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INVESTIGATING THE EFFECTS OF GRASSLAND MANAGEMENT TECHNIQUES ON VEGETATION AND WILDLIFE AT LEWA WILDLIFE CONSERVANCY, KENYA

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A dissertation submitted in partial fulfilment of the requirements for the degree of M.Res. Wildlife Conservation.

As the nominated University supervisor of this M.Res. project by Rebecca Sargent, I confirm that I have had the opportunity to comment on earlier drafts of the report prior to submission of the dissertation for consideration of the award of M.Res. Wildlife Conservation.

Signed.....…….....

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As the nominated Marwell Wildlife Supervisor of this M.Res. project by Rebecca Sargent, I confirm that I have had the opportunity to comment on earlier drafts of the report prior to submission of the dissertation for consideration of the award of M.Res. Wildlife Conservation.

Signed.....…….....

MW Supervisor's name: Dr Zeke Davidson

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Abstract

Managing ecosystems for the benefit of multiple stakeholders is one of the greatest challenges of modern conservation. In the world's rangelands, several methods are being developed to promote coexistence between wildlife, livestock and human communities. Using cattle as a way to manage grassland has become an increasingly popular technique to assist with both conservation and community development. Lewa Wildlife Conservancy (LWC) is a protected area in north-central Kenya that is attempting to use the principles of holistic grazing management, an intensive cattle grazing technique, to improve the quality of the grassland. This study aimed to investigate the difference between cattle grazing and an alternative strategy of mowing, to determine whether they were meeting the desired goals of improving the quality of the grassland and providing a diversity of habitats for wildlife. Results of vegetation surveys indicated that, initially, cattle grazing did reduce grass biomass and increase the amount of green vegetation. However, in less than one year biomass had returned to its original level and within two years the grassland was indistinguishable from control zones. Mowing was a more successful strategy resulting in long term change, with biomass remaining low over two years and showing some increases in grass species diversity. Methods to study wildlife use, including camera traps and dung transects, appeared to indicate some preferences of particular species for specific treatment areas. However, analysis of behaviour did not show that they were grazing on these plots preferentially; therefore, use of these areas may not be related to feeding preference for the vegetation following management interventions. This research found no clear evidence that grassland management was directly influencing habitat use by wildlife. However, the management techniques did alter the vegetation structure and could be used to create a diversity of habitats. Cattle grazing as a conservation strategy may not be appropriate in this landscape as grazing would need to be very frequent in order to maintain changes in the grassland. Nevertheless it may still be worthwhile on a small scale to assist with community outreach.

Target journal: Rangeland Ecology and Management

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Abbreviations

LWC	Lewa Wildlife Conservancy
LRD	Lewa Research Department
C	Control plots
G	Grazed plots
M	Mowed plots
G+M	Grazed and mowed plots (mowing after grazing)
PM	*Pennisetum mezianum*
PS	*Pennisetum stramineum*
S.E.	Standard error
ZSL-CTAT	Zoological Society of London – Camera trap analysis tool

1. Introduction

1.1 Background

In the last 50 years, humans have severely altered the world's ecosystems, with 20-70% of the area of 11 of Earth's 13 terrestrial biomes converted for human use (Rodríguez *et al.* 2011). Ecosystems are regarded as the natural living capital on which humans and other species depend for survival and well-being; providing services such as nutrient cycling, water regulation and provision of raw materials (M.E.Assessment 2005). Yet approximately 60% of these services are being degraded or used unsustainably (M.E.Assessment 2005). Anthropogenic factors threatening the structure and function of ecosystems include: habitat loss and fragmentation, overexploitation, climate change and invasive species (Ashraf *et al.* 2012). In recent years there has been an increasing recognition of the need for conservation interventions to protect and regenerate ecosystems of ecological and economic importance.

Rangelands in particular are of key concern for conservationists given that they support the needs of a variety of different stakeholders and there are many challenges associated with sustainable use (Williams *et al.* 1968). Typically, rangelands are defined as areas in which livestock production is practised by animals feeding on wild vegetation and the majority are classified as grassland or savannah (du Toit *et al.* 2010). There is a severe need to manage these ecosystems effectively to conserve their function for both the wildlife and pastoralists that rely on them. However, competition and conflict between the needs of wildlife, humans and livestock makes this a complex process (Lamers *et al.* 2014).

Increasingly, methods are being developed using livestock as a tool for conservation. Utilising a domestic grazing species can be hugely beneficial for wildlife and evidence has proven its effectiveness in improving plant species richness (Rupprecht *et al.* 2016), removing biomass of dominant vegetation (Kurtz *et al.* 2016), creating structural diversity and assisting with nutrient cycling (Wallis De Vries *et al.* 1998; Ngatia *et al.* 2015). Using livestock to assist with conservation is a potentially valuable way of marrying the needs of wildlife with those of local people. Management of semi-arid ecosystems is undergoing constant re-evaluation (Young *et al.* 1997); but typically these techniques are not intensive and require careful management to prevent overgrazing.

1.2 African rangelands

In the rangelands of Africa, overgrazing is becoming an increasing problem due to rapid growth in human population leading to increased livestock numbers and a deterioration of nomadic lifestyles (du Toit *et al.* 2010; Fynn *et al.* 2016). While historically pastoralists have maintained sustainable relationships with their environment, these societies have had to adapt their way of life and resource management practices to a new and changing environment (Kuriyan 2002). Overstocking and limited ranging typically result in landscape degradation, such as increased soil erosion and reduced vegetation cover. This has negative impacts on livestock, wildlife and human welfare in terms of food and livelihood insecurity (Rubenstein 2010; van Oudenhoven *et al*. 2015). It is therefore important to develop rangeland management strategies that are sustainable and promote tolerance and cooperation between stakeholders with conflicting interests; such as pastoralists and wildlife conservationists.

In Kenya, where this study is based, arid and semi-arid rangelands cover approximately 80% of the country and support one third of the human population and 70% of its livestock (Shilling *et al.* 2012). Understanding the effects of different livestock management techniques is crucial in order to prevent landscape degradation (Young *et al.* 1997).

1.3 Holistic grazing

One method which seeks to use livestock to maintain grassland is the controversial technique of holistic grazing management. This is a somewhat unusual and intensive form of livestock grazing focussed on preventing and reversing desertification: the erosion of soil and steady loss of biomass and diversity of vegetation (Savory 1989). Previously desertification was believed to have been driven by overgrazing (Kiage 2013). However, removal of livestock from overgrazed lands does not always result in recovery of the ecosystem and has been shown in some instances to cause further degradation (Savory 1991). The Holistic Resource Management framework is based around the principle that grazers are in fact necessary to prevent desertification.

Historically, many of the world's grasslands evolved alongside huge herds of animals grazing and trampling the earth before migrating to other areas (Savory 1989). Overgrazing is therefore likely to be linked to movement and time, rather than number of animals (Voisin 1961). Savory proposes using large herds of livestock as a proxy to

mimic former natural herds (Savory 1991). This kind of intensive grazing strategy is controversial and highly debated in the literature. Several papers claim that this method offers no additional benefits over less intensive continuous grazing strategies (Joseph *et al.* 2002; Carter *et al.* 2014). However, there are also examples of success where aspects such as carbon sequestration and vegetation quality have improved through holistic management (Jacobo *et al.* 2006; Teague *et al*. 2011). It is clear that the effects of livestock grazing strategies are not uniform across ecosystems and monitoring and adaptive management are necessary for success. Holistic management is an idea that may be of particular relevance to African grasslands and savannahs which have historically been grazed by large numbers of wild herbivores. Removal or reduction of these herbivores, for example in protected areas with low animal density, may therefore cause declines in the diversity of vegetation and lead to unproductive ecosystems (Bakker *et al*. 2006).

1.4 Ecology of wild grazers

Variation in grass height, composition and chemical attributes will determine the abundance and distribution of large mammalian herbivores (Hopcraft *et al.* 2012). Herbivore species have diverse preferences relating to the structure and composition of grassland communities. Grassland management techniques may therefore affect wildlife communities in complex and varied ways.

Typically it is assumed that creating shorter grass patches provides higher quality forage due to removal of above ground biomass stimulating regrowth of young, nutritious vegetation (Verweij *et al.* 2006). Taller grass patches, while being less palatable, are valuable as a reserve for dry seasons and droughts (Fynn *et al.* 2016). Yet not all herbivores preferentially graze on shorter grass. Other factors also influence habitat preferences such as predation risk, body size, physiology and feeding strategy. For example, while Thompson's gazelle (*Eudorcas thomsonii*) prefer short open grassland where predators are more easily visible, greater kudu (*Tragelaphus strepsiceros*) use a strategy of concealment in long grass (Fynn *et al.* 2016).

Species foraging strategies are also partitioned by body size and digestive system. Smaller bodied herbivores such as Grant's gazelle (*Nanger granti*) and impala (*Aepyceros melampus*) may be constrained to higher quality forage due to their higher metabolic rate; whereas larger bodied species can be less selective (Hopcraft *et al*. 2010). In addition, hindgut fermentation allows for lower forage requirements but

higher absolute food requirements (Sensenig *et al.* 2010). Species such as elephant (*Loxodonta africana*) and zebra (*Equus quagga*) can therefore be less selective but need to spend a larger proportion of their time eating. Conversely, ruminants such as Grant's gazelle and oryx (*Oryx beisa*) will typically avoid tall, tough grass and feed in more nutritious areas (Schuette *et al.* 2016).

This combination of factors affecting feeding behaviour means that different species will be affected by grassland management and livestock grazing in different ways. If livestock grazing is creating shorter more nutritious grass lawns it may be expected that smaller-bodied ruminants such as gazelle will prefer these areas, while other species such as zebra might be less selective in their grazing areas.

1.5 Lewa Wildlife Conservancy

Lewa Wildlife Conservancy (LWC) in Kenya is one site which is attempting to use the principles of holistic resource management to improve its grassland. Originally managed as a cattle ranch, Lewa converted to a wildlife conservancy in the early 1990s and has since been run purely for the protection of wildlife. Initially, after conversion to a conservancy, grazing pressure dramatically declined following the removal of livestock (Giesen *et al.* 2007). However, wildlife numbers have significantly increased since that time and there is a concern that the grassland will not support these numbers.

In an attempt to improve the forage availability for wildlife, in 2014, monitoring and reviews of LWC's habitats allowed for the development of the Lewa Grassland Management Plan. The goal of this plan is to diversify LWC's habitats by improving the composition of vegetation communities and decreasing the build-up of dominant increaser grasses such as *Pennisetum stramineum* and *Pennisetum mezianum* (Schulz *et al.* 2014). These species, while valuable fodder when still growing, are unpalatable when mature and increase in yield when lightly grazed; forming large stands of long dead grass (Ibrahim & Kabuye 1987; Odadi *et al.* 2011). LWC's grassland is therefore considered to be nutritionally poor (Giesen *et al.* 2007).

To meet the objectives of the Grassland Management Plan, LWC uses the principles of holistic management by allowing community cattle to graze on unproductive grass. This extends economic benefits to local communities whilst improving the quality of the grassland (Schulz *et al.* 2014), thereby promoting a strategy of coexistence between wildlife and pastoralists. The method used is 'strip grazing' which involves confining cattle within a small area for a given number of days based on the Animal Days per

Acre (ADAs): a calculation of the patch size needed to feed one mature cow for one day (Butterfield *et al.* 2006).

In addition to livestock grazing, LWC uses mowing in some areas of the conservancy. This is done unsystematically to reduce the biomass of grass at specific sites using a tractor. Mowing is thought to produce similar results to those of cattle grazing yet it is more expensive, making livestock grazing the preferred technique throughout the conservancy (Schulz *et al.* 2014).

1.6 Rationale for current study

Several aspects of LWC's management warrant investigation. Careful monitoring of stocking rates and the effects of livestock grazing on forage quality and availability are basic to rangeland management. A specific study focussing on the different impacts of mowing and livestock grazing will provide vital information for future management of the conservancy. Furthermore, to assess whether the Grassland Management Plan is meeting its aim of providing a diversity of habitats for wildlife, there is a need to focus on wildlife utilisation of managed areas. This will help to determine whether the management practises are in fact beneficial for LWC's wildlife.

It is essential that management interventions are undertaken cautiously and with constant monitoring to determine if techniques are having the desired effects. This allows for continual evaluation and adaptive management of this sensitive ecosystem. This study will provide a detailed and rigorous analysis of LWC's management techniques and provide evidence for which strategies are most effective at meeting the goals of the Grassland Management Plan. This can inform future management practises not only within LWC, but also potentially in the wider area where the future of Kenya's wildlife depends on sustainable coexistence with pastoralist communities and their livestock.

1.7 Objectives

Objective 1: To investigate the effects of mowing and livestock grazing on vegetation quantity and quality.

Hypotheses:

- Mowing reduces biomass to a greater extent than livestock grazing.
- Mowed areas produce higher species diversity than grazed areas.
- Mowed areas have lower vegetation cover than grazed areas.

- Mowing results in a higher leaf:stem ratio than grazing.

Objective 2: To examine how grassland management techniques affect habitat use by wildlife.

Hypotheses:

- Mowed sites have a different herbivore species composition present on them when compared to grazed sites.
- Small-bodied herbivores show a preference for mowed sites over grazed sites.
- Ruminants show a preference for mowed sites over grazed sites.

The goal of this research was to help inform LWC's grassland management and expand on current understanding of their livestock grazing regime to develop best practise guidelines for future management.

2. Methodology

2.1 Study site

Fieldwork was conducted at Lewa Wildlife Conservancy (LWC), a region comprising 250 km^2 in the Isiolo district of north-central Kenya (0°11'36.03"N, 37°27'4.22"E) (Fig.1.1). LWC has a semi-arid climate with rainfall following a bimodal distribution pattern and rainy seasons occurring between March-May and October-December (Botha 1999). The average annual rainfall on the conservancy is 513 mm (Chege & Kisio 2008). Fieldwork took place over 10 weeks; beginning in the dry season in March 2016 and ending in May 2016 during the rainy season.

LWC is located in the northern foothills of Mount Kenya and extends from these lower slopes to the flatter grasslands of the north. Most of the area of LWC can be described as savannah grassland with tree and shrub cover of more than 2% but less than 20% (Giesen *et al*. 2007) (Fig.1.2).

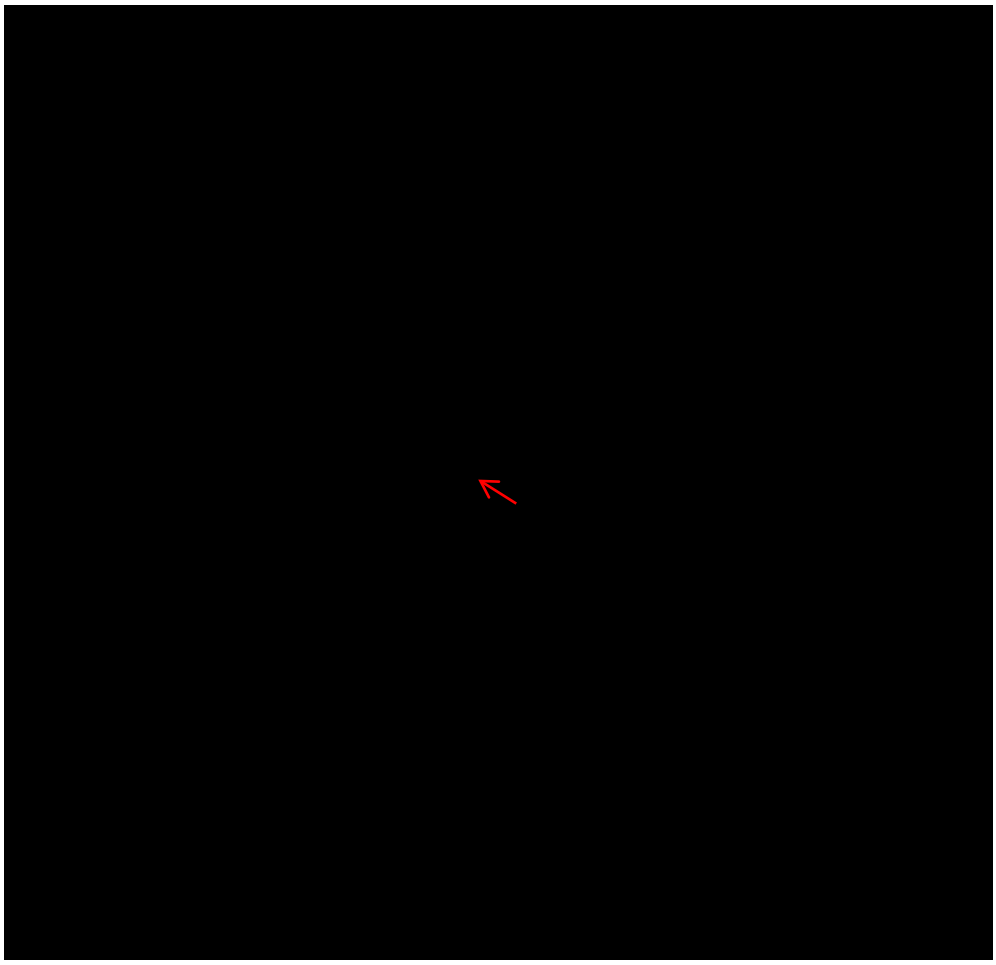

Figure 1.1. Location of LWC in Kenya

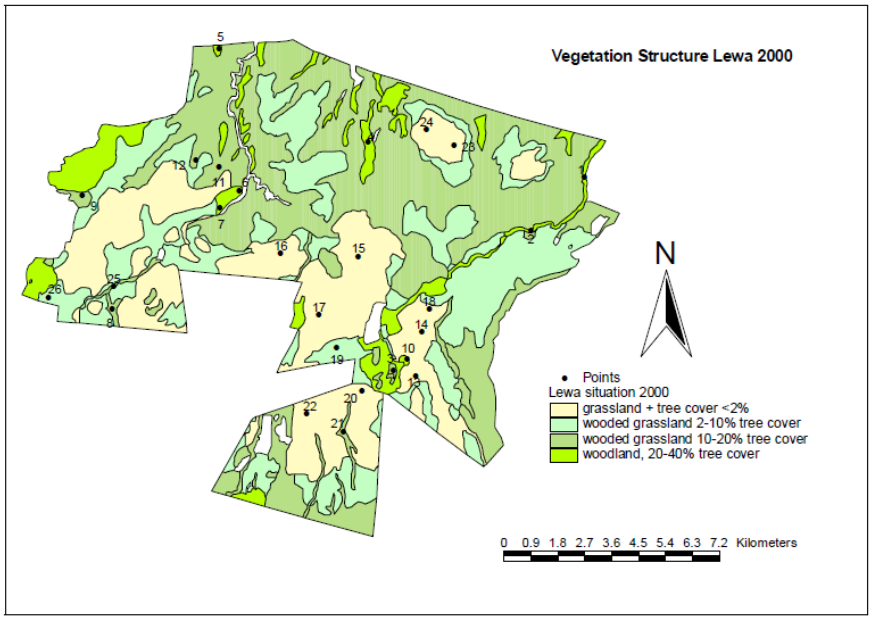

Figure 1.2. Vegetation structure on LWC as surveyed in 2000 (*from* Giesen *et al.* 2007).

LWC is divided in to 100 grazing blocks (Fig.2.1). Blocks with biomass >5000 kg/ha are considered for treatment. Cattle are grazed in herds of approximately 600 in order for the intensive holistic management strategy to be successful. Mowing is conducted unsystematically on LWC to reduce biomass of grass at random sites (Schulz *et al.* 2014).

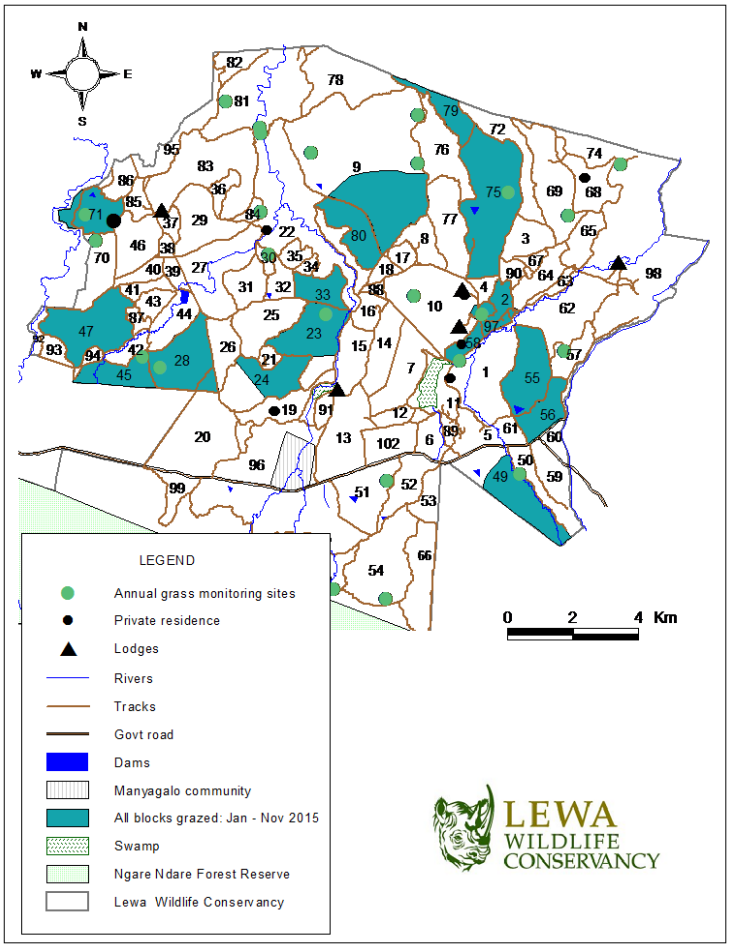

Figure 2.1. LWC divided in to grazing blocks and highlighting areas that were grazed by cattle in 2015 (*from* Lewa Research Department).

2.2 Data collection

2.2.1 Plot selection

At the beginning of the study period cattle were still present on blocks where grazing had begun in 2016. This study therefore focused primarily on blocks which had been grazed in 2015 where cattle were no longer present. Three areas of the conservancy had been mowed in 2015. These areas were matched with a grazed and a control area; resulting in three plots for each treatment (Fig.2.2).

Selecting comparable grazed sites was difficult due to the number of variables that differed between blocks. Attempts were made to select sites which were of a similar age (defined as the number of rainy seasons that had passed since management took place)

and had undergone a similar intensity of livestock grazing. Grazing pressure was estimated using the number of cattle present, the size of the block and the number of grazing days. Sites were also chosen so that they could be grouped approximately by location within LWC (west, central and east), in an attempt to account for differences in environmental variables such as rainfall and soil type (Fig.2.2; for details of surveyed plots see appendix A).

In addition to plots dating from 2015, LWC has an experimental site that was set up in 2014 (Fig.2.2). At this location 6 acre adjacent plots were set up to survey vegetation on mowed, grazed, grazed and mowed, and control sites. This plot was monitored in order to compare current data to Lewa Research Department's (LRD) existing data at this site.

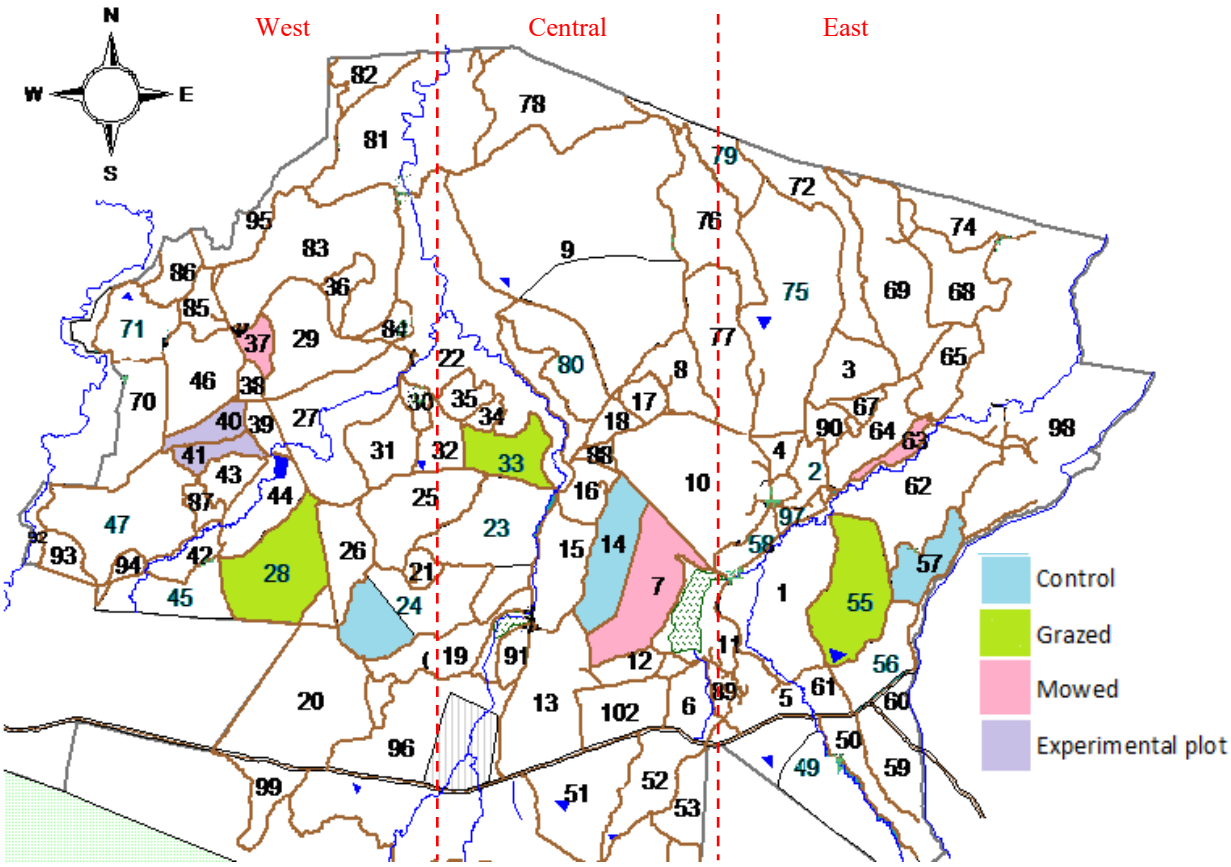

Figure 2.2. Location of survey sites on LWC: 3 mowed, 3 grazed and 3 control; plus the 2014 experimental plot.

2.2.2 Vegetation

The techniques used to monitor vegetation were based on those already in use by the LRD. Vegetation surveys were conducted on arrival at the study site and repeated at the

end of the study period. This resulted in two sets of data: one before the rains and one towards the end of the rainy season when vegetation productivity is highest.

At each site, three 100 m transects were placed at random and several different measurements were taken.

Grass quantity: To measure grass quantity a disc pasture meter was dropped at every 1 m along the transect. The disc pasture meter consists of a base plate and a long aluminium rod calibrated for use under rangeland conditions (Botha 1999). The mean settling height of the base plate on the grass can be used to estimate the standing crop of grass.

Species diversity and ground cover: To measure species diversity a pin was dropped at every 1 m along the transect. The grass species that touched the pin was recorded. If nothing was touching the pin it was recorded as 'bare ground' and any other vegetation was recorded as 'forb', 'sedge' or 'shrub'. Grass species were identified with the help of the LRD and Ibrahim & Kabuye (1987). If a species could not be identified it was labelled alphabetically for inclusion in calculations of diversity.

Leaf:stem ratio: To measure leaf:stem ratio a pin frame was used at 20 m, 60 m and 80 m along the transect. The pin frame consisted of a 1 m long bar with a pin at every 20 cm along its length. The number of leaves and stems touching each pin was counted, making note of whether they were brown or green, along with whether the pin was touching litter or bare ground. Leaf:stem ratio gives a measure of the quality and palatability of the vegetation, with more leaves to stems indicating better quality fodder.

2.2.3 Wildlife use

The central part of the study period focused on collecting data relating to wildlife utilisation of the plots. At each site a number of methods were used to gather information on wildlife presence and behaviour.

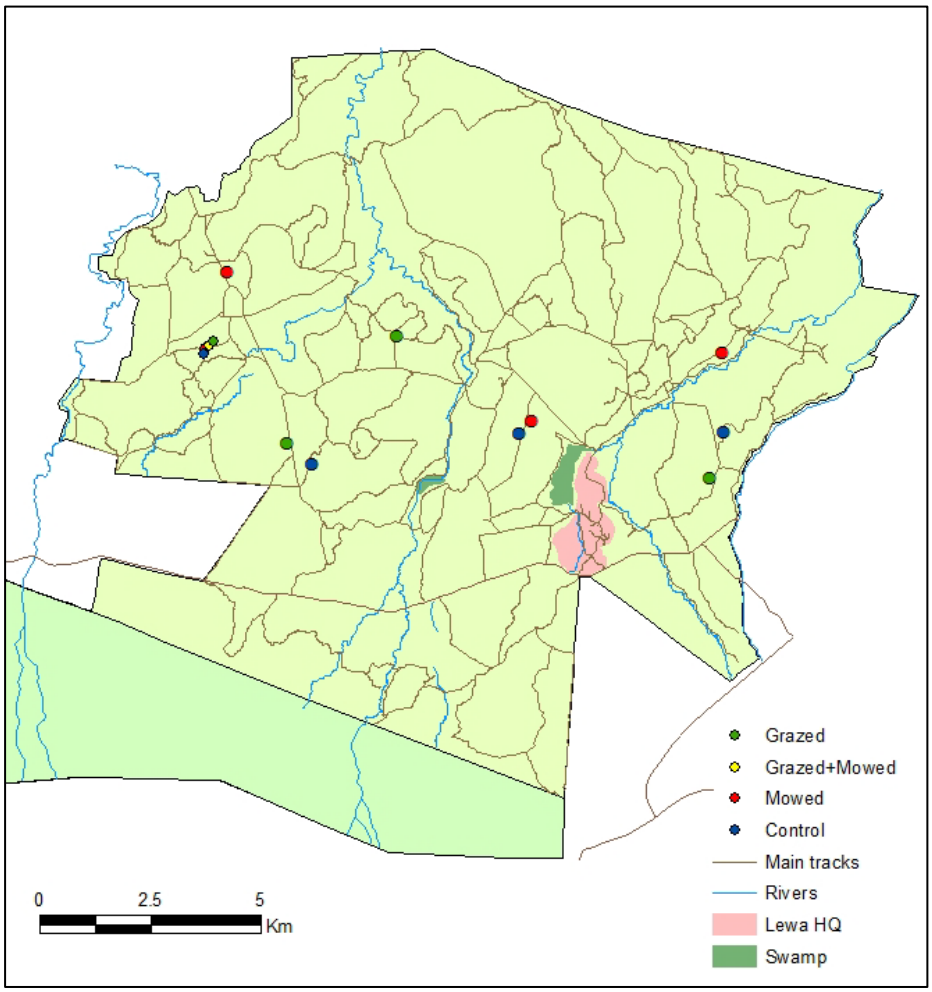

Figure 2.3. Location of camera traps on LWC

Camera traps: The majority of the wildlife use data came from camera traps which were placed at random on each block (Fig.2.3). Due to equipment limitations one camera was used for each block. All cameras were in place by the third week of the study and were then serviced every two weeks for the remainder of the study period. When photos were collected the following items were recorded for each sequence of photos: date, time, species, no. animals in photo and their behaviour.

Dung transects: To supplement the camera trap data, dung transects were also conducted. At each site four belt transects of 50 x 2 m were set up. During set up the transects were cleared of all dung to remove issues relating to age of dung and disappearance rates. Every two weeks the number of dung piles found along the transects was counted. The source species was identified using Stuart & Stuart (2013). After each survey the transects were cleared, either by removing or squashing the dung, in order that the same pile was not recounted at the next survey. In total 3 rounds of transects were conducted throughout the study period.

Exclusion cages: To asses which treatment was preferred by wildlife the grass offtake by herbivores was measured using exclusion cages. 1 m x 1 m movable cages were constructed at the LWC workshop using metal bars and chicken wire. These cages protected the vegetation inside and prevented it from being grazed by wildlife. At each of the 2015 plots, initial biomass of grass inside the cage was measured using the disc pasture meter. Biomass outside was measured on all four side of the cage. Each cage was re-visited after two weeks and the same measurements were taken in order to compare the change in biomass inside and outside the cage. Cages were then moved to another position on the plot and the process was repeated.

2.3 Data analysis

All analyses, plots and maps were completed using R statistical software (version 3.2.5) and ArcGIS (version 10.2.2), with a confidence interval of 95% and $p<0.05$ considered to be significant.

2.3.1 Vegetation data

To obtain a value for species diversity the Shannon-Wiener index was used. The Shannon-Wiener index is a popular diversity index in ecological literature and takes into account the relative abundance of different species in addition to the number of species present (Spellerberg & Fedor 2003). It is calculated using the formula:

$$H' = \sum_{i=1}^{s} (p_i)(\ln p_i)$$

where *S* is the total number of species, and *p* is the proportion of individuals belonging to species *i* (Shannon & Weaver 1949). Higher values indicate a high number of species and high uniformity/evenness, while values closer to zero indicate lower diversity.

Grass biomass was calculated using the settling height of the disc pasture meter and a conversion equation developed by Botha (1999) for use on LWC:

$$\text{Biomass (kg/ha)} = -3340 + 2323(\sqrt{\text{height}}).$$

To test the effect of different treatment types on vegetation data, one-way ANOVAs were used; with Levene's and Shapiro-Wilk tests confirming that assumptions of

normality and homogeneity of variances were met. If assumptions were violated a non-parametric Kruskal-Wallis test was run as an alternative to ANOVA. For the 2015 plots, nested one-way ANOVA was used in order to consider variation between plots within each treatment.

In addition, for the 2014 experimental site, results of previous years' data allowed for investigation of the effect of management techniques on vegetation over time. These data were analysed using linear models.

2.3.2 Wildlife use data

Camera trap photos were entered in to the software ZSL-CTAT (version 2.1.42; Davey *et al.* 2015). This tool allows large numbers of camera trap photos to be stored and managed. Summaries of numbers of photos of different species and numbers of wildlife events for each camera trap can be generated using this software. A series of photos was considered to be a new event when there had been an interval of at least 30 minutes between sets of photos.

Trapping rate was calculated for each species on each plot as the total number of individuals sighted divided by the number of days the camera was operational. Using trapping rate as a method of calculating abundance or density of a species can be difficult due to differences in detection probabilities and individual identification (Rowcliffe *et al.* 2008). However, in this instance it is being used as a way to assess numbers of animals across treatments and is suitable as a comparative measure to show the number of individuals captured on each camera per day.

Mammal species diversity was calculated for each plot using the number of individuals of each species and the Shannon-Wiener index (see above).

The behaviour of the animals in each photo was used to calculate the percentage of the total number of photos where animals were grazing, walking or other (unknown, resting, interacting).

Data from camera traps on the 2015 plots were then compared using one-way ANOVA to determine whether trapping rates, species diversity or behaviour differed between the treatments. For the 2014 experimental site, only one camera per plot resulted in no replicates and camera data could not be analysed statistically.

Dung transects yielded very little data and so dung piles were totalled over all transects for the 2015 plots; giving a total number of piles for each species per plot over the 6 weeks of data collection. For the 2014 plots, dung was totalled per transect over

the 6 weeks, resulting in 4 replicates per plot which could then be analysed statistically. These data were also analysed using one-way ANOVA to test for differences in number of dung piles between treatments.

The camera trapping rates and number of dung piles were grouped in a variety of ways to test for differences in distribution of species and groups. Initially, four key species were selected. These were Grant's gazelle (*Nanger granti*), Plains zebra (*Equus quagga*), Common eland (*Tragelaphus oryx*) and African buffalo (*Syncerus caffer*). These were selected as focal species as they were among the highest trapping rates based on camera trap photos and represent a range of sizes and feeding strategies.

Furthermore species were grouped by body size based on weight, as identified from the ZSL-CTAT species list and Alden *et al.* (1995). In addition species were grouped by their digestive strategy, either hindgut fermenters or ruminants, using information from Alden *et al*. (1995) and Kingdon (2015). Details of species groupings can be found in appendix B.

To estimate offtake by wild herbivores, change in biomass inside and outside the exclusion cages was compared using paired t-tests.

3. Results

3.1 Vegetation data

3.1.2 Biomass

Across all plots there was a detectable effect of treatment on grass biomass (Table 1). In all instances M plots had significantly lower biomass than G or C, which did not differ from each other (Table 1; Fig.3.1). At the 2014 experimental site the G+M area also had a lower biomass than G or C (Table 1; Fig.3.1).

Combining current data with existing data for the 2014 experimental site indicated that there was a significant effect of time ($F_{1,18}$=11.29, $p<0.01$), treatment ($F_{3,18}$=68.31, $p<0.001$) and their interaction ($F_{3,18}$=4.78, $p=0.013$); and that the model had high explanatory power (LM: $F_{7,18}$=32.97, R^2=0.90). The biomass did not change over time in the C plot (LM: b=-1.462, S.E.=3.755, p=0.70; Fig.3.2). The slopes of the G+M and M did not differ from C (LM: G+M: b=6.985, S.E.=5.949, p=0.17; M: b=7.342, S.E.=4.981, p=0.09), thereby indicating that biomass did not change over time in these plots (Fig.3.2). However, G did differ from C and calculations using standard error indicated that biomass increased over time in this plot (LM: b=17.379, S.E.=5.022, $p<0.01$; Fig.3.2). Furthermore, the net effect was an overall increase in biomass in the grazed treatment (Fig.3.2).

While both grazing and mowing reduced biomass immediately after treatment, in grazed plots biomass returned more quickly to its original level (Fig.3.2). This was further confirmed by a visit to a 2016 grazed plot. Year of treatment had a detectable effect on biomass ($F_{5,30}$=19.05, $p<0.001$). Biomass was lower in the 2016 plot than in the 2014 and 2015 grazed plots which had undergone a recovery period (Fig.3.3; appendix C). The 2016 graze had comparable biomass to the mowed plots despite these plots being older (Fig.3.3; appendix C).

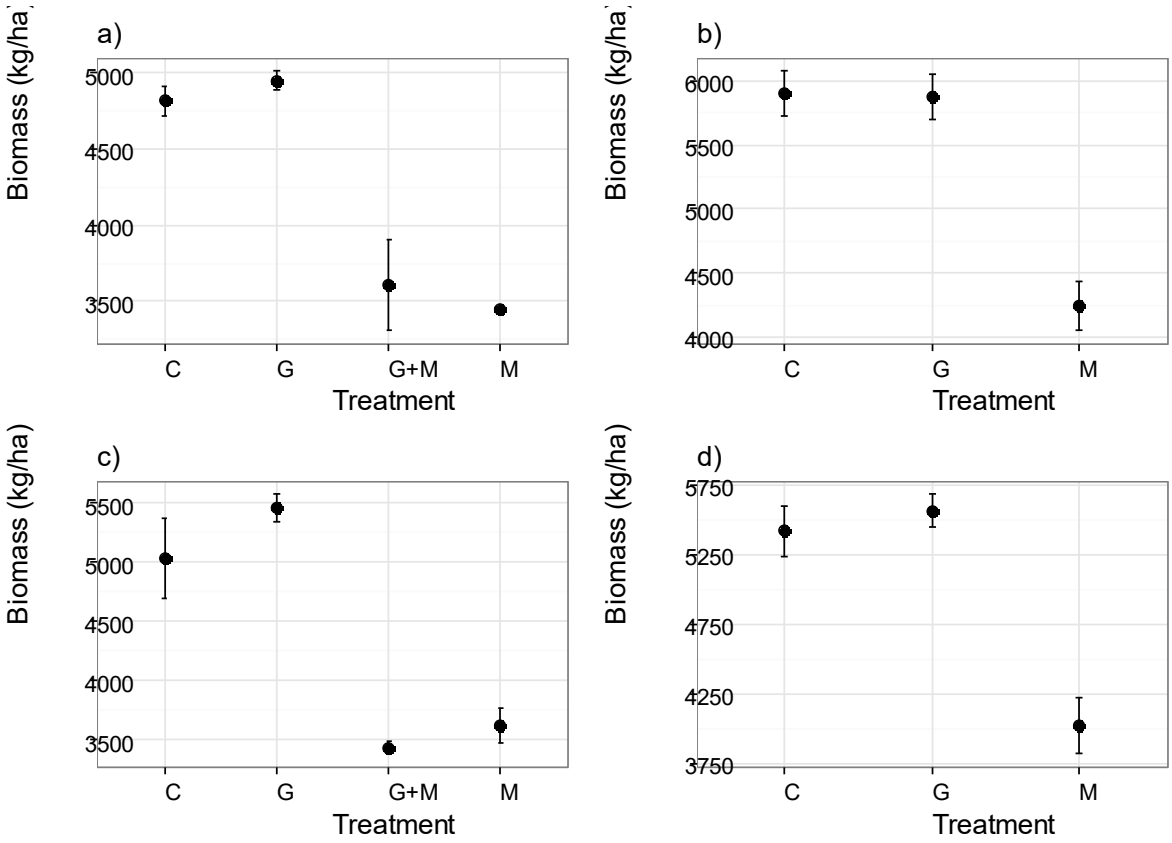

Figure 3.1. Mean biomass (kg/ha) ± S.E. for the 2014 experimental plot in the dry (a) and wet seasons (c) and for the 2015 plots in the dry (b) and wet seasons (d).

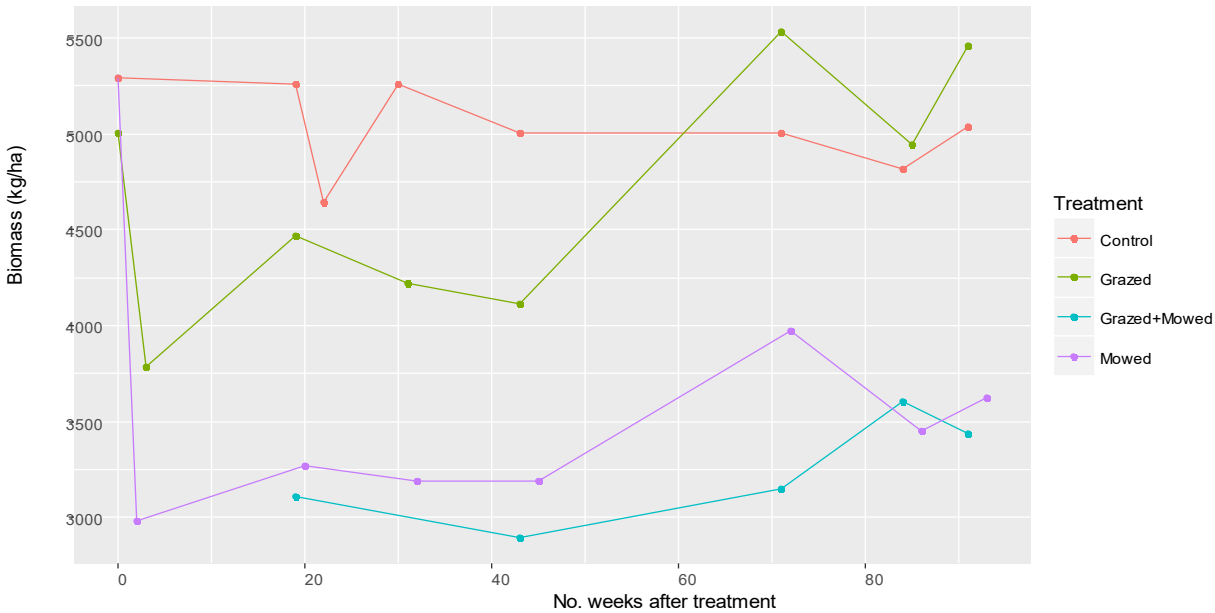

Figure 3.2. Change in biomass over time at the 2014 experimental plot.

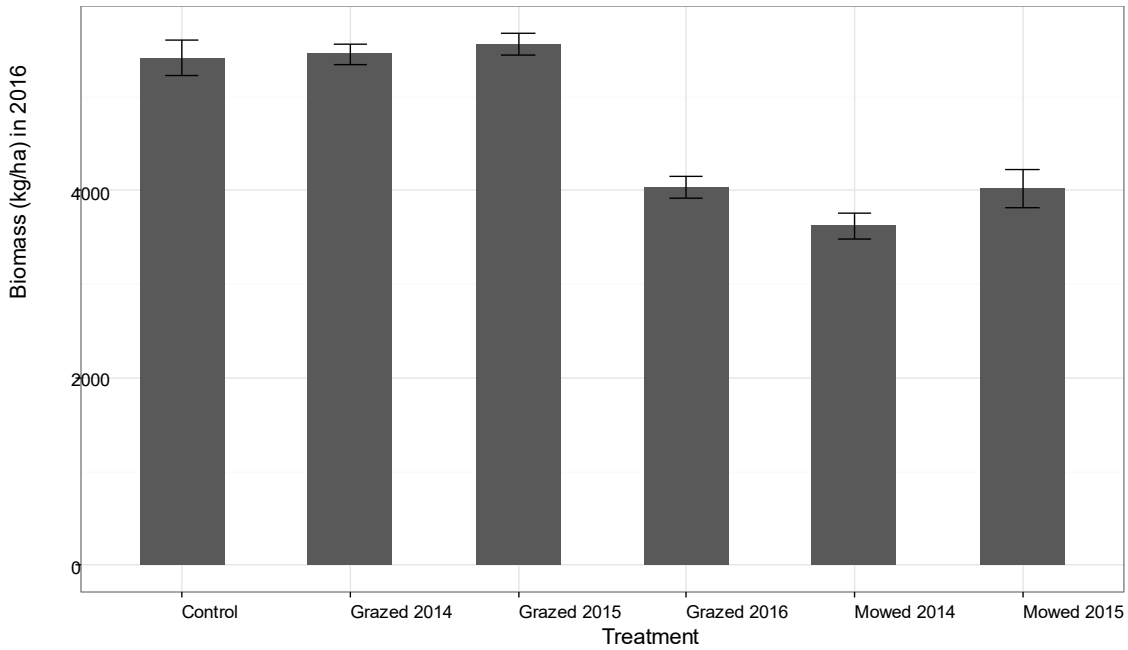

Figure 3.3. Mean biomass ± S.E. in plots that underwent management in different years.

Table 1. Means ± standard errors of vegetation characteristics. Different letters denote significant differences between means (TukeyHSD). Treatment means in rows with no letters were not significantly different from each other. Results of one-way ANOVA for effect of treatment given in final two columns. P-values highlighted in bold are considered significant.

	Season	Control	Grazed	Mowed	Grazed + Mowed	F-value*	p-value
		2014 experimental plot					
Biomass (kg/ha)	Dry	$4814^{a} \pm 93$	$4946^{a} \pm 63$	$3447^{b} \pm 16$	$3604^{b} \pm 298$	24.31	**<0.001**
	Wet	$5034^{a} \pm 340$	$5458^{a} \pm 111$	$3624^{b} \pm 140$	$3439^{b} \pm 56$	27.02	**<0.001**
Species diversity	Dry	0.68 ± 0.37	1.09 ± 0.24	1.13 ± 0.12	1.15 ± 0.15	0.85	0.51
(Shannon index)	Wet	$0.72^{a} \pm 0.14$	$0.83^{ab} \pm 0.10$	$1.22^{bc} \pm 0.03$	$1.53^{c} \pm 0.13$	11.9	**<0.01**
% vegetation cover	Dry	$90^{a} \pm 2.08$	$86^{a} \pm 2.60$	$62^{b} \pm 1.00$	$58^{b} \pm 3.53$	43.3	**<0.001**
	Wet	$84^{a} \pm 1.86$	$91^{a} \pm 1.76$	$81^{ab} \pm 2.00$	$70^{b} \pm 3.76$	11.55	**<0.01**
% litter cover	Dry	98 ± 2.22	91 ± 5.88	80 ± 3.85	82 ± 4.44	3.64	0.06
	Wet	100 ± 0	98 ± 2.22	96 ± 2.22	91 ± 5.88	1.30	0.12
Leaf:stem ratio	Dry (brown)	1.74 ± 0.32	1.17 ± 0.15	1.40 ± 0.24	1.93 ± 0.26	1.85	0.22
	Wet (green)	2.42 ± 0.46	2.40 ± 0.82	4.92 ± 1.86	8.17 ± 1.59	4.33	**0.043**
Green:brown vegetation	Wet	0.49 ± 0.12	0.66 ± 0.28	0.66 ± 0.10	0.59 ± 0.12	0.22	0.88
		2015 plots					
Biomass (kg/ha)	Dry	$5898^{a} \pm 177$	$5874^{a} \pm 177$	$4240^{b} \pm 191$		47.05	**<0.001**
	Wet	$5418^{a} \pm 183$	$5564^{a} \pm 117$	$4022^{b} \pm 205$		57.91	**<0.001**
Species diversity	Dry	0.89 ± 0.11	0.86 ± 0.72	1.07 ± 0.11		2.64	0.10
(Shannon index)	Wet	1.08 ± 0.10	1.11 ± 0.12	0.92 ± 0.17		1.49	0.25
% vegetation cover	Dry	$73^{a} \pm 4.54$	$87^{b} \pm 5.70$	$67.22^{a} \pm 7.84$		8.94	**<0.01**
	Wet	$97^{a} \pm 0.90$	$94^{a} \pm 1.60$	$84^{b} \pm 3.65$		33.09	**<0.001**
% litter cover	Dry	94 ± 2.43	90 ± 9.62	99 ± 0.74		2.48	0.11
	Wet	99 ± 0.98	99 ± 0.74	99 ± 0.98		**	**
Leaf:stem ratio	Dry (brown)	1.24 ± 0.19	1.59 ± 0.23	1.60 ± 0.39		1.28	0.30
	Wet (green)	$4.11^{ab} \pm 1.11$	$8.00^{a} \pm 3.91$	$3.19^{b} \pm 0.75$		3.65	**0.047**
Green:brown vegetation	Wet	$0.12^{a} \pm 0.03$	$0.29^{b} \pm 0.07$	$0.34^{b} \pm 0.06$		5.82	**0.011**

* 2014 experimental plot (df=3,8); 2015 plots (df=2,18).

** data had non-homogenous variances so Kruskal-Wallis test used in place of ANOVA (H_2=0.47, p=0.79)

3.1.2 Species diversity

The dominant grass species across all plots were *Pennisetum stramineum* (PS) and *Pennisetum mezianum* (PM). During the dry season these grasses made up >70% of the grass cover in every block for both the 2014 and the 2015 plots (Fig.3.4; Fig.3.5).

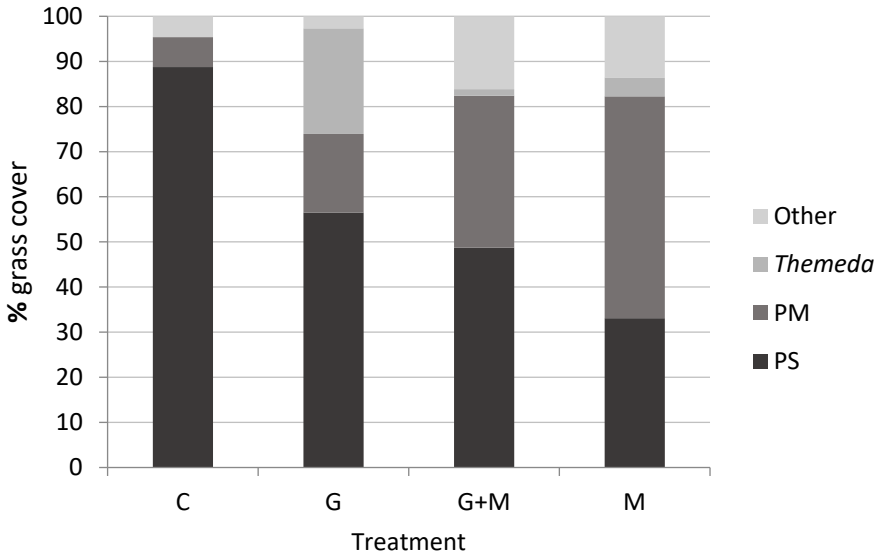

Figure 3.4. Dry season grass species composition at the 2014 experimental plot.

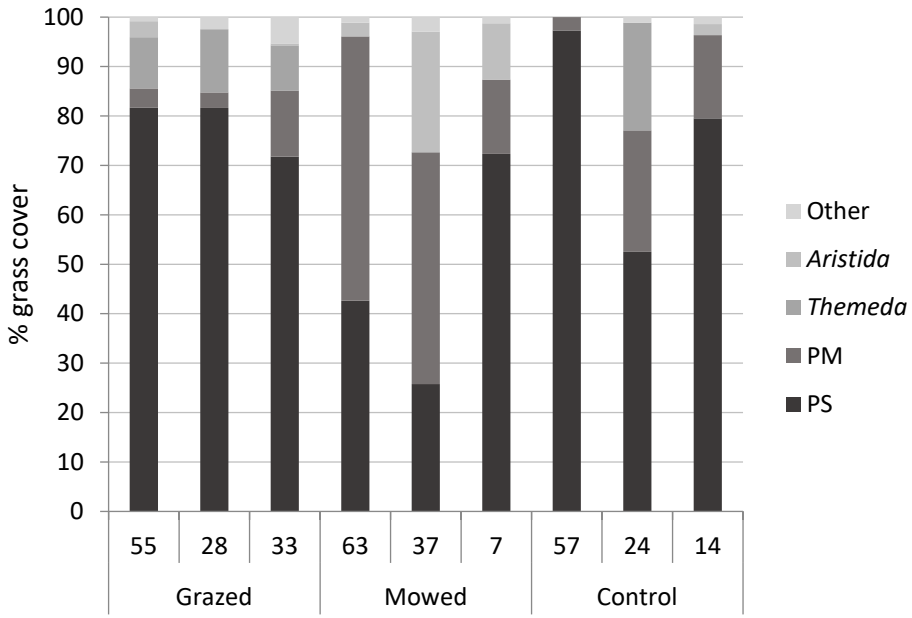

Figure 3.5. Dry season grass species composition for the 2015 plots.

During the wet season, PS and PM still dominated the G and C areas of the 2014 plot. However, M and G+M appeared to have a higher proportion of other grass species (Fig.3.6; see appendix D for list of all identified grass species). PS and PM made up >50% grass cover in all blocks for the 2015 treatments (Fig.3.7).

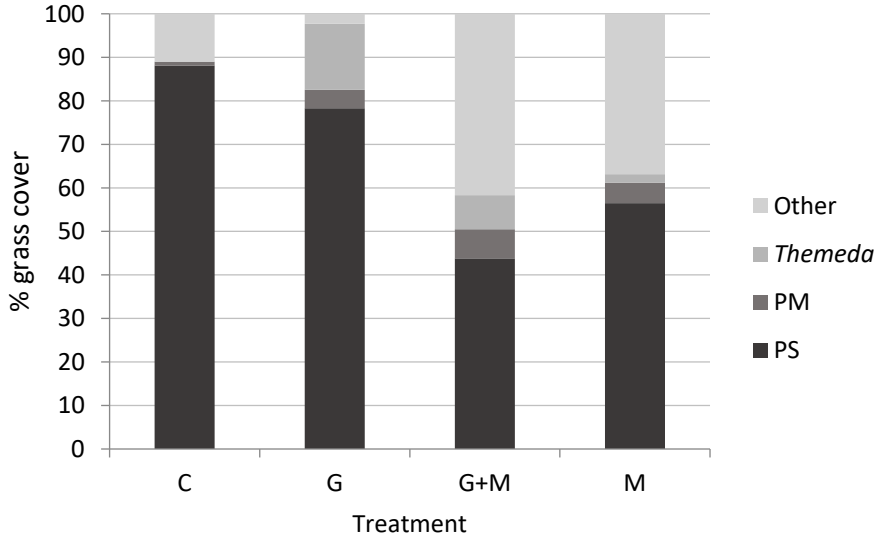

Figure 3.6. Wet season grass species composition at the 2014 experimental plot.

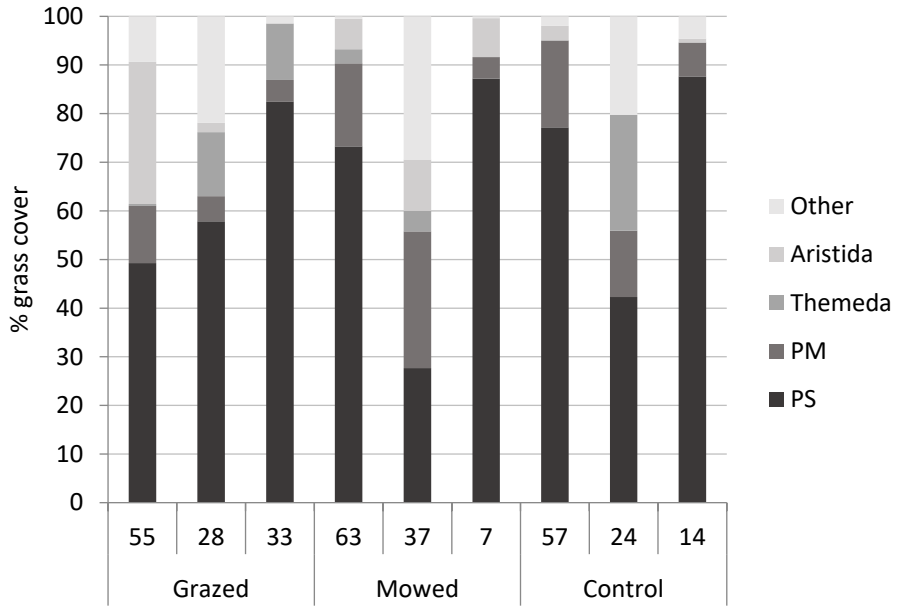

Figure 3.7. Wet season grass species composition for the 2015 plots.

During the dry season there was no difference in grass species diversity across any of the treatments (Table 1; Fig.3.8). Similarly for the 2015 plots during the wet season species diversity did not differ between treatments (Table 1; Fig.3.8). For the 2014 experimental plot, wet season data showed a detectable difference in species diversity, with the G+M area having higher species diversity than the G and C areas (Table 1). Additionally, M species diversity was higher than C (Table 1; Fig.3.8).

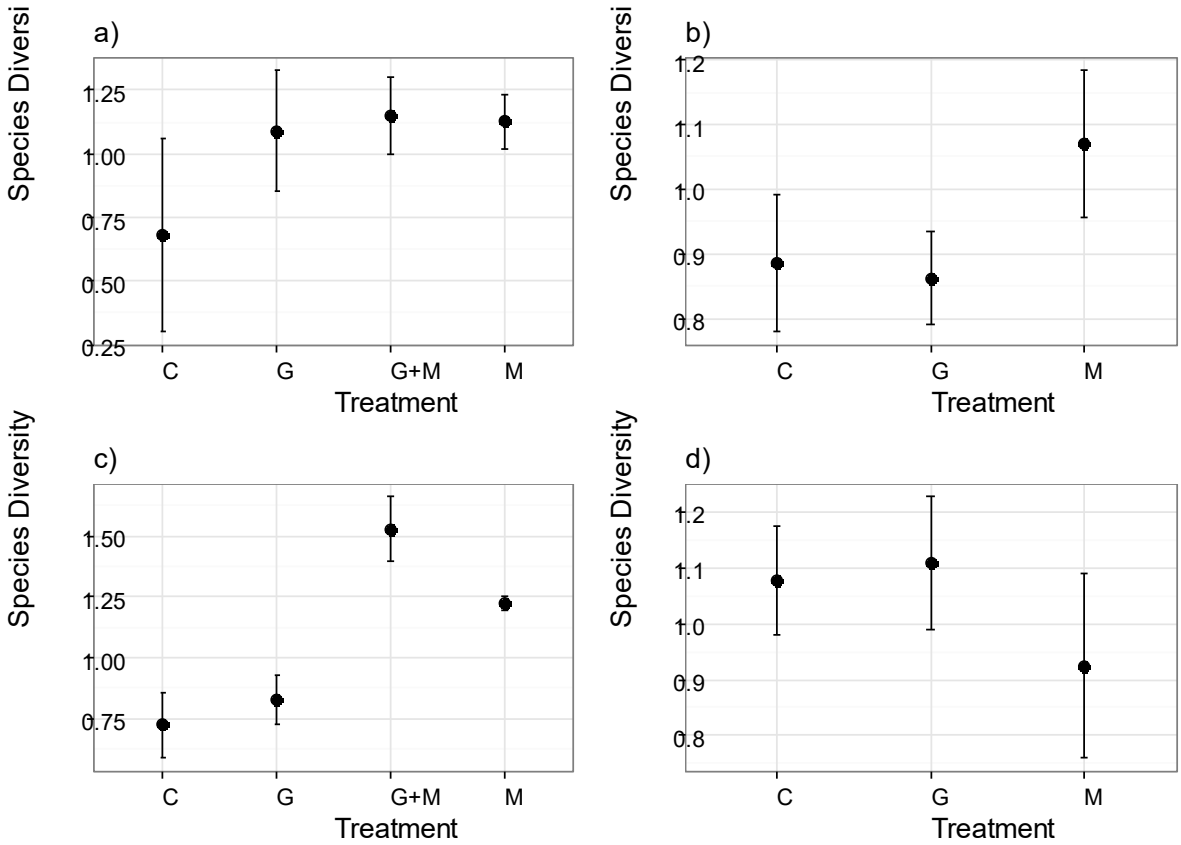

Figure 3.8. Mean species diversity ± S.E. for the 2014 experimental plot in the dry (a) and wet seasons (c) and for the 2015 plots in the dry (b) and wet seasons (d). Diversity is represented using the Shannon-Wiener index.

There was significant variation between blocks within each treatment for the 2015 plots, both during the dry ($F_{6,18}$=4.97, p<0.01; Fig.3.9) and wet ($F_{6,18}$=7.46, p<0.001, Fig.3.9) seasons. Including area in the model showed that there was a difference in species diversity between areas for both the dry ($F_{2,18}$=8.29, p<0.01) and wet ($F_{2,18}$=19.45, p<0.001) seasons (Fig.3.10). Blocks in the western section of LWC had higher species diversity overall. Additionally, during the wet season blocks in the eastern sections had higher diversity than those in the centre (Fig.3.10; appendix C).

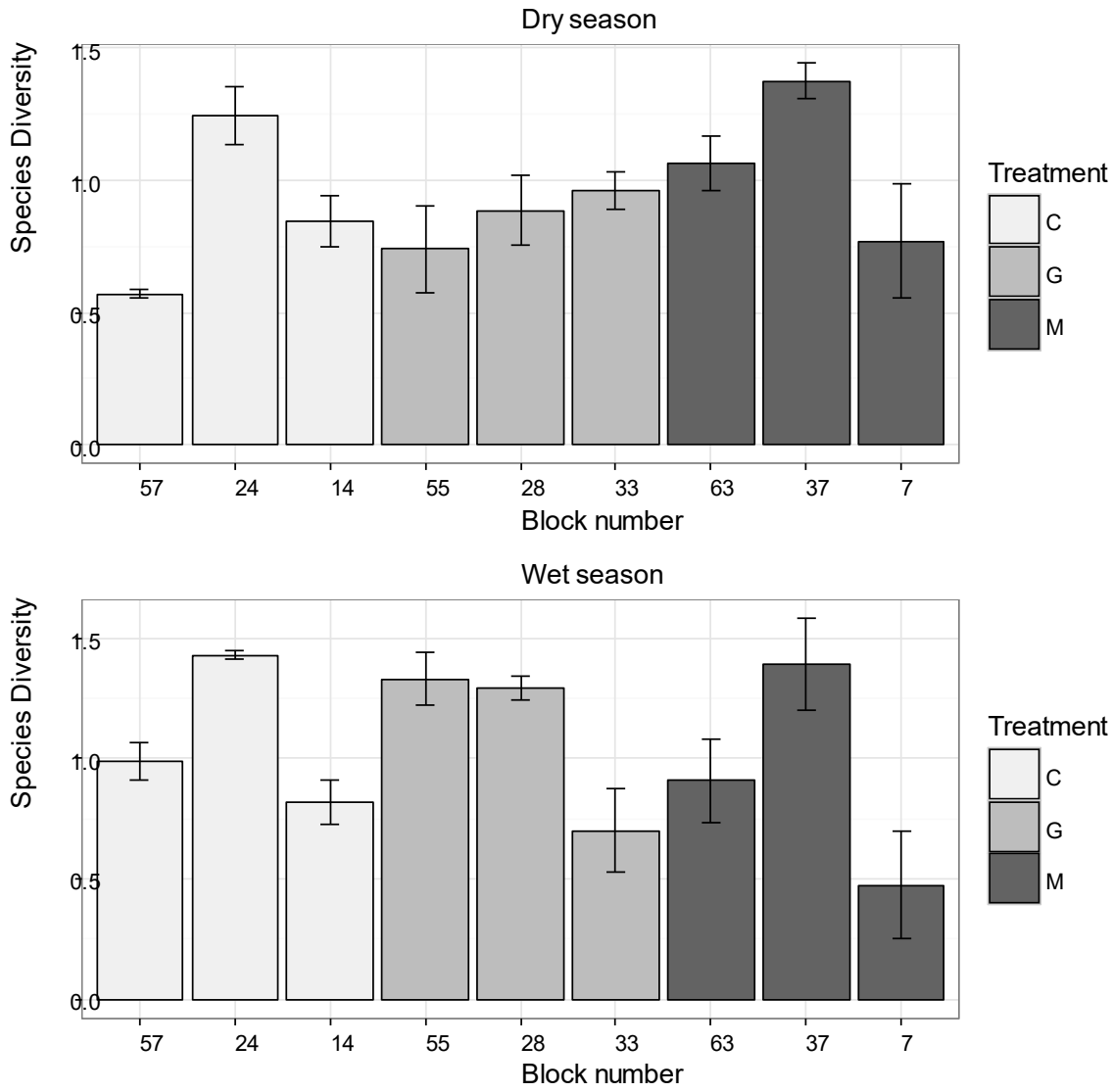

Figure 3.9. Mean species diversity (Shannon-Wiener index) ± S.E. for each of the 2015 blocks during the dry and wet seasons.

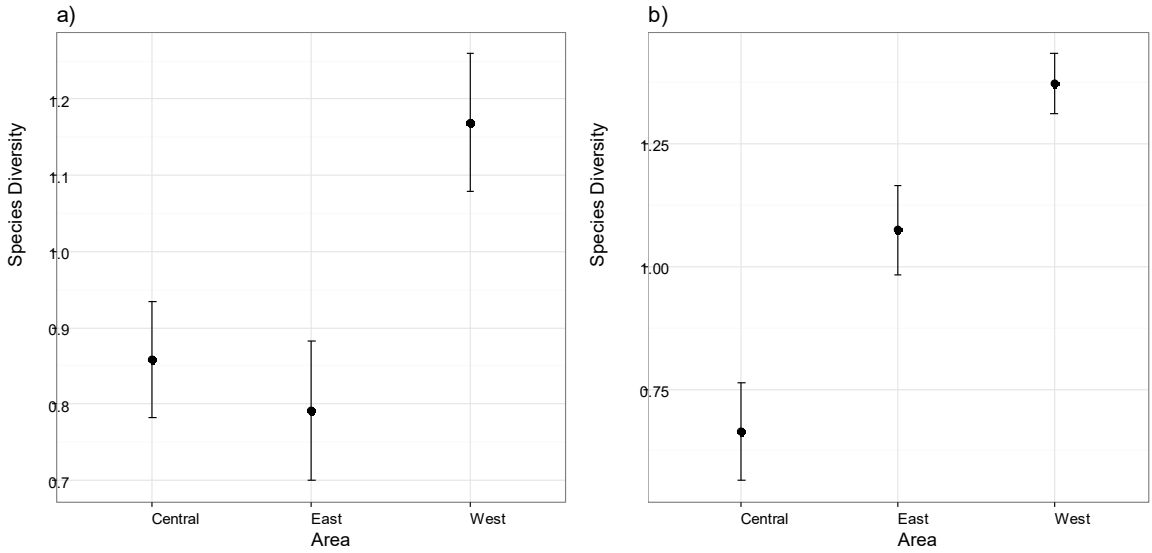

Figure 3.10. Mean species diversity (Shannon-Wiener index) ± S.E. for the 2015 plots in the dry (a) and wet (b) seasons, grouped by area of LWC.

3.1.3 Vegetation cover

There was a detectable effect of treatment on percentage of standing vegetation cover (Table 1). For the 2014 experimental site, the G+M treatment had lower vegetation cover than the G and C both during the dry and wet seasons (Table 1; Fig.3.11). In the dry season, the M area also had less cover than the G and C (Table 1; Fig.3.11). For the 2015 plots, G had a higher percentage cover than M or C during the dry season (Table 1; Fig.3.11). While in the wet season M had a lower percentage cover than G or C (Table 1; Fig3.11).

There was no difference in litter cover between treatments (Table 1; Fig.3.12). Across all treatments and seasons litter cover was >80% (Fig.3.12), indicating that there was a negligible percentage of bare ground.

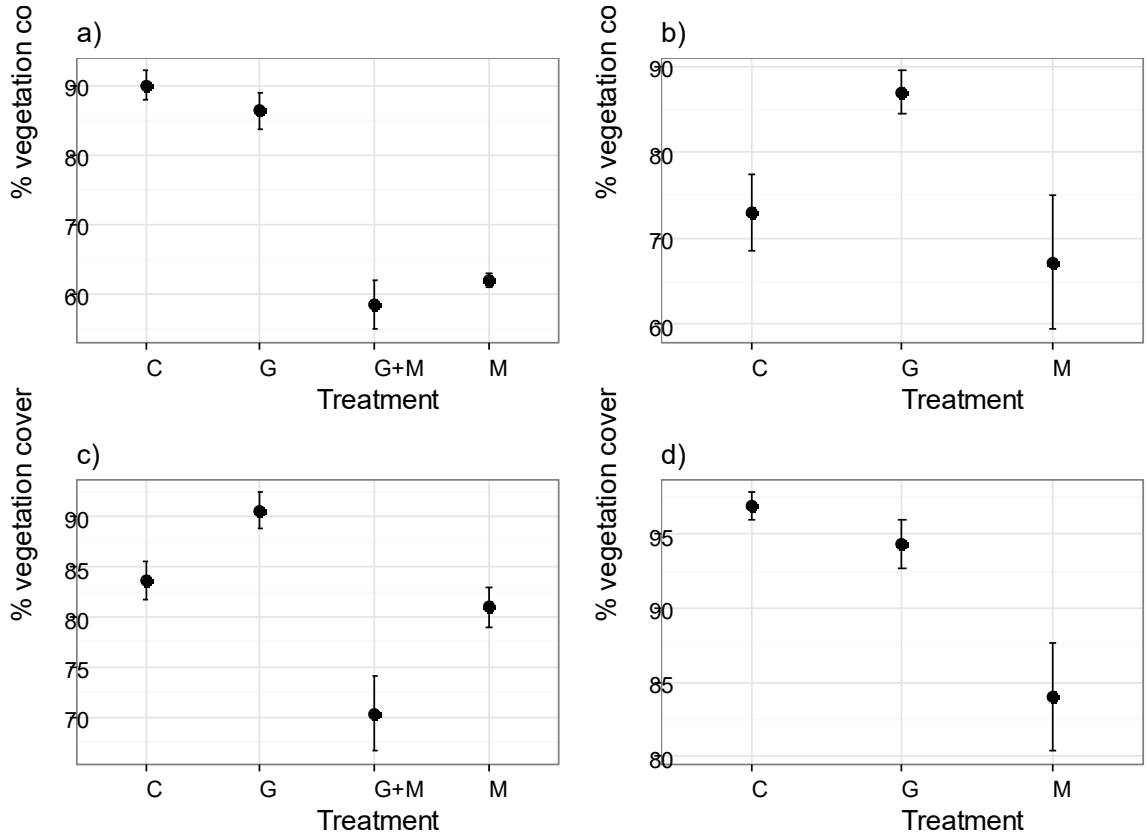

Figure 3.11. Mean percentage vegetation cover ± S.E. for the 2014 experimental plot in the dry (a) and wet seasons (c) and for the 2015 plots in the dry (b) and wet seasons (d).

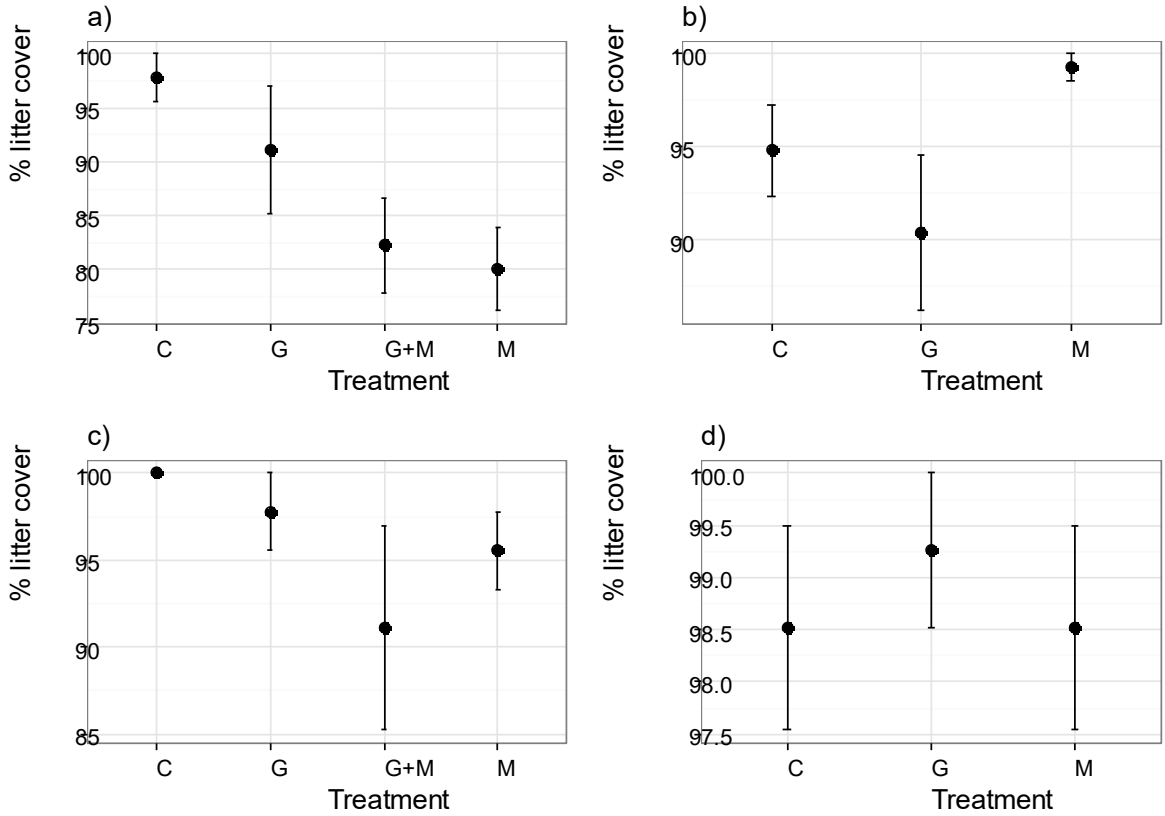

Figure 3.12. Mean percentage litter cover ± S.E. for the 2014 experimental plot in the dry (a) and wet seasons (c) and for the 2015 plots in the dry (b) and wet seasons (d).

3.1.4 Vegetation characteristics

In the dry season, the ratio of brown leaves to brown stems did not differ between the treatments (Table 1; Fig.3.13). However, there was an effect of treatment on green leaf:stem ratio in the wet season (Table 1; Fig.3.13). TukeyHSD tests could not detect differences between treatments for the 2014 experimental plot (Table 1). The largest difference in means lay between the G+M and C and the G+M and G. In the 2015 plots the G treatment had a higher leaf:stem ratio than the M, but did not differ from the C (Table 1; Fig.3.13).

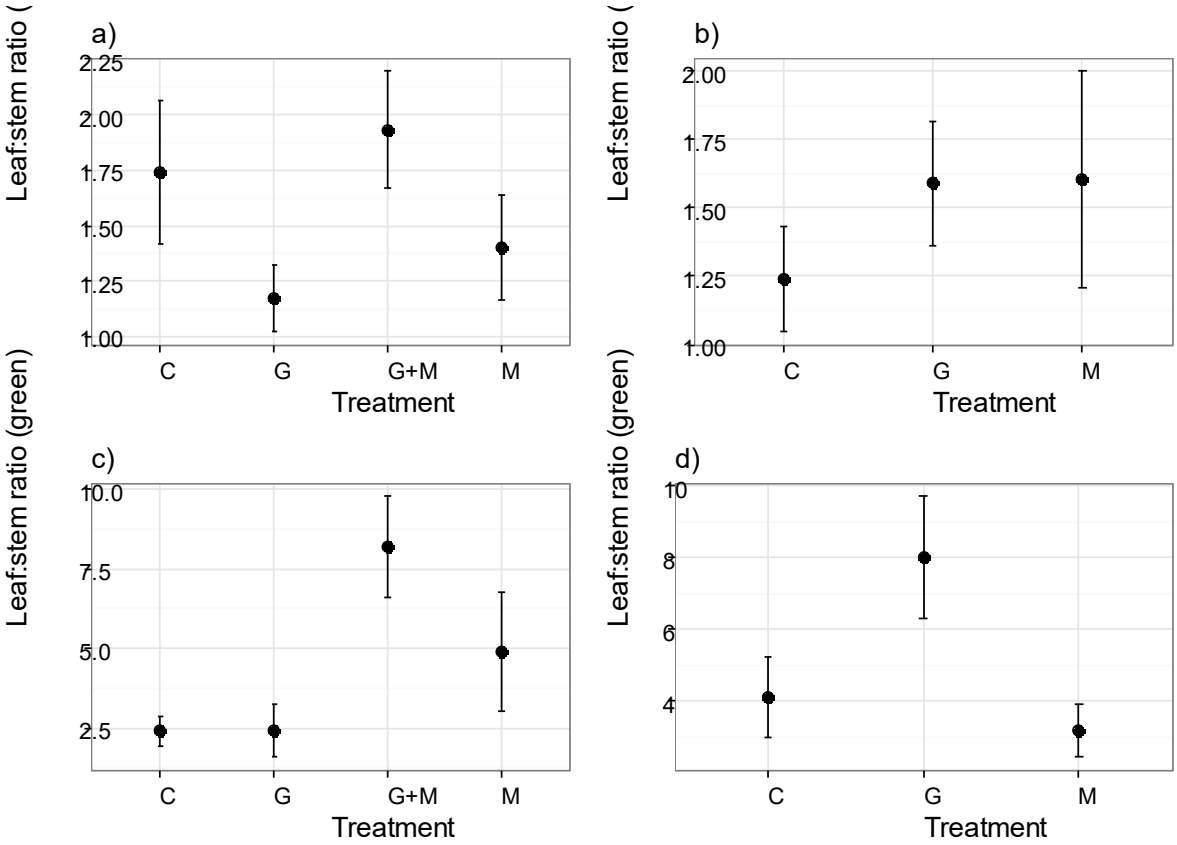

Figure 3.13. Mean leaf:stem ratio ± S.E. for the 2014 experimental plot in the dry (a) and wet seasons (c) and for the 2015 plots in the dry (b) and wet seasons (d).

During the wet season it was possible to investigate the ratio of green:brown plant material. For the 2014 experimental plot there was no difference in proportion of green:brown vegetation between treatments (Table 1; Fig.3.14). However, in the 2015 plots both G and M plots had a higher proportion of green:brown vegetation than C (Table 1; Fig.3.14).

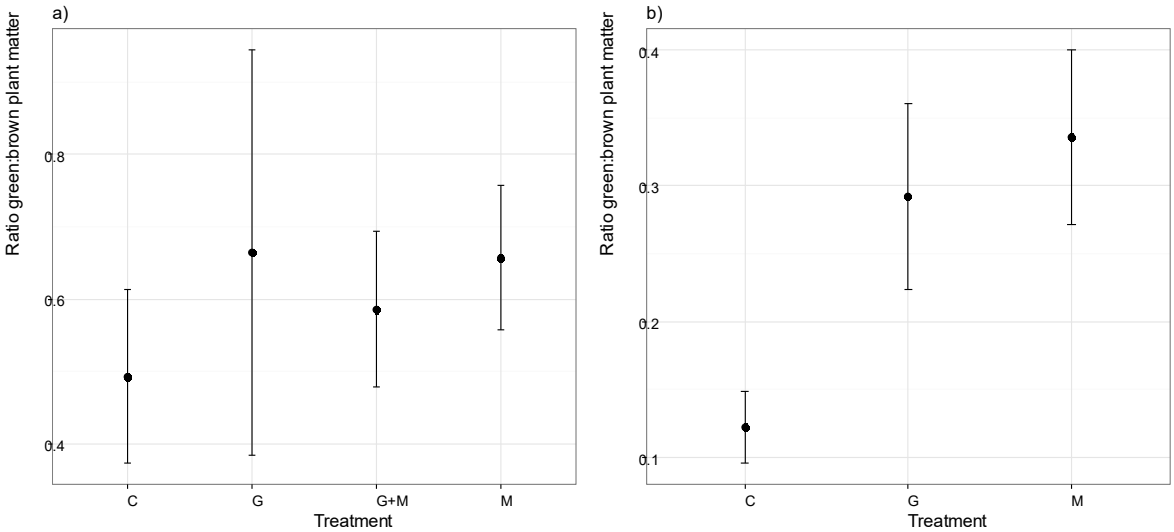

Figure 3.14. Mean ratio of green:brown vegetation ± S.E. during the wet season for the 2014 experimental plot (a) and the 2015 plots (b).

3.2 Wildlife use data

3.2.1 Overview

In total, across all camera traps 19 mammal species were captured (see appendix E for list of mammal species). There were a total of 386 wildlife events captured over the study period and just 13 of these were carnivore captures. Dung transects were less efficient at capturing species presence, with dung of only 12 species found over all plots. There was very little dung in comparison to the number of wildlife events captured by camera traps.

Mammal species diversity did not differ between the treatments for the 2015 plots (Camera data: $F_{2,6}$=0.38, p=0.70; Dung data: $F_{2,6}$=0.28. p=0.76; Fig.3.15).

For the 2014 experimental plot, formal analysis of camera trap data was not possible due to lack of replicates. However, the results appeared to be consistent with 2015 plots indicating similar species diversity across treatments (Fig.3.16).

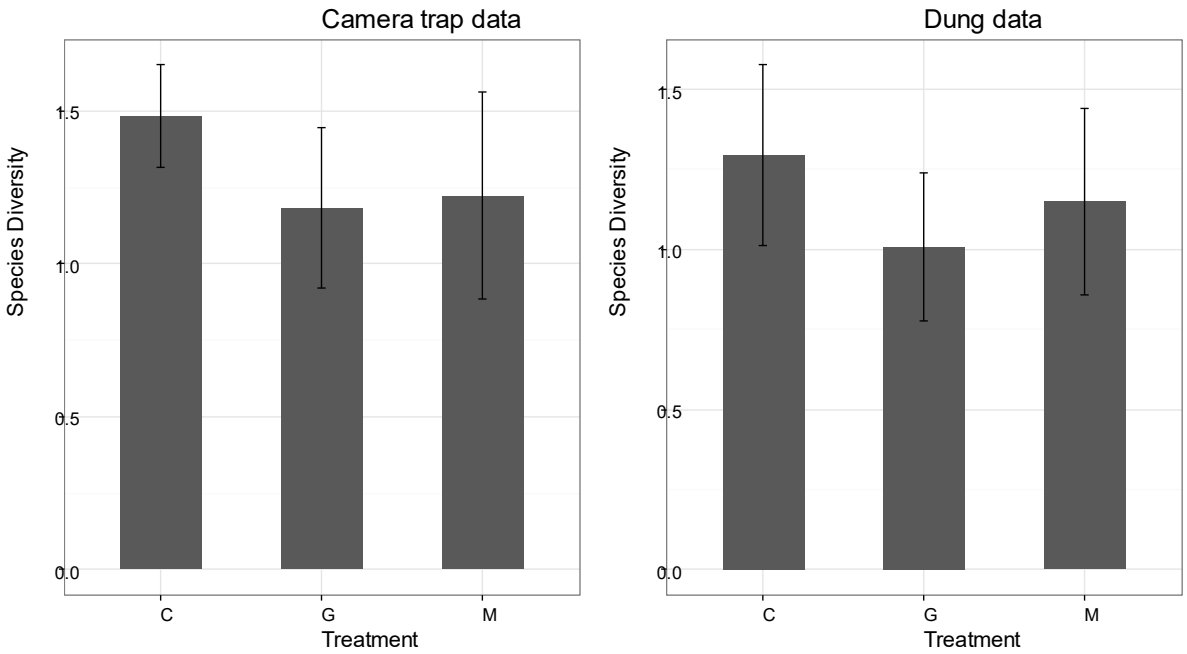

Figure 3.15. Mean mammal species diversity (Shannon-Wiener index) ± S.E. for the 2015 plots using camera traps and dung counts.

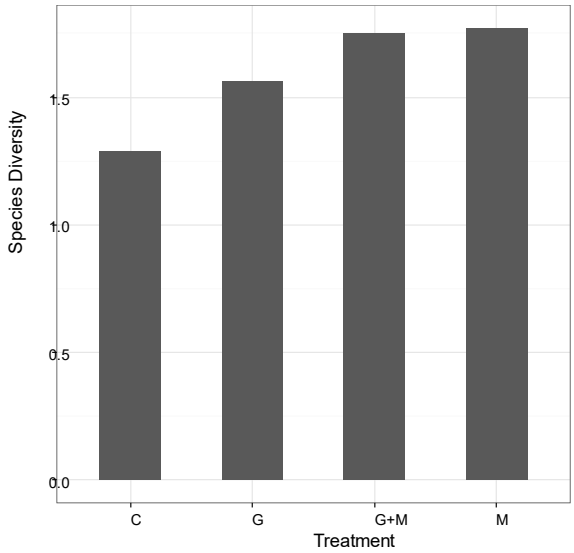

Figure 3.16. Mean species diversity (Shannon-Wiener index) for the 2014 experimental plot based on camera trap data.

Camera trap data indicated no difference in number of wildlife events between treatments for the 2015 plots ($F_{2,6}$=0.87, p=0.47; Fig.3.17a). In addition, there was no difference in number of events based on area ($F_{2,6}$=2.97, p=0.13; Fig.3.17b). On the 2014 experimental plot, there was some indication that number of wildlife events was lower on the G plot (Fig.3.18).

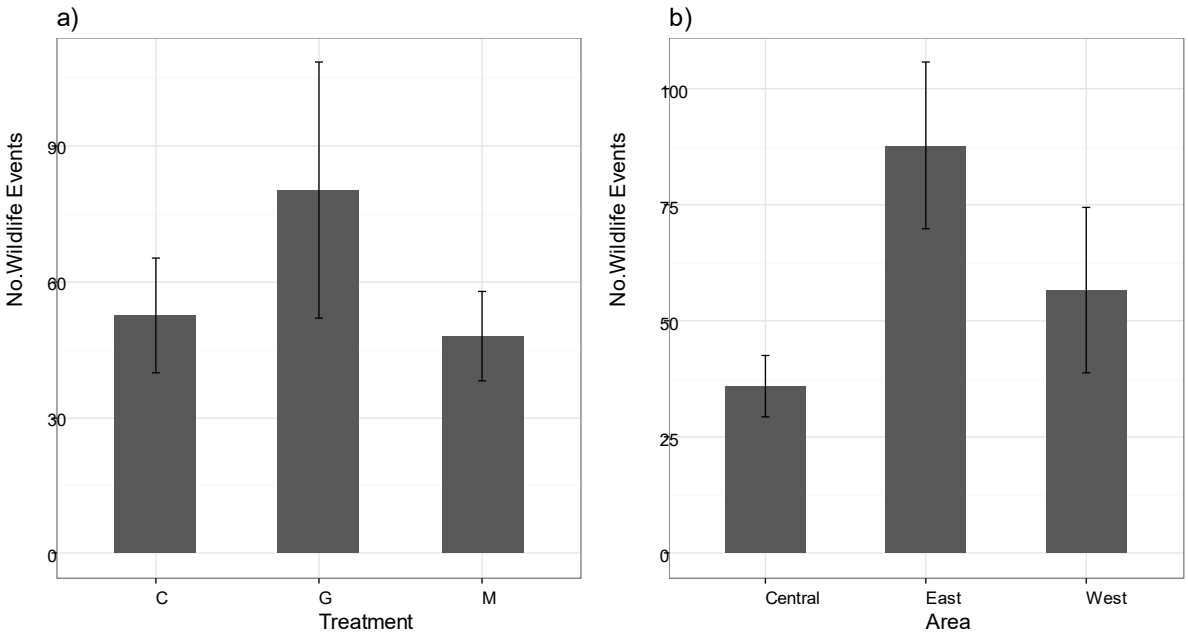

Figure 3.17. Mean no. wildlife events over 100 days ± S.E. for the 2015 plots across treatments (a) and areas of LWC (b).

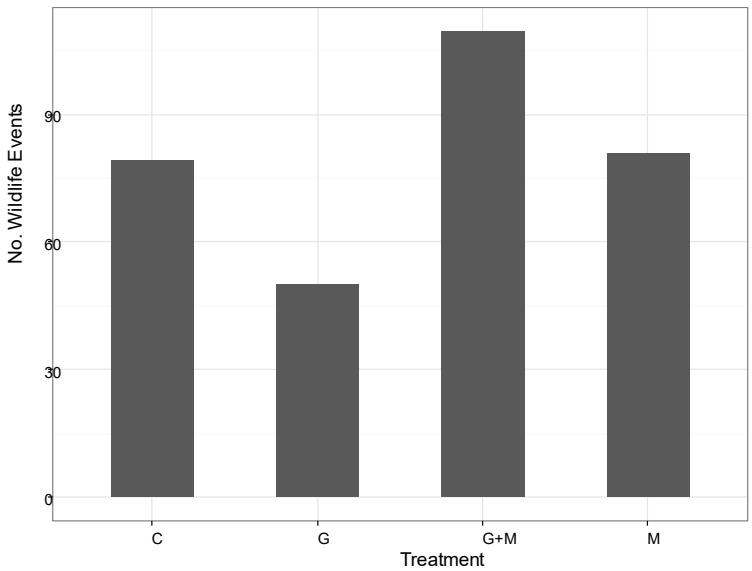

Figure 3.18. Mean no. of wildlife events over 100 days for the 2014 experimental plot.

3.2.2 Focal species

There was no effect of treatment on trapping rate for Grant's gazelle ($F_{2,6}$=0.83, p=0.48), or buffalo ($F_{2,6}$=0.21, p=0.82) on the 2015 plots. Plains zebra trapping rate was higher on G plots than on C ($F_{2,6}$=6.20, p=0.035; Fig.3.19; appendix C). In addition eland trapping rate differed between treatments ($F_{2,6}$=10.89, p=0.0101) with a greater number of eland found on C plots than on G or M (Fig.3.19; appendix C). Using data from dung transects did not uncover any difference in number of dung piles between treatments for Grant's gazelles ($F_{2,6}$=0.33, p=0.73), zebra ($F_{2,6}$=1.32, p=0.34), buffalo ($F_{2,6}$=0.57, p=0.60) or eland ($F_{2,6}$=0.43, p=0.67; Fig.3.19). It is not possible to distinguish between dung of Plains and Grevy's zebra, therefore dung data represents both species. However Plains zebra occur at higher density and it is likely that most of the dung found belonged to this species.

For both camera trap and dung data there was large variation around the mean. Testing whether this variation was explained by area showed that there was no difference in camera trapping rate for Grant's gazelle ($F_{2,6}$=0.08, p=0.93), Plains zebra ($F_{2,6}$=0.30, p=0.75), buffalo ($F_{2,6}$=1.71, p=0.26) or eland ($F_{2,6}$=0.13, p=0.88) based on areas of LWC.

Blocks 28 and 55 had higher trapping rates of Plains zebra than other blocks (Fig.3.20). This was not due to one/two large groups, as they also had a high number of

zebra events (Fig.3.21). Dung data confirmed that there were more zebra on blocks 28 and 55, although unlike the camera trapping rate there was also evidence of a large number of zebra on block 24 (Fig.3.20).

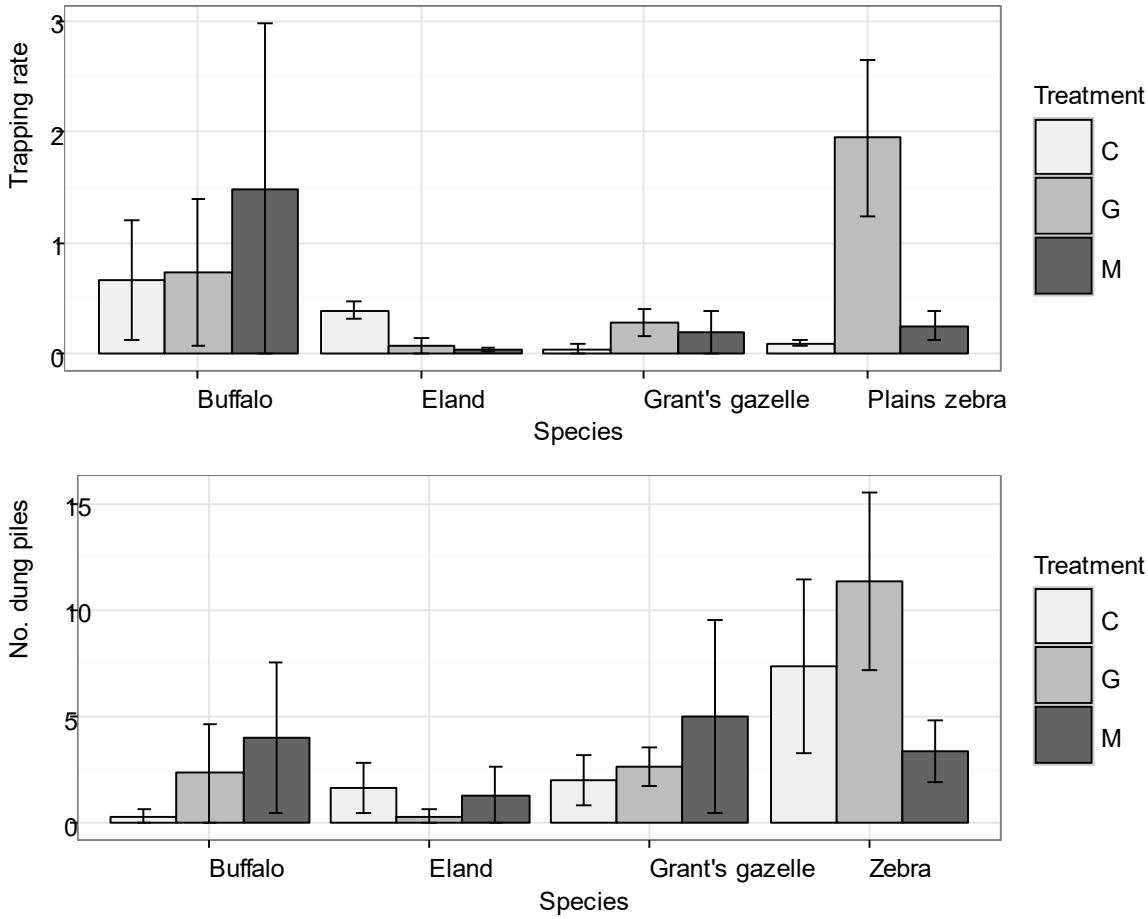

Figure 3.19. Mean trapping rate and mean number of dung piles ± S.E. of four focal species for the 2015 plots.

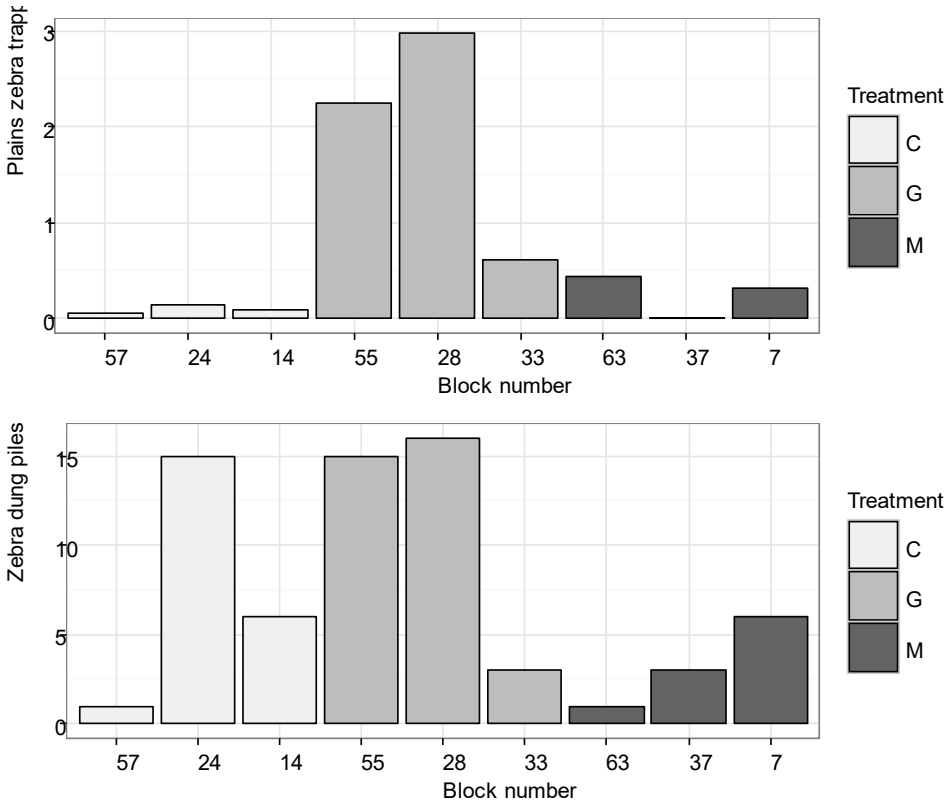

Figure 3.20. Trapping rates and dung counts of zebra for each block of the 2015 treatments.

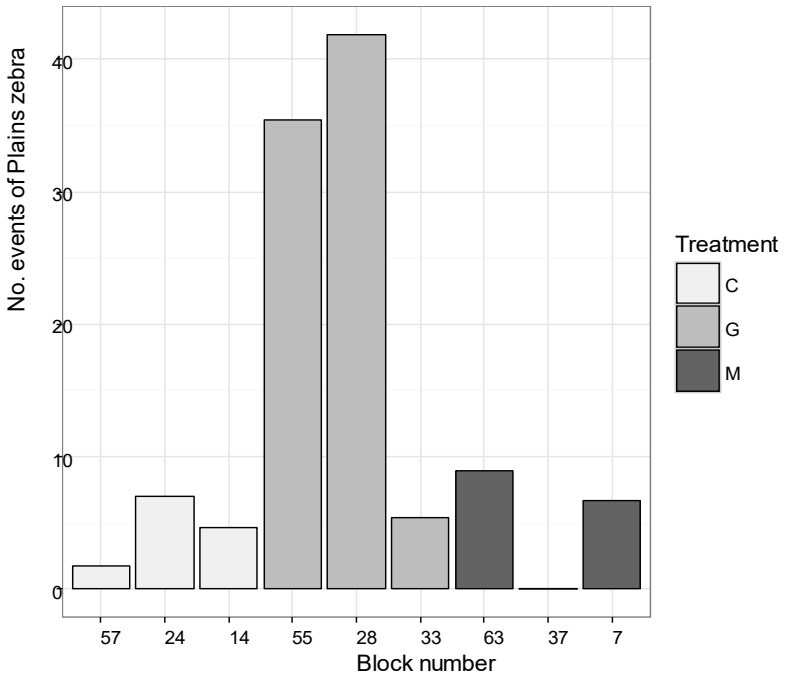

Figure 3.21. Number of Plains zebra events for each block of the 2015 treatments.

On the 2014 experimental plot, only 3 piles of eland dung and 4 piles of buffalo dung were found throughout the entire study period, meaning statistical analysis for these focal species was not possible. There was a difference in the number of piles of Grant's gazelle dung found on each treatment ($F_{3,12}$=4.67, p=0.022). The G+M plot had higher numbers than the G, although it did not differ from the C or M (Fig.3.22; appendix C). Plotting camera trapping rate showed that G+M and C had high numbers of Grant's gazelle; although the difference between treatments appeared low. The number of piles of zebra dung did not differ between plots ($F_{3,12}$=1.11, p=0.38) and camera trap data seemed to concur with this (Fig.3.22).

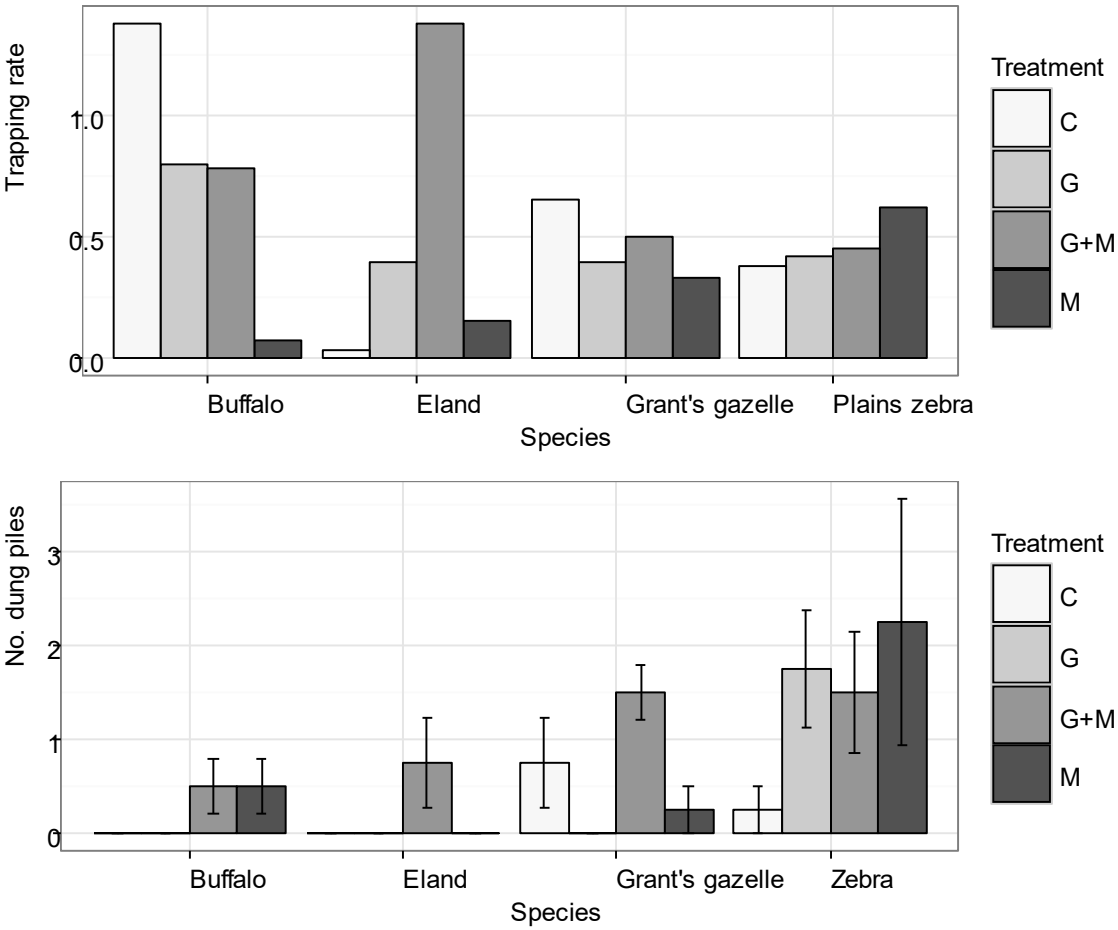

Figure 3.22. Mean trapping rate and mean number of dung piles ± S.E. of four focal species on the 2014 experimental plot.

3.2.3 Body size

There was no difference in trapping rate of small-bodied ($F_{2,6}$=1.04, p=0.41), large ($F_{2,6}$=0.13, p=0.88) or extra-large herbivores ($F_{2,6}$=1.11, p=0.39) between treatments on the 2015 plots (Fig.3.23). There was a difference in trapping rate of medium-bodied herbivores, made up of Plains and Grevy's zebra, oryx and hartebeest ($F_{2,6}$=5.37, p=0.046). TukeyHSD tests could not detect differences between treatments; however, there appeared to be a trend towards higher trapping rates on grazed plots (Fig3.23). Dung data showed no difference in number of dung piles between treatments for small ($F_{2,6}$=0.37, p=0.71), medium ($F_{2,6}$=1.29, p= 0.34) or large-bodied herbivores ($F_{2,6}$=0.57, p=0.60; Fig.3.23). There were a higher number of dung piles belonging to extra-large herbivores on the C plots, than on the G or M ($F_{2,6}$=10.5, p=0.011; Fig.3.23; appendix C).

Given the low number of dung piles found on the 2014 experimental plot, grouping species by body size or other attributes was not possible. Results simply reflected earlier findings based on Grant's gazelle and Plains zebra, the only two species contributing sufficient amounts of dung for analysis.

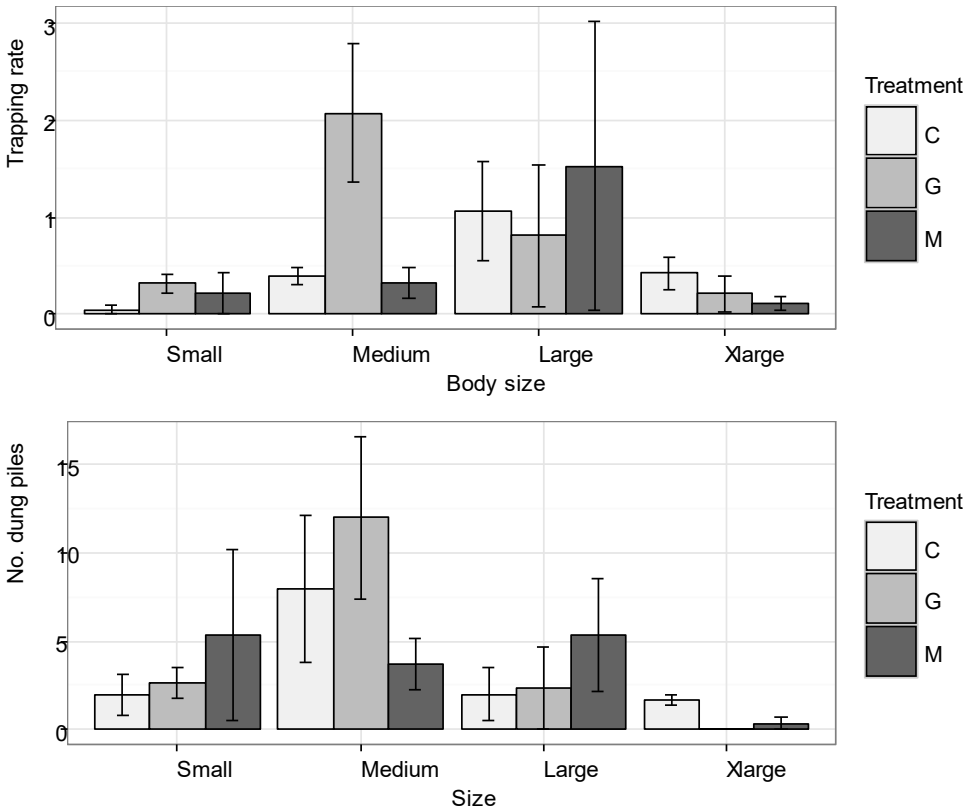

Figure 3.23. Mean trapping rate ± S.E. and mean number of dung piles ± S.E. of mammal species grouped by body size for the 2015 plots.

3.2.4 Digestive system

Grouping species by digestive strategy found no detectable difference in trapping rates of hindgut fermenters ($F_{2,6}$=4.58, p=0.06) or ruminants ($F_{2,6}$= 0.13, p=0.88) across treatments in the 2015 plots (Fig.3.24). Although, there may be a trend towards higher numbers of hindgut fermenters in G plots which could become apparent with more replicates. The same is true for dung transect data as numbers of hindgut fermenters ($F_{2,6}$=1.33, p=0.33) and ruminants ($F_{2,6}$=0.64, p=0.56) did not differ between treatments (Fig.3.24).

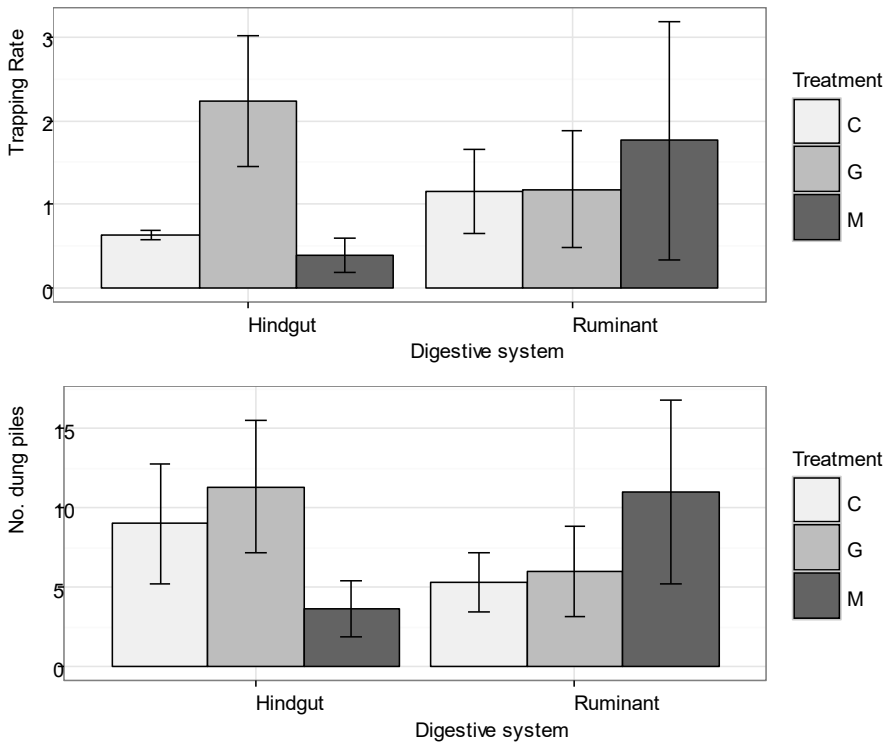

Figure 3.24. Mean trapping rate and mean number of dung piles ± S.E. of mammal species grouped by digestive system for the 2015 plots.

3.2.5 Animal behaviour

To investigate the behaviour of animals on the plots, the percentage of photos where animals were grazing was calculated. This showed no difference between the treatments for the 2015 plots ($F_{2,6}$=0.33, p=0.73; Fig.3.25).

Focussing specifically on Plains zebra showed that although trapping rates were higher for the grazed plots the percentage of photos where zebra were grazing did not vary between treatments ($F_{2,6}$=0.18, p=0.84). Furthermore, there was no difference in percentage of photos where zebra were walking ($F_{2,6}$=1.09, p=0.40). The same is true for eland which did not differ in percentage of photos grazing ($F_{2,6}$=0.19, p=0.83) or walking ($F_{2,6}$=0.05, p=0.95) between treatments. Therefore, although trapping rates differed between treatments for these species, this was not due to behavioural differences or preferential feeding on these plots.

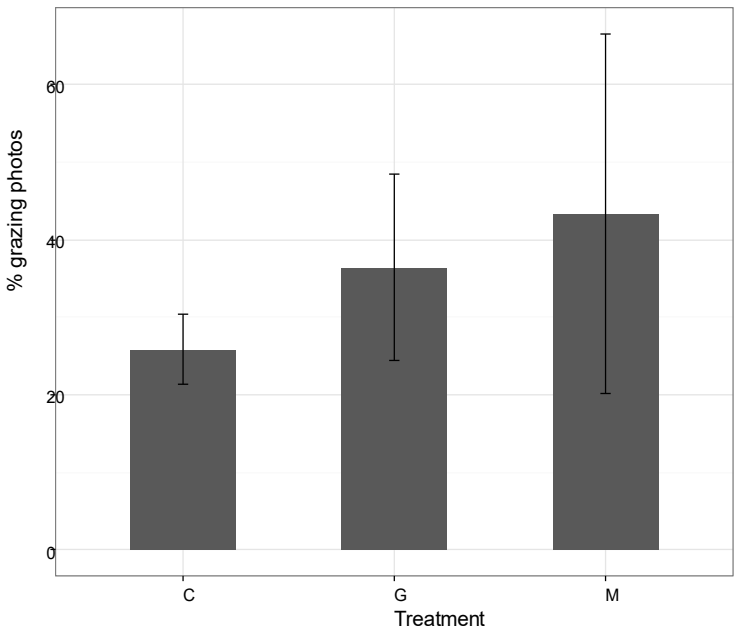

Figure 3.25. Mean percentage of photos where animals were grazing ± S.E. for 2015 plots.

3.2.6 Grass offtake

Using exclusion cages to measure offtake of grass by wild herbivores did not yield reliable results. Comparing plant growth inside the cages with the level of change outside should have been an indicator of grazing by herbivores. However, paired t-tests showed that there was no change in biomass inside the cages after 2 weeks for the C or G plots (C: t_8=0.85, p=0.42; G: t_4=0.35, p=0.75; Fig.3.26). M was the only treatment where an increase in biomass was observed, indicating grass growth inside the cage (t_8=2.81, p=0.02; Fig.3.26). Biomass outside the cages did not change over the two week sample periods (C: t_8=0.04, p=0.97; G: t_4=0.14, p=0.89; M: t_8=0.79, p=0.45; Fig.3.26). There was very large variance around the mean for the G plots (Fig.3.26), which is likely why no change could be detected for this treatment. The reason for this may have been the small sample size (n=5) having lost two cages from grazed plots due to human interference and elephant damage.

Although M plots showed an increase in biomass inside the cage, there was no difference between the change in biomass inside and outside (t_8=0.67, p=0.52). It was therefore not possible to judge offtake by mammalian herbivores from these results.

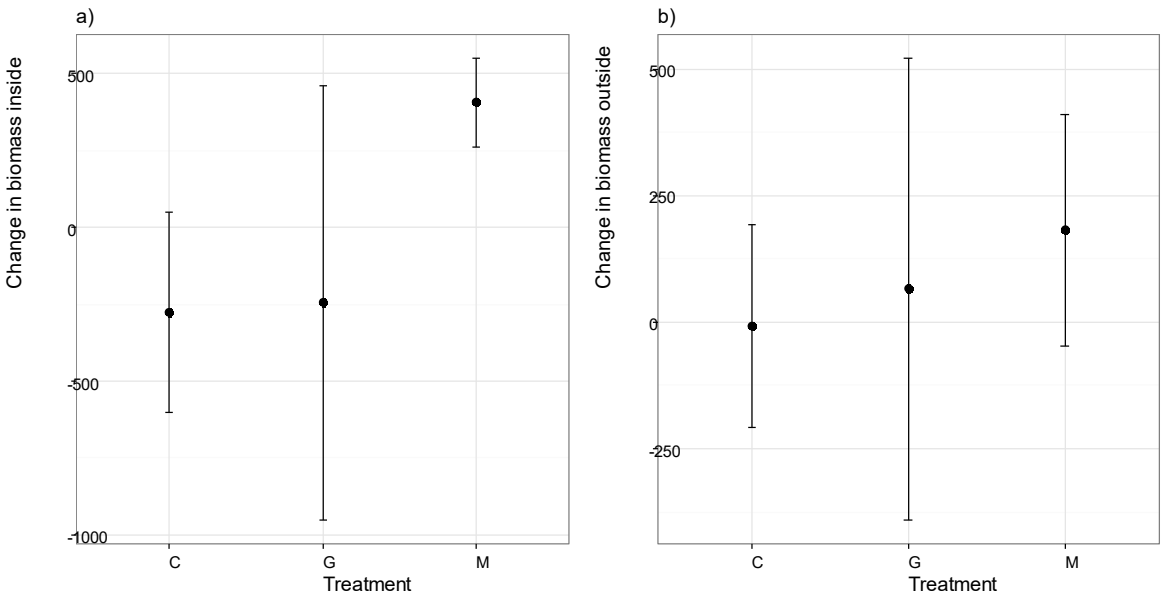

Figure 3.26. Mean change in biomass (kg/ha) ± S.E. inside (a) and outside (b) exclusion cages after two weeks.

4. Discussion

4.1 Vegetation survey results

Across all plots, the results indicated that while cattle grazing reduced grass biomass initially, it quickly returned to its original level. The reason for this swift recovery may in part be due to the cattle's foraging strategy. Generally, ruminants remove only the uppermost parts of plants and cattle graze only a proportion of the sward on offer (Woodward 1998; Gregorini *et al.* 2009), meaning that they avoid grazing into low sward that contains reproductive stems (Flores *et al.* 1993). The dominant grass species on LWC, *P.stramineum* and *P.mezianum*, are increaser species which typically grow close to the ground and possess physiological mechanisms that help them to recover from grazing (Trollope 1999). A plant's ability to recover from disturbance depends on its reproductive ability to re-establish leaves and renew photosynthesis (Hendrickson & Olsen 2006). Increaser species have high compensation abilities, resulting in a higher standing biomass after disturbance compared with species that are not tolerant to grazing (Del-Val & Crawley 2005). Interestingly, it appears that grazing results in a net increase in biomass, with biomass in grazed plots ultimately becoming higher than before any treatment was applied. This is the opposite of the intended aim of the management strategy, which is to remove large stands of unpalatable grass. *P.stramineum* and *P.mezianum* are types of increaser species which purportedly increase in yield when mildly grazed, not when heavily grazed (Trollope 1999), thereby suggesting that the grazing may not be intensive enough and the principles of holistic management are not being met.

In contrast, mowing did decrease biomass significantly across all plots and these low levels were sustained over time. Although information on the height of mowing is unavailable, it seems likely that the grass was mowed to below its apical meristem, the point from which primary growth occurs. Once the reproductive point is removed new growth must begin from dormant buds instead of continuing from the original stem, making regrowth slower and resulting in reduced biomass production over time (Turner & Seastedt 1993; Lemaire *et al*. 2000). Reducing biomass is important as field observations made on LWC since 1992 suggest that once the standing crop of grass exceeds 5000 kg/ha the sward is in a moribund condition and is of limited grazing value for both bulk and selective grazers (Trollope 1999). Digestibility and nutrient content are negatively correlated with grass height (Kleynhans *et al.* 2011) and studies suggest

that regular removal of above ground biomass is an important factor in long-term maintenance of species richness (Klimeš & Klimešová 2001).

This is evident on the 2014 experimental plot where G+M and M treatments had lower biomass than G and C and also higher levels of grass species diversity during the wet season. Mowing is believed to have an equalising effect on the competitive balance of species, allowing for regeneration of subdominant species during the growing season (Socher *et al*. 2013). Several studies have reported the benefits of mowing on plant species diversity (Collins *et al*. 1998; Yang *et al*. 2012; Socher *et al.* 2013) and this may lead to advantages for wildlife by providing more palatable and nutritious forage species.

Results suggest that cattle grazing does not increase grass species diversity. Studies of the effects of herbivores on plant diversity have produced mixed results: in some cases suggesting positive effects, while in others they are negative or neutral (McNaughton 1985; Hobbs & Huenneke 1992; Olff & Ritchie 1998). LWC is considered to be a nutrient-poor ecosystem, and there is some evidence that high grazing levels reduce species richness in these environments (Proulx & Mazumder 1998). However, it is clear that plant community composition is highly dependent on abiotic context, as evidenced by the large variation between the 2015 plots depending on their location. Environmental heterogeneity has been shown to modulate the effect of herbivores on plant communities (Young *et al.* 2013), making it difficult to interpret results from different sites.

At the 2015 plots differences in species diversity were explained by area of LWC. This suggests that there are underlying drivers of grassland structure and function which are more important than management regime. The interaction between herbivory and vegetation depends on many factors including climatic conditions, hydrology and topography and separating this background level variation from changes caused by management is very challenging (Kiage 2013). Plant community composition relates strongly to rainfall and soil properties (Young *et al.* 2013) and within LWC rainfall patterns, soil type and topography vary across the conservancy (appendix F). These are confounding factors when studying plots in different areas of LWC such as those that underwent management in 2015. The 2014 experimental plot is incredibly valuable as it allows for the study of the immediate effects of management regime on blocks which are adjacent to each other and subject to the same environmental variables. Consequently, while there is some evidence that mowing may increase species

diversity, results must be interpreted with caution as they come from only one site and effects of mowing may vary depending on location.

In addition, mowed plots typically had a lower percentage of standing vegetation cover than grazed and control which may have negative consequences such as increased water runoff and erosion (Gutierrez & Hernandez 1996). However, there is no difference in litter cover between treatments. Although attached plant matter is more effective at preventing runoff, litter is still valuable for protecting the soil, encouraging water retention and promoting nutrient cycling (Facelli & Pickett 1991). Nevertheless, if biomass is slow to recover in M plots the litter component may dissipate and thereafter result in erosion. However, all treatments had vegetation cover of >50%, which some authors suggest is enough to control runoff (Gifford 1985). Additionally, a decrease in vegetation cover may be beneficial as sward gaps have been shown to enhance seedling establishment and increase light availability, promoting germination of subdominant species (Bissels *et al.* 2006; Socher *et al*. 2013).

There was some evidence that G plots had a higher leaf:stem ratio than M. This was only observed in the wet season for the 2015 plots. As the wet season is the growing season this may be why this is the period when this became evident. A higher leaf:stem ratio indicates higher nutritive value and digestibility, given that leaves have a high ratio of protein and soluble carbohydrates to fibre, unlike the tougher stems (Gwynne & Bell 1968). Potentially, M had a lower ratio than G due to the slow grass recovery having been cut below its growth point. Cattle typically graze in the uppermost part of the sward and therefore select more leaves and avoid stems (Benvenutti *et al*. 2008). The higher leaf:stem ratio in grazed plots may consequently be a symptom of the increaser grass species compensating for their defoliation (Del-Val & Crawley 2005). Defoliation releases plants from the limitations imposed by old, dead tissue and stimulates increased growth rates and larger allocation to leaf growth (Oesterheld & McNaughton 1990). However, as neither G nor M leaf:stem ratios differed from C it appears that the effect of management is negligible. At the 2014 site there was some indication that the G+M plot had a higher leaf:stem ratio in the wet season than the G or C. This could suggest that, after a recovery period, mowing can improve leaf:stem ratio as the grass begins to compensate for its defoliation.

There was also an effect of treatment on the overall proportion of green:brown plant material at the 2015 plots. Both the G and M plots had a higher proportion of green vegetation than the C. Brown plant matter is a measure of moribund resource which is

of low quality for grazers (Trollope 1999). A higher proportion of green, photosynthetically active material indicates healthier, more nutritious grassland. The grassland management strategies may therefore be succeeding at improving the quality of grassland and stimulating new growth. However, the overall level of green vegetation remained low, with ratios <1 indicating that brown vegetation dominated all sites. Additionally, within two years, as indicated by the 2014 plots, the proportion of green:brown vegetation had returned to its original state.

Overall, results of the vegetation surveys appear to indicate that cattle grazing in its current form is not an effective method for improving the grassland of LWC in the long term. Although there may be some initial benefits for leaf:stem ratio and increased green vegetation, the grassland recovers quickly. Within one year biomass reaches levels higher than before treatment was implemented and within two years leaf:stem ratio and green:brown plant material have returned to their original state. There is also no effect on species diversity. The evidence suggests that mowing may be a more successful strategy for reducing biomass and increasing species diversity.

4.2 Wildlife use results

Results from camera traps and dung surveys indicated that grassland management regimes did not have any effect on the diversity of mammal species found on the plots or the number of wildlife events that occurred.

At the 2014 experimental plot, dung data indicated a possible preference by Grant's gazelle for the G+M plot over the G plot. This treatment had lower biomass, higher grass species diversity and some indication of higher green leaf:stem ratios than the grazed plot. Therefore this finding corresponds with previous studies suggesting that small-bodied ruminants select areas of short, high-quality grass, due to their high nutritional requirements (Hopcraft *et al.* 2012). In addition, gazelle prefer short grasslands due to the high-visibility allowing for increased protection from predator ambush (Fynn *et al.* 2016).

There was some evidence that Plains zebra preferred grazed plots over controls for the 2015 treatments. Zebra are considered to be non-selective roughage grazers and have been shown to forage in intermediate height grass swards (Voeten & Prins 1999) likely due to the high food intake requirements of hindgut fermenters. Zebra select feeding sites with higher digestibility (Voeten & Prins 1999), and have a higher intake of leaves in their diet during the wet season (Gwynne & Bell 1968). Therefore they may

be attracted to the increased green vegetation and leaf:stem ratios at G sites. However, analysis of behaviour in camera traps did not find that zebra were grazing more on G plots than on other treatments. In addition, there was no difference between Plains zebra trapping rate on G and M plots, suggesting that they do not show a preference for higher grass biomass over short grass swards. Data suggested that two of the grazed plots had very high numbers of zebra in comparison to the remaining grazed plot in the centre of LWC. An alternative reason for the high numbers of zebra on these plots may be that they were situated within their core range; with home range sizes depending on resource availability and group composition (Ransom & Kaczensky 2016). Additionally, field observations indicated that Block 28 was situated on a regular route to water, with large herds of zebra moving across it at specific times each day. Equids are dependent on water sources and water strongly influences movement patterns and distribution (Ransom & Kaczensky 2016). It is therefore difficult to determine if the results indicate an active preference for a grazed area. At the 2014 experimental plot there did not appear to be any preference for the grazed treatment, but this is in accordance with the vegetation data showing that the grazed plot did not differ from the control.

Results also suggested that eland and extra-large herbivores prefer control plots. The extra-large grouping refers solely to elephants in this instance, as no rhino dung was found on any plot. This corresponds with both eland and elephant ecology in that they are able to feed on courser, less palatable forage (Kingdon 2015). Elephants in particular prefer longer grass as they are bulk feeders and are unable to feed on short grass due to a minimum height being required in order to grasp it with their trunk (Sinclair & Arcese 1995).

Unlike previous studies, grouping by body size and by digestive system did not indicate any differences in herbivore use of the plots (Illius & Gordon 1987; Arsenault & Owen-Smith 2002; Kleynhans *et al*. 2011). Although there was some evidence of a difference for medium-bodied herbivores, this is conceivably attributable to the response of Plains zebra, which made up the majority of this group due to very low numbers of oryx and Grevy's zebra. The reason for the apparent lack of preference for a particular treatment is likely due to differences between species within the groups.

Feeding behaviour is mediated by several factors including the animals' anti-predator strategy, body size, digestive strategy, mouth width and seasonal differences in food availability (Kleynhans *et al.* 2011). One study found that grass height preferences of different herbivores did not follow the pattern expected based on body size but

instead were due to the relationship between body size and mouth width (Arsenault & Owen-Smith 2008). While larger herbivores are able to tolerate tall, less nutritious grass, they may feed more efficiently on shorter grass patches if their mouth morphology is suited to taking wide, shallow bites (Fynn *et al*. 2016). Small herbivores with narrow mouths can feed efficiently in tall grass by plucking leaf components from surrounding stems (Arsenault & Owen-Smith 2008). This combination of factors affecting feeding behaviour means that habitat preferences are highly species-dependent and explains why no trends were observed when grouping animals broadly by body size or digestive system.

Interpreting offtake of grass by herbivores using exclusion cages was not possible. This was due to there being no detectable change in biomass inside the cages for the majority of the plots. With no comparison to normal growth rate inside the cage, it was difficult to discern how much the grass outside had been grazed. The set of cages in mowed plots did show an increase in grass biomass inside. However, the change in biomass inside and outside did not differ. A possible reason for the failure of this methodology is that the time restraint resulted in cages not being left in place long enough for a measurable change to occur. In order to achieve enough replicates for data analysis, cages were left at one site for two weeks. Grass growth, particularly in the dry season, may be negligible over this length of time. Additionally the disc pasture meter is not a fine scale measure, meaning it may not be precise enough to detect small changes. In future, use of this method to investigate grass offtake on LWC would require a longer study period.

Overall, the wildlife use data must be interpreted with caution. Although there was some suggestion that distribution of particular species was affected by treatment, it is unclear whether this is related to changes in vegetation as wildlife were not behaving differently or feeding preferentially on different treatments. Other aspects such as location of water sources, home ranges, topography or predator avoidance may be influencing herbivore distribution. In general, large variation between blocks was evident. Unfortunately due to equipment limitations there was a low sample size, particularly for the 2014 experimental plot where camera trap data could not be analysed statistically due to a lack of replicates. Dung transects were unreliable as very small amounts of dung were found and this method was less effective at detecting presence of species than camera traps.

4.3 Extension of this research

A significant limitation of this study was the random distribution of the grazed and mowed blocks on LWC and the large variation between blocks in different locations. Although efforts were made to select plots with similar environmental attributes this was not completely achieved (appendix A); meaning that these plots may have differed in vegetation composition and amount of wildlife present due to factors other than the management regime. A longer term and more controlled investigation would be beneficial for understanding these factors.

Further experimental areas, such as that set up in 2014, would be incredibly valuable, as implementing treatments adjacent to each other controls for many of the differing environmental variables. In addition, if grazing is to continue across multiple sites, it would be advantageous to create a small control area at each location. This would allow for direct comparison between the treated area and its original state. Alternatively, completing vegetation transects before and after the grazing or mowing would allow for a better understanding of how the treatment has affected each individual plot.

Continued monitoring of wildlife use would be highly worthwhile and would result in a larger dataset that could be used to aid understanding of the effects of management type on wildlife. Dung transects are not recommended due to their inefficiency at detecting species presence. However, further and more extensive camera trap surveys would be useful for discovering which areas are being used most regularly. This could be supplemented with direct observations to determine how animals are behaving on plots.

Another area of management which may warrant investigation is the impact and recovery of cattle boma sites. Overnight cattle corrals (bomas) are rotated every 7 days during the dry season and every 1-3 days during the wet season (Schulz *et al.* 2014). Traditional bomas remain in position for several months and the accumulated dung layer results in the development of highly-productive glades that are dominated by palatable rhizomatous plant species and used preferentially by wildlife (Veblen & Young 2010). Modern methods using short-term mobile bomas leave large scars on the landscape and it is unclear if cattle have been concentrated long enough for conditions to allow for development of a nutrient-rich glade. Although one study has shown that short-term bomas do initiate major changes in nutrient concentrations and develop in to

wildlife hotspots (Porensky & Veblen 2015) it is unclear whether LWC's nutritionally poor landscape will respond in the same way. Monitoring of these boma sites and their recovery would be advisable.

4.4 Management implications and recommendations

With regard to the continued management of LWC, it must be undertaken cautiously and with ongoing monitoring. The management practices to be used should be decided carefully based on what the exact aim of the plan is.

If the primary aim of the management is to benefit the wildlife and improve the forage availability it is unclear whether the current strategies are effective. There was some suggestion that certain species were found more often on particular plots. Yet, neither treatment was shown to be grazed preferentially by herbivores. The large variation in the wildlife use results means it is uncertain whether the management is influencing wildlife use of particular areas. However, the treatments did result in changes in vegetation structure and characteristics and do produce a diversity of habitats for wildlife. Converting entire landscapes to one dominant structural state means they will lack the diversity to sustain multiple herbivore guilds or provide resilience during drought (Fynn *et al.* 2016). Therefore it is important to maintain a variety of habitat types on LWC and both mowing and grazing may assist with this.

Mowing should be used preferentially if the aim is to reduce biomass of dead grass in the long term. In contrast, livestock grazing appears to be having the reverse effect and is resulting in a net increase in grass biomass.

Excluding the net increase in biomass, cattle grazing does not appear to be having a negative effect on the grassland in terms of vegetation characteristics. Thus, it may still be a useful tool on LWC as a method of community outreach and to provide grazing concessions for local people (Fynn *et al.* 2016). However, as this grassland is considered to be nutritionally poor, adding additional grazers may result in competition with wildlife for limited resources. In conditions where forage is limited, cattle have been shown to compete with wildlife and in particular show evidence of competition for the grass species *Pennisetum stramineum* during the dry season (Voeten & Prins 1999; Odadi *et al.* 2011). Additionally, it has been suggested that competition is greater at sites with lower productivity (Pringle *et al.* 2007). It is therefore recommended that should cattle grazing continue on LWC it must be done with caution, and that if the

primary aim is to improve the grassland, rather than promote community outreach, this method may not be effective.

The increases in biomass in grazed areas may be indicative of a failure to meet the requirements of holistic grazing. *Pennisetum* species increase in yield when mildly grazed, suggesting that grazing may not be intensive enough. Previously LWC has failed to meet the principles of holistic management due to inconsistent stocking rates and lack of capacity to manage herds in a controlled manner (Schulz *et al.* 2014). Further investigation would be needed to determine if these problems still remain and whether adjusting the grazing regime would yield different results. Although, as stated, increasing numbers of cattle may be detrimental if it results in increased competition with wildlife.

Mowing is an effective technique for managing the grassland as it results in a decrease in biomass, potential increases in species diversity at some sites and an increase in green vegetation. This method should also be undertaken cautiously with further monitoring in order to determine the appropriate height, frequency and season that mowing should take place. Different grass species exhibit different tolerance to mowing based on characteristics such as height and growth form (Klimeš & Klimešová 2001; Fynn *et al.* 2005). Strategic management and planning is required to ensure successful grassland management.

4.5 Conclusions

The current cattle grazing strategy does not appear to be an effective tool for improving the vegetation structure and quality of LWC's moribund grasslands in the long term. Although initially it results in increased green vegetation and leaf:stem ratio, within one year biomass has reached levels higher than before treatment has taken place. The initial benefits therefore exist only briefly and to maintain them would require frequent re-grazing, leaving limited time in which they may benefit wildlife. Mowing is effective at removing biomass over the long term and increasing grass species diversity.

LWC's cattle grazing regime does not result in the advantages to the grassland that have been advocated by proponents of holistic management and grazing as a conservation intervention does not appear to be appropriate in this ecosystem. It may however, still be possible to use cattle grazing as an outreach strategy to promote community involvement and encourage tolerance for wildlife.

The grass composition found on LWC, consisting of decreaser *T.triandra* and climax communities of increaser *P.stramineum* and *P.mezianum*, occurs across a wide range of highlands in Kenya and Tanzania (Trollope 1999). Results of this study may therefore be applicable at a wider scale, suggesting that in these areas cattle grazing is not an effective strategy for conserving the grassland or harmonising the needs of wildlife and pastoralists. However, outcomes of management are likely to be highly site-specific and require consideration of many factors including soil type, rainfall and recovery periods.

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Appendix A

Table A. Summary of surveyed plots and their attributes

Block	Treatment	Location on LWC	Date of treatment	No. rainy seasons since treatment	Grazing intensity: (No.cows/ Block size)*ADAs	Soil type	Trees and shrubs	Distance to water (approx.)	Notes
55	Grazed	East	July - Sept 2015	1	31	Mixed	Some	200m–1km	
28	Grazed	West	May - July 2015	1	34	Mixed	Few	1-3km	Surrounded by hills
33	Grazed	Central	June - July 2015	1	23	Mixed	Few	1-3km	
63	Mowed	East	June 2015	1	-	Mixed	Few	200m-1km	At base of hill
37	Mowed	West	Feb 2015	2	-	Mixed	None	200m-1km	On top of hill
7	Mowed	Central	Aug - Oct 2015	1	-	Black cotton	None	1-3km	
57	Control	East	-	-	-	Mixed	Some	1-3km	
24	Control	West	-	-	-	Black cotton	Few	1-3km	Ground is rocky in places
14	Control	Central	-	-	-	Black cotton	None	1-3km	

Appendix B

Table B. List of herbivore species grouped by body size

Scientific name	Common name	Size	Feeding strategy	Digestive system
Aepyceros melampus	Impala	Small (40-70kg)	Mixed	Ruminant
Nanger granti	Grant's gazelle	Small (40-80kg)	Mixed	Ruminant
Oryx beisa	Beisa oryx	Medium (120-210kg)	Mixed	Ruminant
Alcelaphus buselaphus	Hartebeest	Medium (130-230kg)	Grazer	Ruminant
Equus grevyi	Grevy's zebra	Medium (350-450kg)	Grazer	Hindgut
Equus quagga	Plains zebra	Medium (200-260kg)	Grazer	Hindgut
Syncerus caffer	African buffalo	Large (500-700kg)	Grazer	Ruminant
Tragelaphus oryx	Common eland	Large (400-900kg)	Mixed	Ruminant
Ceratotherium simum	White rhinoceros	Extra-large (1700-2300kg)	Grazer	Hindgut
Loxodonta africana	African elephant	Extra-large (3000-6000kg)	Mixed	Hindgut

Appendix C

Table C1. Mean ± standard error of current biomass for plots that underwent management in different years. Different letters denote significant differences between treatments (TukeyHSD).

	Control	Grazed 2014	Grazed 2015	Grazed 2016	Mowed 2014	Mowed 2015
Biomass (kg/ha)	$5418^{a} \pm 183$	$5458^{a} \pm 111$	$5564^{a} \pm 117$	$4042^{b} \pm 116$	$3624^{b} \pm 140$	$4022^{b} \pm 205$

Table C2. Mean ± standard error of vegetation characteristics that differed significantly between areas of LWC (one-way ANOVA). Different letters denote significant differences between areas (TukeyHSD).

	Season	Central	East	West
Species diversity (Shannon index)	Dry	$0.86^{a} \pm 0.08$	$0.79^{a} \pm 0.10$	$1.17^{b} \pm 0.10$
	Wet	$0.66^{a} \pm 0.10$	$1.07^{b} \pm 0.10$	$1.37^{c} \pm 0.06$
Leaf:stem ratio	Dry	$2.23^{a} \pm 0.26$	$0.78^{b} \pm 0.15$	$1.44^{b} \pm 0.17$

Table C3. Mean ± standard error of species trapping rates/no. dung piles that differed significantly between treatments (one-way ANOVA). Different letters denote significant differences between treatments (Tukey HSD).

	Data type	Control	Grazed	Mowed	Grazed + Mowed
		2014 experimental plot			
Grant's gazelle	Dung	$0.75^{ab} \pm 0.48$	0^{a}	$0.25^{ab} \pm 0.25$	$1.50^{b} \pm 0.29$
		2015 plots			
Plains zebra	Camera	$0.10^{a} \pm 0.03$	$1.94^{b} \pm 0.70$	$0.25^{ab} \pm 0.13$	
Eland	Camera	$0.39^{a} \pm 0.08$	$0.08^{b} \pm 0.07$	$0.04^{b} \pm 0.02$	
X-large species (elephant)	Dung	$1.67^{a} \pm 0.33$	0^{b}	$0.33^{b} \pm 0.33$	

Appendix D

Table D. List of all identified grass species found during the study (Ibrahim & Kabuye 1987; Trollope 1999)

Scientific Name	Common name	Category*	Habitat	Notes
Aristida adoensis	Mountain needle grass	II	Bushland/open grassland/overgrazed areas	Grazed when young, unpalatable at seeding
Aristida adscensionis	Annual bristle grass	II	Overgrazed/degenerated areas	Grazed when young, unpalatable at maturity
Arisitda kenyensis	Kenya needle grass	II	Bushland/dry or eroded soils	Grazed when young
Cynodon dactylon	Common star grass	II	Unsettled/mildly overgrazed areas	Variable forms with different grazing values
Cynodon nlemfuensis	Naivasha star grass	II	Bushland/old cultivated land	Valuable for grazing
Digitaria ciliaris	-	-	Disturbed ground	No significance for grazing
Digitaria milanjiana	Wooly finger grass	II	Grassland/black or sandy soils	Excellent grazing grass
Digitaria scalarum	African couch grass	II	Disturbed ground	Valuable for grazing
Digitaria swazilandensis	Swaziland finger grass	II	Disturbed ground/lawn grass	-
Digitaria ternata	-	-	Disturbed ground	Grazed but low forage production
Enneapogon cenchroides	-	-	Bushland/semi-arid grassland/overgrazed areas	Low grazing value
Michrochola indica	-	-	Open grassland	No significance for grazing
Pennisetum mezianum	Bamboo grass	I	Semi-arid grassland/Open bushland	-
Pennisetum stramineum	Wire grass	I	Bushland	Grazed when young, low palatability at maturity
Sporobolus agrostoides	-	II	-	-
Themeda triandra	Red oat grass	D	Open grassland/varied soils	Valuable for grazing

*D: species which decrease in yield when under or over utilised; I: species which increase in yield when lightly grazed; II: species which increase in yield when moderately or heavily grazed.

Appendix E

Table E. List of all mammal species captured on camera traps (output from ZSL-CTAT)

Family or Subfamily	Scientific name	Common name	IUCN Status*	Habitat	Trophic level
Canidea	*Canis adustus*	Side-striped jackal	LC	Woodland	Carnivore
Felidae	*Acinonyx jubatus*	Cheetah	VU	Mixed	Carnivore
Felidae	*Panthera leo*	African lion	VU	Mixed	Carnivore
Hyaenidae	*Crocuta crocuta*	Spotted hyaena	LC	Mixed	Carnivore
Bovidae	*Aepyceros melampus*	Impala	LC	Savanna	Herbivore
Bovidae	*Alcelaphus buselaphus*	Hartebeest	LC	Grassland	Herbivore
Bovidae	*Nanger granti*	Grant's gazelle	LC	Grassland	Herbivore
Bovidae	*Oryx beisa*	Beisa oryx	NT	Scrubland	Herbivore
Bovidae	*Syncerus caffer*	African buffalo	LC	Mixed	Herbivore
Bovidae	*Tragelaphus oryx*	Common eland	LC	Mixed	Herbivore
Giraffidae	*Giraffa camelopardalis*	Giraffe	LC	Savanna	Herbivore
Suidae	*Phacochoerus africanus*	Common warthog	LC	Mixed	Omnivore
Equidae	*Equus grevyi*	Grevy's zebra	EN	Grassland	Herbivore
Equidae	*Equus quagga*	Plains zebra	LC	Grassland	Herbivore
Rhinocerotidae	*Ceratotherium simum*	White rhinoceros	NT	Savanna	Herbivore
Rhinocerotidae	*Diceros bicornis*	Black rhinoceros	CR	Scrubland	Herbivore
Cercopithecidae	*Papio anubis*	Olive baboon	LC	Savanna	Omnivore
Elephantidae	*Loxodonta africana*	African elephant	VU	Mixed	Herbivore
Orycteropodidae	*Orycteropus afer*	Aardvark	LC	Mixed	Insectivore

*LC: least concern; NT: near threatened; VU: vulnerable; EN: endangered; CR: critically endangered.

Appendix F

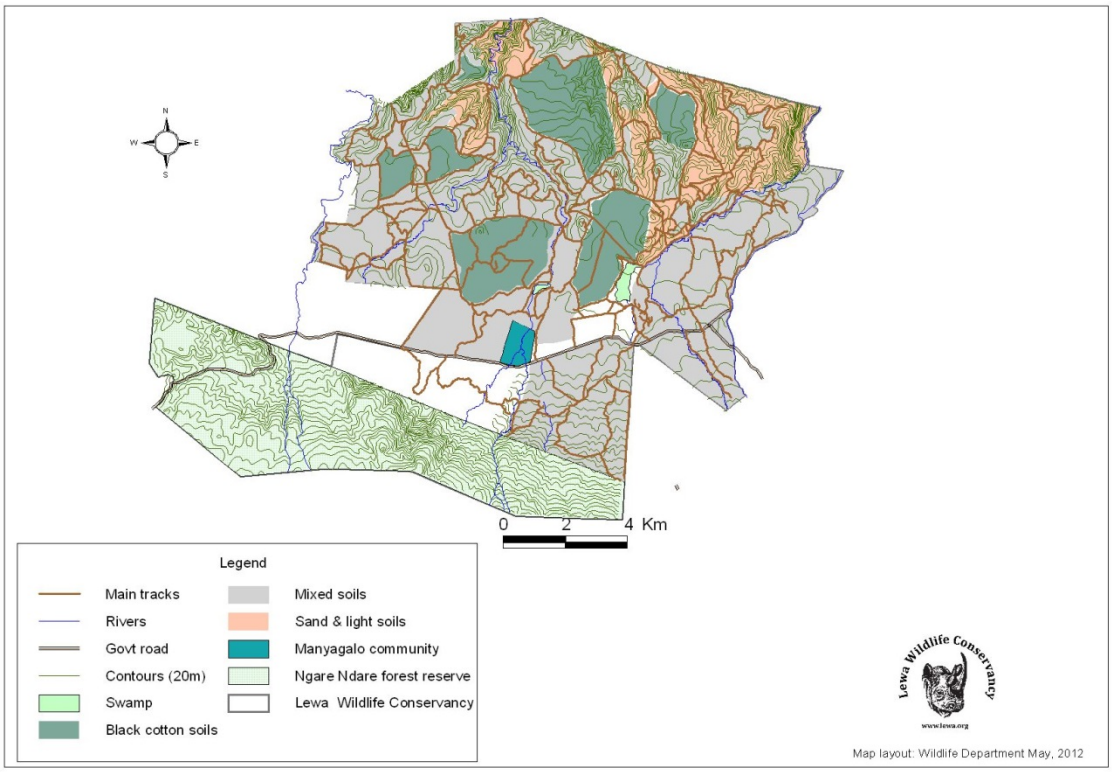

Figure F1. Map of soil types on LWC. *From* LRD.

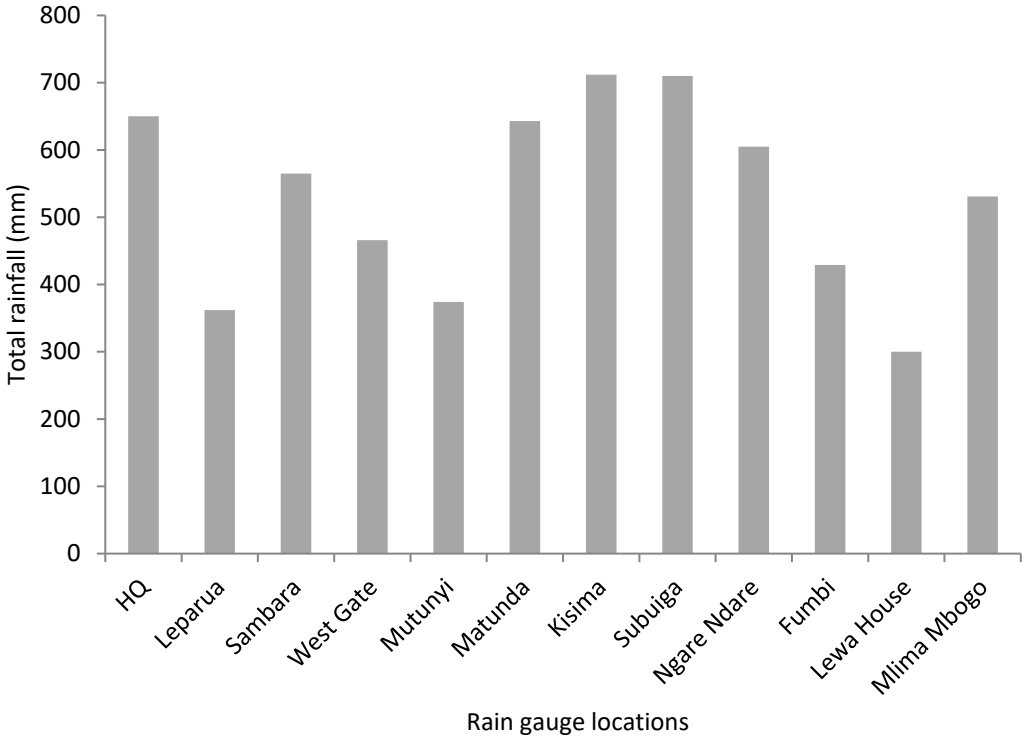

Figure F2. Total rainfall across multiple locations on LWC in 2015. Rain gauges are stationed at gates and guest houses. *From* LRD.