Long-term Vegetation Change Before and After Converting from Livestock farming to Game Ranching in Asante Sana Game Reserve, South Africa

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Abstract

Although wildlife production is widely considered beneficial for semi-arid environments, few studies have reported on the long-term environmental effects of converting from livestock production to game ranching. Asante Sana Game Reserve in the Eastern Cape has centuries old land use history, during which it was cultivated and heavily overgrazed by domestic livestock with associated loss in vegetation productivity and subsequent soil erosion. After 1996 game ranching was adopted in the reserve, with observed positive results on vegetation productivity. This thesis investigates the long-term (1987-2017) spatial and temporal change in vegetation in the reserve. It documents the change in vegetation types and cover using Landsat Top of Atmosphere (TOA) reflectance multispectral data and Soil Adjusted Vegetation Index (SAVI). Correlative relationships between vegetation cover and different drivers (e.g. rainfall, fire and stocking density) are explored using generalized linear mixed models and the implications of the findings for reserve management are discussed. The results show that the relative area of Thicket has increased over time at the expense of Grassland and Shrubland while Bare-ground has expanded into Shrubland and Thicket. A ground-truthing exercise revealed a significant ($p<0.01$, $R^2 = 0.6$) positive relationship between the vegetation cover estimated on the ground and satellite derived SAVI values, suggesting that SAVI can be used as a proxy for vegetation cover. Overall vegetation productivity increased over time, with the greatest increases in Thicket and Cultivated land and the lowest in Shrubland and Bare-ground. Grassland and Riverine thicket experienced surprisingly small increases in productivity, which can be explained by high prevalence of grazing ungulates and elephants in areas of Grassland and Riverine thicket respectively. Rainfall, burning and stocking numbers all had an effect on productivity in the reserve. Rainfall had clearly the strongest influence, supporting the non-equilibrium theory for semi-arid rangelands. The management can undertake restoration actions such as tree thinning, erosion control and prevention as well as fencing off affected Grassland and Riverine thicket. A long-term ecological monitoring programme should be established for the reserve for improved understanding of the vegetation dynamics so that effective evidence-based management decisions can be undertaken.

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1. Introduction

1.1 Rationale and aims

Arid and semi-arid rangelands are characterised by erratic rainfall, making them especially vulnerable to a multitude of environmental and anthropogenic factors (Fynn & O'Connor, 2000). The number of game ranches and game farms in South Africa has increased rapidly within the last two decades, many of which have been converted from small stock farms and occur in South Africa's semi-arid regions. It is commonly thought that wildlife production is more beneficial for the land relative to small stock production (Du Toit & Cumming, 1999), but few studies have reported on the long-term environmental effects of converting from livestock production to game ranching (Lindsey et al., 2009). One locality where such land conversion has occurred is the Asante Sana Game Reserve (Boshoff & Kerley, 1997), which is situated in the ecotone between the Nama Karoo, Grassland and Albany Thicket biomes (Mucina & Rutherford, 2006). The area was settled by people from the mid-18th century and stocked with sheep, goats and ostriches until 1995. During this time, it was cultivated and heavily overgrazed with an associated loss in vegetation productivity and subsequent soil erosion (Shearing, 1997). After 1996 all domestic livestock were removed and game ranching was adopted in the reserve (Boshoff & Kerley, 1997), with observed positive responses in vegetation productivity despite the relatively high stocking rates which were in place at the time. However, regardless of observed improved general veld condition, bush encroachment has been a major problem in the reserve.

This thesis investigates the long-term (1987-2017) spatial and temporal change in vegetation in the reserve. It aims to document the changes in vegetation types and their cover using Landsat Top of Atmosphere (TOA) multispectral data and the Soil Adjusted Vegetation Index (SAVI), and explore correlative relationships between vegetation cover and a suite of potential drivers (i.e. rainfall, overall animal numbers and burning). Findings from this research will help guide the reserve in their game management practices, and provide the scientific community with information on the patterns and dynamics of changes in vegetation type and productivity following a switch in land use practices from commercial livestock production to wildlife ranching. It will also show how the overall vegetation cover in the eastern Karoo is influenced by different drivers over relatively long time periods. The scope of this research is to study changes in vegetation type and cover within thirty years of time, and to explore the general impacts of rainfall, animal numbers and burning on the overall vegetation cover.

This dissertation takes the form of an extended research report. After the introduction, it starts with a literature review on the land degradation, carrying capacity and remote sensing literature, which is followed by a description of the study area and methods used. The results are presented and then discussed in context with an emphasis on their relevance for management of the reserve.

1.2 Research questions and hypotheses

The research questions address long-term changes in vegetation type with special focus on changes in Bare-ground, Shrubland, Grassland and Thicket, as well as changes in vegetation cover in the different vegetation types. They furthermore deal with the relationship of vegetation cover to three different drivers including rainfall, burning and stocking numbers. Finally, management implications of these changes are addressed (Table 1).

Table 1. Research questions and hypotheses for investigating long-term vegetation changes in Asante Sana Game Reserve.

Question	Hypothesis	Support	Evidence	References
How are the main vegetation types distributed in space and how have they changed over time?	Thicket communities have increased over time at the expense of Grassland and Shrubland communities	Thicket communities have been shown to increase at the expense of Grassland and Shrubland communities throughout semi-arid southern Africa in both commercial and communal lands. Preliminary assessment of vegetation cover from GeoTerraImage (2015) products revealed an increase in Thicket cover in Asante Sana Game Reserve. Furthermore, according to the management observations in the reserve Thicket communities have increased in distribution over time.	Empirical, anecdotal	O'Connor et al. (2014), Belayneh and Tessema (2017),Skowno et al. (2017), Devine (2017), GeoTerraImage (2015).
	Bare-ground has decreased over time	Although there are no studies to date which address the comparative impact of small stock and game on vegetation, fence-line experiments show a drastic negative impact of sheep grazing on vegetation in semi-arid rangelands. Reducing grazing pressure resulting from removing livestock and introducing indigenous wild animals is therefore assumed to alleviate grazing pressure and erosion. According to the observations and perceptions of the reserve management, Bare-ground has decreased since the introduction of wild animals.	Hypothetical, anecdotal	Lindsey et al (2009), O'Connor and Roux (1995), Todd and Hoffman (1999, 2009) and Seymour et al. (2010).
How does vegetation productivity vary spatially and temporally in the reserve?	Overall vegetation productivity has increased over time	Records from the reserve show an increase in rainfall over time. According to the non-equilibrium theory for semi-arid rangelands, increase in rainfall is likely to result in an increase in overall productivity. A reduction in grazing pressure as a result of the removal of livestock and introduction of indigenous wild animals is also assumed to have alleviated grazing pressure and therefore result in increased productivity.	Theoretical, empirical	Vetter (2005), Heshmati & Squires (2010), Todd and Hoffman (1999, 2009) and Seymour et al. (2010).
	Productivity in Grassland has increased proportionally at higher rate compared to other vegetation types, and productivity in Shrubland has increased proportionally at lower rate, and in parts of the Riverine thicket, it has decreased.	Reducing grazing pressure as a result of the removal of livestock and introduction of indigenous wild animals is assumed to have alleviated grazing pressure and result in an increase in the productivity Grassland. Increasing browsing pressure as a result of introducing wild indigenous browsers is likely to have resulted in increased browsing pressure in Shrubland, therefore leading to relatively low increase in productivity compared to other vegetation types. Elephants have been demonstrated to impact Riverine vegetation by e.g. trampling and toppling trees therefore reducing vegetation productivity in these areas.	Hypothetical, empirical	van Niekerk (1980), Roques et al. (2001), O'Connor et al. (2014), Fornara and du Toit (2008), (Owen-Smith et al. 2006).
What is the relationship between vegetation productivity and rainfall, fire and stocking numberss?	Rainfall and burning affect vegetation productivity positively while stocking density affects it negatively	Rainfall and burning have been recorded to affect vegetation productivity positively and stocking densities have been recorded to effect it negatively.	Empirical	Wessels et al. (2007), Case and Staver (2017), Scholtz et al. (2017).
	Rainfall is the strongest driver compared to stocking numbers and burning	Non-equilibrium theory for semi-arid rangelands predicts that abiotic factors such as rainfall and fire have a greater influence on vegetation change than biotic factors such as herbivory.	Theoretical, empirical	Sullivan and Rohde (2002), Vetter (2005), Heshmati & Squires (2010), Wehrden et al. (2012) .
What are the management implications of the patterns of change over space and time for Asante Sana Game Reserve?	Long-term monitoring, Thicket thinning, fencing off areas of concern and active restoration interventions in Shrubland can be undertaken	Long-term ecological monitoring programs have been shown to be beneficial for long-term sustainability of game reserves. Thicket thinning has been shown to help restore areas under bush encroachment. Fencing off areas of concern have been shown to reduce grazing/browsing pressure and therefore contribute to restoration of the degraded landscape. Active restoration, however, is necessary to improve rangelands	Theoretical, Empirical	Lindemayer and Likens (2010), Lindemayer et al. (2012a, 2012b), Seymour et al. (2010),

2. Literature review

2.1 Importance of arid and semi-arid rangelands

Rangelands have great global economic and ecological importance, and can be defined as areas which are covered by native grasses, shrubs and woody vegetation and which are used extensively by grazing animals (Lund, 2007). Rangelands provide habitat for biodiversity, sequester carbon, and support ecosystem services such as fodder and livestock production upon which farmer's livelihoods depend (Vetter, 2009). They are moreover social-ecological systems that are influenced by complex interactions between people, domestic and wild animals, vegetation and the physical environment (El-Shorbagy, 1998). Rangelands are therefore susceptible not only to changes in the environment, but also to changes in government policies, markets and management practices.

The United Nations Food and Agricultural Organisation (1987) defines arid and semi-arid areas as having an annual rainfall between 0-300 mm and 300-600 mm respectively. Because of the low and erratic rainfall which characterises such areas, the cultivation of crops is difficult and people rely more on pastoralism for income and food security (Dean and Macdonald, 1994). Pressured by recent global and local population growth and the subsequent increase in food demand, stocking rates have significantly increased in developing countries (Steinfeld et al., 2006). Through mismanagement and overstocking, many of these rangelands have become degraded, resulting in a loss of habitat and soil quality as well as other ecosystem services (Wessels et al., 2007).

2.2 Converting from livestock farming to game ranching in South Africa

South African arid and semi-arid rangelands are characterised by erratic rainfall and fluctuating grass, shrub and tree cover as well as a variety of grazing and browsing herbivores. Rangelands in South Africa have been grazed for millennia by indigenous pastoralists (Vogel et al., 1997). Following the mid-16th century settlement and subsequent expansion of European farmers in southern Africa the number of domestic livestock increased across the region. A lack of ecological knowledge and increasing pressure to produce more animals often led to overstocking of the rangelands (Dean et al., 2003), especially during the 19th and early 20th centuries (Hoffman & Ashwell, 2001). Increased hunting of wild herbivores by European settlers also reduced wildlife numbers, transforming the dynamics of plant-animal interactions over large areas (O'Connor et al., 2014).

Economic pressure such as the rapid increase in the demand for merino wool in the beginning of the 1950's resulted in an increase in the number of sheep in South Africa (Conradie et al., 2013) exacerbating the impact of grazing especially on the grassy component of the vegetation and on soil quality. More recently, many commercial farmers in South Africa have either stopped farming or have transformed their livestock farming practices into game ranches. This has been facilitated by a shift in South Africa's policy around wildlife in the early 1990s which enabled private individuals to own and farm with wildlife (Snijders, 2012). With the rise of the South African eco-tourism industry, game ranching also became more profitable making the shift from livestock to game ranching more attractive. This has led to a rapid increase in the number of private game reserves, which stock indigenous ungulates, mega herbivores and carnivores in South Africa (Smith & Wilson, 2002; Meissner et al., 2013). The number of game ranches in the country has grown from only four in the 1960's to more than 10 000 in 2016 (Peel, 2017). Because of economic incentives, however, many game reserves confine large numbers of indigenous herbivores into relatively small areas (Dean et al., 2003), repeating the high impacts of herbivory on the vegetation that took place during livestock farming.

Current literature, which explores the effects of converting from livestock to game ranching in South Africa focuses on the socio-economic impacts (Cloete et al., 2007; Cousins et al., 2008; Lindsey et al., 2013). Very few studies have investigated the long-term impacts of game ranching on the vegetation of southern Africa, particularly in terms of its comparison with the impacts of domestic livestock (Werger, 1977; Lindsey et al., 2009). Milton et al. (1998), Van der Waal (2000) and Carruthers (2008) present ways to assess rangeland health on game ranches and outline preferred management practices for semi-arid environments. While these studies are helpful for planning sustainable management practices, they leave a gap in our understanding of the relative effects of small stock and game ranching on rangeland vegetation. This has been addressed, in part by O'Connor and Roux (1995), Todd and Hoffman (1999, 2009) and Seymour et al. (2010) who studied the effects of sheep grazing on semi-arid rangeland vegetation through fence-line experiments. However, no studies to date have investigated the impact of the change from small stock farming to game ranching on vegetation composition and productivity at a site over decadal time scales.

2.3 Semi-arid rangelands

Arid and semi-arid rangelands can be considered as non-equilibrium ecosystems, which are driven by stochastic environmental factors such as rainfall (Vetter, 2005). These factors often push the ecosystem between two or more alternative states, instead of following a directional path to one equilibrium state (Heshmati & Squires, 2010). In semi-arid Savanna, for example, the two extreme states are Grassland and Woodland. In the semi-arid eastern Nama Karoo biome, they are Shrubland and Grassland and in some places also Thicket (Masubelele et al., 2014). Either dwarf shrubs and trees or grasses and woody plants compete for dominance. Both co-exists in the area, but depending on environmental factors such as rainfall and fire, as well as past and present grazing and browsing impacts, either one of them will dominate over the other. The dominance of one species can change rapidly however, when factors affecting the system change (Sankaran et al., 2014; Heshmati & Squires, 2010; Belayneh & Tessema, 2017).

2.4 Land degradation in semi-arid rangelands

Land degradation is considered one of the most serious global environmental problems of today (Wessels et.al., 2007), and is a major problem especially for rangelands in arid and semi-arid areas

(Seymour et al., 2010). Early accounts of land degradation in South Africa were reported by Acocks (1988), who hypothesised a north eastward expansion of karroid shrubs into the more productive Grassland biome. Recently, however, this hypothesis has been contested due to lack of evidence (Dean et al., 1995) and the increase of grasses in the Nama-Karoo biome has been described (Masubelele et al., 2014). In South Africa, communal lands have been historically overstocked by up to four times the recommended rate, which has resulted in heavy land degradation and soil erosion with subsequent decrease in vegetation productivity and its capacity to feed both domestic and wild herbivores (Wessels et al. 2007). The problem with land degradation in semi-arid areas is that it is characterised by non-linear dynamics. Reversing degradation, therefore, requires substantial intervention which can often turn out to be costly (Turnbull et al., 2008).

There are multiple definitions for “land degradation” depending on the context in which it is used. This makes the objective assessment of landscape condition difficult (Wessels et al., 2007). To what extent land is degraded depends on the subjective view of what is considered as a “desirable” or “undesirable” state for a particular area. To a sheep farmer, a rangeland may be desirable when perennial grasses are abundant, whereas a game rancher that stocks Kudu and Giraffe might prefer a greater abundance of taller palatable shrubs and trees. A conservationist on the other hand might be concerned with the loss of biodiversity and resilience, and would therefore seek to maximize species richness, redundancy and connectivity (Oliver et al., 2015).

Belayneh and Tessema (2017) reviewed publications on land degradation and categorized them into units that looked at degradation through a) biotic and abiotic qualities including the loss of topsoil, changed composition in flora and fauna and continuous reduction of primary productivity, b) a decrease in above ground biomass production, carbon storage, water infiltration and retention capacity as well as soil quality and c) socio-economic issues such as a reduction in the land’s capacity to provide ecosystem services including primary productivity and livestock production. From the perspective of studying land degradation in a semi-arid rangeland with a strong reference to farm productivity, I will be using the United Nations Convention to Combat Desertification (UNCCD, 2009) definition adapted from Balayneh and Tessema (2017), where land degradation is defined as: "*The decline or complete damage of the biological productivity and/or socioeconomic benefit of land in general*" (p. 2). This definition is well suited for exploring trends in bush encroachment and vegetation productivity and cover in response to environmental factors and land use practices.

There is an ongoing debate on how much of land degradation in arid and semi-arid regions is caused by global drivers such as climatic factors and how much by local drivers such as fire and grazing practices (Turnbull et al., 2008; Vetter, 2009; Belayneth & Tessema, 2017). A growing body of literature discusses how climate change and subsequent changes in rainfall and temperature affect arid and semi-arid areas (Reynolds, 2004; O'Connor et al., 2014; Sankaran et al., 2014). Evidence suggests that climate change imposes increasing pressure on these ecosystems, often by limiting water availability and changing fire dynamics (Kraaij & Milton, 2006). A few studies also address the impact of an increase in atmospheric CO_2, and the role of nitrogen on the unwanted increase in woody cover (Idso, 1992; Eldridge et al., 2011; Bond and Midgley, 2012). Selective grazing and lack of browsing has been observed to reduce perennial grasses and promote erosion and the increase of unpalatable woody plants (O'Connor et al., 2014; Belayneh & Tessema, 2017; Devine, 2017). The increase in woody cover in areas dominated by grassy or shrubby vegetation is a form of land degradation and is known as "bush encroachment" (Belayneh & Tessema, 2017). "The increase in woody cover in areas dominated by grassy or shrubby vegetation is a form of land degradation and is known as "bush encroachment" (Belayneh & Tessema, 2017). In literature both terms "bush encroachment" and "bush thickening" are used when referring to increase in woody cover in grassland and savanna. Bush encroachment, however, is increasingly used to refer to the invasion of alien woody cover at the expense of native grasses and shrubs (O'Connor et al., 2014, Divine, 2017), whereas bush thickening is used to refer to the increase of indigenous woody cover (Joubert et al., 2012; Joubert et al.,2014). In this report, I use "bush encroachment" for both increase in alien and indigenous woody cover for the sake of consistency and clarity. A distinction between increase in alien and indigenous woody cover is made in the discussion section however although the same term is used for both.

2.5 Bush encroachment in semi-arid rangelands

Bush encroachment is usually portrayed in a negative light. Ward (2005) defined bush encroachment as an increase in unwanted woody vegetation, which causes a decrease in biological diversity and in the grazing potential of both wild and domestic herbivores in semi-arid rangelands. Due to the shift from grass cover to woody plants and the subsequent decrease in palatable grass productivity, the carrying capacity for grazing animals also decreases (Dean & Macdonald, 1994) and ultimately leads to wider socio-economic problems (Kraaij & Ward, 2006). A total of 10-20 million ha of agricultural and rangeland in South Africa are thought to be affected by bush encroachment (Ward, 2005). Skowno et al. (2017) found that over 57 000 km^2 of grassland has been replaced by woodland within 23 years in South Africa compared to only 30 000 km^2 of woodland being replaced by grasslands. Oba et al. (2000) furthermore estimated that a 10% increase in wood cover will result in a 7% decrease in grazing potential, and therefore a net loss of income for the farmer. Bush encroachment has additional negative impacts on species diversity and soil quality (Puttick et al., 2014a).

A newly published overview, however, points out that there are some positive impacts associated with bush encroachment as well (Belayneh & Tessema, 2017). Woody plants can sometimes be a good source of fodder for livestock, firewood, medicine and mulch (Smit, 2004; Eldridge et al., 2011). N_2 fixing woody plants may also increase below ground stocks of carbon and nitrogen (Maestre et al., 2009) contributing to carbon sequestration and improving soil quality.

2.6 Drivers of bush encroachment

Bush encroachment is thought to be driven by multitude of global and local drivers (Table 1). The intensity and timing of rainfall partly determine how much water gets absorbed into the soil (Reynolds et al., 2004). Plants get most of their water through their roots, and therefore do not necessarily show any change in productivity in response to a small rainfall pulse. An impermeable top soil layer, together with evaporation, prevent small rainfall events from getting absorbed into the ground and to the roots of most species (Reynolds et al., 2004). If rain falls at night, it is more likely to reach the deeper soil layer due to lower evaporation from the surface of the soil compared to daytime. The intensity and timing of rainfall events are therefore important determinants of vegetation response in arid and semi-arid rangelands (Reynolds et al., 2004).

Different vegetation types also respond to rainfall differently. Grasses generally have shallower roots compared to woody plants. Grasses can therefore utilise water from the soil's top layer, whereas mature woody plants extract water from the soil's deeper layers (Devine et al., 2017). According to Walter's two-layered model, grasses outcompete woody plants in dry lands using moisture from the soil's top layer. They therefore inhibit woody plants from accessing moisture in the deeper layers (Walter, 1939). A thick grassy layer is also likely to compete with tree seedlings for water which also have their roots embedded in the top layer. Sandy soils are thought to provide a more favourable environment for the establishment and growth of woody plants as rainfall easily flushes down from the top layer deeper into the ground. Heavily textured soils, such as clays, on the other hand support grasses because they are richer in nutrients and rainfall is often confined to the top layer of soil. It follows that areas with sandy soils are more susceptible to bush encroachment compared to clay soils (O'Connor, 2014). Ward (2005) however, points out that bush encroachment has been widespread in areas with a single soil layer, therefore contradicting Walter's two layered model.

Fire has been suggested to play an important role in supporting grassy vegetation cover in semi-arid areas (Case & Staver, 2017; Scholtz et al., 2017; Devine et al., 2017). Grass cover in savannas acts as fuel for fire. The taller and drier the grass cover, the more intense the fire will be. Although fire rarely kills mature trees, it adversely impacts seedling growth, survival and regeneration (Bond & van Wilgen, 1996). Depending on intensity, fire may also facilitate germination of some woody species by stopping seed dormancy or destroying it by breaking the seed cover (Schultz et al., 1955).

Given that frequent fire, especially after germination, is important for grasses to prevail in a savanna, fire suppression impacts this growth form negatively. Fire in southern Africa has been suppressed both directly and indirectly. Fire was commonly applied by indigenous communities in southern Africa to increase productivity of the veld for grazing livestock (O'Connor et al., 2014). From the 17th to early 20th centuries, however, fire was suppressed by successive colonial governments, because of its supposed adverse effects on the veld. Natural fires were also prevented from spreading as well. Man-made fires were even criminalised by the colonial authorities in Botswana since 1880 (Jacobs, 2000).

Selective grazing by livestock also indirectly suppresses fire. Grass biomass acts as a fuel for fire. Grazing removes grass biomass, and therefore removes the fuel (Devine, 2017). This was well demonstrated during the Rinderpest epizootic in 1890 that swept across eastern and southern Africa killing off 95% of livestock (Spinage, 2012). As a consequence of decreasing grazing pressure that resulted from decreased livestock densities, fire frequency increased, which in turn resulted in a decrease in woody cover (Norton-Griffiths, 1979; Dublin et al., 1990).

Another way grazing promotes bush encroachment is by reducing grass competition (O'Connor et al., 2014; Balayneth & Tessema, 2017; Devine, 2017). Grazing eliminates perennial grasses that compete with woody seedlings and adult shrubs (Ward & Elser, 2011) for below and above ground resources. Grazing may subsequently make below and above ground resources available for tree seedlings to establish themselves (February et al., 2013). In the Kalahari of Botswana for example, grazing had a positive effect on the density of woody species over a period of four years (Kambatuku et al, 2013). Similarly, in Namibia encroachment was positively correlated with grazing pressure during four years of above average rainfall and two years of drought (Joubert et al., 1966). At another site in

Namibia, sowing perennial grass seeds on the land had a negative effect on woody plants (Donaldson, 1969). All these cases support the idea that grazing promotes the increase of woody plants at the expense of grass cover in arid and semi-arid savannas.

In contrast to grazing, browsing can prevent woody seedlings from establishing as well as hinder the growth of shrubs. This, in turn, can suppress the time it takes for shrubs and trees to reach maturity and can worsen the impact of fire on them (Roques et al., 2001). In contrast, browsing of seed pods and the dissemination of the seeds in dung can enhance shrub recruitment (Roques et al., 2001). Depending on whether the negative effect of browsing outweighs the positive impact on seed dispersal, browsing may reduce the competitive advantage that shrubs and trees have over grasses and therefore slow down the rate of bush encroachment. It follows that the loss of browsers and subsequent browsing pressure results in increased growth and persistence of shrubs and trees, which in combination with grazing will cause woody plant cover to increase relative to grass cover.

Decreased wild browsing pressure has been partly compensated by introducing domestic browsing livestock such as goats. Goats are shown to increase adult tree mortality, but they have had little effect on tree seedlings (O’Connor et al., 2014). In Botswana wood cover declined by 50% within eight years (McKay, 1968) and in Namibia by 60% within 13 years as a result of intense goat browsing (van Niekerk, 1980). Sheep on the other hand eat low growing plants which sometimes also includes the consumption of tree seedlings (O’Connor et al., 2014). It seems that moderate browsing effects by goats and sheep can have a positive impact on grass cover in savannas. However, goat and sheep farming in the Karoo has been practiced with such an intensity (Dean & Macdonald, 1994) that any positive impact on grass competition would have been outweighed by the negative impacts of grazing.

Elephants have been shown to transform woodlands in savanna to open vegetation. (Spinage, 2012; O’Kane et al., 2014). They consume large amounts of vegetation, strip bark, trample and uproot trees with significant effects on adult trees and seedlings (O’Connor et al., 2007). Although some scholars think that elephant’s top down foraging strategy maintains the structure and ecological function of shrubs (Stuart-Hill, 1992), Landman et al. (2014a) demonstrated that elephants have the ability to drastically change the composition and structure of the canopy shrub community as well. Elephants can therefore effectively limit bush encroachment, but at high densities they can also damage the overall productivity of the landscape. Evidence suggests that the impacts of elephants will increase near water, because of the relatively longer time they spend there (Landman et al., 2014a).

Table 2. A summary of the main drivers of bush encroachment.

Driver	Effect	Support	Reference
Soil moisture	Light rainfall only moisturizes the soil's shallow layers and is therefore only available for grasses. Heavy rainfall causes water to infiltrate the soil's deeper layers benefitting woody plants.	Walter's two-layered model, pulse-reserve theory.	Walter (1939), Reynolds et al. (2004), O'Connor et al. (2014), Devine (2017).
Soil texture	Sandy soils allow easier infiltration of water to deeper soil layers benefitting woody plants, whereas clay soils hinder water availability for deeper rooted woody plants.	Walter's two-layered model, pulse-reserve theory.	Walter (1939), Reynolds et al. (2004), O'Connor et al. (2014), Devine (2017).
Carbon dioxide	Increase in atmospheric carbon dioxide benefits C_3 woody plants by increasing their growth rates, while only effecting small C_4 grasses.	FACE-experiments, paleo-evidence.	Eldridge et al. (2011), Ehleringer et al. (2005), Idso (1992), Polley et al. (1992), Belayneh and Tessema (2017), O'Connor et al. (2014), Bond and Midgley (2012), Devine et al. (2017).
Grazing	Grazing reduces grass biomass, therefore reducing grass competition, benefitting woody plants. Grazing animals help woody plants by dispersing their seeds.	Empirical.	Norton-Griffiths (1979), Dublin et al. (1990), Kambataku et al (2013), Donaldson (1969), Miller (1994), Miller (1995), O'Connor et al. (2014), Devine (2017).
Browsing	Helps to remove seedlings improving grass competition, Elephants feed on bulk, remove trees, trample and topple branches impacting the vegetation.	Empirical.	Roques et al. (2001), McKay (1968), van Niekerk (1980), O'Connor et al. (2014), Landman et al. (2014a), Stuart-Hill (1992), Landman et al. (2014b).
Fire	Disrupts seedling survival benefitting grassy plants; Helps grasses natural cycle of replacement of moribund climax species by palatable pioneer species.	Empirical.	Bond and van Wilgen (1996), Ward (2005), Case and Staver (2017), Scholtz et al. (2017), Devine (2017).
Nitrogen	Additional nitrogen benefits species incapable of fixing nitrogen and therefore often benefits grasses.	Hypothetical.	Kraaij and Ward (2006), O'Connor et al. (2014).

2.7 Stocking rates for arid and semi-arid rangelands

Overstocking rangelands with both domestic and wild herbivores can have detrimental impacts on the vegetation and ability of the land to provide ecosystem services (Ward, 2005). Therefore, it is important to determine how many animal units can be kept on the land over longer periods of time without degrading it. Fritz and Duncan (1994) defined carrying capacity for wildlife (also known as ecological carrying capacity) as the maximum number of animals that an area can sustain on a long-term basis without deterioration of habitat. Carrying capacity for livestock (also known as production or economic carrying capacity) on the other hand is the number of animals stocked that produces the maximum sustainable yield (Fritz & Duncan, 1994).

Coe et al. (1976) explained that both rainfall and evapotranspiration are positively correlated with above ground primary productivity (ANPP) which, in turn, determines how much food is available for herbivores. Coe et al. (1976), therefore, suggested that carrying capacity for wild animals in savannas can be predicted from rainfall. Carrying capacity is commonly calculated as predicted stocking rates which is expressed in metabolic biomass per unit area ($W^{0.75}$)/km^2. The weight is raised to the power of 0.75 to account for the relatively higher metabolic rate of smaller animals. Predicted stocking rates can be also expressed in Large Stock Units (LSU)/ha. Large Stock Unit is equivalent to an adult cow at 450 kg, which gains 0,5 kg per day on forage with a digestible energy percentage of 55% (Meissner, 1982; Messinger et al., 2013). Other units like Grazers Unit (GU) and Browsers Unit (BU) have also been developed to compete with LSU when comparing grazers and browsers. GU and BU allow for distinguishing between the different ecological impacts of wild grazers and browsers due to their different habitat and food pool (Dekker et al.,1996). Grazer Unit is based on the metabolic biomass of adult blue wildebeest of 180kg and Browser unit based on the metabolic biomass of adult Kudu of 140 kg (Dekker et al., 1996). Coe et al. (1976) and Cumming and Cumming (2003) outlined Unit Mass for different African herbivores, which can be used to calculate the predicted stocking rate for a specific species. The total predicted stocking rate is expressed as:

$$PSR = 0.02 \times AP^{1.69}$$

Where PSR is the predicted stocking rate expressed in metabolic biomass/km^2 and AP is the mean annual rainfall.

Cumming and Cumming (2003) stated that herbivore communities with large animals such as cows or elephants are more likely to have a greater impact on the vegetation compared to communities, in which equal biomass is better distributed across different body sizes. This is not only due to feeding practices but also trampling. Bigger animals have relatively larger trampling effects on the vegetation compared to smaller animals (Cumming & Cumming, 2003).

Furthermore, East (1984) pointed out that calculating carrying capacities based solely on rainfall is likely to be an oversimplification of the situation. He explained that in addition to rainfall, the availability of soil nutrients is an important factor contributing to ANPP. Lack of proper nutrients in the soil may limit ANPP even with adequate rainfall. Other environmental factors influencing the carrying capacity include terrain and the type of vegetation. Generally, the steeper the terrain, the lower the carrying capacity will be. Also, an area with better grass cover naturally has higher carrying capacity for grazing animals compared to an area with poor grass cover, and an area with high numbers of palatable woody plants can better support browsers than an area with few shrubs and trees.

Fritz and Duncan (1995) found that herbivore species richness in natural sites was a small but significant factor in determining carrying capacity as well. They compared the carrying capacities of pastures to natural areas and concluded that a mixture of different grazers and browsers which is characteristic of many natural areas, is more likely to sustain higher metabolic biomass of animals compared to pastures that were stocked only by grazing livestock. This is because niche separation and better niche utilisation characterised by higher herbivore diversity allows a more efficient utilization of the available plant resources.

Herbivores in African savannas are commonly classified into grazers (feeding on grass), browsers (feeding on woody plants) and mixed feeders (feeding on both). Collinson (1995) estimated that the stocking rate for browsers in the bushveld with moderate soil fertility and a balance between woody and grass plants should be lower than 20% of the total carrying capacity, and the combined stocking rate of browsers and mixed feeders less than 40%. The above-mentioned classification of herbivore types based on their feeding can be further broken down into bulk feeders and concentrate feeders. Bulk feeders consume plant material with high fibre and low nutrient content, whereas concentrate feeders consume plants that have low fibre but high nutrient content. Generally, the bigger the herbivore, the more likely it is to be a bulk feeder and the smaller it is the more likely it is to be a concentrate feeder (Collinson 1995). According to Collinson (1995), bulk feeders should make up much higher proportion of the stocking rate compared to concentrate feeders in areas that have high rainfall and/or low soil fertility, and concentrate feeders should be stocked higher compared to bulk feeders in areas low rainfall and/or high soil fertility.

2.8 Using remote sensing to monitor arid and semi-arid rangelands

Vegetation indices derived from satellite data have been widely used in vegetation and rangeland studies because of their correlation with plant productivity (Wessels et al., 2007; Mermer et al., 2015; Palmer et al. 2017). Vegetation indices can be used for monitoring net primary productivity (NPP) and long-term plant productivity (Asrar, 1984) for instance through Leaf Area Index (Palmer et al., 2017), detection of vegetation change (Pettorelli et al., 2005), land degradation assessment (Thiam, 2003) and the mapping of vegetation types (Reed et al., 1994). The popularity of remotely sensed vegetation indices, has been enabled by freely available databases and cloud computing platforms such as Google Earth Engine (Gorelick et al., 2017), from which vegetation indices and multispectral satellite data are relatively easy and cost-effective to extract (Johansen et al., 2015; Gorelick et al., 2017). High spatial and temporal resolution in these data enables increasing precision in fine-scale vegetation research.

Vegetation indices are based on the proportion of near infrared and red spectra that is reflected from photosynthetically active vegetation. Multispectral and multitemporal Normalized

Difference Vegetation Index (NDVI) product is commonly used in vegetation surveying via Landsat thematic mapper (TM) and the moderate-resolution imaging spectrometer (MODIS) satellite sensors (Sesnie et al., 2012; Palmer et al., 2017). Both sensors produce data in the red and near infrared spectra, which can be used to detect photosynthetically active plant material. The difference between TM and MODIS satellite imagery is that they offer different spatial and temporal resolutions. TM produces images with 30 m spatial and 16 days temporal resolution, whereas MODIS produces images with 250 m spatial resolution once a day. Landsat TM imagery is available from 1984 to date with varying consistency, whereas MODIS imagery is available from 2000 to date with high consistency.

In arid and semi-arid areas, the choice of vegetation indices is further complicated by the sparse vegetation cover and exposed top soil that are characteristic of these areas. The subsequent "noise" created by the reflectance from bare soil often disturbs the vegetation index signal (Huete, 1988). The most commonly used vegetation index, NDVI, has been widely used in rangeland assessments (Hobbs, 1995; Weiss, 2001; Wessels et al., 2004), but it has been observed to be sensitive to background noise especially in arid and semi-arid areas. Several other indices were developed to combat the shortcomings of NDVI, including Enhanced Vegetation Index (EVI) (Huete, 1988), Transformed Soil Adjusted Index (TSAVI) (Baret et al., 1989) and Soil Adjusted Vegetation Index (Huete, 1988). SAVI has been used increasingly because of its reduced atmospheric noise, canopy background variation (Huete, 1988) and soil brightness (Solano et al., 2010), and can be used in vegetation monitoring in semi-arid rangelands.

3. Methods

3.1 Study site

Asante Sana Game Reserve (Figure 1) is located 20 km east of Graaff-Reinet in the Eastern Cape Province, South Africa (S32.311202°, E24.972620°). It consists of seven different farms that were purchased and joined into a single 12 000 ha reserve in 1996. During the 18th and 19th centuries the area was inhabited by colonial farming communities who established the village of Petersburg on the valley bottom (Shearing, 1997). During this period the land was heavily utilized by the village inhabitants and sheep, goats and ostriches were intensely farmed in the area (Shearing, 1997). Commercial farming practices continued into late 20th century until the area was declared a game reserve in 1996 (Bosshoff & Kerley, 1997). Current perceptions are that the area was heavily overgrazed during the period when livestock farming practices took place, and large parts of the landscape have experienced gully and sheet erosion. This erosion has been exacerbated by the cultivation and irrigation practices that occurred on the valley bottom. The relatively high stocking rates of the historical period are also thought to be responsible for a reduction in grass cover and the suppression of fire both of which are known to result in an increase in woody plant cover (O'Connor et al., 2014). An assessment of the land cover datasets which are available for South Africa for the periods 1990 and 2014 (GeoTerraImage, 2015) (See Appendx B) reveal a significant increase in woody plant cover, especially in low-lying areas of the valley bottom and on the lower slopes of the mountains. However, the GeoTerraImage products are derived at subcontinental scale with potential inaccuracies when used at local scale, and should, therefore, be used with caution.

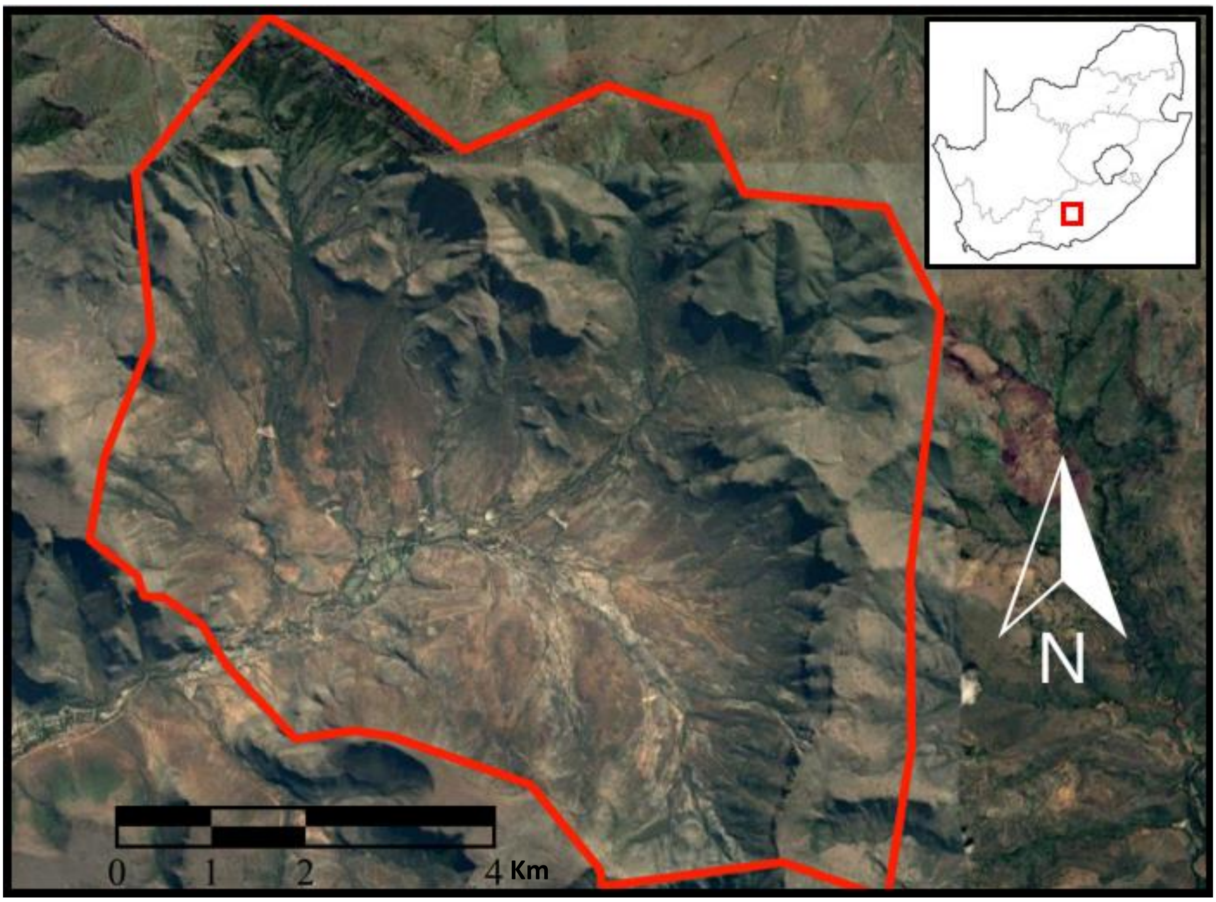

Figure 2. Satellite map of Asante Sana Game Reserve extracted from Google Earth. Insert map is modified from Bosman et al. (2013).

Elevation in the reserve ranges from 1 000 to 2 100 m. The landscape is mountainous with slopes of varying steepness (Judd & Hobson, 1999). In the northern section of the reserve, mountains encircle the extensive valley bottom below. The region receives summer rainfall with 70% of the rain falling between October and March. The mean annual rainfall varies between 380-410 mm/year (Stewart & Campbell, 2001) although this range is contested by Collinson (1995) who suggests that mean annual rainfall is between 600-800 mm. Day and night temperatures vary greatly. In the summer months, there can be extended periods of hot temperatures (above 30 °C), and in the winter, snowfalls accompanied by freezing temperatures are recorded regularly (Boshoff & Kerley, 1997).

The geology of the area is characterised by grey and red mudstone and sandstone of the Middleton formation overlain by greenish grey and red mudstones and shales and sandstones of the

Balfour formation (Stewart & Campbell, 2001). The valley region is comprised of alluvial deposits of importance to agriculture. However, the alluvial deposits, as well as the shales and mudstone are prone to erosion (Stewart & Campbell, 2001). Sheet and gully erosion are present in the valley bottom. The valley pediment is generally dominated by loamy sand to loamy duplex type soils. Records show very high amounts of potassium, copper and boron values while zinc values are relatively low (Stewart & Campbell, 2001). Soil fertility in the valley bottom is relatively high, but on the slopes, especially near the top of the escarpment, it is low and acidic due to leaching (Collinson, 1995).

The reserve falls between the border of the Nama Karoo, Grassland and Albany Thicket biomes (Mucina & Rutherford, 2006). In the initial vegetation surveys of the reserve Boshoff and Kerley (1997) described four main vegetation types present in the area (Figure 2). Montane grassland, which is situated on the high mountain slopes and mountain tops, makes up 42.1% of the total area, and it is dominated by *Merxmuellera disticha* with some *Themeda triandra* present as well. It also hosts a few shrubs or low growing trees. Karoo Shrubland, which occurs predominantly on the valley bottom on both sides of Melk River, makes up 18.4% of the total area and is dominated by scattered trees such as *Vachellia karroo* (*Acacia karroo* according to the old convention) and *Boscia albitrunca*. Medium shrubs such as *Rhigozum obovatum, Gymnosporia polyacantha* (*Maytenus polyacantha* according to the old convention) dwarf shrubs such as *Pteronia spp* and *Pentzia incana* are also present. Succulent plants such as *Psilocaulon absimile* and *Aloe ferox* inhabit the Karoo Shrubland as well. The following grasses are also found there: *Aristida spp* and *Stipagrotis spp*. Woodland vegetation is situated on the valley bottom and envelopes Karoo Shrubland extending in all directions to the mountain slopes where it is in turn encircled by Grasslands at higher elevations. It was noted by Boshoff and Kerley (1997) that Woodland cover had increased in the area and comprised 36.3% of the total area in 1997. It is dominated by evergreen shrubs of 1-3 m height and some low growing trees. Dominant species include *Vachellia karroo*, *Celtis africana*, *Olea europaea*, *Gymnosporia polyacantha*, *Grewia robusta* and *Pappea capensis*. *Aloe ferox* is present as well. The riverine vegetation, as defined by Boshoff and Kerley (1997), comprised only 4.2% of the total area. The dominant species in this vegetation type were considered to be *Vachellia karroo*, *Rhus lancea* and the grass species, *Cynodon dactylon*. There was also a significant cover of alien species including *Populus spp*., *Schinus mollii*, *Opuntia ficus-indica*, *Eucalyptus spp*., *Atriplex nummularia* and *Agave spp.* (Boshoff & Kerley, 1997).

After the removal of livestock from the area in 1995 a mix of indigenous and non-indigenous grazers and browsers were introduced. Collinson (1995) estimated the sustainable stocking rate for the area as 1 Large Stock Unit (LSU)/15 ha. In other words, approximately 800 LSU can be sustainably kept on the reserve. Stocking rate is expressed as the area allotted to each animal unit per length of the grazeable/ browseable period of the year (Booysen, 1967), and a Large Stock Unit represents an equivalent of an animal that has a mass of 450 kg and gains 0,5 kg per day on forage with a digestible energy percentage of 55% (Meissner, 1982). When talking about how many animals can be sustainably kept on a piece of land Grazing/Browsing capacity is often used, which Booysen (1967) expressed as the area of land required to maintain one Large Stock Unit indefinitely without deterioration to

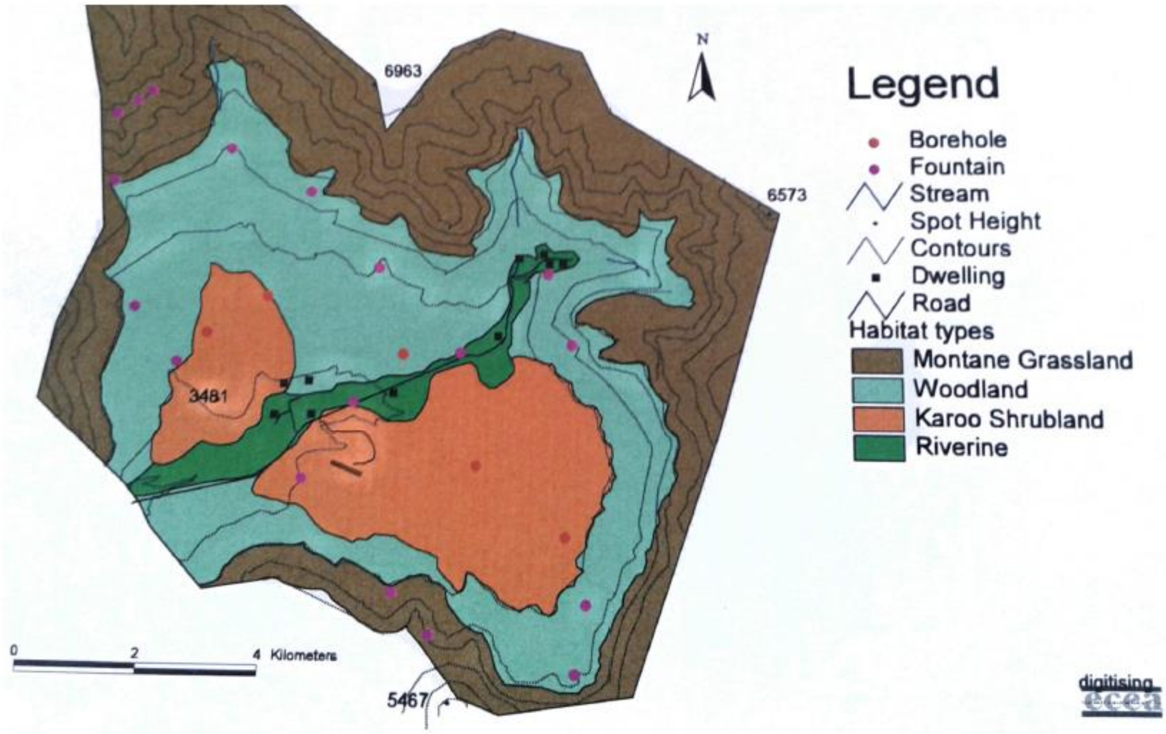

Figure 2. Vegetation map of Asante Sana Game Reserve with the main vegetation types as determined by Boshoff and Kerley (1997)

vegetation or soil. Collinson (1995) furthermore suggested that, based on the mix of vegetation on Asante Sana Game Reserve, 40% of the total stocking rate should be comprised of grazers (of which 25% of the species should be bulk grazers and 15% concentrate feeders), 40% should be comprised of mixed feeders (25% bulk grazers and 15% concentrate feeders) and 20% of should be comprised of browsers (10% bulk grazers and 10% concentrate feeders). Grazer species that were introduced to Asante Sana Game Reserve in the initial stocking period included Blue Wildebeest (*Connochaetes taurinus*) and Warthog (*Phacochoerus africanus*). The bulk mix feeders that have been introduced include Elephant (*Loxodonta africana*) and Eland (*Taurotragus oryx*) while the concentrate mixed feeders include Klipspringer (*Oreotragus oreotragus*). Bulk browsers in the reserve include Giraffe (*Giraffa camelopardalis*) while the concentrate browsers include Greater Kudu (*Tragelaphus strepsiceros*) (Boshoff & Kerley, 1997).

3.2. Vegetation types

3.2.1 Spatial pattern of vegetation types

Boshoff and Kerley's (1997) vegetation map (Figure 2) was used as a basis to determine the vegetation types for the analysis, although some changes were made to their original classification. What was called "Woodland" in Boshoff and Kerley (1997) was changed to "Thicket" to align it more directly with Mucina and Rutheford's (2006) concept of Albany Thicket as well as to better reflect the bush encroached nature of the Woodland. Riverine vegetation as proposed in Boshoff and Kerley (1997) was removed from the analysis and merged with Thicket due to the difficulty of the supervised classification process of being able to distinguish between these two vegetation types. Based on observations in the field, the Riverine vegetation was later demarcated by creating a 75 m buffer on either side of the Melk River which runs through the study area. Vegetation types "Cultivated land" and "Bare-ground" were also added to the new classification.

A supervised classification was conducted for the study area's six major vegetation types for 1987 and 2017. The years were partially determined by the relatively good availability of cloud-free images for the two years. The supervised classification was done in Google Earth Engine using the supervised classification function and minimum distance algorithm with the Mahalanobis metric (Gorelick et al., 2017), promoted by e.g. Perumal and Bhaskaran (2010). The accuracy of the classification for both years was generally good (Table 3 & 4). Multispectral Landsat 8 Top of Atmosphere (TOA) Reflectance Orthorectified (LANDSAT/LC8_L1T_TOA) and Landsat 5 TM TOA Reflectance Orthorectified (LANDSAT/LT5_L1T_TOA) products (courtesy of the U.S. Geological Survey) were used with 30 m resolution for years 1987 and 2017 respectively. For both years and for each vegetation type, 500 training points were selected and classified and used to train the classification algorithm. Annual composites were computed for both years based on pixel median values for each band to correct for atmospheric disturbances. All seven of the Landsat 5 bands (B1, B2, B3, B4, B5, B6 and B7) were used for the classification of the main vegetation types for both years. To aid in comparison, the colour ramp used to demarcate different vegetation types was adopted from Boshoff and Kerley's (1997) vegetation map of the study area.

Table 3. Confusion matrix from the supervised classification showing the relative accuracy of each classification for 1987.

		Predicted vegetation class				
		Bare-ground (n=108)	Shrubland (n=221)	Grassland (n=205)	Cultivated land (n=76)	Thicket (n=222)
Actual vegetation class	Bare ground (n=97)	**0.90**	0.01	0.00	0.07	0.01
	Shrubland (n=141)	0.00	**0.64**	0.00	0.00	0.00
	Grassland (n=183)	0.03	0.33	**0.89**	0.00	0.12
	Cultivated land (n=71)	0.02	0.00	0.10	**0.93**	0.00
	Thicket (n=193)	0.06	0.02	0.00	0.00	**0.87**

Table 4. Confusion matrix from the supervised classification showing the relative accuracy of each classification for 2017.

Actual vegetation class	Predicted vegetation class: Bare-ground (n=102)	Shrubland (n=226)	Grassland (n=183)	Cultivated land (n=69)	Thicket (n=187)
Bare ground (n=95)	**0.93**	0.04	0.02	0.00	0.30
Shrubland (n=178)	0.00	**0.79**	0.00	0.00	0.00
Grassland (n=178)	0.01	0.18	**0.97**	0.00	0.05
Cultivated land (n=69)	0.00	0.00	0.01	**0.99**	0.00
Thicket (n=120)	0.06	0.00	0.01	0.00	**0.64**

3.2.2 Long-term change in vegetation types

For a description of the broad changes in vegetation types, QGIS r.report algorithm (QGIS Development team, 2017) was used to calculate the area that each vegetation type occupied in 1987 and 2017, from which percentage changes could be derived. For further analysis, 500 random sites were selected in the landscape and proportionally distributed for each vegetation type using the random point constructor function in base package R (RStudio Team, 2015). Vegetation type was assigned for each site based on the results of the supervised classifications for 1987 and 2017. Riverine thicket was excluded from the temporal analysis of vegetation types because it was classified independent of the supervised classification. Therefore, in this analysis the area for Riverine thicket remained the same over time. Generalized linear mixed models from lme4 package (RStudio Team, 2015; Bates et al., 2005) were used to determine if there was a significant change in the number of sites in each vegetation type between 1987 and 2017 (see Table 3). Generalized linear mixed models were chosen over generalized linear models to avoid the problem of pseudo replication, which arises when the same sites are sampled multiple times. All predictor variables were scaled and centred and the 95% confidence intervals profiled to determine the significance of each parameter. All statistical analyses were conducted in R Statistical Software 3.3.3 (RStudio Team, 2015). Additionally, the proportion of sites which did not change from one vegetation type to another between 1987 and 2017 was calculated and compared to the proportion of sites which changed from one vegetation type to another.

3.3 Vegetation cover and productivity

3.3.1 Spatial pattern of vegetation cover and productivity

The Normalized Difference Vegetation Index (NDVI) and the Soil Adjusted Vegetation Index (SAVI) are commonly used to determine changes in vegetation productivity and cover over space and time (Palmer & van Rooyen, 1998; Colditz et al., 2007; Hüttic et al., 2009; Wessels et al., 2011) and are defined by the following formulas:

$$\text{NDVI} = \frac{\rho\text{NIR} - \rho\text{RED}}{\rho\text{NIR} + \rho\text{RED}}$$

$$\text{SAVI}=\frac{\rho\text{NIR} - \rho\text{RED}}{\rho\text{NIR} + \rho\text{RED} + 0.5} \text{ x } 1.5$$

where ρ is the atmospherically corrected reflectance in the near infrared and red spectra, and coefficients are: C1 = 6.0 and C2 = 7.5. The soil adjustment factor is furthermore expressed as: L = 1.0 with a gain factor of G = 2.5 (Jensen, 2009; Sesnie et al., 2012). However, because SAVI is a corrected version of the Normalized Difference Vegetation Index (NDVI) and is better suited for analysing vegetation cover change in semi-arid and arid regions (see literature review for full details), it was used in this study.

Landsat data was chosen over MODIS because of MODIS's lower spatial resolution. For establishing medium-scale spatial and long-term temporal trends it was considered better to use high spatial resolution rather than high temporal resolution. Multispectral Landsat 5 surface reflectance pre-collection (LANDSAT/LT5_SR), Landsat 7 surface reflectance pre-collection (LANDSAT/LE7_SR) and Landsat 8 surface reflectance pre-collection (LANDSAT/LC8_SR) products (courtesy of the U.S. Geological Survey) with 30 m resolution were used to derive SAVI values.

3.3.2 Long-term change in vegetation cover and productivity

The mean of the small integrated value has been proposed as a proxy for annual plant productivity (Eklundh & Jönsson, 2012; Wessels et al., 2011). However, to calculate the small integrated value, monthly SAVI data are required. Because the availability of Landsat imagery varies between years it was not possible to obtain monthly time series data for the study area. Therefore, the approach used here was to compute SAVI annual composites based on median pixel values (corrected for atmospheric disturbances) for all pixels in the study area in Google Earth Engine (Gorelick et al., 2017). These values were then used to calculate the mean values for three-year periods at the beginning (1989-1991) and at the end of the analysis (2014-2016). The three-year mean values were used to determine the long-term change in vegetation cover for each vegetation type. The three-year mean values derived from one-year median values were chosen to better represent the overall SAVI scores compared to one-year median value, which is easily skewed by drought or exceptional rainfall for that year. The limitation of this approach is that it is likely to underestimate the annual productivity of deciduous plants compared to the mean of the small integrated value as deciduous plants usually drop their leaves during the winter months.

The reason for not starting the three-year time series in 1987 is because the quality of the satellite data for the study area in 1988 was very poor. Both years 1987 and 1988 were therefore, discarded from the analysis because of the discontinuity that would have occurred by only eliminating year 1988. Year 2017 was also eliminated from the analysis because it was still incomplete when the analysis was done. Including only half of the year into the composite could have potentially influenced the median SAVI values for that year.

The 500 random points created to analyse the change in vegetation types were each assigned SAVI values based on the three-year mean values. This was done to analyse the change in vegetation productivity between the start and the end of the study period. Generalized linear mixed models from lme4 package (RStudio Team, 2015; Bates et al., 2005) were used to explore whether changes in SAVI in each vegetation type were significant (Table 3, Figure 12). Significant differences in the change in SAVI were also analysed for areas of special interest (e.g. Thicket that remained Thicket compared to Thicket that changed to Grassland or Shrubland and vice versa). All predictor variables were scaled and centred and the 95% confidence intervals profiled to determine the significance of each parameter. All statistical analysis was conducted in R Statistical Software 3.3.3 (RStudio Team, 2015).

3.3.3 Field estimates of vegetation type and cover

Field determinations of vegetation type and estimates of cover in the Asante Sana Game Reserve were undertaken from 23-28 October 2017. This was done to assess the accuracy of the supervised classification with on-the-ground determinations of vegetation type as well as to relate field measurements of vegetation cover with satellite-derived estimates of vegetation productivity. Sampling sites were determined in a stratified random manner to represent the full SAVI gradient in each of the six vegetation classes (Figure 3). A total of 60 sites (7 Bare-ground, 15 Shrubland, 12 Grassland, 5 Cultivated, 10 Thicket and 11 Riverine thicket (Figure 4) were surveyed for vegetation cover, species composition and other biophysical characteristics. The results which show species composition and biophysical characteristics of each vegetation type are described separately from those which show estimates of vegetation cover. Due to the inaccessibility of mid-northern and north-eastern slopes, these areas were excluded from the field survey.

Biophysical characteristics of the sites were recorded to aid in description of the vegetation types. A measuring tape was used to demarcate plots of 4 x 4 m in which the total vegetation cover, and cover within the ground layer (<0.5 m), mid-layer (0.5-3 m) and canopy layer (>3 m) was estimated. The percentage cover was estimated by demarcating sections equivalent to half, quarter, eight and sixteenth of the 4 x 4 m plot, and systematically using those reference areas to assess the total vegetation cover. Independent estimates were given by two people, which were averaged for the final cover value. The percent cover of the dominant species in each plot was also estimated as was the cover of rock, litter and bare-ground. The incidence of recent herbivory was indicated by the amount of dung in each plot and assessed on a 5-point scale (0=none; 1=rare; 2=occasional; 3=common; 4=abundant) while a subjectively-determined disturbance score (ranging from 1=none to 10=highly disturbed) was also provided for each plot. This was assessed according to the degree and extent of erosion visible in a plot as well as the composition and cover of plant species in the plot relative to other sites in the area considered as being benchmark sites for a particular vegetation type. Following discussion and independent assessment by two recorders a consensus disturbance value was agreed upon for each plot. Soil texture was recorded using the Feel Flow Chart approach outlined in Ritchey et al. (2015). GPS coordinates and elevation for each of the sampling sites were also recorded. Photographs of unidentified common species were taken for further identification.

To assess the strength of the relationship between satellite- and field-derived estimates of productivity and cover, SAVI values and vegetation cover estimates from the field surveys were regressed against each other. A measure of fit (the R^2 value) and the significance level were derived from a linear model using base functions in R Statistical Software (RStudio Team, 2015) (Table 3).

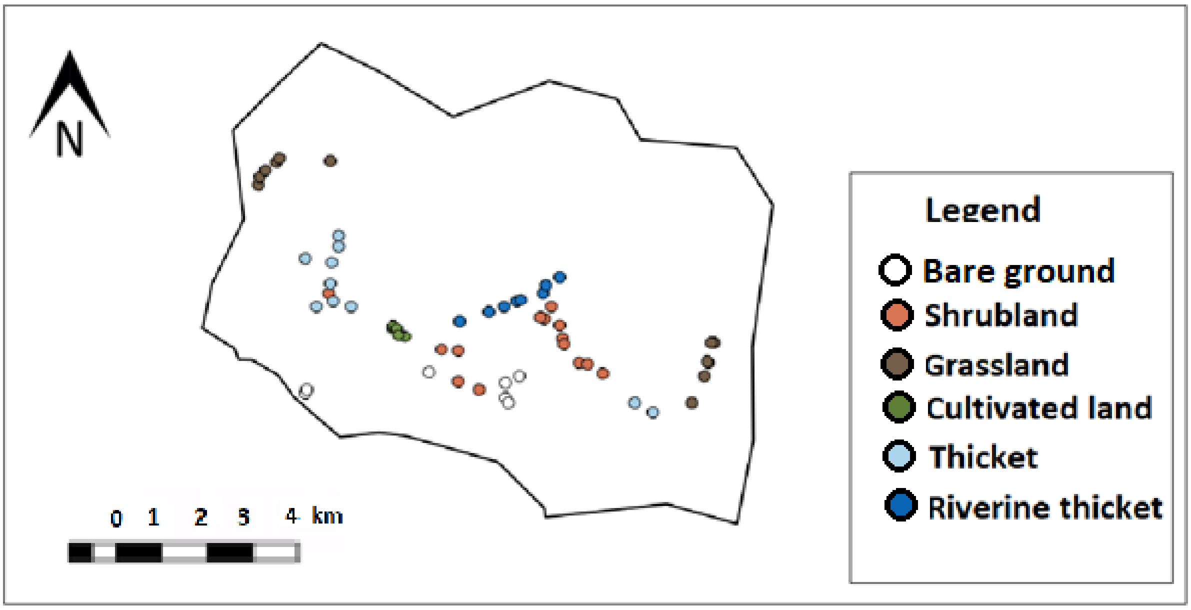

Figure 3. The location of the 60 plots sampled in the six vegetation types of Asante Sana Game Reserve in October 2017.

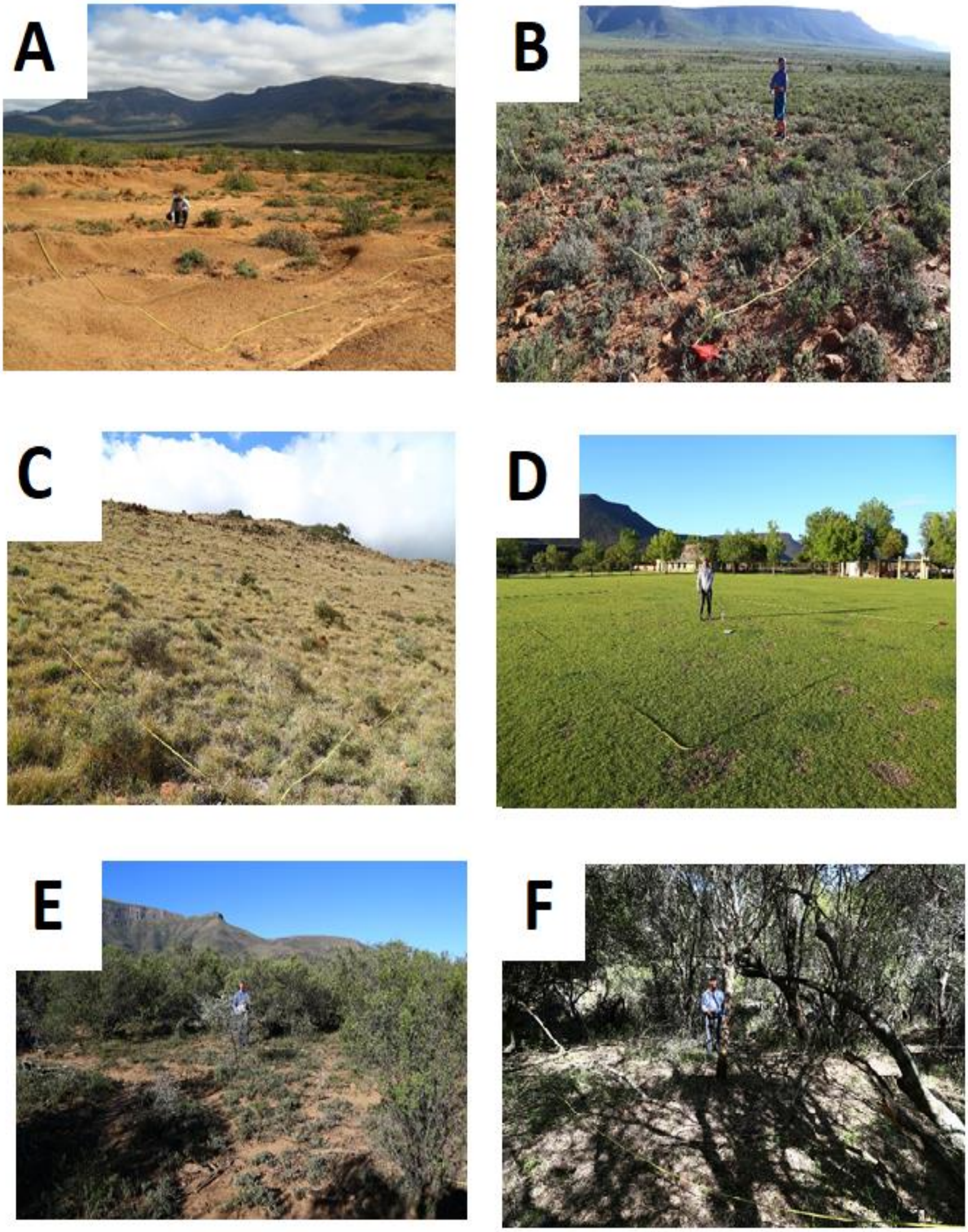

Figure 4. Main vegetation types in Asante Sana Game Reserve. A) Bare-ground B) Shrubland C) Grassland D) Cultivated land E) Thicket F) Riverine thicket.

3.4 *The effect of drivers on vegetation cover and productivity*

Data on rainfall, fire and herbivory were used as covariates to understand the changes observed in SAVI over time. Annual SAVI composites based on yearly pixel median values were calculated for each year between 1989 and 2016. Annual rainfall data for the period from 1989 to 2016 were obtained from a single rain gauge located at the reserve manager's house. No other rainfall records for the reserve are available. The spatial resolution of rainfall data remains at the reserve level and, therefore, determines the scale at which management considerations can be discussed. Publicly-available remotely sensed rainfall data such as The Tropical Rainfall Measuring Mission (TRMM) did not have any better spatial resolution compared to the data obtained from the reserve, and therefore were not used in the analysis. MCD45A1.051 Burned Area Monthly L3 Global 500 m were extracted from Google Earth Engine (Gorelick et al., 2017) for all years between 2000-2016. Game stocking number data were obtained from aerial game counts undertaken by Asante Sana Management and were only available for the years 2007, 2009, 2010, 2013, 2015 and 2017 (see the detailed aerial count data in Appendix D). Data on each driver (rainfall, burning and stocking numbers) was calculated for the 500 random sites created for previous analyses. Generalized linear mixed models were used to assess the relationship between the change in SAVI in each vegetation type and rainfall, burning and stocking numbers (Table 5). Vegetation types determined from the 1987 supervised classification were used to assign SAVI values for each vegetation type up until 2000 and those determined from the 2017 analysis were used from 2001 onwards. Again, all predictor variables were scaled and centred and the 95% confidence intervals profiled to determine the significant of each parameter. All statistical analysis was conducted in R Statistical Software 3.3.3 (RStudio Team, 2015). All methods are summarized in Figure5.

Table 5. Statistical tests used to check for significant results for changes in vegetation type and cover.

Response variable	Explanatory variable(s)	Sample size	Random effect	Test	Distribution
Vegetation type	Year	500	Cell ID	GLMM	Binomial
SAVI	Year	500	Cell ID	GLMM	Gaussian
Vegetation cover	SAVI	60		LM	Gaussian
SAVI	Rainfall + Burning + Stocking density	500	Cell ID	GLMM	Gaussian

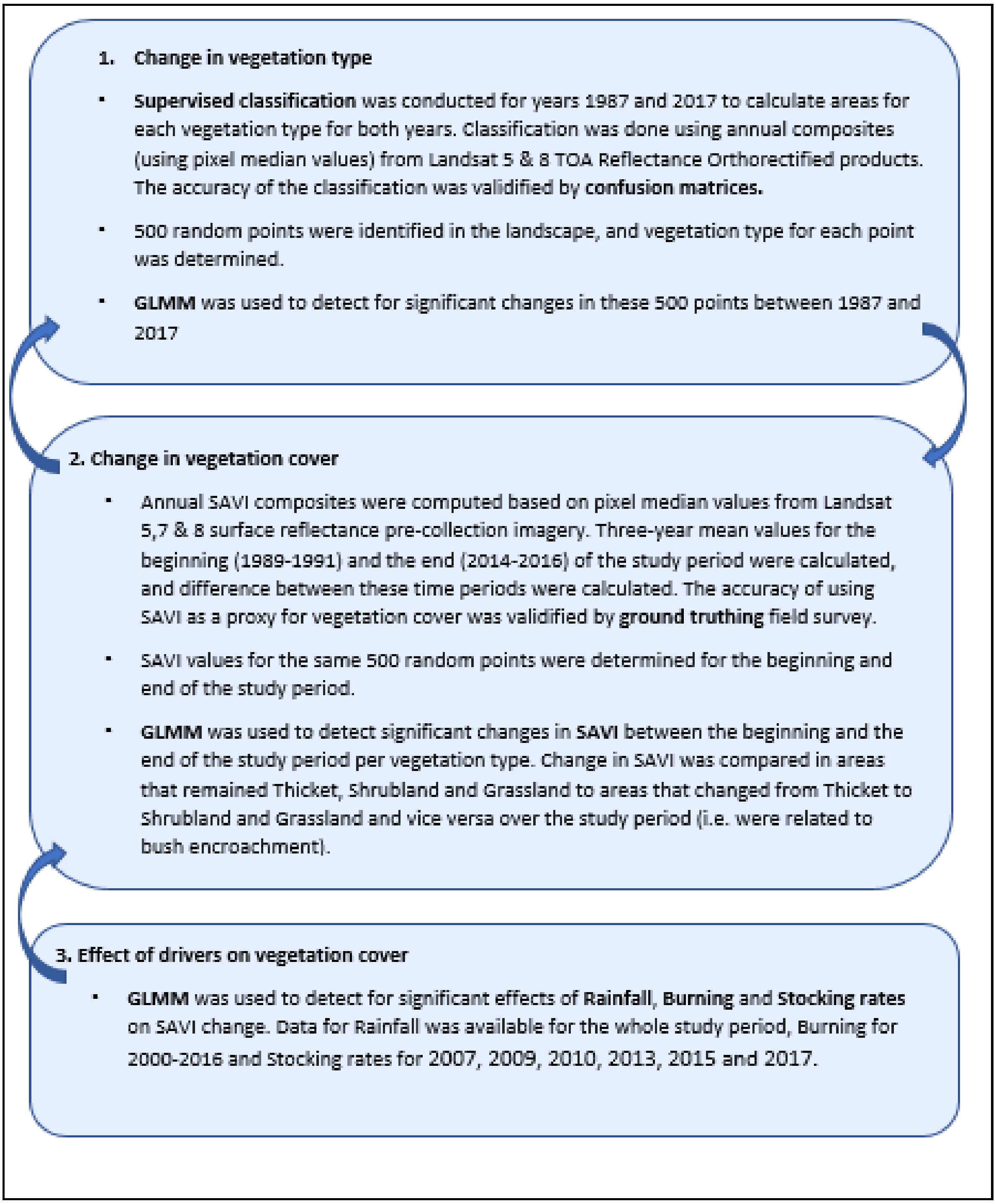

Figure 5. Summary of methods for analysing 1. Changes in vegetation type, 2. Changes in vegetation cover and 3. The effects of drivers on vegetation cover change.

4. Results

4.1 Vegetation types

4.1.1 Spatial pattern of vegetation types

The Landsat based supervised classification maps for both 1987 and 2017 (Figure 6) show the distribution of each of the six vegetation types in Asante Sana Game Reserve and how they have changed between the two time steps. In both maps Bare-ground was distributed in the central area of the valley bottom and was surrounded by either Shrubland or Thicket. Extremely eroded soils, usually associate with Bare-ground were found where sheep, goat and ostrich farming had been the most intense before the start of game farming. Shrubland was also located on the valley bottom on both sides of Melk River surrounding the houses and Cultivated land. Thicket surrounded the entire valley bottom with Grassland encircling it at elevations above 1400 m. A 75 m buffer on either side of the Melk River was used to demarcate Riverine thicket. Field observations showed, however, that typical Riverine thicket species such as *Olea europaea*, *Celtis africana* and *Diospyros lycioides* were also associated with the many smaller streams which occur on the south-facing slopes of the reserve, particularly in the northwest.

Based on our ground-truthing exercise the supervised classification closely matched the results from the field survey which provided additional information on each of the main vegetation types (Table 6). Results from the field survey showed that vegetation cover ranged between 0-95% across the different vegetation types and was lowest in Bare-ground and highest in Riverine thicket. Shale and dolorite dominated the geology of the area although Riverine thicket occurred on alluvium and sandstone. Where vegetation was present in locations mapped as Bare-ground the sites were dominated by disturbance-adapted dwarf shrubs such as *Chrysocoma ciliata.* Shrubland vegetation was dominated by dwarf shrubs typical of the Nama Karoo such as *Pentzia incana* and *Eriocephalus ericoides.* Grassland on the other hand was dominated by perennials such as *Themeda triandra* on the north-western slopes *and Merxmuellera disticha* on the eastern slopes. Cultivated land was dominated by a mixture of commercially produced lawn grass species but also *Cynodon dactylon*. Thicket was dominated by trees such as *Vachellia karroo* and *Searsia lucida* (*Rhus lucida* according to the old convention) as well as some medium shrubs (e.g. *Lycium cinereum*) and dwarf shrubs (e.g. *Pentzia incana*). Riverine thicket was dominated by trees such as *Vachellia karroo* and *Rhus spp*. with *Olea europaea* and *Celtis africana* present in some locations.

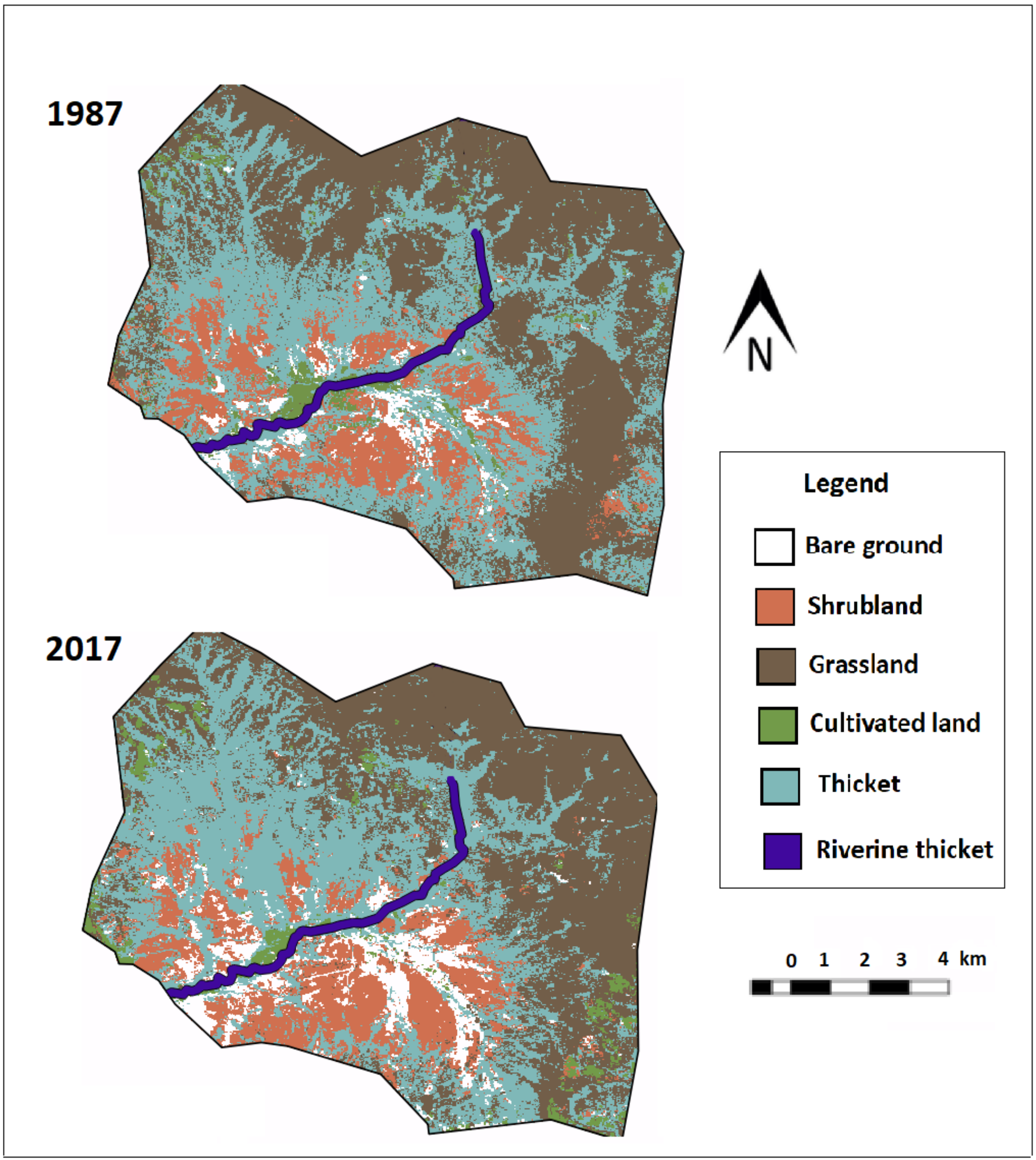

Figure 6. Main vegetation types of Asante Sana reserve in 1987 and 2017. The classifications were derived from annual composites (based on yearly median values) of multispectral Landsat 5 and 8 satellites from Google Earth Engine.

Table 6. Geology, soil texture, vegetation cover, degradation, dung abundance and dominant species composition per vegetation type of 60 ground survey sites in Asante Sana Game Reserve.

Vegetation type	Geology	Range in soil texture	% vegetation cover (mean±std; range)	Dung abundance (mean±std)	Degradation score (mean±std)	Dominant species
Bare-ground	Shale	Loamy sand – Sandy clay	7.7±5.6 0-15	0.7±0.75	9.1±0.4	*Pentzia incana, Eriocephalus ericoides, Chrysocoma ciliata, Lycium cinereum, Psilocaulon absimile, Cynodon dactylon, Vachellia karroo*
Shrubland	Dolorite/ shale	Loamy sand – Sandy clay	42.9±19.1 8-70	2.3±1.2	5.6±2.3	*Pentzia incana, Chrysocoma ciliata, Eriocephelus ericoides, Lycium cinereum, Walafrida sp., Pteronia incana, Helichrysum sp., Psilocaulon absimile*
Grassland	Dolorite	Loamy sand – Sandy clay	65.0±17.8 40-90	1.7± 0.7	3.4± 2.0	*Cymbopogon plurinodis, Themeda triandra, Anthospermum sp., Felicia filifolia, Euclea sp., Elytropappus rhinocerotis, Merxmuellera sp., Diospyros austro-africana*
Cultivated land	-	Sandy clay	86.2±9.6 70-93	1.6±1.5	10.0± 0	*Mixed grass species but mainly Cynodon dactylon*
Thicket	Dolorite/ shale	Loamy sand – Sandy clay	60.0±14.1 40-80	2.6±1.2	5.0± 1.6	*Vachellia karroo, Searsia lucida, Pentzia incana, Chrysocoma ciliata, Panicum maximum, Lycium cinereum, Cynodon dactylon*
Riverine thicket	Alluvium/ sandstone	Loamy sand – Silty clay loam	72.2±14.2 45-95	2.5± 0.7	3.3±1.6	*Vachellia karroo, Searsia lucida, Olea europaea, Celtis africana, Diospyros lycioides*

4.1.2 Long-term change in vegetation types

The vegetation maps that resulted from the supervised classification (Figure 6) together with Figure 7 show the change in the area covered by each vegetation type over time. The results showed an increase in the proportional area of Bare-ground. Shrubland, changed little overall, but closer inspection suggests that it decreased on the mid-northern section and increased in the mid-southern section of the reserve. Grassland decreased slightly overall and especially on the north western slopes of the reserve. However, there are also indications that it increased in extent on the north eastern slopes. Cultivated land appears to have increased over time, but the extent to which the increase seems realistic is discussed further on. Thicket shows a slight decrease in the area covered by this vegetation type, but closer observations reveal that it has decreased on the mid-southern and eastern sections of the reserve, while it has increased substantially in the western areas of the reserve.

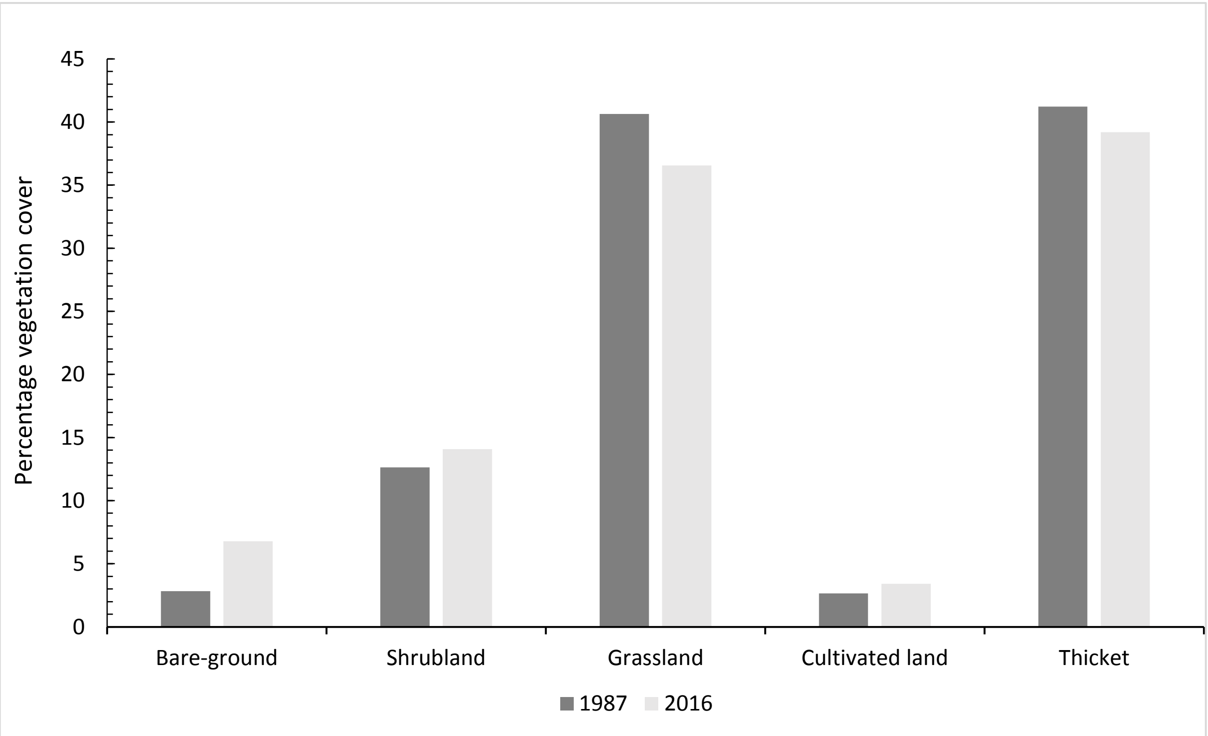

Figure 7. Percentage vegetation cover per vegetation type in Asante Sana Game Reserve in 1987 and 2017 obtained from supervised classifications.

A further analysis was carried out to understand the relative proportion of pixels in each vegetation type that was classified as having remained the same over time or that was classified as having changed to one of the other vegetation types between 1987 and 2017 (Table 7). For instance, nearly 25% of the pixels that were classified as Grassland in 1987 were classified as Thicket in 2017, whereas the same proportion that was classified as Thicket in 1987 was classified as Grassland in 2017. In contrast, 22% of the pixels classified as Shrubland in 1987 were classified as Thicket in 2017 while only 13% of Thicket was classified as Shrubland.

Table 7. Proportion of each vegetation type of either staying same or changing to another vegetation type between 1987 and 2017 in Asante Sana Game Reserve

Initial vegetation type in 1987	Final vegetation type in 2017			
	Bare-ground	Shrubland	Grassland	Thicket
Bare-ground	***93.7***	0.1	0.0	6.3
Shrubland	2.6	***70.1***	5.2	22.1
Grassland	0.0	0.9	***74.3***	24.8
Thicket	5.0	12.8	24.7	***57.5***

4.2 Vegetation cover and productivity

4.2.1 Spatial pattern of vegetation cover and productivity

The mean SAVI values for Asante Sana Game Reserve over the three-year period (2014-2016) were the highest for Thicket and Riverine thicket vegetation types particularly where these vegetation types occur along smaller streams and along the banks of the Melk River (Figure 8). The second highest values were for Grassland and Cultivated land, the second lowest for Shrubland and the lowest for Bare-ground. Each vegetation type had significantly different SAVI signatures ($p<0.05$) from each other, except for two pairs: Thicket and Riverine thicket and Grassland and Cultivated land.

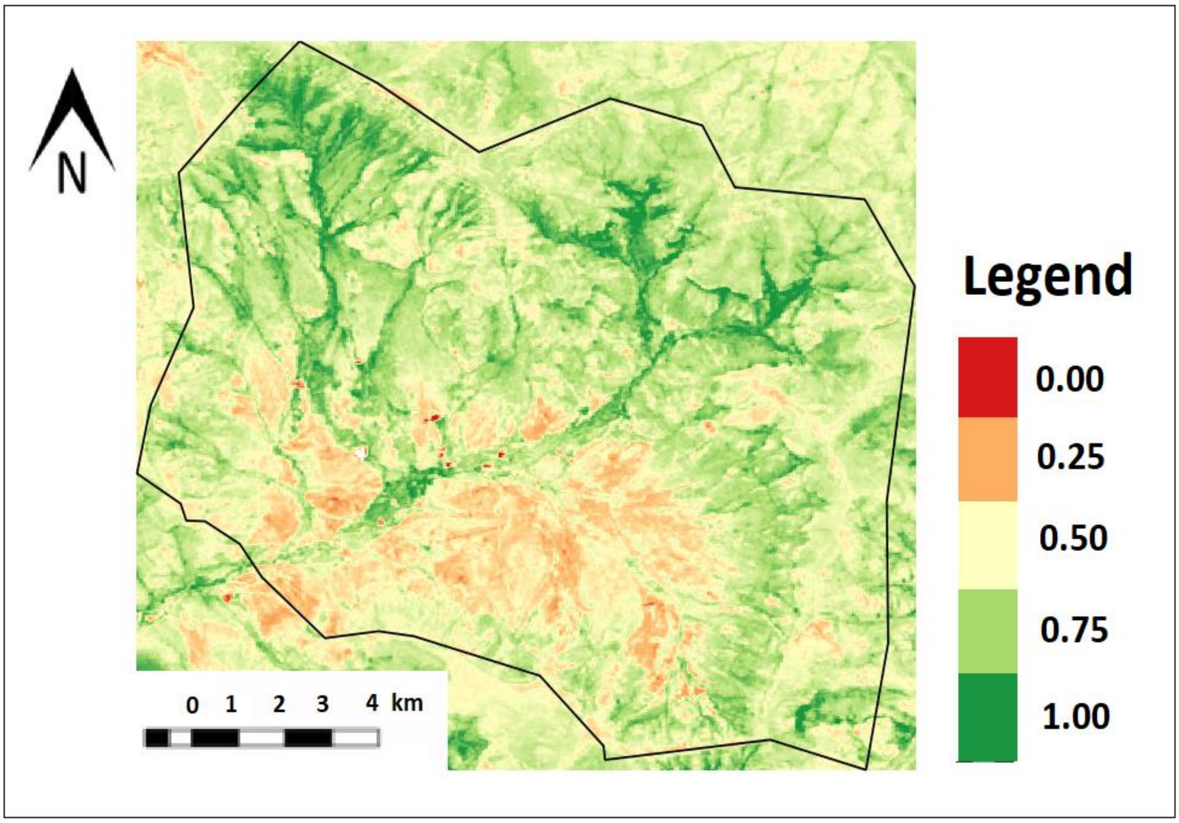

Figure 8. Mean SAVI values between 2014 and 2016 for the vegetation of Asante Sana Game Reserve

4.2.2 Long-term change in vegetation cover and productivity

A comparison of mean SAVI values between 1989-1991 and 2014-2016 indicated a positive long-term trend (Figures 9, 10 and 11) in this value over most of the reserve. SAVI has increased the most in areas that were used for ostriches and areas that were previously cultivated for fodder crops such as Lucerne. It has also increased steadily over most of the Thicket and Shrubland areas. On the mid-northern highland plateau areas, which are dominated by Grasslands, SAVI appears to have decreased between the two time periods.

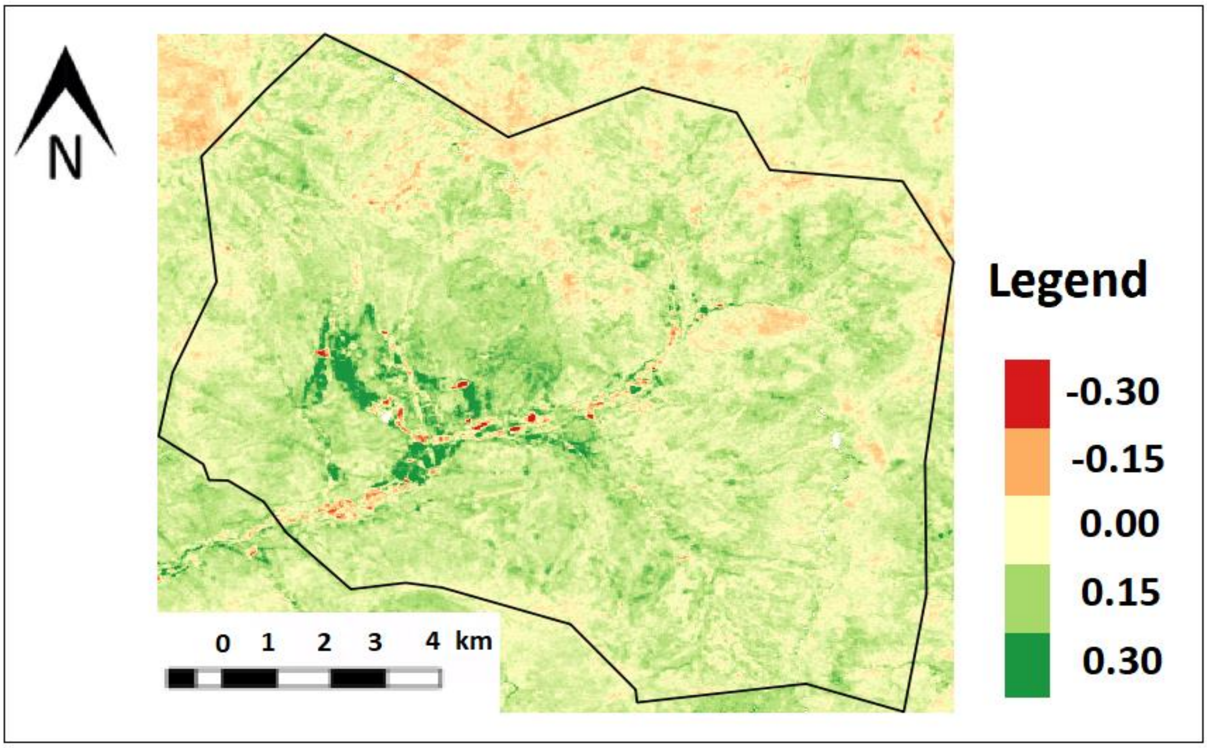

Figure 9. Long term change in SAVI reflecting the difference between the mean values of 1989-1991 and 2014-2016.

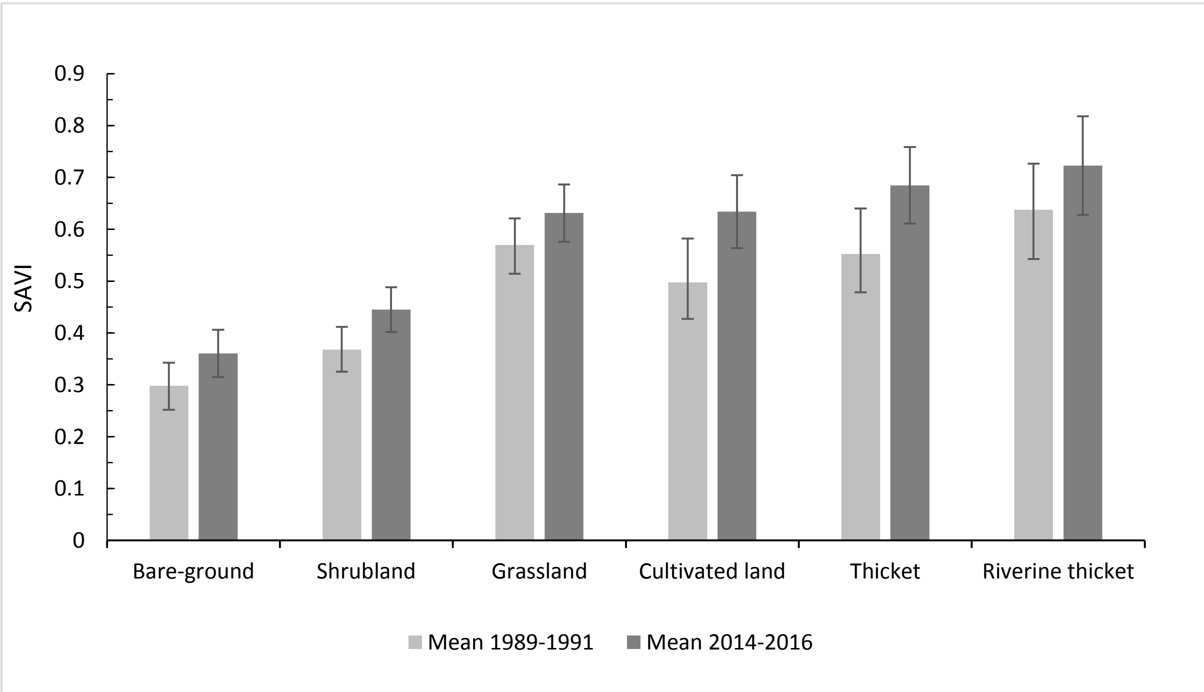

Figure 10. A comparison of the mean SAVI values for six vegetation types on Asante Sana Game Reserve between 1989-1991 and 2014-2016. All differences between mean SAVI values in a vegetation type over the period 1989-1991 were significantly lower than those for the same vegetation type over the period 2014-2016 ($p<0.05$).

The rate of change (i.e. the slope of the linear trend line) in long-term SAVI values were closest to the overall mean values (B=0.0031) in Grassland (B=0.0025), higher in Thicket (B=0.0056) and Riverine thicket (B=0.0038) and Cultivated land (B=0.0046) and lower in Bare-ground (B=0.0013) and Shrubland (0.0019) (Figure 11). The highest SAVI increase was in Thicket and lowest on Bare-ground.

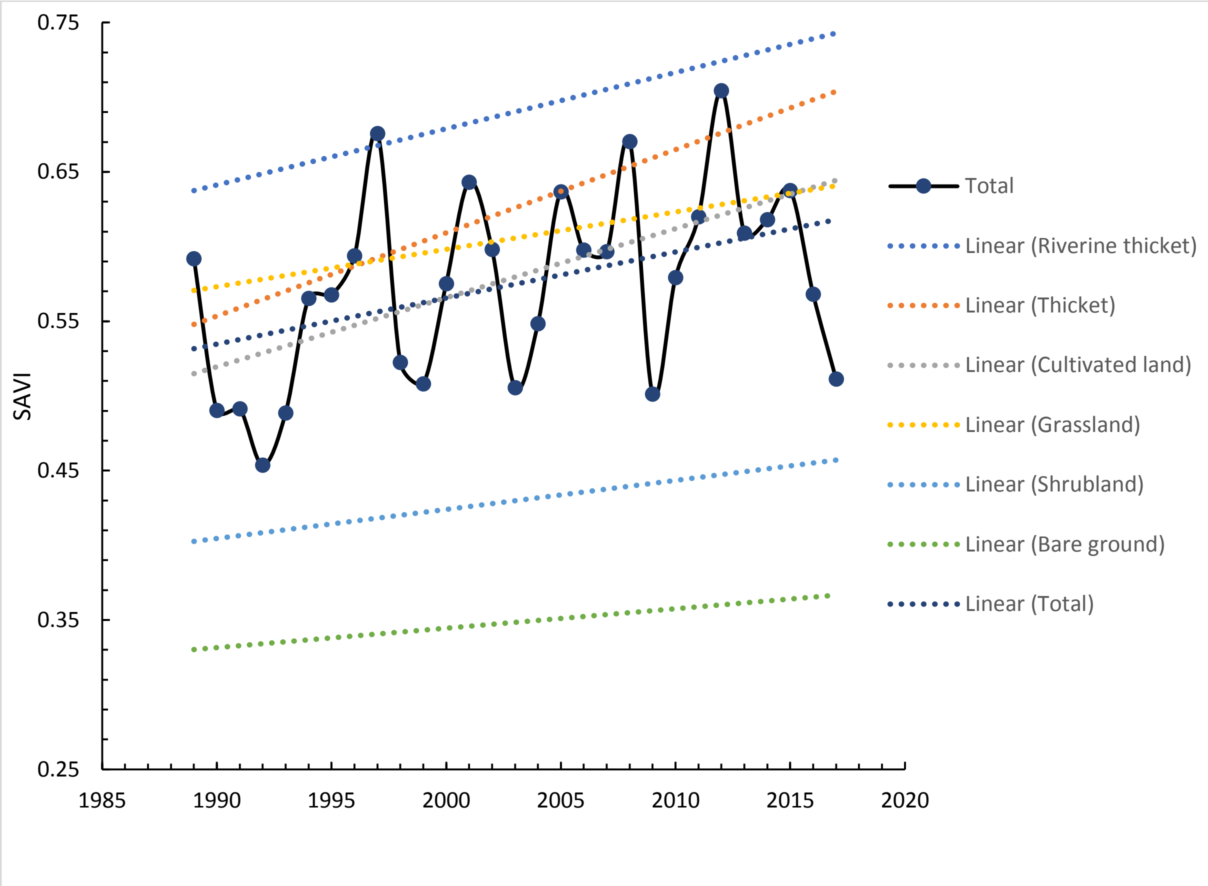

Figure 11. Time series of SAVI values in each vegetation type between 1989 and 2016 in Asante Sana Game Reserve showing the rate of change in SAVI for each type.

To show whether changes from one vegetation type to another had a significant impact on SAVI values, generalised linear mixed models were used and slope coefficient evaluated to further assess the magnitude and direction of the change. Results show significant differences in SAVI values for pixels that have changed in vegetation type compared to pixels that have maintained vegetation type (Table 8). The main findings from the analysis are that pixels of Grassland and Shrubland that changed to Thicket experienced significantly higher SAVI increase compared to those that maintained the same vegetation type. Furthermore, pixels of Thicket that changed to Shrubland had significantly lower values for change in SAVI compared to those pixels that were classified as Thicket vegetation in 1987 and that remained Thicket vegetation in 2017.

Table 8. Coefficients of difference between SAVI values in areas that maintained vegetation type compared to those that changed from one vegetation type to another between 1987 and 2017. Empty cells indicate that sample sizes were too small for performing a GLMM on the subset in question. Values in bold type are those in which the 95% confidence intervals do not overlap zero and therefore reflect significant differences.

Initial vegetation type	**Final vegetation type**			
	Bare-ground	Shrubland	Grassland	Thicket
Bare-ground				
Shrubland	-0.0521		**-0.0943**	**0.12942**
Grassland		-0.0534		**0.04164**
Thicket	**0.208**	**-0.1924**	-0.0279	

4.2.3 Field estimates of vegetation cover

Most of the plots surveyed (80%) corresponded to the vegetation class assigned by the supervised classification. Only 2 plots that were identified as Shrubland in the supervised classification were marked as Bare-ground in the field. All plots that were identified as Grassland on the north-western slopes in the field were classified as Cultivated land in the supervised classification. A total of 4 plots identified as Thicket in the field were misclassified in the supervised classification, one as Shrubland, one as Grassland and two as Bare-ground.

Regression analysis shows a positive correlation between the field vegetation cover estimates and SAVI values (B = 0.006; R^2=0.6 and p<0.01) (Figure 12). This suggests that SAVI values can be used as a proxy for vegetation cover in the study area.

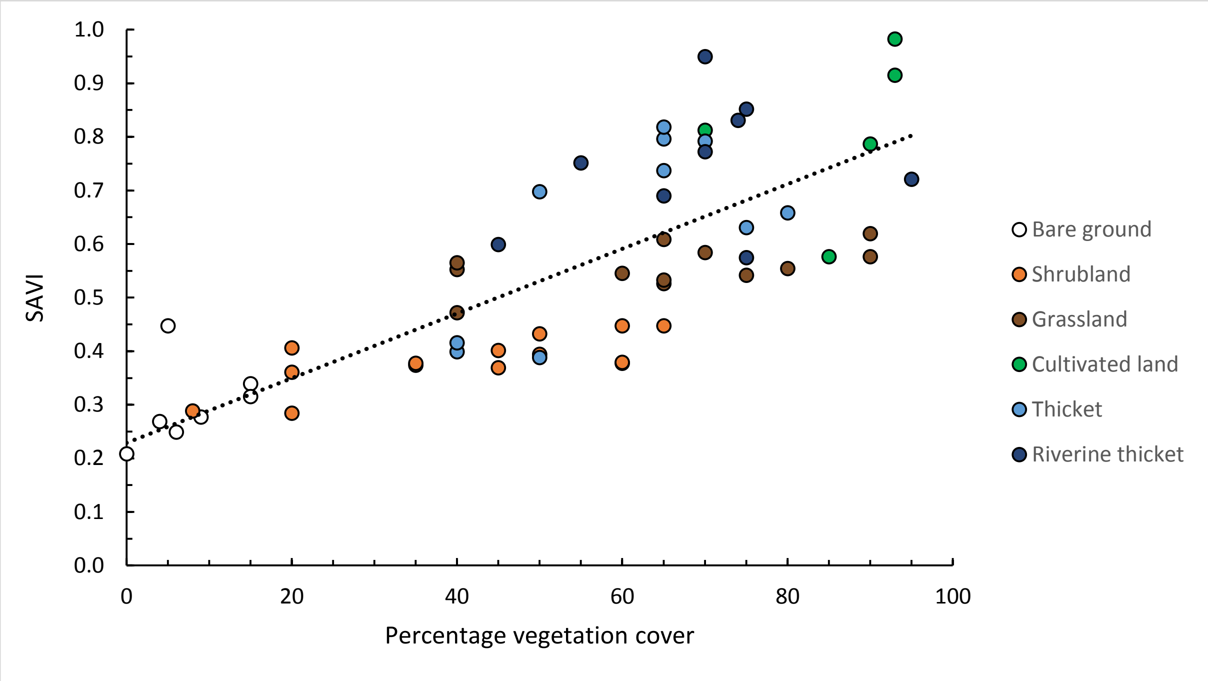

Figure 12. Results from the regression analysis of SAVI in relation to percentage vegetation cover as estimated in the field survey.

3.4 Drivers of change

4.2.4 Rainfall

Annual rainfall totals increased significantly (B = 8.36, $P < 0.01$) between 1989 and 2016 (Figure 13). Years of low rainfall (annual rainfall less than 400 mm) occurred in 1991, 1992, 1993, 1999, 2003, 2008 and 2016 and years of high rainfall (more than 600 mm) occurred in 2001, 2006, 2011 and 2015.

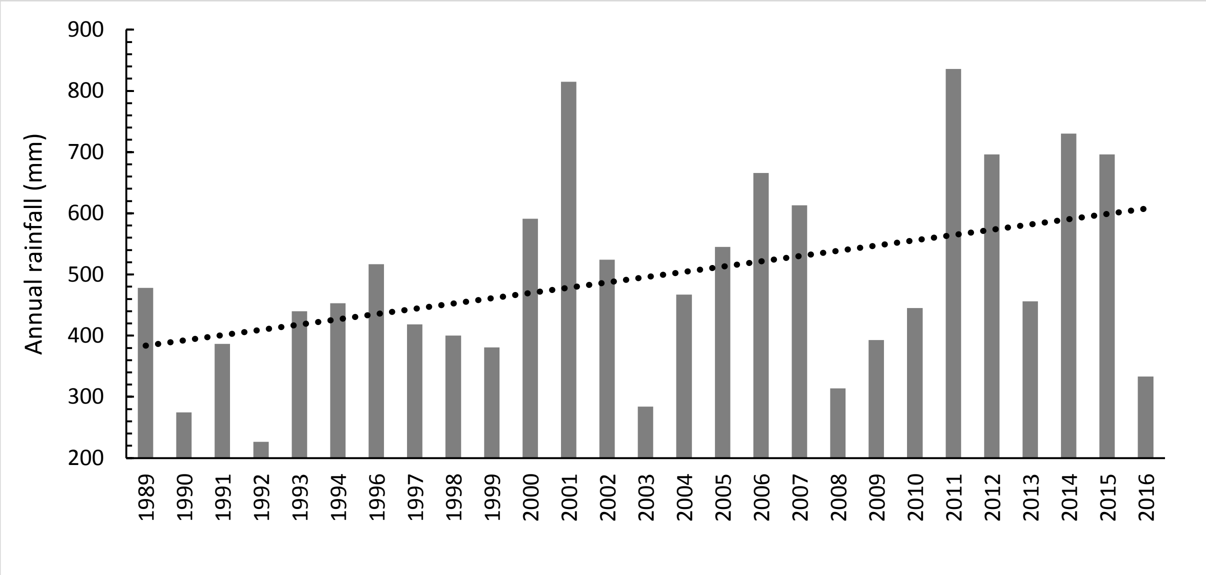

Figure 13. Annual rainfall in Asante Sana Game Reserve between 1989 and 2016. A linear trend line is also shown.

4.2.5 Burning

Since 2000, three substantial fires have occurred on Asante Sana Game Reserve (Figure 14). Two small fires occurred on the norther part of the highlands in 2003, and 2008 while a relatively larger fire took place on the north eastern Grassland in 2016.

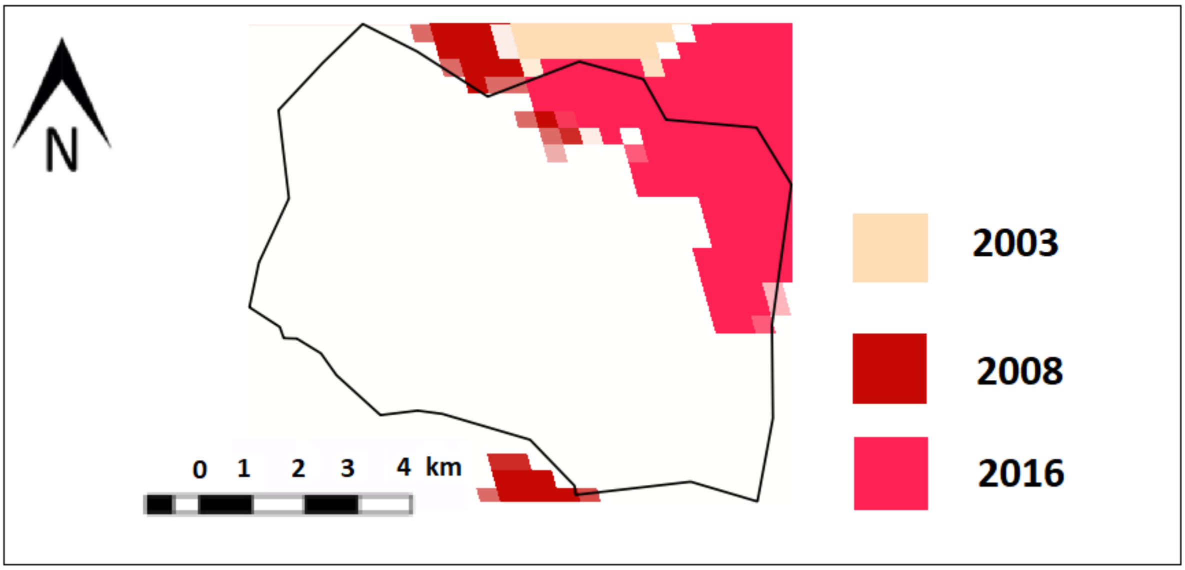

Figure 14. The incidence of fire in the Asante Sana Game Reserve in 2003, 2008 and 2016. Data are from MCD45A1.051 Burned Area Monthly L3 Global 500m extracted from Google Earth Engine for the period 2000-2016.

4.2.6 Stocking numbers

The number of animals on Asante Sana Game Reserve, expressed as the number of Large Stock Unit equivalents (LSU) remained relatively high (1382 LSU) over the period 2007-2009 compared to the recommended number of ~800 LSU from Collinson (1995) (Figure 15). In 2010, however, the number of animals dropped to 594 LSU largely because large numbers of animals were sold. Grazers and mixed feeders have been consistently more prevalent over time compared to browsers. See Appendix D for detailed areal counts for the different game species in the reserve

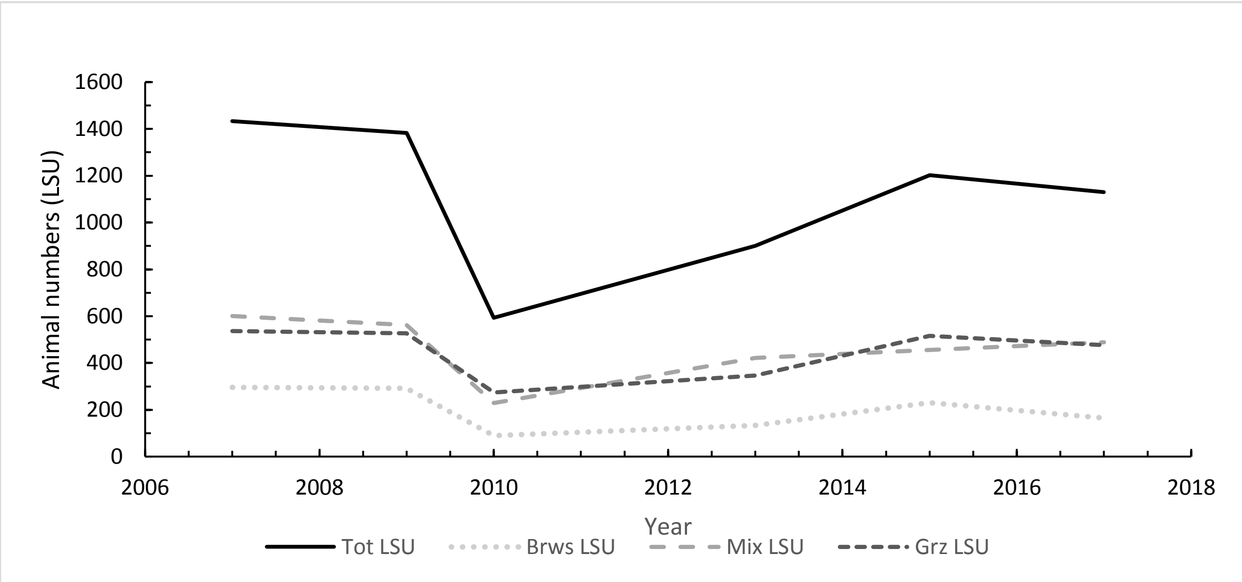

Figure 15. Animal numbers of game in Asante Sana by feeding type in Large Stock Units (LSU). Data were obtained from aerial surveys by the park management.

4.2.7 The effect of drivers on vegetation cover and productivity

The regression analysis (Figure 16) showed that there was a significant positive (B=1560, $P < 0.01$) correlation between rainfall and SAVI suggesting a clear link between these two factors. For unknown reasons, years 1997 and 2008 had big disturbances in the median SAVI values for the area. When they were removed from the analysis R^2 value increases from 0.39 to 0.67.

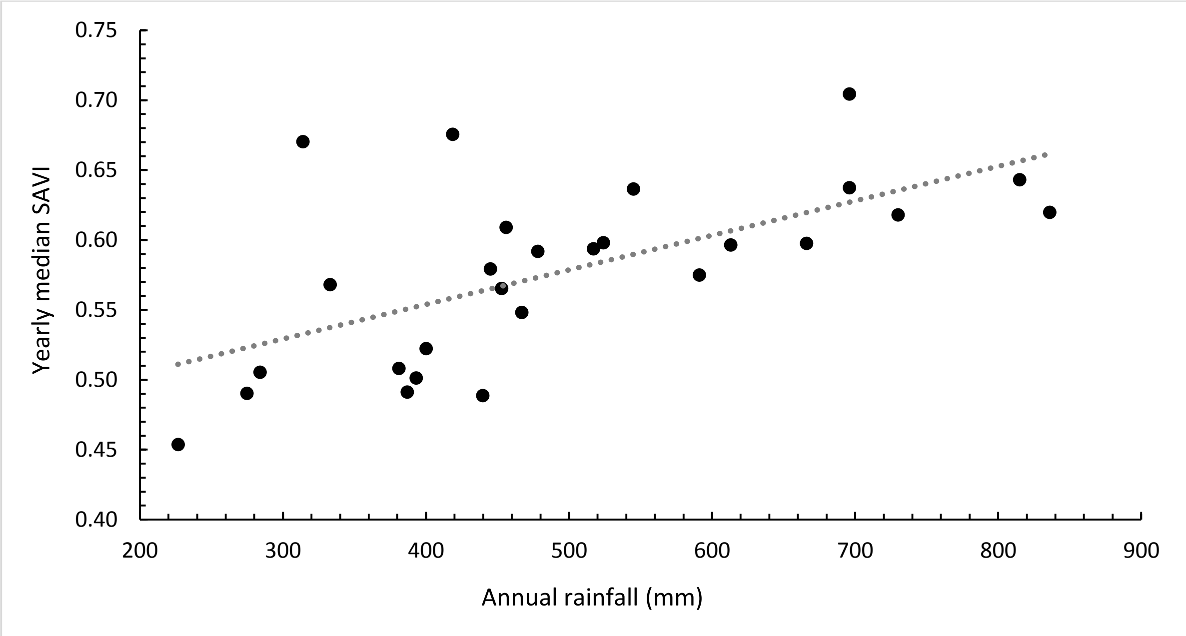

Figure 16. Relationship between annual rainfall and yearly mean SAVI.

Results from Generalized linear mixed models show that rainfall (B=0.04, CI[0.039, 0.041]) and burning (B=0.003, CI[-0.003, -0.001]) had a positive impact on SAVI, whereas stocking numbers had a negative impact (B=-0.002, CI[0.002,0.004] in each vegetation type (Appendix B). All predictor variables were scaled, and based on the magnitude of the coefficient, rainfall had a 10 times greater impact on SAVI compared to fire and stocking numbers.

5. Discussion

The Karoo has a centuries-old history of sheep, goat and ostrich farming (Dean & Macdonald 1994, Shearing, 1997). Over the last few decades, however, many small stock farms have been converted to game farms (Snijders, 2012). Intense, small stock farming practices are thought to have contributed to the deterioration of southern Africa's rangelands through the loss of vegetation cover, erosion and in some areas, through bush encroachment (Belayneh & Tessema, 2017; Devine, 2017). The removal of small stock, together with the introduction of indigenous wild herbivores on the other hand, has been observed to contribute to higher productivity of the land (Du Toit & Cumming, 1999). This study is the first to address the change in vegetation type and productivity after converting from small stock farming to game farming in the Karoo. The location offers an interesting case study for investigating this change, not only because of its long history of land use, but also because of its location at the ecotone of Nama Karoo, Grassland and Albany Thicket biomes (Mucina & Rutherford, 2006). Although the absence of suitable control sites limits the generality of the study's findings, and does not allow for the establishment of causality between vegetation type and cover changes and the conversion from small stock to game farming, the results shed light on the dynamics of the vegetation types and productivity over time. They are also in accord with observations on vegetation change, including bush encroachment in other parts of semi-arid South Africa (O'Connor et al., 2014, Belayneh & Tessema 2017; Devine, 2017). Rainfall, fire and stocking numbers were all related to changes in vegetation cover, with rainfall having the strongest influence in the model. This study also shows that open source platforms such as Google Earth Engine provide a cost-effective means to use satellite imagery to classify vegetation types and cover and to monitor how they change over time (Johansen et al., 2015). However, field work is essential to identify inaccuracies in the output and to contextualise and ground-truth the changes observed in the satellite data.

5.1 Spatial and temporal changes in different vegetation types

5.1.1 Thicket increase at the expense of Shrubland and Grassland

As predicted, findings from this study suggest that Thicket vegetation has expanded into both Shrubland and Grassland over the study period. Field observations confirmed these transitions, and are in accord with previous findings on Thicket invasion in semi-arid Shrubland in South Africa (Puttick et al., 2014a, 2014b). Areas that were clearly identified as open Shrubland in the initial classification, especially in the valley bottom, are now dominated by Thicket taxa. Similarly, areas that were Grassland, especially along the river courses which drain the high mountains below the escarpment, now support Thicket communities. What is more difficult to explain, however, is the switch from Grassland to Thicket over relatively large areas between 1987-1989 and 2014-2016. Thicket communities are generally stable and are unlikely to change back to Grassland and Shrubland unless through major disturbances such as land clearing or sometimes extensive and frequent fire. This discrepancy, however, is best explained by problems associated with the supervised classification. The field survey revealed high densities of Renosterbos (*Elytropappus rhinocerotis*) in both north western and eastern Grassland sites, which coincide with areas where the supervised classifications indicated large changes from Thicket to Grassland over time. Given the similar spectral signals of dense Renosterbos and Thicket communities (dominated by *Vachellia karroo* and *Searsia lucida*), it is likely that parts of Grassland were misclassified as Thicket in the initial map of the distribution of vegetation types. This problem is clearly evident from the confusion matrices. In the 1987 confusion matrix 12% of Grassland was inaccurately assessed as Thicket, compared to only 5% in 2017. Mature Thicket communities are usually very stable and fire resistant and lack herbaceous cover, whereas Renosterbos, which can become dominant in Grassland vegetation (Levyns, 1926) burns relatively easily. It follows that Grassland areas which were misclassified as Thicket due to the presence of Renosterbos in 1987, were burnt in the fire in 2016, and were therefore classified as Grassland in 2017. Consequently, the analysis showed that Thicket had turned into Grassland in these areas which does not reflect the true sequence of events on the ground. This discrepancy also implies that the relative area designated as Thicket in both 1987 and 2017 is likely to be an over-estimation of the real values.

Despite the errors in classification, bush encroachment is still a matter of concern as it can often be negative for game farmers when the proportion of palatable shrub and grass species decreases in favour of more unpalatable shrubs and low trees (O'Connor et al., 2014; Belayneh & Tessema, 2017). The field survey showed that the most prevalent species to encroach into Grassland and Shrubland were *Searsia lucida* and *Vachellia karroo*, both with different implications for the reserve. An increase in the abundance and biomass of the unpalatable tall shrub, *Searsia lucida*, has obvious negative consequences for wildlife production since it competes with palatable shrub and grass species for water, nutrients and light. An increase in the nitrogen-fixing *Vachellia karroo* on the other hand, can be seen in a positive light for the reserve (as pointed out by Belayneh and Tessema (2017)) as it provides habitat and food for elephant (*Loxodonta africana*) and giraffe (*Giraffa caelopardalis*). Also, when it is knocked over it can give rise to an increase in palatable grasses that provide for a multitude of grazers (O'Connor et al., 2014). An increase in *Vachellia karroo*, especially in areas dominated by Shrubland can also be beneficial for the soil, due to increasing soil nitrogen and carbon sequestration (Belayneh & Tessema, 2017). An increase in *Vachellia karroo* is not a new phenomenon, in the region and has been recorded in both communal and commercial semi-arid rangelands in South Africa since at least the 1940s (Frost, 1999; Eckhardt et al, 2000; Puttick et al., 2014a, 2014b). However, an increase in *Searsia lucida*, which is an indigenous species is rarely mentioned in literature.

5.1.2 Increase in Bare-ground

Contradictory to our predictions, the relative area of Bare-ground increased over time at the expense of both Shrubland and Thicket vegetation. This is concerning for the overall productivity of the reserve and has implications for the type and scale of management interventions and restoration activities that are necessary on the reserve. Low levels of animal dung in these areas, as revealed from the field survey, suggest little utilization by wildlife, which in turn affects the proportion of the reserve that is suitable for wildlife. Bare-ground appears to have expanded primarily on low-lying areas and often at locations where cultivation occurred historically. These results are in accord with findings by Kakembo and Rowntree (2003), who reported severe erosion as a consequence of cultivation and its subsequent abandonment in the communal areas of the semi-arid Eastern Cape. It is not clear to what extent the high stocking numbers of browsers such as black wildebeest (*Connochaetes gnou*) and nyala (*Tragelaphus angalii*), are responsible for the expansion of Bare-ground. It is possible, however, that they could impose high levels of browsing pressure on these low-lying areas, thus exacerbating the spread of erosion.

5.1.3 Change in Cultivated land

Surprisingly, the results show an increase in Cultivated land in the high Grassland areas of the reserve. In addition to problems of accuracy in relation to the classification of Thicket vegetation, another point of uncertainty arose in the classification of Cultivated land. Pixels that indicated the presence of Cultivated land within a matrix of Grassland vegetation on the north eastern and south western corners of the reserve both in 1987 and in 2017 have been incorrectly assigned to this vegetation class. Field observations confirmed that areas classified as Cultivated land near the houses on the valley bottom were correctly assigned. However, those areas on the higher slopes that were classified as Cultivated land were not correct. The discrepancy can be explained by the almost identical spectral signals emanating from both Grassland and Cultivated land in some instances. The increase in productivity in Grassland, especially in early post-fire environments, adds to the difficulty of being able to differentiate Grassland from the highly productive Cultivated land vegetation type. Both are comprised of grasses which are not easily separated under certain circumstances.

5.2 Vegetation cover and drivers of change

5.2.1 Overall changes in SAVI

The field survey supported our prediction in that SAVI accurately reflects the vegetation cover in the reserve. This suggests that SAVI can be confidently used to assess changes in vegetation cover in the reserve as part of, for example, an ongoing monitoring programme for the reserve. The results also confirmed the reserve manager's anecdotal observations that woody plant cover, and therefore productivity, had increased over time in each vegetation type.

Although the long history of land-use in the area is likely to have shaped the reserve's vegetation today, as shown by studies in similar environments (Puttick et al., 2014a, 2014b), the analysis suggested that rainfall was the primary driver of the changes recorded in vegetation cover over the last 30 years. Various studies show the link between vegetation indices, rainfall and primary productivity (Tucker et al., 1991; Nicholson, 1998; Martiny et al., 2006; Palmer et al., 2017) in semi-arid environments. Our findings are in accord with many other studies in semi-arid rangelands which show that abiotic drivers such as rainfall influence vegetation dynamics more strongly than biotic drivers such as grazing (Fynn & O'Connor, 2000; Sullivan & Rohde, 2002; Vetter, 2005). This observation has been used by some authors as support for a non-equilibrium view of arid and semi-arid rangelands dynamics (e.g. Behnke et al., 1993; Sullivan & Rohde, 2002). However, as pointed by Fynn and O'Connor (2000), even though rainfall might have a dominant influence on vegetation productivity, biotic and abiotic factors are likely to work in conjunction with each other in shaping the vegetation of semi-arid rangelands. Therefore, the impact of grazing and browsing on the vegetation of the reserve should not be underestimated. Although this study did not specifically address the relative impacts of different game species on the different vegetation types, specific impacts of certain species such as blue wildebeest and elephant are discussed in the context of changes in vegetation cover.

Areas at the bottom of the valley and west of the Melk River were heavily utilized historically by ostrich farming and for lucerne cultivation, and were consequently severely degraded (Shearing, 1997). As a result of the restoration activities undertaken in this area by management following the introduction of game in the reserve, these areas exhibited a large increase in SAVI values over the study period. Our findings are supported by those from Van den Berg and Kellner (2005) and Snyman (2003), who studied the effect of various restoration methods in semi-arid Karoo, and concluded that restoration can have significant positive impacts on vegetation productivity.

While the long-term trend in Grassland productivity as measured by SAVI was positive, it was lower than predicted from previous studies that dealt with the impact of eliminating livestock and reducing grazing pressure on the veld (e.g. Seymour et al., 2010; Ward & Elser, 2011; Kambataku et al., 2013). In some areas on the reserve the trend in Grassland productivity was negative over the study period. The most negative trends in Grassland productivity were found on the montane plateaus, where grazing antelopes such as blue wildebeest (*Connochaetes taurinus*) have been observed to spend most of their time foraging. It seems that blue wildebeest utilize these areas more than other Grassland areas due to the convenience that flat ground brings for foraging. The reserve is stocked above recommended rates, which can elicit high grazing/ browsing pressure on the landscape. Maintaining animal numbers within the recommended values is especially important for those species which are known to focus on particular patches of palatable plants in a landscape such as blue wildebeest (Codron & Brink, 2007). Such patch-selective grazing might explain the negative trends in SAVI on the montane plateaus.

Although the eastern Grassland areas in general showed an increase in SAVI over time, after closer inspection the changes occurred may entail a negative impact at the scale of the whole reserve. It is likely that the increase in productivity in these areas was due to subsequent encroachment of Renosterbos (*Elytropappus rhinocerotis*) in these areas. Although the increase of Renosterbos (facilitated by lack of fire) in *Danthonia disticha* Grassland may be attributed with an increase in productivity in the landscape, it may entail decrease in occupancy by grazing animals due to decrease in palatable grass cover. In other words, despite the seemingly positive impact on vegetation productivity, the increase in Renosterbos might be accompanied by negative impact on the grazing animals due to reduced forage.

5.2.2 Decrease in SAVI in Riverine thicket

Although overall, and as predicted, SAVI increased in Riverine thicket over time, parts of this vegetation type along the banks of the Melk River exhibited a significant decrease in SAVI. Conversations with the reserve manager and observations of dung abundance indicated that the elephants in the reserve spend most of their time in Riverine thicket areas. This is the best explanation for the decline in SAVI in parts of the Riverine thicket and is supported by Spinage (2012) and Landman et al. (2014) who recorded the prevalence of elephant damage proportional to the distance to water sources. Elephant damage in the reserve is concerning because of its wider impacts of the ecosystem. Elephants have been recorded to increase run-off zones where soil nutrients are lost. Furthermore, open patches created by elephants have been shown to reduce browse availability for other species (Kerley & Landman, 2006). Fornara and du Toit (2008) explained that elephants can have major impacts on trees and therefore control bush encroachment, suggesting a beneficial role for elephant damage. The removal of elephants on the other hand can promote bush encroachment. Interestingly, concentrations of similar damage found in the vicinity of Melk River elsewhere in the reserve were not detected from the SAVI analysis. This is despite the obvious signs of elephant impact recorded over parts of the reserve during the field surveys. Their impact on individual trees such as *Celtis africana*, *Rhus lancea* and *Euclea undulata*, located several kilometres from the Melk River was clearly evident. A possible explanation for why only localised evidence of elephant impact was recorded from the satellite imagery is that Riverine areas and areas close to the ephemeral streams exhibit a higher SAVI signal relative to other areas, therefore possibly masking elephant damage detectable from SAVI in this vegetation type. The concentrations of decreased SAVI in the low-lying, western part of the reserve, interspersed with relatively high increases in SAVI, do not reflect elephant damage, but rather the presence of newly built dams that elicit weak SAVI signals.

5.3 Limitations

The lack of fine-scale temporal and spatial data for rainfall and wildlife numbers limited the extent to which an understanding of the dynamics of vegetation productivity in the reserve was possible. The spatial resolution of rainfall data was generally poor due to the reliance on a single weather station only. With multiple measuring points, rainfall could be more accurately modelled for the entire reserve based on its relationship to elevation. More spatially-explicit rainfall data would, for instance, allow for the exploration of microclimate effects on vegetation productivity. Similarly, a more comprehensive and spatially-explicit data set of animal numbers on the reserve would help with an assessment of the relative impacts of different species on the different vegetation types, plant functional groups and species. This would, in turn, allow for greater determination of the most appropriate stocking rate for the reserve. With the help of high resolution data, future studies on the reserve could also address the combined effect of the relationship between different feeding types (grazers, browsers and mix-feeders) and rainfall on the vegetation. Different browsers, grazers and mix feeders may interact in different ways. Different combinations of these species may have varying impacts on the vegetation cover and productivity in different vegetation types. Therefore, it is an important next step to compare the impacts of the combination of different animals on the different vegetation types. As part of this next step it is useful to derive Carrying capacities for all different game species and compare them to the current stocking numbers in the context of vegetation cover changes in the different vegetation types.

5.4 Management implications

From our analysis, we found no indication that rangeland condition on Asante Sana had deteriorated over time relative to the period when domestic livestock utilised the land. This is despite the relatively high stocking numbers that have been employed in the reserve since the adoption of game farming. Neither the field survey nor the satellite based analyses which documented changes in both vegetation types and productivity over time suggested that the area was more degraded in 2017 than it was before livestock were replaced by wild ungulates. However, this is not to suggest that there are no issues of concern. Several indicators highlight issues that could be addressed by the management team responsible for the sustainable utilisation of the resource on the reserve. Although the overall increase in productivity occurred on the reserve level, most points of concern for the long-term sustainability of wildlife production in the reserve were at the level of individual vegetation types. These points of concern, include the increase in Thicket communities at the expense of Grassland and Shrubland communities, the overall increase in Bare-ground and the decline in vegetation productivity in some parts of the Grassland and Riverine thicket vegetation types, often with wider indirect impacts at the reserve level (Table 9).

Increase in Thicket communities, especially of *Searsia lucida*, can have major negative impacts on the carrying capacity of the land at the reserve level, thereby affecting the number of animals that can be sustainably kept in the reserve. Smit (2004) outlined how to effectively restore areas affected by bush encroachment, and explained that thinning rather than clearing should be practiced in these areas. Thinning should focus on the balance between removing the negative effects of competition that trees pose to the herbaceous layer, and maintaining the potential benefits they bring through, for example, improved soil quality. Preservation of mature trees is important, because they help prevent seedling recruitment and improve nutrient availability for herbaceous species (Smit, 2004). The most effective results are furthermore achieved through long-term commitment rather than once-off restoration efforts (Smit, 2004).

An increase in Bare-ground can have significant adverse effects on wildlife production in the whole reserve, and therefore its restoration should be prioritised. Van den Berg and Kellner (2005) outlined effective ways to restore bare patches in semi-arid areas in South Africa. Best measures outlined include covering the affected area with organic material such as cow and horse manure to increase soil's carbon content, aeration and water retention capacity as well as ripping of the surface of the bare patches to improve water infiltration and root moisture as well as seedling recruitment. Lastly the authors suggested over-sowing the patches with indigenous grass or shrub species. A combination of all methods outlined above were shown to be the most effective treatment compared to any treatment on its own (Van den Berg & Kellner, 2005). In addition to restoration, management could also employ preventative measures such as fencing off erosion prone patches and limiting the population densities of target browsers, such as black wildebeest (*Connochaetes gnou*) and nyala (*Tragelaphus angalii*).

Decreases in productivity in parts of Grassland and Riverine thicket are also areas of concern. Despite the overall increase in Grassland productivity, the conspicuous decrease in productivity on the high montane plateaus is concerning at the reserve level, and should be addressed to prevent further degradation in those areas. Fence-line experiments have demonstrated the positive impact of removing grazing animals on Grassland productivity from an area (Todd & Hoffman, 1999, 2009). The best way to reverse the negative trend in these areas is probably reduce the population numbers of target grazer species (e.g. blue wildebeest) although preventing patch selective grazing is difficult.

The reduced productivity in Riverine thicket due to elephant damage was significant, although the long-term implications of this are not clear. While some damage from elephants should be anticipated, the development of a management plan would help to anticipate further impacts from the megaherbivores on the reserve. As explained by Owen-Smith et al. (2006), no easy solution exists for elephant management. A combination of different strategies should be undertaken and tested with adaptive management. Population management through contraceptives and zoning of areas as “elephant sanctuaries” and “tree sanctuaries” can help alleviate pressure on target areas (Owen-Smith et al., 2006).

In order to better understand the long-term dynamics of vegetation and its impacts on wildlife production, a well-thought long-term monitoring program should be established in the reserve. There is increasing literature on the importance of long-term ecological monitoring in wildlife resource management (Lindenmayer & Likens, 2010; Lindenmayer et al., 2012a, 2012b). Benefits of long-term monitoring include improved understanding of complex ecological phenomena that occur over long periods of time and obtaining data to support evidence based management of ecosystems (Lindenmayer et al., 2012a). Such monitoring is important for Asante Sana, so that effective and science-based management action (e.g. the determination of stocking rates, and supplemental feeding) can be planned with confidence. Methods used in this study act as a basis for long-term monitoring of vegetation types and productivity in the reserve. Satellite-based monitoring should be accompanied by annual, field-based observations and measurements perhaps accompanied by a photo-monitoring survey.

Table 9. Summary of proposed management activities for bush encroachment, increase in Bare-ground, reduction in vegetation productivity in Grassland and Riverine thicket and overall long-term changes in Asante Sana Game Reserve.

Point of concern	Implications	Proposed restoration or activity	Support for the restoration or activity	Reference
Bush encroachment	Reduction in grass and shrub cover is negative for wildlife production	Thinning of targeted species	Thinning of target species rather than clearing is shown to help restore affected areas. Activities should be repeated long-term rather than doing it once off	Smit (2004).
Increase in Bare-ground	Increase in Bare-ground and erosion is negative for vegetation cover which influences wildlife production negatively	Organic material, ripping, over-sowing. Fencing, reduction of stocking rates of target browsers	Various measures have been shown to improve the productivity of the land	Van den Berg and Kellner (2005).
Decrease in Grassland productivity	Decrease in Grass productivity is negative for wildlife production of those species that rely on Grass	Fencing, reduction of stocking rates of target grazers	Reduction/elimination of grazers from a plot has been widely demonstrated to improve the productivity of Grassland throughout southern Africa	O'Connor and Roux (1995), Todd and Hoffman (1999, 2009) and Seymour et al. (2010).
Decrease in Riverine thicket productivity	Reduction of productivity in Riverine thicket has negative impacts on the landscape making the soil more susceptible to erosion	Elephant population control through contraceptives. Determining areas for "elephant sanctuaries" and "tree sanctuaries"	Elephants have been shown to affect Thicket communities significantly, but there seems to be no easy one-size fits all elephant management strategy. Strategies are case dependent and therefore should focus on principles of adaptive management	Fornara and du Toit (2008), Owen-Smith et al. (2006).
Long-term changes in the reserve	Because of land use practices and changing environmental conditions the state of the vegetation is susceptible to changes in the reserve	Establishing long-term monitoring program for vegetation type and productivity as well as rainfall and stocking rates	Long-term ecological monitoring programs have been shown to be beneficial for long-term sustainability of game reserves	Lindemayer and Likens (2010), Lindemayer et al. (2012a, 2012b).

6. Conclusions

Vegetation change resulting from converting from livestock to game farming has been little studied in ecological context. This study monitored the spatial and temporal changes in vegetation type and cover in Asante Sana Game Reserve in the Eastern Cape, South Africa, and found an overall increase in vegetation productivity. Some concerns associated with the changes in the reserve arose, including trends in land degradation, bush encroachment and decreased productivity in areas of Grassland and in Riverine thicket, the latter as a result of elephant damage. Although fire and stocking densities seem to have an impact on SAVI values, the overall increase in SAVI can be mainly attributed to an increase in rainfall over the study period. Although these results should be approached with caution due to the lack of suitable control sites, they suggest that rainfall may have an overriding effect on the vegetation compared to stocking numbers in this individual reserve, therefore supporting the non-equilibrium hypothesis for semi-arid rangelands. The management can, howerver, undertake restoration actions to reduce the impact of bush encroachment and erosion. Furthermore, the fencing off of ecological sensitive areas can be an effective tool for management. For the long-term sustainability of the reserve, an ongoing ecological monitoring programme should be established for the reserve.

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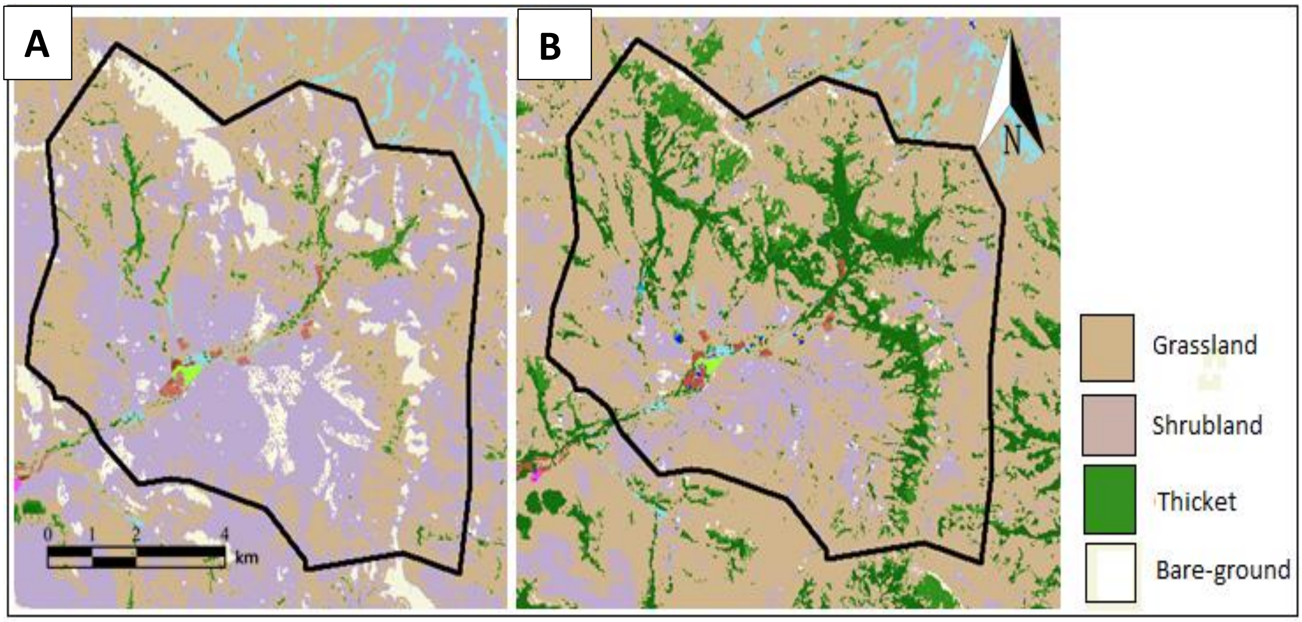
Appendix A

Results from generalised linear mixed models showing the effect of rainfall, stocking numbers and burning on the change in SAVI between 1989 and 2016 on the different vegetation types and other areas of interest. All predictor variables were scaled and centred and the 95% confidence intervals profiled to determine the significance of each parameter. Coefficients that are indicated in bold have 95% confidence intervals that do not overlap zero and therefore represent a significant effect.

Vegetation type/ area of interest	Variable	Coefficient	SE
Reserve as a whole	Rain	**0.040**	0.005
	Tot.LSU	**-0.002**	0.006
	Burned	**0.003**	0.06
Bare ground	Rain	**-0.009**	0.003
	Tot.LSU	**-0.009**	0.003
Shrubland	Rain	**0.035**	0.012
	Tot.LSU	**-0.013**	0.013
Grassland	Rain	**0.030**	0.010
	Tot.LSU	**-0.012**	0.011
Cultivated land	Rain	**0.036**	0.034
	Tot.LSU	**-0.012**	0.035
Thicket	Rain	**0.033**	0.011
	Tot.LSU	**-0.014**	0.012
From Grassland to Thicket	Rain	**0.044**	0.021
	Tot. LSU	-0.003	0.020
From Thicket to Grassland	Rain	**0.040**	0.019
	Tot. LSU	**-0.004**	0.018
From Shrubland to Thicket	Rain	**0.046**	0.030
	Tot. LSU	-0.003	0.030
From Thicket to Shrubland	Rain	**0.04**	0.021
	Tot. LSU	**-0.009**	0.021
From Shrubland to Bare ground	Rain	**0.024**	0.047
	Tot. LSU	0.001	0.046
From Shrubland to Grassland	Rain	**0.050**	0.060
	Tot. LSU	0.005	0.060
From Cultivated land to Bare ground	Rain	**0.036**	0.060
	Tot. LSU	0.003	0.060
From Cultivated land to Thicket	Rain	**0.053**	0.066
	Tot. LSU	-0.005	0.065

Appendix B

Land cover data from GeoTerraImage (2015), showing the increase in Thicket in Asante Sana Game Reserve between A) 1990 and B) 2013.



Appendix C

The feeding type, Metabolic biomass expressed as **mass$^{0.75}$**, Large Stock Unit (LSU) expressed as Metabolic biomass of the species per Metabolic biomass of an adult Cow of 450 kg, Grazer Unit (GU) expressed as Metabolic biomass of the species per Metabolic biomass of an adult Blue Wildebeest of 180 kg and Browser Unit (BU) expressed as Metabolic biomass of the species per Metabolic biomass of an adult Kudu of 140 kg for all the different game species in Asante Sana Game Reserve. Grazers are expressed in LSU and GU, Browsers in LSU and BU while Mix feeders in all LSU, GU and BU.

Species	Feeding type	Metabolic biomass	Large Stock Unit	Grazer Unit	Browser Unit
Black Wildebeest	Grazer	39.20	0.40	0.80	
Blesbuck	Grazer	19.60	0.20	0.40	
Blue Wildebeest	Grazer	49.14	0.50	1.00	
Bufallo	Grazer	97.70	1.00	1.99	
Eland	Mix feeder	79.20	0.81	1.61	1.95
Elephant	Mix feeder	267.70	2.74	5.45	6.58
Gemsbuck	Grazer	42.90	0.44	0.87	
Giraffe	Browser	143.30	1.47		3.52
Grey Duiker	Mix feeder	7.60	0.08	0.15	0.19
Grey Rhebok	Grazer	7.60	0.08	0.15	
Impala	Mix feeder	17.40	0.18	0.35	0.43
Klipspringer	Browser	5.60	0.06		0.14
Kudu	Browser	40.70	0.42		1.00
Lechewe	Grazer	24.70	0.25	0.50	
Mountain Reedbuck	Grazer	11.20	0.11	0.23	
Nyala	Mix feeder	25.00	0.26	0.51	0.61
Ostrich	Mix feeder	31.60	0.32	0.64	0.78
Reedbuck	Grazer	15.90	0.16	0.32	
Red Hartebeest	Grazer	37.40	0.38	0.76	
Sable	Grazer	50.20	0.51	1.02	
Springbuck	Mix feeder	11.50	0.12	0.23	0.28

Steenbuck	Browser	4.80	0.05		0.12
Warthog	Mix feeders	17.40	0.18		
Waterbuck	Grazer	45.00	0.46	0.92	
White Rhino	Grazer	241.00	2.47	4.90	
Zebra	Grazer	53.20	0.54	1.08	

Appendix D

Aerial counts for the different game species in Asante Sana Game Reserve for 2007, 2009, 2010, 2013, 2015 and 2017. Aerial counts are expressed in animal numbers.

Species	2007	2009	2010	2013	2015	2017
Black Wildebeest	192	100	64	111	117	92
Blesbuck	127	150	77	116	168	128
Blue Wildebeest	285	220	85	89	206	208
Bufallo	35	16	16	15	22	39
Eland	575	500	246	346	416	449
Elephant		9		16	18	21
Gemsbuck	91	150	52	87	153	90
Giraffe	34	40	34	34	41	42
Grey Duiker		250				
Grey Rhebok		20				5
Impala	192	250	50	116	144	140
Klipspringer		30		6		
Kudu	695	600	116	237	483	296
Lechewe	25	30	30	49	33	39
Mountain Reedbuck	82	100	23			25
Nyala		200		16		
Ostrich				52		
Reedbuck				25		
Red Hartebeest	164	120	64	80	121	128
Sable	13	19	15	4	6	6
Springbuck	76	100	50	98	52	36
Steenbuck		100				
Warthog	297	250	60	170	114	107
Waterbuck	61	80	36	107	187	183
White Rhino	16	13	15	13	17	13
Zebra	148	150	46	86	120	110
Total	3108	3497	1079	1873	1460	2118