brown, D.G. 2007. Multi-dimensional vegetation structure in modelling avian habiat. *Ecological Informatics* 2: 9-22.

Biggs, HC and Rogers, KH (2003). An adaptive management system to link science, monitoring, and management in practice. In: (Eds. Du Toit, JT, Rogers, KH & Biggs, HC.) *The Kruger experience: ecology and management of savanna heterogeneity.* Washington DC, Island Press**:** 61-82.

Birkett, A and Stevens-Wood, B (2005). Effect of low rainfall and browsing by large herbivores on an enclosed savannah habitat in Kenya. *African Journal of Ecology* **43**: 123-130.

Chape, S, Harrison, J, Spalding, M and Lysenko, I (2005). Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society Biological Sciences* **360**: 443-455.

Chen, Q, Baldocchi, D, Gong, P and Dawson, T (2008). Modeling radiation and photosynthesis of a heterogeneous savanna woodland landscape with a hierarchy of model complexities. *Agricultural and Forest Meteorology* **148**: 1005-1020.

Dzerefos, CM, Witkowski, ETF and Shackleton, CM (2003). Host preference and density of woodrose-forming mistletoes (Loranthaceae) on savanna vegetation, South Africa. *Plant Ecology* **167**: 163-177.

Fensham, RJ, Fairfax, RJ and Archer, SR (2005). Rainfall, land use and woody vegetation cover change in semi-arid Australian savanna. *Journal of Ecology* **93**: 596-606.

Franklin, DC, Petty, AM, Williamson, GJ, Brook, BW and Bowman, DMJS (2008). Monitoring Contrasting Land Management in the Savanna Landscapes of Northern Australia. *Environmental Management* **41**: 501-515.

Gaston, KJ, Charman, K, Jackson, SF, Armsworth, PR, Bonn, A, Briers, RA, Callaghan, CSQ, Catchpole, R, Hopkins, J, Kunin, WE, Latham, J, Opdam, P, Stoneman, R, Stroud, DA and Tratt, R (2006). The ecological effectiveness of protected areas: The United Kingdom. *Biological Conservation* **132**: 76-87.

Govender, N, Trollope, WSW and Van Wilgen, BW (2006). The effect of fire season, fire frequency, rainfall and management on fire intenstiy in savanna vegetation in South Africa. *Journal of Applied Ecology* **43**: 748-758.

Halaj, J, Ross, DW and Moldenke, AR (2000). Importance of habitat structure to the arthropod food-web in Douglas-Fir canopies. *Oikos* 90: 139–152.

Helm, CV, Witkowski, ETF, Kruger, L, Hofmeyr, M and Owen-Smith, N (2009). Mortality and utilization of Sclerocarya birrea subsp. caffra between 2001 and 2008 in the Kruger National Park, South Africa. *South African Journal of Botany* **75**: 475-484.

Higgins, SI, Shackleton, CM and Robinson, ER (1999). Changes in woody community structure and composition under constrasting landuse systems in a semi-arid savanna, South Africa. *Journal of Biogeography* **26**: 619-627.

Hiscocks, K (1999). The impact of an increasing elephant population on the woody vegetation in southern Sabi Sand Wildtuin, South Africa. *Koedoe* **42**: 47-55.

Joubert, S (2007). *The Kruger National Park: A History, Volume 1*. Johannesburg, High Branching Pty(Ltd).

Kirton, A, Scholes, RJ, Verstraete, M, Archibald, S, Mennell, K and Asner, GP (2009). Detailed structural characterisation of the savanna flux site at Skukuza, South Africa. *Proceedings of the International Geoscience and Remote Sensing Symposium*, IEEE Catalog number CFP09IGA-CDR.

Latif Khan, M, Menon, S and Bawa, KS (1997). Effectiveness of the protected area network in biodiversity conservation: a case study of Meghalaya state. *Biodiversity and Conservation* **6**: 853-868.

Lefsky, MA, Cohen, WB, Parker, GG and Harding, DJ (2002). Lidar remote sensing for ecosystem studies. *BioScience* **52**: 19-30.

Levick, SR, Asner, GP, Kennedy-Bowdoin, T and Knapp, DE (2009). The relative influence of fire and herbivory on savanna three-dimensional vegetation structure. *Biological Conservation* **142**: 1693-1700.

Levick, SR and Asner, GP (2013). The rate and spatial pattern of treefall in a savanna landscape. *Biological Conservation* **157**: 121-127.

Lumsden, LF and Bennett, AF (2005). Scattered trees in rural landscapes: foraging habitat for insectivorous bats in south-eastern Australia. *Biological Conservation* **122**: 205-222.

Manning, AD, Fischer, J and Lindenmayer, DB (2006). Scattered trees are keystone structures - Implications for conservation. *Biological Conservation* **132**: 311-321.

McGarigal, K and Marks, BJ (1995). FRAGSTATS: spatial pattern analysis program for quantifying landscape structure, USDA For. Serv. Gen. Tech. Rep.

Mori, SA, Rabelo, BV, Tsou, CH and Daly, D (1989). Composition and structure of an Eastern Amazonian forest at Camaipi, Amapa, Brazil. *Boletim do Museu Paraense Emilio Goeldi, Serie Botanica* **5**: 3-18.

Mucina, L and Rutherford, MC (Eds.) (2006). <u>The vegetation of South Africa, Lesotho and Swaziland,</u> Pretoria, *Strelitzia 19. South African National Biodiversity Institute*, pp. 439-539.

Neke, KS (2005). Woody plant allometry and demography in a semi-arid Africa savanna. The regeneration ecology of savanna woodlands in relation to human utilization.

Owen-Smith, N (1988). *Megaherbivores: The influence of very large body size on ecology*, Cambridge University Press.

Papanastasis, VP (2009). Restoration of Degraded Grazing Lands through Grazing Management: Can It Work? *Restoration Ecology* **17**: 441-445.

Parrish, JD, Braun, DP and Unnasch, RS (2010). Are we conserving what we say we are? Measuring ecological integrity within protected areas. *BioScience* **53**: 851-860.

Popescu, S.C. and Zhao, K. (2008) A voxel-based LiDAR method for estimating crown base height for deciduous and pine trees. *Remote Sensing of Environment* **112**: 767-781

Ripple, WJ and Beschta, RL (2004). Wolves and the ecology of fear: Can predation risk structure ecosystems? *BioScience* **54**: 755-766.

Scholes, RJ and Archer, SR (1997). Tree-grass interactions in savannas. *Annual Review of ecology and Systematics* **28**: 517-544.

Scholes, RJ and Walker, BH (1993). *An African savanna - Synthesis of the Nylsvley study*. Cambridge, Cambridge University Press.

Seymour, CL and Dean, WRJ (2009) The influence of changes in habitat structure on the species composition of bird assemblages in the southern Kalahari. *Austral Ecology* **35**: 581–592.

Shackleton, CM (2000). Comparison of plant diversity in protected and communal lands in the Bushbuckridge lowveld savanna, South Africa. *Biological Conservation* **94**: 273-285.

Smart, R, Whiting, MJ and Twine, W (2005). Lizards and landscapes: integrating field surveys and interviews to assess the impact of human disturbance on lizard assemblages and selected reptiles in a savanna in South Africa. *Biological Conservation* **122**: 23–31.

Tobler, MW, Cochard, R and Edwards, PJ (2010). The impact of cattle ranching on large-scale vegetation patterns in a coastal savanna in Tanzania. *Journal of Applied Ecology* **40**: 430-444.

Treydte, AC, Grant, CC and Jeltsch, F (2009). Tree size and herbivory determine below-canopy grass quality and species composition in savannahs. *Biodiversity and Conservation* **18**: 3989-4002.

Turner, W, Spector, S, Gardiner, N, Fladeland, M, Sterling, E and Steininger, M (2003). Remote sensing for biodiversity science and conservation. *Trends in Ecology & Evolution* **18**: 306-314.

van Wilgen, BW, Govender, N and Macfadyen, S (2008). An assessment of the implementation and outcomes of recent changes to fire management in the Kruger National Park. *Koedoe* **50**: 22-31.

Wessels, KJ, Mathieu, R, Erasmus, BFN, Asner, GP, Smit, IPJ, van Aardt, J, Main, R, Fisher, J, Marais, W, Kennedy-Bowdoin, T, Knapp, DE, Emerson, R and Jacobson, J (2011) Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* **261**: 19–29.

Whitecross, MA, Archibald, S and Witkowski, ETF (2012). Do freeze events create a demographic bottleneck for *Colophospermum mopane*? *South African Journal of Botany* **83**: 9-18.

Wiegand, K, Ward, D, Thulke, H-h and Jeltsch, F (2000). From snapshot information to long-term population dynamics of Acacias by a simulation model. *Plant Ecology* **150**: 97-114.

Williams, SE, Marsh, H and Winter, J (2002). Spatial scale, species diversity, and habitat structure: Small mammals in Australian tropical rain forest. *Ecology* **83**: 1317–1329.

Witkowski, ETF (1983) Ecology of the Klaserie Private Nature Reserve. MSc dissertation. University of the Witwatersrand, Johannesburg. 362 pp.

Witkowski, ETF and O'Connor, TG (1996). Topo-edaphic, floristic and physiognomic gradients of woody plants in a semi-arid African savanna woodland. *Vegetatio* **124**: 9-23.

Chapter 3 : Human-modified landscapes: patterns of fine-scale woody vegetation structure in communal savannah rangelands**

J. T. FISHER[1, 5]*, E.T.F. WITKOWSKI[1], B.F.N. ERASMUS[1], J. VAN AARDT[2], G. P. ASNER[3], K. WESSELS[4] AND R. MATHIEU[5]

[1]School of Animal, Plant and Environmental Sciences, University of the Witwatersrand, Private Bag 3, WITS, Johannesburg 2050, South Africa, [2]Chester F. Carlson Center for Imaging Science, Rochester Institute of Technology, 54 Lomb Memorial Drive, Rochester, New York 14623, USA, [3]Department of Global Ecology, Carnegie Institution, 260 Panama Street, Stanford CA 94305, USA, [4]Remote Sensing Research Unit, Council for Scientific and Industrial Research (CSIR)-Meraka Institute, PO Box 395, Pretoria 0001, South Africa, and [5]Ecosystems Earth Observation, Natural Resource and Environment, Council for Scientific and Industrial Research (CSIR), PO Box 395, Pretoria 0001, South Africa

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*Correspondence: J.T. Fisher Tel: +27 11 717 6408 Fax: +27 11 717 6494 e-mail: jolenefisher@gmail.com

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3.1 Summary

Despite electrification, over 90% of rural households in certain areas of South Africa continue to depend on fuelwood, and this affects woody vegetation structure, with associated cascading effects on biodiversity within adjacent lands. To promote sustainable use, the interactions between anthropogenic and environmental factors affecting vegetation structure in savannahs need to be understood. Airborne light detection and ranging (LiDAR) data collected over 4758 ha were used to examine woody vegetation structure in five communal rangelands around 12 settlements in Bushbuckridge, a municipality in the Kruger to Canyons Biosphere Reserve (South Africa). The importance of underlying abiotic factors was evaluated by measuring size class distributions across catenas and using canonical correspondence analysis. Landscape position was significant in determining structure, indicating the importance of underlying biophysical factors. Differences in structure were settlement-specific, related to mean annual precipitation at one site, and human population density and intensity of use at the other four sites. Size class distributions of woody vegetation revealed human disturbance gradients around settlements. Intensity of use affected the amplitude, not the shape, of the size class distribution, suggesting the same height classes were being harvested across settlements, but amount harvested varied between settlements. Highly used rangelands result in a disappearance of disturbance gradients, leading to homogeneous patches of low woody cover around settlements with limited rehabilitation options. Reductions in disturbance gradients can serve as early warning indicators of woodland degradation, a useful tool in planning for conservation and sustainable development.

Keywords: Carnegie Airborne Observatory, communal rangelands, LiDAR, resource gradients, size structure, sustainable resource use

3.2 Introduction

South African savannahs are home to over nine million rural residents, with over 90% of households dependent on fuelwood as a primary energy source, even where electricity is available (Twine *et al*. 2003). This dependence changes savannah vegetation structure (Freitag-Ronaldson & Foxcroft 2003), however, the interactions between socioeconomic and environmental factors that determine the level and type of use are complex, often resulting in non-linear trajectories of change that are difficult to quantify (Giannecchini *et al.* 2007).

Since the first South African democratic elections in 1994, the traditional authorities' control over natural resource use within the tribal trust lands has weakened (Kaschula *et al*. 2005; Twine 2005), people often being disinclined to limit personal consumption when others have unrestricted access due to diminished control (Scholes 2009). Population growth, coupled with non-residents using vehicles to collect large amounts of fuelwood for commercial purposes, has contributed to increased demand and subsequent decline in natural resources (Twine 2005). Distances walked to collect fuelwood increased from 100 m in the 1980s to approximately 1000 m in the 1990s, indicating the development of gradients of wood resource availability around settlements (Giannecchini *et al*. 2007). Since natural resources provide a buffer against adversity (Dovie *et al*. 2002; Shackleton *et al*. 2007), demand is unlikely to diminish.

These rural landscapes require continued management to ensure sustained availability of natural resources (Hobbs *et al*. 2006). Given that rural areas in South Africa are often situated around protected areas, resource use not only affects ecosystem services and function in the immediate area, but also the sustainability of neighbouring protected areas (Joppa *et al*. 2009). Biosphere reserves are intended to reconcile the real and perceived differences between conservation and sustainable use of natural resources (UNESCO [United Nations Educational, Scientific and Cultural Organization] 1996). However, since the inception of the Kruger to Canyons (K2C) Biosphere Reserve in South Africa in 2001, where this study is based, degradation of woodlands has continued. Between 1993 and 2006, intact natural vegetation, a priority conservation class, decreased by 7.3% in K2C (Coetzer *et al*. 2010). Settlement areas increased by 39.7%, predominantly in Bushbuckridge, with a concurrent increase of 6.8% for human-impacted vegetation (Coetzer *et al*. 2010). Between 1972 and 1994, human population density in Bushbuckridge doubled, and is currently estimated at 300 people km^{-2}, resulting in increased land use intensity and economic impoverishment (Pollard *et al*. 2003).

An understanding of local interactions between the biophysical factors, socioeconomics and natural resources is required to manage the resources sustainably (Hobbs *et al*. 2006; Giannecchini *et al*. 2007). The 'top-down' effect of fire and herbivory on savannah dynamics is relatively well understood (Scholes & Archer 1997; Sankaran *et al.* 2005; Helm *et al.* 2011); however, the factors influencing human use are not. The way people use savannahs depends on governance, socioeconomics, and individual and group behaviour, among other aspects (Scholes 2009), making the effects on savannah dynamics difficult to quantify and predict. Previous studies in Bushbuckridge suggested that patterns of use were settlement specific (Shackleton *et al*. 1994; Giannecchini *et al*. 2007), indicating the importance of village-level characteristics on resource extraction.

It is important to understand if patterns of vegetation structure are indeed settlement-specific, or whether generalizations across areas and communities can be made. Additional variables affecting patterns in rangelands are underlying biophysical factors. Higgins *et al*. (1999) included landscape position in their study of woody vegetation structure for three settlements. High levels of harvesting pressure in uplands relative to lowlands resulted in new vegetation patterns that did not reflect the undisturbed topographical differences measured in surrounding protected areas. However, at lower levels of use they showed that an interaction between abiotic factors and human impacts determine vegetation structural patterns. Given the ever-evolving human dynamics, the expectation is that vegetation structure will change within 10–20 years.

Light Detection and Ranging (LiDAR) sensors measure the three-dimensional structure of vegetation and the underlying terrain. Small-footprint, discrete return LiDAR allows for objective fine-scale (1.12 m spot spacing) measurement of woody vegetation over land areas much larger than those measured by field techniques to assess effects of fire, herbivores (Asner *et al*. 2009; Levick *et al*. 2009; Smit *et al*. 2010), reserve management and land use (Wessels *et al*. 2011). The overarching aim here was to quantify anthropogenic impacts on the finer-scale nature of patterns in woody vegetation structure in communal rangelands, relative to elements of underlying biophysical factors (rivers, topography, slope and aspect). The following questions were addressed: (1) How does rangeland woody vegetation structure, measured using size class distributions (SCDs), change with distance from settlements? (2) What are the relative effects of topographic position and distance from settlement on woody vegetation structure? (3) How do environmental variables, such as distance from settlements, roads and rivers, elevation above closest major river channel, slope, aspect and geology, influence the spatial and vertical distribution of woody vegetation in communal rangelands? We examined woody vegetation structure in five communal rangelands surrounded by 12 settlements using airborne LiDAR data collected in 2008 over large parts of Bushbuckridge.

3.3 Methods

3.3.1 Study area

Bushbuckridge Municipality is located in the southernmost portion of Limpopo Province (South Africa) (centred on 24.731°S, 31.181°E; Appendix 3.1, see supplementary material at Journals.cambridge.org/enc), a savannah region with three vegetation types: granite lowveld (dominant), gabbro grassy bushveld and legogote sour bushveld (Rutherford *et al*. 2006). In the granite lowveld, typical species include *Terminalia sericea*, *Combretum zeyheri* and *C. apiculatum* on the deep sandy uplands, while *Acacia nigrescens*, *Dichrostachys cinerea* and *Grewia bicolor* grow in the more clay-rich lowland soils. In the two other vegetation types, additional common species include *Sclerocarya birrea*, *Lannea schweinfurthii*, *Ziziphus mucronata*, *Dalbergia melanoxylon*, *Peltophorum africanum* and *Pterocarpus rotundifolius*. Mean annual precipitation, predominantly summer rainfall, ranges from > 900 mm in the west to 500 mm in the east, with a mean annual temperature of 22˚C. The geology is dominated by granite, with Timbavati gabbro intrusions (Venter *et al*. 2003).

The study encompassed five areas of communal rangelands (A–E) associated with 12 settlements (Appendix 3.1, see supplementary material at Journals.cambridge.org/enc). The human population in these settlements varies in the total number of people, density, age and gender (Appendix 3.3, see supplementary material at Journals.cambridge.org/enc), thereby exerting different resource extraction pressures on each associated rangeland. Although rangelands are predominantly used by the closest settlements, they are not exclusive use areas, especially with regard to the immigration of foreigners (both South Africans from surrounding areas and immigrants from neighbouring countries) who do not adhere to the local traditional authority's regulations (Twine 2005). Sites A and C are exceptions since their rangelands cannot be accessed from more than one settlement.

3.3.2 Light detection and ranging (LiDAR) data

LiDAR data were collected over 4578 ha by the Carnegie Airborne Observatory (CAO) in April 2008, using an airborne laser scanner. A pulse was actively emitted in the direction of the ground and the return time from emission to detection was measured to estimate the distance from the sensor to the object (ground or any land cover, i.e. tree or roof) (Wehr & Lohr 1999). The CAO was operated in Alpha mode, intended for high-resolution mapping of up to 20 000 ha day^{-1} at a 0.5–1.5 m spatial resolution of the raster of interpolated points. The CAO LiDAR sub-system provides three-dimensional (3-D) vegetation structural information, as well as high resolution digital elevation models. For this study, the discrete-return LiDAR data were collected 2000 m above ground level with a laser pulse repetition frequency of 50 kHz, laser spot spacing of 1.12 m, and four returns per pulse. The first LiDAR return typically indicated the top of canopy, or the sole return in the case of a ground hit, while the last return was often associated with the ground, unless dense vegetation hindered signal penetration. Algorithms based on between-return angles are used in pre-processing steps to classify ground versus non-ground returns. This resulted in a 3-D point cloud (x,y,z), providing a detailed representation of woody vegetation height structure.

A canopy height model (CHM) was first derived by subtracting a digital elevation model (DEM) from a digital surface model (DSM) of first canopy returns (van Aardt *et al*. 2006). The DSM and DEM are triangulated models generated through linear interpolation of all first (DSM) and ground (DEM) returns per 1.12 m grid cell. The CHM was resampled into one metre height increments to be used for vegetation structural analysis. For 3-D vegetation analysis (woody structure-environment relationships), the xyz point cloud was divided into volumetric pixels (voxels) of 5 × 5 × 1 m (length × width × height). The value of each voxel was represented by the number of LiDAR returns m^{-3} relative to the total number of returns in the entire 5 × 5 m column. Each column in the dataset was normalized to equal a total of 1000 returns (Asner *et al*. 2008). Ground validation of vegetation heights was conducted concurrent to the aerial data collection in 2008 (Wessels *et al*. 2011).

3.3.3 Vegetation structure with increasing distance from settlements and between landscape positions

Settlements, roads, rivers, crop fields and rangelands (used for natural resource extraction and grazing) were manually digitized across the study area using a combination of SPOT 5 imagery (panchromatic-multispectral merge (480–890 nm), 2.5 m spatial resolution, www.spotimage.com) and hyperspectral imagery collected by the CAO (1.12 m spatial resolution, 400–1050 nm; Asner *et al*. 2007). Distance classes of 200 m, radiating away from the settlements as sequential buffers, excluding riparian areas, roads and fields (Appendix 3.1, see supplementary material at Journals.cambridge.org/enc), were created using ArcMap 9.3 (Esri 2009). If the rangeland was surrounded by settlements (sites B and D), the resulting distance classes were 'circular' with the furthest zone as a midpoint between adjacent settlements (Appendix 3.2, see supplementary material at Journals.cambridge.org/enc). Seven distance classes were created for each site, except site B which, due to the circular nature of the distance classes and small area, could only accommodate six classes. For sites A, C and E, the maximum number and direction of distance classes were determined by a combination of the extent of the LiDAR data and either the distance to the Sabi Sand Wildtuin Private Game Reserve (SSW) boundary (Appendix 3.1, see supplementary material at Journals.cambridge.org/enc, sites A and C) or the distance to a natural landscape boundary (for example hills, site E). Upland and lowland areas were delineated manually using a winter SPOT 5 image (2.5 m spatial resolution) and the CAO DEM (1.12 m spatial resolution) within the study sites situated on granite (sites C, D and E). We were unable to reliably differentiate between topographic positions for sites occurring on gabbro, which has a much more subdued relief relative to granite, and hence topographic position was not included for these sites (sites A and B).

Within each distance class, 10% of the pixels in the top-of-canopy image were randomly sampled in ENVI v4.7 (ITT Vis [ITT Visual Information Systems] 2009), with five repeats of each. This 10% allowed for a representative number of pixels to be sampled per site (n_A = 391 747; n_B = 576 572; n_C = 168 603; n_D = 533 684; n_E = 1 103 537; n_{total} = 2 765 143 pixels), while ensuring pixels were not spatially autocorrelated (Asner *et al*. 2009). We recorded the mean value of the five repeats per distance and height class. Woody vegetation was defined as vegetation above 1 m. Per cent woody cover of each height (1–12 m) and distance class was calculated from the top-of-canopy data to derive a SCD of woody vegetation with increasing distance from each settlement. SCDs are useful indicators of vegetation change and population structure (Lykke 1998; Wilson & Witkowski 2003; Botha *et al.* 2004). Care must be taken when assessing SCDs at a landscape scale, as many species with various height structures are present. A SCD with an inverse-J shape is generally characteristic of vegetation with good rejuvenation and continuous replacement, whereas a flatter distribution indicates a lack of recruitment (Mwavu & Witkowski 2009). In disturbed savannah landscapes, people influence SCDs through harvesting of live wood and trees respond by coppicing (Neke *et al*. 2006), resulting in increased density of vegetation below three metres. Alternatively, the selective conservation of taller more mature trees for fruit and/or shade may be practised (Luoga *et al*. 2005; Twine 2005; Wessels *et al*. 2011).

ANOVA was used to test for differences in the mean per cent cover, as measured from the top of canopy images, between sites (five categories) in relation to distance (six categories) and height classes (14 categories) (Fig. 1). For each site separately, ANOVAs were used to explore differences in SCDs between distance (seven categories for sites A, C, D and E, and six for site B) and height classes (14 categories). For sites C, D and E, an additional ANOVA including topography was conducted (treatment = topography [two categories], factors = height and distance class). Significant differences between treatment combinations were evaluated using a Tukey post-hoc test (α = 0.05) (Zar 1999).

3.3.4 Woody structure-environment relationships

The relationship between three-dimensional woody vegetation structure and environmental variables was investigated using canonical correspondence analysis (CCA), a constrained ordination technique (ter Braak & Smilauer 2002). CCA represents synthetic environmental gradients from ecological datasets, in this case how woody vegetation density in different height classes extracted from the voxel dataset was affected by the environmental variables (Leps & Smilauer 2003). Environmental variables were chosen according to available data and their hypothesized influence on woody vegetation structure. All variables were classified into one of two categories: 'anthropogenic' (distance to closest settlement, distance to closest road), or 'natural' (horizontal distance to closest river channel, geology, slope, aspect and elevation relative to the nearest river channel [REM = relative elevation model]). The 'anthropogenic' variables were selected according to their perceived effect on resource use: fuelwood is more accessible closer to settlements and closer to roads and therefore use should be higher closer to these features. 'Natural' variables were chosen due to their known effect on savannah vegetation structure (Scholes & Walker 1993). Fire was not included in the set of 'natural' variables as there is no reliable fine-scale fire scar data for the area, but due to high human use and thus low fuel loads, fire is generally a less important variable than in conservation areas (Archibald *et al*. 2009).

Raster maps of distances to settlement, rivers and roads were created using the spatial analyst function in ArcMap 9.3, with a spatial resolution of 5 m, corresponding to the voxel data. Slope and aspect were calculated at 5 m spatial resolution in ENVI 4.7 using the topographical modelling feature and the CAO DEM. Only north (exposed slopes) and south (sheltered slopes) aspects were included in the analysis. The REM was constructed using the 'terrain: relative heights and slope position' module in SAGA (weighting = 5, search window = 100 m; see www.saga-gis.org). The 'normalized height' product was used, which is a normalized version of the slope heights output (values recalculated to range from 0–1; calculated as AACL/(AACL + ABRL), where AACL = altitude above closest channel and ABRL = altitude below ridge line [Bock *et al*. 2007]).

A minimum distance between each sampling point (voxel) was enforced to ensure points were not spatially autocorrelated, since vertical data from each voxel were used for the CCA and not mean of top-of canopy values. The minimum distance over which sampling points should be spread was determined using semivariograms, calculated in ENVI 4.7, as the range at which the sill occurs on the semivariogram was 150 m. Points were randomly sampled across the study area, using Hawth's analysis tools for ArcGIS, with a minimum distance of

150 m enforced between points to negate the effects of spatial autocorrelation, resulting in a total of 1651 points across the study area. Environmental variables for each point were extracted in ArcGIS and the frequency of LiDAR returns per voxel in the column was extracted in ENVI 4.7. By using the voxel data, which is a measure of vegetation density in 1 m height increments, we were able to characterize the actual structure of the vegetation. CANOCO v5 (ter Braak & Smilauer 2002) was used to perform the CCA.

Partial canonical correspondence analysis (PCCA) was conducted for all sites to establish the contribution of each group of explanatory variables (‘natural’ versus ‘anthropogenic’) to the total variance explained by a combination of the factors. A difference in the contribution of each group of variables was analysed using a t-test. PCCA is conducted by using the variable of interest as the explanatory variable (for example distance to settlement) and the other factors as covariates (all other natural and environmental explanatory variables) (Pysek & Leps 1991; Leps & Smilauer 2003). Once the variation explained by ‘natural’ and ‘anthropogenic’ variables was calculated, ordinations were performed for all sites combined, and then site-specific ordinations to establish which natural and anthropogenic factors influenced vertical vegetation structure. Geology was not included in the site specific ordinations, as each site only fell within a single geological type. All variables were tested for normality before performing the CCA, while rare height classes (such as > 10 m) were down-weighted. Forward selection by Monte Carlo tests (9999 permutations) were used to select significant environmental variables ($p < 0.05$) in the ordination, however, all variables were depicted. The total variance in each dataset accounted for by the explanatory variables was calculated as a percentage of the canonical eigenvalue contribution to the sum of all eigenvalues.

3.4 Results

3.4.1 Vegetation structure with increasing distance from settlements and between landscape positions

Mean per cent woody vegetation cover was significantly different between sites (Fig. 3.1; $F_{4,258} = 923.35$, $p < 0.0001$), except between sites A and D ($p > 0.05$). There was a significant interaction between site and distance from settlement (Fig. 3.1; $F_{20, 258} = 3.57$, $p < 0.0001$), with only site B experiencing a decrease in per cent canopy cover with increased distance from settlements (8.6 × less cover in the furthest distance class; Fig. 3.1). Increases in per cent canopy cover with increased distance from a settlement were as follows: site A = 1.7, site C = 1.2, site D = 2.0 and site E = 1.3 ×. Site E had significantly higher woody cover than all others for all distance classes ($p < 0.0001$), while site B had significantly lower woody cover across all height classes (Fig. 3.1). The overall trend was an increase in canopy cover with increased distance from settlement, although the opposite was true for site B (Fig. 3.1; site A: $F_{6,76} = 6.2$, $p < 0.0001$; site B: $F_{5,65} = 16.35$, $p < 0.0001$; site C: $F_{6,78} = 47$, $p = 0.0006$; site D: $F_{6,78} = 3.29$, $p = 0.0061$; site E: $F_{6,78} = 45$, $p = 0.0006$).

SCDs at increased distances from settlements followed an approximate inverse J-shape for sites A, C and D (Fig. 3.1*a*, *c* and *d*). There was a significant interaction between height class and distance from settlement ($F_{65,258} = 1.82$, $p = 0.0005$). The trend for sites A, C and D was a decreasing disturbance gradient with increased distance from settlement; however, the woody cover in each height class was site specific (Fig. 3.2*a*, *c* and *d*). Site B was severely impacted, with reduced vegetation cover in all size classes relative to the other sites (Fig. 3.2*b*).

3.4.2 Size class distributions of per cent cover on uplands and lowlands with increased distances from settlements

In the analysis that included topography as a factor (sites C, D and E only), there were significant interactions for sites D and E between height and distance class (Fig. 3.3; site D, distance class: $F_{65,65}= 1.77$, $p = 0.011$; topography: $F_{13,65} = 8.3$, $p < 0.0001$; site E, distance class: $F_{78,78} = 1.51$, $p = 0.0356$; topography: $F_{13,78} = 63$, $p < 0.0001$). However, for site C, only topography was significant (Fig. 3.3; distance class: $F_{36,36} = 0.88$, $p = 0.65$; topography: $F_{12,36} = 5.91$, $p < 0.0001$). The difference in SCDs between landscape positions is therefore greater than differences at increased distances from settlement, reflecting the greater importance of the physical template.

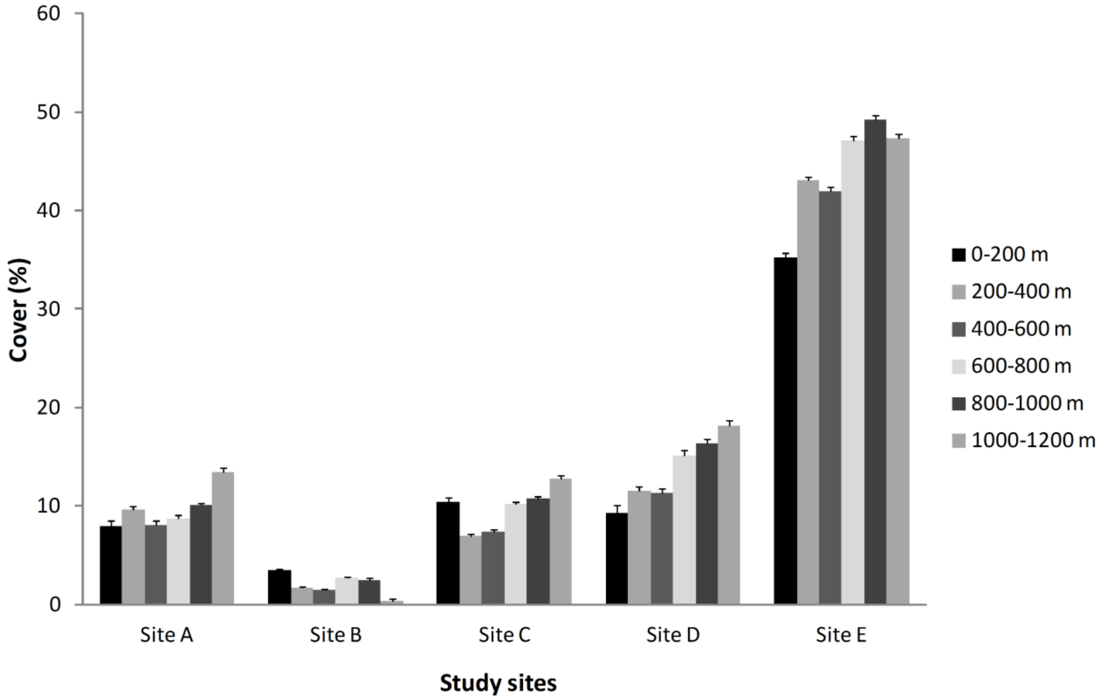

Figure 3.1: Per cent woody cover in rangelands at increasing distances from settlements for five sites in Bushbuckridge municipality, Limpopo Province, South Africa. Error bars denote standard deviation.

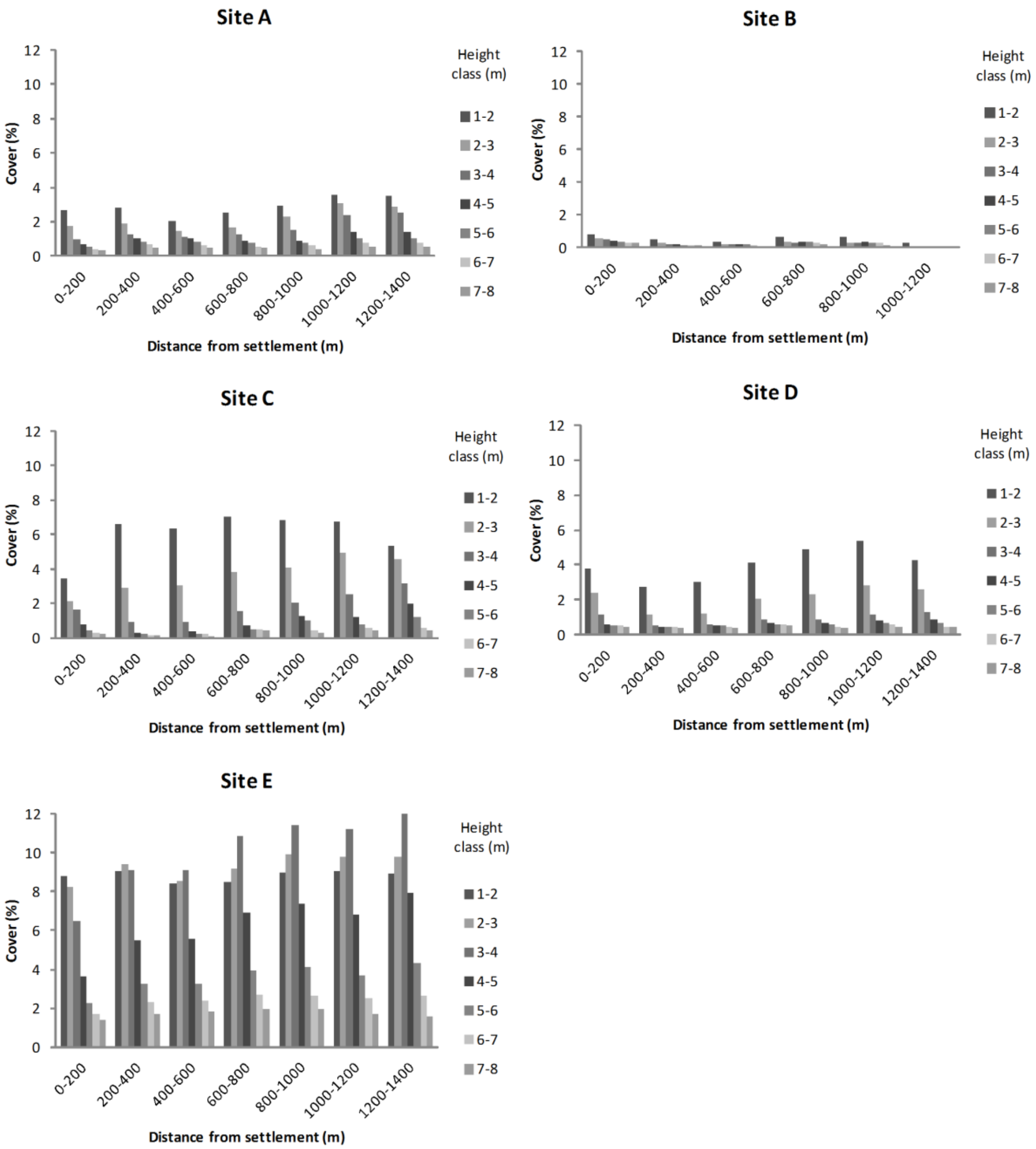

Figure 3.2: Size class distributions (SCD) of per cent canopy cover in distance classes of 200 m with increasing distance from settlement/s (sites A, B, C, D and E) in Bushbuckridge municipality, Limpopo Province, South Africa.

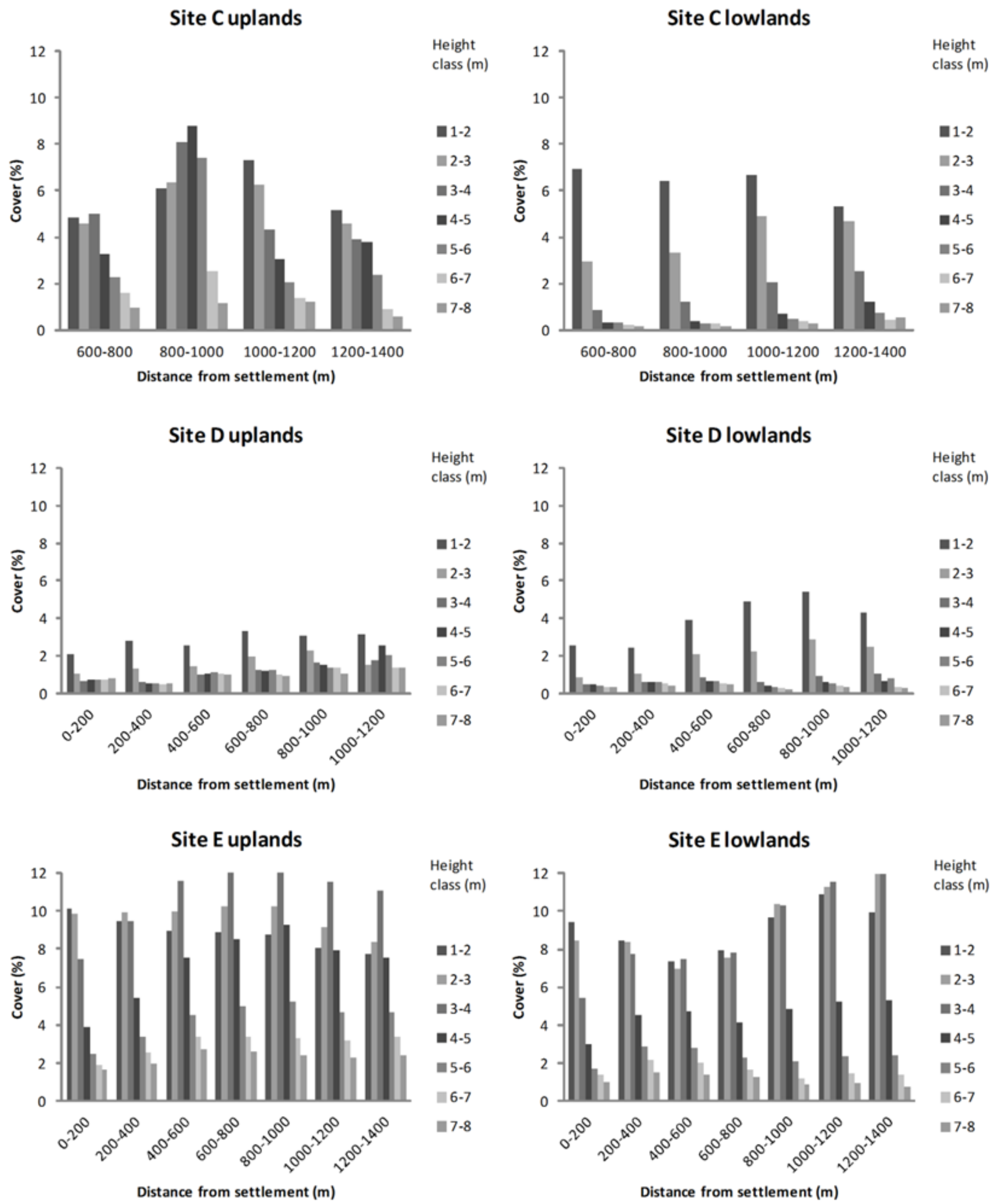

Figure 3.3: Size class distributions (SCD) of per cent canopy cover in distance classes of 200 m with increasing distance away from settlement/s on uplands and lowlands in Bushbuckridge municipality, Limpopo Province, South Africa.

3.4.3 Woody structure-environment relationships

Total variance accounted for in the spatial (horizontal and vertical) distribution of woody vegetation, measured by the explanatory variables from the voxel data, was relatively low (site A = 7.4%, B = 24.8%, C = 29.5%, D = 17.7%, site E = 3.6%). Even so, results of the PCCA showed 'natural' variables contributed more to total variance than 'anthropogenic' variables for each site, as well as for all sites combined (Fig. 3.4; $t_4 = 3.75$, $p = 0.0199$). However, this was expected as there are five ‘natural’ and only two ‘anthropogenic’ variables. For ‘all sites’, the proportion of the total variance explained by ‘natural’ and ‘anthropogenic’ variables included in the PCCA was 52%, implying that 48% of the total variance may be attributed to interactions between these variables and others not measured. For ‘all sites’ (A–E), ‘natural’ variables contribute more to the total variance explained than the ‘anthropogenic’ variables (Fig. 3.4).

The exploration of the spatial effects of individual ‘natural’ and ‘anthropogenic’ factors on vegetation density at 1 m height increments, measured from the voxel data using CCA, was first performed on a dataset including all sites (1650 samples). A combination of ‘anthropogenic’ and ‘natural’ variables was significant, with only aspect and geology not significant at this large scale (Fig. 3.5*a*). Distance to settlement was the most significant factor explaining the spatial distribution of vegetation density, followed by REM (both positively correlated with tall vegetation). These were followed by distance to roads, distance to rivers and finally slope, which was positively correlated with the tallest vegetation (10–12 m) (Fig. 3.5*a*). At this broad scale of analysis, a combination of ‘anthropogenic’ (increasing distances from settlement) and ‘natural’ (REM) factors was most important in affecting vertical structural heterogeneity. However, this pattern changed at finer site-specific scales (Fig. 3.5*b*–*f*).

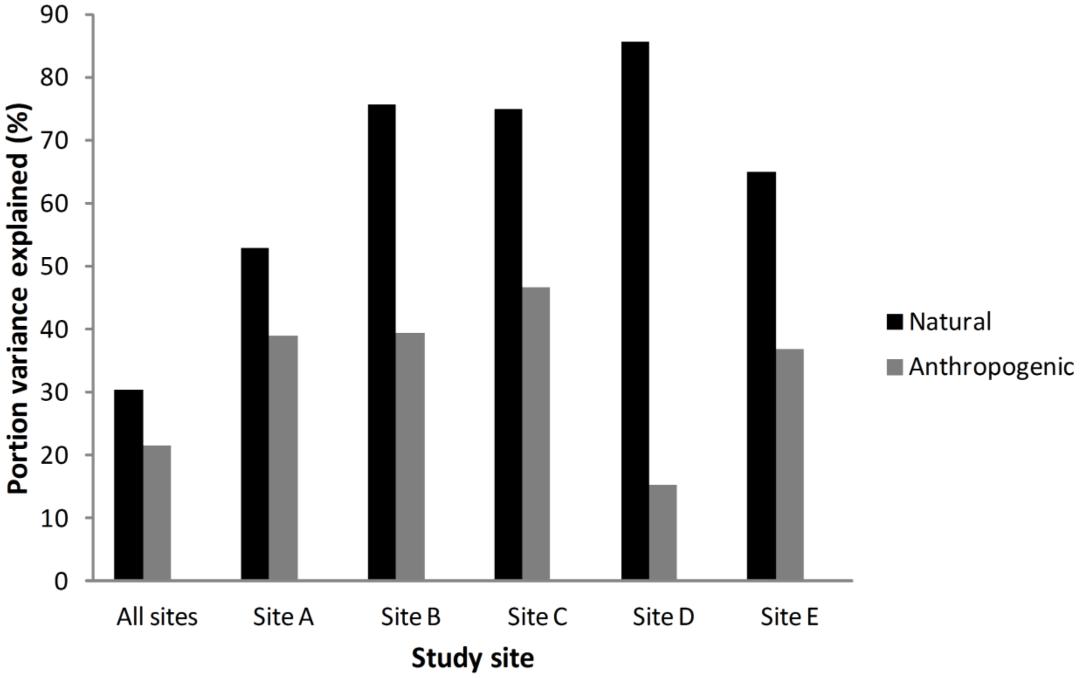

Figure 3.4: Contribution of ‘natural’ and ‘anthropogenic’ factors to total variance explained in the spatial distribution of vegetation within rangelands in Bushbuckridge Municipality, Limpopo Province, South Africa..

Vegetation in the 1–2 m height class was always separate from all other vegetation and not strongly correlated with any explanatory variable (Fig. 3.4*a*–*f*). Distance to settlement was significant in explaining the spatial distribution of vegetation for sites A, C and E (Fig. 3.5*b*, *d* and *f*), the three sites where the rangelands were only used by one settlement. Distance to roads, the other ‘anthropogenic’ factor, was only significant for site E. Only ‘natural’ explanatory variables were significant for sites B and D (Fig. 3.5*c* and *e*), the two sites where the rangelands were used by more than one settlement. We therefore identified trends across the sites related to the intensity of use (inferred from number of settlements accessing the rangeland), with vegetation structure on intensively used sites being more related to ‘natural’ variables (Fig. 3.5*c* and *e*) and those less intensively used related to ‘anthropogenic’ variables (Fig. 3.5*b*, *d* and *f*).

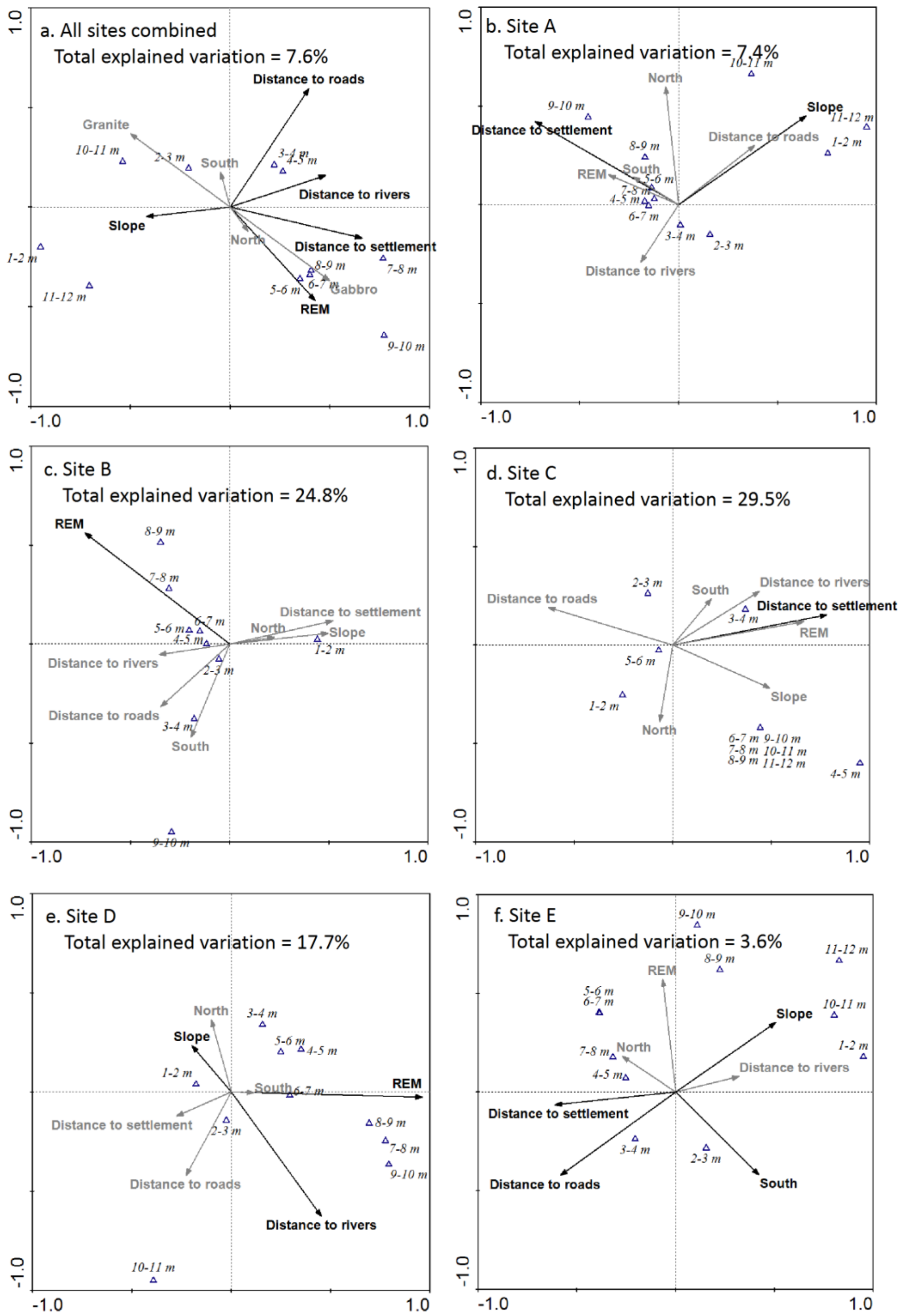

Figure 3.5: Natural' (distance to settlements and distance to roads) and 'anthropogenic' (Distance to rivers, elevation above river channel (REM), slope and aspect – North and South) factors in relation to patterns of three-dimensional woody vegetation structure. (a) all study sites combined, (b) Site A, (c) Site B, (d) Site C, (e) Site D and (f) Site E. Significant explanatory variables are shown in bold, non-significant indicated in grey. Total explained variation was calculated as sum of all canonical eigenvalues as a percent of all eigenvalues.

Vegetation within sites A, B and C tended to be more homogeneous, with many height classes occurring in close proximity in the ordination and thus indicative of greater spatial cohesion. When examining the SCDs of vegetation around settlements (Fig. 3.2), we saw that for all sites except site B, the cover of vegetation within size classes < 3 m was far greater compared to classes > 4 m. The same pattern emerged in the ordinations, where, for all sites, although to a lesser degree in site B, the lower height classes were more dispersed from the taller height classes, whereas taller vegetation was more grouped. Taller vegetation (> 5 m) was usually positively correlated with either slope or REM, high values of each indicating a drainage line or crest in the landscape, respectively. We would therefore expect short vegetation to be spatially widespread across the landscape, while tall vegetation would be clumped and tending to occur on crests and near rivers.

3.5 Discussion

In rural landscapes, understanding the interactions between underlying biophysical factors and human activities is critical for predicting future changes and planning for sustainable development. Our study covered 4578 ha, larger by orders of magnitude than the sampling areas examined by Shackleton *et al*. (1994) (0.81 ha) and Higgins *et al*. (1999) (1.08 ha). The findings of Shackleton *et al*. (1994) have held true over this greater sampling area, with disturbance gradients present around settlements that are only moderately used, as opposed to those with either high or low use intensity, where a gradient is not apparent. However, while Higgins *et al*. (1999) found vegetation structure within the rangelands to 'fall outside the topographic continuum' due to use, we found that significant differences in structure still existed across slope position (Figs 3.3 and 3.5). Woody vegetation structural patterns were a result of a combination of anthropogenic and natural factors (Figs 3.2–3.5), although the total variation explained in the CCA was relatively low (< 30%, Fig. 3.5). Much of the unexplained variation is likely to be due to species-specific variation in height structure along disturbance and topoedaphic gradients (Witkowski & O'Connor 1996).

Wessels *et al*. (2011) compared the overall tree canopy cover and height distributions between communal rangelands (the same rangelands of sites A, B, C and D, this study) and conservation areas at the landscape scale. They found geology to be an overriding factor affecting vegetation structure across this land-use gradient. At the finer scale of our investigation, geology was not significant (Fig. 5*a*), but landscape position was (Figs 3.3 and 3.5), highlighting that the hierarchical abiotic determinants of vegetation structure (Gillson 2004) remain true even in human-modified landscapes. The significant difference in the shape of SCDs between uplands and lowlands in the rangelands (Fig. 3.3) indicated that underlying fine scale abiotic factors have a stronger influence than resource extraction at moderate levels of land use.

The presence or absence of disturbance gradients around settlements and the shape of SCDs appear to be settlement specific. Giannecchini *et al*. (2007) highlighted the importance of settlement specific studies that incorporate local information, as broad-scale studies often neglect fine-scale variation. At site B, the low cover and lack of disturbance gradient was attributed to high use intensity, with the rangeland being surrounded by five settlements (Appendix 3.1, see supplementary material at Journals.cambridge.org/enc). One settlement using site B, Lillydale B, had a human population increase of 67.1% over the period 1993–2008, greater than for any other settlement in the area (Appendix 3.3, see supplementary material at Journals.cambridge.org/enc). As this increase cannot be attributed to births (1.1% increase), it seems most likely to be a result of immigration. This has negative impacts on sustainable resource use, as outsiders are less likely to respect traditional authorities (Kaschula *et al*. 2005). Similarly, settlements around site D (Ireagh A, Ireagh B and Kildare A; Appendix 3.3, see supplementary material at Journals.cambridge.org/enc) showed signs of immigration, as there was a decline in the birth rate and population decreases in the 5–19 year old age group, yet the overall population increased.

We found that 'natural' factors were more significant in determining the spatial pattern of woody vegetation for sites B and D, both used by more than one settlement (Fig. 3.5*c* and *e*). This result was confirmed by the SCDs and absence of disturbance gradients (Fig. 3.2*b* and *d*). High and increasing demand on these rangelands, caused by surrounding settlement density and thus higher population density (Appendix 3.3, see supplementary material at Journals.cambridge.org/enc), therefore appear to create a homogeneous landscape as a result of high use across the entire site. Homogeneous landscapes are negative for biodiversity, as habitat decreased especially for small-bodied fauna (Manning *et al*. 2006) and landscape function related to ecosystem services such as fruit, shade and fuelwood also decreased.

Alternatively, for the rangelands used by only one settlement (sites A, C and E), distance to settlement was a significant explanatory variable of the spatial distribution of vegetation (Fig. 3.5*b*, *d* and *e*). Settlements using these areas (Justicia A and Xanthia) showed relatively high population increases of 27.5% and 23.6%, respectively (Appendix 3.3, see supplementary material at Journals.cambridge.org/enc). However, use intensity remained low because use of the rangelands was geographically restricted to one settlement. Each of these three sites showed human-driven disturbance gradients (Fig. 3.2*a*, *c* and *e*), although differences in SCDs are greater between sites than between distance classes. Although the amount of cover is settlement specific, the presence of disturbance gradients is common in this landscape, as shown here and by Shackleton *et al*. (1994).

With increased demand on natural resources and more people collecting fuelwood using vehicles (Twine 2005), we expect disturbance gradients to diminish and few to develop as more areas become accessible, especially in these areas with dense settlements and reduced control over resource use. Disturbance gradients are expected in a human-modified landscape (Shackleton *et al*. 1994). Is the decline of these gradients into homogenous highly-used patches coupled with low woody cover a cause for concern? Coppice regrowth of harvested trees could change the tree's structure to a shrub form, which at a broad-scale might be viewed as bush encroachment (Luoga *et al*. 2005). In addition, adult coppicing trees are prevented from reaching sexual maturity, resulting in a lack of juvenile recruitment and therefore limited regeneration ability. A potential result of unsustainable harvesting of coppice regrowth following this trajectory is woodland degradation (Banks *et al*. 1996) unless community action is taken (R. Matsika, unpublished data 2011).

In conclusion, although results are inherently settlement specific and potentially dependent on an array of socioeconomic factors, some generalizations can be made. The shapes of the SCDs are similar for each settlement, but the cover of woody vegetation present within each size class is dependent on the use intensity. High use intensity in rangelands results in a disappearance of disturbance gradients, creating homogeneous patches of low woody cover. This will ultimately decrease structural diversity and thus biodiversity and woodlands will be unable to provide the necessary ecosystem services of fuelwood, shade and fruit. Therefore, land and conservation planners within the Kruger to Canyons Biosphere Reserve can use the early warning sign of initial development and later reduction of disturbance gradients, or indicators of them, to focus their conservation and sustainable development efforts. The continued high reliance on natural resources, especially fuelwood (Twine *et al*. 2003), highlights the need for continuous monitoring of this resource base to assess sustainability and provide solutions if use is unsustainable. Using LiDAR, it is possible to quickly and reliably measure and map woody vegetation structure across entire rangelands without observer bias. Repeated data collection will permit monitoring of the changes in woodland structure and biomass, change in patterns of rangeland use as natural resources decrease, and the effectiveness of management interventions (such as rotational harvesting). LiDAR will thus facilitate adaptive management of natural resources by providing an objective monitoring tool.

3.5 Acknowledgements

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3.6 References

Archibald, S., Roy, D.P., van Wilgen, B.W. & Scholes, R.J. (2009) What limits fire? An examination of drivers of burnt area in Southern Africa. *Global Change Biology* **15**: 613–630.

Asner, G.P., Knapp, D.E., Kennedy-Bowdoin, T., Jones, M.O., Martin, R.E., Boardman, J. & Field, C.B. (2007) Carnegie Airborne Observatory: in-flight fusion of hyperspectral imaging and waveform light detection and ranging (wLiDAR) for three-dimensional studies of ecosystems. *Journal of Applied Remote Sensing* **1**: 013536.

Asner, G.P., Hughes, R.F., Vitousek, P.M., Knapp, D.E., Kennedy-Bowdoin, T., Boardman, J., Martin, R.E., Eastwood, M. & Green, R.O. (2008) Invasive plants transform the three-dimensional structure of rain forests. *Proceedings of the National Academy of Science USA* **105**: 4519–4523.

Asner, G.P., Levick, S.R., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R., Jacobson, J., Colgan, M.S. & Martin, R.E. (2009) Large-scale impacts of herbivores on the structural diversity of African savannas. *Proceedings of the National Academy of Science USA* **12**: 261–278.

Banks, D.I., Griffin, N.J., Shackleton, C.M., Shackleton, S.E. & Mavrandonis, J.M. (1996) Wood supply and demand around two rural settlements in a semi-arid savanna, South Africa. *Biomass Bioenergy* **11**: 319–331.

Bock, M., Bohner, J., Conrad, O., Kothe, R. & Ringeler, A. (2007) Methods for creating functional soil databases and applying digital soil mapping with SAGA GIS. In: *Status and Prospect of Soil Information in South-Eastern Europe: Soil Databases, Projects and Applications*, ed. T. Hengl, P. Panagos, A, Jones & G. Toth, pp. 149–163, Italy: European Commission.

Botha, J., Witkowski, E.T.F. & Shackleton, C.M. (2004) The impact of commercial harvesting on *Warburgia salutaris* ('pepper-bark tree') in Mpumalanga, South Africa. *Biodiversity and Conservation* **13**: 1675–1698.

Coetzer, K.L., Erasmus, B.F.N., Witkowski, E.T.F. & Bachoo, A.K. (2010) Land cover change in the Kruger to Canyons Biosphere Reserve (1993–2006): a first step towards creating a conservation plan for the subregion. *South African Journal of Science* **106**: 26–35.

Dovie, D.B.K., Shackleton, C.M. & Witkowski, E.T.F. (2002) Direct-use values of woodland resources consumed and traded in a South African village. *International Journal of Sustainable Development and World Ecology* **9**: 269–283.

Esri (2009) *ArcMap*. Redlands, USA: ESRI Inc.

Freitag-Ronaldson, S. & Foxcroft, L.C. (2003) Anthropogenic influences at the ecosystem level. In: (Eds. Du Toit, JT, Rogers, KH & Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity.* Washington, DC, USA: Island Press: 3–21.

Giannecchini, M., Twine, W. & Vogel, C. (2007) Land-cover change and human-environment interactions in a rural cultural landscape in South Africa. *The Geographical Journal* **173**: 26–42.

Gillson, L. (2004) Evidence of hierarchical patch dynamics in an east African savanna? *Landscape Ecology* **19**: 883–894.

Helm, C., Wilson, G., Midgley, J., Kruger, L. & Witkowski, E.T.F. (2011) Investigating the vulnerability of an African savanna tree (*Sclerocarya birrea* ssp. *caffra*) to fire and herbivory. *Austral Ecology* (in press). doi: 10.1111/j.1442-9993.2010.02232.x

Higgins, SI, Shackleton, CM and Robinson, ER (1999). Changes in woody community structure and composition under contrasting landuse systems in a semi-arid savanna, South Africa. *Journal of Biogeography* **26**: 619-627.

Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W.., Valladares, F., Vila, M., Zamora, R. & Zobel, M. (2006) Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography* **15**: 1–7.

ITT Vis (2009) ENVI, ITT Visual Information Solutions. Boulder, Colorado, USA: ITT Corporation

Joppa, L.N., Loarie, S.R. & Pimm, S.L. (2009) On population growth near protected areas. *PLoS ONE* **4**: e4279.

Kaschula, S.A., Twine, W.E. & Scholes, M.C. (2005) Coppice harvesting of fuelwood species on a South African common: utilizing scientific and indigenous knowledge in community based natural resource management. *Human Ecology* **33**: 387–418.

Leps, J. & Smilauer, P. (2003) *Multivariate analysis of ecological data using CANOCO*. Cambridge, UK: Cambridge University Press.

Levick, S.R., Asner, G.P., Kennedy-Bowdoin, T. & Knapp, D.E. (2009) The relative influence of fire and herbivory on savanna three-dimensional vegetation structure. *Biological Conservation* **142**: 1693–1700.

Luoga, E.J., Witkowski, E.T.F & Balkwill, K. (2005) Land cover and use changes in relation to the institutional framework and tenure of land and resources in eastern Tanzania Miombo woodlands. *Environment, Development and Sustainability* **7**: 71–93.

Lykke, A.M. (1998) Assessment of species composition change in savanna vegetation by means of woody plants' size class distributions and local information. *Biodiversity and Conservation* **7**: 1261–1275.

Manning, A.D., Fischer, J. & Lindenmayer, D.B. (2006) Scattered trees are keystone structures: implications for conservation. *Biological Conservation* **132**: 311–321.

Mwavu, E.N. & Witkowski, E.T.F. (2009) Population structure and regeneration of multiple-use tree species in a semi-deciduous African tropical rainforest: implications for primate conservation. *Forest Ecology and Management* **258**: 840–849.

Neke, K.S., Owen-Smith, N. & Witkowski, E.T.F. (2006) Comparative resprouting response of savanna woody plant species following harvesting: the value of persistence. *Forest Ecology and Management* **232**: 114–123.

Pollard, S., Shackleton, C. & Curruthers, J. (2003) Beyond the fence: people and the lowveld landscape. In: (Eds. Du Toit, JT, Rogers, KH & Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*. Washington, DC, USA: Island Press: 422-446.

Pysek, P. & Leps, J. (1991) Response of a weed community to nitrogen fertilization: a multivariate analysis. *Journal of Vegetation Science* **2**: 237–244.

Rutherford, M.., Mucina, L., Lotter, M.C., Bredenkamp, G.J., Smit, J.H.L., Scott-Shaw, C.R., Hoare, D.B., Goodman, P.S., Bezuidenhout, H., Scott, L., Ellis, F., Powrie, L.W., Siebert, F., Mostert, T.H., Henning, B.J., Venter, C.E., Camp, K.G.T., Siebert, S.J., Matthews, W.S., Burrows, J.E., Dobson, L.N., Schmidt, E., Winter, P.J.D., Ward, R.A., Williamson, S. & Hurter, P.J. (2006) Savanna biome. In: *The Vegetation of South Africa, Lesotho and Swaziland,* ed. L. Mucina & M.C. Rutherford, pp. 439–539. Strelitzia 19. Pretoria, South Africa: South African National Biodiversity Institute.

Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux, J., Higgins, S.I., Xavier, Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G., Caylor, K.K., Coughenour, M.B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C., Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K., Prins, H., Rigrose, S., Sea, W., Tews, J., Worden, J. & Zambatis, N. (2005) Determinants of woody cover in African savannas. *Nature* **438**: 8–11.

Scholes, R.J. & Walker, B.H. (1993) *An African Savanna. Synthesis of the Nylsvley Study*. Cambridge, UK: Cambridge University Press.

Scholes, R.J. & Archer, S.R. (1997) Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics* **28**: 517–544.

Scholes, R.J. (2009) Syndromes of dryland degradation in southern Africa. *African Journal of Range and Forest Science* **26**: 113–125.

Shackleton, C.M., Griffin, N.J., Banks, D.I., Mavrandonis, J.M. & Shackleton, S.E. (1994) Community structure and species composition along a disturbance gradient in a communally managed South African savanna. *Vegetatio* **115**: 157–167.

Shackleton, C.M., Shackleton, S.E., Buiten, E. & Bird, N. (2007) The importance of dry woodlands and forests in rural livelihoods and poverty alleviation in South Africa. *Forest Policy and Economics* **9**: 558–577.

Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E. & Jacobson, J. (2010) Effects of fire on woody vegetation structure in African savanna. *Ecological Applications* **20:** 1865–1875.

ter Braak, C.J.F. & Smilauer, P. (2002) *CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (Version 4.5)*. Ithaca, New York, USA: Microcomputer Power.

Twine, W., Moshe, D., Netshiluvhi, T. & Siphugu, V. (2003) Consumption and direct-use values of savanna bio-resources used by rural households in Mametja, a semi-arid area of Limpopo province, South Africa. *South African Journal of Science* **99**: 467–473.

Twine, W.C. (2005) Socio-economic transitions influence vegetation change in the communal rangelands of the South African lowveld. *African Journal of Range and Forest Science* **22**: 93–99.

UNESCO (1996) *Biosphere Reserves: The Seville Strategy and the Statutory Framework of the World Network*. Paris, France: UNESCO.

van Aardt, J.A.N., Wynne, R.H. & Oderwald, R.G. (2006) Forest volume and biomass estimation using small-footprint LiDAR-distributional parameters on a per-segment basis. *Forest Science* **52**: 636–649.

Venter, F.J., Scholes, R.J. & Eckhardt, H.C. (2003) The abiotic template and its associated vegetation pattern. In: (Eds. Du Toit, JT, Rogers, KH & Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity.* Washington, DC, USA: Island Press: 83-129

Wehr, A. & Lohr, U. (1999) Airborne laser scanning - an introduction and overview. *ISPRS Journal of Photogrammetry & Remote Sensing* **54**: 68–82.

Wessels, K.J., Mathieu, R., Erasmus, B.F.N., Asner, G.P., Smit, I.P.J., van Aardt, J., Main, R., Fisher, J., Marais, W., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R. & Jacobson, J. (2011) Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* **261**: 19–29.

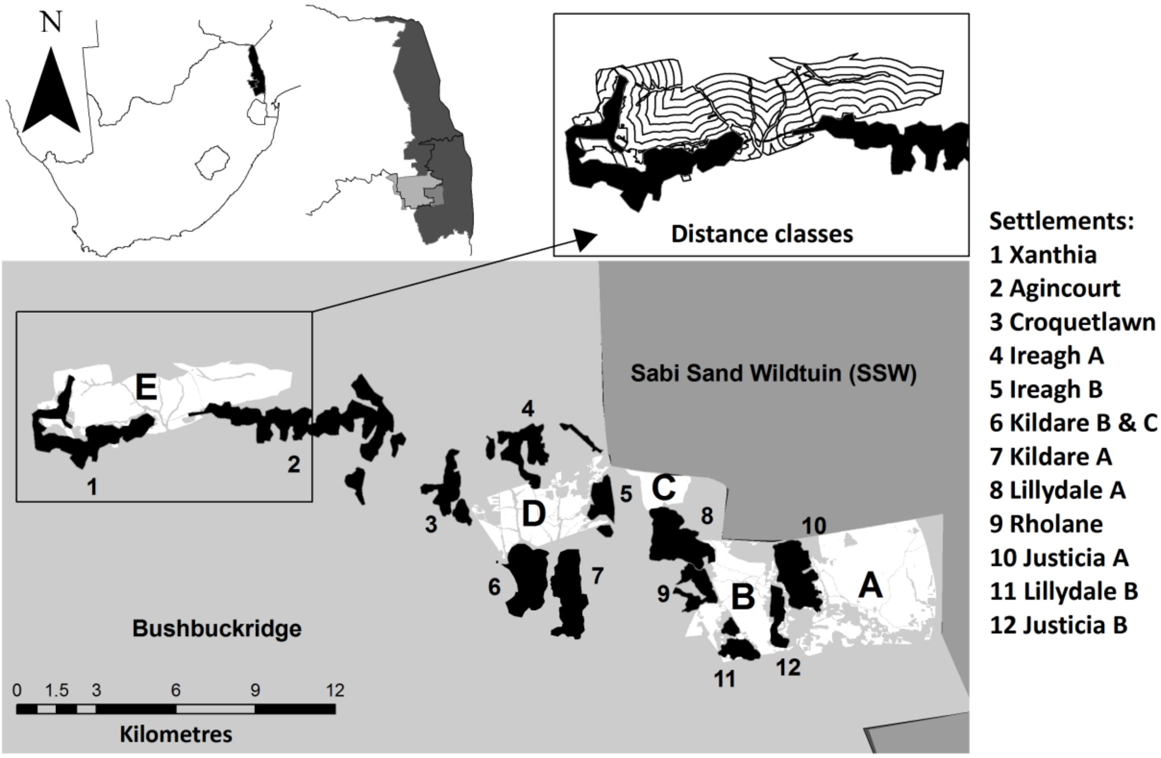
Wilson, B.G. & Witkowski, E.T.F. (2003) Seed banks, bark thickness and change in age and size structure (1978–1999) of the African savanna tree, *Burkea africana*. *Plant Ecology* **167**: 151–162.

Witkowski, E.T.F. & O'Connor, T.G. (1996) Topo-edaphic, floristic and physiognomic gradients of woody plants in a semi-arid African savanna woodland. *Vegetatio* **124**: 9–23.

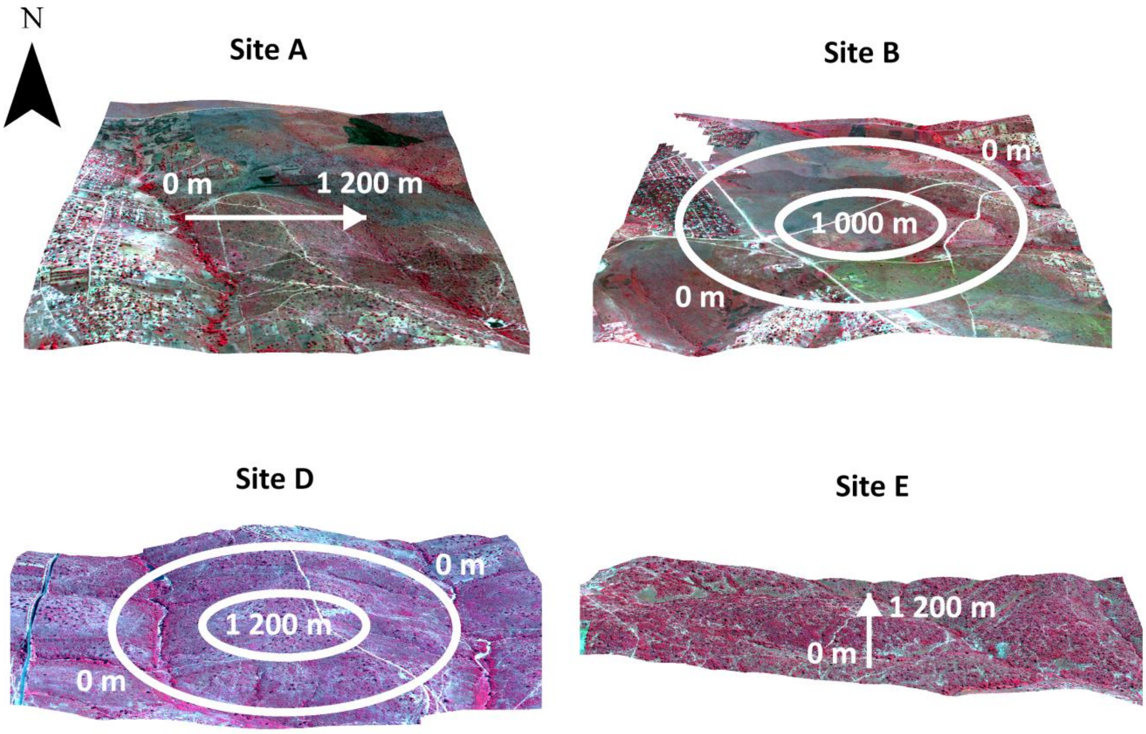
Zar, J.H. (1999) *Biostatistical Analysis*. Fourth edition. Upper Saddle River, USA: Pearson Education.

3.7 Appendix

* As it appears in Journals.cambridge.org/enc



Appendix 3.1: Location of study sites within Bushbuckridge Municipality, Mpumalanga Province, South Africa. Settlements are shown in black, areas of communal rangelands with LiDAR data appear in white (sites A–E). Left inset: map of South Africa, showing Kruger National Park with Bushbuckridge highlighted; right inset: an example of distance classes of 200 m radiating from settlements for site E.



Appendix 3.2: Colour infra-red (CIR) images draped over the digital elevation model for sites A, B, D and E. CIR imagery was not available for site C. For sites A and E, distance classes extend in a linear progression away from the settlement, while for sites B and D, distances classes converge at a central point as the rangelands are surrounded by settlements.

Appendix 3.3: Settlement data for 2008, except Croquetlawn (where date derive from 2005), and per cent change since 1993 (for Croquetlawn since 1992). In 1992, 468 Mozambicans lived in Croquetlawn, in 2005, 43 Mozambicans lived in Croquetlawn (a –90.8% change). Croquetlawn years = 1992, 2003, 2004 and 2005. *Information included in this table was obtained from MRC/Wits Rural Public Health and Health Transition Research Unit (Agincourt).

Settlement	*Households*		*Population*		*Male*		*Female*		*Children under 5*		*5–19 year olds*	
	n	*% change since 1993*	*n*	*% change since 1993*	*n*	*% change since 1993*	*n*	*% change since 1993*	*n*	*% change since 1993*	*n*	*% change since 1993*
Xanthia	760	23.6	4180	8.3	2008	7.2	2172	9.3	494	0.6	1330	-15.9
Agincourt	1068	10.1	6564	3.1	3193	2.7	3371	3.4	739	-24.1	2199	-11.9
Ireagh A	622	20.3	3733	5.7	1885	11.9	1848	0.1	433	-27.1	1300	-10.7
Ireagh B	386	0.8	2411	3.8	1122	6.1	1289	1.8	275	-33.1	865	-1.7
Kildare A	835	19.1	4747	9.8	2225	6.8	2522	12.5	510	-19.9	1555	-7.9
Kildare B	973	24.6	5864	11.8	2826	14.8	3038	9.4	694	-28.7	2268	12.5
Lillydale A	1520	17.9	8828	7.3	4255	8.5	4573	6.3	984	-28.3	3136	-0.3
Lillydale B	426	67.1	2348	61.7	1122	65.5	1226	58.4	281	1.1	895	61.3
Justicia A	1190	27.5	6329	1.9	2330	-20	3399	3	752	-32.1	2267	-4.2
Croquetlawn	516	16	3089	12.1	1524	13	1564	11.2	316	-29.8	1065	4.3

Chapter 4 : Savanna woody vegetation classification – now in 3D**

Jolene T. Fisher, Barend F.N. Erasmus, Ed T.F. Witkowski, Jan van Aardt, Konrad J. Wessels and Gregory P. Asner

Fisher, J.T. (corresponding author jolenefisher@gmail.com) Centre for African Ecology; Restoration and Conservation Biology Research Group, School of Animal, Plant & Environmental Sciences, University of the Witwatersrand, WITS, Johannesburg, 2050, South Africa

Erasmus, B.F.N. (Barend.Erasmus@wits.ac.za): Centre for African Ecology, School of Animal, Plant & Environmental Sciences, University of the Witwatersrand, WITS, Johannesburg, 2050, South Africa

Witkowski, E.T.F. (Ed.Witkowski@wits.ac.za): Restoration and Conservation Biology Research Group, School of Animal, Plant & Environmental Sciences, University of the Witwatersrand, WITS, Johannesburg, 2050, South Africa

Van Aardt, J. (vanaardt@cis.rit.edu): Chester F. Carlson Center for Imaging Science, Rochester Institute of Technology: 54 Lomb Memorial Drive, Rochester, New York, 14623, USA

Wessels, K.J. (kwessels@csir.co.za): Remote Sensing Research Unit, Council for Scientific and Industrial Research (CSIR)-Meraka Institute, P.O. Box 395, Pretoria, 0001, South Africa

Asner, G.P. (gpa@standford.edu): Department of Global Ecology, Carnegie Institution for Science, 260 Panama Street, Stanford, CA 94305, USA

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4.1 Abstract

Question: The co-existence of woody plants and grasses characterize savannas, with the horizontal and vertical spatial arrangement of trees creating a heterogeneous biotic environment. To understand the influence of biogeophysical drivers on the spatial patterns of 3-D structure of woody vegetation, these patterns need to be explained over large areas to capture the context. Is there a spatially explicit, ecologically meaningful way to capture the patterns and context of 3-D woody vegetation structure?

Location: Classification development and testing sites: Landscapes in Bushbuckridge Municipality, Sabi Sand Wildtuin, and Kruger National Park, Mpumalanga province, north east South Africa.

Methods: The aforementioned structural classification approach requires appropriate 3D and spatially explicit remote sensing data. A LiDAR-based canopy height model (CHM) and volumetric pixel (voxel) data from the Carnegie Airborne Observatory-Alpha system were used to create the structural classification. Firstly, we segmented the CHM images using multi-threshold and multi-resolution image segmentation techniques, and classified the image segments into four height classes, namely shrub (1-3 m), low tree (3-6 m), high tree (6-10 m) or tall tree (>10 m). A hierarchical *a priori* approach was used to develop classification criteria. The following metrics were calculated for 0.25 ha grid cells based on the cover and spatial arrangement of the four height classes: Canopy Cover, Sub-canopy Cover, Canopy Layers, Simpson's Diversity Index and Cohesion. Top of canopy vegetation was classified using each metric at the 0.25 ha scale, with canopy cover being the primary classification metric. Subsequently, individual layers identified within the canopy were classified using the voxel data. We use a code system for describing classes to ensure standardization between different regions; a more traditional naming system may be used in addition for interpretation.

Conclusion: This system provides a more comprehensive classification of the horizontal and vertical structural diversity of savannas compared to the traditional vegetation classification systems. The description of multi-layers within the canopy should allow for a sensitive change detection method. The classification can be used in many current focus areas,

including habitat suitability mapping for biodiversity conservation, strategic adaptive management and monitoring land cover change.

Key words

Carnegie Airborne Observatory; heterogeneity; LiDAR; object-based image classification; vegetation structure; woody vegetation

Abbreviations

Light Detection and Ranging = (LiDAR); Canopy Cover = (CC); Sub-canopy Cover = (SCC); Canopy Layers = (CL); Simpson's Diversity Index = (SDI); Land Cover Classification System = (LCCS); National Land Cover = (NLC); International Geosphere-Biosphere Project = (IGBP); Global Land Cover Classification = (GLCC)

4.2 Introduction

Woody vegetation classification maps are inherently two-dimensional based on the remotely sensed data used. The position, extent and connectivity of the woody vegetation layer are captured; however, the vertical arrangement of woody plant components is not visible from standard two-dimensional (2D) passive remotely sensed data. Recording the three dimensional (3D) structure of vegetation in the field is time consuming and often not feasible, and is not possible at all with standard multispectral images. However, with the development of LiDAR (Light Detection and Ranging), it is now possible to objectively and repeatedly collect measurements of vertical structure over large areas (Goatley & Bellwood 2011). Savannas, defined by a continuous herbaceous layer with a discontinuous woody layer, possess a complex woody architecture best described in three dimensions. This complex vertical and horizontal structure provides habitat for a broad range of vertebrates and invertebrates; and has implications for conservation and natural resource provision as the importance of the vertical dimension in habitat heterogeneity across large extents is not well understood (Tews et al. 2004; Hall et al. 2011).

LiDAR has previously not been used extensively in structurally heterogeneous African savannas; however, it has recently been used to successfully map savanna biomass (Colgan et al. 2012), investigate the effects of fire and herbivory on vegetation structure (Asner et al. 2009, Levick et al. 2009, Smit et al. 2010, Asner & Levick 2012), explore patterns of structure around communal rural villages (Wessels et al. 2011; Fisher et al. 2012) and map riparian condition indicators (Johansen et al. 2010). LiDAR produces large amounts of data, so it is often necessary to derive summary statistics of canopy height and inferred estimates of diameter at breast height and aboveground biomass for ecological applications (Lefsky et al. 2002 a, b; Naesset 2002; Blaschke et al. 2004; Anderson et al. 2006). While this may provide acceptable measurements of structural attributes in areas with homogeneous vegetation, these statistics do not properly describe the variation especially at large extents in heterogeneous landscapes like African savannas, which have more complex vertical structure with high spatial variability. A method to remedy this over-simplification of LiDAR data, while still reducing data volume and complexity and providing relevant ecological information, would be to use it in a spatially explicit 3D classification. Small footprint LiDAR can address the deficiencies of conventional 2D savanna classifications (Appendix S1) by providing a 3D component (plant height as well as layers within the canopy), without the need for extensive field work at a scale relevant to capturing the heterogeneity of savannas.

Such an approach remains challenging, since the spatio-temporal heterogeneity in the horizontal and vertical structure of woody plants in savannas adds complexity when studying pattern and process (Levick & Rogers 2011). Complex patterns in vegetation 3D structure cannot be effectively characterized by a single measure, as they are driven by climate, rainfall, geology, topography, fire and herbivory (Scholes & Archer 1997, Sankaran et al. 2008), which vary across space and time (Levick & Rogers 2011). The resultant patterns in savannas are not only determined by an individual component, but more importantly, by the interactions between them (Pickett et al. 2003). The spatial context of woody vegetation in the landscape is therefore necessary for appropriate application of the knowledge to management and conservation (Levick & Rogers 2011).

Woody vegetation structure refers to the position, extent, quantity, type and connectivity of the aboveground components of woody vegetation (Lefsky et al. 2002 a) in three dimensions. Therefore, each of these characteristics needs to be measured in order to adequately represent savanna vegetation. Although theoretical methods such as volumetric neutral models capture 3D spatial structure of vegetation (Kirkpatrick & Weishampel 2005), no ecologically-based classification currently exists that captures this type of heterogeneity in savanna vegetation structure. Five land cover classifications include a savanna component: Structural Classification of Edwards (1983), the Land Cover Classification System (LCCS) (Di Gregorio & Jansen 2000), the National Land Cover (NLC) for South Africa (Thompson 1996), International Geosphere-Biosphere Project (IGBP) (Loveland & Belward 1997) and the Global Land Cover Classification (GLCC) (Hansen et al. 2000) (Appendix 4.1). Except for Edwards and LCCS, which are acknowledged for including structural measures, the other classifications (NLC, GLCC and IGBP) do not account for a shrub layer interspersed within the tree layer. The inclusion of the shrub layer is essential as increases in shrubs may indicate bush encroachment with implications for ecosystem function and biodiversity (Eldridge et al. 2011; Buitenwerf et al. 2012). In addition, the finer scale spatial arrangement of the woody layer as a whole, and of each cover type (tree/shrub), is not taken into consideration in any of these classifications.

It can be argued that a spatial metric such as "cohesion" would give information on the extent, subdivision and contagion of cover classes, indicating habitat suitability (Ishii et al. 2004). At a landscape scale, a low cohesion value could indicate fragmentation, which affects ecological flows within the landscape (McGarigal et al. 2002). Furthermore, habitat suitability is not only determined by the cohesion or fragmentation of the plant canopies in the area, but also the diversity of vegetation structure. Diversity indices such as Shannon-Weiner and Simpson's are commonly used to characterise species diversity (Magurran 2004) and can be applied to structural data (MacArthur & MacArthur 1961). However, diversity indices have not been used in existing land cover classifications, possibly because different height classes are not identified using conventional multispectral imagery.

Remotely sensed vegetation structural classifications have evolved over the years as image resolution improved, making it possible to now include data to capture the third dimension. There is much benefit from such a classification in savannas, ranging from mapping natural resource availability for ecosystem services, to improved biomass and thus carbon estimates, and enhanced habitat modelling for biodiversity conservation (Ishii et al. 2004; Hall et al. 2011). The aim of the study is therefore to develop an ecologically meaningful savanna classification that captures both the vertical and horizontal heterogeneity of the woody plant canopy, using novel 3D remote sensing approaches.

4.3 Methods

4.3.1 Light Detection and Ranging (LiDAR)

Woody vegetation was mapped across approximately 35 000 ha of semi-arid savanna in South Africa in April 2008 with the Carnegie Airborne Observatory Alpha System (CAO-Alpha; http://cao.ciw.edu). The CAO-Alpha combined both imaging spectroscopy (hyperspectral imaging) and LiDAR technologies to study ecosystems at the regional scale (Asner et al. 2007). The spectrometer was co-mounted with the LiDAR sensor that acquires both waveform- and discrete-return data; however, only discrete-return data were used in this study.

The integrated Global Positioning System-Inertial Motion Unit sub-system in the CAO provides the position and orientation of the sensors in 3D, while the CAO algorithms ensure that data inputs from both the spectrometer and the LiDAR system are co-located and precisely projected to ensure geographically aligned output (Asner et al. 2007). The LiDAR data were collected at 2000 m above ground level with a laser pulse repetition frequency of 50 kHz, laser spot spacing of 1.12 m, and up to four returns per pulse. These specifications are considered to be a minimum requirement for the classification to remain consistent across data sets.

LiDAR produces a 3D xyz point cloud. A digital elevation model (DEM – interpolated from the LiDAR ground returns) was subtracted from the digital surface model (DSM – LiDAR first return interpolation) to produce the canopy height model (CHM, 1.12 m horizontal pixel resolution). The point cloud frequency values were binned into volumetric pixels (voxels) of 5x5x1 m (X, Y, Z) for 3D vegetation analysis. The value in the voxel represents the frequency of LiDAR returns/25m^3 relative to the sum of returns for the entire 5x5 m vertical column and is used to assess sub-canopy vegetation. Ground validation of vegetation heights was conducted concurrent to the aerial campaign in 2008. It should be noted that trees less than 2 m tall may be underestimated (Wessels et al. 2011) due to the laser pulse not hitting their small and often sparse canopies.

4.3.2 Test data – site description

The classification was created and tested on sites in communal rangelands in Bushbuckridge Municipality (BBR), and two adjacent protected areas, Sabi Sand Wildtuin (SSW; a private game reserve) and Kruger National Park (KNP), in Mpumalanga Province, north-eastern South Africa (Fig. 4.1). Due to the mosaic of land management techniques and land-use intensities, spatial heterogeneity is high in these areas. This property makes them appropriate sites on which to develop the classification as they are representative of a wide variety of situations present in global savannas. The sites form a west to east gradient in climate and topography. Mean annual precipitation over the study area ranges from >1 200 mm in the west, and gradually reduces to an average of 550 mm in the east, with mean annual temperature of 22˚C. The geology of the region is dominated by granite, with Timbavati gabbro intrusions. All sites fall within three vegetation units of the savanna biome: Granite lowveld (dominant), Gabbro grassy bushveld and Legogote sour bushveld (Mucina & Rutherford 2006). Typical woody plant species in the granite lowveld include:

Terminalia sericea, *Combretum zeyheri* and *C. apiculatum* on the deep sandy toplands, and *Acacia nigrescens*, *Dichrostachys cinerea* and *Grewia bicolor* on the more clayey soils of the bottomlands. In the two other vegetation units additional common species include *Sclerocarya birrea*, *Lannea schweinfurthii*, *Ziziphus mucronata*, *Dalbergia melanoxylon*, *Peltoforum africanum* and *Pterocarpus rotundifolius*.

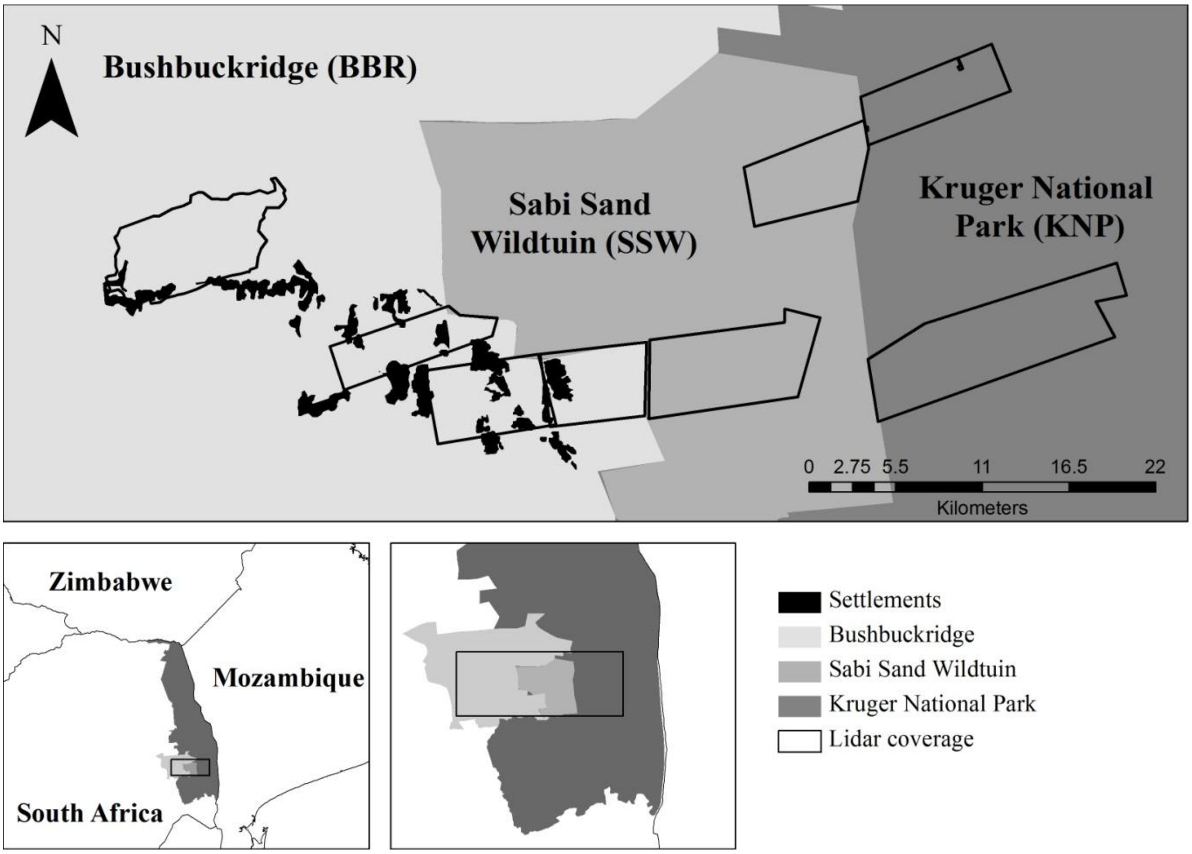


Figure 4.1: Study location – Bushbuckridge (BBR), Sabi Sand Wildtuin (SSW) and Kruger National Park (KNP) in Mpumalanga Province, South Africa.

4.3.3 Conceptual approach of the classification

Our classification approach was based on a compilation of criteria used in the National Vegetation Classification System (FGDC 1997) and the LCCS (Di Gregorio & Jansen 2000). The classification must furthermore adhere to the following:

- have ecologically meaningful metrics
- be based on a sound scientific approach that is a logical progression from historical methods and can be repeated
- meet the needs of a variety of users

- provide a common reference system, and facilitate comparisons between classes used in different classifications
- be a flexible system, which can be used at different scales and at different levels of detail allowing cross-reference of local and regional features with continental maps without loss of information
- be hierarchically organized such that it can be applied at multiple scales
- identify spatial units that are appropriately scaled to meet objectives for biodiversity conservation, as well as resource and ecosystem management needs
- be a flexible system which is open ended such that it will allow for additions, modifications and continuous refinement
- be a well documented system that is easy to execute

We used a hierarchical *a priori* approach to develop the classification criteria. When conducting a global classification it is often easier to use a data driven (*a posteriori*) classification in order to reduce the amount of user interaction as no prior or local knowledge of the area is a pre-requisite (Achard et al. 2001). However, such methods rely on spectral separability being equated to ecologically meaningful classes which is not necessarily the case, especially when performing a structural classification. The wealth of existing information about savannas ensures that we can define *a priori* classes that adequately represent the 3D nature of savannas (Scholes & Walker 1993; Scholes & Archer 1997; Sankaran et al. 2008). The nature of an *a priori* classification system is such that category definitions are independent of (i) the area mapped, (ii) the data properties and (iii) the mapping techniques, thus making the classification more robust and universally applicable. The hierarchy can be described as a compositional containment hierarchy, where no one metric is more important than the others; however, each by itself is meaningless without the context provided by other metrics and size classes (Parsons 2002).

The classification was based on two levels (Fig. 4.2). The first level in the hierarchy classifies the top of canopy vegetation based on canopy cover, percent canopy layers present (derived from the voxel product), cohesion and diversity (Fig. 4.2). Top of canopy vegetation includes all vegetation captured by the CHM and does not include understory vegetation. The second level of the classification categorizes each height class present, including both vegetation that appears on the CHM and understory (sub-canopy) vegetation (Fig. 4.3).

There are four possible canopy layers: shrub (1-3 m), low tree (3-6 m), high tree (6-10 m) and tall tree (>10 m). The cover and cohesion of each layer is described, starting with shrubs and ending with tall trees. If a layer is not present it is excluded from the description (Fig. 4.2, 4.4). Level I of the classification is a top-down classification, while Level II is a bottom-up classification (Fig. 4.2).

Traditional land cover classifications place the emphasis on the name of the class; however this may lead to confusion as one land cover type may be called a different name under two classifications systems (Appendix 4.1; Fig. 4.4). We therefore adopted the technique used in the LCCS whereby a code is used to define a class. This makes the classification comparable between countries which might use alternative names for a vegetation type. In addition, the code system is more robust when investigating change, as the specific metric of the class that is changing, for example, the level of aggregation, is identified.

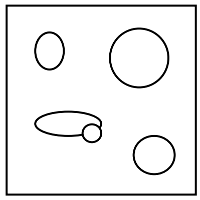
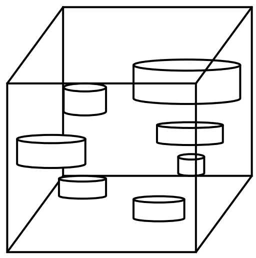
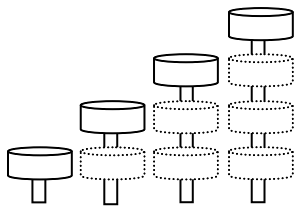
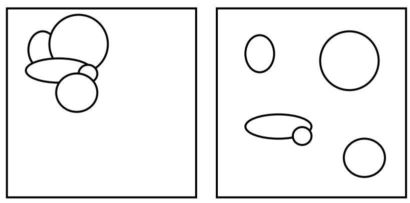
4.3.4 Classification development

The building blocks of the classification are individual trees and shrubs. In accordance with Edwards (1983), four growth forms were classified in agreement with the canopy layers: shrub (1-3 m), low tree (3-6 m), high tree (6-10 m) and tall tree (10+ m). Class intervals at the lower end were inclusive and exclusive at the upper end (i.e. 1-3 m height class: 1 m ≤ trees <3 m). The selected height categories are ecologically meaningful and relate to fire, herbivory and human use. Vegetation <3 m in height are in the fire trap (Govender et al. 2006; Smit et al. 2010) and heavily browsed by small- to medium-size herbivores; vegetation in the 3-6 m height class are targeted by mega-herbivores (elephant (*Loxodonta africana*) and giraffe (*Giraffa camelopardalis*)); finally, vegetation >6 m is less influenced by fire and herbivory (Owen-Smith 1988; Scholes & Walker 1993; Birkett & Stevens-Wood 2005; Neke et al. 2006). People are known to harvest wood for fuel and poles, predominantly from <3 m height class (Neke et al. 2006), although in miombo woodlands where wood is used for charcoal production the entire tree is often harvested (Luoga et al. 2000). Trees >10 m are important in the savanna landscape, providing shade, reducing evapotranspiration of the below canopy herbaceous layer, and increasing local nutrients accumulated close to the root systems (Belsky 1994; Manning et al. 2006; Treydte et al. 2009), thereby creating high quality grazing which may attract greater abundances of ungulates.

Individual vegetation units were identified on the CHM and voxel layers using an object based image analysis (OBIA) in eCognition Developer v8.7 (Trimble Geospatial Imaging, Munich, Germany, 2011). The CHMs were treated with a 3x3 low pass filter prior to segmentation to remove noise. A multi-threshold segmentation was performed using the following height thresholds: 0.25; 0.5; 1; 1.5; 2; 3; 4; 5; 6; 7; 8; 9 and 10 m. These thresholds allowed for hierarchical segmentation aggregation from a very fine sub-canopy scale (Appendix 4.2a) to individual tree canopies (Appendix 4.2e). Image objects with a mean and/or maximum vegetation height <1m were classified as 'background' and removed from further classifications. After initial segmentation, each object was classified into one of the four height classes (1-3 m, 3-6 m, 6-10 m, >10 m) based on maximum height in each image object. As a result of height uniformity in large clumps of trees, and inter-canopy variation in large trees, image objects in areas with high woody cover were not adequately segmented and large trees were often too finely segmented with canopies consisting of multiple segments (Appendix 4.2a); however, coarser height thresholds did not identify smaller, often isolated tree canopies. These fine image objects were therefore merged according to their height classification; and a subsequent multi-resolution segmentation was performed on these merged image objects (Segmentation parameters used in eCognition v8.7 (2011): Scale parameter = 12 (determines size of segmentation in relation to the landscape), Shape weighting = 0.5 (0 = irregular shape; 1 = regular shape) and Compactness weighting = 0.9 (0 = high perimeter: area ratio; 1 = low perimeter: area ratio)) creating a second segmentation layer. The resulting image objects, which contained finer detail in areas of dense woody cover, were then reclassified into the four vegetation height classes based on maximum height in an image object.

The 3D structural classification is such that once image objects of individual vegetation units have been created, the classification can be carried out at a variety of user specified scales according to user need. The minimum grid size for the classification was determined using semivariograms calculated in ENVI 4.7 (ITT Vis [ITT Visual Information Systems] 2009) on the CHM and it was established that the variogram sill occurred at a range of 50 m, translating to a grid size of 0.25 ha (Wessels et al. 2011). Metrics were calculated for each grid cell using the four vegetation height classes exported from eCognition. The following metrics were calculated in ArcMap 10.0 (ESRI 2010, Redlands, USA, www.esri.com): Canopy Cover (CC), Sub-canopy Cover (SCC), Canopy Layers (CL), Cohesion and Simpson's Diversity Index (SDI) (Table 1). CC is classified into nine cover classes which were chosen because of overlap with existing classifications (Appendix S4.1, Fig. 4.2).

Table 4.1: Description of metrics used to construct a three-dimensional structural classification of savanna woody vegetation.

Metric	Description and schematic representation
Canopy cover (CC) Units: % Range: 0-100	 CC refers to the vertical projection of the tree/shrub crown onto the ground, given as a percent of the area. Cover is measured for the overall woody cover (all height classes). The dominant cover class is measured from the CC metric as the class that constitutes $\geq$50% of the total woody canopy cover. Canopy cover is measured from the top of canopy objects produced in eCognition v8.7 (2011) based on the Canopy Height Model (CHM) LiDAR product.
Sub-canopy cover (SCC) Units: % Range: 0-100	 SCC is a measurement of the percent cover of each height class (1-3 m, 3-6 m, 6-10 m and >10 m) as it occurs below the dominant cover classes. That is, a tree of >10 m may obscure vegetation below. It is measured as a percent of the grid cell for each height class. Each individual SCC measurement for each height class will fall within the range of 0-100%. SCC is only used to calculate Cohesion for each individual height class, and is used as the Cover metric in the description of Level II – Plant Layers of the classification. SCC is measured from the volumetric pixel (voxel) data.
Canopy Layers (CL) Units: % Range: 0-100	 CL is a measure of the percent of canopy layers present within the canopy. This metric quantifies the thickness of the woody layer. An increase in CL over time might be an indication of bush encroachment. CL is calculated for the entire grid cell of interest using the SCC product (a presence /absence measure – indicated by the solid cylinders and dashed cylinders respectively in the figure). It is a measurement of the number of vertical canopy layers present relative to the total possible number of canopy layers (for a tree >10 m four layers are possible) available in each grid cell including the top of canopy object.
Cohesion Units: % Range: 0-100	 Cohesion is a measure of how aggregated the vegetation components (trees and shrubs) are within the designated area in the horizontal plane. Values range between 0 and 100, with 100 representing greater aggregation or clumping. Due to the mix of grass and woody components defining savannas, spatial arrangement is an important consideration with

Metric	Description and schematic representation
	implications for habitat suitability and utilisation. At a fine scale cohesion has implications for organisms' movement and use of the landscape, while at a landscape scale cohesion gives an indication of edge effects (Fischer & Lindenmayer 2007). An increase in cohesion of one or more vertical height classes may indicate increased bushiness. Often cohesion is inversely proportional to interspersion, with a high cohesion value indicating a low level of interspersion of cover types. Cohesion was measured for both the entire woody layer within the grassland matrix (using the CC metric), as well as for each height class (using the SCC metric) to measure the cohesion of each height class. The following equation was used to calculate cohesion (McGarigal *et al*. 2002): $$COHESION = \left[1 - \frac{\sum_{j=1}^{n} p_{ij}}{\sum_{j=1}^{n} p_{ij}\sqrt{a_{ij}}}\right]\left[1 - \frac{1}{\sqrt{A}}\right]^{-1} \times (100)$$ Where: P_{ij} = perimeter of patch *ij* (either woody vegetation, or each height layer) in terms of number of pixels a_{ij} = area of patch *ij* in terms of pixels A = total number of pixels in the landscape Values were then corrected according to percent area covered.
Simpson's Diversity Index (SDI) Units: % Range: 0-100	SDI takes into account both the number of height classes present, and their relative abundance. SDI is a measure of structural diversity, the higher the value the greater the likelihood that two objects within a grid cell are different (i.e. mixture of shrubs, low trees, high trees and tall trees; i.e. greater diversity). The metric is calculated from the tree and shrub objects layer based on the CHM using the following equation (McGarigal *et al*. 2002): $$SDI = (1 - \sum\left(\frac{n_i(n_i - 1)}{N(N-1)}\right)) \times 100$$ Where: n_i = number of individuals of height class *i* N = total number of individuals (trees identified using object-based image analysis) of all height classes

An example of using the classification is as follows: A grid cell contains one tall tree (>10m), one high tree (6-10 m), one low tree (3-6 m) and one shrub (1-3 m) (Fig. 4.3). The tall tree has three layers within its canopy, representing vegetation in the >10 m, 6-10 m, and 1-3 m height classes (Fig. 4.3). Top of canopy vegetation (CC) covers 37% of the grid cell, percent of canopy layers present (CL) is 60% (i.e. 60% of the possible sub-canopy layers within the 0.25 ha grid cell are present, Fig. 4.3), cohesion equals 57, SDI is 87, and all four height classes are present, therefore we use the code E4. Since no height class covers >50% of the total percent cover, we use the code E40. The classification of Level I in the hierarchy is therefore A6B60C57D87E40 (moderately covered, evenly dispersed, diverse savanna, with understory vegetation, Table 4.2). Sub-canopy layers are described in height order from shrub to tall tree. Subsequent layers are therefore given the following codes: e1a3c15, e2a2c0, e3a3c15 and e4a2c0. Lowercase letters are used to indicate layers within the sub-canopy. The resulting full code for the grid cell is thus: A6B60C57D87E40 e1a3c15 e2a2c0 e3a3c15 e4a2c0.

SAVANNA WOODY VEGETATION CLASSIFICATION

CLASSIFICATION HIERARCHY

I – Top of canopy

- A. Canopy Cover (CC)
- B. Canopy Layers (CL)
- C. Cohesion of woody layer
- D. Diversity and Evenness (SDI)
- E. Number of life forms present & dominant life form

II – Understory plant layers

Layer 1

- e. Life form
- a. Cover (SCC)
- c. Cohesion

Subsequent layers

A. Canopy Cover (CC)

Code	Canopy cover	Cover (%)
A1	Bare	0
A2	Grassland	0.1-5
A3	Sparse	5.1-10
A4	Open	10.1-20
A5	Open-Moderate	20.1-30
A6	Moderate	30.1-40
A7	Moderate-Closed	40.1-60

B. Canopy Layers (CL)

Code	Canopy layers (%)
B	$0 \leq CL \leq 100$ 0 = no vegetation present 100 = each understory layer of the canopies present is occupied by vegetation

*For naming purposes use the following 3 categories:
0: no understory, mention of understory is omitted from the name

C. Cohesion

Code	COHESION
C	$0 \leq COHESION \leq 100$ 0 = no coverage 100 = continuous coverage

*For naming purposes use the following 3 categories:
<25: evenly dispersed

D. Simpson's Diversity Index (SDI)

Code	Simpson's Diversity Index
D	$0 < SDI \leq 100$ 0 = low diversity 100= high diversity

*For naming purposes use the following 2 categories:
<50: low diversity
>50: diverse

E. Height classes

Code	Height classes	Range (m)
E1	Shrub	1-3
E2	Low tree	3.1-6
E3	High tree	6.1-10
E4	Tall tree	>10

Figure 4.2: Classification metrics and how they are combined for a savanna woody structural classification.

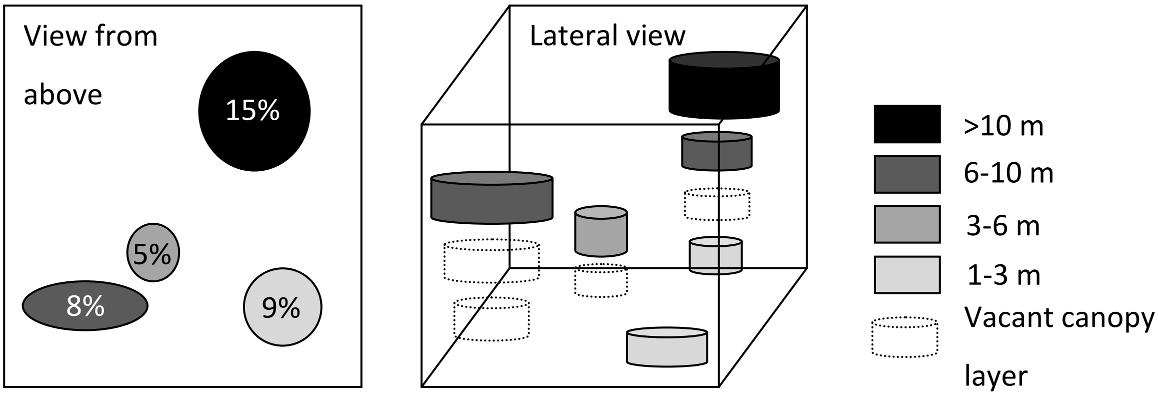
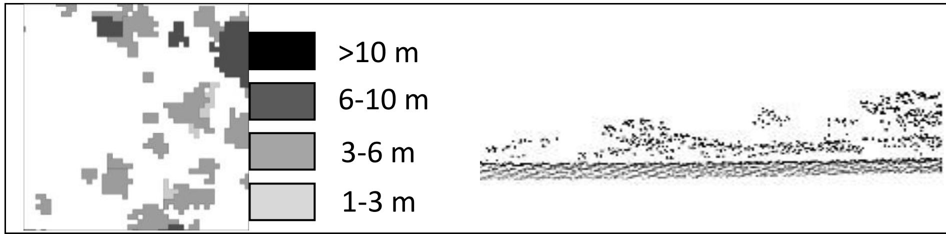


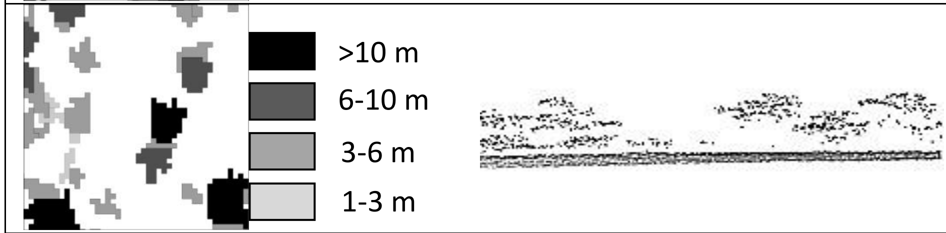
Figure 4.3: Schematic representation of savanna plants within a 0.25 ha area shown in both 2D (top of canopy vegetation – View from above) and 3D (lateral view). Vacant canopy layers are shown (4 layers) and indicate, along with filled canopy spaces (6 layers), the number of canopy layers present (6/10; 60%) in the 0.25 ha area.

Table 4.2: Examples of two areas classified using the structural classification and suggested names for each code. The breaks in the code shown below (canopy cover and volume, cohesion and diversity, etc) are suggested break points along the classification hierarchy where users may end their classification.

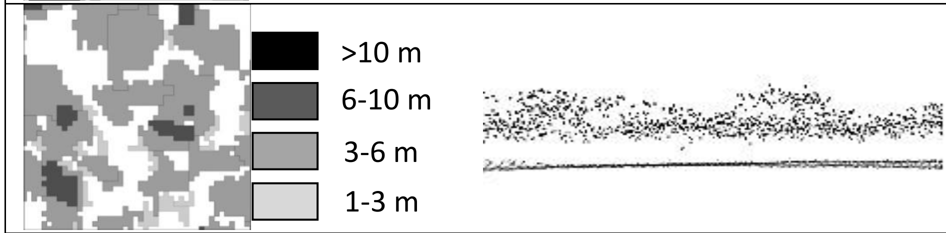
Classifier used	**Code**	**Suggested name**
Example 1 – no dominant layer		
Canopy cover & % layers	A6B60	Moderately covered savanna with understory vegetation
Cohesion and Diversity	A6B60C57D87	Moderately covered evenly dispersed diverse savanna with understory vegetation
Life forms present and dominance	A6B60C57D87E40	Moderately covered evenly dispersed diverse savanna with understory vegetation
Example 2		
Canopy cover & % layers	A4B4	Open savanna with understory vegetation
Cohesion and Diversity	A4B4C82D23	Aggregated, even open savanna with understory vegetation
Life forms present and dominance	A4B4C82D23E32	Low tree aggregated even open savanna with understory vegetation
Understory layers	A4B4C82D23E32 e1a2c12 e2a3c61	Multi-layered low tree aggregated open savanna with shrubs



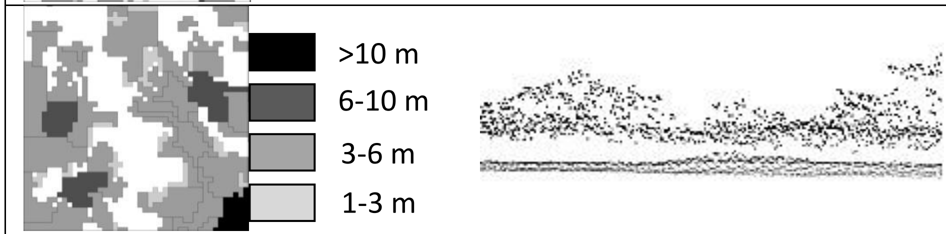
3D Classification: A5B74C27D54E42 (Low tree dominated open-moderate moderately cohesive diverse savanna with understory vegetation present)
Edwards: Short closed woodland
LCCS: Open woodland
NLC: Wooded grassland/Bushland
GLCC: Wooded grassland/shrubland
IGBP: Savanna



3D Classification: A5B54C30D60E40 (Open-moderate moderately cohesive diverse savanna with understory vegetation present)
Edwards: Tall closed woodland
LCCS: Open multi-layered woodland
NLC: Wooded grassland/Bushland
GLCC: Wooded grassland/shrubland
IGBP: Savanna



3D Classification: A8B13C64D65E32 (Low tree dominated closed moderately cohesive diverse savanna with understory vegetation present)
Edwards: Low closed woodland
LCCS: Closed woodland
NLC: Bushland
GLCC: Mixed forest
IGBP: Mixed forest



3D Classification: A8B24C67D57E42 (Low tree dominated closed moderately cohesive diverse savanna with understory vegetation present)
Edwards: Low closed woodland
LCCS: Closed woodland
NLC: Bushland
GLCC: Mixed forest
IGBP: Mixed forest

Figure 4.4: Aerial view and transects through the 3D point cloud of four 0.25 ha areas of semi-arid savanna with corresponding classifications. The height key refers to vegetation in the aerial view. Point clouds depict actual vegetation and height classes are not differentiated by colour. The corresponding classifications of the area using Edwards (1983), Land Cover Classification System (LCCS; Di Gregorio & Jansen 2000), National Land Cover Classification of South Africa (NLC; Thompson 1996), Global Land Cover Classification (GLCC; Hansen et al. 2000) and International Geosphere-Biosphere Program (IGBP; Loveland & Belward 1997).

4.4 Discussion

The 3D structural classification put forward in this paper creates a standard for comparison with existing vegetation classifications (Appendix 4.1), while at the same time incorporating novel 3D technology creating an ecologically meaningful and useful classification. Using an adaptation of the well-known LCCS code system, the structural classification can be used on LiDAR data in different countries with different naming conventions, but remain comparable. We propose a set of suggested names for classes (Fig. 4.2; Table 4.2); however, the order of the names for each metric within the full name may be modified as long as the code remains consistent. While the code may become cumbersome, it can be shortened according to user needs (i.e. only report Level I) or according to available information (Table 4.2). An intermediate option is to present the classification for the top of canopy layer and for just the dominant height class layer if one is present (Table 4.2).

While other classifications can identify changes in land cover (e.g. Edwards 1983; Di Gregorio & Jansen 2000), the change has to be considerable before being detected. Conversion from one land cover category to another through land use change is easily identified as cover is drastically altered (e.g. clear cutting or planting trees). However, modifications within one land cover category through land use intensification, especially when the changes are occurring below the top canopy (e.g. fuelwood harvesting or coppice regeneration), are difficult to detect with traditional land cover classifications (Jansen & Di Gregorio 2002). The 3D classification provides an advantage over Edwards' (1983) classification, which classifies the amount of cover of four life forms (trees, shrubs, grasses and herbs) and describes vegetation based on aerial cover of dominant life form (e.g. high closed woodland). Spatial configuration and number of layers within the canopy are not included. LCCS does provide more detail than Edwards, such as leaf type and phenology (e.g. broadleaved, deciduous), as well as information on the stratification of the canopy; however, stratification only refers to life forms that can be identified from an aerial view and does not include sub-canopy layers (Di Gregorio & Jansen 2000).

The 3D structural classification will be able to identify subtle changes in sub-canopy vegetation density and spatial arrangement before a state shift occurs by identifying changes in height class dominance, as well as changes in cover, cohesion and diversity. The

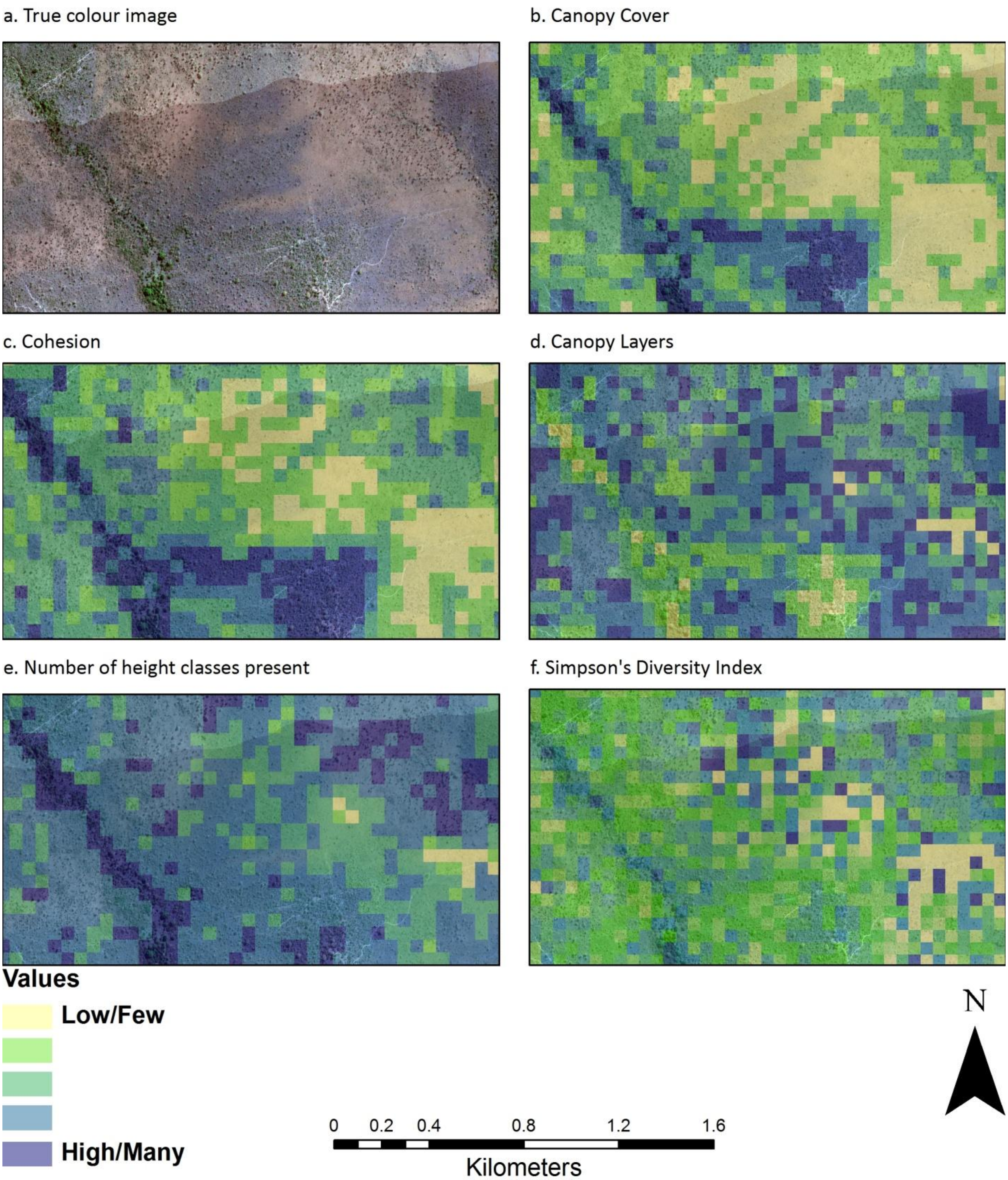


Figure 4.5: True colour image of an area in Bushbuckridge Municipality (a) Mpumalanga Province, north-west South Africa and the corresponding Level I classifications (b. Canopy cover, c. Cohesion, d. Canopy layers, e. Number of height classes present and f. Simpson's diversity index) using the 3-D woody structural classification for savannas.

metrics can be used as part of a monitoring system contributing to better management by early detection of areas of concern, such as areas with woody encroachment, loss of big trees or excessive fuelwood removal; which might not be detected with traditional 2D classifications that only use percent canopy cover as a classifier. For change detection, we would recommend using the greatest amount of detail (i.e. use the code for all layers within the canopy) to ensure greater sensitivity to identify change. Changes may also be investigated separately for each metric for ease of interpretation (Fig. 4.5, Appendix 4.3).

Areas with the same cover may have different structural compositions (Fig. 4), which will result in dissimilar functional habitats. The two areas might also contain varying assemblages of height classes, different arrangement of these height classes within the area and different canopy layers. Vegetation structural complexity has been shown to increase species richness and diversity of both small mammals and reptiles (Price et al. 2010). Birds (MacArthur & MacArthur 1961; Bergen et al. 2007; Seymour & Dean 2009), arthropods (Halaj et al. 2000), mammals (Williams et al. 2002; Lumsden & Bennet 2007), and reptiles (Smart et al. 2005) rely on fine-scale spatial niches created by complex vertical architecture present in savannas for their habitat. In addition, ungulates, both browsers and grazers, and predators show definite preferences for areas with different amounts of woody cover, ranging from grasslands to densely wooded areas (de Knegt et al. 2007; Winnie et al 2008). The structural classification provides the level of detail needed to map areas of suitable habitat which is essential for effective management and conservation of biodiversity. Furthermore, the classification makes it possible to monitor heterogeneity throughout the landscape. Maintenance of this heterogeneity is an explicit management goal pursued by some conservation agencies, (e.g. SANParks Thresholds of Potential Concern for heterogeneity; http://www.sanparks.org/) to facilitate biodiversity conservation (Rogers 2003; Ishii et al. 2004).

A further advantage of a 3D classification, over one that is two-dimensional, is that it is a combination of plant structure and percent cover that influences biomass, and subsequently estimates of carbon. While numerous methods are available to monitor carbon stocks using satellite remote sensing (see Goetz et al. 2009), it is unclear how accurate and precise these estimates of biomass and biomass change are (Maniatis & Mollicone 2010). The 3D structural classification can be used to improve understanding on the relationship between biomass and habitat structure. Current biomass estimates for savanna vegetation are derived from adult, often single-stemmed trees (Colgan et al. 2012), yet they are applied to multi-stemmed coppicing vegetation and may contain up to 40% error. In communal rangelands in southern Africa, where large areas of vegetation are coppicing, biomass might be overestimated using the standard allometry. The necessity to estimate biomass more accurately highlights the need to map the three-dimensional structure of vegetation (Hall et al. 2011). 3D maps of savannas would provide greater monitoring potential to identify subtle changes and increased thickening of these woody components (Jansen & Di Gregorio 2002; Hall et al. 2011).

In order to ensure that the proposed structural classification method is comparable with existing classifications (Appendix 4.1), we used codes as a naming system and using percent canopy cover as a primary classifier. The cover classes chosen here are narrow, but can be combined to be directly comparable to existing classifications (Appendix 4.1). The classification adds to existing classifications not only by including understory layers, but also in the description of the spatial arrangement of the woody components in terms of their cohesion and diversity of woody layers within each area. These metrics aid in classifying the landscape in an ecologically meaningful way especially for habitat suitability mapping (McGarigal et al. 2002). Although we used a static grid for the classification, we do acknowledge that a grid has arbitrary boundaries and vegetation often has no clearly defined boundaries (Fisher 1997). A solution would be to use a moving window analysis, with the kernel size equal to distance at which spatial autocorrelation ceases, in this case 50 m, providing a spatially continuous description of the vegetation. This may, however, impact on change detection analyses.

We provide a classification method to reduce the large volumes of data associated with LiDAR while still capturing the spatially variable structural heterogeneity in savannas. In addition, since the classification can be done over large extents, the context of the structural patterns is captured. This aids in understanding the drivers of savanna woody structure and can be used in regional change predictions. 3D maps of woody vegetation structure for conservation and resource planning would be invaluable. In addition, the structural maps can be used to model the potential percolation of fire through the landscape (Archibald et al. 2012) as well as mapping surface roughness parameters which will affect storm surge (Medeiros et al. 2012). Although future satellite borne LiDAR campaigns are in the process of being planned, such as ICESat II, airborne LiDAR is currently the only method available to collect high resolution 3D information to detect individual tree canopies (Hall et al. 2011).

4.5 Acknowledgements

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4.6 References

Achard, F., Eva, H. & Mayaux, P. 2001. Tropical forest mapping from coarse spatial resolution satellite data: production and accuracy assessment issues. *International Journal of Remote Sensing* 22: 2741-2762.

Anderson, J.E., Martin, M.E., Smith, M-L., Dubayah, R.O., Hofton, M.A., Hyde, P., Peterson, B.E., Blair, J.B. & Knox, R.G. 2006. The use of waveform lidar to measure northern temperate mixed conifer and deciduous forest structure in New Hampshire. *Remote Sensing of Environment* 105: 248-263.

Archibald, S., Staver, A.C. & Levin, S.A. 2012. Evolution of human-driven fire regimes in Africa. *Proceedings of the National Academy of Science* **109**: 847-852

Asner, G.P., Knapp, D.E., Kennedy-Bowdoin, T., Jones, M.O., Martin, R.E., Boardman, J. & Field, C.B. 2007. Carnegie Airborne Observatory: in-flight fusion of hyperspectral imaging and waveform light detection and ranging (wLiDAR) for three-dimensional studies of ecosystems. *Journal of Applied Remote Sensing* 1, 013536.

Asner, G.P., Levick, S.R., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R., Jacobson, J., Colgan, M.S. & Martin, R.E. 2009. Large-scale impacts of herbivores on the structural diversity of African savannas. *Proceedings of the National Academy of Science* 12: 261-278.

Asner, G.P. & Levick, S.R. 2012. Landscape-scale effects of herbivores on tree fall in African savannas. *Ecology Letters* 15: 1211-1217.

Blaschke, T., Tiede, D. & Heurich, M. 2004. 3D landscape metrics to modelling forest structure and diversity based on laser scanning data. *International Archives of Photogrammetry, Remote Sensing and Spatial Information Sciences* XXXVI-8/W2: 129-132.

Belsky, A J. 1994. Influences of trees on savanna productivity: tests of shade, nutrients, and tree-grass competition. *Ecology* 75: 922-932.

Bergen, K.M., Gilboy, A.M. & Brown, D.G. 2007. Multi-dimensional vegetation structure in modelling avian habiat. *Ecological Informatics* 2: 9-22.

Birkett, A., & Stevens-Wood, B. 2005. Effect of low rainfall and browsing by large herbivores on an enclosed savannah habitat in Kenya. *African Journal of Ecology* 43: 123-130.

Buitenwerf, R., Bond, W.J., Stevens, N. & Trollope, S.W. 2012. Increased tree densities in South African savannas: >50 years of data suggest CO_2 as a driver. *Global change Biology* 18: 675-684

Colgan, M.S., Asner, G.P., Levick, S.R., Martin, R.E. & Chadwick, O.A. 2012. Topo-edaphic controls over woody plant biomass in South African savannas. *Biogeosciences Discussions* 9: 957-987.

De Knegt, H.J., Hengeveld, G.M., van Langevelde, F., de Boer, W.F. & Kirkman, K.P. 2007. Patch density determines movement patterns and foraging efficiency of large herbivores. *Behavioral Ecology* 18: 1065-1072

Di Gregorio, A. & Jansen, L.J.M. 2000. *Land Cover Classification System (LCCS): Classification concepts and user manual LCCS version 1.0*. GCP/RAF/287/ITA Africover, Food and Agriculture Organization of the United Nations Publishing Service, Rome.

Edwards, D. 1983. A broad-scale structural classification of vegetation for practical purposes. *Bothalia* 14: 705-712.

Eldridge, D.J., Bowker, M.A., Maestre, F.T., Roger, E., Reynolds, J.F. & Whitford, W.G. 2011. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. *Ecology Letters* 14:709-722.

Fischer, J. & Lindenmayer, D.B. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* 16: 265-280

Fisher, P. 1997. The pixel: A snare and a delusion. *International Journal of Remote Sensing* 18: 679-685

Fisher, J.T., Witkowski, E.T.F., Erasmus, B.F.N., van Aardt, J., Asner, G.P., Wessels, K.J. & Mathieu, R. 2012. Human-modified landscapes: patterns of fine-scale woody vegetation structure in communal savannah rangelands. *Environmental Conservation* 39: 72-82.

FGDC (Federal Geographic Data Committee, Vegetation Subcommittee) 1997. *Vegetation Classification Standard.* FGDC-STD-005

Goatley, C.H.R. & Bellwood, D.T. 2011. The roles of dimensionality, Canopies and complexity in ecosystem monitoring. *PLoS ONE* 6(11): e27307

Goetz, S.J., Baccinin, A., Laporte, N.T., Johns, T., Walker, W., Kellndorfer, J., Houghton, R.A. & Sun, M. 2009. Mapping and monitoring carbon stocks with satellite observations: a comparison of methods. *Carbon Balance and Management* 4:2.

Govender, N., Trollope, W.S. & van Wilgen, B.W. 2006. The effect of fire season, fire frequency, rainfall and management on fire intensity in savanna vegetation in South Africa. *Journal of Applied Ecology* 43: 748-758.

Halaj, J., Ross, D.W. & Moldenke, A.R. 2000. Importance of habitat structure to the arthropod food-web in Douglas-Fir canopies. *Oikos* 90: 139–152.

Hall, F.G., Bergen, K., Blair, J.B., Dubayah, R., Houghton, R., Hurtt, G., Kellndorfer, J., Lefsky, M., Ranson, J., Saatchi, S., Shugart, H.H. & Wickland, D. 2011. Characterizing 3D vegetation structure from space: Mission requirements. *Remote Sensing of Environment* 115: 2753-2775.

Hansen, M.C., DeFries, R.S., Townshend, J.R.G. & Sohlberg, R. 2000. Global land cover classification at 1 km spatial resolution using a classification tree approach. *International Journal of Remote Sensing* 21: 1331-1364.

Ishii, H.T., Tanabe, S. & Hiura, T. 2004. Exploring the relationships among canopy structure, stand productivity, and biodiversity of temperate forest ecosystems. *Forest Science* 50: 342-355.

Jansen, L.J.M. & Di Gregorio, A. 2002. Parametric land cover and land-use classifications as tools for environmental change detection. *Agriculture, Ecosystems and Environment* 91: 89-100.

Johansen, K., Arroyo, L.A., Armston, J., Phinn, S. & Witte, C. 2010. Mapping riparian condition indicators in a sub-tropical savanna environment from discrete return lidar data using object-based image analysis. *Ecological Indicators* 10: 796-807

Kirkpatrick, L.A. & Weishampel, J.F. 2005. Quantifying spatial structure of volumetric neutral models. *Ecological Modelling* 186: 312-325

Lefsky, M.A., Cohen, W.B., Parker, G.G. & Harding, D.J. 2002a. Lidar remote sensing for ecosystem studies. *BioScience* 52: 19-30.

Lefsky, M.A., Cohen, W.B., Harding, D.J., Parker, G.G., Acker, S.A. and Gower, S.T. 2002b. Lidar remote sensing of above-ground biomass in three biomes. *Global Ecology and Biogeography* 11: 393-399.

Levick, S.R., Asner, G.P., Kennedy-Bowdoin, T. & Knapp, D.E. 2009. The relative influence of fire and herbivory on savanna three-dimensional vegetation structure. *Biological Conservation* 142: 1693-1700.

Levick, S.R & Rogers, K.H. 2011. Context-dependent vegetation dynamics in an African savanna. *Landscape Ecology* 26: 515-528.

Loveland, T.R. & Belward, A.S. 1997. The International Geosphere Biosphere Programme Data and Information System Global Land Cover Data Set (DISCover). *Acta Astronautica* 41: 681-689.

Lumsden, L.F., & Bennett, A.F. 2005. Scattered trees in rural landscapes: foraging habitat for insectivorous bats in south-eastern Australia. *Biological Conservation* 122:205-222.

Luoga, E.J., Witkowski, E.T.F. & Balkwill, K. 2000. Subsistence use of wood products and shifting cultivation within a miombo woodland of eastern Tanzania, with some notes on commercial uses. *South African Journal of Botany* 66: 72–85.

MacArthur, R.H. & MacArthur, J.W. 1961. On bird species diversity. *Ecology* 42: 594-598

Magurran, A. E., 2004. *Measuring Biological Diversity*. Blackwell Publishing, Oxford, UK.

Manning, A.D, Fischer, J. & Lindenmayer, D.B. 2006. Scattered trees are keystone structures – Implications for conservation. *Biological Conservation* 132: 311-321.

Maniatis, D. & Mollicone, D. 2010. Options for sampling and stratification for national forest inventories to implement REDD+ under the UNFCCC. *Carbon Balance and Management* 5: http://www.cbmjournal.com/content/5/1/9

McGarigal, K., Cushman, S.A., Neel, M.C., & Ene, E. 2002. *FRAGSTATS: spatial pattern analysis program for categorical maps*. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available from http://www.umass.edu/landeco/research/fragstats/fragstats.html

Medeiros, S.C., Hagen, S.C. & Weishampel, J.F. 2012. Comparison of floodplain surface roughness parameters derived from land cover data and field measurements. *Journal of Hydrology* 452-453: 139-149

Mucina, L. & Rutherford, M.C (Eds.) 2006. *The vegetation of South Africa, Lesotho and Swaziland*, Pretoria, Strelitzia 19. South African National Biodiversity Institute, pp. 439-539.

Naesset, E. 2002. Predicting forest stand characteristics with airborne laser scanning using a practical two-stage procedure and field data. *Remote Sensing of Environment* 80: 88-99.

Neke, K.S., Owen-Smith, N. & Witkowski, E.T.F. 2006. Comparative resprouting response of Savanna woody plant species following harvesting: the value of persistence. *Forest Ecology and Management* 232: 114-123.

Owen-Smith, N. 1988. *Megaherbivores: The influence of very large body size on ecology*. Cambridge University Press.

Parsons, D. 2002. *Object Oriented Programming in C++, 2nd Edition*., pp. 110–185. Cengage Learning, Continuum, London, UK.

Pickett, S.T.A., Cadenasso, M.L. & Benning, T.L. 2003. Biotic and abiotic variability as key determinants of savanna heterogeneity at multiple spatiotemporal scales. In: Du Toit, J.T., Rogers, K.H. & Biggs, H.C. (eds) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*, pp. 422-446. Island Press, Washington DC, USA.

Price, B., Kutt, A.S. & McAlpine, C.A. 2010. The importance of fine-scale savanna heterogeneity for reptiles and small mammals. *Biological Conservation* 143: 2504-2513.

Rogers, K.H. 2003. Adopting a heterogeneity paradigm: Implications for management of protected savannas. In: Du Toit, J.T., Rogers, K.H. & Biggs, H.C. (eds.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*, pp. 41-58. Island Press, Washington DC, USA.

Sankaran, M., Ratnam, J & Hanan, N. 2008. Woody cover in African savannas: the role of resources, fire and herbivory. *Global Ecology and Biogeography*. 17: 236-245.

Scholes, R.J., & Walker, B.H. 1993. *An African savanna – Synthesis of the Nylsvley study*. Cambridge University Press, Cambridge, UK.

Scholes, R.J., & Archer, S.R. 1997. Tree-grass interactions in savannas. *Annual Review of ecology and Systematics* 28:517-544.

Seymour, C.L., & Dean, W.R.J. 2009. The influence of changes in habitat structure on the species composition of bird assemblages in the southern Kalahari. *Austral Ecology* 35: 581–592.

Smart, R., Whiting, M.J. & Twine, W. 2005. Lizards and landscapes: integrating field surveys and interviews to assess the impact of human disturbance on lizard assemblages and selected reptiles in a savanna in South Africa. *Biological Conservation* 122: 23–31.

Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E. & Jacobson, J. 2010. Effects of fire on woody vegetations tructure in African savanna. *Ecological Applications* 20: 1865-1875.

Tews, J., Brose, U., Grimm, V., Tielborger, K., Wichmann, M.C., Schwager, M. & Jeltsch, F. 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. *Journal of Biogeography* 31: 79-92

Thompson, M. 1996. A standard land-cover classification scheme for remote sensing applications in South Africa. *South African Journal of Science* 92: 34-42.

Treydte, A.C, Grant, C.C. & Jeltsch, F. 2009. Tree size and herbivory determine below-canopy grass quality and species composition in savannahs. *Biodiversity and Conservation* 18: 3989-4002.

Wessels, K.J., Mathieu, R., Erasmus, B.F.N., Asner, G.P., Smit, I.P.J., van Aardt, J., Main, R., Fisher, J., Marais, W., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R. & Jacobson, J. 2011. Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* 261: 19-29.

Williams, S.E., Marsh, H. & Winter, J. 2002. Spatial scale, species diversity, and habitat structure: Small mammals in Australian tropical rain forest. *Ecology* 83: 1317–1329.

Winnie, J.A., Cross, P. & Getz, W. 2008. Habitat quality and heterogeneity influence distribution and behavious in African buffalo (*Syncerus caffer*). *Ecology* 89: 1457-1468.

4.7 Appendices

4.1. Key features of vegetation structure classifications and land cover classifications and the subsequent classification of semi-arid savannas using each type.

4.2 The process of using first a multi-threshold segmentation (a-d) and then a multi-resolution segmentation (e & f) to identify savanna woody vegetation tree canopies using object-based image analysis on a canopy height model derived from Light Detection and Ranging (LiDAR).

4.3. Woody vegetation structural metrics (canopy cover, number of canopy layers present, canopy cohesion, dominant height classes, number of height classes present and Simpson's Diversity Index) in 0.25 ha grid cells for eight sites across Kruger National Park, Sabi Sand Wildtuin and Bushbuckridge, South Africa.

Appendix 4.1: Key features of vegetation structure classifications and land cover classifications and the subsequent classification of semi-arid savannas using each type.

<table>
<tr><th>Key features</th><th colspan="3">Cover classes</th><th>Classification of semi-arid savannas</th></tr>
<tr><td colspan="5">Edwards (1983)</td></tr>
<tr>
<td rowspan="9">Primary set of four growth forms – trees, shrubs, grasses and herbs
Primary set of four cover classes – high, tall, short, low
Set of four height classes for each form type
Growth form x cover = structural group
Structural group x height = formation class
Cover class given as a measure of both % cover and crown:gap ratio
Developed for field data but can be applied to aerial and satellite images</td>
<td>Cover class descriptors</td><td>Cover class (%)</td><td>Crown:gap</td>
<td rowspan="9">
Forest and woodland
Total tree cover >0.1%, shrub cover <10% if >1m high
<table>
<tr><th>Dominant height class</th><th>Total tree cover
100-75%</th><th>75-10%</th><th>10-1%</th><th>1-0.1%</th></tr>
<tr><td>Trees>20m</td><td>1. High forest</td><td>5. High closed woodland</td><td>9. High open woodland</td><td>13. High sparse woodland</td></tr>
<tr><td>Trees 10-20m</td><td>2. Tall forest</td><td>6. Tall closed woodland</td><td>10. Tall open woodland</td><td>14. Tall sparse woodland</td></tr>
<tr><td>Trees 5-10m</td><td>3. Short Forest</td><td>7. Short closed woodland</td><td>11. Short open woodland</td><td>15. Short sparse woodland</td></tr>
<tr><td>Trees 2-5m</td><td>4. Low forest</td><td>8. Low closed woodland</td><td>12. Low open woodland</td><td>16. Low sparse woodland</td></tr>
</table>
Thicket and bushland
Total tree cover >1%, shrub cover >10% & >1m height
<table>
<tr><th></th><th>Total tree cover
100-10%</th><th>10-1%</th></tr>
<tr><td>Trees 5-10m & shrubs 2-5 m</td><td>17. Short thicket</td><td>19. Short bushland</td></tr>
<tr><td>Trees 2-5 m & shrubs 1-5m</td><td>18. Low thicket</td><td>20. Low bushland</td></tr>
</table>
</td>
</tr>
<tr><td>Closed</td><td>10-100</td><td>0-2</td></tr>
<tr><td>Open</td><td>1-10</td><td>2-8.5</td></tr>
<tr><td>Sparse</td><td>0.1-1</td><td>8.5-30</td></tr>
<tr><td>Scattered</td><td><0.1</td><td>>30</td></tr>
<tr><td>Continuous</td><td>76-100</td><td><0.1</td></tr>
<tr><td>Sub-continuous</td><td>51-75</td><td>0.1-0.3</td></tr>
<tr><td>Moderately closed</td><td>26-50</td><td>0.3-0.9</td></tr>
<tr><td>Semi-open</td><td>11-25</td><td>0.9-2.0</td></tr>
</table>

Key features	Cover classes		Classification of semi-arid savannas
LCCS (Di Gregorio and Jansen 2000)			
Hierarchical *a priori* Variables defined by a set of independent diagnostic criteria Dichotomous phase (use of 3 classifiers: presence of vegetation, edaphic condition and artificiality of cover = 8 cover types) Modular hierarchical phase (different for each of the 8 cover types. Classifiers for savannas = life form, height, spatial aspects (cover) stratification)	**Cover class descriptors** Closed Open Sparse	**Cover class (%)** >(60-70) between (60-70) & (10-20) below (10-20) but >1	**Dichotomous phase: initial level distinction** -primarily vegetated **Dichotomous phase: second level distinction** -edaphic condition: terrestrial Dichotomous phase: tertiary level distinction Artificiality of cover: (semi-) natural **Modular-hierarchical phase:** Life form (physiognomy) – woody plants divided into trees and shrubs, and herbaceous into forbs, graminoids and lichens/mosses Height: >5m = tree, <5m = shrub (if clear physiognomy shows trees and >3m then trees) Cover: Closed (>(60-70)%), open (between (60-70) & (10-20)%) and sparse (below (10-20)% but >1%) Range from forest (multilayered forest, forest with shrubs, multi-layered forest with emergents, forest with shrubs and emergents, forest with emergents), woodland (woodland with shrubs, woodland with herbaceous, woodland with shrubs and emergents, woodland with herbaceous and emergents), Thicket (thicket with emergents, thicket with shrubs emergents, thicket with tree and shrub emergents), shrubland 1 (shrubland with herbaceous layer and emergents, shrubland with herbaceous layer and shrubs emergents, shrubland with trees and shrubs emergents), shrubland 2 (shrubland with herbaceous layer, shrubland with emergents, shrubland with shrubs emergent) and grassland (grassland with sparse trees, grassland with sparse shrubs, grassland with sparse trees and shrubs)

Key features	Cover classes		Classification of semi-arid savannas
National Land Cover (NLC) Classification for South Africa (Thompson 1996)			
Hierarchical *a priori* 3 hierarchical levels I: 12 broad land cover types II: 23 subclasses III: flexible, user-defined subcategories Based on Edwards and LCCS	**Cover class descriptors** Closed Open Sparse Forest Woodland Wooded grassland Thicket Scrub Bushland Bush >70% but pockets of clumps in bushland or grassland	**Cover class (%)** 10-100 1-10 <1.0 >70 40-70 10-40 >70 >70 <70 >70, pockets of clumps – 10-40, 40-70 and >70	Level I: Forest and woodland — Level I: Thicket, bushland II: Forest, Woodland, wooded grassland — II: Thicket , Bush clumps **Forest and woodland**: All wooded areas with greater than 10% tree canopy cover,[1] where the canopy is composed of mainly self-supporting, single stemmed,[2] woody plants >5 m in height. Essentially indigenous tree species,[3] growing under natural or semi-natural conditions (although it may include some localized areas of self-seeded exotic species). Excludes planted forests (and woodlots).Typically associated with the Forest and Savanna biomes in South Africa. **Forest:** Tree canopy cover > 70%. A multi-strata community, with interlocking canopies, composed of canopy, sub-canopy, shrub and herb layers. **Woodland**: Tree canopy cover between 40-70%. A closed-to-open canopy community, typically consisting of a single tree canopy layer and a herb (grass) layer. **Wooded grassland**: Tree canopy cover between 10-40%. An open-to-sparse canopy community, typically consisting of a single tree canopy layer and a herb (grass) layer. **Thicket, bushland, scrub forest and high fynbos:** Communities typically composed of tall, woody, self-supporting, single and/or multi-stemmed plants (branching at or near the ground), with, in most cases, no clearly definable structure. Total canopy cover > 10%, with canopy height between 2-5 m. Essentially indigenous species, growing under natural or semi-natural conditions (although it may include some localized areas of self-seeded exotic species, especially along riparian zones). Typical examples are Valley Bushveld, Mopane Bush, and tall fynbos. Dense bush encroachment areas would be included in this category. **Thicket**: Areas of densely interlaced trees and shrub species (often forming an impenetrable community). Composed of multi-stemmed plants with no clearly definable structure or layers, with > 70% cover. A typical example would be Valley Bushveld. **Scrub**: Vegetation intermediate in structure between forest true forest mad thicket. A multi-layered community with interlocking canopies, with > 70% cover. **Bushland**: Similar to 'thicket', but more open in terms of canopy cover levels. Composed of multi-stemmed plants with no definable structure or layers, and with < 70% cover. An example would be Mopane Bush. **Bush**: Scattered islands of thicket-like clumps vegetation (i.e. > 70% cover) within a matrix of more open bushland or grassland.

<table>
<tr><th>Key features</th><th colspan="2">Cover classes</th><th>Classification of semi-arid savannas</th></tr>
<tr><td colspan="4">Global Land Cover Classification (Hansen et al. 2000)</td></tr>
<tr><td rowspan="7">Global coverage

1km resolution

Hierarchical tree classification

12 classes

Decision tree classification using spectral metrics</td><td>Cover class descriptors</td><td>Cover class (%)</td><td rowspan="7">Mixed Forests: lands dominated by trees with a per cent canopy cover > 60% and height exceeding 5 m. Consists of tree communities with interspersed mixtures or mosaics of needeleaf and broadleaf forest types. Neither type has < 25% or > 75% landscape coverage.

Woodlands: lands with herbaceous or woody understories and tree canopy cover of > 40% and < 60%. Trees exceed 5 m in height and can be either evergreen or deciduous.

Wooded grasslands/shrublands: Lands with herbaceous or woody understories and tree canopy cover of > 10% and < 40 %. Trees exceed 5 m in height and can be either evergreen or deciduous.

Closed bushlands or shrublands: lands dominated by bushes or shrubs. Bush and shrub per cent canopy cover is > 40%. Bushes do not exceed 5 m in height. Shrubs or bushes can be either evergreen or deciduous. Tree canopy cover is < 10%. The remaining cover is either barren or herbaceous.

Open shrublands: lands dominated by shrubs. Shrub canopy cover is >10% and <40%. Shrubs do not exceed 2 m in height and can be either evergreen or deciduous. The remaining cover is either barren or of annual herbaceous type.</td></tr>
<tr><td>Mixed forest</td><td>>60 (but not <25 or >75)</td></tr>
<tr><td>Woodlands</td><td>40-60</td></tr>
<tr><td>Wooded grasslands/ shrublands</td><td>10-40</td></tr>
<tr><td>Closed bushlands or shrublands</td><td>>40%, trees < 10</td></tr>
<tr><td>Open shrublands</td><td>10-40</td></tr>
<tr><td>Grasslands</td><td><10%</td></tr>
<tr><td colspan="4">IGBP DIS Land cover working group vegetation classes (Loveland and Belward 1997)</td></tr>
<tr><td rowspan="7">Exhaustive global classification

17 DISCover classes (no specification as to how they were chosen, only descriptions available)

Regional validation

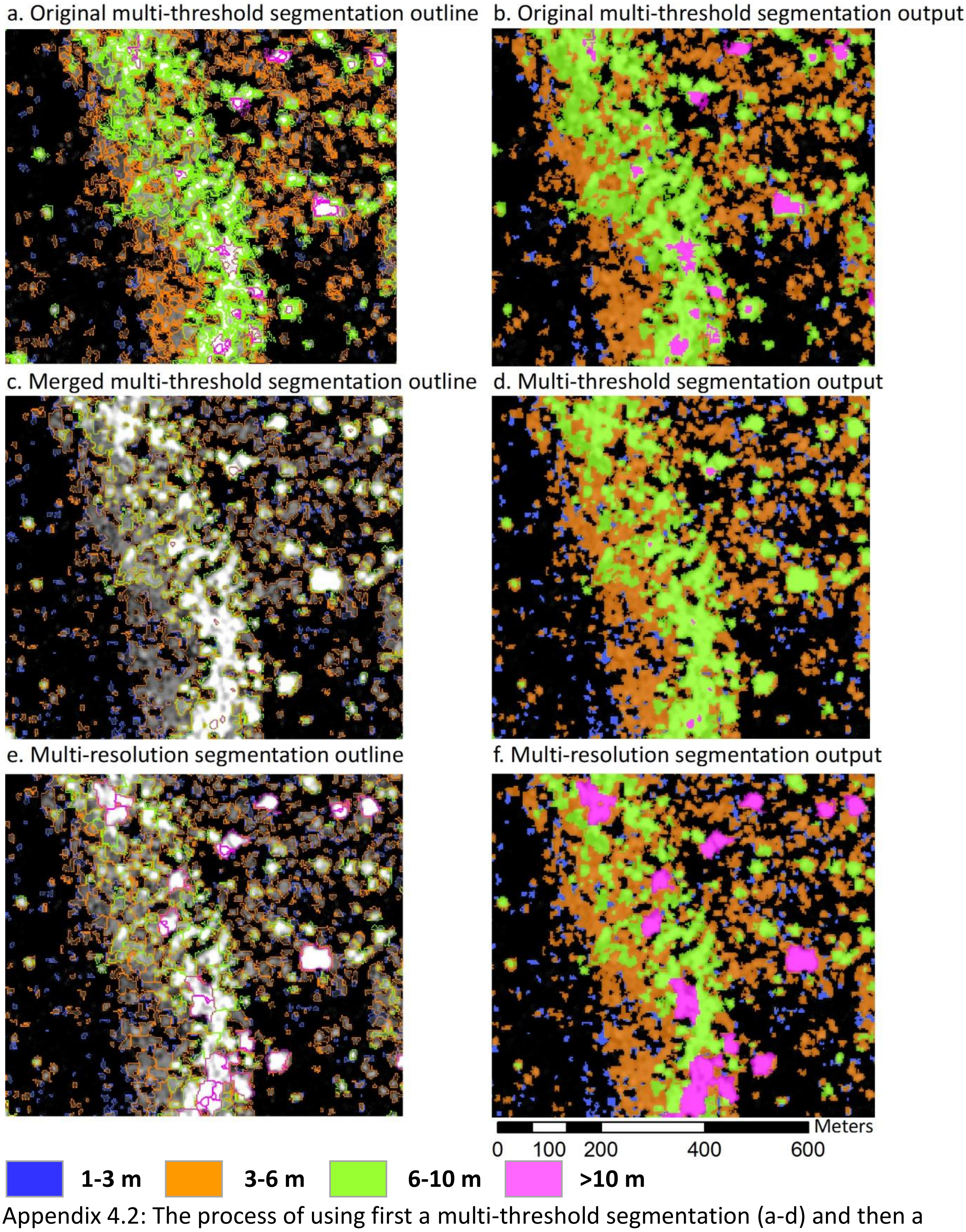
1 km resolution</td><td>Cover class descriptors</td><td>Cover class (%)</td><td rowspan="7">Mixed forests: lands dominated by trees with a per cent canopy cover > 60% and height exceeding 2 m. Consists of tree communities with interspersed mixtures or mosaics of the other four forest cover types. None of the forest types exceeds 60% of the landscape.

Woody savannas: lands with herbaceous and other understory systems, and with forest canopy between 30-60%. The forest cover height exceeds 2 m.

Savannas: lands with herbaceous and other understory systems and with forest canopy between 10-30%. The forest cover height exceeds 2 m.

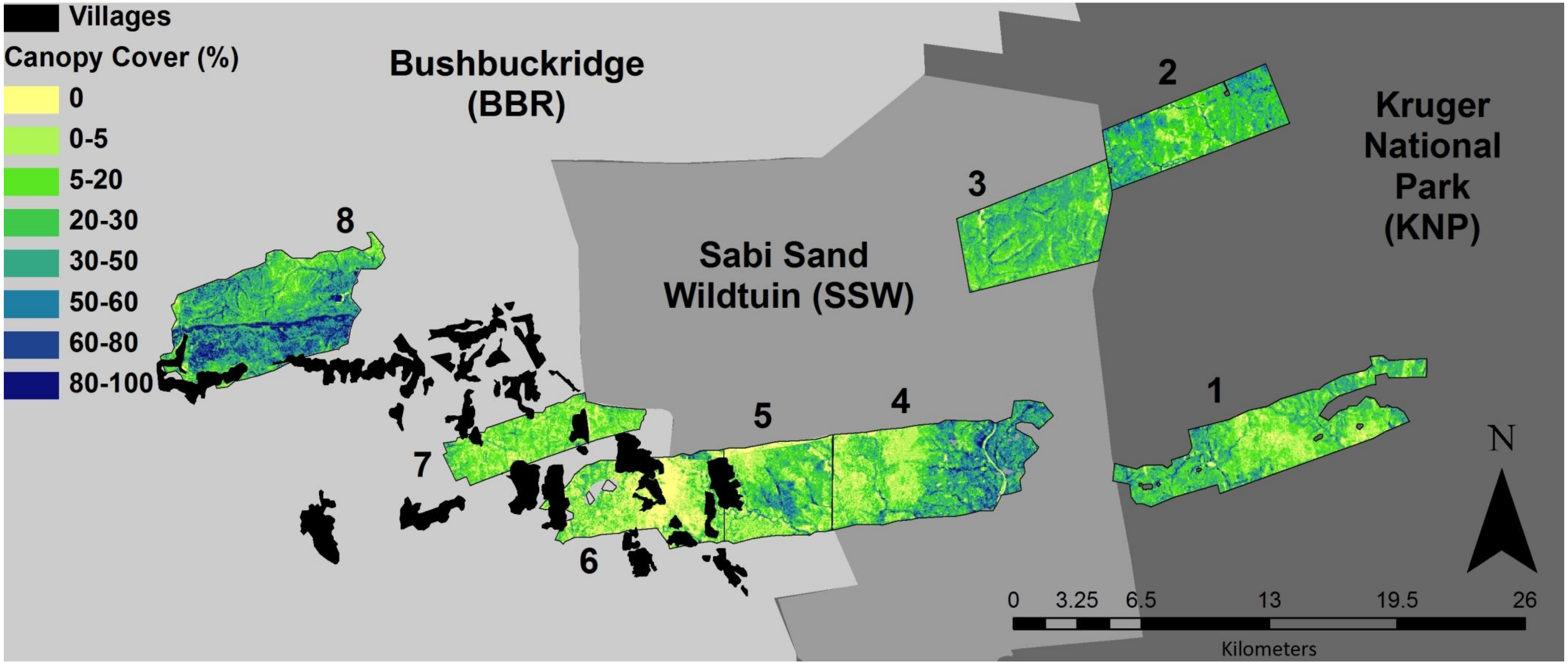
Closed shrublands: lands with woody vegetation less than 2 m tall and with shrub-canopy cover > 60%. The shrub foliage can be either evergreen or deciduous.

Open shrublands: lands with woody vegetation less than 2 m tall and with shrub canopy cover between 10-60%. The shrub foliage can be either evergreen or deciduous.</td></tr>
<tr><td>Mixed forest</td><td>>60</td></tr>
<tr><td>Woody savannas</td><td>30-60</td></tr>
<tr><td>Savannas</td><td>10-30</td></tr>
<tr><td>Closed shrublands</td><td>>60</td></tr>
<tr><td>Open shrublands</td><td>10-60</td></tr>
<tr><td>Grasslands</td><td><10%</td></tr>
</table>

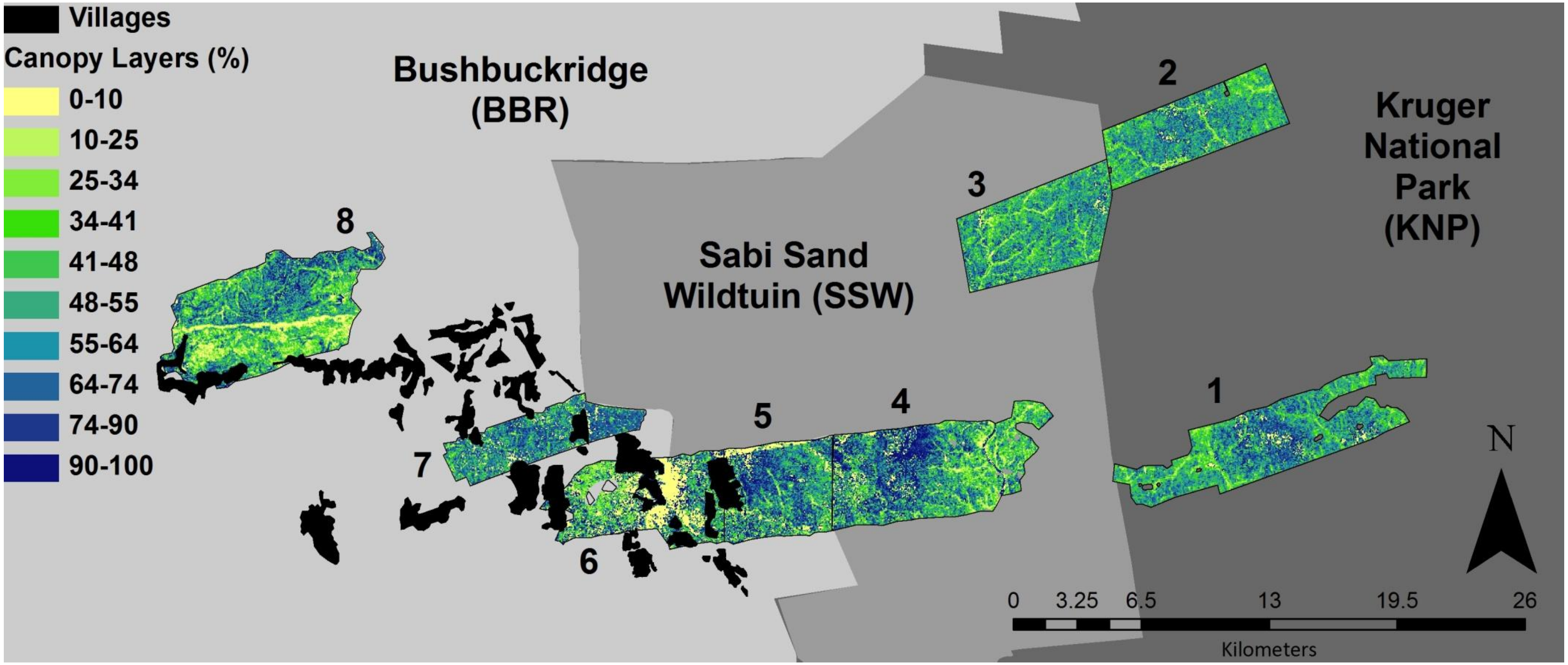


Appendix 4.2: The process of using first a multi-threshold segmentation (a-d) and then a multi-resolution segmentation (e & f) to identify savanna woody vegetation tree canopies using object-based image analysis on a canopy height model derived from Light Detection and Ranging (LiDAR). a. The original multi-threshold segmentation classified into the 4 height classes, b. The multi-threshold segmentation output, c. The multi-threshold segmentation once classes have been merged, d. Output of the multi-threshold segmentation, e. Multi-resolution segmentation on the merged multi-threshold segmentation output, and f. The final product of trees and shrubs after the multi-threshold and then multi-resolution segmentations.

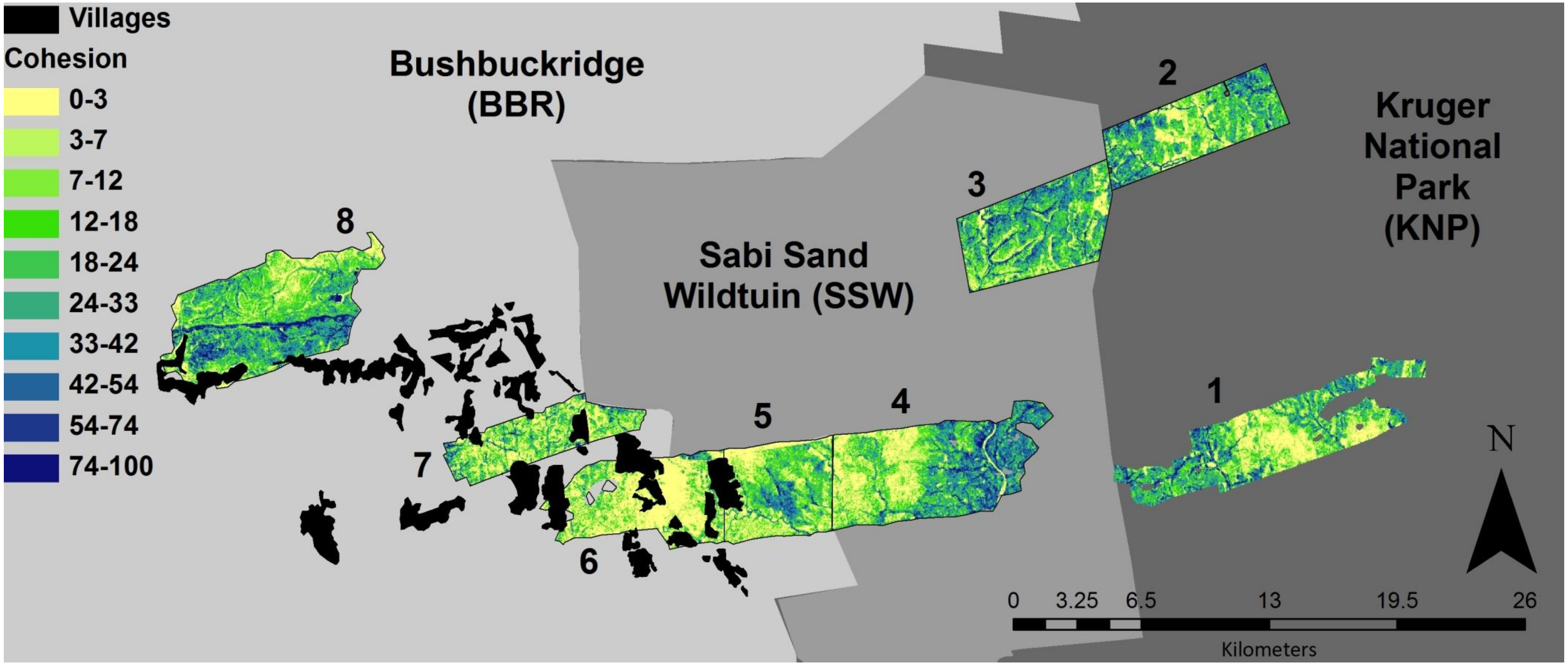
Appendix 4.3: Six woody vegetation structural metrics (canopy cover, number of canopy layers present, canopy cohesion, dominant height classes, number of height classes present and Simpson's Diversity Index) in 0.25 ha grid cells for eight sites across Kruger National Park, Sabi Sand Wildtuin and Bushbuckridge, South Africa. The sites form an east to west gradient of increasing rainfall from 550 mm (Site 1 & 2) to >1200 mm (Site 8). Sites 1 & 2 are in a national protected area (KNP), Sites 3 & 4 are in a private game reserve (SSW) and sites 5-8 are communal rangelands. Site 8 is a low use intensity site, and Site 6 is the highest intensity of use site.



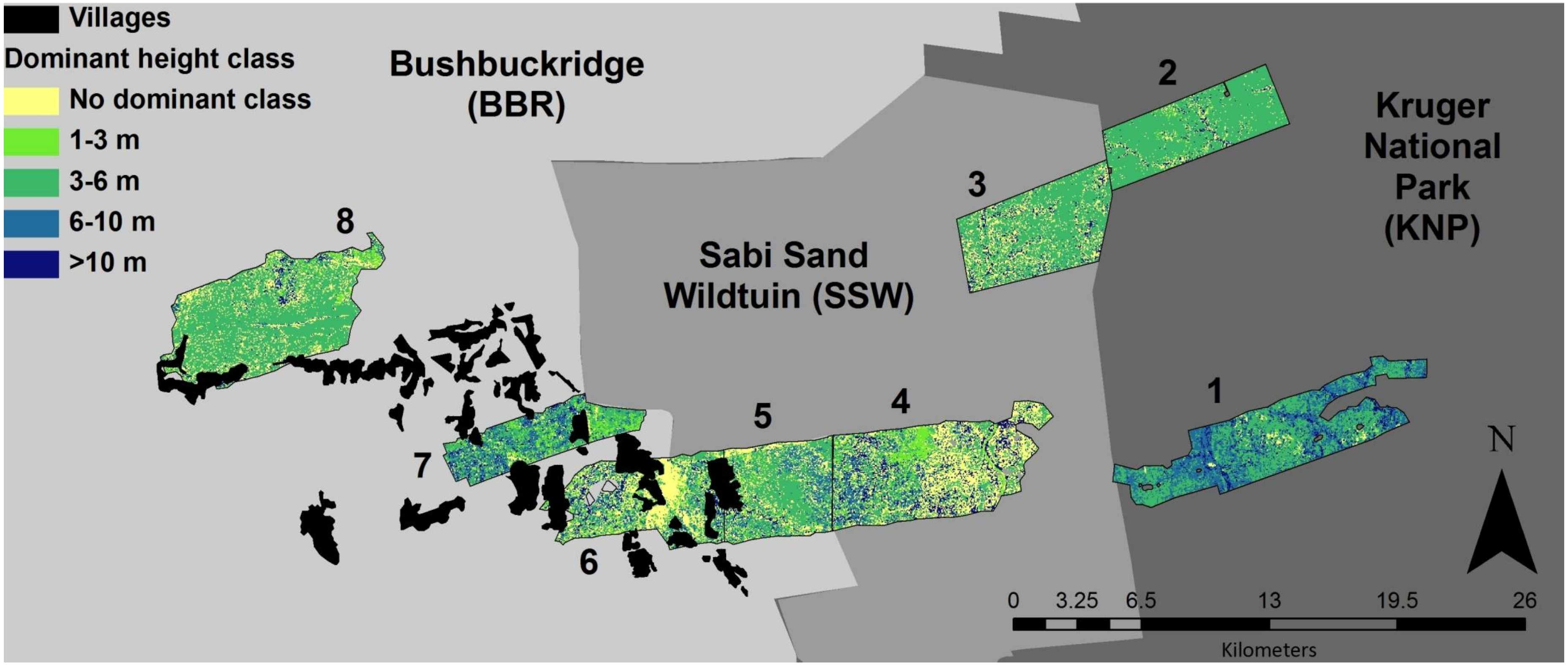
4.3a: Canopy cover (%)



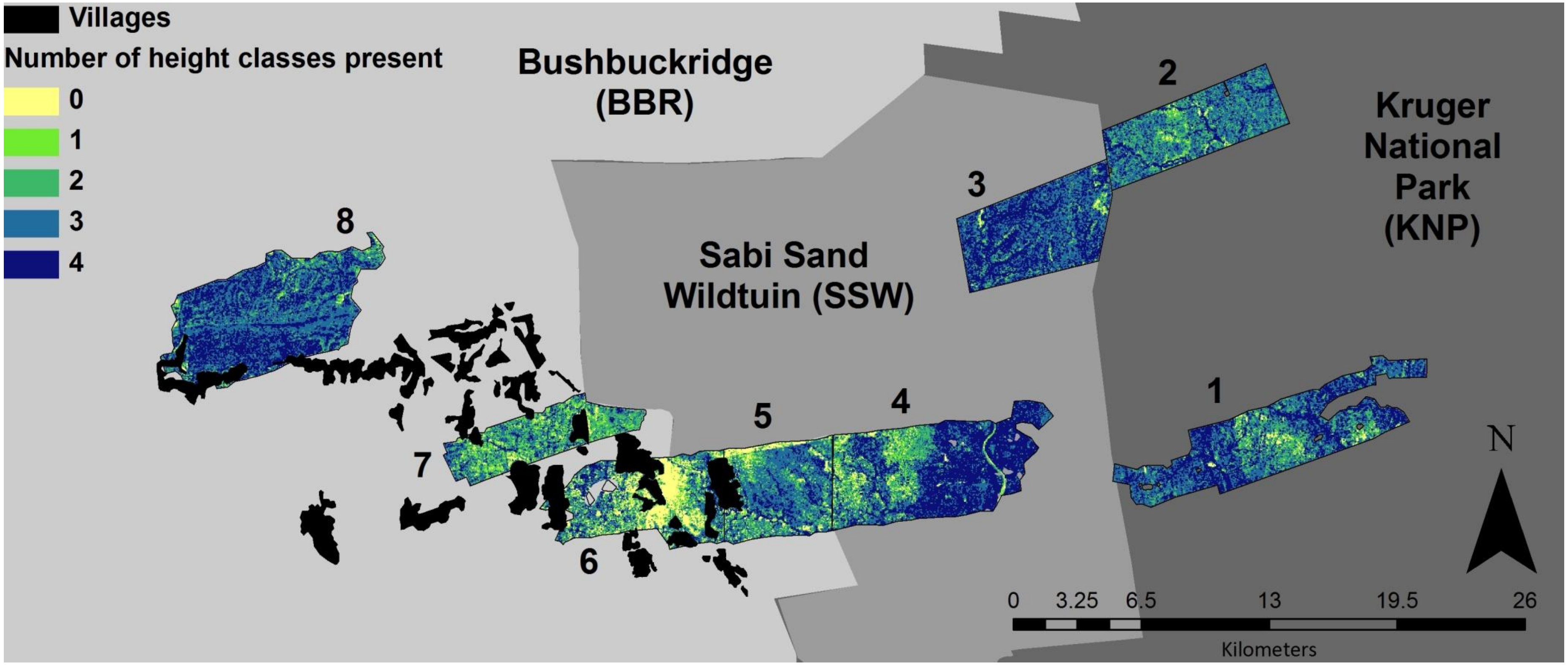
4.3b: Number of canopy layers present (%)



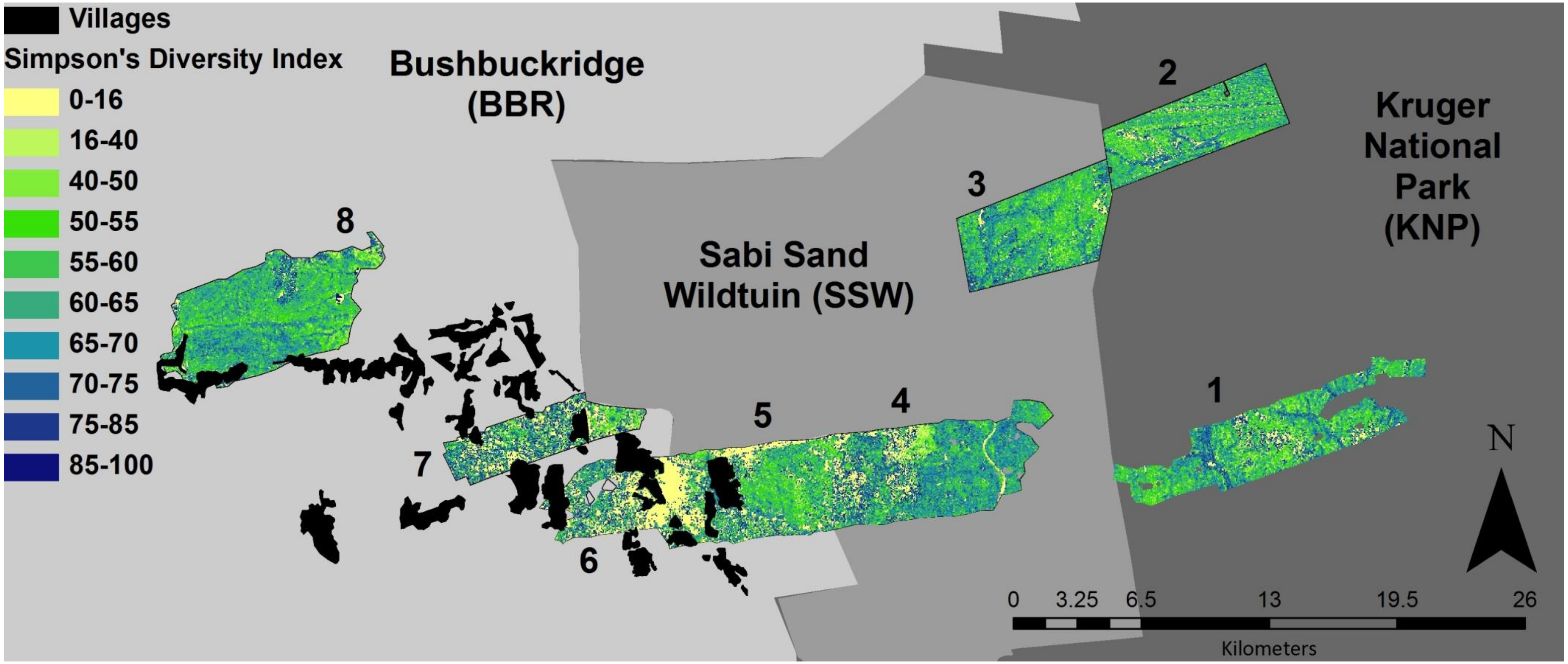
4.3c: Cohesion of woody layer



4.3d: Dominant height class



4.3e: Number of height classes present. Height classes are 1-3 m, 3-6 m, 6-10 m and >10 m



4.3f: Simpson’s Diversity Index (SDI) of height classes present

Chapter 5 : Detecting structural change in a protected area and communal rangeland: Application of a 3D classification method

Fisher, J.T.[1,2*], Witkowski, E.T.F.[1], Erasmus, B.F.N.[2], Asner, G.P.[3], van Aardt, J.A.N.[4], Wessels, K.J.[5]

[1]Restoration and Conservation Biology Research Group, School of Animal, Plant & Environmental Sciences, University of the Witwatersrand, WITS, Johannesburg, 2050, South Africa
[2]Centre for African Ecology, School of Animal, Plant & Environmental Sciences, University of the Witwatersrand, WITS, Johannesburg, 2050, South Africa
[3]Department of Global Ecology, Carnegie Institution for Science, 260 Panama Street, Stanford, CA 94305, USA
[4]Chester F. Carlson Center for Imaging Science, Rochester Institute of Technology: 54 Lomb Memorial Drive, Rochester, New York, 14623, USA
[5]Remote Sensing Research Unit, Council for Scientific and Industrial Research (CSIR)-Meraka Institute, P.O. Box 395, Pretoria, 0001, South Africa

5.1 Abstract

The feedbacks between increasing population pressure, socio-economic development and associated natural resource use in savannas are resulting in large scale land cover change which can be mapped using remote sensing. However, change in vegetation structure is difficult to quantify using traditional remote sensing methods which typically detects two dimensional (2D) changes only. We use a three-dimensional (3D) woody vegetation structural classification applied to LiDAR (Light Detection and Ranging) data to investigate change in fine-scale woody vegetation structure over a two-year period in a protected area (PA) and a communal rangeland (CR). This effectively quantifies the advantages of a 3D versus a 2D classification and provides an assessment of the effect of human use and management on woody structural change. LiDAR data were collected in April 2008 and 2010 over 3 300 ha of savanna in north-east South Africa. Individual tree canopies were identified using object-based image analysis and classified into four height classes: 1-3 m, 3-6 m, 6-10 m and >10 m. Five structural metrics were then calculated for 0.25 ha grid cells using the height classes and sub-canopy cover measured using volumetric pixels (voxels): Canopy Cover, Number of Canopy Layers Present, Cohesion, Dominant Height Class and Number of height classes present. The relationship between top of canopy cover and sub canopy cover was investigated using regression. Gains, losses and persistence (GLP) of each height class and the five structural metrics over the two years were calculated for each site. GLP of clusters of each structural metric (calculated using LISA (Local Indicators of Spatial Association) statistics) were calculated to assess the changes in clusters of each metric over time. Top of canopy cover is not a good predictor of sub-canopy cover. In addition, the number of canopy layers present and cohesion of the canopy cover showed gains and losses with some persistence in canopy cover over time, necessitating the use of a 3D classification to detect fine scale changes, especially in structurally heterogeneous savannas. Trees >3 m showed recruitment and gains up to 2.2 times higher in the CR where they are protected, but losses of up to 3.2 times more in the PA compared to the 'poorly' managed CR due to treefall caused by elephant, as well as direct fire management. Land use has affected the structure in the adjacent sites, with the low intensity use CR showing greater structural diversity and resilience to change. We show that a 3D approach is successful in detecting

fine scale, short term changes between land uses and can thus be used as a monitoring tool for savanna woody vegetation structure.

Keywords: change detection, ecosystem function, ecosystem services, fire, geology, land-use, Local Indicators of Spatial Association (LISA), savanna

5.2 Introduction

The effects of biodiversity loss on ecosystem function and services has been a major focus of global change research (e.g. Balvanera et al. 2006; Hector & Bagchi 2007; Hooper et al. 2005; Naeem 2002). Landscape modification and habitat fragmentation are two of the key drivers of biodiversity loss (Fischer & Lindenmayer 2007, Sala et al. 2000), with unsustainable natural resource use further exacerbating the problem. In South African savannas, which are home to over nine million rural poor (Twine et al. 2003), both the strong dependence on natural resources and expansion of settlements into intact vegetation has altered vegetation structure in this biome (Freitag-Ronaldson & Foxcroft 2003; Coetzer et al. 2010).

Savannas, with their discontinuous woody layer in a continuous grassy matrix, are particularly structurally heterogeneous with a mosaic of woody patches and complex vertical structure. The complex structural heterogeneity created by the diversity of life forms in savannas contributes towards ecosystem services (supporting, provisioning, regulating and cultural, MA 2005). Structural diversity provides habitat for a wide range of fauna (Atauri & de Lucio 2001; Lumsden & Bennett 2005; Manning et al. 2006; McArthur & McArthur 1961; Palminteri et al. 2012; Seymour & Dean 2010; Smart et al. 2005; Tews et al. 2004). With regard to regulating services, scattered large trees function as keystone structures, for example by regulating microclimate and elevating localised soil nutrients (Manning et al. 2006). Savannas are also prone to bush encroachment arising from overgrazing/browsing intensity, over harvesting and an unsuitable fire regime, resulting in an increase in the density of woody vegetation and a subsequent reduction in palatable grasses (Oba et al. 2000).

Provisioning services such as fuelwood and fencing poles are linked to vegetation structure, in that trees and saplings of various diameters are harvested for different purposes in communal rangelands (Luoga et al. 2004; Neke et al. 2006). Fisher et al. (2012) found that increasing pressure on natural resources in rural savannas is resulting in a decline in disturbance gradients around settlements, leading to increased homogenization of structure in highly utilized areas (Fisher et al. 2012). In protected areas, provisioning services are related to habitat structure and forage. Ungulates utilise woody vegetation up to 3 m, and elephants target trees in the 5-9 m range, thereby creating an 'elephant trap' (Asner & Levick 2012). Intensified land use practices, either through increased natural resource use or through intensive management leads to land cover change (MA 2005). The effect of land cover change on biodiversity loss highlights the need to investigate causes and products of structural change, yet these investigations should be relevant to the ecosystem services associated with various structures.

Changes in woody vegetation structure are detectable both between land uses and over time. Traditionally, fine-scale measurements of structure such as tree height, diameter and number of stems are field-based, while large-scale but coarse measurements of structure such as woody cover and spatial patterns are often estimated using remote sensing methods. Time and financial constraints usually limit field surveys to measuring structure at one point or a few points in time. Therefore, field measurements of structural dynamics are possible although they are often collected by different researchers using different protocols (e.g. woody structure in a riparian area in 1996 by Garner & Witkowski (1997) and in 2005 by Beater et al. (2008); and woody structure in two villages by Banks et al. (1996) in 1992 and Matsika et al. (2012) in 2009). Matsika et al. (2012) found a reduction in wood stocks in both villages over time, although the finding was more pronounced in one village where fuelwood harvesting was unsustainable and the rangeland was being encroached by the settlement. The differences in rate of decline indicate patterns are settlement specific, highlighting the need for change detection studies to be carried out over more extensive areas.

Remote sensing is necessary for long-term change studies over large regions or in areas that have not had field work previously applied to them. Giannecchini et al. (2007) conducted a 23-year historical analysis of woody cover change (percent cover and number of woody patches) for three villages using aerial photographs. The results were site specific and related to intensity of use, population density, natural resource availability, diversification of livelihood strategies and drought, the findings of which support Matsika et al. (2012). Aerial photographs and satellite imagery such as Landsat are commonly used for change detection studies (e.g. Asner et al. 2003; Brink & Eva 2009; Coetzer et al. 2010; Luoga et al. 2005; Mwavu & Witkowski 2008) as they date back to the 1930's (aerial imagery) and 1970's

(Landsat), thus allowing for long term change to be measured (Buitenwerf et al. 2012). However, in the absence of field work, it is unclear what fine-scale changes are occurring within a landscape, including potential structural changes below the canopy linked to the use of the landscape. Passive remote sensing products can be used to detect more than just changes in canopy cover by including changes in life form, spatial distribution, leaf type and phenology and stratification such as in the Land Cover Classification System (LCCS) (Di Gregoria & Jansen 2002). However, plants below the canopy cannot be detected. Therefore, if woody vegetation encroachment occurs, or if the distribution of vegetation size classes change, the change would go undetected (Jansen & Di Gregorio 2002).

Current woody vegetation change detection methods in savannas are limited to fine-scale field measurements across land uses over small geographic areas, or to remotely sensed coarse-scale information gathered over large areas. Airborne LiDAR (Light Detection and Ranging) provides a powerful middle ground between field data and satellite remote sensing. LiDAR is an active remote sensing technology that measures sub-canopy information at fine resolution over large extents via measurement of laser travel time (Lefsky et al. 2002). As this technology is relatively new, historical change detection is not yet common; however, data collected now can be used as baseline information for future investigations. The Carnegie Airborne Observatory (CAO; Asner et al. 2007) collected LiDAR data across two land uses (communal rangelands and protected areas) in the Lowveld region of South Africa in 2008 and 2010. We use a three-dimensional (3D) woody structural classification (Fisher et al. 2013, Chapter 4 this thesis) to investigate change in fine-scale woody vegetation structure over a two-year period in the two different land uses to address the following: (1) What are the advantages of a 3D over a 2D vegetation structural classification for detection of change over time? (2) How does human use of the landscape affect woody vegetation structural dynamics?

5.3 Methods

5.3.1 Site description

The two study sites border one another on the boundary between Sabie Sands Wildtuin (SSW), a private game reserve, and the village of Justicia in Bushbuckridge Municipality (BBR) situated in Mpumalanga province, north-east South Africa (Fig. 5.1). The total area is 3 300 ha (2 034 ha in SSW and 1 266 ha in BBR). As the two sites border one another, they share essentially the same biophysical characteristics. Rainfall is predominantly in the form of convection thunderstorms, with a mean annual precipitation of 650 mm, while the mean annual temperature is 21°C, with hot summers and mild winters (Shackleton et al. 1994). Topography is undulating with an altitudinal range of 310 – 460 m above sea level and the geology in the region is predominantly granite with Timbavati gabbro intrusions. However, only gabbro was present in BBR while both gabbro and granite were present in SSW. Granite lowveld is the dominant vegetation unit in the area, with gabbro grassy bushveld and legogote sour bushveld also occurring (Mucina & Rutherford 2006). Typical woody plant species in the granite lowveld include: *Terminalia sericea*, *Combretum zeyheri* and *C. apiculatum* on the deep sandy toplands, and *Acacia nigrescens*, *Dichrostachys cinerea* and *Grewia bicolor* on the more clayey soils of the bottomlands. In the two other vegetation units additional common species include *Sclerocarya birrea*, *Lannea schweinfurthii*, *Ziziphus mucronata*, *Dalbergia melanoxylon*, *Peltoforum africanum* and *Pterocarpus rotundifolius*.

Bushbuckridge consists of two former Apartheid homelands, Gazankulu and Lebowa (Thornton 2002), which were formed with the Native Land Act (No. 27) of 1913. Between 1972 and 2012 human population density increased in the area to 209 people/km^2 (Stats SA 2012), with resulting increase in land utilization intensity and economic impoverishment (Pollard *et al*. 2003). In 1994 the region was divided into Tribal Trust Lands and governed by Tribal Authorities. Subsistence livelihoods are practiced, and land utilization tends to be more intensive near the villages (Shackleton et al. 1994, Fisher et al. 2012). Historically, cultural values of the people in the area meant harvesting of live trees used for medicine, fruit and culturally important activities was discouraged; however, the demand for fuel wood and timber now overrides these values (Higgins et al. 1999).

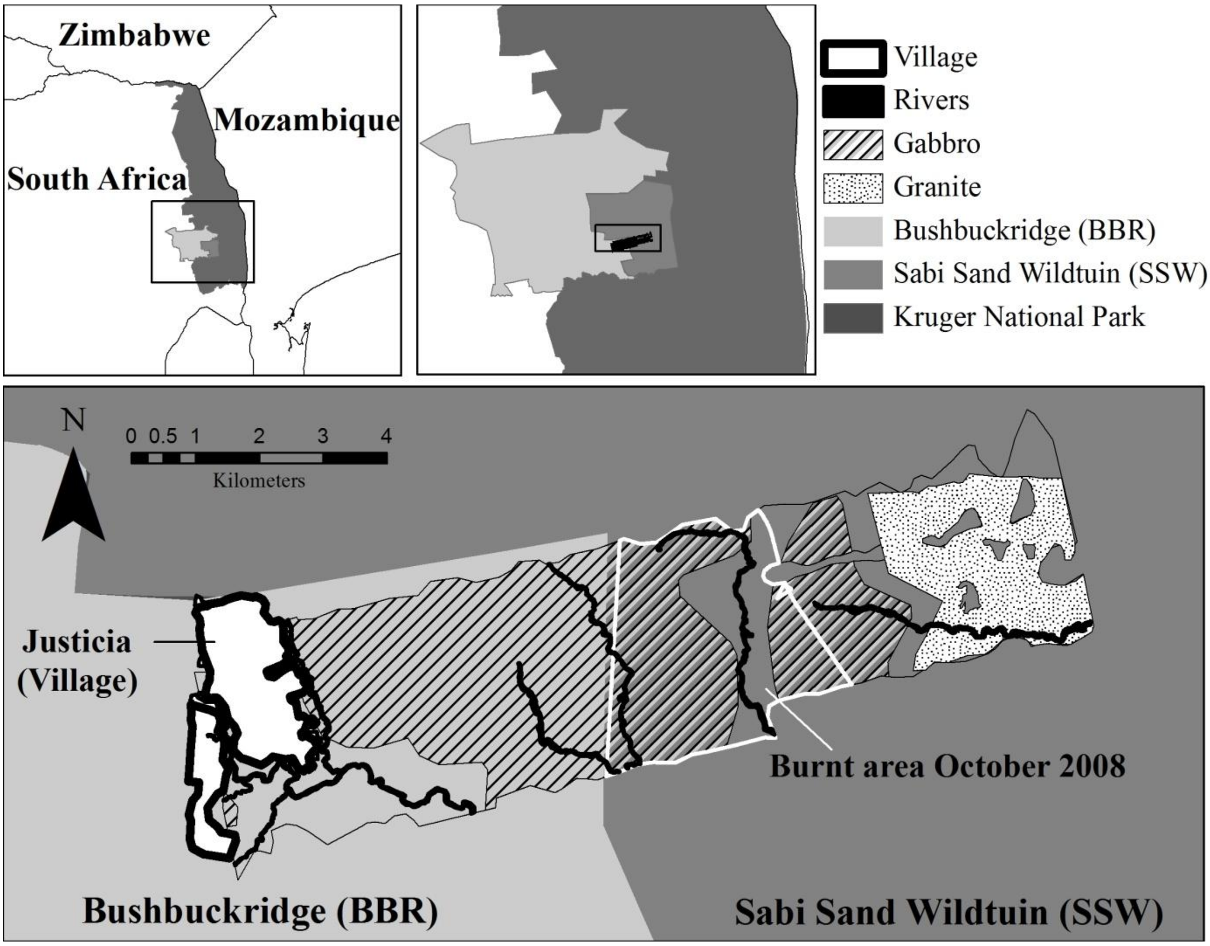


Figure 5.1: Location of study sites within Bushbuckridge municipality (BBR) and Sabi Sand Wildtuin (SSW), Mpumalanga Province, South Africa. Justicia village, and granite and gabbro substrates are shown.

Sabie Sands Wildtuin is 65 000 ha, and was only formally proclaimed as a private game reserve in 1965. From 1922 to 1934, it was known as the Sabi Ranch, owned by the Transvaal Consolidated Lands (TCL), and was used for cattle ranching. Additional areas in the current SSW were purchased and used as game reserves around the same time, and in 1938 all cattle were removed due to a foot-and-mouth disease outbreak (J. Swart, pers. comm.). Each land owner within the conservancy manages their own land with regard to bush clearing and fire regimes. With the removal of fences between Kruger National Park and SSW in 1993, there was an influx of elephant into SSW, increasing from 0.0009 elephant/ha in 1993 to 0.007/ha in 1998 (Hiscocks 1999). From 1996 to 1998, although the damage appeared high, only 21% of preferred tree species in southern SSW were damaged (Hiscocks 1999). Elephants primarily affect the structure rather than the species composition of trees, transforming vegetation to short woodland with a low density of large trees (Trollope et al. 1998). Structural changes are often better indicators of disturbance than compositional changes (Shackleton et al. 1994). However, elephant do also tend to target certain species such as Marula, a keystone species, which has declined 25% in a 10 year period (2001 - 2010) in Kruger National Park (Helm and Witkowski 2012). Hiscocks (1999) warned that an increase in elephant population from 0.003 elephant/ha to 0.007/ha in two years required attention. By 2010 the population had increased to 0.013/ha, although it has seen peaks of up to 0.02 elephant/ha in 2007 (the year prior to our first data collection) and in 2012 (M. Grover pers. comm.). Part of the SSW site was burnt in October 2008 (Fig. 5.1).

5.3.2 Light Detection and Ranging (LiDAR)

Woody vegetation was mapped across approximately 3 300 ha of semi-arid savanna in South Africa in April 2008 and April 2010 with the Carnegie Airborne Observatory Alpha System (CAO-Alpha; http://cao.ciw.edu). See Asner et al. (2007) for detailed specifications of the CAO-Alpha system and Fisher et al. (2012) for details on data collection and processing of the 3D point cloud into a digital elevation model (DEM), digital surface model (DSM), canopy height model (CHM – image layer with top of canopy heights), and volumetric pixels (voxels). Voxels are the vertically distributed density of laser returns in 1-metre increments from ground to top-of-canopy (Asner et al. 2008; Lefsky et al. 2002; Popescu & Zhao 2008). Ground validation of vegetation heights was conducted concurrent to the airborne campaign in 2008 (Wessels et al. 2011).

5.3.3 3D classification of woody vegetation structure

A 3D characterization of woody vegetation is necessary to accurately measure structure, which in turn represents biomass, habitat and biodiversity as well as a metric of ecosystem services (Hall et al. 2011; Fisher et al. 2013). Furthermore, a high degree of spatial detail is necessary to detect not only change but also modifications in land cover and vegetation structure. Jansen and Di Gregorio (2002) promote a parametric (classifier) approach to classification for change detection in line with the Land Cover Classification System (LCCS). This type of approach allows for a consistent application of land-cover or land-use criteria, and a consistent use of criteria at the same level of classification, although actual criteria differ for each land cover type ensuring greater specificity and change detection ability (Jansen & Di Gregorio 2002). Fisher et al. (2013) developed a 3D classification of savanna vegetation structure using principles taken from LCCS. The classification is specific to savanna vegetation and uses ecologically meaningful height classes related to fire, herbivory, frost and human use (Fisher et al. 2013). Plants are delineated using object-based image analysis and classified as either shrubs (1-3 m), low trees (3-6 m), high trees (6-10 m) or tall trees (>10 m).

Top of canopy cover is then classified for 0.25 ha grid cells according to canopy cover, number of canopy layers present and cohesion of patches of different height classes (Fisher et al. 2013; Table 5.1). The second level of the classification categorizes the individual height classes within the canopy and sub-canopy using the voxel results from the LiDAR data analysis. Canopy layers are described in height order from shrub to tall tree. Here we explore differences in the metrics and the four height classes between land uses, geology and years.

5.3.4 Woody vegetation structure characterisation

The advantages of a 3D over a 2D classification were investigated by comparing the gains (G; *increase* in the value of the metric under consideration), losses (L; *decrease* in the value of the metric under consideration) and persistence (P; no change in the value of the metric under consideration) (Coetzer et al. 2013) of the percent canopy layers present (CL) and canopy cohesion with persistence in canopy cover from 2008 to 2010 (n = 13 198 0.25 ha grid cells). Canopy cover is a categorical metric (Table 5.1) therefore gains, losses and persistence (GLP) were determined if there was a change in the cover class. For example, if the cover class changed from ‘5’ (20-30%) to ‘6’ (30-40%) a gain would be denoted. For continuous variables such as CL and cohesion (Table 5.1), the value of the metric had to exceed a change of greater the 5% before it was considered a gain or loss of value in order to reduce ‘noise’ (i.e. if a metric changed from 61.7% to 62.3% this would not be considered a gain). Percent cover of the four height classes (1-3 m, 3-6 m, 6-10 m and >10 m) were compared between 2008 and 2010 using box plots (R v2.13.0). We investigated the relationships between the percent cover of the four height classes as measured on the top of canopy image (i.e. seen from above), and the percent cover of the sub-canopy vegetation within in each class (i.e. lateral view) in 2008 and 2010 using regression. Regressions were performed in R Studio (R v2.13.0, UsingR package).

5.3.5 Structural dynamics between land uses

Gains, losses and persistence in the cover of each height class as measured from the voxel data (sub-canopy and top of canopy) in SSW and BBR were compared. In addition, changes in spatial patterns of clusters of high and low values of canopy cover, canopy layers, canopy cohesion, number of height classes and dominant height classes in 2008 and 2010 were measured. Spatial clustering was quantified using a LISA statistic, Local Moran's I (Anselin 1995) in ArcMap 10.0 (ESRI 2010, Redlands, USA, www.esri.com). Local Moran's I is used to assess the influence of locations on the magnitude of the global Moran's I statistic, with significance values giving a representation of the spatial clustering of similar values around each grid cell (Anselin 1995). The z-score (based on each metric's standard deviation), p-value (probability of the observed pattern being created by a random process) and local mean value of the respective classification metric were calculated for each cell, and cells which were significantly different as determined using a permutation approach were classified as follows. The target mean of each cell is compared to the local mean of neighbouring grid cells using an inverse distance spatial relationship (features that are closer together have a larger influence on the local mean than features further away). For grid cells with a strong positive z-score (>1.96) a cell is either classified as HH if the target mean is higher than the local mean, or LL if the target mean is lower. For spatial outliers (grid cells with z-scores <-1.96) grids cells are classified as HL if the target mean is higher than the local mean, and LH if it is lower (ESRI 2010). To simplify these classifications, they may be interpreted as follows: HH (highly significant clusters of high values), LL (highly significant cluster of low values), LH (outlier in which a low value is surrounded by predominantly high values) and HL (outlier in which a high value is surrounded predominantly by low values) (ESRI 2010). The change in size and location of clusters over the two year period was then measured. The spatial intersection of clusters in 2008 and 2010 (as calculated using Anselin Local Morans I indicator of spatial association; Fig. 5.2a & b) was used to determine whether there was a gain (increase in spatial extent of significant clusters), loss (decrease in spatial extent of significant clusters), persistence (no change in clusters) or NS (no change in non significant cells, i.e. non-significant cells did not become clusters; Fig. 5.2c).

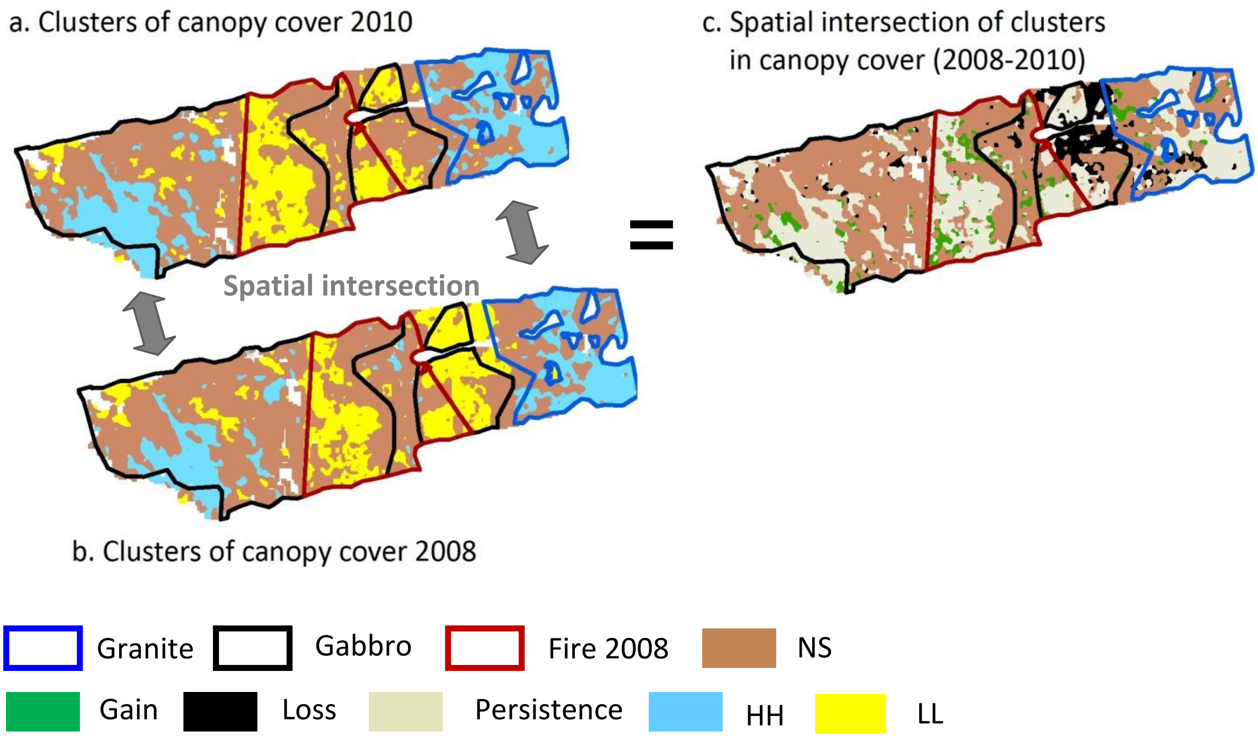


Figure 5.2: Example of how gains, loss and persistence (GLP) of clusters of high / low values of a particular metric, in this case canopy cover, is derived. Maps of clusters (as calculated using Anselin Local Morans I indicator of spatial association; HH: highly significant clusters of high values, LL: highly significant cluster of low values; NS: Non Significant areas i.e. no clusters) were calculated for canopy cover in a. 2010 and b. 2008. The difference between where clusters occur in 2008 and 2010 are depicted in a GLP map (c) where gain indicates an increase in clusters, loss indicates a decrease in clusters and persistence is no change in clusters. NS indicates a persistence of no significant clusters.

Table 5.1:Ecological relevance of 3D woody vegetation structural classifiers and dynamics.

Functional metric	**Ecological relevance**
Canopy Cover (categorical data)	Canopy cover is a key descriptor of biomes, with savannas having around 5-60% woody canopy cover (Mucina & Rutherford 2006). An increase or decrease in cover may be the result of a biome shift.
Canopy Layers (continuous data)	Measurement of sub-canopy density. An increase in sub-canopy vegetation may indicate bush encroachment. Dense sub-canopies, particularly from 1-3 m and 3-6 m may restrict animal movements in the landscape and decrease predator visibility (Ripple & Beschta 2004).
Cohesion (continuous data)	Measure of habitat connectivity (McGarigal et al. 2002). At a fine scale cohesion has implications for organisms' movement through, and use of, the landscape. At a landscape scale high cohesion would reduce edge effects (Fischer & Lindenmayer 2007). An increase in cohesion of one or more vertical height classes may indicate increased bushiness.
Number of height classes (categorical data)	The greater the number of life forms present, the higher the structural heterogeneity. This may also increase faunal diversity as a result of increased habitat niches (Ishii et al. 2004). Higher diversity might also increase resilience to global change and/or intense use/management of the landscape (Fischer et al. 2006).
Dominant life form (categorical data)	Keystone structures in a landscape, such as tall trees, influence nutrient concentrations, surrounding grass palatability and evapo-transpiration (Tews et al. 2004). Shrub dominated areas might result in a decrease of palatable grass species and affect animal movement in the landscape (Oba et al. 2000).

5.4 Results

5.4.1 Woody vegetation structure characterisation

Grid cells with a persistent canopy cover showed gains, losses and persistence in the percent of number of canopy layers present and canopy cohesion from 2008-2010 (Fig. 5.3). Number of canopy layers showed greater variability than cohesion over the two years as indicated by 2.3 times higher percent losses (48%) and 3 times less persistence compared to cohesion (Fig. 5.3). Mean percent cover for each height class is not significantly different over the two year period (Fig. 5.4). The relationship between top of canopy cover and vegetation present within the canopy is not 1:1 (Fig. 5.5). Percent cover of each height class is greater for the sub-canopy compared to the top of canopy (Fig. 5.5), although differences are more pronounced from 1-3 m and 6-10 m where the slopes of the regressions are ≤0.5 (Fig 5.5 a, b, f). A significant relationship is present between top of canopy percent cover and sub-canopy percent cover from 3-6 m ($p<0.0005$; R^2= 0.76 Fig 5.5 c; R^2= 0.82 Fig 5.5 d), which often constitutes the highest amount of cover in a grid cell (Fig 5.12a), and over 10 m (R^2= 0.93, Fig. 5.5g; R^2= 0.78, Fig. 5.5h), often the lowest amount of cover (Fig. 5.12a). Higher sub canopy cover than top of canopy cover is present from 1-3 m, indicating high density of vegetation within this height class which is present under most other height classes. Even when no shrubs are visible in the top of canopy (but other height classes are present), >90% of the vegetation present may contain a shrub layer (Fig. 5.5a). The greatest change in sub canopy cover from 2008 to 2010 is in the 1-3 m and >10 m height classes (Fig. 5.5a, b, g, h), with the sub-canopy cover showing greater variation than top of canopy cover (reduction in R^2 from 0.4 to 0.35 (Fig. 5.5a, b) and 0.93 to 0.78 (Fig. 5.5g, h).

Each height class showed greater persistence than gains or losses, although this result was more pronounced for height classes >6 m (Fig. 5.6c, d). From 1-3 m, GLP were similar for SSW and BBR (Fig 5.6a), while SSW showed consistently higher percent losses than BBR for height classes 3-6 m (2.75 times higher), 6-10 m (3.2 times higher) and >10 m (2.6 times higher). Similarly, BBR showed higher percent gains than SSW for height classes >6 m, particularly from 6-10 m (2.2 times higher) (Fig. 5.6b, c, d).

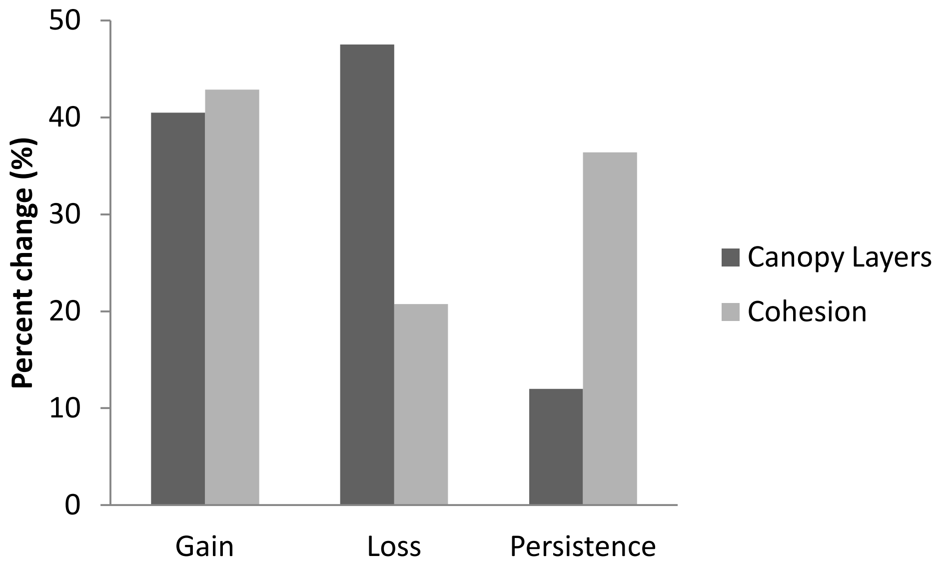


Figure 5.3: Gains, losses and persistence (GLP), with no change in canopy cover (persistence, 2008-2010), in the percent of canopy layers present and canopy cohesion in areas with persistent canopy cover from 2008 to 2010 in Sabi Sand Wildtuin (SSW) and Bushbuckridge (BBR) study sites combined, South Africa (n=6149 0.25 ha grid cells).

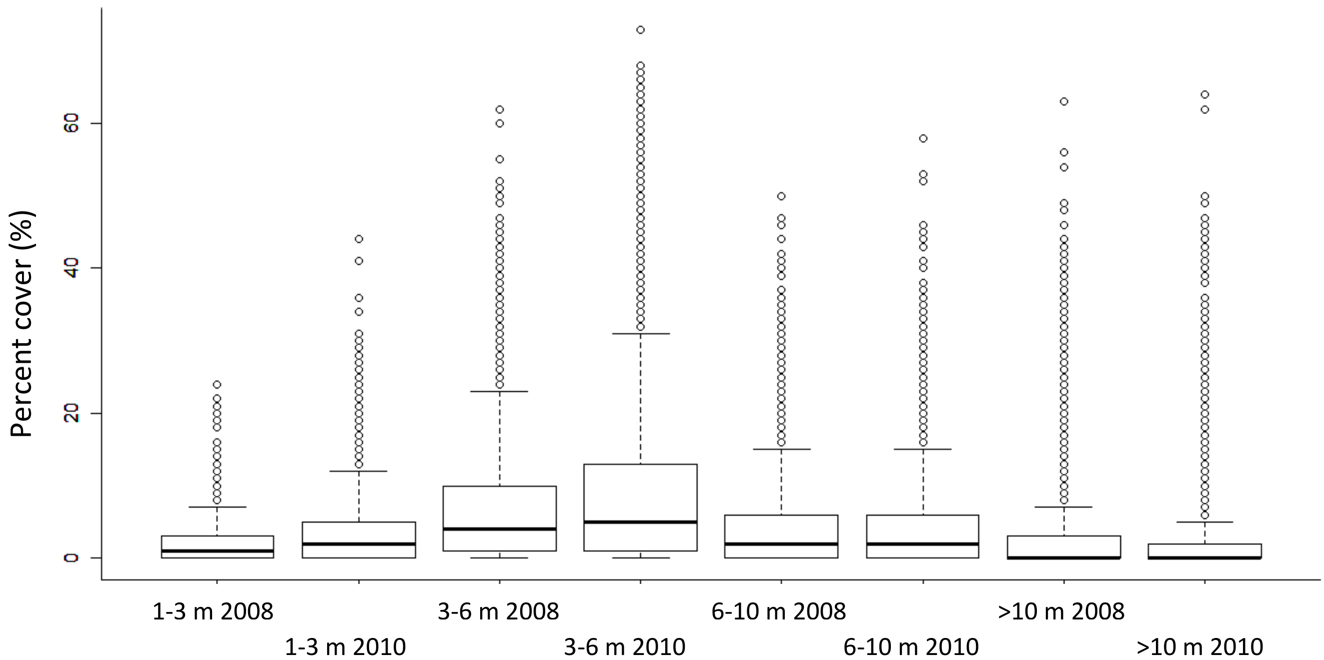


Figure 5.4: Box plot of percent cover for four height classes ((1-3 m, 3-6 m, 6-10 m and >10 m) in 2008 and 2010 in Sabi Sand Wildtuin (SSW) and Bushbuckridge (BBR) study sites (n=6149 0.25 ha grid cells).

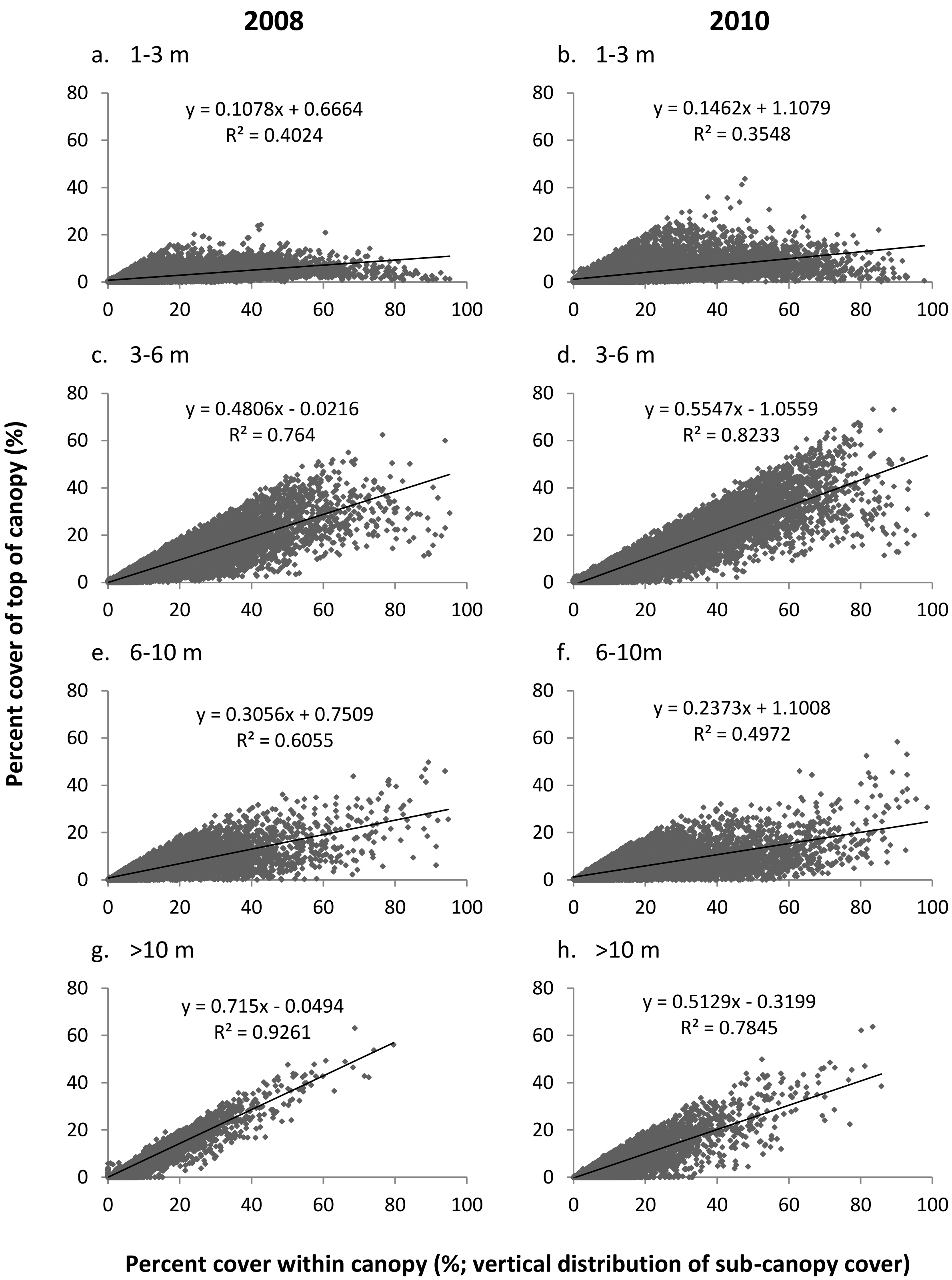


Figure 5.5: Relationship between percent of total canopy from above and the percent cover of vegetation present within the canopy as measured using the slicer LiDAR data for four height classes (1-3 m, 3-6 m, 6-10 m and >10 m) in 2008 and 2010 in Sabi Sand Wildtuin (SSW) and Bushbuckridge (BBR) study sites ($p<0.005$; n= 13 198 0.25 ha grid cells).

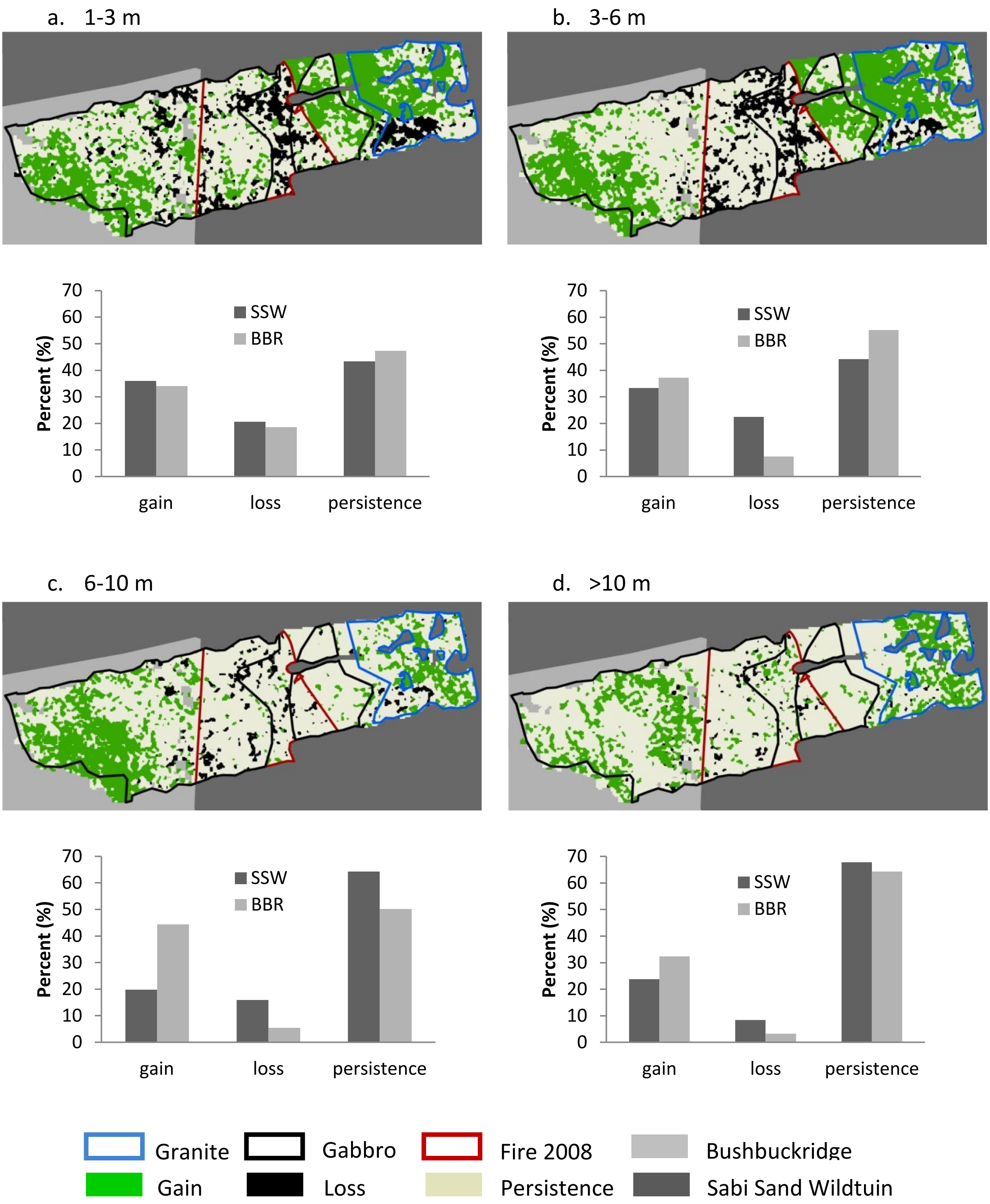


Figure 5.6: Gains, losses and persistence (GLP) for four height classes (a. 1-3 m, b. 3-6 m, c. 6-10 m, d. >10 m) measured using volumetric pixels in 0.25 ha grid cells in Sabi Sand Wildtuin (SSW; n=8136 0.25 ha grid cells) and Bushbuckridge (BBR; n=5062 0.25 ha grid cells) study sites from 2008 to 2010.

a. Gains, losses and persistence of canopy cover

b. Gains, losses and persistence of clusters of high / low values of canopy cover

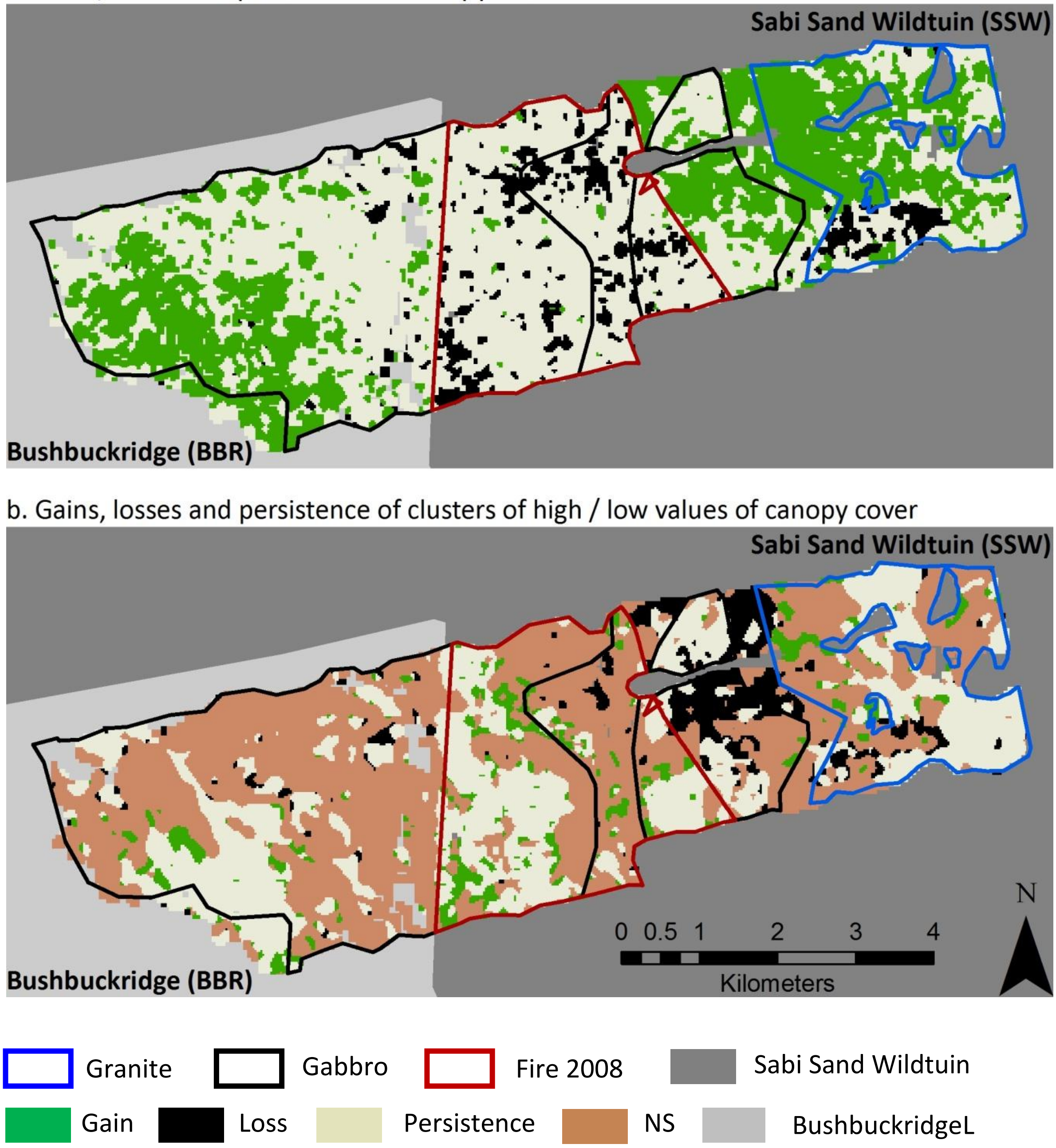


Figure 5.7: Gains, losses and persistence of a. canopy cover and b. clusters of high / low values of canopy cover from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin Local Morans I indicator of spatial association. Gain indicates an increase in clusters, loss indicates a decrease in clusters and persistence is no change in clusters. NS indicates a persistence of no clusters.

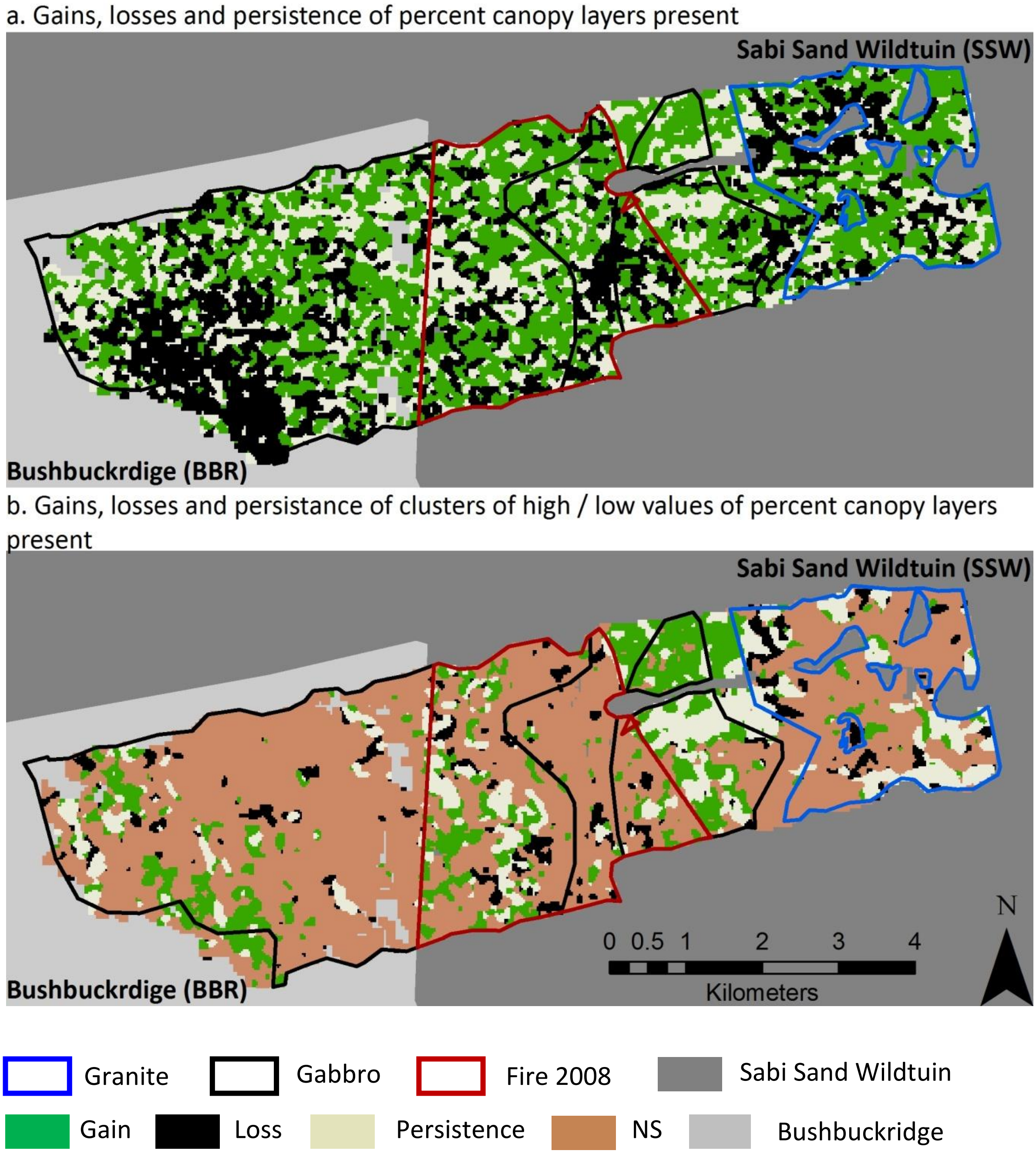


Figure 5.8:Gains, losses and persistence of a. percent number of canopy layers present and b. clusters of high / low values of percent number of canopy layers present from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin Local Morans I indicator of spatial association. Gain indicates an increase in clusters, loss indicates a decrease in clusters and persistence is no change in clusters. NS indicates a persistence of no clusters.

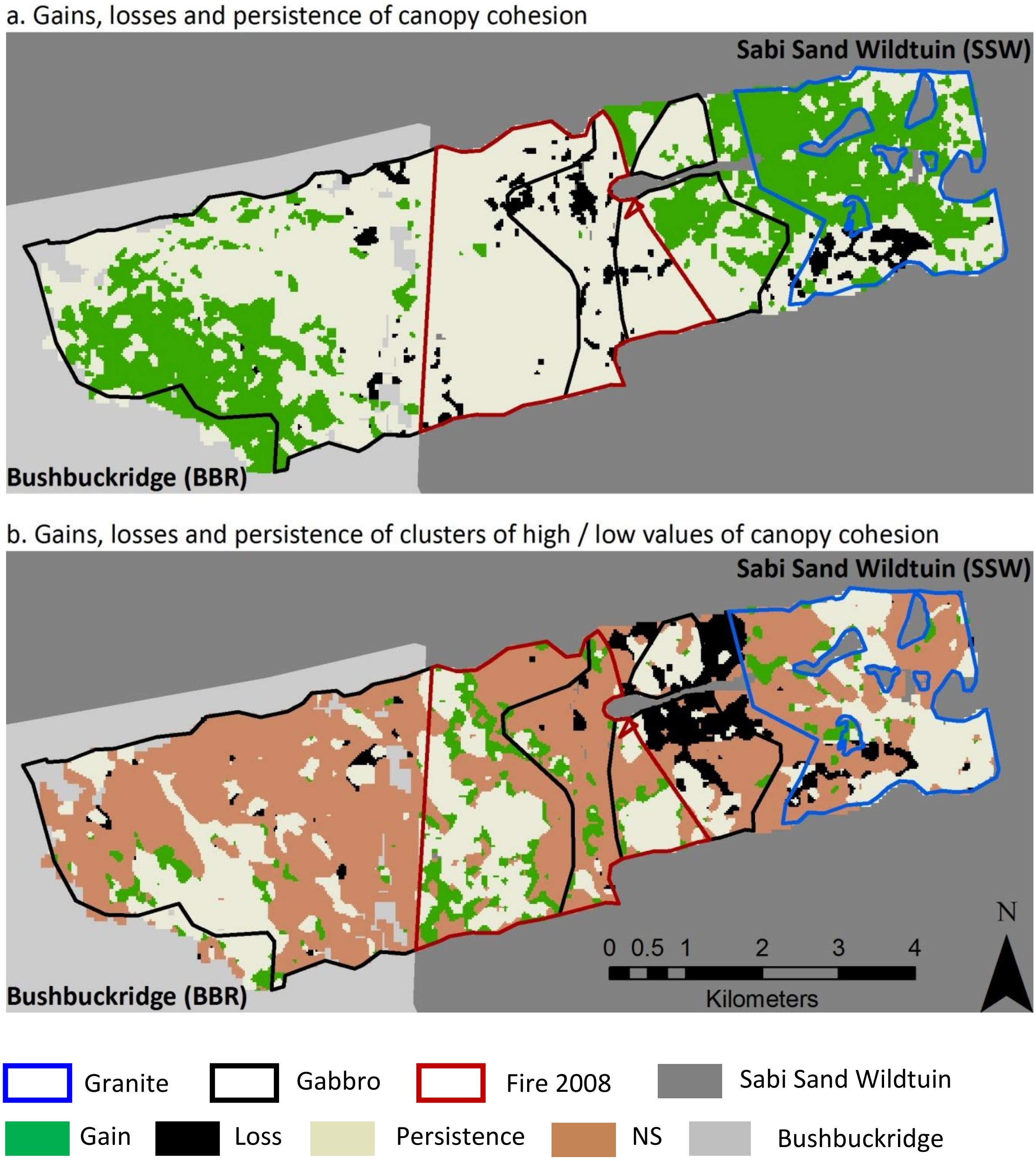


Figure 5.9: Gains, losses and persistence of a. canopy cohesion and b. clusters of high / low values of canopy cohesion from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin Local Morans I indicator of spatial association. Gain indicates an increase in clusters, loss indicates a decrease in clusters and persistence is no change in clusters. NS indicates a persistence of no clusters.

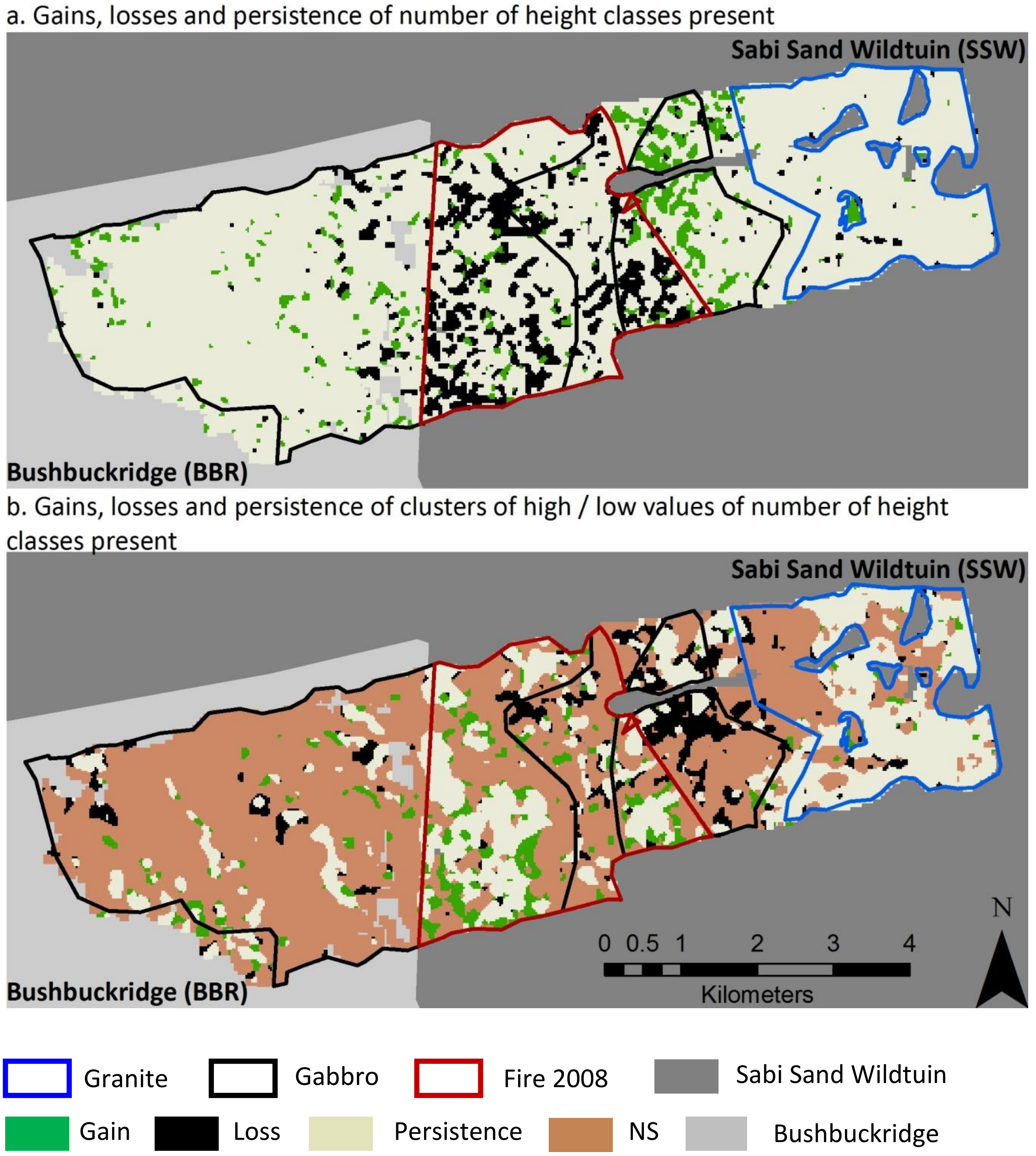


Figure 5.10: Gains, losses and persistence of a. number of height classes present and b. clusters of high / low values of number of height classes present from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin Local Morans I indicator of spatial association. Gain indicates an increase in clusters, loss indicates a decrease in clusters and persistence is no change in clusters. NS indicates a persistence of no clusters.

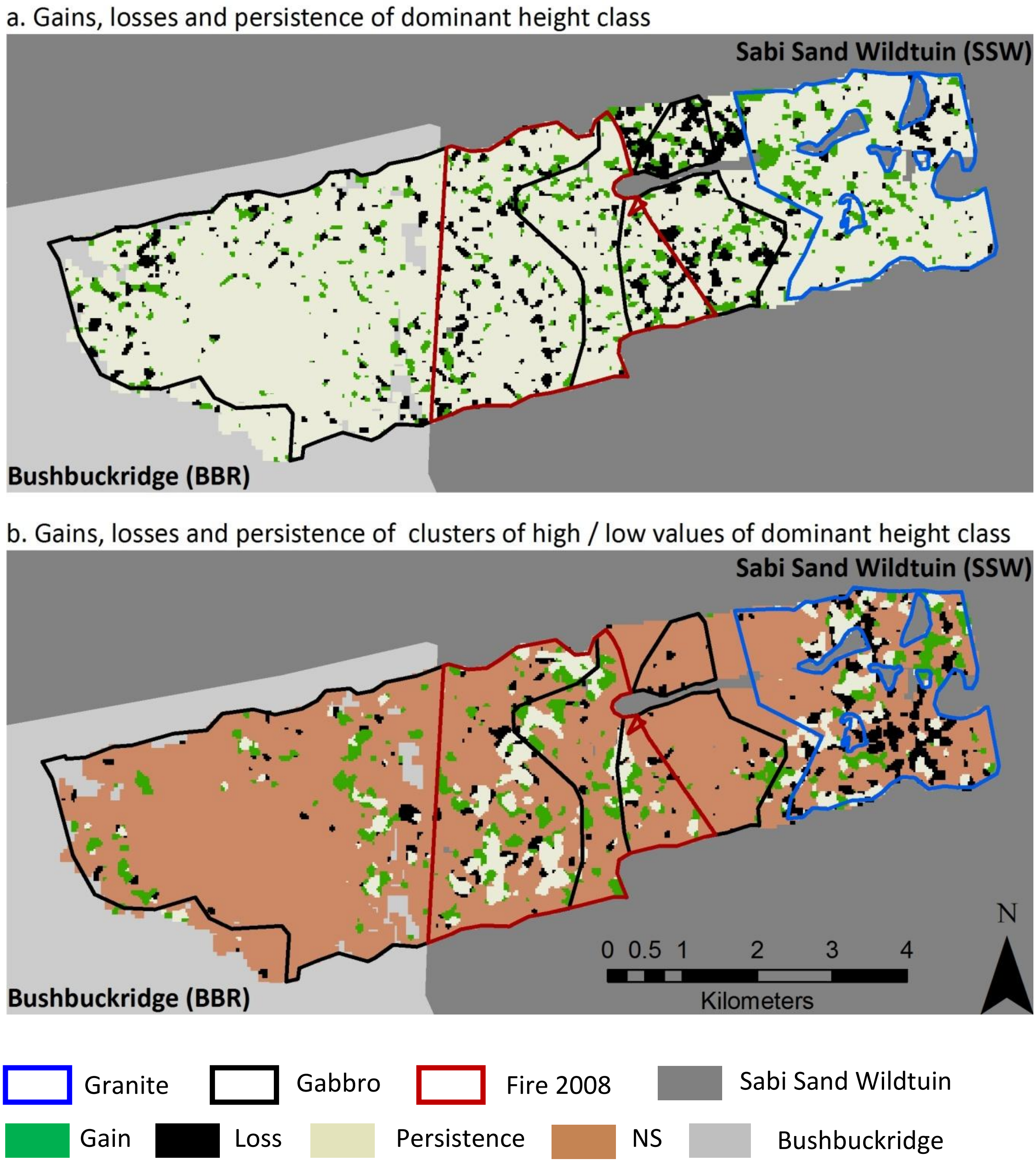


Figure 5.11: Gains, losses and persistence of a. dominant height class and b. clusters of high / low values of dominant height class from 2008 to 2010 in Bushbuckridge (BBR) and Sabi Sand Wildtuin (SSW). Clusters were determined using Anselin Local Morans I indicator of spatial association. A gain in dominant height class means a shift towards taller trees being dominant occurred, and a loss indicates shorter trees became dominant. NS indicates a persistence of no clusters.

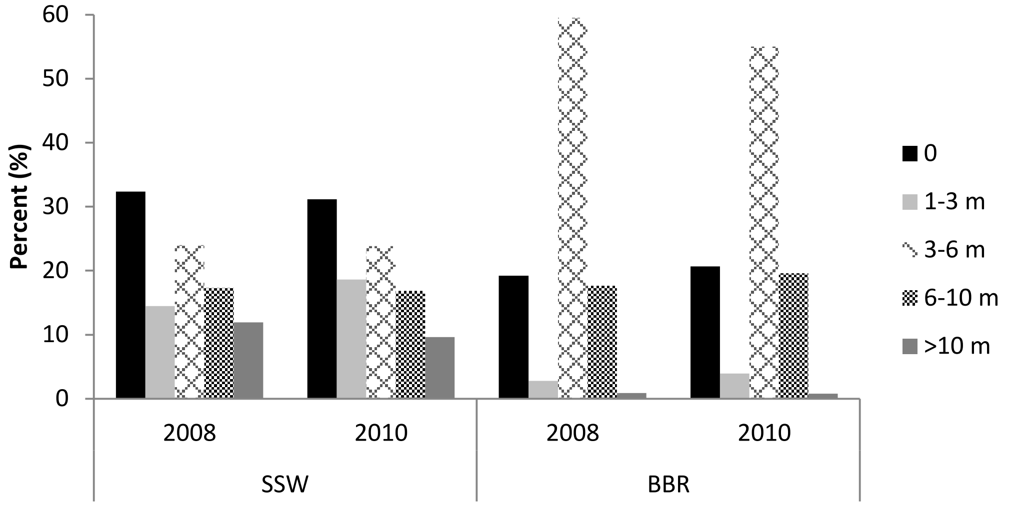
5.4.2 Structural dynamics across land use

Given the relatively short two year time period, noteworthy gains and losses of value for each metric, and changes in how these metrics cluster were observed (Fig. 5.6, Fig. 5.7-

5.11). There was a large decrease in clusters of similar values of canopy cover (Fig. 5.6b) and canopy cohesion (Fig. 5.9b) in SSW, although corresponding gains in the values of these metrics were experienced (Fig. 5.7a & 5.9a respectively). Although block gains in canopy cover and cohesion occurred (Fig. 5.7a & 5.9a respectively), there were corresponding losses of significant clusters of canopy cover (Fig. 5.7b) and canopy cohesion (Fig. 5.9b) in SSW. The gains in cover and cohesion are a result of the gains in height classes <10 m (Fig. 5.6). There was a gain in significant clusters of all metrics (Fig. 5.7 – 5.11) in the burnt areas of SSW with corresponding losses in the value of the metrics (Fig. 5.7 – 5.11) between 2008 and 2010. Although only small gains in dominant height class (i.e. taller trees became more dominant) in the south-east portion of the SSW granites, there were corresponding losses of clusters over the two year time period (Fig. 5.11b). Gains and losses of statistically significant clusters predominantly occurred around existing clusters, i.e. existing clusters of a metric act as a nuclei of change.

The majority of grid cells in SSW had no dominant height classes indicating a more homogeneous mixture of height classes, whereas vegetation in the 3-6 m height class was most dominant in BBR (Fig 5.12a). Although SSW has a greater percent of grid cells with 0, 1, 2 and 4 height classes present than BBR, BBR has a notably higher percent of cells with three height classes present (Fig. 5.12b).

a. Dominant height class



b. Number of height classes present

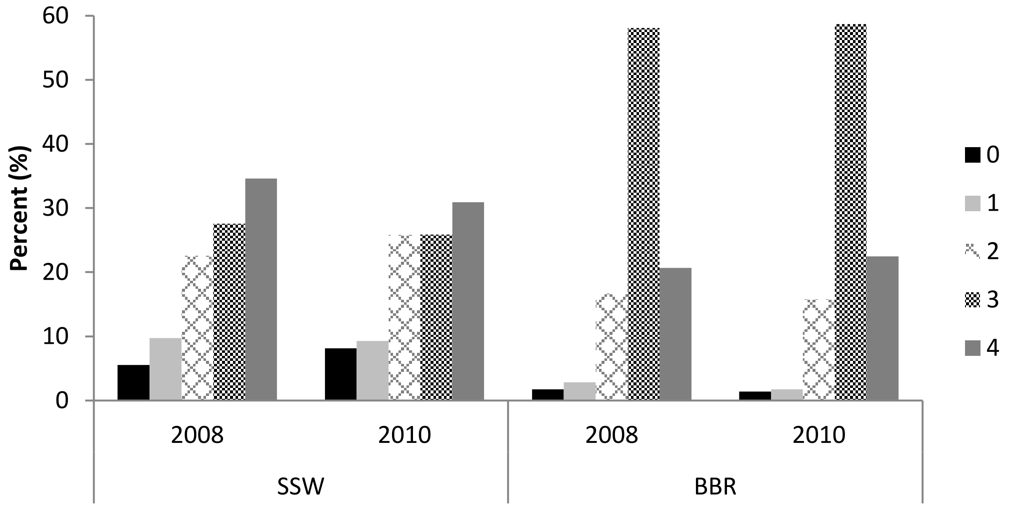


Figure 5.12: Comparisons of a. dominant height classes and b. number of height classes present between Sabi Sand Wildtuin (SSW) and Bushbuckridge (BBR) in 2008 and 2010.

5.5 Discussion

It is a combination of land-use and the biophysical template that affects the distribution of vegetation within each height class (Fig. 5.6-5.12; Fisher et al. 2012; Higgins et al. 1999). However, considering the proximity of the two sites and the similarities in their biophysical templates, differences are due to land use (Fig. 5.12). A 3D approach has been necessary to detect the changes in structure over the two year time period, as a measurement of canopy cover alone does not indicate the changes that are occurring in the understory (Fig. 5.5). Furthermore, persistence in canopy cover, which would be regarded as no change over time using a 2D classification, does not equal persistence in either the vertical or horizontal domain of structure as measured by the number of canopy layers present and cohesion respectively (Fig. 5.3). Similarly, no significant differences occur between mean percent canopy cover for each of the four height classes, a standard 2D measurement (Fig. 5.4). Although phenology may affect results, LiDAR data were collected at the same time of year in 2008 and 2010 to be consistent between monitoring periods. The 0.25 ha grid cell, and the 5% confidence interval of change when calculating gains and losses will further reduce noise in the results.

The presence of vegetation >3 m in BBR is increasing, while SSW shows net losses within these height classes (Fig. 5.6). Tall trees are protected in communal rangelands and special permission is needed to cut them down (Twine 2005). Coupled with recruitment into these taller height classes, the protection explains the gains in these height classes (Fig. 5.5 h) as well as the high percent of persistence especially in trees >10 m in BBR (Fig. 5.6). Fuelwood and fencing poles are harvested from trees predominantly under 3 m (Neke et al. 2006, Twine 2005), thereby resulting in greater losses in vegetation from 1-3 m (Fig. 5.6a). The gains observed in the percent cover of shrubs is due to either coppicing or bush encroachment (Neke 2005). Vegetation within the communal rangelands is therefore increasing with gains exceeding losses in all height classes. While this does point towards densification of the woody layer, it also means there is a greater, and regenerating, wood supply for the rural community.

While anthropogenic use is the predominant cause of change in vegetation structure in the rangelands, changes in the vegetation structure in SWW is affected by fire, herbivory and manual clearing. Although high persistence in vegetation >6 m is evident in SSW, vegetation shows greater losses compared to BBR for height classes >3 m (Fig. 5.6). The tall trees are presently not being effectively conserved in the wildlife area, and the higher percent losses in trees >3 m and especially those >6m in SSW compared to BBR are a result of treefall from elephants (Asner et al. 2012), the effects of which would have been exacerbated by the fire in October 2008. This has been clearly shown for marula trees in the neighbouring southern

Kruger National Park, with some sites showing losses of >25% over the last decade (Helm and Witkowski 2012). The loss of tall trees in SSW is evidence of the increase in elephant densities by 0.012 elephant/ha from 1998-2010 (Hiscocks 1999, M Grover pers comm.) as cautioned by Hiscocks (1999). Thomas et al. (2011) showed elephant spent a large proportion of time in this southern section of SSW from 2003 – 2009, and are likely to account for the loss of tall trees (Fig. 5.6; Helm et al. 2011).

The percent gain of vegetation from 1-3 m and 3-6 m in SSW is almost equal to that of persistence (Fig. 5.6a&b) showing increasing woody vegetation density despite the effects of herbivory and fire, perhaps indicating bush encroachment. The area is prone to bush encroachment as a result of previous cattle farming on the land (http://www.sabisand.co.za/ssw-history.html, Tobler et al. 2010, Papanastasis 2009). When bush encroachment is extensive in protected areas, population numbers of ungulates may decline due to increased predation (as the animals congregate in open areas and there is an increase in good cover for close-range stalking), and a reduction in available forage. Fire is successfully used as a management tool in SSW as a result of the propensity towards bush densification. This can be seen by the gain in significant clusters of high percent of canopy layers present immediately adjacent to the burnt area, but a loss of clusters within the burnt area (Fig. 5.8b). Similarly, canopy cover, cohesion and number of height classes showed losses within the burnt area (Fig. 5.7a, 5.9a & 5.10a). The decrease of canopy cover and canopy cohesion as a result of the fire will affect how animals use the landscape, with most ungulates showing a preference for open spaces (Riginos & Grace 2008, Table 5.1). This phenomenon is advantageous for SSW management which receives revenue from tourism, namely game is more visible in less dense bushveld.

The gain and loss of the various structural metrics from 2008 to 2010 does not necessarily translate into gains or losses of clusters; rather, existing clusters act as nuclei around which new clusters will be formed or clusters will be lost. A gain in canopy cover for example might even mean a loss of clusters (see Fig. 5.7a&b - SSW) indicating the landscape is becoming more heterogeneous as clusters of similar vegetation cover are lost. A gain in clusters around existing clusters can be interpreted as a loss of heterogeneity as there is a spatial aggregation of similar values (either high or low) indicating a more homogeneous landscape.

Management interventions promoting heterogeneity should therefore focus around eliminating clusters of similar vegetation, e.g. as occurs with bush encroachment. Patches of structurally similar vegetation are likely to be less resilient to change.

The communal rangeland shows greater structural diversity than SSW in terms of dominant height classes and number of life forms present (Fig. 5.11a) and temporal stability in structural metrics (Fig. 5.7-5.12). However this finding is probably specific to this low intensity use rangeland site, which is only used by one village. Surrounding communal rangelands under more intense use have been found to exhibit reduced structural diversity, and in some cases are completely degraded (Fisher et al. 2012). Human effects on structure are strongest within 1 km of a village (Wessels et al. 2011) and this rangeland extends to 4 km, therefore it is still relatively intact (Fisher et al. 2012). The low intensity use rangelands are still intact habitats, with the structural diversity present potentially improving the rural landscapes' resilience to change (Fischer et al. 2006). Higher structural diversity has been found to increase ecosystem function and biodiversity (Ishii et al. 2004, Hooper et al. 2005). However, with current levels of development in rural areas, and in BBR in particular (Coetzer et al. 2010), effective management of intact vegetation is essential. Intermediate levels of disturbance have been shown to increase diversity (Shackleton 2000); however, the legacy of cattle farming, increasing elephant densities and intense management of SSW provides a level of disturbance that has decreased structural stability and heterogeneity. In order to manage both areas with the goal of maintaining heterogeneity and biodiversity, Fischer and Lindenmayer (2007) recommend the maintenance and/or restoration of matrices that are structurally similar to native vegetation in order to provide habitat for species, habitat connectivity and reduce the structural contrast between modified and unmodified areas.

5.5.1 Conclusions

A high level of detail such as that provided by a parametric classification is necessary to detect modifications or changes in land cover (Jansen and Di Gregorio 2002). Global or regional land cover classifications such as the National Land cover Classification (NLC) of South Africa (Thompson 1996) and the Global Land Cover Classification (Hansen et al. 2000) define classes based on broad ranges of vegetation cover. Due to the extent of the area covered, and the resolution of these classifications, each class contains a high degree of variability and changes within these classes over time may not be detected. If the change is detrimental to ecosystem functioning and services, the change may be detected too late to rectify with management. Not only is a finer level of detail required to define classes, additional metrics are necessary to properly define and detect change (Fisher et al. 2013). Percent cover may remain constant (or within a range as defined by the cover class) while the spatial arrangement of the cover changes, and/or the structural composition including the vertical arrangement of understory plants is altered (Fig. 5.3). Processes such as bush encroachment within a tall wooded area would not be detected using a classification that only measures the aerial extent of cover (Jansen and Di Gregorio 2002), as the amount of vegetation present within a height class is often higher than what is seen from the top of canopy (Fig. 5.5), yet it is the sub-canopy that is utilised or affected by disturbance. Hence, even though short term change in savanna woody vegetation structure are generally fine scale changes and not readily apparent using 2D methods, we clearly show that we can successfully monitor these dynamics using a 3D classification applied to LiDAR data. Future work could be done to test these relationships across a greater variety of sites spanning a temperature and rainfall gradient.

5.6 Acknowledgements

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5.7 References

Anselin, L. (1995) Local indicators of spatial association—LISA. *Geographical analysis* **27**: 93-115

Asner, G.P., Archer, S.A., Hughes, R.F., Ansley, J.N., and Wessman, C.A. (2003) Net changes in regional woody vegetation cover and carbon storage in North Texas rangelands, 1937-1999. *Global Change Biology* **9**: 316-335.

Asner, G.P., Knapp, D.E., Kennedy-Bowdoin, T., Jones, M.O., Martin, R.E., Boardman, J. and Field, C.B. (2007) Carnegie Airborne Observatory: in-flight fusion of hyperspectral imaging and waveform light detection and ranging (wLiDAR) for three-dimensional studies of ecosystems. *Journal of Applied Remote Sensing* **1**, 013536

Asner, G.P., Hughes, R.F., Vitousek, P.M., Knapp, D.E., Kennedy-Bowdoin, T., Boardman, J., Martin, R.E., Eastwood, M., and Green, R.O. (2008) Invasive plants alter 3-D structure of rainforests. *Proceedings of the National Academy of Sciences* **105**: 4519-4523

Asner, G.P. and Levick, S.R. (2012) Landscape-scale effects of herbivores on treefall in African savannas. *Ecology Letters* **15:** 1211-1217

Atauri, J.A. and de Lucio, J.V. (2001) The role of landscape structure in species richness distribution of birds, amphibians, reptiles and lepidopterans in Mediterranean landscapes. *Landscape Ecology,* 16: 147–159

Balvanera, P., Pfisterer, A.B., Buchman, N, He, J-S., Nakashizuka, T., Raffaelli, D. and Schmid, B. (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters* **9**: 1146-1156

Banks, D.I., Griffin, N.J., Shackleton, C.M., Shackleton, S.E. and Mavrandonis, J.M. (1996) Wood supply and demand around two rural settlements in a semi-arid savanna, South Colgan et al 2012Africa. *Biomass Bioenergy* **11**: 319-331

Beater, M.M.T., Garner, R.D. and Witkowski, E.T.F. (2008) Impacts of clearing invasive alien plants from 1995 to 2005 on vegetation structure, invasion intensity and ground cover in a temperate to subtropical riparian ecosystem. *South African Journal of Botany* **74**: 495-507

Brink, A.B. and Eva, H.D. (2009) Monitoring 25 years of land cover change dynamics in Africa: A sample based remote sensing approach. *Applied Geography* **29**: 501-512

Buitenwerf, R., Bond, W.J., Stevens, N. And Trollope, S.W. (2012) Increased tree densities in South African savannas: >50 years of data suggests CO_2 as a driver. *Global Change Biology* **18**: 675-684

Coetzer, K.L., Erasmus, B.F.N., Witkowski, E.T.F. and Bachoo, A.K. (2010) Land cover change in the Kruger to Canyons Biosphere Reserve (1993-2006): A first step towards creating a conservation plan for the subregion. *South African Journal of Science* **106**: 26-35

Coetzer, K.L., Witkowski, E.T.F., Erasmus, B.F.N., Reyers, B. (2013). The race for space: Tracking land-cover transformation in a socio-ecological landscape. *Environmental Management* DOI: 10.1007/s00267-013-0094-9

Di Gregorio, A. and Jansen, L.J.M. (2000) Land Cover Classification System (LCCS): Classification concepts and user manual LCCS version 1.0. GCP/RAF/287/ITA Africover, Food and Agriculture Organization of the United Nations Publishing Service, Rome

Fisher, J.T., Witkowski, E.T.F., Erasmus, B.F.N., van Aardt, J., Asner, G.P., Wessels, K.J. and Mathieu, R. (2012) Human-modified landscapes: patterns of fine-scale woody vegetation structure in communal savannah rangelands. *Environmental Conservation* **39**: 72-82

Fisher, J.T., Erasmus, B.F.N., Witkowski, E.T.F, van Aardt, J., Wessels, K.J. and Asner, G.P. (2013) Savanna woody vegetation classification – now in 3D. *Applied Vegetation Science* DOI: 10.1111/avsc.12048

Fischer, J., Lindenmayer, D.B. and Manning, A.D. (2006) Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. *Frontiers in Ecology and Environment* **4**: 80-86

Fischer, J. and Lindenmayer, D.B. (2007) Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* **16**: 265-280

Freitag-Ronaldson, S. and Foxcroft, L.C. (2003) Anthropogenic influences at the ecosystem level. In Du Toit, Johan T., Rogers, Kevin H. and Biggs, Harry C. (Eds) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press: 391-421

Garner, R.D. and Witkowski, E.T.F. (1997) Changes in plant diversity after clearing of alien plants along riverine corridors in Mpumalanga, South Africa. Report to the Mpumalanga Water Conservation Programme, p. 34.

Giannecchini, M., Twine, W. and Vogel, C. (2007) Land-cover change and human-environment interactions in a rural cultural landscape in South Africa. *The Geographical Journal* **173**: 26-42

Hall, F.G., Bergen, K., Blair, J.B., Dubayah, R., Houghton, R., Hurtt, G., Kellndorfer, J., Lefsky, M., Ranson, J., Saatchi, S., Shugart, H.H. and Wickland, D. (2011) Characterizing 3D vegetation structure from space: Mission requirements. *Remote Sensing of Environment* **115**: 2753-2775

Hansen, M.C., DeFries, R.S., Townshend, J.R.G. and Sohlberg, R. (2000) Global land cover classification at 1 km spatial resolution using a classification tree approach. *International Journal of Remote Sensing* 21: 1331-1364

Hector, A. and Bagchi, R. (2007) Biodiversity and ecosystem multifunctionality. *Nature Letters* **448**: 188-191

Helm, C., Wilson, G., Midgley, J., Kruger, L. and Witkowski, E.T.F. (2011) Investigating the vulnerability of an African savanna tree (*Sclerocarya birrea* ssp. *caffra*) to fire and herbivory. *Austral Ecology* **36**: 964-973

Higgins, S.I., Shackleton, C.M., and Robinson, E.R. (1999) Changes in woody community structure and composition under constrasting landuse systems in a semi-arid savanna, South Africa. *Journal of Biogeography* **26**:619-627

Hiscocks, K. (1999) The impact of an increasing elephant population on the woody vegetation in southern Sabi Sand Wildtuin, South Africa. *Koedoe* **42**: 47-55

Hooper, D.U., Chapin III, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A.J., van der Meer, J. and Wardle, D.A. (2005) Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs* **75**: 3-35

Ishii, H. T., Tanabe, S. and Hiura, T. (2004) Exploring the relationships among canopy structure, stand productivity, and biodiversity of temperate forest ecosystems. *Forest Science* **50**: 342–355

Jansen, L.J.M. and Di Gregorio, A. (2002) Parametric land cover and land-use classifications as tools for environmental change detection. *Agriculture, Ecosystems and Environment* **91**: 89-100

Lefsky, M.A., Cohen, W.B., Parker, G.G. and Harding, D.J. (2002) Lidar remote sensing for ecosystem studies. *BioScience* **52**: 19-30

Lumsden, L. F. and Bennett, A. F. (2005) Scattered trees in rural landscapes: foraging habitat for insectivorous bats in south-eastern Australia. *Biological Conservation* **122**: 205–222

Luoga, E.J., Witkowski, E.T.F. and Balkwill, K. (2004) Regeneration by coppicing (resprouting) of miombo (African savanna) trees in relation to land use. *Forest Ecology and Management* **189**: 23-35

Luoga, E.J., Witkowski, E.T.F and Balkwill, K. (2005) Land cover and use changes in relation to the institutional framework and tenure of land and resources in eastern Tanzania Miombo woodlands. *Environment, Development and Sustainability* **7**: 71-93

MA (2005) Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC

MacArthur, R.H., and MacArthur, J.W. (1961) On bird species diversity. *Ecology* **42**: 594-598

Manning, A. D., Fischer, J. and Lindenmayer, D. B. (2006) Scattered trees are keystone structures – Implications for conservation. *Biological Conservation* **132**: 311-321

Matsika, R., Erasmus, B.F.N., and Twine, W. (2013) A tale of two villages: assessing the dynamics of fuelwood supply in communal landscapes in South Africa. *Environmental Conservation* **40**:71-83, DOI: http://dx.doi.org/10.1017/S0376892912000264

McGarigal, K., Cushman, S.A., Neel, M.C., and Ene, E. (2002) FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available from http://www.umass.edu/landeco/research/fragstats/fragstats.html

Mucina, L. and Rutherford, M.C (Eds.) (2006) The vegetation of South Africa, Lesotho and Swaziland, Pretoria, *Strelitzia 19. South African National Biodiversity Institute*, pp. 439-539

Mwavu, E.N. and Witkowski, E.T.F. (2008) Sprouting of woody species following cutting and tree-fall in a lowland semi-deciduous tropical rainforest, North-Western Uganda. *Forest Ecology and Management* **255:** 982-992

Naeem, S. (2002) Ecosystem consequences of biodiversity loss: the evolution of a paradigm. *Ecology* **83**: 1537-1552

Neke, K.S. (2005) Savanna woody plant demography: Contrasting the effects of protection and communal subsistence land use on recruitment potential and resource sustainability. In "The regeneration ecology of savanna woodlands in relation to human utilisation." PhD thesis, University of the Witwatersrand, pp 24-59

Neke, K.S., Owen-Smith, N. and Witkowski, E.T.F. (2006) Comparative resprouting response of Savanna woody plant species following harvesting: the value of persistence. *Forest Ecology and Management* **232**: 114-123

Oba, G., Post, E., Syvertsen, P. O. and Stenseth, N. C. (2000) Bush cover and range condition assessments in relation to landscape and grazing in southern Ethiopia. *Landscape Ecology,* 15: 535–546

Palminteri, S., Powell, G.V.N., Asner, G.P. and Peres, C.A. (2012) LiDAR measurements of canopy structure predict spatial distribution of a tropical mature forest primate. *Remote Sensing of Environment* **127**: 98-105

Papanastasis, V.P. (2009) Restoration of Degraded Grazing Lands through Grazing Management: Can It Work? *Restoration Ecology* **17**: 441-445

Pollard, S., Shackleton, C. and Curruthers, J. (2003) Beyond the fence: People and the lowveld landscape. In Du Toit, Johan T., Rogers, Kevin H. and Biggs, Harry C. (Eds) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press: 422-446

Popescu, S.C. and Zhao, K. (2008) A voxel-based LiDAR method for estimating crown base height for deciduous and pine trees. *Remote Sensing of Environment* **112**: 767-781

Riginos, C. and Grace, J. B. (2008) Savanna tree density, herbivores, and the herbaceous community: bottom-up vs. top-down effects. *Ecology,* 89(8): 2228–38

Ripple, W.J., and Beschta, R.L. (2004) Wolves and the ecology of fear: Can predation risk structure ecosystems? *BioScience* **54**: 755-766

Sala, O.E., Chappin III, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., LeRoy Poff, N., Sykes, M.T., Walker, B.H., Walker, M and Wall, D.H. (2000) Global biodiversity scenarios for the year 2100. *Science* **287**: 1170-1174

Seymour, C. L. and Dean, W. R. J. (2009) The influence of changes in habitat structure on the species composition of bird assemblages in the southern Kalahari. *Austral Ecology*, 35(5): 581–592

Shackleton, C.M., Griffin, N.J., Banks, D.I., Mavrandonis, J.M. and Shackleton, S.E. (1994) Community structure and species composition along a disturbance gradient in a communally managed South African savanna. *Vegetatio* **115**: 157-167

Shackleton, C.M. (2000) Comparison of plant diversity in protected and communal lands in the Bushbuckridge lowveld savanna, South Africa. *Biological Conservation* **94**: 273 – 285

Smart, R., Whiting, M. J. and Twine, W. (2005) Lizards and landscapes: integrating field surveys and interviews to assess the impact of human disturbance on lizard assemblages and selected reptiles in a savanna in South Africa. *Biological Conservation* **122**: 23–31

Stats SA (2012) http://interactive.statssa.gov.za/superweb/loadDatabase.do;jsessionid=3038C871551C2B4317A15FC54B4BFFBC?db=Descriptive11_wd

Tews, J., Brose, U., Grimm, V., Tielborger, K., Wichmann, M.C., Schwager, M. and Jeltsch, F. (2004) Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. *Journal of Biogeography* **31**: 79-92

Thomas, B., Holland, J.D. and Minot, E.O. (2011) Seasonal home ranges of elephants (*Loxodonta Africana*) and their movements between Sabi Sand Reserve and Kruger National Park. *African Journal of Ecology* **50**: 131-139

Thompson, M. (1996) A standard land-cover classification scheme for remote sensing applications in South Africa. *South African Journal of Science* **92**: 34-42

Thornton, R. (2002) Environment and land in Bushbuckridge, South Africa

Tobler, M.W., Cochard, R. and Edwards, P.J. (2010) The impact of cattle ranching on large-scale vegetation patterns in a coastal savanna in Tanzania. *Journal of Applied Ecology* **40**: 430-444

Trollope, W.W., Trollope, L.A., Biggs, H.C., Pienaar, D and Potgieter, A.L.F. (1998) Long-term changes in the woody vegetation of the Kruger National Park, with special reference to the effects of elephants and fire. *Koedoe* **41**: 103-112

Twine, W., Moshe, D., Netshiluvhi, T. and Siphugu, V. (2003) Consumption and direct-use values of savanna bio-resources used by rural households in Mametja, a semi-arid area of Limpopo province, South Africa. *South African Journal of Science* **99**: 467–473

Twine, W.C. (2005) Socio-economic transitions influence vegetation change in the communal rangelands of the South African lowveld. *African Journal of Range and Forest Science* **22**: 93-99

Wessels, K.J., Mathieu, R., Erasmus, B.F.N., Asner, G.P., Smit, I.P.J., van Aardt, J., Main, R., Fisher, J., Marais, W., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R. and Jacobson, J. (2011) Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* **261**: 19-29

http://www.sabisand.co.za/ssw-history.html, accessed January 2013

Chapter 6 : Synthesis

6.1 Introduction

Protected areas do not occur in isolation in the landscape, and are often adjacent to land uses that are not aligned with conservation principles such as agriculture, mining or human settlements (e.g. Kruger National Park in the Kruger to Canyons (K2C) Biosphere Reserve; Coetzer et al. 2010). Communal rangelands within rural areas are a threatened land-cover as a result of settlement expansion due to population growth and development (Coetzer et al. 2010; Coetzer et al. 2013; Matsika 2012); and are being degraded as a result of the communities' continued high reliance on natural resources (Dovie et al. 2002; Twine 2005). This investigation of woody vegetation structure in communal rangelands and protected areas revealed management of resources, including direct use of fuelwood and management of herbivory and fire, affects woody structural dynamics.

The spatio-temporal dynamics and drivers of woody vegetation in savannas have been the explicit focus of many studies (Gillson 2004; Levick & Rogers 2011; Sankaran et al. 2005; Sankaran et al. 2008; Scholes & Walker 1993; Scholes & Archer 1997). Recently, the context specific effects of fire (Levick et al. 2012) and herbivory (Asner & Levick 2012; Levick & Asner 2013) in protected areas have been identified. However, what is still lacking is an understanding of the social in addition to the ecological context of natural resource management in both protected areas and communal rangelands. This thesis presents an explicit analysis of the use of LiDAR in savannas and the advantage of measuring vegetation structure in three dimensions (3D) over two dimensions (2D) (section 6.2). I show how woody vegetation structure across multiple land use types and intensities allows for a greater understanding of the context of woody structural patterns and dynamics in human-modified landscapes contributing to better management of natural resources for conservation and sustainable use (section 6.3).Finally, I discuss the necessity of savanna management and interdisciplinary biodiversity research in a global context (section 6.4). This chapter is a discussion of how LiDAR can be successfully used to provide context to advance our understanding of the effects of management of natural resources on spatio-temporal patterns of 3D woody vegetation structure across land uses in a heterogeneous semi-arid savanna system; making a valuable contribution to the field of earth observation for biodiversity management and conservation.

6.2 LiDAR as a monitoring tool for management of woody vegetation structure and biodiversity in semi-arid savannas

The use of LiDAR for vegetation measurement and monitoring has its roots in forestry (Hudak et al. 2008; Lefsky et al. 1999; Lefsky et al. 2002; Pascual et al. 2008; Skowronski et al. 2007). LiDAR has been successfully used in temperate and tropical environments, landscapes which typically have low species richness and discrete vertical canopies, characteristics which make it easier to construct structural metrics (Drake & Weishampel 2000, 2001) and measure biomass. Even though African savannas are complex systems with intricate vertical architecture, LiDAR has been used successfully to measure the effects of herbivory (Asner et al. 2009; Asner & Levick 2012; Levick & Asner 2013) , fire (Smit et al. 2010; Levick et al. 2012), a combination of the two (Levick et al. 2009), and the effect of land management and land-use (Chapter 2, 3 (Fisher et al. 2012) & 4 (Fisher et al. 2013), Wessels et al. 2011; Wessels et al. 2013) on woody vegetation structure.

Underestimation of tree height below 2 m in the 2008 LiDAR dataset (Fig. 3.3, Chapter 3) was a result of low leaf area index in 2008 and the processing techniques used on the low resolution (1.12 m versus 56 cm resolution) data. This issue has since been corrected with different processing techniques in 2010 and the launch of the new CAO-2 AToMS (Airborne Taxonomic Mapping System) sensor in 2011 (Asner et al. 2012). The vertical uncertainty/error on the CAO-2 LiDAR is less than 15 cm (Asner et al. 2012). Although the underestimation of tree height is no longer an issue with the CAO LiDAR data, data obtained using a spaceborne sensor such as GLAS (Geosciences Laser Altimeter System) on board the ICESat (Ice, Cloud and Land Elevation Satellite) can have a 1-2 m vegetation height error on surfaces with zero slope with increasing error on steep slopes (Hall et al. 2011). For global applications these height errors may be acceptable; however, for fine scale applications such as mapping fuelwood availability, the data could be inadequate. To date, a high resolution spaceborne LiDAR sensor has not been successfully launched.

The Centre for LiDAR Information Coordination and Knowledge (CLICK), managed by the USGS (United States Geological Survey) was started as a result of the demand to use all information derived from LiDAR and not just digital elevation models (http://lidar.cr.usgs.gov/, accessed February 2013). With the realisation that LiDAR costs and the technical learning curve associated with using the entire LiDAR point cloud are prohibitive to research, CLICK's focus is on facilitating data access and LiDAR education to make LiDAR more readily available. The high costs and processing overheads mean LiDAR is not typically available to conservation and land-use planners; however, the value gained from the high spatial coverage of LiDAR which provides socio-ecological context of woody vegetation structural patterns is important. I developed the woody vegetation structural classification (Chapter 4) as a user friendly, simplified product with the needs of land-use planners and conservation managers in mind. The height classes and metrics used are ecologically relevant for a wide range of savannas; they account for the high horizontal and vertical heterogeneity characteristic of savannas and are applicable to both protected areas and communal rangelands. Ultimately, the structural classification could be produced as a LiDAR-derived product available to land-use managers, which is more cost-effective than collecting the same data in the field. Potential applications of the classification include sustainable resource extraction plans, site specific management of areas of concern in protected areas, assessing restoration success and identifying areas for restoration or development.

6.3 Advancing understanding of management through explicit consideration of socio-ecological context

The term socio-ecological systems (SES) refers to all human-environment relationships, including both those in communal rangelands where people are dependent on natural resources for their livelihoods, as well as protected areas where conservation is a priority and people interact with the environment through tourism and management (Table 6.1). I refer to Ostrom's (2009) framework of SES being divided into the resource system, resource units, users and governance in Chapter 1 (Fig. 1.1). The SES dealt with in this thesis can be categorized according to the framework (Table 6.1). The structure, function and composition of woody vegetation (resource unit) is a defining feature of savannas (resource system) (Table 6.1, Sankaran et al. 2005; Scholes & Archer 1997). Society is concerned about the state of the resource system because of its value, in terms of the productivity of the system for subsistence livelihoods such as those practiced in rural areas (Dovie et al. 2002), as well as their conservation value in protected areas (users in the SES). However, the governance of the SES determines how effectively the users interact with the resource system and resource units (Fig. 1.1, Table 6.1). Understanding these complex interactions is necessary for better informed management decisions that are specifically relevant to the local community ensuring long term sustainability.

The advantage of investigating woody structural patterns and dynamics in protected areas and communal rangelands is that it allows for a more cohesive understanding of how management for different uses affects savannas. Although the abiotic template is a key determinant of savanna structure (Venter et al. 2003), the social context of use (users and governance of the system; Table 6.1) is the ultimate determinant of vegetation patterns (for example see Chapter 5). This thesis is not simply an analysis of woody vegetation structure in communal rangelands (Chapter 3) and protected areas (Chapter 2), but also includes patterns of vegetation structure after a land use succession from cattle farming (which is also practiced in communal rangelands) to conservation (Chapter 2) and a comparison of structure and change in structure over time in the two types of land-uses (Chapter 5). Such comparisons are necessary to differentiate between the effects of management and history of use (Table 6.1) and the abiotic template on vegetation structure as well as how each separate SES affects the other (Fischer & Lindenmayer 2007; Ostrom 2009). This thesis therefore contributes towards a holistic understanding of two SES which will lead to sustainable management of their natural resources.

A broad comparison of the vertical distribution of woody vegetation shows different patterns in communal rangelands and protected areas (Fisher et al. 2009; Appendix 6.1). Each site within the two land uses also shows very different vertical profiles (intra-land-use variability; Chapter 2, 3, 5; Fisher et al. 2009; Appendix 6.2) and spatial patterns of structural metrics (Chapter 4 [Appendix S4.2a-f]; Chapter 5) depending on their topography (Chapter 3), management (Chapter 2) and intensity of use (Chapter 3). Even when merely examining percent canopy cover of woody vegetation across the land-use gradient, it is evident that factors other than land-use alone are responsible for spatial patterns (Fig. 6.1). For example, canopy cover is similar for parts of site 1 (statutory protected area - KNP), 4 (private game reserve – SSW) and 5 (communal rangeland – BBR). The work presented in this thesis highlights the importance of context not only when investigating abiotic influences (Levick & Rogers 2011) but also when considering SES (Table 6.1).

While communal rangelands are thought to be impacted by resource harvesting and human use, the gains from 2008 to 2010 in the amount of woody vegetation present >3 m in a low intensity use site were up to 2 x greater than in the neighbouring protected area, which also showed corresponding losses up to 3 x more than in the communal rangeland (Chapter 5). This trend will not hold true for all rangelands, as this study (Chapter 3) as well as previous efforts (Giannecchini et al. 2007; Shackleton et al. 1994) show that resource extraction is not constant across the landscape. The resource system in communal rangelands is spatially variable (Table 6.1), with areas under higher utilisation pressure showing a depletion of resources (Chapter 3). Disturbance gradients around settlements are evident, although they diminish at higher levels of utilization (inferred from population/density of settlements utilizing the rangeland, Table 4.1, Chapter 4). The result is a landscape mosaic of heterogeneous and homogeneous rangelands. While heterogeneity at a large scale may persist (Fig. 6.1), the fine scale losses of heterogeneity are pre-cursors to block losses of vegetation within the K2C Biosphere Reserve (Matsika et al. 2013; Coetzer et al. 2010). A decline in disturbance gradients resulting in a more structurally homogeneous landscape should therefore be seen as an early warning sign of woodland degradation. Monitoring of these gradients and an understanding of vegetation structural patterns within the context of knowledge of SES, such as short term maximisation of resources for personal use in communal rangelands (Table 6.1), should be incorporated into land-use and sustainable resource use planning.

Table 6.1: Contrasts between the core subsystems within two socio-ecological systems (SES) in semi-arid savannas in South Africa, namely communal rangelands and protected areas

	Communal Rangelands	Protected Areas
	Resource system	
Biome	Savanna	Savanna
Clarity of resource boundaries	Defined (village boundaries, ignored at times)	Defined (Park fences)
Size of resource system	Small, limited by settlement expansion (Coetzer et al. 2010; Matsika 2012)	2 200 000 ha - KNP 65 000 ha - SSW
Productivity of resource system	Productive, but declining	Productive
Predictability of system dynamics	Disturbance driven system, predictable within constraints (Chapter 3 & 6)	Disturbance driven system, predictable within constraints (Chapter 2 & 6)
Location	Bushbuckridge, Limpopo Province, South Africa	Kruger National Park & Sabi Sand Wildtuin, Limpopo Province, South Africa
	Resource units	
Resource unit	Woody vegetation	Woody vegetation
Growth or replacement rate	Coppice, recruitment to adult size classes (not necessarily reproductively mature) (Luoga et al. 2004; Mwava & Witkowski 2009; Neke et al. 2006)	Coppice, fast growth to escape fire, herbivore and frost trap (Scholes & Archer 1997; Whitecross et al. 2012)
Economic value	Fuelwood, edible herbs and thatch grass contribute 80.6% total direct use value of annual household income (Dovie et al. 2002)	Tourism contributed 7.9% to national GDP in 2009 (http://www.info.gov.za/aboutsa/tourism.htm, accessed 14 Jan 2013)

	Communal Rangelands	Protected Areas
Spatio-temporal distribution	Vegetation structure and composition varies with catenal position, geology and climate. Temporally vegetation is affected by temperature, rainfall, herbivory, fire and harvesting (Gillson 2004; Picket et al. 2003; Sankaran et al. 2005; Neke et al. 2006)	Vegetation structure and composition varies with catenal position, geology, climate. Temporally its affected by temperature, rainfall, herbivory and fire (Gillson 2004; Pickett et al. 2003; Sankaran et al. 2005)
	Governance system	
Governance organisations	Local government, municipality, tribal authorities	SANParks (KNP), Local conservation agencies
Property rights systems	No land ownership by individuals, managed by local authority	National land (KNP) and private land (SSW)
	Users	
Number of users	Many – residents dependant on natural resources	Few - tourists
Socioeconomic attributes of users	Low income, rural users	Middle/high income holiday makers
History of use	Pre-1913: Rural landscape, low population density, no land ownership 1913: Designated homelands under Apartheid Natives Land Act (No. 27) (Gazankulu & Lebowa) (Thornton 2002) 1994: Region divided into Tribal Trust Lands, ruled by Tribal Authority. High population density (209 people/km^2 in 2012; Stats SA 2012). No private land ownership, communal resource use	1922-1934: Present day SSW called Sabi Ranch, owned by Transvaal Consolidated Lands (TCL) and used for cattle farming 1926: Present day KNP proclaimed a national protected area (National Parks Act). 1938: Cattle shot on Sabi Ranch due to foot-and-mouth disease outbreak. 1965: SSW formally proclaimed a conservancy. Private ownership, game viewing safaris

	Communal Rangelands	Protected Areas
Location	Bushbuckridge	Kruger National Park, Sabi Sand Wildtuin
Knowledge of SES/mental models	Short term maximisation of resources for self, but also understand resources are declining (Twine 2005)	Understand importance of conservation
Dependence on resource	Rural communities are highly dependent on woodland resources (Dovie et al. 2002; Twine et al. 2003, Matsika et al. 2013)	People are not dependant on the resource for direct use, rather vegetation is a necessity for conservation, providing cultural value
	Interactions	
Harvesting levels of diverse users	Fuelwood continues to be primary source of energy despite electrification (Matsika et al. 2013)	No harvesting, but clearing in the private reserve to improve visitor amenity (improved game viewing)
Information sharing among users	Information shared about location of resource	Strategic adaptive management (Rogers 2003)
Conflicts among users	Present: limited resource base and high demand	Conservation priorities versus needs of rural communities
	Outcomes	
Social performance measures	Accountability of tribal authority, equity of resource use	Accountability of SANParks with regard to conservation and social responsibility
Ecological performance measures	Mosaic of overharvested and intact areas	Landscape heterogeneity

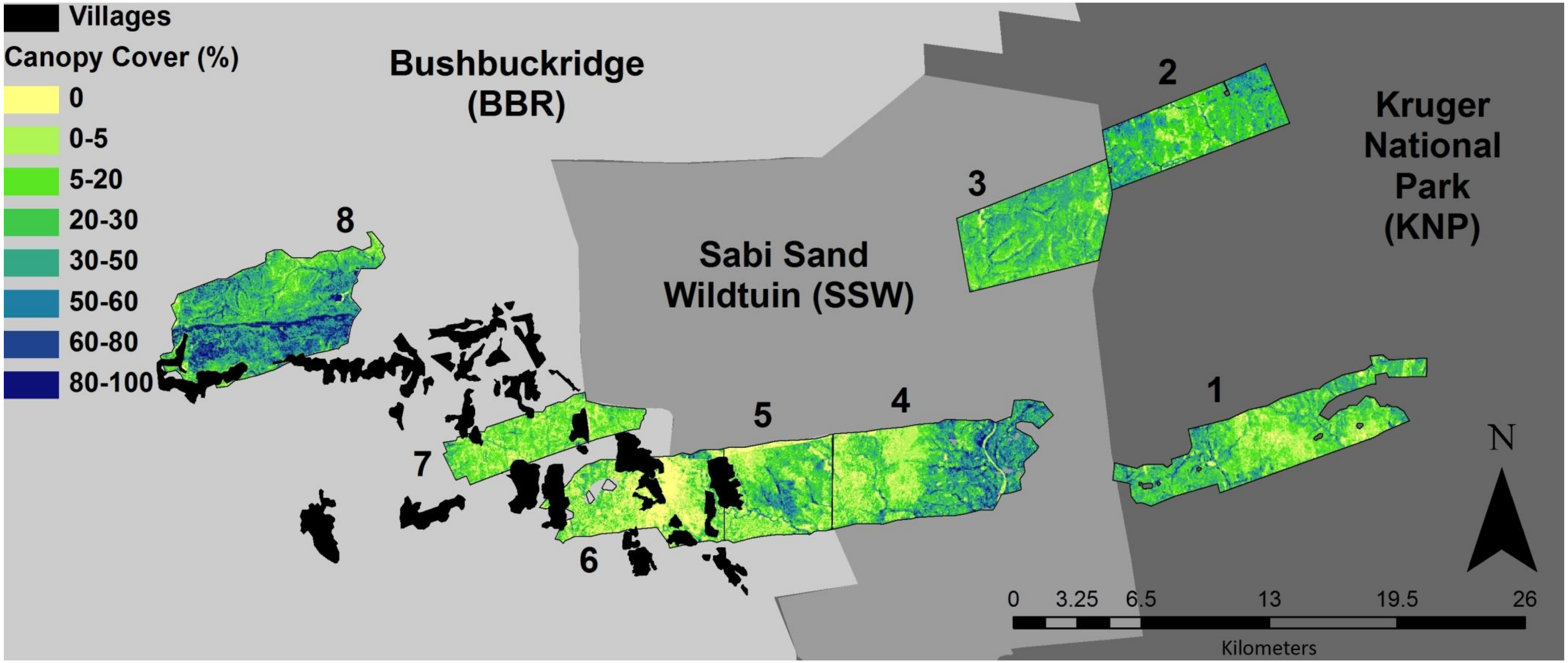


Figure 6.1: Percent canopy cover present in 0.25 ha grid cells for eight sites across Kruger National Park, Sabi Sand Wildtuin and Bushbuckridge, South Africa. The sites form an east to west gradient of increasing rainfall from 550 mm (Site 1 & 2) to >1200 mm (Site 8). Sites 1 & 2 are in a national protected area (KNP), Sites 3 & 4 are in a private game reserve (SSW) and sites 5-8 are communal rangelands. Site 8 is a low intensity of use site, and Site 6 is the highest intensity of use site (Figure appears as Appendix S4.2a, Chapter 4. Replicated here for ease of reading).

Action can be taken to rehabilitate impacted vegetation as is seen in KNP and MalaMala which were successfully converted from cattle ranches to conservation (Chapter 2). Although the two reserves are effective protected areas, differences in their management objectives have already caused changes in vegetation structure and possibly ecosystem function (Table 6.1). Continuing on the current trajectory, KNP stands to lose functionally important tall trees, with a similar fate in MalaMala if elephant densities continue to increase in this private reserve. In the southern section of SSW, 3 x more tall trees (> 6 m) were lost over 2 years compared to the neighbouring communal rangelands as a result of increasing elephant density (Chapter 5). The effectiveness of these protected areas is not only important to local and global biodiversity conservation, but also to the local economy. Tourism, including eco-tourism and village tourism, contributes 7.9% towards South Africa's Gross Domestic Products (GDP) (http://www.info.gov.za/aboutsa/tourism.htm, accessed 14 Jan 2013; Table 6.1).

Not all protected areas show the same level of heterogeneity (Chapter 2) as a result of land use history and management (Table 6.1; Fig. 6.1), and not all communal rangelands are degraded, it depends on intensity of use as well as abiotic factors such as rainfall, geology and topography (Chapter 3; Fig. 6.1). Both types of SES (communal rangelands and protected areas) show feedbacks between the resource system and users/governance (Table 6.1). People will alter their harvesting patterns based on their knowledge and awareness of the system (Table 6.1), thereby avoiding complete woodland degradation (Matsika 2012). Preconceptions about the state of savannas based on land use alone (i.e. protected areas have intact vegetation, communal rangelands have degraded vegetation; Fig. 6.1) therefore do not hold true when fine scale empirical evidence is obtained. Knowledge and understanding of local systems (Table 6.1) is vital for effective management.

6.4 Socio-ecological systems and LiDAR in savannas: a global perspective

The ecosystems and human wellbeing synthesis report of the Millenium Ecosystem Assessment identified land cover change as one of the five key drivers of biodiversity loss (Millenium Ecosystem Assessment 2005), and human activities are resulting in the degradation and loss of woodlands globally. Savannas constitute at least half of Africa and over a third of the land cover of South Africa (Scholes & Archer 1997), supporting almost a quarter of the population (Shackleton 2000). People living in rural areas continue to depend heavily on savanna-derived natural resources for survival (Dovie et al. 2002, Kirkland et al. 2007). With increasing population levels, as well as urbanisation within these areas, the result is an amplified use of natural resources (Banks et al. 1996, Fisher et al. 2012 (Chapter 3), Kirkland et al. 2007) and settlement areas are encroaching increasingly into intact vegetation (Coetzer et al. 2010). Although the data collected for this thesis covers only a small percent of the earth's surface, the range of land use types and intensities, and a better understanding of the SES (Table 6.1), allows for lessons to be extrapolated across greater areas.

For example, the use of fuelwood as a primary energy source is practiced across Africa. Even though fuelwood extraction may not cause complete woodland loss, intense removal from understory forest layers limits the regeneration potential of these forests and alters the structure (Fisher et al. 2012 (Chapter 3); Furukawa et al. 2011, Christensen & Heilmann-Clausen 2009). Traditional remote sensing methods to measure vegetation dynamics would not detect changes below the canopy (Jansen and Di Gregorio 2002). The advantage of using LiDAR, and in particular a 3D vegetation structural classification (Chapter 4), is that subtle changes in sub-canopy vegetation density and spatial arrangement can be identified before a state shift occurs. As a monitoring system, the 3D classification can be used to mitigate the changes in land cover by identifying areas of concern, such as areas with bush encroachment (Chapter 5) or declining disturbance gradients (Chapter 3), and take action towards better management. LiDAR is more cost effective than collecting field-based measurements over larger extents. When a spaceborne LiDAR system is successfully launched, it will be a valuable tool for global vegetation monitoring (Hall et al. 2011).

Mapping and monitoring of biodiversity has become a major global focus (Asner 2013) and initiatives such as iDiv (German Centre for Integrative Biodiversity Research; http://www.idiv-biodiversity.de/) are being launched with the aim of assessing biodiversity globally.

6.5 Conclusion and recommendations

Socio-ecological systems in savannas are complex adaptive systems which require a multi-disciplinary approach in order to understand and manage them. I integrate the use of remote sensing with the knowledge of savanna ecology and an understanding of community-based natural resource management to provide both the social and ecological context necessary to understand the effects of management on natural resources in this heterogeneous semi-arid savanna system. Sustainable management of these systems is vital with communal rangelands being threatened by unsustainable development, creating a fuelwood crisis as an immediate local consequence (Matsika et al. 2013; Wessels et al. 2013). Long term, the loss of biodiversity as a result of land-cover change (Sala et al. 2000) has dire global implications. This thesis is the first instance of characterising woody vegetation structure from the individual tree to the landscape scale across multiple land use types and intensities which was made possible with the use of LiDAR. Such detail is necessary to understand the socio-ecological drivers of vegetation structure, providing context for well-informed management decisions.

The CAO conducted an additional flight campaign in 2012 and has a future campaign planned for 2014. Data were collected over more areas of Bushbuckridge and less of the protected areas in 2010 and 2012 than were presented in this thesis. These data will allow for further context specific studies to be conducted and will address various gaps in the field. I put forward the following recommendations for future studies:

- With repeat LiDAR campaigns woody vegetation structural change detection studies can be conducted over greater areas of Bushbuckridge. One such area is Welverdiend village which was the focus of the thesis by Matsika (2012). In depth field research and household surveys were conducted in 2009 (Matsika 2012) and can be used in conjunction with the LiDAR data to extrapolate the village scale effects of fuelwood harvesting on structural diversity coupled with socio-economic data providing a more holistic view of the system.
- Using participatory Geographic Information Systems (GIS), the structure of areas suitable for fuelwood harvesting can be defined (fuelwood hotspots), allowing the Tribal Authorities to make pre-emptive decisions regarding resource allocation.

- The classification developed in Chapter 4 can be time consuming to conduct, especially for a new user unfamiliar with the software. I therefore suggest the classification becomes an add-in to a software package. The reason it was not done as part of the thesis is because eCognition is proprietary software and licences are expensive. Therefore in order to make the classification a standard add-in, free software is needed to segment the images. This can be done in programs such as R.
- CAO-Alpha included in-flight fusion of visible to near infrared (VNIR) hyperspectral imagery which allowed for species identification of common, abundant savanna species, for example *Acacia nigrescens* and *Terminalia sericia and* (Cho et al. 2012). CAO-2 AToMS includes a further hyperspectral sensor covering the visible to shortwave infrared (VSWIR) which will allow for improved species identification as well as making it possible to measure canopy chemistry, hydraulic fluxes, carbon storage and fluxes as well as plant physiology (Asner et al. 2012). Mapping these attributes across Bushbuckridge will provide greater insight on the use and sustainability of the system.
- Although changes in biomass have been recorded around settlements (Matsika et al. 2013), they have not been spatially explicit and over large extents. With the repeat CAO LiDAR campaigns it will be possible to measure the spatio-temporal change in woody biomass around multiple settlements. Investigation of biomass change can also be used as an indicator of the economic status of the villages as the fuelwood harvesting to electricity use ratio is related to dependence on natural resources rather than paying for electricity if finances are tight.
- Heterogeneity is not a defined measurement, and different levels of heterogeneity are useful /necessary based on the use of the landscape (Table 6.1). Using the Thresholds of Potential Concern (TPC) framework (Biggs & Rogers 2003), a defined set of acceptable levels of heterogeneity for various users could be constructed, together with a defined measurement of heterogeneity, to establish what the thresholds of heterogeneity are for various SES.

6.6 References

Asner, G.P, Levick, S.R., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R., Jacobson, J., Colgan, M.S. and Martin, R.E. (2009) Large-scale impacts of herbivores on the structural diversity of African savannas. *Proceedings of the National Academy of Science* **12**: 261-278

Asner, G.P. and Levick, S.R. (2012) Landscape-scale effects of herbivores on treefall in African savannas. *Ecology Letters* **15:** 1211-1217

Asner, G.P., Knapp, D.E., Boardman, J., Green, R.O., Kennedy-Bowdoin, T., Eastwood, M., Martin, R.E., Anderson, C. And Field, C.B. (2012) Carnegie Airborne Observatory-2: Increasing science data dimensionality via high-fidelity multi-sensor fusion. *Remote Sensing of Environment* **124**: 454-465

Asner, G.P. (2013) Biological diversity mapping comes of age. *Remote Sensing* **5**: 347-376

Banks, D.I., Griffin, N.J., Shackleton, C.M., Shackleton, S.E. and Mavrandonis, J.M. (1996) Wood supply and demand around two rural settlements in a semi-arid savanna, South Colgan et al 2012Africa. *Biomass Bioenergy* **11**: 319-331

Biggs, H.C. and Rogers, K.H. (2003) An adaptive system to link science, monitoring, and management in practice. In Du Toit, Johan T., Rogers, Kevin H. and Biggs, Harry C. (Eds) *The Kruger experience: ecology and management of savanna heterogeneity*. Washington DC, Island Press**:** 59-80

Cho M.A., Mathieu, R., Asner, G.P., Naidoo, L., Ramoelo, A., Debba, P., Wessel, K., Main, R., van Aardt, J.A.N., Smit, I. and Erasmus, B. (2012) Mapping tree species composition in South African savannas using an integrated airborne spectral and LiDAR system. *Remote Sensing of Environment* **125**: 214-226

Christensen, M. and Heilmann-Clausen, J., 2009. Forest biodiversity gradients and the human impact in Annapurna Conservation Area. *Nepal Biodiversity and Conservation* **18**: 2205-2221

Coetzer, K.L., Erasmus, B.F.N., Witkowski, E.T.F. and Bachoo, A.K. (2010) Land cover change in the Kruger to Canyons Biosphere Reserve (1993-2006): A first step towards creating a conservation plan for the subregion. *South African Journal of Science* **106**: 26-35

Coetzer, K.L., Witkowski, E.T.F., Erasmus and B.F.N., Reyers, B. (2013). The race for space: Tracking land-cover transformation in a socio-ecological landscape. *Environmental Management* DOI: 10.1007/s00267-013-0094-9

Dovie, D.B.K., Shackleton, C.M. and Witkowski, E.T.F. (2002) Direct-use values of woodland resources consumed and traded in a South African village. *International Journal of Sustainable Development and World Ecology* **9**: 269-283

Drake, J.B. and Weishampel, J.F. (2000) Multifractal analysis of canopy height measures in a longleaf pine savanna. *Forest Ecology and Management* **128**: 121-127

Drake, J.B. and Weishampel, J.F. (2001) Simulating vertical and horizontal multifractal patterns of a longleaf pine savanna. *Ecological Modelling* **145**: 129-142

Fischer, J. and Lindenmayer, D.B. (2007) Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* **16**: 265-280

Fisher, JT, Erasmus, BFN, Witkowski, ETF, van Aardt, J, Asner, GP, Kennedy-Bowdoin, T, Knapp, DE, Mathieu, R and Wessels, K (2009). Three-dimensional woody vegetation structure across different land-use types and land-use intensities in a semi-arid savanna. *Proceedings of the International Geoscience and Remote Sensing Symposium*, IEEE Catalog number CFP09IGA-CDR.

Fisher, J.T., Witkowski, E.T.F., Erasmus, B.F.N., van Aardt, J., Asner, G.P., Wessels, K.J. and Mathieu, R. (2012) Human-modified landscapes: patterns of fine-scale woody vegetation structure in communal savannah rangelands. *Environmental Conservation* **39**: 72-82

Fisher, J.T., Erasmus, B.F.N., Witkowski, E.T.F, van Aardt, J., Wessels, K.J. and Asner, G.P. (2013) Savanna woody vegetation classification – now in 3D. *Applied Vegetation Science* DOI: 10.1111/avsc.12048

Furukawa, T., Fujiwara, K., Kiboi, S.K. and Chalo Mutiso, P.B. (2011) Threshold change in forest understory vegetation as a result of selective fuelwood extraction in Nairobi, Kenya. *Forest Ecology and Management* **262**: 962-969

Giannecchini, M., Twine, W. and Vogel, C. (2007) Land-cover change and human-environment interactions in a rural cultural landscape in South Africa. *The Geographical Journal* **173**: 26-42

Gillson, L. (2004) Evidence of hierarchical patch dynamics in an east African savanna? *Landscape Ecology* **19**: 883-894

Hall, F.G., Bergen, K., Blair, J.B., Dubayah, R., Houghton, R., Hurtt, G., Kellndorfer, J., Lefsky, M., Ranson, J., Saatchi, S., Shugart, H.H. and Wickland, D. (2011) Characterizing 3D vegetation structure from space: Mission requirements. *Remote Sensing of Environment* **115**: 2753-2775

Hudak, A.T., Crookston, N.L., Evans, J.S., Hall, D.E. and Falkowski, M.J. (2008) Nearest neighbor imputation of species-level, plot-scale forest structure attributes from LiDAR data. *Remote Sensing of Environment* **112**: 2232-2245

Jansen, L.J.M. and Di Gregorio, A. (2002) Parametric land cover and land-use classifications as tools for environmental change detection. *Agriculture, Ecosystems and Environment* **91**: 89-100

Kirkland, T., Hunter, L.M. and Twine, W. (2007) "The bush is no more": Insights on institutional change and natural resource availability in rural south Africa. *Society and Natural Resources* **20**: 337-350

Lefsky, M.A., Cohen, W.B., Acker, S.A., Parker, G.G., Spies, T.A. and Harding, D. (1999) Lidar Remote Sensing of the Canopy Structure and Biophysical Properties of Douglas-Fir Western Hemlock Forests. *Remote Sensing of Environment* **70**: 339-361

Lefsky, M.A., Cohen, W.B., Harding, D.J., Parker, G.G., Acker, S.A. and Gower, S.T. (2002) Lidar remote sensing of above-ground biomass in three biomes. *Global Ecology and Biogeography* **11**: 393-399

Levick, S.R., Asner, G.P., Kennedy-Bowdoin, T. and Knapp, D.E. (2009) The relative influence of fire and herbivory on savanna three-dimensional vegetation structure. *Biological Conservation* **142**: 1693-1700

Levick, S.R. and Roger, K.H. (2011) Context-dependant vegetation dynamics in an African savanna. *Landscape Ecology* **26**:515-528

Levick, S.R., Asner, G.P. and Smit, I.P.J. (2012) Spatial patterns in the effects of fire on savanna vegetation three-dimensional structure. *Ecological Applications* **22**: 2110-2121

Levick, S.R. and Asner, G.P. (2013) The rate and spatial pattern of treefall in a savanna landscape. *Biological Conservation* **157**: 121-127

Luoga, E.J., Witkowski, E.T.F., Balkwill, K. (2004) Regeneration by coppicing (resprouting) of miombo (African savanna) trees in relation to land use. *Forest Ecology and Management* **189**: 23-35

Matsika, R. (2012) 'Cultural landscapes in motion: Tracing changes in land-use and land-cover and communal woodland loss in rural South Africa'. In The spatio-temporal dynamics of woody biomass supply and demand in response to human utilisation in an African savanna woodland. PhD thesis, University of the Witwatersrand, South Africa

Matsika, R., Erasmus, B.F.N. and Twine, W. (2013) A tale of two villages: assessing the dynamics of fuelwood supply in communal landscapes in South Africa. *Environmental Conservation.* DOI: http://dx.doi.org/10.1017/S0376892912000264

Mwavu, E.N. and Witkowski, E.T.F. (2009) Population structure and regeneration of multiple-use tree species in a semi-deciduous African tropical rainforest: implications for primate conservation. *Forest Ecology and Management* **258**: 840–849

Millenium Ecosystem Assessment (2005) Ecosystems and human wellbeing: Synthesis. Island Press, Washington (DC)

Neke, K.S., Owen-Smith, N. and Witkowski, E.T.F. (2006) Comparative resprouting response of savanna woody plant species following harvesting: the value of persistence. *Forest Ecology and Management* **232**: 114–123

Ostrom, E. (2009) A general framework for analyzing sustainability of social-ecological systems. *Science* **325**: 419-422

Pascual, C., Garcia-Abril, A., Garcia-Montero, L.G., Martin-Fernandez, S. and Cohen, W.B. (2008) Object-based semi-automatic approach for forest structure characterization using lidar data in heterogeneous *Pinus sylvestris* stands. *Forest Ecology and Management* **255**: 3677-3685

Pickett, S.T.A., Cadenasso, M.L. and Benning, T.L. 2003. Biotic and abiotic variability as key determinants of savanna heterogeneity at multiple spatiotemporal scales. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*. Washington DC, *Island Press*: 422-446

Rogers, K.H. 2003. Adopting a heterogeneity paradigm: Implications for management of protected savannas. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*. Washington DC, *Island Press*: pp. 41-58

Sala, O.E., Chappin III, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., LeRoy Poff, N., Sykes, M.T., Walker, B.H., Walker, M and Wall, D.H. (2000) Global biodiversity scenarios for the year 2100. *Science* **287**: 1170-1174

Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux, J., Higgins, S.I., Le Roux, X., Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G., Caylor, K.K., Coughenour, M.B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C., Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K.L., Prins, H.H.T., Ringrose, S., Sea, W., Tews, J., Worden, J. and Zambatis, N. (2005) Determinants of woody cover in African Savannas. *Nature* **438**: 846–849

Sankaran, M., Ratnam, J. and Hanan, N. (2008) Woody cover in African savannas: the role of resources, fire and herbivory. *Global Ecology and Biogeography* **17**: 236-245

Scholes, R.J. and Walker, B.H. (1993) *An African savanna - Synthesis of the Nylsvley study*. Cambridge, Cambridge University Press

Scholes, R.J. and Archer, S.R. (1997) Tree-grass interactions in savannas. *Annual Review of ecology and Systematics* **28**: 517-544

Shackleton, C.M., Griffin, N.J., Banks, D.I., Mavrandonis, J.M. and Shackleton, S.E. (1994) Community structure and species composition along a disturbance gradient in a communally managed South African savanna. *Vegetatio* **115**: 157-167

Shackleton, C.M. (2000) Comparison of plant diversity in protected and communal lands in the Bushbuckridge lowveld savanna, South Africa. *Biological Conservation* **94**: 273-285

Skowronski, N., Clark, K., Nelson, R., Hom, J. and Patterson, M. (2007) Remotely sensed measurements of forest structure and fuel loads in the Pinelands of New Jersey. *Remote Sensing of Environment* **108**: 123-129

Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E. and Jacobson, J. (2010) Effects of fire on woody vegetation structure in African savanna. *Ecological Applications* **20:** 1865–1875

Stats SA (2012) http://interactive.statssa.gov.za/superweb/loadDatabase.do;jsessionid=3038C871551C2B4317A15FC54B4BFFBC?db=Descriptive11_wd, accessed January 2013

Thornton, R. (2002) Environment and Land in Bushbuckridge, South Africa. Sciences-New York**:** 1-31

Twine, W., Moshe, D., Netshiluvhi, T. and Siphugu, V. (2003) Consumption and direct-use values of savanna bio-resources used by rural households in Mametja, a semi-arid area of Limpopo province, South Africa. *South African Journal of Science* **99**: 467–473

Twine,W.C. (2005) Socio-economic transitions influence vegetation change in the communal rangelands of the South African lowveld. *African Journal of Range and Forest Science* **22**: 93–99

Venter, F.J., Scholes, R.J. and Eckhardt, H.C. (2003) The abiotic template and its associated vegetation pattern. In: (Eds. Du Toit, JT, Rogers, KH and Biggs, HC.) *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*. Washington DC, *Island Press*: pp. 83-129

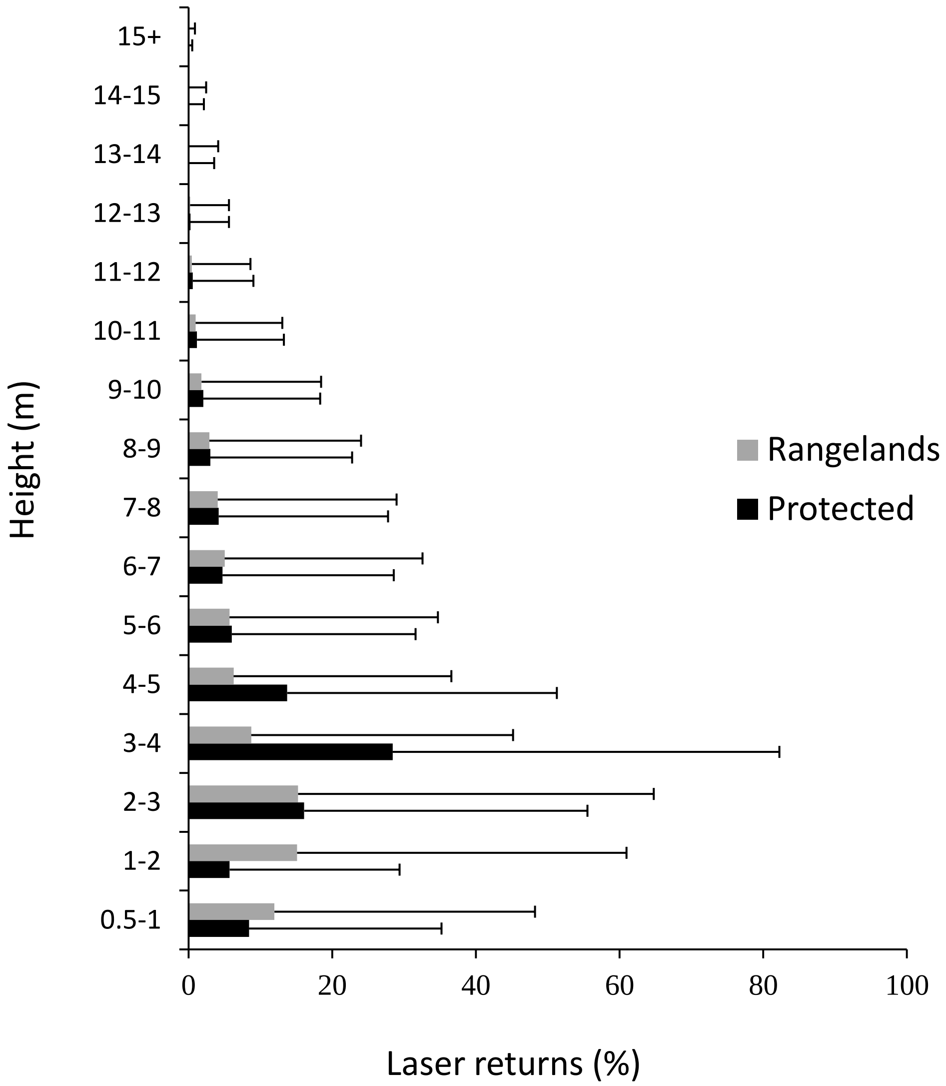
Wessels, K.J., Mathieu, R., Erasmus, B.F.N., Asner, G.P., Smit, I.P.J., van Aardt, J., Main, R., Fisher, J., Marais, W., Kennedy-Bowdoin, T., Knapp, D.E., Emerson, R. and Jacobson, J. (2011) Impact of contrasting land use on woody vegetation structure in the Lowveld savannas of South Africa. *Forest Ecology and Management* **261**: 19–29

Wessels, K.J., Colgan, M.S., Erasmus, B.F.N., Asner, G.P., Twine, W.C., Mathieu, R., van Aardt, J.A.N., Fisher, J.T and Smit, I.P.J. (2013) Unsustainable fuelwood extraction from South African Savannas. *Environmental Research Letters* **8**: 10pp. doi:10.1088/1748-9326/8/1/014007

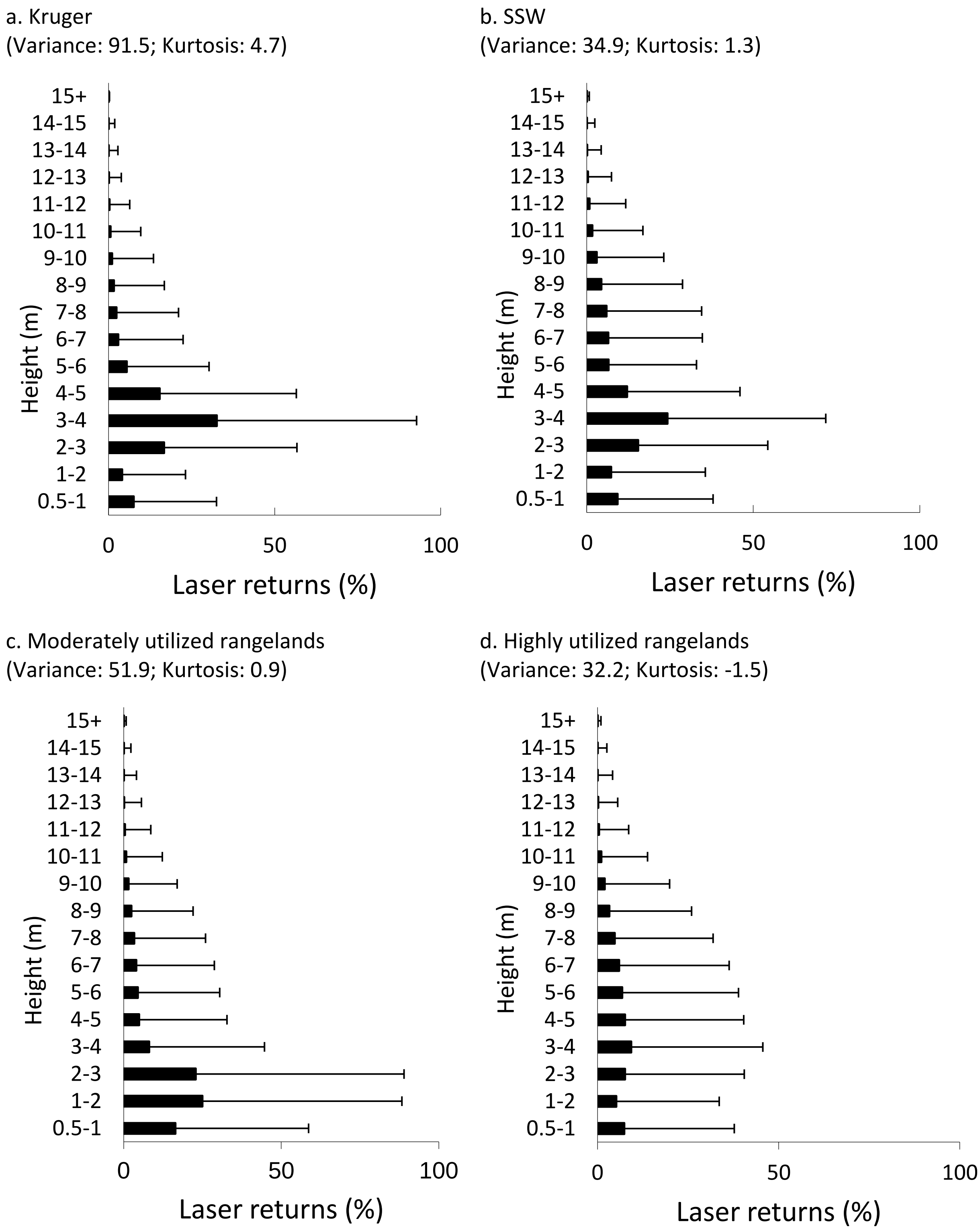
Whitecross, M.A., Archibald, S. And Witkowski, E.T.F. (2012) Do freeze events create a demographic bottleneck for *Colophospermum mopane*? *South African Journal of Botany* **83**: 9-18

6.7 Appendix

For ease of reference, figures referred to from Fisher et al. (2009) are presented in this appendix.



Appendix 6.1: Vertical distribution of vegetation density in the protected areas (Kruger National Park and Sabi Sands Wildtuin study areas) and the communal rangelands (near the towns of Justicia and Kildare, Bushbuckridge) in South Africa. Percentage laser returns refers to the mean of all laser returns in a 200 ha sample area in a specific height class. Error bars indicate standard deviation. Vegetation height classes should be interpreted as follows: 1-2 m includes vegetation from 1-1.9 m, 2-3 includes vegetation from 2-2.9 etc.



Appendix 6.2: Vertical distribution of vegetation density in 200 ha (a) Kruger National Park (Kruger), (b) Sabi Sands Wildtuin (SSW), (c) moderately utilized rangelands in Bushbuckridge and (d) highly utilized rangelands in Bushbuckridge. Percentage laser returns refers to the mean of all laser returns in a 200 ha sample area in a specific height class. Error bars indicate standard deviation. Vegetation height classes should be interpreted as follows: 1-2 m includes vegetation from 1-1.9 m, 2-3 includes vegetation from 2-2.9 etc.