

**HABITAT SELECTION AND DISTRIBUTION OF SOUTHERN  
WHITE RHINOCEROS (*Ceratotherium simum simum*, Burchell 1817)  
TRANSLOCATED TO OL PEJETA CONSERVANCY, KENYA**

**BY**

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REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY  
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KENYA**

**2014**

## DECLARATIONS

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University of Eldoret, Kenya

## DEDICATION

To those in conservation who strive to use the environment responsibly

for

When God delegated some of His authority (“be masters over”) to the human race, He expected us to take responsibility for the environment and the other creatures that share our planet. We must not be careless and wasteful as we fulfil this charge (Life Application Study Bible).

God blessed them and told them, “Multiply and fill the earth and subdue it. Be masters over the fish and birds and all the animals”  
(Genesis 1:28).

## ABSTRACT

Translocation has been used extensively as a management tool in rhinoceros conservation. However, information on post release habitat preference and home range size has been modest. Thus six southern white rhinoceros, three females and three males, translocated to Ol Pejeta Conservancy in Laikipia District of Kenya were studied between July 2006 and December 2008. The study was aimed at assessing the post release habitat preference and home range size of these rhinoceros by quantifying habitat quality of six study sites namely; Morani, Zebra, Oryx, Serat, Grants and Loldru. Study site quality was determined by estimating grass biomass, quantifying the relationship between normalised difference vegetation index values and faecal nitrogen while study site preference and estimates of home ranges were established by recording GPS locations of each or group of rhinoceros sighted. Study site preference index was calculated by comparing the proportion of white rhinoceros occurrence in each study site and the proportion of the study site sizes within the study area. A disc pasture meter was used to estimate grass biomass. The mean grass biomass for Ol Pejeta Conservancy was predicted to be  $700 \pm 434$  kg/ha at a mean disc height of  $7.51 \pm 0.35$  cm with the highest being in Loldru site ( $642 \pm 169$  kg/ha) and the lowest in Oryx site ( $266 \pm 106$  kg/ha). A significantly positive linear correlation between faecal nitrogen and the normalized difference vegetation index values was recorded in Zebra site ( $r = 0.997$ ;  $p < 0.001$ ), Oryx site ( $r = 0.996$ ;  $p < 0.001$ ) and Morani site ( $r = 0.714$ ;  $p < 0.001$ ) while a significantly negative linear correlation ( $r = -0.9$ ;  $p < 0.001$ ) was recorded in Loldru site. Zebra site was the highest utilized (18%) site when all the seasons were considered simultaneously while Grants site was the highest utilised site in both wet (19%) and dry (52%) seasons. In terms of site preference, Zebra site was the most preferred (+1) when all the seasons were considered simultaneously while Oryx and Serat sites were the most preferred (+1) in wet and dry seasons respectively. During post release monitoring phase, Females exhibited larger home ranges ( $22-25.4\text{km}^2$ ) than males in the wet season and smaller home range ( $28.6\text{km}^2$ ) than Males ( $29.7\text{km}^2$ ) in the dry season. Zebra site exhibited high quality forage and was consequently the most utilised and preferred site, hence the core areas of all rhinoceros studied were within this site. It is concluded that the post release distribution of these rhinoceros followed the principles of the Ideal Free Distribution theory, and the Zebra site within Ol Pejeta Conservancy is important for the conservation of these rhinoceros. The study recommends active management of these sites and a study on the role of water in distribution of white rhinoceros should further translocation of white rhinoceros be considered.

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**ABBREVIATIONS AND ACRONYMS**

ASAL	:	Arid and Semi Arid Land
AVHRR	:	Advanced Very High Resolution Radiometer
CETRAD	:	Centre for Training and Integrated Research in ASAL Development
DPM	:	Disc Pasture Meter
EMU	:	Ecological Monitoring Unit
GIS	:	Geographical Information System
GPS	:	Global Positioning System
IFD	:	Ideal Free Distribution
IUCN	:	International Union for Conservation of Nature
KWS	:	Kenya Wildlife Service
MODIS	:	MODerate-resolution Imaging Spectroradiometer
MCP	:	Minimum Convex Polygon
NASA	:	National Aeronautics and Space Administration
NDVI	:	Normalised Difference Vegetation Index
NIR	:	Near Infra Red
OPC	:	OI Pejeta Conservancy
PI	:	Preference Index

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## CHAPTER ONE

### INTRODUCTION

#### 1.1 Background to the study

Translocation is the deliberate and mediated movement of wild individuals or populations from one part of their range to another (IUCN, 1995; Emslie *et al.*, 2009) thus providing them with a new home. This is done for several reasons including for example; to improve the chances of survival of a species, to establish new populations, to reintroduce a species back into its former range where it had been eliminated, to keep established populations productive (that is at or below estimated maximum productivity carrying capacity), introduce new genetic traits into a population, or to restock suitable habitats without sufficient natural corridors between them and thus natural dispersal may no longer take place (IUCN, 1995; Emslie and Brooks, 1999).

One of the basic principles of translocation involves taking a species to a suitable habitat that is better or at least equal to where the animal is being removed. In the case of animal species, post release monitoring to determine habitat preference, estimate home range sizes and general adaptation of the translocated animal is essential. Thus species undergoing translocation for any reason offer a unique opportunity to understand their specific niche requirements, more easily, than long established populations because they are likely, if sufficiently mobile, to colonise high quality habitats first.

This expectation is the foundation of the Ideal Free Distribution (IFD) theory which suggests that all high quality habitats will be preferentially used until the

moment that species density is so high that the low quality habitat becomes at least as attractive (Fretwell and Lucas, 1970; Fretwell, 1972). The aspects of IFD theory thus brings into consideration the importance of knowledge on wild herbivore habitat quality when considering its translocation, as habitat quality is likely to influence the distribution of the animals. In addition, estimation of indices of habitat quality such as the amount and nutrient contents of the above ground plant biomass become essential when assessing the suitability of a site as a herbivore habitat. Therefore, post release monitoring of translocated animals could provide information that has important consequences for feeding ecology, social organization, mating strategies and conservation of the species (White *et al.*, 2007).

While simple in concept, translocation as a tool in wildlife management remains controversial (Van Zant and Wooten, 2003). Typical controversy focuses on the lack of long-term quantitative information on the post-release impact (Scott and Carpenter, 1987; Linklater *et al.*, 2006; Patton *et al.*, 2010), difficulty in establishing the success or failure criteria (Seddon, 1999), and concerns that extensive gene flow can interfere with local adaptations (Storfer, 1998). In Africa, most translocations involve animals from the wild. Since these wild populations are important to the long-term recovery strategies especially for endangered species, it is important to consider factors affecting the persistence of the wild populations translocated to a new habitat (Todd *et al.*, 2002). Fundamentally then, issues related to habitat quality, preference, and potential home range sizes in the new habitat need to be taken into consideration prior to selecting a new home to introduce an animal.

The southern white rhinoceros (*Ceratotherium simum simum*) is not indigenous to Kenya having been introduced in 1965 from Hluhluwe-Imfolozi National Park, South Africa (Okita-Ouma *et al.*, 2007). These consisted of the initial 6 southern white rhinoceros that were taken to Meru National Park, followed by 45 more that were translocated between 1970 and 1994 from Hluhluwe-Imfolozi National Park to Solio, Lewa and Lake Nakuru National Park. By the end of 2008, over 300 southern white rhinoceros were estimated to exist in Kenya spread over 15 national parks, private and community wildlife sanctuaries (Okita-Ouma *et al.*, 2007). The continued stay of this out-of-range species in Kenya is justified as the southern white rhinoceros has largely been used for tourism given that it has certain characteristics that make it very ideal for tourism (Bothma, 1996). These characteristics include; tendency to occur in small groups, and hence can be kept on smaller areas and in higher numbers thus making them relatively cheaper to maintain per given acreage, their preference to open wooded savannas that facilitates viewing them, their territorial nature, thus tendency to occur in a specific area, therefore tourists on hiking trails can track them easily as they are also less aggressive.

Although the southern white rhinoceros does not qualify for inclusion in the threatened category of the IUCN Red List, and thus rated as Near Threatened (Brooks, 2008), it was the focal animal for this study since it is still the focus of continuing taxon-specific conservation programmes, and its survival is still considered conservation dependant due to the continued poaching threat and high illegal demand for rhinoceros horn (Emslie and Brooks, 1999; Brooks, 2008).

Similarly, requests for translocating white rhinoceros across conservation areas in Kenya for tourism purposes are increasing. Consequently southern white rhinoceros translocated to Ol Pejeta Conservancy between July 2006 and December 2007 from Lewa and Kigio Wildlife Conservancies, as a breeding nucleus for this species and to enhance tourism in the long term were studied between July – December 2006 (post release settling phase) and January 2007 - December 2008 (post release monitoring phase).

## **1.2 The problem statement**

Since the introduction of the southern white rhinoceros in Kenya in 1965 data collection on this species in all the national parks, private and community wildlife sanctuaries conserving it, has been limited to sightings and records of births and deaths (Mulama and Okita-Ouma, 2004). Analysis of this information is also limited to sighting frequency and annual net numbers but has not involved aspects of habitat preference and/or post release home range sizes of this species. The fact that this species is not indigenous to Kenya could be the reason for lack of documentation of its habitat preference and distribution. Even in private and community wildlife sanctuaries, some of which never recorded births since introduction of the white rhinoceros, such as Mugie, Delta Crescent Farm, Kigio and Mt. Kenya Wildlife Conservancy, no studies had been undertaken to assess if the poor reproduction records was related to the habitat status. These sanctuaries have since been closed down as they no longer have rhinoceros on them. No birth had been recorded in these sanctuaries even when a pair of male-female was resident on each property. This means that post release monitoring was and is still not done effectively across the national parks, private and community sanctuaries



conserving white rhinoceros and as a result, information on adaptation of this species to its new habitat is not available in Kenya. Despite the poor birth record in 57% of the parks and sanctuaries with white rhinoceros (KWS, 2010), translocation of white rhinoceros in Kenya has continued without considering the aspects of habitat quality, preference and potential home range sizes in their new habitats, unlike that of the black rhinoceros, which has explicit pre- and post-translocation protocols. Thus the effects of habitat quality on habitat preference and home range size of the white rhinoceros in a new habitat are not well understood in Kenya.

Due to lack of translocation protocols for white rhinoceros in Kenya and the information gap on its habitat preference and potential home range size, white rhinoceros translocations in Kenya are guided mainly by requests from different stakeholders involved in wildlife conservation, availability of the white rhinoceros and funds to undertake the translocation. In the process, the underlying objectives of rhinoceros translocations have rarely been achieved and the reasons for the failure of the translocated white rhinoceros to adopt and successfully breed in their new habitat remain an area of conservation concern.

### **1.3 Rationale of the study**

Information on successful adaptation of animal species translocated to a new habitat and their movement and distribution in their new habitat is a fundamental requirement for setting effective recovery strategies for any endangered species or implementing management interventions, more so, for species that are being conserved out of their natural range. This information is equally important to

justify the continued use, review or discontinue use of some wildlife management intervention strategies for some animal species. The translocation of the southern white rhinoceros to Ol Pejeta Conservancy in 2006 and subsequent post release monitoring provided an opportunity to understand their post release adaptation. Upon release, the rhinoceros were assumed to be “ideal” in their judgement of habitat quality and “free” to move from habitat to habitat in their quest for the best as there were no physical barriers separating the habitats. These are critical assumptions of the IFD theory, and by fulfilling the assumptions, this translocation provided another opportunity to test if the post release distribution of the southern white rhinoceros in Ol Pejeta conservancy followed the IFD theory or not.

Other than determining the numbers in each national park, private and community wildlife sanctuaries, and the daily patrols for sightings and anti-poaching purposes, little research on this species has been undertaken in Kenya. For example, only one study (Mukewa, 1996) is documented in Kenya that attempted to address adaptation of translocated white rhinoceros. However, this study did not determine the home range sizes of the rhinoceros but estimated the distribution of the white rhinoceros around water sources. Similarly it did not look at habitat selection of the translocated rhinoceros. The translocation of six southern white rhinoceros (three females and three males) to Ol Pejeta Conservancy thus provided a chance to study them and generate the missing information on habitat selection and distribution of this species when moved to a new habitat. Five of the six rhinoceros were translocated from Lewa Wildlife Conservancy with the

remaining one female from Kigio Wildlife Conservancy although it was also originally from Lewa to Kigio.

With the continued use of translocation as a management tool in rhinoceros conservation in Kenya, it is important to understand how translocated white rhinoceros selects its ideal habitat within the new habitat and how it utilises this new habitat. Although guidelines for keeping and managing white rhinoceros in Kenya were developed in 2007 (Okita-Ouma *et al.*, 2007), they are not clear on habitat quality aspects and require that procedures to assess white rhinoceros habitat and ecological carrying capacity be developed and implemented to manage the existing and assist in developing new white rhinoceros conservation areas in Kenya and former historical ranges of this species that border Kenya. This study therefore seized the opportunity to generate information that would contribute to the understanding of habitat selection and development of procedures for assessing white rhinoceros habitat quality and ecological carrying capacity.

#### **1.4 Broad objective**

The broad objective of this study was to assess the post release habitat selection and distribution of the southern white rhinoceros translocated to Ol Pejeta Conservancy by quantifying habitat quality of the study sites where the southern white rhinoceros moved, quantifying the movement size within these sites as well as determine if their distribution followed the IFD theory. This study focussed on grass biomass, faecal nitrogen and normalise difference vegetation index (NDVI) as they are indicators of habitat quality. While grass biomass and faecal nitrogen are both qualitative measures, the NDVI is a qualitative measure of habitat

quality. Likewise, habitat preference is a consequence of habitat selection while home range size is an indicator of animal distribution.

#### **1.4.1 Specific objectives**

1. To estimate grass biomass in the study sites,
2. To determine the relationship between NDVI and faecal nitrogen measure, in the study sites,
3. To determine habitat preference by the translocated white rhinoceros and
4. To estimate the home range size of the translocated white rhinoceros

#### **1.4.2 Hypotheses**

This study tests the IFD theory through the following specific hypotheses:

1. Frequency of sighting white rhinoceros per study site is not inversely proportional to the quantity of grass biomass in the study site
2. Study sites with high NDVI values will not have corresponding high faecal nitrogen value
3. White rhinoceros will not utilise study sites in proportion to their availability, but will prefer sites with high quality forage
4. The home range size of the white rhinoceros will not vary with seasons.

### **1.5 STUDY AREA**

#### **1.5.1 Location**

The Ol Pejeta Conservancy, a private wildlife sanctuary is located on the Laikipia Plateau, (0°00'N–0°02' S; 36°44' – 36°59'E) Kenya and covers an area of 90,000 acres (approximately 370km<sup>2</sup>). It is located about 260 km North of Nairobi and

approximately 20 km east of Nanyuki town and situated approximately between the foothills of the Aberdare ranges and Mount Kenya. Administratively, the Conservancy is demarcated into the western and eastern sectors by the Ewaso Nyiro River (Fig. 1.1). Current land use activities in the Conservancy include wildlife conservation, cattle ranching, and wheat farming. An elaborate history of establishment and management of Ol Pejeta Conservancy is available (Games, 2008; Kahiro, 2009).

### **1.5.2 Climate**

Rainfall has been measured in Ol Pejeta Conservancy since 1969 and, in recent years the Ol Pejeta Conservancy Ecological Monitoring Department, with the help of Kenya Meteorological Department has established 10 rain stations across the Conservancy. Ten years data (1998-2008) from these stations indicate that the mean annual rainfall on Ol Pejeta Conservancy is  $731 \pm 220$ mm. Generally, the Conservancy experiences two rainfall seasons with the long rains occurring between March and May while the short rains occur between October and December (Waweru and Musyoki, 1992; Birkett, 1999; Games, 2008). Two years data (2006-2008) from the Ecological Monitoring,

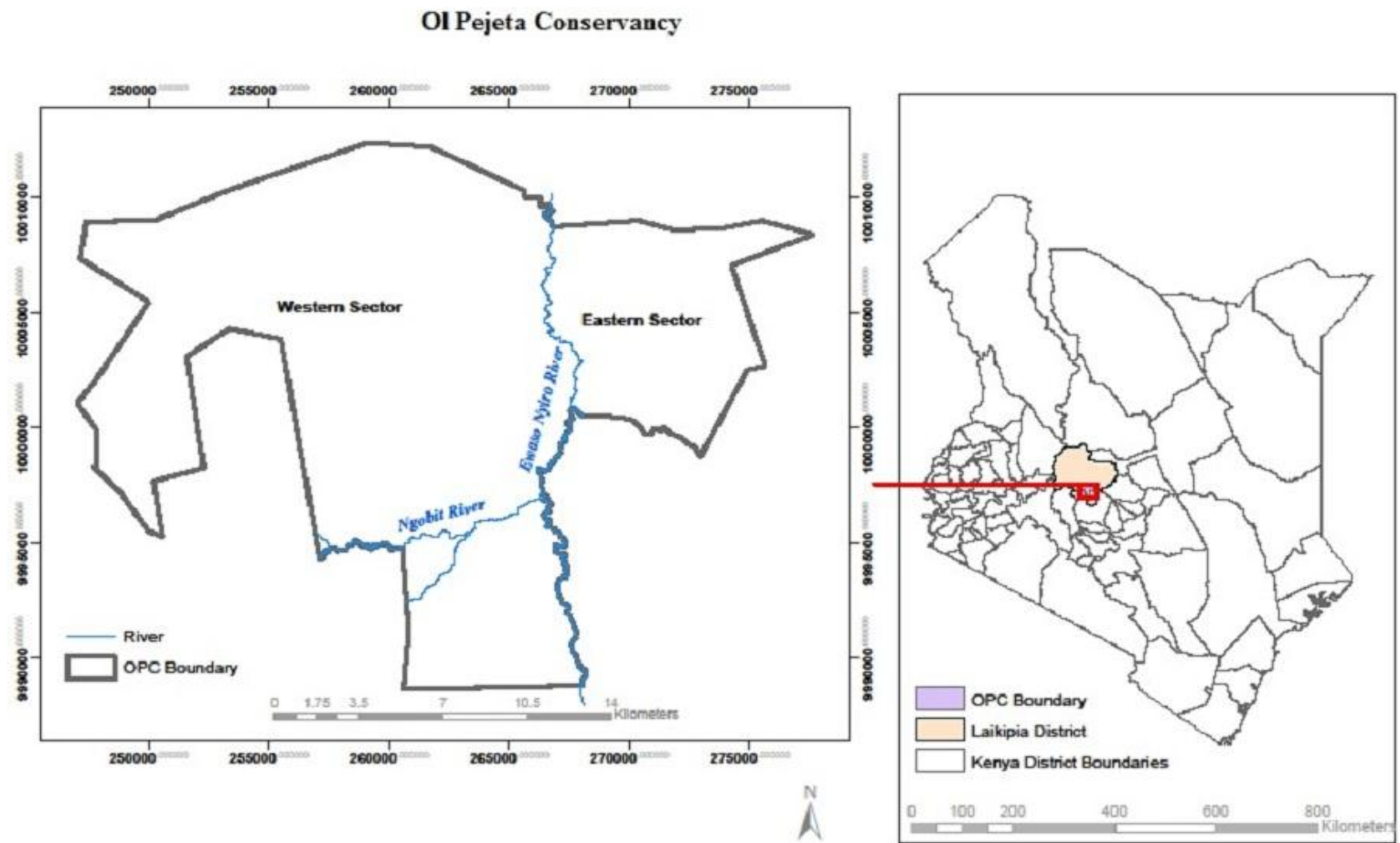


Figure 1.1: Ol Pejeta Conservancy (Source: OPC EMU and Clifford Okembo, 2014)

Department indicate that the mean annual maximum temperature is 22°C with a mean minimum of 10°C.

### **1.5.3 Geology, drainage and soils**

The Conservancy is about 800m above sea level on average but rises to 1700 m above sea level on the western sector (Waweru and Musyoki, 1992; Birkett,1999). The topography consists of low undulating hills with two relatively large hills found on the western sector at Sirima and Githera (Games, 2008). This lava plateau which is primarily used for cattle and sheep ranching is generally flat but slopes gently where rivers cut through it. The two major rivers that traverse this plateau are the Ewaso Narok and the Ewaso Nyiro. The latter flows in a North-South direction close to the western boundary of the Conservancy and is the only permanent source of water to Ol Pejeta Conservancy.

The dominant soils within the Conservancy are the “black cotton” soils formed from decomposed volcanic lava (Games, 2008). These soils are rich in alkaline and are poorly drained thus become easily waterlogged during the rains and rapidly lose moisture during the dry season (Waweru and Musyoki, 1992; Games, 2008). It is likely that the area was within the volcanic plume emanating from Mount Kenya and most of the soils were deposited during the volcano (Games, 2008). Clay loams are found on the colluvial deposits and clay deposits along the larger rivers and drainage while alluvial deposits are found along the valleys of Ewaso Nyiro and Ngobit rivers (Games, 2008).

#### 1.5.4 Fauna species composition

Though the major goal of the Conservancy is to enhance the population of the black rhinoceros, it is also home to a wide range of wildlife species and birds (Waweru and Musyoki, 1992). Among the most common plains game include the Burchell's zebra (*Equus burchelli*), Grant's gazelle (*Gazella granti*), Thompson's gazelle (*Gazella thomsoni*), Impala (*Aepyceros melampus*), Kongoni (*Alcelaphus buselaphus*), Eland (*Taurotragus oryx*) and the Oryx (*Oryx gazella*) (Waweru and Musyoki, 1992). Other large mammals include black rhinoceros, southern white rhinoceros, northern white rhinoceros, African elephant (*Loxodonta africana*), African buffalo (*Syncerus caffer*), Hippopotamus (*Hippopotamus amphibius*) and Reticulated giraffe (*Giraffa camelopardalis reticulata*). Ol Pejeta Conservancy also has several carnivores namely Lion (*Panthera leo*), Cheetah (*Acinonyx jubatus*), Striped hyena (*Hyaena hyaena*), Spotted hyena (*Crocuta crocuta*) and the Black backed jackal (*Canis mesomelas*). Primates found in the Conservancy include the Vervet Monkey (*Chlorocebus pygerythrus*), yellow Baboon (*Papio cynocephalus*), the rare Pallas Monkey (*Erythrocebus patas*) and the exotic chimpanzee (*Pan troglodytes* spp.). Reptiles in the Conservancy are represented by the Crocodile (*Crocodylus niloticus*) and several species of snakes, such as olive hissing snake (*Psammophis mossambicus*). Boran cattle (*bos indicus*) are also found in the Conservancy (Waweru and Musyoki, 1992).

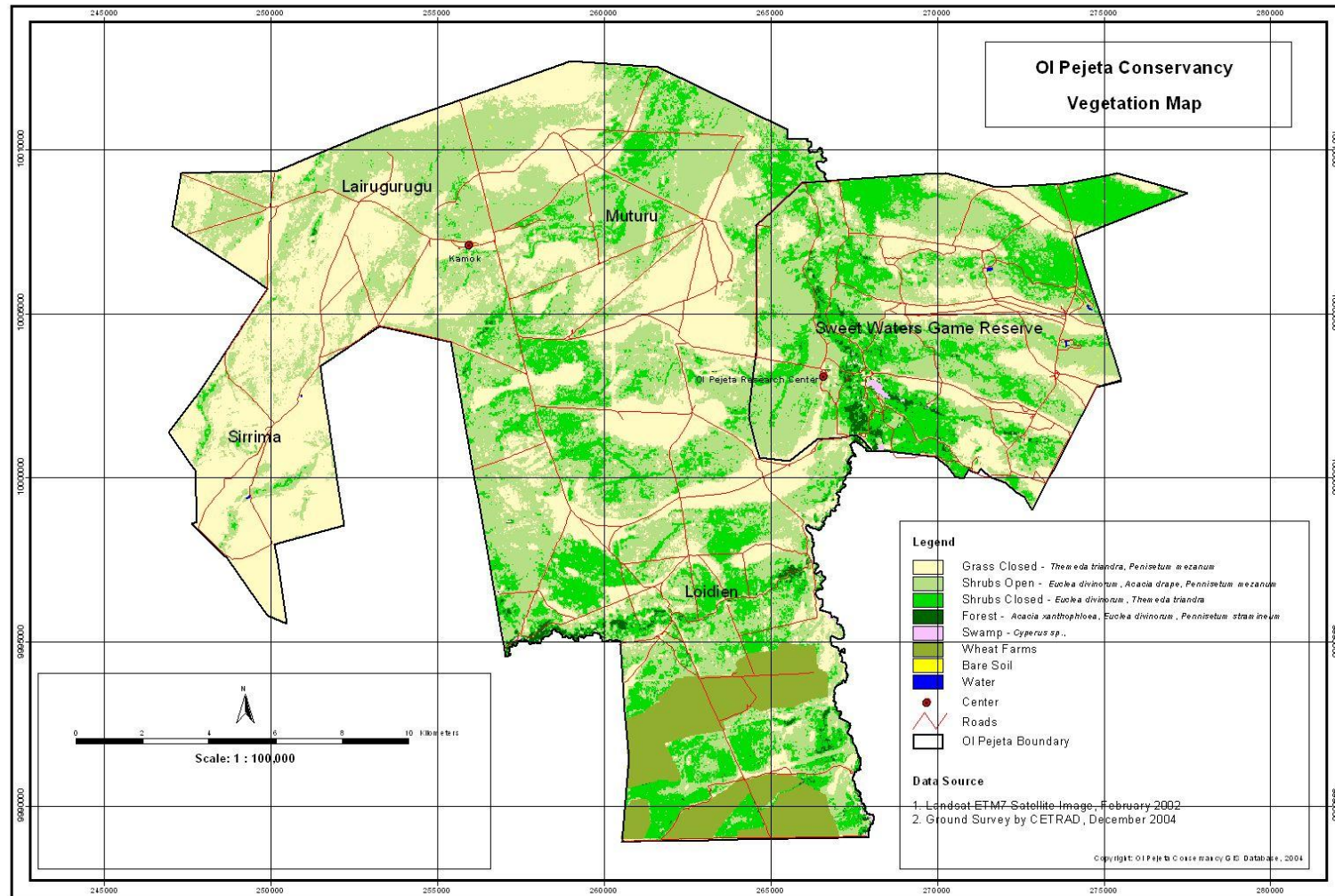
More than 230 species of birds have been recorded in Ol Pejeta Conservancy (Birkett, 1999). Notable among these include the Maasai Ostrich (*Struthio camelus massaicus*), Kori bustard (*Ardeotis kori struthiunculus*), the secretary bird (*Sagittarius serpentarius*), Ruppell's vulture (*Gyps rueppellii*), Egyptian vulture



(*Neophron pecnopterus*), Marabou stork (*Leptoptilus crumeriferus*), Pelican (*Pelecanus* spp), Augur buzzard (*Buteo augur*), Starling (*Lamprotornis* spp.), Shrike (*Lanius* spp.), Kingfisher (*Halcyon* spp. and *Alcedo* spp.), African Spoonbill (*Platalea alba*), various species of Eagles such as Tawny eagle (*Aquila rapax*), Francolin (*Francolinus* spp.) and the Guinea fowl (*Acryllium* spp.).

### 1.5.5 Vegetation types

OI Pejeta Conservancy has four main types of vegetation (Waweru and Musyoki, 1992; Games, 2008) that is, open grassland, thick bushes, riverine forests and swamp (Fig. 1.2 and Plate 1.1). The open grasslands are characterized by grasses such as *Themeda triandra*, *Pennisetum mezianum*, *Sporobolus pyramidalis*, *Digitaria* spp. *Cynodon dactylon* and *Cymbopogon nardu*. The thick bushes are characterized by *Acacia drepanolobium*, *Euclea divinorum*, *Rhus natalensis*, *Maerua triphylla* and *Scutia myrtina*. Some of the common tree species in the riverine areas include *Acacia xanthophloea*, *Euclea divinorum*, *Carissa edulis* and *Maerua triphylla* while the riverine forest floor is dominated by herbs and shrubs that include *Hibiscus aponeuris*, *Lippia javanica*, *Hypoestes verticillaris*, *Abutilon longiscuspe* and *Psiadia punctulata*. The major plant species in the swampy areas is *Cyperus* spp.



**Figure 1.2: Vegetation types of OI Pejeta Conservancy (Source: OPC EMU, 2006)**



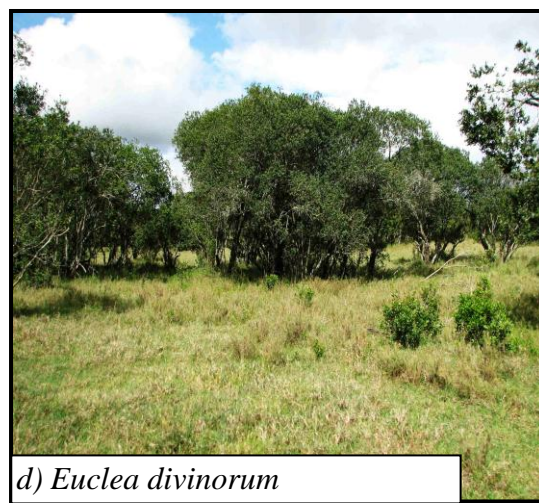
a) *Acacia xanthophloea*



b) *Acacia drepanolobium*



c) *Grassland*



d) *Euclea divinorum*

**Plate 1.1: Open grassland and dominant woody species of Ol Pejeta Conservancy (Source: Games, 2008)**

## CHAPTER TWO

### LITERATURE REVIEW

#### **2.1 Taxonomy, distribution and the status of southern white rhinoceros in Kenya**

The rhinoceros is classified in the Rhinocerotidae family which consists of five extant species, two of which are found in Africa and three in Asia. Those in Africa are the African white rhinoceros (*Ceratotherium simum*) or square-lipped and the black rhinoceros (*Diceros bicornis*) or hook-lipped rhinoceros, while those in Asia are the Indian (*Rhinoceros unicornis*), Javan (*Rhinoceros sondaicus*), and Sumatran rhinoceros (*Dicerorhinus sumatrensis*). The white rhinoceros is not only morphologically different from the black rhinoceros but also behaviourally different as it exhibits a higher degree of sociality (Owen-Smith, 1988; Shrader and Owen-Smith, 2002) compared to the black rhinoceros.

The African white rhinoceros is the third largest land mammal, after the African bush elephant (*Loxodonta africana africana*) and the Asian elephant (*Elephas maximus*), with estimated live body weights of 1600 kg in adult females and up to 2300 kg in males (Owen-Smith, 1988; Balfour and Balfour, 1991). Foster (1960) recorded maximum adult weight of 3200 – 3600 kg, but it is feasible to assume that their maximum body weight would gradually decrease over time due to the larger individuals fetching greater prices from trophy hunters and poachers alike (Pedersen, 2009). The white rhinoceros was separated into two subspecies in 1900 when a skull was discovered in Sudan and confirmed to be distinct from the South African variety due to the depth of dorsal concavity of the skull (Pedersen, 2009). The variety found in Sudan was then referred to as the Northern white rhinoceros

(*Ceratotherium simum cottoni*) subspecies while the well-known southern form was referred to as the Southern white rhinoceros (*Ceratotherium simum simum*) subspecies. Just over a century later, Brooks (2008) reported the Northern white rhinoceros to be on the brink of extinction.

Historically, the white rhinoceros had a restricted distribution in Africa with the northern white rhinoceros known to have been significantly more numerous than its southern relative in the 19<sup>th</sup> century (Emslie and Brooks, 1999). Poaching for the horn, hide, and meat resulted in great reduction of the numbers of the northern white rhinoceros with only 4 animals reported to exist in the Democratic Republic of Congo in 2006 and only 1 having been sighted in 2007 (Brooks, 2008). The future of this subspecies now lies on the only 4 individuals (2 females and 2 males) in the world with potential to breed that are now conserved in Ol Pejeta Conservancy having been translocated from Dvur Kralove Zoo, the Czech Republic, in 2009. Similarly, the relentless hunting of rhinoceros in Africa for sport and meat by Europeans and killing of rhinoceros in areas being opened up for agriculture, left between 20 to 100 southern white rhinoceros (Brooks and Emslie, 1999) that survived in one population in Hluhluwe-Imfolozi National Park within South Africa. This population was protected and closely monitored until 1961, when translocation techniques were successfully developed and surplus animals from this population were routinely translocated from Hluhluwe-Imfolozi National Park to establish new populations in Parks, Conservancies and Community areas within South Africa. By the end of 2007, over 17,000 southern white rhinoceros were estimated to exist in Africa (Brooks, 2008).

The southern white rhinoceros was introduced in Kenya by the Government in 1965 from Hluhluwe-Imfolozi National Park, South Africa, on conservation basis and was called a re-introduction. This was based on the presence 3000 years ago of another white rhinoceros amongst East African fauna based on fossil records and cave paintings. The initial 6 southern white rhinoceros (3 males and 3 females) were translocated to Meru National Park but were poached except one male namely “*Mukora*” that was translocated to Lewa Conservancy in 1989. Meanwhile, another 20 white rhinoceros had also been translocated from Hluhluwe-Imfolozi National Park, in the early 1970s to Solio Game Reserve, Kenya (Okita-Ouma *et al.*, 2007). In 1992, a further 5 southern white rhinoceros were translocated from South Africa to Lewa Wildlife Conservancy, Kenya, with the last translocation being of 20 individuals in 1994 by the Kenya Wildlife Service (KWS) to Lake Nakuru National Park and Ol Chororwa Conservancy (Okita-Ouma *et al.*, 2007). The surpluses from Solio, Lewa and Lake Nakuru National Park sub-populations have since been used to establish other sub-populations in Kenya and Uganda through translocations.

By the end of 2008, over 300 southern white rhinoceros were estimated to exist in Kenya in 4 large (>30 rhinoceros) populations and 11 small populations (Appendix I) six of which lived in pairs (Okita-Ouma *et al.*, 2007). A majority (57%) of the small populations have not recorded births since they were translocated to their new habitats for reasons not documented. Considering that an out-of-range population of southern white rhinoceros had been established in Kenya; that there was no supporting legislation and management strategy; and the considerable value that the species has for tourism and for driving community conservation initiatives;



Kenya Wildlife Service and its partners in 2007 declared the overall strategic goal for managing the white rhinoceros in Kenya as being: “*a species for community conservation, education, and tourism and as a conservation resource for stocking white rhinoceros ranges outside Kenya*” (Okita-Ouma *et al.*, 2007). Similarly, minimal guidelines for keeping and management of the white rhinoceros in Kenya were developed with four strategic objectives of biological management, monitoring for management, protection and coordination and support. Part of the biological management objective requires that procedures to assess white rhinoceros habitat and ecological carrying capacity be developed and implemented to manage the existing white rhinoceros sub-populations and to assist in developing new sub-populations of this species in Kenya and its historical ranges that border Kenya.

## **2.2 Translocation in wildlife management and lessons learnt**

Translocation has been used as a tool in wildlife management for many decades. In India for example, it has been found to be the best non-destructive control measure for the rhesus monkey (*Macaca mulatta*) (Southwick *et al.*, 1984; Forthman-Quick, 1986; Else, 1991). Further, Imam *et al.*(2002) noted that studies in India mainly used translocation as a management intervention to reduce human-wildlife conflict. In Africa, translocation of African bush elephants for management purposes was first adopted in South Africa in the 1970s (Whyte, 2001a) mainly to relieve the ecosystems from high densities of elephants. Since early 1990s, several elephant translocations have also been done in East Africa, mainly in Kenya. These translocations have focussed on reducing human-elephant conflict and thus the post release monitoring has concentrated on assessing the alleviation of the conflict

problem in the area where the elephants were removed and ensuring that the elephants do not become a menace in the area where they have been moved to.

Van Zant *et al.* (2003) monitored 18 translocated Choctawhatchee beach mice (*Peromyscus polionotus allophrys*) as they attempted to integrate into an established population and showed that translocations into already occupied habitats, even in the most conducive situations, may yield unanticipated outcome. Their study recorded a loss of nearly 100% of the translocated mice; although they had anticipated at most a 50% loss. The study thus concluded that additional knowledge on home range of the existing population is necessary before translocations are used to supplement existing populations. Studying the eastern barred bandicoots (*Perameles gunnii*), Todd *et al.* (2002) observed that the removal of individuals for translocation to other populations clearly contributed to the decline of this species in its original habitat. Their results highlighted the need for careful evaluation of the status of any potential source population before translocation.

Translocation of rhinoceros has been conducted extensively in Southern Africa (Hitching, 1984; Adcock *et al.*, 1998; Linklater *et al.*, 2006) and Kenya (Annon, 1993; Okita-Ouma, 2004; Okita-Ouma, 2007; Patton *et al.*, 2010). Unfortunately there is little known on the social behaviour of translocated rhinoceros during their early stages of establishing a home range (Linklater *et al.*, 2006; Patton *et al.*, 2010) mainly due to lack of long-term quantitative information on the post-release monitoring (Scott and Carpenter, 1987). Post-release monitoring is thus an integral component of rhinoceros translocation as it provides knowledge of their spatial and



temporal dynamics; and their habitat use which is a crucial aspect of ecosystem management (Bailey *et al.*, 1996) and has important consequences for feeding ecology, social organization, mating strategies and conservation management (White *et al.*, 2007) for this species.

An analysis of introduction of 24 black rhinoceros into Pilanesberg National Park, in South Africa, showed a combined post-release mortality rate of 12.5% with the mortality rate being highest during the introduction phase (Adcock *et al.*, 1998). Hitchins (1984) also reported an overall post-release mortality rate of 6% for Natal Parks Board translocations from 1962 to 1984. From 1986 to 1995, post-release mortality rate among South African and Namibian translocations (excluding those to Pilanesberg) was 8.4% (Adcock, 1995; 1996). In Kenya, a post-release mortality rate of 16% out of 118 translocations was reported by Brett (1998) for the period 1984 to 1995, while Mulama and Okita-Ouma (2005) reported a 5% post-release mortality rate out of 90 translocations for the period 1994 to 2005. Patton *et al.* (2010) describes the post release monitoring of the largest single translocation of rhinoceros in Kenya and notes that there was no post-release mortality. The experience in both South Africa and Kenya indicates that the management of black rhinoceros with respect to translocations has faced challenges in post-release adaptation.

Nevertheless, translocation is inevitable to effectively manage rhinoceros meta-populations and attain high population growth rate. In most African rhinoceros range states, rhinoceros sanctuaries are stocked at or below the desirable stocking rate at which the maximum population growth rates can be attained, usually

estimated as 75% of ecological carrying capacity (Emslie and Brooks, 1999). The effect of translocation on black rhinoceros population growth rate and on productivity maintenance has been well documented by several authors (e.g. Brett, 2001; Goodman, 2001; Owen-Smith, 2001; Okita-Ouma, 2004), while the capture and translocation procedures in Kenya are covered in Okita-Ouma *et al.* (2007). Although translocation has been a key component of successful rhinoceros conservation in Africa and in developing viable populations of rhinoceros, this has not been the case with the white rhinoceros in most national parks, private and community wildlife sanctuaries in Kenya.

### **2.3 Habitat quality and animal distribution**

A variety of definitions and use of terms related to habitat exist (Hall *et al.*, 1997). However, in the simplest form, the habitat of an organism is the place where it normally lives, grows and reproduces (Odum, 1971). A habitat therefore, must contain specific resources needed by an organism such as food, cover, water, nutrients and the right climate for it to sustain the organism (Melton, 1987). On the other hand, habitat use is the way an animal uses the physical and biological resources in a habitat (Krausman, 1999), whereas habitat selection is a hierarchical process involving a series of innate and learned behavioral decisions made by an animal about what habitat it would use at different scales of the environment (Hutto, 1985). On the other hand, habitat preference is the consequence of habitat selection, resulting in the disproportional use of some resources over others. Since animals are able to move and select a suitable habitat within a given area, many animal species occupy different habitats at different times of the year or stages in their life cycle. Even within a habitat, some sites are more preferred than others,

indicating that distribution patterns of animals in a habitat may be influenced directly by a combination of biotic (e.g. food, water) and abiotic (e.g. wallows, shade) factors. The accessibility and procurability of these biotic and abiotic components of a habitat by animals is defined as habitat availability (Krausman, 1999).

Consequently, not all habitats are occupied because animals will choose a habitat based on its suitability; that is, its ability to provide the animal with its requirements for growth, survival and reproduction (Bailey *et al.*, 1966; Redfern *et al.*, 2003). Hall *et al.* (1997) defines these habitat characteristics as habitat quality and further suggest that it is a continuous variable, ranging from low (i.e. based on resources only available for survival), to medium (i.e. based on resources available for reproduction), to high (i.e. based on resources available for population persistence). For example, grassland habitats are grazed by a variety of grazers, many of which are notable for the range and diversity of consumers that depend on the grass resource for food (Farnsworth *et al.*, 2002). Information on habitat quality is thus important to conservation of grazers in a grassland ecosystem (Bailey *et al.*, 1966). Even though information on habitat quality in grassland ecosystems is scarce since grass production is variable in time and space thus limiting adequate monitoring and requiring enormous resources for long-term monitoring, it is important that these ecosystems are studied so as to improve their ecological management and conservation as they constitute approximately one third of the Earth's vegetative cover (Jacobs *et al.*, 1999).

The Ideal Free Distribution theory proposed by Fretwell and Lucas (1970) and Fretwell (1972) describes how animals will disperse within an environment containing habitat patches of varying suitability. The theory assumes that individuals have similar competitive abilities and move freely among patches in relation to rewards expected from each patch. Consequently, the theory predicts that the number of individuals per patch will be proportional to the fraction of resources in that patch and that the intake per individual will be equal across all patches (Darling, 2000). The IFD theory has been modified to take into account variation in competitive ability among members of foraging groups (Harper, 1982; Godin and Keenlyside, 1984; Alonso *et al.*, 1995).

Several studies (Conradt *et al.*, 1999; Pettorelli *et al.*, 2001, 2002) on ungulates have tested whether they conform to the predictions of IFD and the results suggest a rather strong spatial structuring thus violating the IFD predictions. Similarly, in a study on white-tailed deer (*Odocoileus virginianus*), deviations were found around IFD in an experimental setting with two artificial patches (Kohlmann and Risenhoover, 1997). Only a few studies, none of them fully experimental, have shown that selection of ideal habitat patch is indeed density dependent (Freeland and Choquenot, 1990; Choquenot, 1991; Mellado *et al.*, 2003; Kausrud *et al.*, 2006).

Several direct and indirect methods for assessing habitat quality exist of which direct harvesting is rated as the most accurate for determining above-ground standing biomass (Causton, 1988), an indicator of habitat quality. However, this method is not practical for monitoring extensive rangelands (Holechek *et al.*, 1989)

since above-ground standing biomass varies rapidly in time and space. Consequently the Disc Pasture Meter (DPM) still remains a preferred method for estimating standing biomass of herbaceous material (Bransby and Tainton, 1977). The DPM has been successfully calibrated and used in savanna areas including for example in Kruger National Park (Trollope and Potgieter, 1986), the Zululand Coastal Plain (Brockett, 1996), Lewa Wildlife Conservancy (Botha, 1999), Nylsvley Nature Reserve (Dorgeloh, 2002), and in the Arid and Semi Arid Land (ASAL) of Northern Kenya (Chege, 2004). The use of DPM method to estimate biomass compares closely to the traditional methods of estimating yield, but it is more attractive because it is a rapid and accurate way by which yield can be measured in a non-destructive manner (Botha, 1999).

Indirect methods for assessing habitat quality include faecal analysis (for total nitrogen and phosphorous) and the use of remote sensing techniques. Faecal indices are easy to acquire, relatively inexpensive to analyse, and do not interfere with the daily feeding strategy of the study animal (Ryan *et al.*, 2003). However, when habitat quality data are required over a long-term period, then logistical and financial constraints render this measure impractical. Remote sensing techniques take advantage of some earth observation satellites (e.g. Landsat, SPOT, NOAA, and MODIS) each carrying specialised sensors with varying spatial and temporal resolutions (Komp, 1991). Satellites with sensors that have a high temporal resolution are suitable for savannah resource monitoring since grass at productive juvenile growth form changes very fast and would require satellites with a high revisit capability (Prince and Tucker, 1986) such as MODIS.

The sensors are designed to measure indicators of vegetation activity, which is mostly related to photosynthesis. The main indicators of photosynthetic activity are a combination of visible (R) and near infra-red (NIR) reflectance (i.e. proportion of the incoming solar light reflected by the earth's surface) measurements. The simplest vegetation index, often called "simple ratio", is computed by dividing NIR by R. The higher the ratio, the more the absorption of R and emission of NIR for photosynthesis, and thus the more active and productive the plants are. The most utilised vegetation index worldwide is the normalized difference vegetation index (NDVI), a normalized ratio of the NIR and R, and computed as follows:  $NDVI = (NIR - R) / (NIR + R)$  (Reed *et al.*, 1994; Myneni *et al.*, 1995). This index varies by a fixed interval of [-1, +1] and is easier to compare in time for monitoring purposes (Prince and Tucker, 1986).

It is the utilisation of the solar energy, its interception and absorption by plants that is measured from space by the satellite sensors (e.g. Multi-Spectral Scanner, High Resolution Visible Instruments or Advanced Very High Resolution Radar) and thereby offering a relatively cheap and reliable source of data, especially in savanna ecosystems, that can be used to monitor grass "greenness". Numerous studies now use the satellite-derived NDVI to link habitat quality and herbivore life-history traits (Pettorelli *et al.*, 2007; Ryan, Knechtel and Getz 2007; Wiegand *et al.*, 2008) as well as in plant and animal population ecology (Hamel *et al.*, 2009).

In most cases, however, remote sensing techniques such as satellite-derived NDVI must be combined with ground-truthing in order to establish the relationships between remotely sensed data and what actually exists on the ground. Three types

of remote sensing methods most widely used in eastern Africa for monitoring extensive rangelands are: satellite imagery, aerial photography and observations made from light aircraft (Gwynne and Croze, 1975; Norton-Griffiths, 1975; Clarke, 1986; Loth, 1990). The type of remote sensing method used usually depends on the objective of the study, level of details required and the budget available. Satellite imagery, that is acquisition of black and white and false colour images of the earth's surface from satellites, was preferred in this study given the high revisit capability required and that the associated cost was to be covered outside the budget of this study.

#### **2.4 Habitat preference, ranging patterns and home range of animals**

Bailey *et al.* (1996) and Redfern *et al.* (2003) observed that the distribution patterns of large herbivores may be directly influenced by the combination of biotic and abiotic factors, while Melton (1987) has shown that habitat selectivity can affect an animal's potential to reproduce and survive, which for ungulates depends on readily available food, water, shelter from harsh weather and protection from predators. According to Du Toit and Owen-Smith (1989), large African savanna herbivores utilise disproportionately large quantity of resources, thus allowing them to survive in a wider range of habitats not suitable for smaller species. It is therefore important to consider this disproportionate use when assessing the surrounding habitat suitability for large (African) savanna herbivores.

Finlayson *et al.* (2008) further noted that it is critical to have adequate understanding and details about habitat preferences or avoidance and patterns of use when making informed conservation and management decisions or plans on

threatened animals e.g. the white rhinoceros. According to Owen-Smith (1973; 1988), there are a number of requirements in a habitat that will determine its preference by white rhinoceros. Primarily, these requirements are food, water, wallows, rubbing posts, shelter and often mineral licks when the nutrient quality of the available grasses is particularly low. All of these requirements define habitat preferences if more than one is abundant in a particular habitat.

Animal range is the total area used over a specified time period, while home range is that fraction of the total area habitually used (Newton-Fisher, 2003). Ranging behaviour or pattern of an animal thus gives rise to home range, a concept that has been a source of debate among ecologists. However, the definition most often encountered in literature as given by Osborn (2004) states that home range is “the area traversed by an individual in its normal activities of food gathering, mating and caring for young”. Osborn (2004) further noted that the “dispersal and occasional ventures outside the area, perhaps exploratory in nature, should not be considered as part of home range”. White and Garrott (1990) also noted that home range is not the entire area over which an animal moves but the area over which it normally moves. The problem in both definitions is that mammals exhibit widely diverse movement patterns that are influenced by a combination of factors. These factors include; availability of resources (e.g. food and water), social behaviour, predator avoidance and human disturbance (Osborn, 2004). Some animals may thus regularly shift their range in response to changing environmental conditions.

On the other hand, Jewell (1966, as cited in Osborn 2004) defined home range as “an area with a certain productivity that meets the energy requirements of an



individual that occupies it". Similarly, McNab (1963, as cited in Osborn 2004) states that home range size could be expressed as a function of body weight that was directly comparable to the function relating basal metabolic rate to body weight. In a bid to clarify the concept, Jewell (1966, as cited in Osborn 2004) suggested the term lifetime range, meaning the "total area, within which an animal has become familiar, including seasonal home ranges, excursions for mating and routes of movement". It then follows from this baseline definition that home range assessments attained for relatively long-lived and highly mobile animals are "snapshots" and do not represent all the places the animals have traversed in their lifetime.

More recently, White *et al.* (2007) has defined home range as a non-random area within which the animal travels in the process of normal activities such as feeding, mating or resting. However, Hayne (1949, as cited in Osborn 2004) observed that animals do not use their entire home range with equal intensity but occupy certain areas, known as core areas, with greater frequency than others, while Harris *et al.* (1990) appreciated that core areas are rarely defined precisely. Efforts to establish core areas, therefore, actually identify areas of high animal activity and exclude occasional ventures (Osborn, 2004).

Estimates of home range are an important component of species conservation and management strategies. However, most authors do not state why they chose one method of analysing home range over another (Harris *et al.*, 1990). The criteria on which they base their home range size estimation, the number of GPS fixes and auto correlation or determination of core areas is also not reported consistently.

However, Osborn (2004) notes that it is generally believed that determining home range can be useful for a variety of reasons if the objectives are clearly defined and the techniques used are stated. According to Kenward (1990), there are at least six fundamentally different approaches for representing an animal's home range.

A review of the literature on home range analysis by Osborn (2004) indicate that there was little agreement among authors about which technique is generally the most appropriate. The decision on the technique to be used depends heavily on the question to be addressed and the type of data to be collected. Osborn (2004) grouped the methods for calculating home range into those based on a statistical distribution of activity loci and non-statistical methods. Techniques for estimating home range size non-statistically involve either drawing polygons (convex, concave or restricted) around the outer fixes or by overlaying grid cells (White and Garrott, 1990); whereas probabilistic methods include drawing probabilistic circles or ellipses around all the fixes (Jennrich and Turner, 1969, as cited in Osborn 2004).

The simplest way to estimate the size of a home range is to draw a polygon that encloses all the Global Positioning System (GPS) fixes, and then estimate the area in the polygon (Osborn, 2004). The minimum convex polygon (MCP) is simple to calculate and is the most widely published estimate of home range size despite its limitation in that the range estimate continues to increase as more GPS fixes are added (Osborn, 2004) and that the range estimate is a function of the number of locations used to generate the range (White and Garrott, 1990). Minimum convex polygons are also heavily influenced by "outliers" and sample size (Schoener,

1981). The harmonic mean method has also been widely used as a measure of animal activity centres (Dixon and Chapman, 1980). However, this method has some drawbacks in that the contours that include all fixes tend to “balloon” into areas never visited by an animal (Kenward and Holder, 1995). The kernel method proposed by Worton (1989) is similar to harmonic mean method but uses the kernel fix estimator instead of the harmonic mean centre and tends to give a more accurate representation of home range (Osborn, 2004). The kernel method is preferable to the harmonic mean method because the output is the actual probability value.

The kernel method, which appears to give the most accurate representation of the structure of an animal’s range and more precise estimates of total range and core area sizes, was thus used in this study. Its choice was guided by its ability to give an accurate representation of the animal’s range structure (Osborn, 2004) and the knowledge that MCP method is often biased by extraordinary movements beyond typical range (Leggett, 2006). The kernel method is also more adept at locating geographical features such as rivers, lakes, inhospitable terrain, and incorporating these into the analyses (Getz *et al.*, 2007).

Methods available for the measurement of habitat use by wild ungulates have been documented by Prins and Bokdam (1990) in their review of methods that can be used to assess the occupancy of the habitat by animal species. The methods reviewed can be divided into two groups namely; direct and indirect methods. Direct methods are those in which direct visual or other contact with the animal(s) to be observed is used to establish its presence or activity and indirect methods are

those in which traces of various kinds are interpreted as indicating presence or activity. The direct methods include visual observations that are enhanced by marking of individual through ear notching and/or radio transmitters, while indirect methods included spoor, dung depositions and bite marks. Information gathered on occupancy can then be used to deduce reliable information on habitat preference and ranging patterns of animals.

Furthermore, there have been many papers published on the different analyses for evaluating resource and habitat use versus availability (Neu *et al.*, 1974; Johnson 1980; Byers *et al.*, 1984; Alldredge and Ratti 1986; Scogings *et al.*, 1990; Alldredge and Ratti, 1992; Aebischer *et al.*, 1993; Mysterud and Ims, 1998; Conner *et al.*, 2003). However, the preference index of Viljoen (1989) was considered suitable for this study as the study animal had been individually marked.

## CHAPTER THREE

### MATERIALS AND METHODS

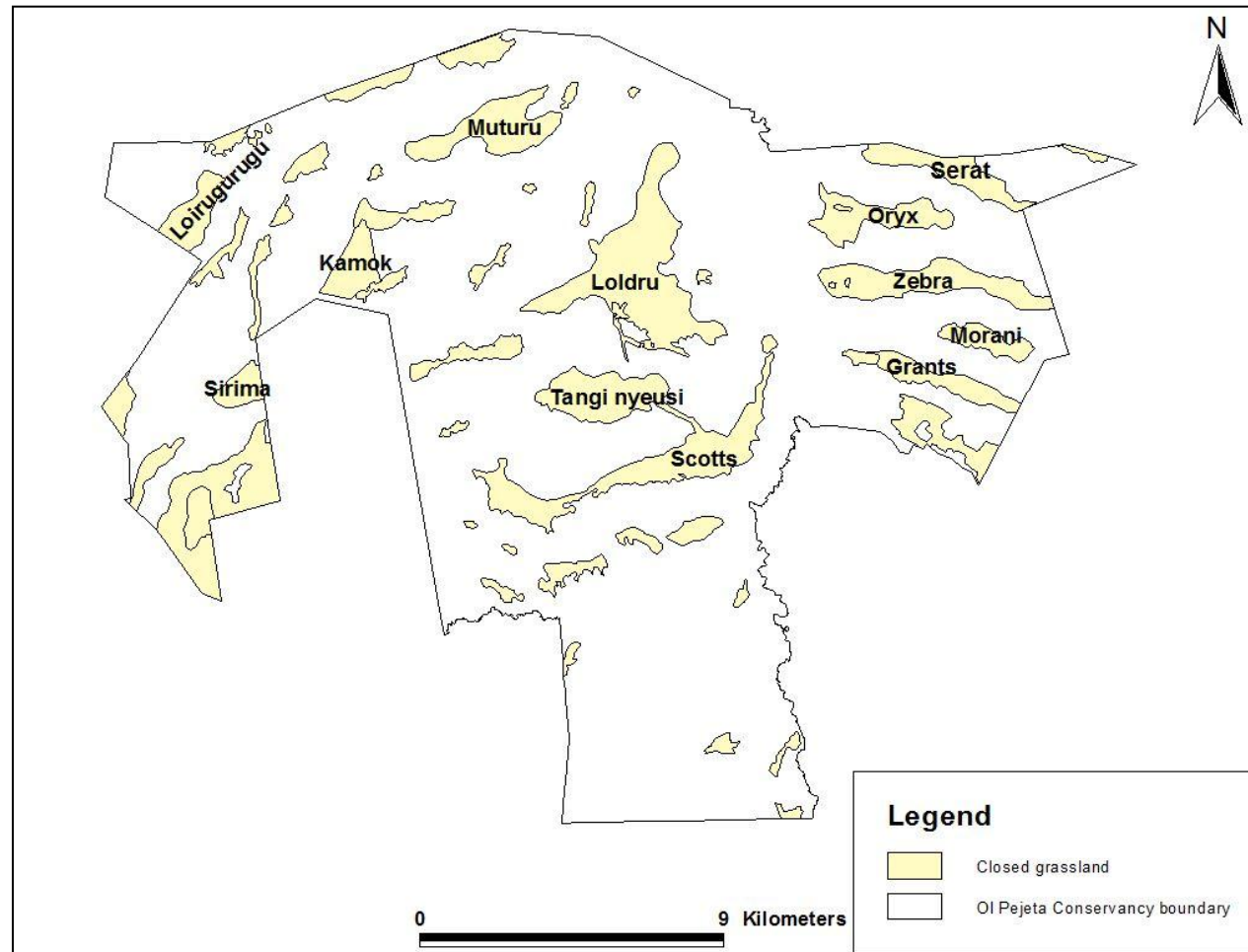
#### 3.1 Selection of study sites

Given that the focal animal in this study was a grazer, grassland types in Ol Pejeta Conservancy were considered as the study sites. In order to achieve this, the vegetation map of Ol Pejeta Conservancy was divided into 1km<sup>2</sup> grids using ArcView 3.3 GIS software, where all the open grasslands were highlighted, and those that fell within the 1km<sup>2</sup> grids or more were confirmed as the study sites (Fig 3.1). The minimum 1km<sup>2</sup> grid was used as a confirmation criterion for study site so as to accommodate the NDVI data, which had a spatial resolution of 1km (Komp, 1991).

The study was conducted in six study sites namely; *Morani*, *Zebra*, *Oryx*, *Serat*, *Grants* and *Loldru* (Fig. 3.1) that fall in both the western (*Loldru*) and the eastern (*Morani*, *Zebra*, *Oryx*, *Serat* and *Grants*) sectors of the Conservancy. However, calibration of the disc pasture meter was undertaken during the post release settling phase of the southern white rhinoceros.

#### 3.2 Estimating grass biomass in the study sites

A disc pasture meter was used for estimating grass biomass. This is a simple inexpensive instrument, which consists of a 45.8 cm base plate sliding over a 180 cm long calibrated aluminium rod (Bransby and Tainton, 1977). It yields a significant linear regression between disc height (cm) and standing crop (kg/ha). However, the regression coefficients yielded by the linear regression equation may

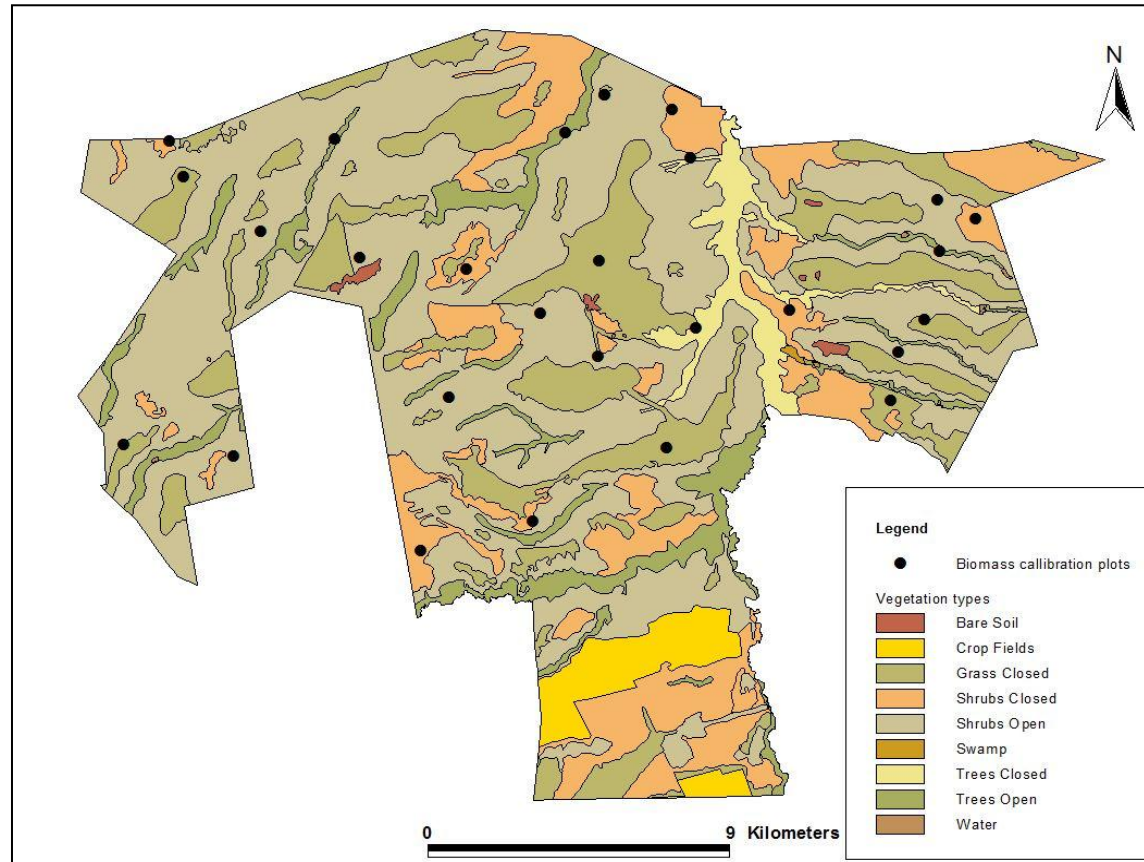


**Figure 3.1: Location of the study sites in OI Pejeta Conservancy (Source: OPC EMU, 2006)**

change with geographical area, and it was therefore important to calibrate the DPM using data collected in OPC (area-specific data) as recommended by Mitchell *et al.*, (1993) before using it to estimate grass biomass in the study sites.

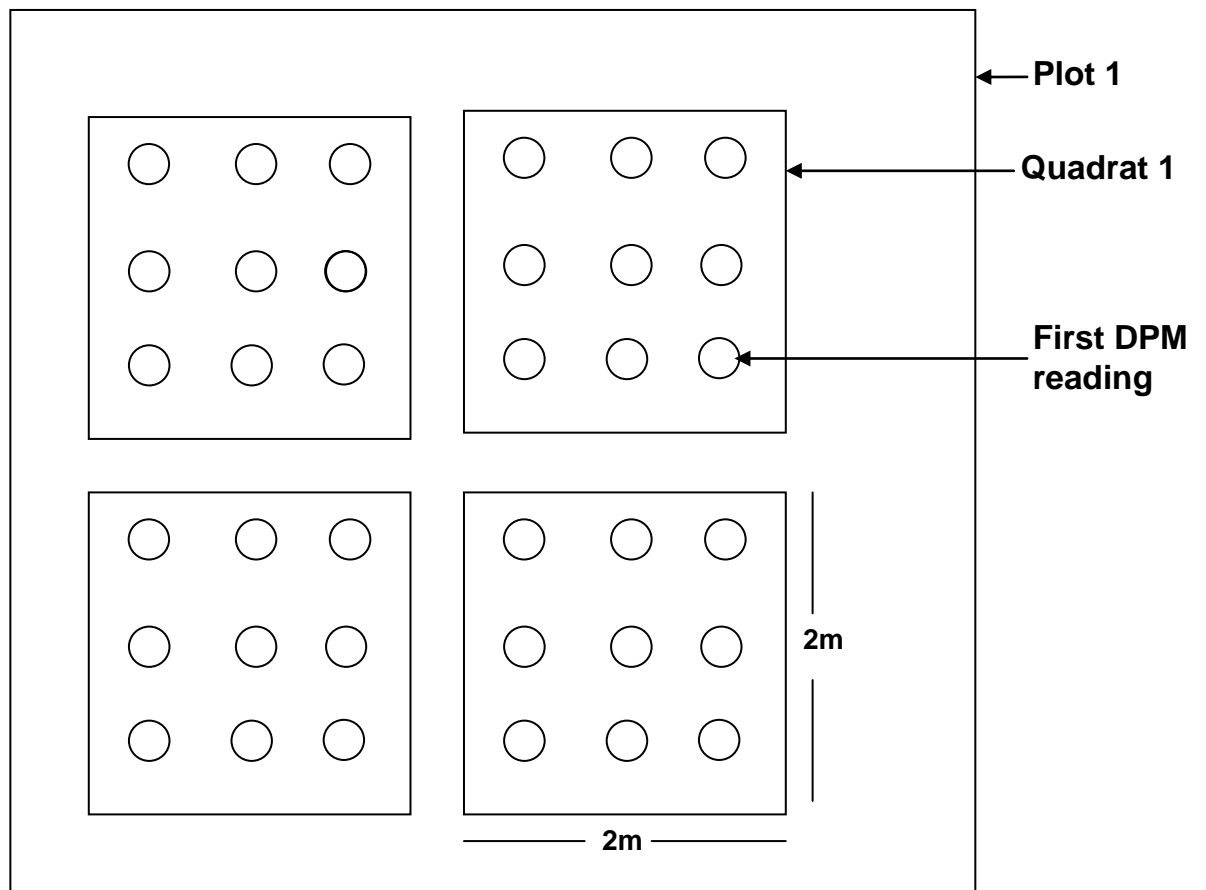
For the purpose of calibrating the DPM, all the vegetation types within Ol Pejeta Conservancy were considered so as to capture the difference in grass biomass within the Conservancy. Consequently, a total of 28 plots were established for calibration purpose (Fig. 3.2), while estimation of the grass biomass focused on the six study sites. The procedure for calibrating the disc pasture meter followed that outlined by Trollope and Potgieter (1986) and Dorgeloh (2002) whereby four separate square quadrates (4 m<sup>2</sup> each) were used at each of the 28 plots distributed over Ol Pejeta Conservancy vegetation types, thus giving a total of 112 calibration surveys of (4 x 4) m<sup>2</sup>.

The 28 plots were stratified randomly to represent grass swards within the vegetation types that had been lightly, moderately and heavily grazed. Sampling at each plot was done using the four square meter quadrats that were placed close to each other (Fig. 3.3). In each quadrat, 9 readings of the settling height of the DPM were recorded and then averaged to obtain the mean settling height of the DPM per quadrat. The grass layer within each quadrat was then harvested as close to the ground level as possible, the wet weight determined using a weighing scale at the Ol Pejeta research centre and from this sample 9 sub samples collected randomly. Wet weight of each sub sample was measured, dried at 100°C for 48 hours and then dry weight of each determined.



**Figure 3.2: Location of Disc Pasture Meter calibration plots in Ol Pejeta Conservancy (Source: OPC EMU, 2006)**





**Figure 3.3: Typical sampling scheme for Disc Pasture Meter calibration at one of the twenty eight plots (Source: modified from Botha, 1999)**

Dry weight of the above-ground standing biomass (kg/ha) was estimated for each quadrat based on the percentage moisture loss from the 9 sub samples. These data were used to calibrate the DPM by developing a linear regression equation between the disc height (cm) and the standing crop (kg/ha).

The calibrated DPM was then used to estimate grass biomass in the six study sites. In each site, a permanent monitoring point was established by marking its coordinates using a GPS unit. From this point, the DPM settling height was recorded every 1 meter for 25 points in the four main compass directions. This gave a total of 100 disc meters in each study site. Trollope and Potgieter (1986) found that 100 disc meter readings per sampling site were adequate for estimating the grass biomass in a given habitat. Harvesting within the quadrats was only done during the calibration and not during grass biomass estimation for the six study sites. Grass biomass estimates were obtained every two weeks for 18 months.

### **3.2.1 Data synthesis**

In order to calibrate the disc pasture meter, the relationship between mean disc settling heights (cm) and biomass (kg/ha) per quadrat was determined using simple linear regression. Data from 90 of the total 112 calibration survey plots were randomly selected for developing a regression model and the remaining data from the 22 survey plots was later used to test the model. In order to find the best fit using the coefficient of determination ( $r^2$ ), further syntheses were done by subjecting the untransformed mean disc height (independent variable) as well as the transformed mean disc height and biomass (dependent variable) per quadrat to a linear regression (Dorgeloh, 2002). The different transformations of the mean

disc height conducted to obtain best fit were; (i) square ( $x^2$ ), (ii) square root ( $\sqrt{x}$ ), (iii) reciprocal ( $1/x$ ), and (iv) natural log ( $\ln x$ ).

The grass biomass (kg/ha) per site was estimated by averaging the disc height readings for the 100 points at each site. The mean height was converted into an estimate of grass biomass using the equation obtained from the linear regression of the calibration model:  $y = -2248x^{-1} + 999$ , where:  $y$  = estimated biomass (kg/ha) and  $x$  = mean disc height (cm). A mean monthly biomass estimate was computed since biomass estimates were obtained every two weeks, within a month.

### **3.3 Determining the relationship between faecal nitrogen and normalized difference vegetation index**

Although remotely sensed NDVI data are potentially suitable for resource monitoring, it is important that a relationship be established with quantitative data if this is to be used as a regular and long-term predictor of habitat quality (Loth, 1990).

Vegetation indices from low resolution satellite images are generally not computed from daily images but “synthesised” or “composited” on a period of 16 days (Holben, 1986). This method reduces the cloud cover in the images and selects the best quality pixels out of the integration period. The NDVI values were derived from the MODerate-resolution Imaging Spectroradiometer (MODIS) on board NASA's Terra satellite with a spatial resolution of 250m in 16-day interval composite images starting from 1<sup>st</sup> January 2007 (<http://modis.gsfc.nasa.gov>) to 31<sup>st</sup> December 2008. The MODIS NDVI images were downloaded at Mpala

Research Centre, for the whole area covering central and northern Kenya. The areas corresponding to the study sites were then extracted from the images, converted into grids and stored in one folder for further processing using ArcGIS.

Given that both southern white rhinoceros and common zebra are water dependent, non-ruminants, non-selective grazers and that protein expressed as faecal nitrogen is the most limiting nutrient for grazers, it was expected that the standing biomass would be a good surrogate for habitat quality and this would be reflected in the faecal nitrogen (Ryan *et al.*, 2003). It was also noted that the density of the southern white rhinoceros (0.02 rhinos/km<sup>2</sup>) was very low compared to that of the common zebra (10.39 zebra/km<sup>2</sup>). Thus considering the digestive system, the density and therefore availability of faecal samples, and the feeding behaviour of the southern white rhinoceros and the common zebra, the faecal samples of common zebra were used for nutritional analysis to estimate faecal nitrogen. Fresh faecal samples were collected from each study site, where wetness and lack of insect damage were used as indicators of freshness within 24 hours as recommended by Stowe (2003).

Two faecal samples were collected from each study site assuming that they came from the common zebras that grazed within the study site and that each sample belonged to a different individual. According to Leite and Stuth (1994), faecal samples can be collected up to seven days after defecation. Samples were collected into a polythene paper bag, clearly labelled (e.g. Zebra 1/Morani site/20<sup>th</sup> July 2007), and then dried approximately 24 hours after collection, in an oven at 80° C for approximately 25 hours following Stowe (2003) method. After drying, the

samples were put in a manila paper and stored awaiting the digestion process to extract total nitrogen.

The extraction of total nitrogen was done using the block digester and following the procedures described by Gathua *et al.* (2002). A weight of 0.3 g oven dried (80°C) faecal sample was put into a digestion tube and 4.4 ml digestion mixture (hydrogen peroxide (H<sub>2</sub>O<sub>2</sub>) + H<sub>2</sub>SO<sub>4</sub> + selenium (Se) + salicylic acid) added. Two reagent blanks were also included and the samples digested at 360°C for 3 hours until the solution became colourless and then allowed to cool. A volume of 25 ml distilled water was added, quantitatively transferred to 50ml volumetric flask and then made to the mark with distilled water for total nitrogen determination.

### 3.3.1 Data synthesis

Normalised difference vegetation index data obtained were processed as follows; means of the entire time-series of NDVI images were calculated for 250m, 500m, and 1km MODIS using the cell statistics tool in ArcGIS; the NDVI values of each point were then generated using paxeextract tool; the "Extract Values to Point" tool under the Spatial Analyst tools was then used to transfer the values of the mean NDVI raster to their corresponding points and then plotted against respective dates.

Determination of percentage faecal nitrogen was done in two steps, with the first step being the calorimetric determination of total nitrogen in the faecal matter digests. Standard solutions of 0, 2.5, 5, 7.5, 10.0 and 15.0 mg N/litre were prepared from stock solution (2500 mg N/litre as (NN<sub>4</sub>)<sub>2</sub>S<sub>0</sub><sub>4</sub>) in 50 ml volumetric flasks).

The entire digest and the blanks were diluted to the ratio of 1:9(v/v) with distilled water to match the standards. A 0.2 ml volume of the sample digest, the blanks were pipetted using a micropipette into clearly labelled test tubes. 5.0 ml of the reagent N1 was added and allowed to stand for 15 minutes, and 5.0 ml of reagent N2 added and shaken well. Contents were allowed to stand for at least 2 hours and the absorbance was read at 650 nm. Calibration curve was plotted from the standard solutions and the concentrations of nitrogen in the solution obtained. {Reagents N1 and N2 were prepared as follows:

- i) *Reagent N1*: 34 g sodium salicylate, 25g sodium citrate and 25g sodium titrate were dissolved together in 750ml distilled water. 0.12 g sodium nitroprusside was added and made up to 1.0 litre with distilled water.
- ii) *Reagent N2*: 3g sodium hydroxide was added in 750 ml of distilled water. This was allowed to cool and 10.0 ml sodium hypochlorite was added, mixed well and made up to 1.0 litre}.

The second step was calculation of percentage nitrogen which was done using the following expression:

$$\% N = \frac{\{(a-b) * v * 100\}}{(1000 * w * al * 1000)}$$

Where; a = concentration of N in the solution, b = concentration of N in the blank, v = total volume at the end of analysis procedure, w = weight of the dried sample, and, al = aliquot of the solution taken.

### 3.4 Determining habitat preference and home range size

Upon translocation of the southern white rhinoceros to Ol Pejeta Conservancy, it was important to understand their post-release dispersal and study site preference or avoidance as well as determine their home range sizes.

At capture site in Lewa and Kigo, the rhinoceros were immobilised and fitted with radio transmitters in the anterior horn following procedures described by Pienaar and Hall-Martin (1991), which briefly included drilling a hole on the lateral side of the anterior horn, placing a transmitter in this hole and sealing the hole with dental acryl. However, due to the small size of the anterior horn of some of the rhinoceros, only two (Male 1 and 2) of the six rhinoceros were fitted with horn transmitters. Similarly, the rhinoceros were ear notched following the procedures described by Adcock and Emslie (2004) that briefly included cutting a “V” shaped notch at top, centre and/or bottom of either the left and/or right earlobe depending on the rhinoceros identity number. Notches on the left ear of the rhinoceros were read in “ones” (top being 1, centre being 3 and bottom being 5) while those on the right ear were read in “tens” (top being 10, centre being 30 and bottom being 50). For example, rhinoceros with identity number 35 (Female 2) was ear notched at the centre on the right ear and bottom on the left ear.

After fixing the transmitters and notching, the rhinoceros were then transported and released in the eastern sector of Ol Pejeta Conservancy. Post release monitoring was done by individual rhinoceros identification following the Kenyan rhino monitoring protocols (Amin *et al.*, 2006). This was possible since the rhinoceros had in addition to the ear notch and the horn transmitter, a name and an

identification number, thus making it possible to follow and record the daily activities of each one of the introduced white rhinoceros.

The rhinoceros were monitored from a vehicle and on foot daily for at least two weeks of every month between July 2006 and December 2008. The monitoring was done during early morning and late afternoon hours known to be the peak activity hours for white rhinoceros (Owen-Smith, 1973; White *et al.*, 2007; Pedersen 2009). Radio telemetry (using a handheld receiver and a two-element antenna) was used to help in locating the rhinoceros that had transmitters while spoor was used to help locate those without transmitters. The exact location of the rhinoceros was not reliant on the transmitter signals, but tracking continued till the rhinoceros was physically sighted as recommended by Pienaar *et al.* (1993) and Amin *et al.* (2006) where other identification features such as ear notching, natural scars, shape of the horns, were used to confirm the identity of the individual.

The white rhinoceros locations were recorded using a Garmin 12 XL GPS hand set.

Other information recorded included:

- i) Identification and ear notch numbers of the rhinoceros,
- ii) Number, if in a group, sex and age composition of the group members

Due to the low density of rhinoceros and the vast area of Ol Pejeta Conservancy, the rhinoceros could potentially travel long distances daily thus making physical sightings random and not guaranteed on a daily basis. When the rhinoceros were sighted, they were approached from downwind to minimize disturbance and allow time for individual identification.



### 3.4.1 Data synthesis

The GPS location fixes for each rhinoceros were counter checked to ensure that there was no duplications and that no more than two GPS locations were recorded for each individual rhinoceros or group in one day. Ten GPS fixes per individual or group are considered minimum for the estimation of range size (Conway and Goodman, 1989) for any period, although recent research work on home range of black rhinoceros has used 25 GPS fixes (Tatman *et al.*, 2000), while research into sample size of black rhinoceros home range analysis has suggested 35 GPS fixes (Lent and Fike, 2003). In the current study, the number of minimum GPS fixes per individual or group of rhinoceros that was required for home range analysis was 25. Data storage and synthesis were done using Microsoft excel while the home range sizes were constructed and calculated using the Arc View 3.3 animal movement extension and based on the Kernel method where 95% contour was used for home range and the 50% contour for core area estimates.

Home ranges for rhinoceros whose movements appeared to be very sporadic (e.g. Male 1 and 3 wet and dry season datasets) or whose GPS location sample size was less than 25 (e.g. Females 2 and 3 settling phase and dry season datasets) were not calculated as these data could not be reliably used for the Kernel estimator. All the dataset was organized into post release settling (July 2006 to December 2006) and intensive (January 2007 to December 2008) phase. The intensive phase was further split into wet and dry season.

Since the movements of calf 1 and 2 were always in close proximity to their mothers, Female 1 and 2 respectively as they were still yearlings (< 1 year old) for most of the study period, the calf and its mother were grouped as one. Information used for females was split as follows; Female 1: Five months (August 2006 to December 2006) post release settling phase as the rhinoceros settled followed by twenty three months (January 2007 to December 2008; excluding March 2008 data) post release intensive monitoring; Female 2: twenty months (March 2007 to December 2008; excluding June and July 2008 data) post release intensive monitoring; Female 3: thirteen months (December 2007 to December 2008) post release intensive monitoring. Both Females 2 and 3 were moved from Lewa Wildlife Conservancy and Kigio Wildlife Conservancy respectively to Ol Pejeta Conservancy after the post release settling phase (July 2006 to December 2006) had been completed. Thus the information used to determine preference and estimate home range sizes can be summarized as: three males and one female for the post release settling phase while four females and two males (three female and one male for the wet season and one male and one female for the dry season) were used for the post release intensive monitoring phase.

The adult males (Male 1, 2 and 3) were occasionally sighted with the females or alone thus their home ranges were calculated individually. Information for the males was split as follows; Male 1: six months (July 2006 to December 2006) post release settling phase as the rhinoceros settled followed by nineteen months (January 2007 to December 2008; excluding March, July, October, November and December 2008 data) post release monitoring; Male 2: five months (August 2006 to December 2006) post release settling phase as the rhinoceros settled followed by

twenty two months (January 2007 to October 2008) post release monitoring. This male died two months before the end of the study period. Male 3: five months (August 2006 to December 2006) post release settling phase as the rhinoceros settled followed by twenty two months (January 2007 to December 2008; excluding January and March 2008 data) post release monitoring.

Information emanating from the two monitoring phases (post release settling and post release monitoring) was synthesised and mapped per monitoring phase. Although the rainfall pattern during the monitoring phase (2007-2008) did not follow the long-term rainfall pattern (1996-2005) for Ol Pejeta Conservancy, the home range sizes for each rhinoceros or pair of rhinoceros were compared for the two monitoring phases. The monitoring phase was further split into wet (October - December) and dry (May - June) seasons for further comparison based on the amount of rainfall. The post release settling phase largely fell within the short rains (August – December) of 2006.

In regard to study site preference, the information was first processed by entering the GPS location fixes of all white rhinoceros sightings onto the map of the study area with the study sites using ArcView 3.3 and then exported to a database in order to ascertain which study site each GPS location fell within. Frequency of rhinoceros occurrence in each study site was then calculated for the study period and further categorized by season. Evaluation of study site preferences, as a consequence of habitat selection, by the southern white rhinoceros followed the method outlined by Viljoen (1989) and Pienaar *et al.* (1992; 1993a), where the preference index (PI) was calculated by comparing the proportion of southern

white rhinoceros occurrence in each study site throughout the study period and the proportion of the study site sizes within the study area. The preference index based on this method ranges from  $-1$  to  $+1$ , with  $-1$  indicating avoidance, a zero indicating random association (suggesting utilisation is in proportion to availability), and  $+1$  indicating preference. The four variables used to calculate these values were:

$n_x$  = the number of rhinoceros in study site 'x'.

$N_t$  = the total number of rhinoceros observed.

$a_x$  = the surface area of study site 'x' ( $\text{km}^2$ ).

$A_t$  = the total area available to the rhinoceros ( $\text{km}^2$ ).

$n_x/N_t$  = the proportion of rhinoceros recorded in study site 'x' relative to the total number of rhinoceros sightings.

$a_x/A_t$  = the proportion of the study area covered by study site 'x'.

IF  $n_x/N_t > a_x/A_t$ , then P.I. (x) =  $1 / (1 - a_x/A_t) \times (n_x/N_t - a_x/A_t)$ ,

but,

IF  $n_x/N_t < a_x/A_t$ , then P.I. (x) =  $1 / (a_x/A_t) \times (n_x/N_t - a_x/A_t)$ .

The preference index obtained does not provide an indication of statistical significance as it is only an indication of study site use against study site availability (Pedersen, 2009). Thus a chi-square test was conducted to test for goodness-of-fit of the study site use to study site availability as described by Byers *et al.* (1984) and Pienaar *et al.* (1992). In this case the proportion of expected

utilisation was compared to the confidence intervals, and those values that did not fit within the intervals were considered as significantly different.

### **3.5 Statistical analysis**

Data storage, sorting, initial processing, and analysis of the descriptive statistics were done using Microsoft excel while Minitab Statistical Software, Minitab Inc. (<http://www.minitab.com>) was used for the regression analysis for estimation of grass biomass and correlation analysis for establishing the relationship between faecal nitrogen and NDVI.

## CHAPTER FOUR

### RESULTS AND DISCUSSION

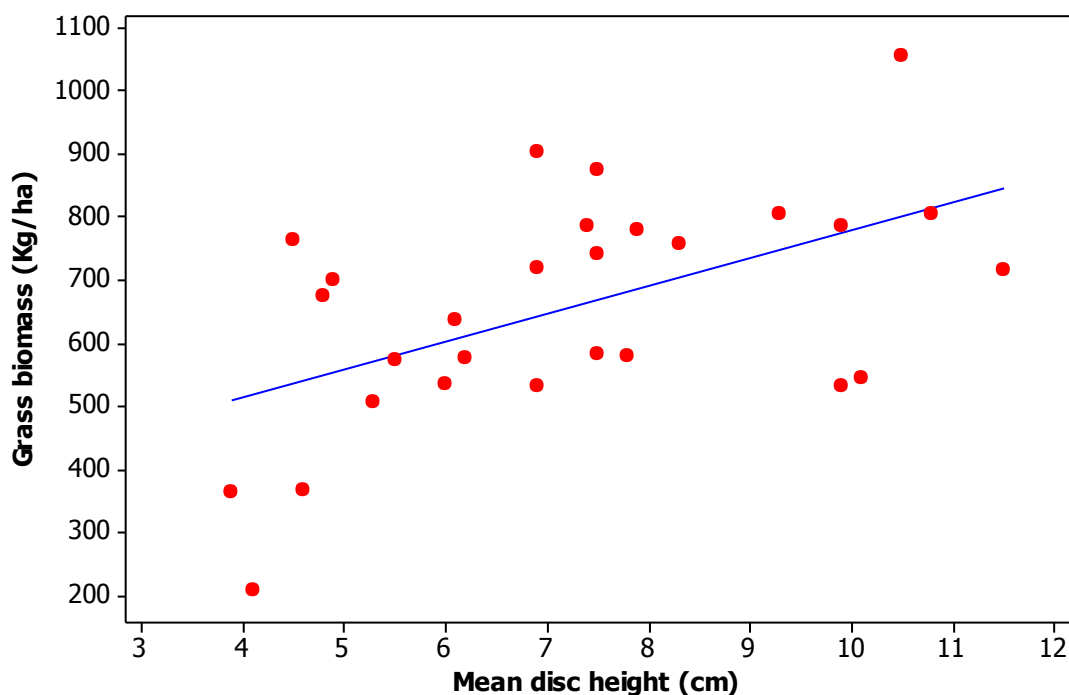
#### 4.1 GRASS BIOMASS IN OPC AND THE STUDY SITES

##### 4.1.1 Calibration of pasture disc meter and OPC grass biomass

The results for the regression of disc meter values and biomass measurements using Ol Pejeta Conservancy data are presented in Figure 4.1 which shows the scatter plot of the whole untransformed data set. A significant positive linear relationship ( $r = 0.41$ ;  $df = 26$ ;  $p = 0.004$ ) was obtained between grass biomass and mean disc height. The linear regression results of the rest of the transformed mean disc height and grass biomass are shown in Table 4.1. The table indicates that the reciprocal transformation ( $1/x$ ) of the disc height provided the best fit for the linear regression, with a significant negative correlation coefficient ( $r = -0.60$ ;  $df = 26$ ;  $p = 0.001$ ) (Table 4.1; Figure 4.2). The linear regression equation representing this calibration model is:  $y = -2247x^{-1} + 999$ , where:  $y$  = estimated biomass (kg/ha) and  $x$  = mean disc height (cm).

When compared to other calibration models obtained for South Africa, the correlation coefficient obtained in this study compared well to that obtained in Zululand Coastal plains (Brockett, 1996) and Nylsvley Nature Reserve (Dorgeloh, 2002) but not with that obtained in Sourish Mixed Bushveld (Theron, 1991) although the independent variable in the three studies in South Africa was untransformed mean disc height.

Similarly, the correlation coefficient obtained in this study is lower than that obtained for calibration models in Kruger National Park (Trollope and Potgieter,

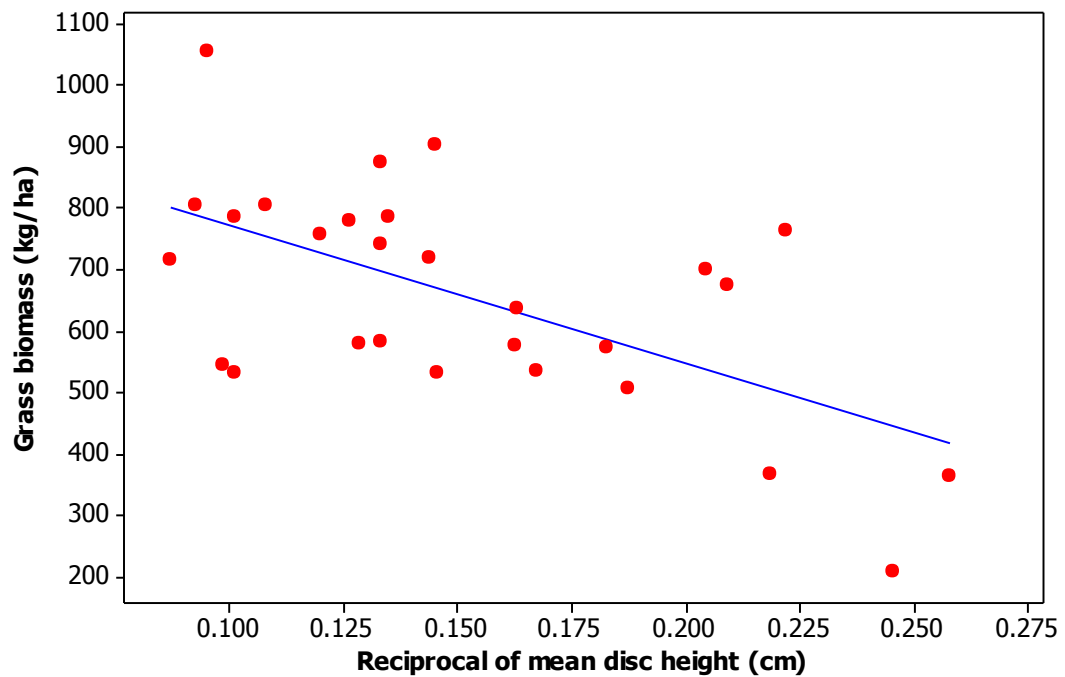


**Figure 4.1:** Scatter diagram for the linear regression between untransformed mean disc height and grass biomass (Source: Author, 2010)

**Table 4.1:** Linear regression between untransformed and transformed mean disc height and grass biomass respectively in Ol Pejeta Conservancy

Transformation	n	$r^2$	r	a	b	p-value
x	28	0.17	0.41	357.74	8.384	0.004
lnx	28	0.33	0.57	15.857	332.35	0.001
$x^2$	28	0.24	0.49	506.12	2.6744	0.008
$\sqrt{x}$	28	0.31	0.56	5.186	245.6	0.003
1/x	28	0.36	0.60	999	-2246.5	0.001

Key: x = mean disc height (cm); n = sample size; r = simple correlation coefficient;  $r^2$  = coefficient of determination; a = intercept/constant; b = slope



**Figure 4.2: Scatter diagram for the linear regression between reciprocal of the mean disc height and grass biomass used for calibration of disc pasture meter (Source: Author, 2010)**

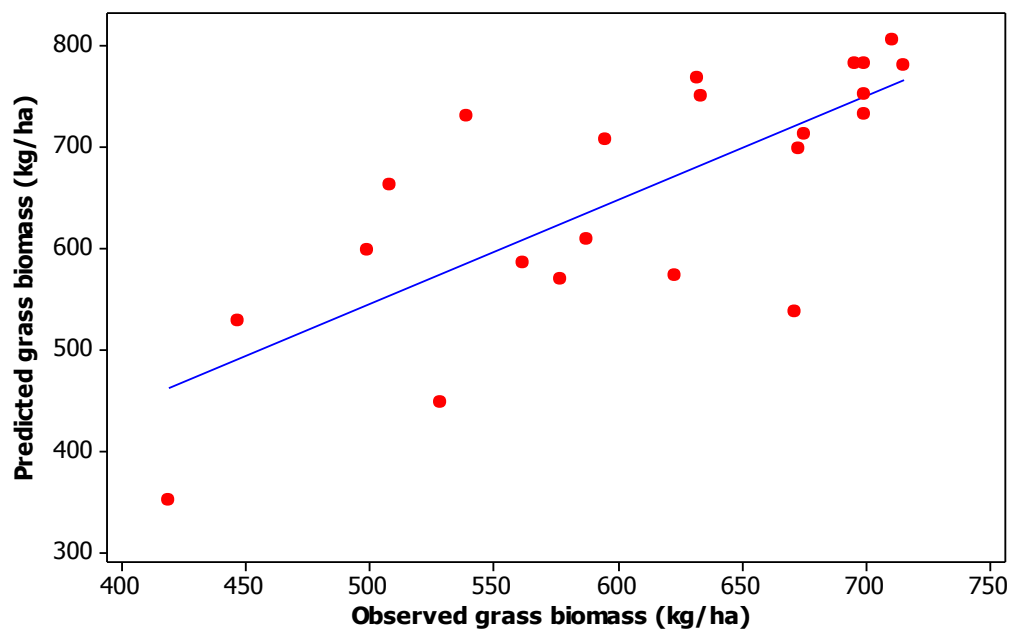
1986), Lewa Conservancy (Botha, 1999), and Arid and Semi-arid Lands (ASAL) areas of Northern Rangelands (Chege, 2004) even though the independent variable in the three studies was a transformed mean disc height. When the correlation coefficients of the calibration models obtained in this study using untransformed and transformed mean disc height are compared, that obtained from the untransformed mean disc height does not provide a model with the best fit.

Likewise among the different transformations done, the model with the best fit was obtained from the reciprocal transformation even though the natural log transformation ( $\ln x$ ) also had a significant correlation coefficient (same p-value with reciprocal transformation). These observations thus supports the need to calibrate the DPM using area-specific data as recommended by Mitchell, Elderkin



and Lewis (1993) before using it to estimate grass biomass in a particular area as well as the need to use both untransformed and transformed mean disc height as the independent variable in order to get a model with the best fit linear regression. The two processes i.e. calibration and using both untransformed and transformed mean disc height have the net effect of improving the grass biomass estimates.

When tested by regressing the observed grass biomass (independent variable) against the predicted grass biomass (dependent variable) using independent data from the 22 survey plots (Fig 4.3), the calibration model was found to be accurate with the disc pasture meter accounting for 75.1% of the variation in observed grass biomass ( $r = 0.751$ ;  $df = 21$ ;  $p\text{-value} = 0.001$ ). Similarly, there was no significant



**Figure 4.3: Scatter diagram for the linear relationship between observed and predicted grass biomass (Source: Author, 2010)**

difference between observed and predicted grass biomass ( $t = -1.52$ ;  $p\text{-value} = 0.135$ ) hence the calibration model was considered to be valid for use in Ol Pejeta Conservancy. The mean predicted grass biomass for Ol Pejeta Conservancy was  $700 \pm 434$  kg/ha at a mean disc height of  $7.51 \pm 0.35$  cm. Range of the data used in the regression analysis for calibrating the disc pasture meter was obtained under the conditions indicated in Table 4.2.

The mean predicted grass biomass for Ol Pejeta Conservancy compared well to  $738 \pm 33.31$  kg/ha obtained by Chege (2004) in his study of the areas outside protected areas within the arid and semi-arid lands in northern Kenya. However, the mean predicted grass biomass for Ol Pejeta Conservancy was half of that estimated by Chege (2004) ( $1492.11 \pm 76.11$  kg/ha) inside the protected areas within the arid and semi-arid lands in northern Kenya. Equally, the Ol Pejeta Conservancy estimate was a fifth of that obtained by Dorgeloh (2002) ( $3661.51 \pm 868.41$  kg/ha) in Nylsvley Nature Reserve. When compared to the Botha (1999) study sites in Lewa Wildlife Conservancy, the mean predicted grass biomass for Ol Pejeta Conservancy was four times lower than the lowest recorded ( $2769$  kg/ha) in the Lewa Wildlife Conservancy study sites and seven times lower than that recorded ( $4792$  kg/ha) in the short closed grassland study site of Lewa Wildlife Conservancy.

#### **4.1.2 Grass biomass in the study sites**

Grass biomass estimated in the six study sites is presented in Table 4.3. This indicates that the mean monthly grass biomass was highest in *Loldru* ( $642 \pm 169$

**Table 4.2: Range of the data used for the regression analysis**

Variable	Mean	Minimum	Maximum
Disc height (cm)	7.2	3.9	11.5
Grass biomass (kg/ha)	658	208	1057
Grass moisture (%)	52	29	76

**Table 4.3: Mean grass biomass estimates in the six study sites for the period February 2007 to December 2008**

	Estimated grass biomass (kg/ha) in the study sites					
	<i>Serat</i>	<i>Loldru</i>	<i>Morani</i>	<i>Zebra</i>	<i>Oryx</i>	<i>Grants</i>
Mean	510	642	461	389	266	638
S.E	211	169	171	140	106	97
Maximum	822	832	804	678	437	812
Minimum	100	451	250	62	100	437

Key: S.E. = Standard Error

kg/ha) and lowest in the Oryx ( $266 \pm 106$  kg/ha), with a range of 451 kg/ha to 832 kg/ha and 100 kg/ha to 437 kg/ha respectively. The maximum grass biomass estimated in Oryx (437 kg/ha) and Zebra (678 kg/ha) was lower than the mean predicted grass biomass for Ol Pejeta Conservancy. According to Vesey-Fitzgerald (1965) grazing lawns consist of short grasses ( $\leq 7$ cm) that are at a productive juvenile growth form. This growth form also signifies nutritious forage as it has less fibre content and more nitrogen content. Grass height, especially in its growing phase, is also indicative of grass biomass such that short grass implies low grass biomass and tall grass high grass biomass, with a variety of mesoherbivores (herbivores  $< 1000$ kg) preferring grazing lawns (Waldram *et al.*, 2007). The low grass biomass in *Oryx* and *Zebra* sites compared to the other four study sites can thus be interpreted as a result of high quality forage (reflected in low grass biomass

in these two sites that they were preferred by diverse herbivores over the other four sites, thus leading to a sustained lower grass biomass).

Consequently, the mean predicted grass biomass of Ol Pejeta Conservancy represents better forage quality (lower grass biomass) study sites compared to those in Lewa Wildlife Conservancy, inside the protected areas within the arid and semi-arid lands in northern Kenya and Nylsvley Nature Reserve.

The successful calibration of the disc pasture meter has a practical value for pasture management within Ol Pejeta Conservancy as grass biomass can be estimated rapidly thus setting for stocking rates or assessing the Conservancy's capability to support a specific number of grazing animals can be achieved. The grass biomass estimates can also be used to guide pasture management interventions in Ol Pejeta Conservancy such as prescribed burning and intensive grazing by livestock, assessing hydrologic properties and value of any site within the Conservancy with regards to its suitability for any grazing animals. The disc pasture meter will thus provide a practical, low cost means of undertaking long term assessment of grass biomass within Ol Pejeta Conservancy and its surrounding environs.

## **4.2 THE RELATIONSHIP BETWEEN FAECAL NITROGEN AND NORMALISED DIFFERENCE VEGETATION INDEX**

### **4.2.1 Normalised difference vegetation index and percentage faecal nitrogen**

A list of common grass species as recorded in the four study sites is presented in Table 4.4 and shows that *Oryx* site recorded the highest number (85%, n = 13) of

**Table 4.4: Grass species commonly found in the four study sites**

Grass species	Loldru site	Morani site	Oryx site	Zebra site
<i>Eragrostis tenuifolia</i>	√	√	√	√
<i>Sporobolus discosporus</i>	√	√	√	√
<i>Aristida keniensis</i>	√			√
<i>Microchloa caffra</i>	√	√	√	
<i>Pennisetum stramineum</i>	√	√	√	√
<i>Sporobolus africanus</i>	√		√	√
<i>Cynodon dactylon</i>	√	√	√	√
<i>Hyperchni schimperi</i>	√		√	
<i>Themeda triandra</i>	√	√	√	√
<i>Sporobolus pyramidalis</i>		√	√	
<i>Panicum maximum</i>		√		
<i>Bothriocloa insculpta</i>		√	√	√
<i>Pennisetum mezianum</i>		√	√	√

Key: √ = present in site

these common grass species while Zebra and Loldru sites recorded the least number (69%, n = 13). Normalised difference vegetation index values were processed for four out of the six study sites, that is, sites that had both NDVI and faecal nitrogen data. The highest NDVI ratio (0.526) was recorded in Zebra site while the lowest was recorded in Loldru site (0.4802). Table 4.5 indicates that the mean percentage faecal nitrogen (2.664, S.E = 0.475) was highest in the Zebra site and lowest in the Loldru site (1.928; S.E = 0.053).

While Zebra and Lodru sites had the same percentage of common grass species found in the four study sites, the grass species composition within these sites differed. Equally the NDVI values and the percentage faecal nitrogen for these two sites differed with Zebra site recording higher values compared to Loldru site. Similarly Oryx site with the highest percentage of the common grass species in these sites

recorded lower values for both NDVI and faecal nitrogen than Zebra site. Also of importance to note is that *Panicum maximum* was only recorded in Morani study site, while *Sporobolus africanus* was only absent in the same study site. Santos et al. (2006) reported that *Panicum maximum* had high nutritive value while Andrews (1995) observed that *Sporobolus africanus* had significant weedy characteristics with low nutritive value and was unpalatable to most grazers.

**Table 4.5: Faecal nutritive value (%FN) of zebra within the four study sites**

Study site	Mean %FN	S.E.
<i>Morani</i>	2.247	0.313
<i>Loldru</i>	1.928	0.053
<i>Oryx</i>	2.021	0.193
<i>Zebra</i>	2.664	0.475

Key: FN = Faecal nitrogen; S.E. = standard error

Comparing the findings of this study to those of Santos *et al.* (2006) and Andrews (1995), *Morani* site would have been expected to record the highest percentage of faecal nitrogen compared to the other sites considering that *Panicum maximum* was found in only this site and *Sporobolus africanus* absent in only this site. However findings of this study showed that *Zebra* site recorded the highest percentage faecal nitrogen followed by *Morani* site.

#### **4.2.2 Relationship between faecal nitrogen and normalised difference vegetation index**

Correlation analysis was done for each of the four study sites that had both NDVI and percentage faecal nitrogen values (Table 4.6). There was a strong positive linear

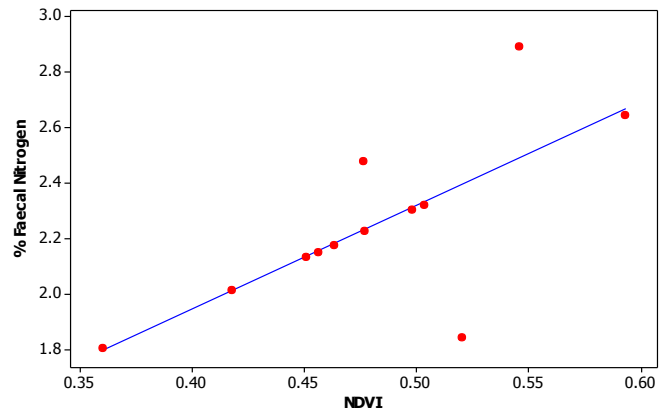
**Table 4.6: Pearson correlation coefficients across the different study sites**

Study site	Pearson correlation	p-value
<i>Morani</i>	0.714	0.009
<i>Loldru</i>	-0.9	0
<i>Oryx</i>	0.996	0
<i>Zebra</i>	0.997	0

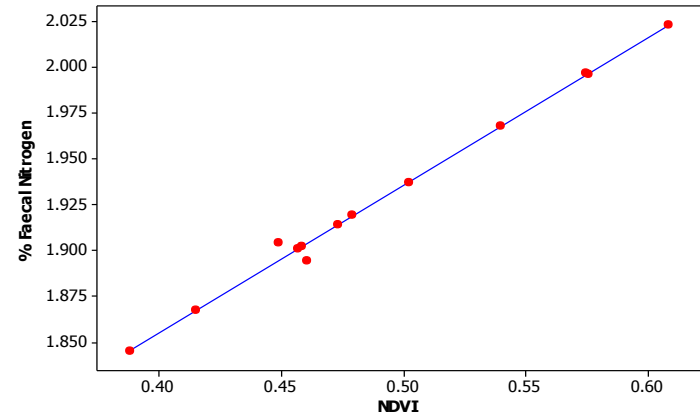
relationship between NDVI and percentage faecal nitrogen in all the study sites except *Loldru* site which showed a strong negative linear relationship ( $r = -0.9$ ;  $p < 0.001$ ) (Table 4.6, Fig 4.4) described by the equation:  $y = 3.154 - 2.152x$  where  $y$  represents the % faecal nitrogen and  $x$  the NDVI.

Despite the fact that the information on NDVI obtained from satellite imagery is usually qualitative, results obtained in this study show that there is a strong relationship with a quantitative measure of habitat quality, the faecal nitrogen. This observation is in line with Ryan *et al.* (2003) study in Kruger National Park where a positive correlation between NDVI and faecal nitrogen was obtained for African buffalo. In Okavango Delta, Stowe (2003) also reported nitrogen levels in buffalo dung that were within standard deviations of the baseline, indicating that faecal nitrogen can be used as an indicator of habitat quality.

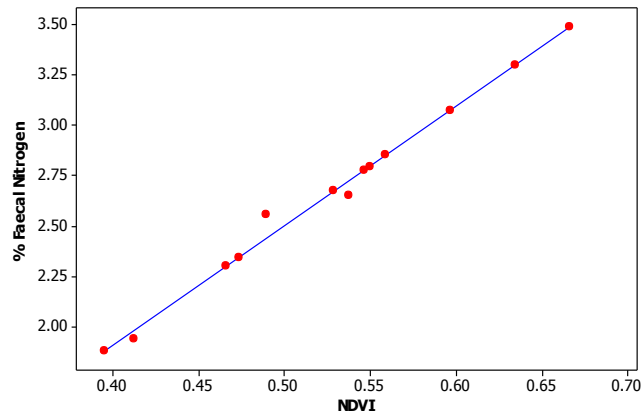
Liversidge and Berry (1996) observed that wild ungulates, e.g. white rhinoceros and common zebra, need a minimum of 0.8% nitrogen in their diet to maintain body weight. In the case of Ol Pejeta Conservancy, the lowest concentration of nitrogen was 1.93% implying that the common zebra, whose faecal samples were used, had



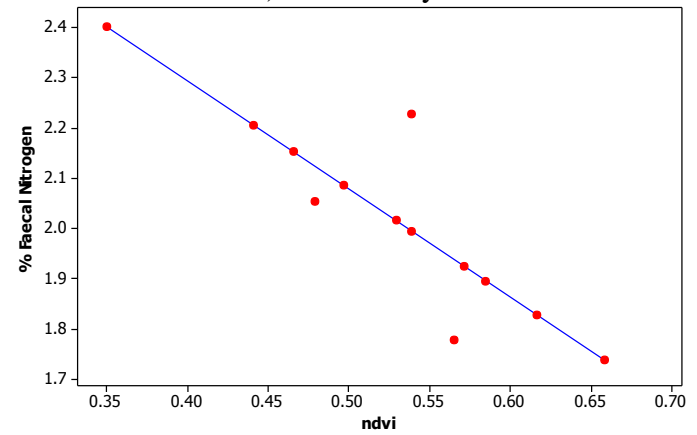
a) Morani study site



b) Zebra study site



b) Oryx study site



d) Loldru study site

**Figure 4.4: Linear correlation between NDVI and percentage faecal nitrogen in the four study sites (Source: Author, 2010)**



adequate forage quality within the *Loldru* site but to a higher forage quality in *Morani*, *Oryx*, and *Zebra* sites, which exhibited higher faecal nitrogen.

Although a relationship between NDVI and faecal nitrogen exists in *Loldru* site, it was a negative relationship due to relatively more forbs within this site compared to the other three sites thus enhancing the reflectance and consequently the NDVI values while in reality the forbs were unpalatable and not being grazed by the common zebra thus the percentage faecal nitrogen in this site does not increase proportionally to the perceived increase in NDVI values. This observation is supported by Pettorelli *et al.* (2005) caution on interpretation of NDVI values in monitoring herbivore habitat quality as they may inaccurately represent productivity due to the difference in reflectance in heterogeneous habitats, such as those with interspersed woody and herbaceous vegetation (Elvidge and Lyon, 1985; Huete, Jackson and Post, 1985; Huete and Tucker, 1991).

Given that the results of this study demonstrated a significant relationship between faecal nitrogen and NDVI, it is suggested that NDVI data could provide a surrogate measure of habitat quality for common zebra and therefore the southern white rhinoceros with respect to crude protein. Moreover, the finding supports the use of NDVI data in understanding the effects of habitat quality on population parameters, individual life-history traits, and habitat use as illustrated by other studies (Garel *et al.*, 2006; Herfindal *et al.*, 2006; Pettorelli *et al.*, 2006; Gustine *et al.*, 2006; Marshal *et al.*, 2006; Wittemyer, Rasmussen and Douglas-Hamilton, 2007;

Wiegand *et al.*, 2008). More recently Hamel *et al.* (2009) has demonstrated that a multi-year time series of NDVI can reliably measure yearly changes in the timing of the availability of high-quality vegetation for temperate herbivores. Likewise Ryan *et al.* (2003) concluded that faecal nitrogen is highly correlated with forage digestibility, dietary protein concentration, intake and changes in live mass of cattle.

The process of quantifying the study sites habitat quality therefore required the calibration of a disc pasture meter, estimation of grass biomass as well as determining a relationship between NDVI and faecal nitrogen measure. These are of significant value for continued effective monitoring of pasture in these study sites considering that both wildlife and livestock exist in the Conservancy with the two major grazing species, boran cattle and common zebra, accounting for over 60% of the herbivore biomass on the Conservancy (Games, 2008).

The suggestion in Section 4.1.2 that *Oryx* and *Zebra* sites might have had better forage quality (low grass biomass) than the other four sites thus being preferred by diverse herbivores and leading to a sustained lower forage biomass, is supported by the finding in Section 4.2.1 which reveals that these two sites including *Morani* had higher forage quality that translated to a higher faecal nitrogen in these sites. Further, findings in Section 4.1 and 4.2 show that as the forage biomass in the different sites increase, the nitrogen content of forage decreases, implying that forage quality declines with increase in forage quantity.

### 4.3 POST RELEASE HABITAT PREFERENCE OF THE SOUTHERN WHITE RHINOCEROS

A total of 210 GPS location fixes of rhinoceros sighting were collected and used for subsequent analysis of habitat use and preference. The results indicate that the *Grants* site recorded the highest use (19%) when data for the six study sites were pooled for the entire study period as well as when data were categorized into wet (19%) and dry (52%) seasons, even though only two sites were used in the dry season (Table 4.7). The results further indicate that the *Grants* site was the least preferred (PI ranking) site in the dry (PI = 0.833), wet (PI = 0.055), as well as when all data for study period were pooled (PI = 0.447). The wet season preference index indicates that the site was used in proportion to availability (PI = 0.055). The finding that *Grants* site was the most utilized in all the seasons and the least preferred in all the seasons is in line with the findings of Pedersen (2009) that the Baobab/Mopane landscape, which was the most utilized landscape in three out of four seasons, was only preferred in one season and avoided in the others.

Overall preference for the six study sites by the southern white rhinoceros during the study period is illustrated in Figure 4.5. The results indicate that five out of the six sites were preferred during the study period albeit with varying preference indices. When all the data were pooled, *zebra* and *Morani* sites recorded the highest preference index (PI = 1) followed by *Oryx* site (PI = 0.867), then *Serat* (PI = 0.809) and *Grants* (PI = 0.447). During the dry season, only two sites were preferred with

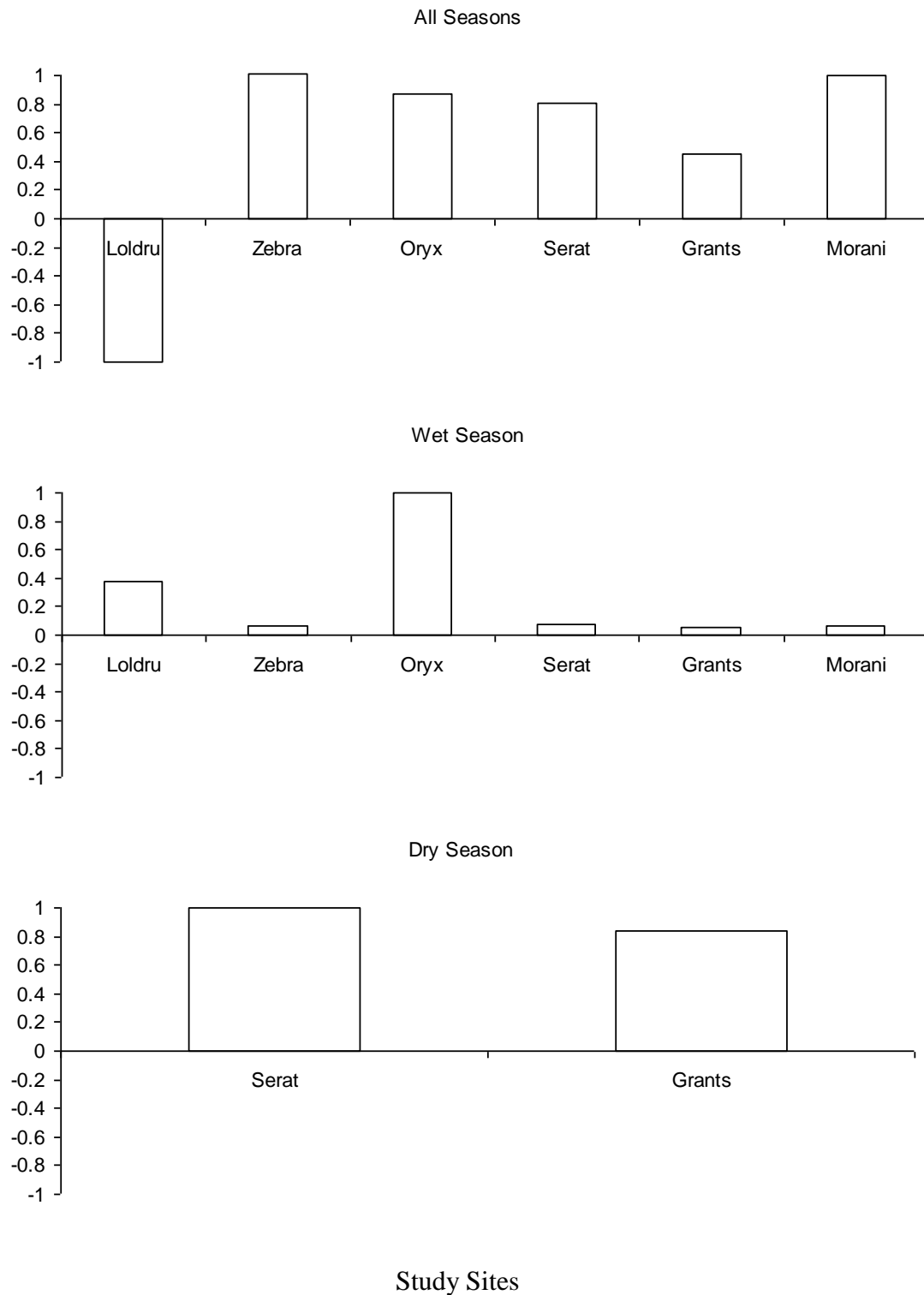
**Table 4.7: The proportion of occurrence of all rhinoceros per the study site, season, and their preference index ratings**

Study site	Site size	Wet season (N=117)		Dry Season (N=93)		Pooled data (N=210)	
	%	%	PI	%	PI	%	PI
<i>Loldru</i>	23.9	15.0997151	0.374268	-	-	16.35802	-1.00057
<i>Zebra</i>	10.4	18.5185185	0.064365	-	-	18.20988	1.007853
<i>Oryx</i>	6.8	15.0997151	1.000945	-	-	15.24691	0.867483
<i>Serat</i>	6.7	15.954416	0.071814	47.31183	1.0000112	15.61728	0.809227
<i>Grants</i>	4.9	19.3732194	0.054724	52.68817	0.83354464	18.95062	0.446929
<i>Morani</i>	4.0	15.954416	0.067784	-	-	15.61728	1.004383

*Serat* site being highest preferred (PI = 1), followed by *Grants* site (PI = 0.834) while the other four sites were not utilized at all. In the wet season, *Oryx* site was highest preferred (PI = 1) followed by *Lodru* site (PI = 0.374), while *Serat*, *Morani*, *Zebra* and *Grants* sites had preference indices close to zero. *Loldru* site was, however, avoided (PI = -1) when all seasons (pooled data) were considered.

Table 4.8 summarizes the significance of preference or avoidance during the study period. It shows that the *Morani*, *Zebra*, *Oryx*, *Serat* and *Grants* sites were considered as significantly preferred ( $p = 0.05$ ), and thus the null hypothesis that study sites utilization occurs in proportion to their availability, was rejected. The hypothesis was also rejected for the *Loldru* site as it was also significantly avoided. Thus none of the sites appeared to be utilised in proportion to their availability.

*Zebra* site was the most preferred study site when data for all seasons were pooled. This site is characterized by not only high quality forage signified by low grass biomass, but also more water sources that include a permanent water trough and two *Serat* site being highest preferred (PI = 1), followed by *Grants* site (PI = 0.834) while the other four sites were not utilized at all. In the wet season, *Oryx* site was highest preferred (PI = 1) followed by *Lodru* site (PI = 0.374), while *Serat*, *Morani*, *Zebra* and *Grants* sites had preference indices close to zero. *Loldru* site was, however, avoided (PI = -1) when all seasons (pooled data) were considered.



**Figure 4.5: The preference index rating for each study site in the study area, reflecting preference (+1) and avoidance (-1) per season and for all the seasons combined (Source: Author, 2010)**

Table 4.8 summarizes the significance of preference or avoidance during the study period. It shows that the *Morani*, *Zebra*, *Oryx*, *Serat* and *Grants* sites were considered as significantly preferred ( $p = 0.05$ ), and thus the null hypothesis that study sites utilization occurs in proportion to their availability, was rejected. The hypothesis was also rejected for the *Loldru* site as it was also significantly avoided. Thus none of the sites appeared to be utilised in proportion to their availability.

*Zebra* site was the most preferred study site when data for all seasons were pooled. This site is characterized by not only high quality forage signified by low grass biomass, but also more water sources that include a permanent water trough and two streams, with one being right on the edge of the site. Studies by Lock (1972); Olivier and Laurie (1974) and Owen-Smith (1975) reported that white rhinoceros prefer to feed in areas of short grass (signifying low grass biomass), a habit to which they are suited by their wide mouth and low slung head. The streams in this study site, which are run-offs from the Ewaso Nyiro River, were used a lot when the rhinoceros were moving between water sources, and vegetation along them also acted as good sources of shade for resting, as well as grazing. Based on these results it is clear that the study sites were not utilized in proportion to their availability as they were either preferred or avoided, like *Loldru* site.

According to White (1983), the net gain of using a resource may to some extent be influenced by variations in quantity and quality of the resource in relation to costs

**Table 4.8: Study site habitat preference or avoidance by white rhinoceros during the study period**

Study Site	$\chi^2$	Confidence Interval	Expected utilisation	Observed utilisation	Remarks
<i>Loldru</i>	104.676	$0.1635769 \leq p \leq 0.1635835$	0.239282303	0.1635802	avoided
<i>Zebra</i>	2134.698	$0.1820949 \leq p \leq 0.1821027$	0.103971099	0.1820988	Preferred
<i>Oryx</i>	22.046	$0.1524661 \leq p \leq 0.1524721$	0.067847995	0.1524691	Preferred
<i>Serat</i>	149.654	$0.1561697 \leq p \leq 0.1561759$	0.067149617	0.1561728	Preferred
<i>Grants</i>	8354.860	$0.189502 \leq p \leq 0.1895104$	0.049059719	0.1895062	Preferred
<i>Morani</i>	149.654	$0.1561698 \leq p \leq 0.156176$	0.039792542	0.1561729	Preferred

associated with searching for and exploiting the resource. Yarrow (2009) observed that while most animals can survive for weeks without food, they can only survive for days without water as they need it for digestion and metabolism, reducing body temperature and removal of metabolic wastes among others. Thus the move by all white rhinoceros to the *Serat* and *Grants* sites during the dry season was indicative of the declining water availability in the other study sites and also in support of Yarrow (2009) observation that water is critical.

Although there is the Ol Pejeta dam around *Oryx* site, there is no water trough and the only stream dries up during dry season leaving the dam as the only source of water for all the other herbivores using this site. The location of *elephant* dam, a permanent water source, within *Grants* site in addition to the permanent water trough and the two streams might have helped in drawing rhinoceros to this site during the dry season. However, the proximity of the water trough and the two streams in *Serat* compared to the *Grants* site made it more preferred than the *Grants* site as the cost for searching



this resource would be less. This finding, therefore, shows that during the dry season, the white rhinoceros did not utilize the study sites in proportion to their availability as both *Serat* and *Grants* sites were preferred. The findings are also in line with Smit *et al.* (2007) study which found that white rhinoceros were more often associated with areas close to waterholes.

All study sites were utilized during the wet season with Oryx being the most preferred. During this period, water was easily available in all the streams in addition to that from moisture and lush green herbaceous vegetation in all the study sites. *Oryx* site was preferred during this season due to the fact that the stream and the Ol Pejeta dam are close by as well as the numerous mud wallowing areas available in this site during the wet season. This finding also follows that observed by Owen-Smith (1973; 1988) and further suggests that the utilization of most of the study sites during the wet season was in proportion to their availability.

According to Redfern *et al.* (2003), African savanna herbivores must meet their daily dietary requirements within the limitations set by surface water availability. In this study, *Serat* site include one of the four most frequently utilised study sites as white rhinoceros were seen to utilise it on a regular basis throughout the study period. Despite rhinoceros only needing to drink every 2 – 3 days, they will drink daily if water is readily available (Owen-Smith 1973; 1988), as observed during the wet season of this study. These results further support the suggestion by Owen-Smith (1988) that in the driest parts of their historical range, habitat preferences by white rhinoceros are influenced by available surface water, as a substitute for moisture gained from fresh, green graze.

#### 4.4 HOME RANGE SIZE OF THE TRANSLOCATED SOUTHERN WHITE RHINOCEROS

##### 4.4.1 Post release settling phase home range size

A total of 540 GPS location fixes were recorded throughout the post release monitoring phases with 270 GPS fixes being within the settling phase while 270 GPS fixes were within the rest of the monitoring phase. The four rhinoceros that contributed to the settling phase GPS fixes were; Female 1 and Males 1, 2, 3.

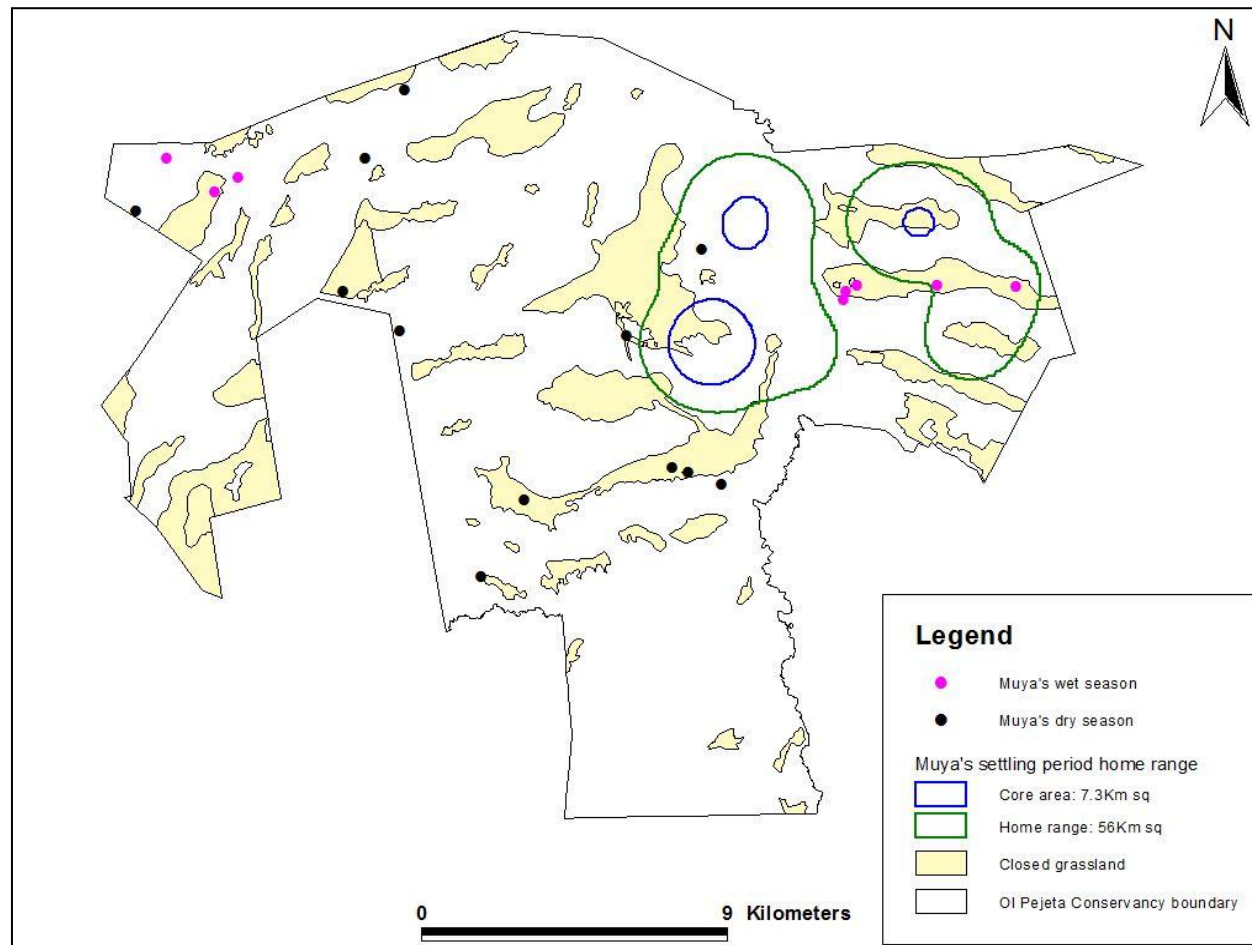
During the post release settling phase, Male 3 had the largest mean home range of 56km<sup>2</sup> with a mean core area of 7.3km<sup>2</sup> while Female 1 exhibited the smallest mean home range of 23km<sup>2</sup> with a mean core area of 2.7km<sup>2</sup> (Table 4.9). Male 3

**Table 4.9: Mean home range and core area estimates for the four rhinoceros during the post release settling monitoring phase – July 2006 to December 2006**

White rhinoceros	Mean Home Range (km <sup>2</sup> )	Mean Core Area (km <sup>2</sup> )	GPS fixes
Female 1	23	2.7	47
Female 2	Nil	Nil	Nil
Female 3	Nil	Nil	Nil
Male 1	30	3.1	116
Male 2	36.9	4.2	55
Male 3	56	7.3	52
Total GPS fixes			270

Female 1: Ariemet; Female 2: Daly; Female 3: Susan; Male 1: Kingi; Male 2 Namunyak; Male 3: Muya; Nil = < 25 GPS fixes thus not used in analysis

moved through the eastern sector of the Conservancy and established its core areas in the western sector (Fig 4.6). Similarly, Male 1 moved through the northern area



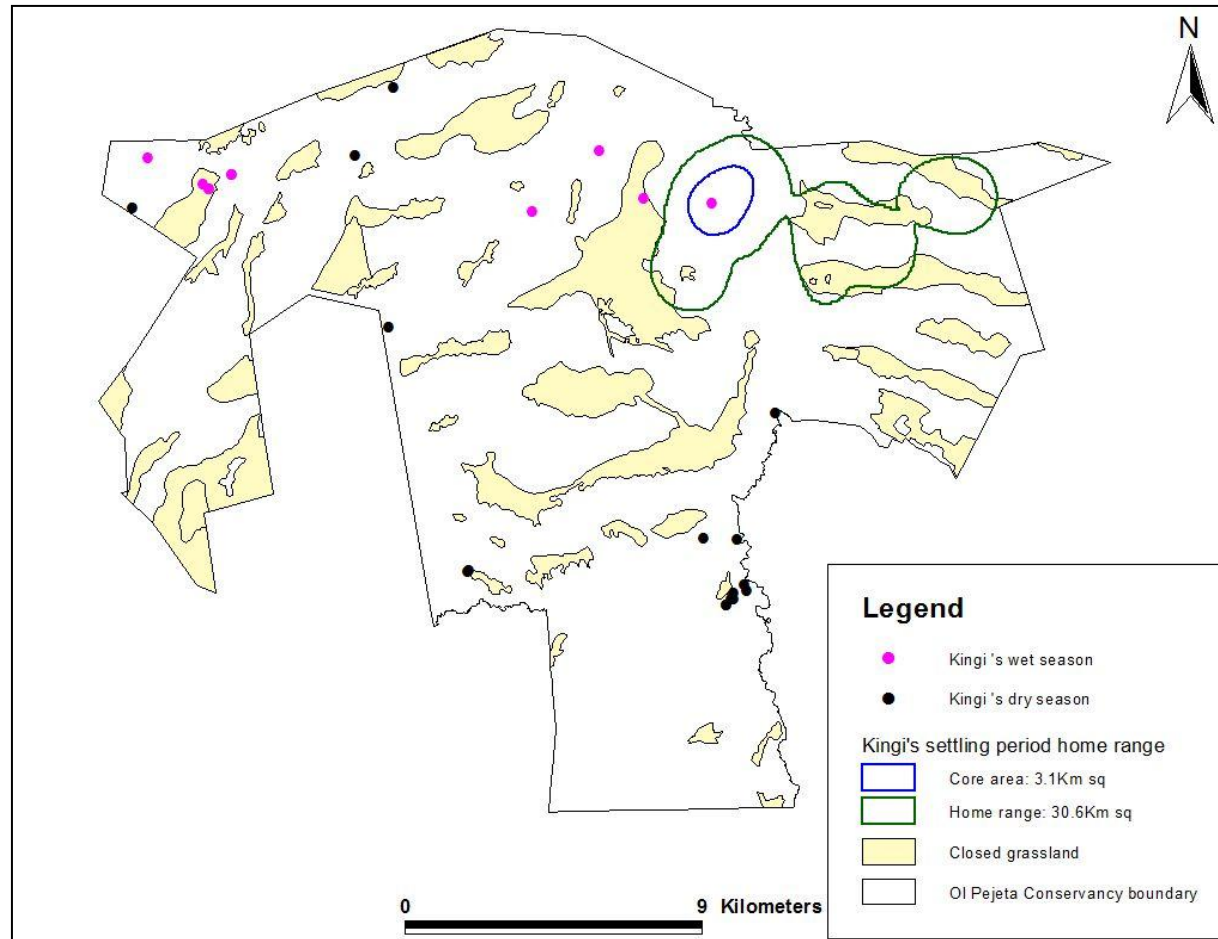
**Figure 4.6: Home range and core area covered by Male 3 during the post release settling monitoring phase – July 2006 to December 2006 (Source: Author, 2010)**

of the eastern sector and established its mean core area of 3.1km<sup>2</sup> in the western sector of the Conservancy (Fig 4.7). Male 2 on the other hand settled in the eastern sector of the Conservancy with its mean home range of 36.9km<sup>2</sup> and a mean core area of 4.2km<sup>2</sup> being around the centre of the eastern sector of the Conservancy and overlapping with that of Female 1 (Fig 4.8).

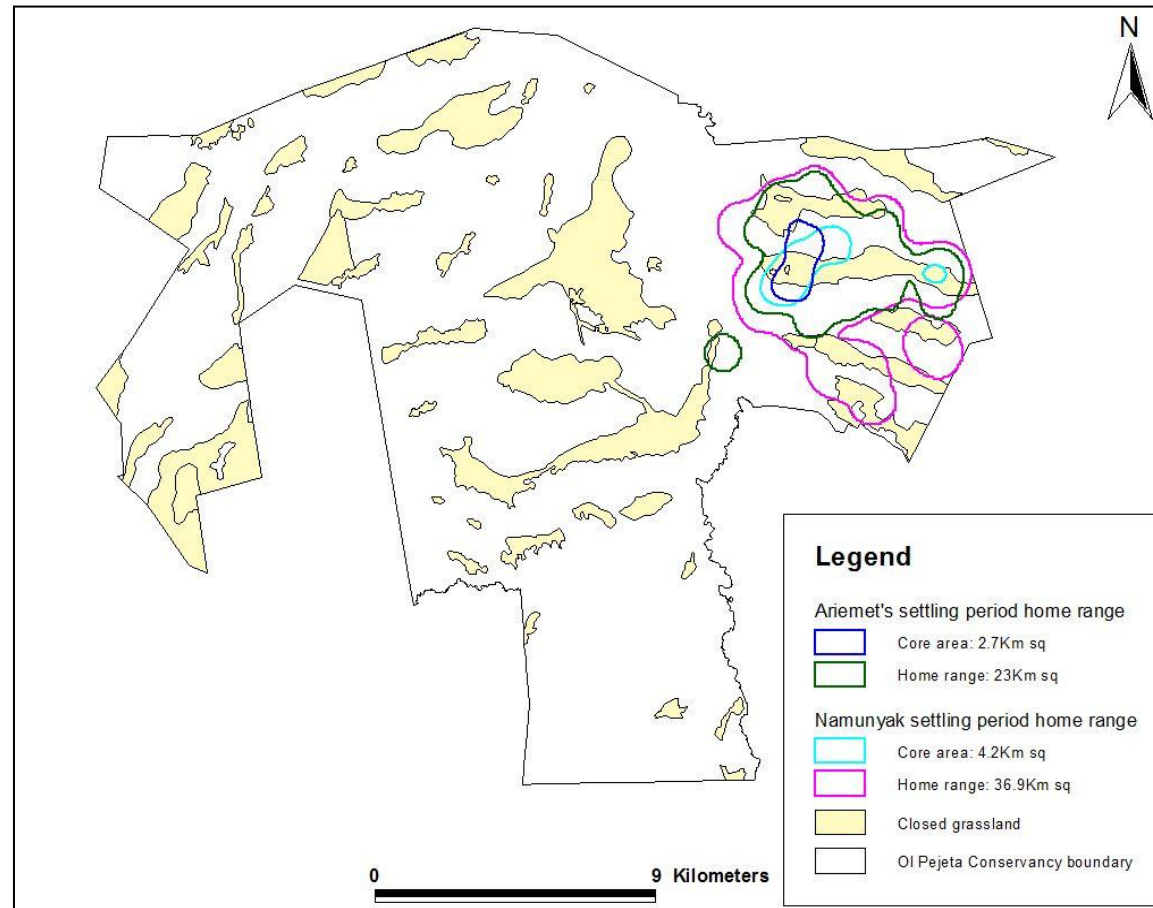
#### **4.4.2 Post release monitoring phase home range size**

The post release monitoring phase was split into wet and dry season where 270 GPS fixes were recorded with 177 fixes being in the wet season and 93 fixes in the dry season. Four rhinoceros (Females 1, 2, 3 and Male 2) contributed to the 177 GPS fixes for the wet season and two (Female 1 and Male 2) contributed to the 93 GPS fixes for the dry season (Table 4.10). Female 2 had the largest mean home range (25.4km<sup>2</sup>) and core area (4.5km<sup>2</sup>) while Female 1 had the smallest mean home range (22 km<sup>2</sup>) and mean core area (2.7km<sup>2</sup>) in the wet season. The mean home range of Female 2 was close to that of all the four rhinoceros combined in wet season (25.7km<sup>2</sup>) and its mean core area was larger than that of the four rhinoceros in this season (2.7km<sup>2</sup>). During the dry season, Male 2 exhibited a larger mean home range (29.7km<sup>2</sup>) and mean core area (3.8km<sup>2</sup>) than Female 1.

Home range results indicate an overlap of core areas by Female 1 and Male 2 in the two seasons (Fig 4.9 & 4.10). The dry season mean home range for all rhinoceros was not significantly larger than the wet season ( $\chi^2 = 0.066$ , d.f. = 1,  $p > 0.05$ ). There was a significant difference in the Female-Male home range across the season ( $\chi^2 = 3.866$ , d.f. = 1,  $p < 0.05$ ) with the dry season recording larger home



**Figure 4.7: Home range and core area covered by Male 1 during the post release settling Monitoring phase – July 2006 to December 2006 (Source: Author, 2010)**

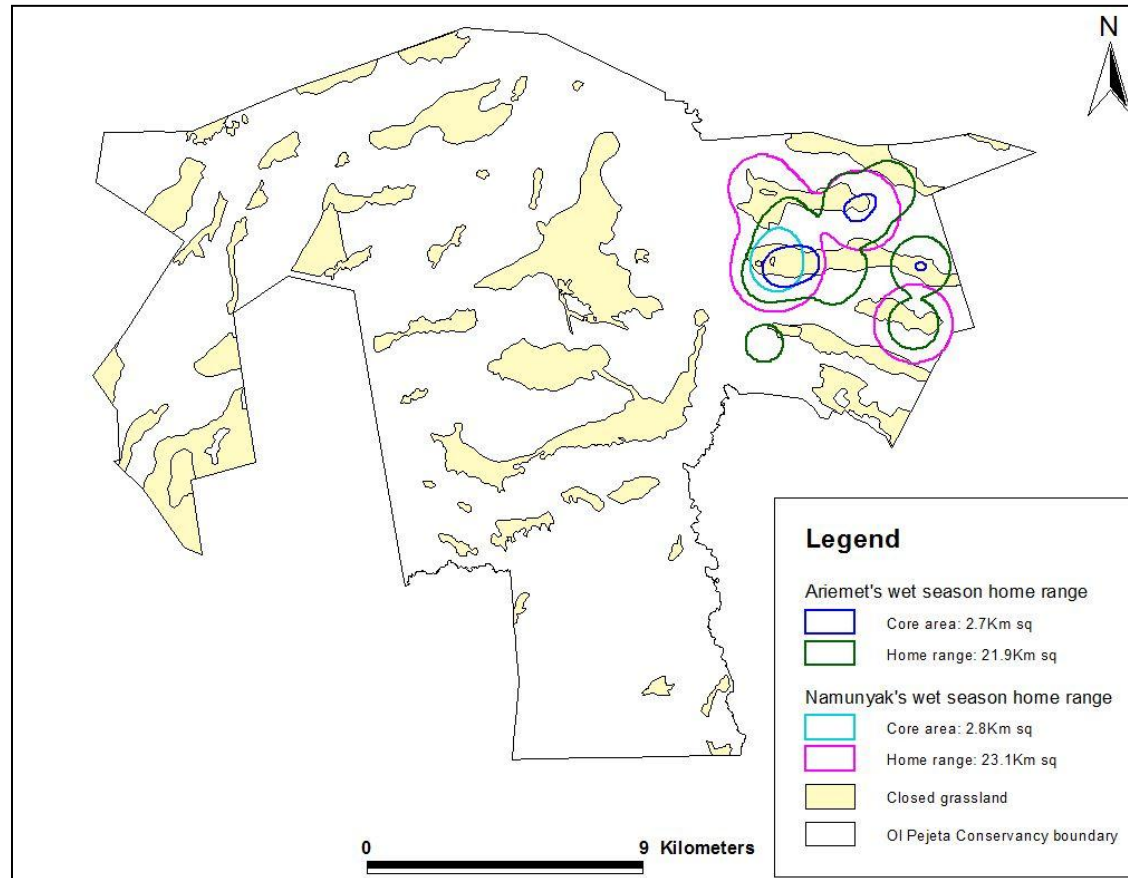


**Figure 4.8: Home range and core area covered by Male 2 and Female 1 during the post release settling monitoring phase – July 2006 to December 2006 (Source: Author, 2010)**

**Table 4.10: Sizes of home ranges and core areas by white rhinoceros during the post release monitoring phase – January 2007 to December 2008**

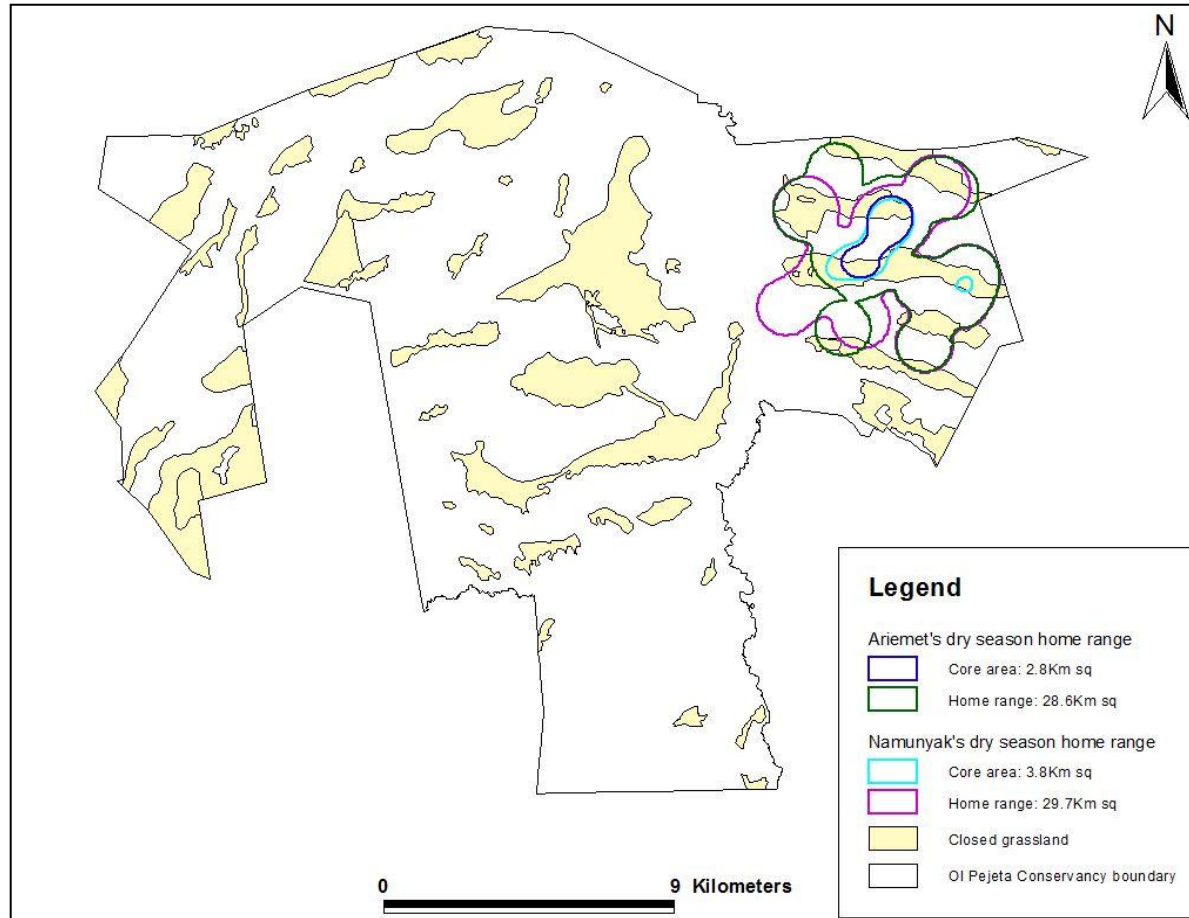
White rhinoceros	Wet season		GPS fixes	Dry season		GPS fixes	Total GPS fixes
	Home Range (km <sup>2</sup> )	Core Area (km <sup>2</sup> )		Home Range (km <sup>2</sup> )	Core Area (km <sup>2</sup> )		
Female 1 <i>d=7</i>	22	2.7	60	28.6	2.8	27	87
Female 2 <i>d=5.1</i>	25.4	4.5	30	Nil	Nil	3	33
Female 3 <i>d=5.6</i>	24	3.8	34	Nil	Nil	5	39
Male 1 <i>d=19.6</i>	Nil	Nil	10	Nil	Nil	17	27
Male 2 <i>d=7.2</i>	23.1	2.8	32	29.7	3.8	29	61
Male 3 <i>d=22.3</i>	Nil	Nil	11	Nil	Nil	12	23
All rhinoceros	25.7	2.7	177	27	2.4	93	270

*d* = distance (km) between the two most distant GPS fixes in each dataset; Nil = females with < 25 GPS fixes and males with sporadic movements thus not used in analysis



**Figure 4.9: Home range sizes and core areas covered by Female 1 and Male 2 during the wet season of post release monitoring phase – January 2007 to December 2008 (Source: Author, 2010)**

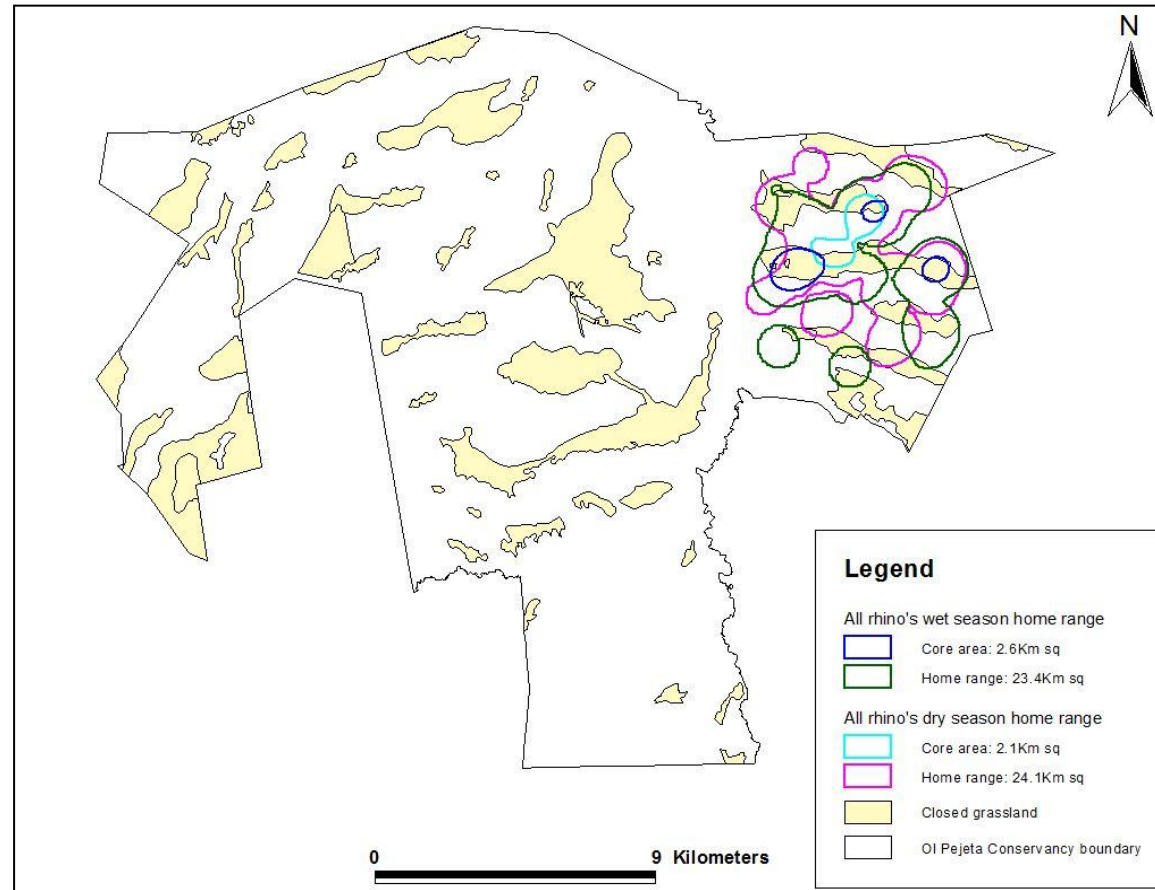




**Figure 4.10: Home range sizes and core areas covered by Female 1 and Male 2 during the dry season of post release monitoring phase – January 2007 to December 2008 (Source: Author, 2010)**

range than the wet season. Within seasons, the Female-Male home ranges did not differ significantly (wet season:  $\chi^2= 0.021$ , d.f. = 1,  $p > 0.05$  and dry season:  $\chi^2= 0.042$ , d.f. = 1,  $p > 0.05$ ). Females 1, 2, 3 and Male 2 concentrated in the eastern sector of the Conservancy with their wet season sightings being evenly distributed in the sector, and the dry season core areas avoiding the southern part of the sector (Fig 4.11). Male 1 and 3 utilized the western sector of the Conservancy throughout the post release monitoring phase concentrating towards the northern part of the sector in the wet season and moving to the southern part during the dry season. overlaps in home ranges were seen for all the Females and Male 2 during wet season.

This study was conducted in a low white rhinoceros density ( $0.02$  rhinos/km<sup>2</sup>) area similar to that conducted in Kruger National Park by Pedersen (2009) with a density of  $0.03$  rhinos/km<sup>2</sup>. When compared to findings reported in Southern Africa on white rhinoceros ranging patterns (Pienaar *et al.*, 1993; White *et al.*, 2007; Pedersen, 2009) the home range sizes in Ol Pejeta Conservancy were found to be larger. This is in line with Pienaar *et al.* (1993) and Pedersen (2009) observation that home ranges in areas with low white rhinoceros density are larger than those with high density. This observation could be attributed to less competition for available resources that include water, food, shelter and wallows in low white rhinoceros density areas. Contrary to the findings of Owen-Smith (1972), Pienaar *et al.* (1993), Rachlow *et al.* (1999) and White *et al.* (2007) that male home ranges are smaller than females', this study observed smaller female home ranges than males', except in the case of Females 2 and 3. The later results



**Figure 4.11: Home range sizes and core areas covered by all white rhinoceros during the wet and dry season of post release monitoring phase – January 2007 to December 2008 (Source: Author, 2010)**

are consistent with those of Pedersen (2009) who reported smaller female home ranges than males and could be attributed to the low density of white rhinoceros leading to low competition with other males.

Although Male 1 had been moved from Lewa to Ol Pejeta Conservancy a month earlier, than Males 2, 3 and Female 1 (all ex-Lewa), it is Male 2 that was often seen expressing territory marking (scraping, spray urination, dung kicking) characteristics and eventually establishing a territory that was favourable to the females. This resulted in Males 1 and 3 being confined to the western sector throughout the post release monitoring phase potentially in search of females. However, Male 3 moved to the eastern sector a week after Male 2 died and was seen utilizing former Male 2 range.

While Rachlow (1997) observed that in areas of high white rhinoceros density, not all adult males are dominant territory holders, results from this study indicate that the same applies for areas with low white rhinoceros density. Details of dominant white rhinoceros male social interactions with oestrus females and other subordinate males have been discussed in several studies (Owen-Smith, 1971, 1972, 2002; Rachlow, 1997; Pedersen, 2009) and generally indicate that the occupation of a territory gives the dominant male a reproductive advantage. Such is the case displayed by Males 2 and 3 respectively where Male 2 was the dominant male and Male 3 took over as soon as Male 2 died.

The home range overlap between Female 1 and Male 2 throughout the post-release monitoring phase and the birth of calf 1 twenty months after translocation concurs

with Owen-Smith (1975, 1988) and White *et al.* (2007) suggestion that females may select a male for mating simply by entering his territory when she is on oestrus. White *et al.* (2007) revealed a link between the utilisation of particular territories by white rhinoceros females, and the amount of grassland habitats within it suggesting that females select male territories with better available graze. The home range overlap between Female 1 and Male 2 thus indicate that Male 2 had a territory with better graze, i.e. *Zebra* site with lower grass biomass (398kg/ha). Partial home range overlap of Female 2 with Male 2, in the company of Females 1 and 3, did not yield successful mating as Female 2 could have been pregnant when she was moved to the Conservancy. This was confirmed by the fact that she gave birth to calf 2 fifteen months after being moved to OPC.

The results from this study show that dry season home size range were larger than those for wet season, possibly due to the decreased availability and quality of forage and water in the dry season as observed by Owen-Smith (1988). Although this seasonal change was not significant when the home range size for all rhinoceros was considered, it was significantly large for Female-Male. Consequently the seasonal variation was as hypothesized, with dry season home range size being larger than those of the wet season. This observation could be attributed to the reduced forage availability and quality, reduced water availability as well as shelter and wallows forcing the white rhinoceros to move widely in search of these resources. The seasonal results obtained in this study, however, differ with Pienaar *et al.* (1993) who observed that home range size increased in the wet season instead of the dry season. This observation was thought to be due to the wider range of available field water in the wet season, suggesting that the

movement of the rhinoceros was not constrained by the few permanent water supplies.

In summary, the high utilization of *Zebra* site when all seasons data were pooled is supported by the findings in Section 4.1 and 4.2 which show that this site had better forage quality (graze) and by observation in Section 4.3 which characterizes this site as having numerous water sources that include a permanent water trough and two streams (water). Further findings in Section 4.4 show that the home ranges of the six rhinoceros in wet, dry and when all seasons data were pooled, had their core areas within *Zebra* site. This observation further supports the suggestion that the site provided the best habitat conditions for the southern white rhinoceros that included high quality forage, adequate water sources, and shade for resting.

In terms of site preference, *Zebra* site was the most preferred followed by Morani then Oryx while in terms of home range, it was followed by Oryx then Morani except in the dry season preference where only Serat and Grants sites were utilized. *Zebra* site also recorded the highest mean percentage faecal nitrogen (2.664%) followed by Morani (2.247%) and then Oryx (2.021%) sites while it had the second least grass biomass estimate (398 kg/ha) followed by Morani (461 kg/ha) with Oryx site recording the least (266 kg/ha). Assuming that the translocated rhinoceros were “ideal” in their judgement of habitat quality and “free” to move from one site to another in their quest for the best as there were no physical barriers separating the site habitats, and knowing that percentage faecal nitrogen and grass biomass estimates are both measures of site quality, it follows that sites with high

quality (low grass biomass and high percentage faecal nitrogen) would be more used.

This observation was noted for *Zebra* site which recorded higher forage quality and thus the most preferred site. Therefore the post release distribution of the southern white rhinoceros in Ol Pejeta Conservancy followed the principles of the Ideal Free Distribution theory (Fretwell and Lucas, 1970; Fretwell, 1972). Further, the observation in Osborn (2004) that animals occupy core areas with greater frequency indicates that the core areas provide the animal with the best dietary requirements. This concurs with the findings of Section 4.4 of the current study where the *Zebra* site formed the core area for most of the southern white rhinoceros moved to Ol Pejeta Conservancy.

The case of the *Grants* site serves to show that a site that is utilized most is not necessarily the most preferred as the utilization could be for proximate reasons such as testing for suitability, and not for ultimate factors such as reproductive success and survival that would make it most preferred since according to Krausman (1999) habitat preference is the consequence of habitat selection, resulting in the disproportional use of some resources over other.

According to Steinheim *et al.* (2005), statistical avoidance of a habitat type does not infer unsuitability thereof as supported by the findings of Section 4.3 of the current study, where *Lodru* site was statistically avoided when all seasons data were pooled and preferred in the wet season. Likewise, the fact that in dry season the rhinoceros preferred study sites with numerous permanent water sources and in

wet season they preferred study sites with numerous water sources in close proximity, not only emphasizes the demand for this resource in a semi-arid area, such as, Ol Pejeta Conservancy, but also the fact that same sites can be preferred in one season and avoided in another season. This is in line with Bailey *et al.* (1966) and Redfern *et al.* (2003) observation that not all habitats are occupied as animals will choose a habitat based on its suitability. Consequently, findings of this study indicate that distribution patterns of the southern white rhinoceros are influenced directly by a combination of biotic and abiotic factors.



## CHAPTER FIVE

### CONCLUSIONS AND RECOMMENDATIONS

#### 5.1 CONCLUSIONS

The most significant finding of this study was that the post release distribution of the southern white rhinoceros translocated to Ol Pejeta Conservancy followed the principles of the Ideal Free Distribution theory and that these rhinoceros were translocated to an area with better habitat quality (signified by low grass biomass) than where they were removed from (Lewa Wildlife Conservancy). Other key conclusions drawn from this study were as follows:

1. Rapid grass biomass assessment can now be achieved in Ol Pejeta Conservancy. This is as a result of successful calibration of the disc pasture meter, during this study. Further, this is of significant value to the pasture management in the Conservancy that includes; controlled burning and intensive grazing by livestock, as this relies on grass biomass estimates.
2. Normalised difference vegetation index values obtained from satellite imagery could provide a surrogate measure of habitat quality for common zebra with respect to faecal nitrogen and the same can be extended to the southern white rhinoceros considering that both are non-ruminants. This is based on this study's finding that showed a strong relationship between the NDVI values and percentage faecal nitrogen.
3. In addition to grass biomass, water resource in a semi-arid area of relatively low rainfall, such as Ol Pejeta Conservancy is critical for southern white rhinoceros.
4. During the settling phase, home ranges for male rhinoceros were larger than those for females while there was a seasonal switch during the intensive

monitoring phase with the male home ranges being larger in the dry season and the female home ranges larger in the wet season.

5. The rhinoceros were exposed to a better forage quality i.e. lower grass biomass compared to the high grass biomass in Lewa Conservancy where they had been.

## **5.2 RECOMMENDATIONS FOR WHITE RHINOCEROS MANAGEMENT AND FUTURE RESEARCH**

The following recommendations are drawn from the conclusions of this study:

1. There is need to continue estimating grass biomass in Ol Pejeta Conservancy in order to capture long term seasonality and compare its relationship with NDVI values.
2. The significantly positive preference by southern white rhinoceros for the *Zebra*, *Oryx*, *Morani* and *Serat* sites, during different seasons, shows the importance of these sites to the conservation and management of this rhinoceros species in Ol Pejeta Conservancy thus management of grass biomass in these sites is necessary to ensure optimum grazing lawns.
3. The role of water in distribution of white rhinoceros in Ol Pejeta Conservancy should be studied if further translocation of white rhinoceros is to be considered for Ol Pejeta Conservancy.
4. Monitoring of these rhinoceros should be continued, especially, to follow up on how Male 3 finally settled in the territory of Male 2 and if it had resistance from Male 1 and also the movements of Females 1 and 2 as their calves approach the weaning stages and the mating successes of Female 3 given that Female 1 and 2 have calves.

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## APPENDICES

**Appendix I: Distribution and population estimate of southern white rhinoceros in Kenya as at the end of December 2010 (updated from Okita-Ouma *et al.*, 2007)**

Sanctuary	Population estimate	Area km <sup>2</sup>	Density	Census precision
Solio	128	72		Census of November 2010 (minimum)
Nakuru National Park	60	144		Based on daily monitoring data
Lewa	40	247		Known population
Meru National Park	38	48		Known population
Oserian	10	397		Known population
Ol Pejeta Conservancy	11	300		Known population
Ol Jogi (Pyramid)	8	50		Known population
Ol Choro oirwa	3	-		Known population
Delta Crescent	2	-		Known population
Mugie Ranch	2	93		Known population
Nairobi National Park	10	117		Known population
Nairobi Safari Walk	1	0.5		Known population
Ill Ngwesi	2	-		Known population
<b>Total</b>	<b>305</b>			

**Appendix II: Comparison of linear regression models for some savanna locations in South Africa and Kenya**

Treatment for independent variable	a	b	r <sup>2</sup>	Location	References
$\sqrt{x}$	-3019	2260	0.895	Kruger National Park	Trollope & Potgieter (1986)
x	567.2	279.9	0.710	Sourish Mixed Bushveld	Theron (1991)
x	998.7	313.7	0.576	Zululand Coastal Plain	Brockett (1996)
$\sqrt{x}$	-3340	2323	0.923	Lewa Conservancy	Botha (1999)
x	681.9	300.4	0.647	Nylsvley Nature Reserve	Dorgeloh (2002)
$\sqrt{x}$	-3340	2323	0.923	ASAL Northern Kenya	Chege (2004)
1/x	999	-2246.5	0.601	Ol Pejeta Conservancy	This study

Key: x = disc height (cm); a = intercept/constant; b = slope; r<sup>2</sup> = coefficient of determination