INFLUENCE OF HABITAT AND ANTHROPOGENIC DISTURBANCE ON THE DIVERSITY, DISTRIBUTION AND ABUNDANCE OF LARGE MAMMALS: A CASE STUDY ACROSS FOUR ADJACENT WILDLIFE CONSERVATION AREAS IN LAIKIPIA COUNTY, KENYA

by

Ryan G. Valdez
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Committee:

___________________________________	Dr. Larry L. Rockwood, Dissertation Director
___________________________________	Dr. Rebecca E. Forkner, Committee Member
___________________________________	Dr. Lee M. Talbot, Committee Member
___________________________________	Dr. David Luther, Committee Member
___________________________________	Dr. Albert Torzilli Graduate Program Director
___________________________________	Dr. Robert Jonas, Department Chairperson
___________________________________	Dr. Donna M. Fox, Associate Dean, Office of Student Affairs & Special Programs, College of Science
___________________________________	Dr. Peggy Agouris, Dean, College of Science
Date:______________________________	Fall Semester 2015 George Mason University Fairfax, VA

Influence of Habitat and Anthropogenic Disturbance on the Diversity, Distribution and Abundance of Large Mammals: A Case Study Across Four Adjacent Wildlife Conservation Areas In Laikipia County, Kenya

A dissertation proposal submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy at George Mason University.

by

Ryan Glenn Valdez
Master of Forest Science
Yale University, 1996
Bachelor of Science.
Texas A&M University, 1993

Dissertation Director: Larry L. Rockwood
Department of Environmental Science and Public Policy

Fall Semester 2015
George Mason University
Fairfax, Virginia

DEDICATION

This dissertation is dedicated to my father, Luciano Enrique "Capt. Johnny" Valdez, who instilled in me his spirit of adventure and a desire to keep learning.

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I can’t imagine anyone surviving the Ph.D. process without the support of friends and family. Life happens to students, too, and to have a supportive foundation in your personal life is important during any academic endeavor. I’ve always had the deep support of my sister, Joanie Valdez, who accompanied me on my very last journey with my GMU students to Kenya. I would like to recognize and thank my partner, Rob Bannister, for his patience, understanding, and unwavering support. Rob has shared the entire Ph.D. journey with me, and any Ph.D. student can tell you it takes real commitment from others to help you get through the process (and for them to get through the process as well). It’s hard to say if I would have recovered from the difficult stretches and unexpected changes during this journey without Rob at my side. All the years of hard work on this Ph.D. were ultimately brought together by the experience of sharing my field site (the Ol Pejeta Conservancy) and a few other Kenyan treasures with my sister, Rob and his parents.

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LIST OF ABBREVIATIONS

African Wildlife Foundation .... AWF
Agriculture and Development Corporation .... ADC [Mutara Ranch]
Akaike's Information Criterion .... AIC
American Association of Geographers .... AAG
Arid and Semi-arid Lands .... ASAL
Association For Rural Advancement .... AFRA
Central Laikipia Collaboration CLC
Centre for Development and Environment .... CDE
Chief Executive Officer .... CEO
Community-based Ecotourism .... CBET
Community-based Organization .... CBO
Department of Range Surveys and Remote Sensing .... DRSRS
Department of Tourism .... DoT
District Development Officer .... DDO
East Africa .... EA
East African Community .... EAC
East African Tourism and Wildlife Conservation Agency .... EATWC
Economic Commission of Africa .... ECA
Eland Downs Ranch .... ED
Environment Management and Coordination Act .... EMCA
Environmental Systems Research Institution .... ESRI
European Space Agency .... ESA
Geographic Information System .... GIS
George Mason University .... GMU
Geographic Positioning System .... GPS
Generalized Linear Model .... GLM
Gross Domestic Product .... GDP
Information Technology .... IT
Intermediate Disturbance Hypothesis .... IDH
International Federation of Tour Operators .... IFTO
International Livestock Research Institute .... ILRI
International Union for the Conservation of Nature and Natural Resources
(The World Conservation Union) .... IUCN
Jomo Kenyatta International Airport .... JKIA
Kenya Association of Tour Operators .... KATO
Kenya National Bureau of Statistics .... KNBS

Kenya National Council for Law Reporting .... KNCLR
Kenyan Shilling .... KES
Kenya Tourism Board .... KTB
Kenya Tourism Federation .... KTF
Kenya Wildlife Service .... KWS
Laikipia Wildlife Forum .... LWF
Lewa Wildlife Conservancy .... LWC
Light Emitting Diode .... LED
Malpai Borderlands Group .... MBG
Ministry of Environment of Natural Resources .... MENR
Mpala Research Centre .... MRC
National Integrated Monitoring and Evaluation System .... NIMES
National Park .... NP
National Science Foundation .... NSF
Nations World Tourism Organization .... NWTO
Non-Governmental Organization .... NGO
Normalized Difference Vegetation Index .... NDVI
Northern Rangelands Trust .... NRT
Ol Pejeta Conservancy .... OPC
Regional Centre for Mapping of Resources for Development .... RCMRD
Remote Sensing .... RS
Rift Valley Adventures .... RVA
Smithsonian Conservation Biology Institute .... SCBI
Segera Ranch .... SEG
Satellite Pour l'Observation de la Terre .... SPOT
Save The Rhino .... STR
Smithsonian Institution .... SI
Statistical Package for the Social Sciences .... SPSS
Sub-Saharan Africa .... SSA
The International Ecotourism Society .... TIES
The Nature Conservancy .... TNC
Tourism Fund .... TF
Tourism Regulatory authority .... TRA
Tourism Secretary .... TS
United Kingdom .... UK
United Nations Economic Forum for Africa .... UNEFA
United Nations World Tourism Organization .... UNWTO
U.S. Agency for International Development .... USAID
Value Added Tax .... VAT
World Commission on Environment and Development .... WCED
World Economic Forum .... WEF
World Travel and Tourism Council .... WTTC
World Wildlife Fund for Nature; World Wildlife Fund .... WWF

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ABSTRACT

INFLUENCE OF HABITAT AND ANTHROPOGENIC DISTURBANCE ON THE DIVERSITY, DISTRIBUTION AND ABUNDANCE OF LARGE MAMMALS: A CASE STUDY ACROSS FOUR ADJACENT WILDLIFE CONSERVATION AREAS IN LAIKIPIA COUNTY, KENYA

Ryan G. Valdez, Ph.D.

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Dissertation Director: Dr. Larry L. Rockwood

The large mammals of East Africa are a functionally diverse component of tropical savanna ecosystems and vital drivers of tourism and, thus, landscape conservation. Laikipia County, Kenya, known for harboring high African mammal diversity and a high density of a flagship species, the eastern black rhino (*Diceros bicornis michaeli*), is a complex mosaic of public and privately owned lands. In this study, I take advantage of a unique opportunity to measure anthropogenic impacts on wildlife among four contiguous, independently managed Laikipia ranches, including one proposed as a future national park. Georeferenced habitat identification was combined with vegetation surveys on all ranches with the majority of the landscape classified as Acacia (39%), followed by grassland (25%), mixed Acacia-Euclea forest (21%), Euclea (12%), and riverine (3%). Sampling of wildlife was conducted through simultaneous camera trapping, resulting in over 150,000 image captures of 49 species. Occupancy modeling was used to test

hypotheses of anthropogenic impacts (fencing, roads, areas of human activity, and artificial waterpoints) on the diversity and distribution of large mammals. Functional guilds were correlated to habitat classification, with grazers and browsers dominating the landscape. Species richness was correlated to size of ranch, but showed no correlations to habitat type or to proximity to artificial waterpoints as was hypothesized. Modeling showed a positive correlation of richness to natural rivers, fencing, and areas of human activity. These unique data across multiple privately owned properties provide an updated and holistic perspective of landscape dynamics in Laikipia. To facilitate data visualization and to promote new technological resources for land managers, a GIS was used to combine habitat classification, anthropogenic structures, and camera trapping records into an accessible, on-line interactive mapping application.

CHAPTER 1. Landscape-level management of large African mammals: A case study of four wildlife conservation areas in Kenya's high country.

1.1 The Importance of Laikipia County, Kenya for wildlife conservation

Within East Africa, the region now known as Laikipia County, Kenya has a recorded history dating to 19^{th} century British colonization (Georgiadis, 2011; Barnes, 2012) of attracting both local and foreign people to its natural beauty and abundant wildlife. Better known as the Central Highlands, the region was once replete with intact native vegetation and herds of migratory mammal species (Cole, 1986; Denney, 1972). Initial growth of rural settlements, agriculture, and livestock production throughout most of the region were soon followed by a network of roads and fencing that had a significant impact on the region's biodiversity (Taiti, 1992). Such transformations altered the natural vegetation of these vast rangelands with the exception, however, of Laikipia, which managed to maintain much of its native flora and fauna. Today, Laikipia County (Figure 1.1) is one of East Africa's primary examples of successful wildlife conservation within a human-dominated landscape (LWF, 2011).

It is the consensus of the greater conservation community in Kenya that

Laikipia's success is due to five key factors. First, Laikipia has a unique and powerful assemblage of dedicated conservation organizations spread throughout the county. The more prominent organizations include the African Wildlife Foundation (AWF),

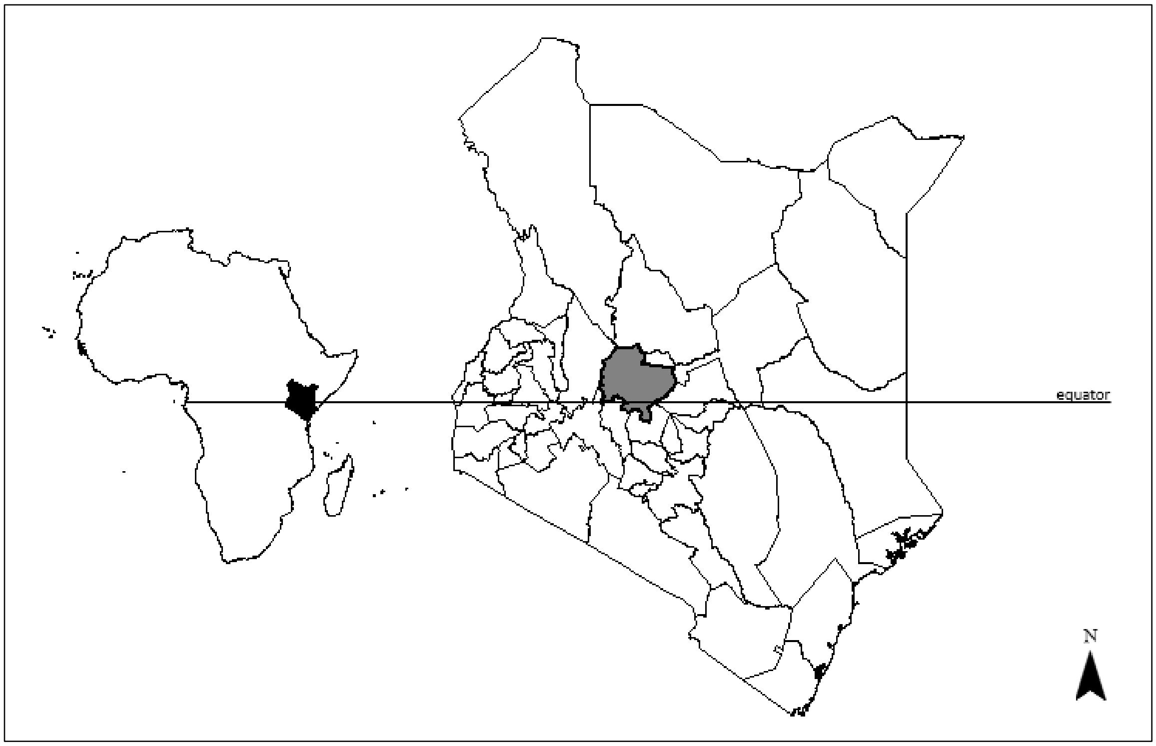


Figure 1.1. Location of Kenya, East Africa (black, left inset) and Laikipia County Kenya (grey, right inset).

the Northern Rangelands Trust (NRT), the Lewa Wildlife Conservancy (LEWA), the Ol Pejeta Conservancy (OPC), the Laikipia Wildlife Forum (LWF), the Mpala Research Centre (MRC), and a relatively recent land acquisition from the Kenya Wildlife Service (KWS). These groups utilize the tools and resources of conservation science to effectively work with local people and numerous small NGOs in addressing the needs for improved human health, welfare, and education (LWF, 2013). The diverse sets of stakeholders invested in conservation are distributed throughout the county in such a way that they maintain a well-balanced representation of the landscape and its tribal communities.

Second, the willingness and cooperation of the people of Laikipia with strategic efforts from private landowners and conservation leaders have made conservation success possible (LWF, 2009). Laikipia harbors large populations of large and charismatic mammals, maintaining the country's second largest population of elephant (*Loxodonta africana*), a significant portion of Kenya's Grevy's zebra *(Equus grevii*) and reticulated giraffe (*Giraffa reticulata*) populations, as well as ecologically important predator species such as cheetah (*Acynonyx jubatus*), wild dog (*Lycaon pictus*), leopard (*Panthera pardus),* hyena (*Crocuta crocuta*), and lion (*Panthera leo*). Its large mammal populations are second only to the well-known Maasai Mara National Reserve, a long-standing protected area bordering the greater Serengeti National Park of Tanzania. The maintenance of such populations is a considerable accomplishment given that Laikipia is a predominantly non-protected landscape where conservation could not have been accomplished without public cooperation.

Third, Laikipia County and all landscapes pertaining to this study are situated within a largely intact and hyperdiverse landscape, the greater Ewaso ecosystem (Figure 1.2) (Georgiadis, 2011; Lane, 2011). Ewaso is known for high biodiversity among invertebrate species (>1000), vascular plants (>700), avifauna (540), and the more well-known species of mammals (95) (LWF, 2013). Many of these are listed among IUCNs globally threatened and endangered species list (IUCN, 2015), adding to the importance of conservation in this region. Of the estimated 800+ black rhino throughout Kenya, over half can be found in Laikipia County, with the largest single concentration at the Ol Pejeta Conservancy (>100 individuals) (OPC, 2013).

Fourth, the wildlife and natural landscapes of Laikipia attract significant tourism, raising Laikipia's profile among business investors. Wildlife tourism can supplement income from livestock production within conservancy ranches (Sundaresan and Riginos, 2010) and throughout the region has been considered a possible avenue of poverty reduction (Manyara and Jones, 2007). Tourism, however, can lead to overdevelopment and a decline in the quality of wild lands, leading some researchers to advise caution in developing these markets so that revenue accrues to

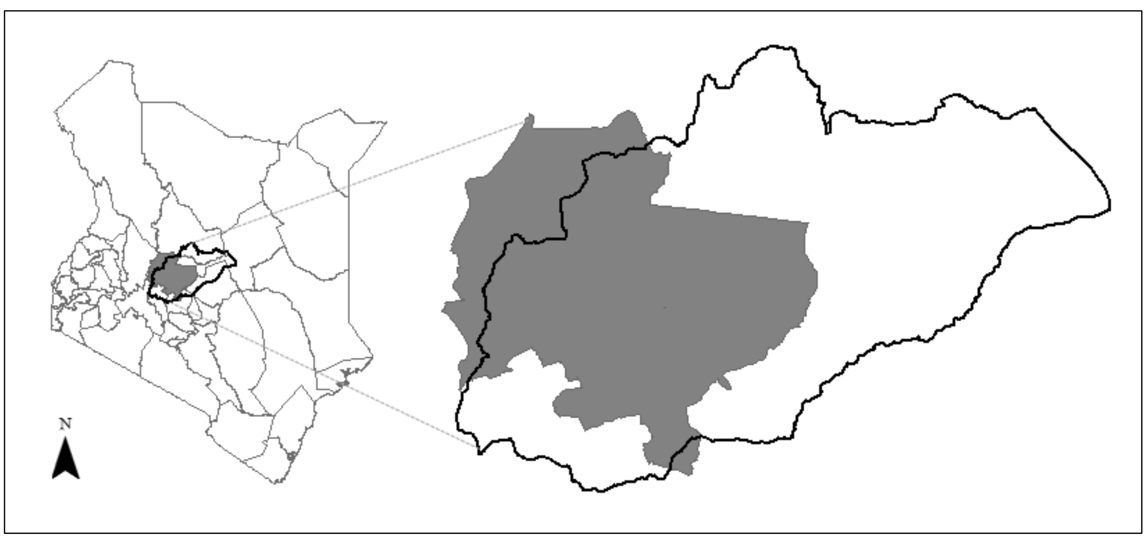


Figure 1.2. The overlay of Laikipia County, Kenya (solid gray) and the Ewaso ecosystem basin (bold outline) of Kenya.

pastoralists rather than foreign companies (Dixit *et al.*, 2013; Homewood *et al.*, 2012).

Finally, the high numbers of conservation organizations within Laikipia have produced a growing body of knowledge and research on the intersection of human, livestock, and wildlife coexistence (LWF, 2013). Importantly, MRC is and has been a vital contributor of peer-reviewed conservation science for the whole of East Africa, representing tropical ecology research in Acacia-savanna ecosystems. Laikipia stands out from other African locations in its capacity to inform wildlife conservation, as organizations and researchers in the region continue to generate funding and support for basic ecological research.

1.2 Research objectives and hypotheses

The overarching goal of this research was to use camera trapping, in combination with surveys of habitat characteristics, to identify causal factors in the diversity, distribution, and abundance of large African mammals. My objectives were to examine differences in diversity and distribution of large mammals in four ranches differing in levels of privatization, use, and protection to improve current management and to make recommendations for the improvement of non-protected areas in Laikipia County (Figure 1.3). I sought to determine the underlying factors that contribute to differences in diversity and distribution of wildlife across management regimes as well as across natural habitat types, and more specifically to address park boundary issues that involve corridors, rhino conservation, and the methodology of camera trapping to improve protection.

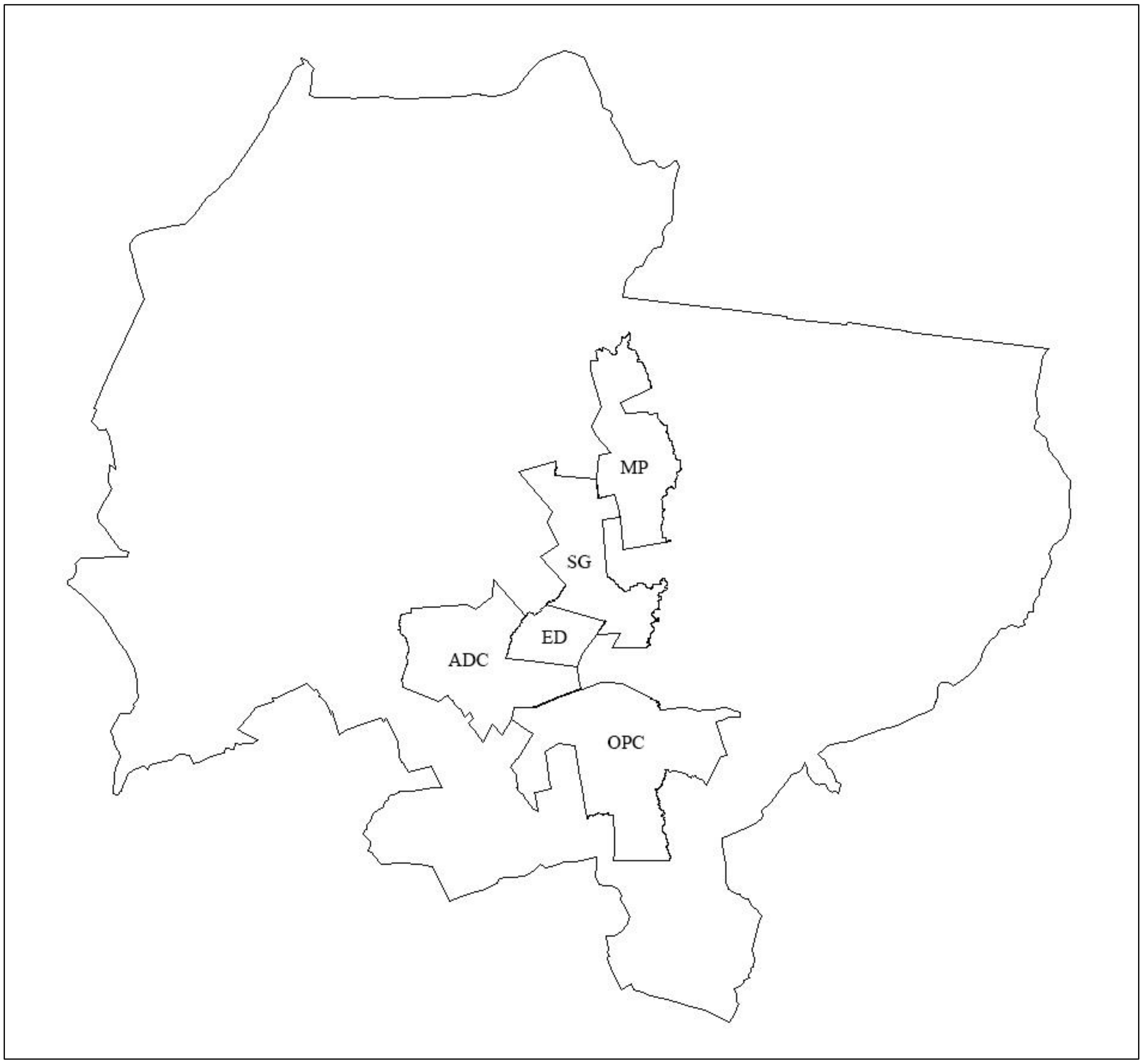


Figure 1.3. The five adjacent ranches that make up the Central Laikipia Collaboration: From North to South: MP (Mpala), SG (Segera), ED (Eland Downs), ADC (ADC Mutara), and OPC (Ol Pejeta Conservancy) within the Laikipia County, Kenya (outer boundary).

Management of wildlife on isolated, private reserves lacks the biological and ecosystem functioning of more contiguous and connected landscapes. Sharing resources across property boundaries, such as access to habitat and natural rivers and water holes for cattle, could advance tourism sustainability and improve quality of habitat for cattle and human livelihood, but might also facilitate overuse if methods of access are not informed by natural patterns of diversity. Though the four adjacent ranches examined in this study share common landscape features, such as Acacia-grassland habitat and vertisol soils, they differ with regard to degrees of cattle management, conservation activity, human activity, and infrastructure. Segera and Ol Pejeta Ranches, for example, are private cattle ranches. ADC Mutara is a government owned ranch that contains a dedicated 20,000 acre conservation area, and Eland Downs is an additional government owned property transitioning from a private cattle ranch to a future national park. In areas where conservancy boundaries abut one another, an opportunity presents itself to form larger cooperatively managed landscapes. This study examines the role for continued joint management planning in reducing anthropogenic impacts of land-use on wildlife, allowing for coexistence of wildlife with livestock, and reducing conflict between humans and predators.

This research addresses a common conservation question: How does human activity affect biodiversity across a landscape? Although this is a broad question, a combination of factors make this study uniquely helpful: the geography and proximity of the focal ranches, the diversity of flora and fauna, the variety of land-use management strategies, an impressive collection of regional stakeholders, and the incorporation of information concerning the coexistence of wildlife with livestock. In addition to the complexity of factors addressed, this research incorporates geographic information systems (GIS) overlaid with field research in order to assess habitat preferences of wildlife and the impacts of anthropogenic factors. Data collected across the landscape include vegetation and camera-trapping surveys, GIS habitat analyses and classification, infrastructure measurement and observations, and an assembly of supplementary wildlife data. Consultants and collaborators provided supplemental data such as precipitation estimates, species identification, land-use history, GIS and GPS data, and management planning strategies. Results of this research are to be made accessible to County stakeholders and contribute to a growing movement of increased wildlife conservation on cattle ranches. In addition to improving general management guidelines for the region, lessons learned from individual ranches and from the greater ecology of the landscape can be used to support plans to expand the black rhino population in Laikipia County.

Camera-trapping surveys were an indispensable part of this investigation. Infrared remote-trip camera manufacturing is a fast growing business and its application highly popular in wildlife management and conservation science. This non-intrusive method for monitoring wildlife provides opportunities to record important data such as species identity, animal condition, behavior, temporal activity, as well as temperature, lunar stages, and audio and video data. Development of numerous software programs such as DISTANCE™, CAPTURE™, MARK™ and PRESENCE™ now allow statistical analyses of camera trapping data. Camera trapping is widespread among research scientists, land managers, and tourism operators in Laikipia County. Managers use camera trapping to combat poaching activity and to monitor and improve general security of protected areas. In this study, I determined that camera trapping was useful for determining species diversity, habitat use, and species responses to environmental disturbances or human modifications (e.g., fences, corridors, water holes, cattle management) (Figure 1.4). Over 80 camera trapping units were randomly assigned using a 50m grid overlay to classified habitat layers per ranch and were additionally positioned on wildlife corridors on the border between the Ol Pejeta Conservancy and ADC Mutara Ranch. Data collected allowed me to answer the following questions:

Figure 1.4. A collection of images from camera trapping deployments during this study: Top row from left to right: lion (*Panthera leo*), reticulated giraffe (*Giraffa camelopardalis*), cheetah (*Acynonyx jubatus*); Second Row: Grant's gazelle (*Nanger granti*), black rhinoceros (*Diceros bicornis*), olive baboon (*Papio anubis*); Third row: black backed jackal (*Canis mesomelas*), leopard (*Panthera pardus*), and hippopotamus (*Hippopotamus amphibius*); last row: humans (local community members).

Question 1: What is the diversity, distribution, and abundance of large mammals across four adjacent and contiguous ranches, and do trophic guilds demonstrate habitat preference?

As a primary goal in understanding the complex nature of wildlife diversity and distribution across a highly managed landscape, it is important to have a proper measure and understanding of the use of habitat represented within the assigned classification. In this study, I estimated species richness per ranch and per habitat and described the distribution of large mammals across the landscape. Habitat preference was determined by guild occupancy estimations. An analysis combining field vegetative surveys through a GIS followed by camera trapping was a necessary first step to determine distribution of wildlife.

To confirm camera trapping was a successful method, I first tested how detection was influenced by effort (amount of time cameras were in use) and camera deployment (number of cameras in use). Relative abundances of species is predicted to be higher on larger ranches, with Eland Downs Ranch as a bottleneck representing the lowest species richness compared to its northern and southern neighboring ranches. Lastly, I address the hypothesis that guilds, particularly grazers and browsers would occupy specific habitats (Table 1.1).

Table. 1.1. List of hypotheses, methods, criteria, and justification testing wildlife habitat preference and occupancy.

	Hypothesis	Method	Criteria	Justification
I	Ranch size and overall condition influence species diversity.	Accumulation curves using EstimateS. PRESENCE using AIC	Larger ranches are likely to have more habitat available to support increased diversity.	Conservancy size and proximity to other conservancies are important criteria for future land acquisition.
II	Trophic guilds exhibit habitat preference.	PRESENCE using AIC, Chi-square analysis	Grazers, browsers, and carnivores will show preference to grasslands, Acacia habitat, and riverine habitat respectively.	Maintaining a mosaic of habitat, or specific habitat is important to observing large mammals.

Question 2. What are the impacts of anthropogenic factors on the diversity, distribution and abundance of large mammals in Laikipia County, Kenya?

The arrival of Maasai pastoralists and later, European settlers, has significantly impacted the landscape of Laikipia County, affecting large scale land cover and wildlife resources (Muchiru *et al.*, 2008, 2009; Kay-Zwiebel and King, 2014). European farmers built dams, boreholes, roads, fences, and various other structures to manage for agriculture and cattle ranching. Today, conservancies have adapted some of this infrastructure to manage for native wildlife. For example, boreholes pumping water to the surface for cattle also provide water for zebra (*Equus quagga*), elephant

(*Loxodonta africana*), and various antelope. During periods of low water availability and drought, these resources sustain populations of local wildlife and reduce human-wildlife conflict when large mammals would normally seek water outside of the conservancy boundaries. General observations in this study indicate that large mammals, in particular elephant, seek out the more available natural and man-made water resources on the Ol Pejeta Conservancy, but travel northward into ADC Mutara and through Eland Downs for greater access and potentially lower competition of browse availability of Acacia forest. I propose that aside from the natural mosaic of habitat throughout the landscape, anthropogenic factors influence the distribution, diversity, and abundance of wildlife in Laikipia County. The combination of public and private land among the four ranches in this study offers a unique opportunity to examine the influence of man-made features at a landscape scale. I investigate associations of wildlife with human occupancy, water availability (boreholes), livestock management, fencing, and habitat (Table 1.2). I hypothesize that water resources outweigh all other factors in their effects on the occupancy and diversity of wildlife, particularly among large mammals. Furthermore, I hypothesize that permanency and intactness of fencing, roads, and human settlements will impact the abundance and distribution of wildlife.

Table. 1.2. List of hypotheses, methods, criteria, and justification testing anthropogenic impacts on species diversity.

	Hypothesis	Method	Criteria	Justification
I	Access and availability of water influences species diversity.	General Linear Model using AIC	Species richness increases with proximity to both water holes and rivers	Creating access and availability to water may be an important investment to future conservancies.
II	Roads and human activity areas impact species richness.	General Linear Model using AIC	Species richness decreases with increased proximity to roads and human activity areas.	Careful planning of conservancy infrastructure can improve species diversity.
II	The use of fencing on conservancies impacts species diversity.	General Linear Model using AIC	Fenced conservancies prevent dispersal of wildlife, exclude illegal cattle grazing, thus will be correlated to higher species richness.	Economic incentives for conservancies are to achieve high species diversity, particularly of large mammals that will support increase ecotourism.

Question 3: How do remote technologies improve public private wildlife conservation effort in Laikipia County, Kenya?

The participation of private lands into wildlife conservation has become a focus for landscape level environmental action. The shared responsibility through stakeholder engagements that involved landowners, NGOs, various state and county agencies, along with economic and regulatory aspects, can make public private ventures challenging and complex. There has been increased interest in utilizing agricultural lands in such agreements, particularly in East Africa where many protected areas share boundaries with agricultural industry. More often, large-scale agricultural landowners are often familiar with aspects of ecosystem management and wildlife sciences, making the process to understand the subject matter quite effective. Conservation leaders quickly identify public-private opportunities strategically for wildlife corridor investment and connecting large landscapes for broader ecosystem-level goals. There are also many social and economic benefits at the local and city level when looking at markets and possible constituencies.

The Kenyan Agricultural Development Corporation (ADC), the Ol Pejeta Conservancy (OPC), and the African Wildlife Foundation (AWF) signed a Memorandum of Understanding (MOU) in 2012 to facilitate the inclusion of 20,000 acres of the ADC Mutara Ranch as a vital component and corridor for regional wildlife conservation. The unique qualities of this parcel of land will boost the local economy from infrastructure and ecotourism while providing a much needed wildlife migration pathway. Under the agreement, AWF provided the initial support for infrastructure, OPC provided equipment and personnel, as well as valuable guidance on ecotourism operations, and guards from ADC Mutara ranch were hired to monitor and protect the conservation area from illegal cattle grazing. Development of ADC Mutara's ecotourism facilities brought new jobs and opportunities at the local level.

The conservation MOU fills a gap that connects OPC to Eland Downs Ranch, slated for national park status in the near future. This connection is vital not only for local wildlife movement and ecological connectivity, but specifically to assist with the expansion of the black rhino population of Laikipia. In the past 40 years, African black rhino populations have drastically decreased from an estimated 65,000 in the 1970s, to present day estimates of 5,500 individuals (KWS, 2007). Poaching and loss of habitat are the primary contributors to this decline (KWS, 2007; Knight, 2011). The population of the Eastern subspecies (*Diceros bicornis michaeli*) is fewer than 1000 individuals, of which over 800 are located throughout Kenya, and a current estimate of 100 on the Ol Pejeta Conservancy (OPC) in Laikipia County.

At present, the Ol Pejeta Conservancy has a carrying capacity of 120 black rhino (OPC, 2015; Mulama, personal communication). This ranch also has a small population of white rhino (*Ceratotherium simon*) that do not compete with the browse availability of black rhino, as they are non-native grazers. Carrying capacity for black rhino is partially determined by availability of its primary food source, the whistling-thorn Acacia (*Acacia drepanolobium*). As the density of black rhino approach carrying capacity, there is an associated decrease in breeding success (OPC, 2010) that will encourage management of OPC to expand the black rhino population north into the 20,000 acre conservation area of the ADC Mutara Ranch.

In this chapter, I test the assumption that increased effort and camera deployment result in increased probability of detection. I then determine whether camera trapping is an effective method of gathering wildlife data needed to answer questions regarding diversity, abundance, and distribution of large mammals in

Laikipia County, Kenya. The combination of remote-trip camera data and on-line mapping technologies will supply ranch managers with a fast, reliable, and more accurate assessment of landscape level resources. The integration of camera trapped wildlife imagery embedded into Esri (Esri, 2015) Story Map™ mapping templates offer on-line, interactive mapping that is an effective visualization tool offering land managers the ability to view multiple layers of data, get access to landscape measuring tools, incorporate updated 2015 satellite imagery, and print and share customized mapping products. Importantly, this resource is virtually without cost. This study provides a unique application tool to visualize research results that will benefit the community of stakeholders in Laikipia County, Kenya.

1.3 Theoretical background for wildlife conservation in Laikipia County, Kenya

Savanna ecosystems

The geographic scope of the research includes four connected ranches within Kenya's Ewaso ecosystem basin: Segera Ranch, Eland Downs, ADC Mutara Ranch, and the Ol Pejeta Conservancy (Figure 1.3). These combined ranches represent a large portion of what is broadly known as the Central Laikipia Collaboration (CLC), an informally arranged effort of conservation leaders to promote wise use of the greater landscape for wildlife and ecotourism, for cattle ranching, and for the benefit of local human communities. The CLC is part of a tropical dry savanna ecosystem. Savanna environments consist of tropical, mixed tree-grass communities that cover nearly 40% of global land surface (Walker and Noy-Mier, 1982; Mistry, 2000, Olsen *et al.*, 2001). Savannas and their associated woodlands represent nearly half of the African continent (Menaut, 1983) and account for roughly 30% of the primary production in Africa (Grace *et al.*, 2006). They support pastoral rangelands (Georgiadis, 2011) as well as commercial livestock production (Georgiadis, 2007a), both of which may be intermixed with conservation areas for wild ungulates (Sankaran *et al.*, 2005, Western et. al, 2009). As the human appropriation of net primary productivity increases across the African continent, protection of savanna biodiversity will depend on safeguarding wildlife as well as sustainably managing livestock (Grootenhuis and Prins, 2000).

Because of their high densities and role as primary consumers, large ungulate populations substantially influence ecosystem dynamics in these regions (Good and Caylor, 2011) (Figure 1.5), which are now at risk from either overuse or abandonment (Constanza *et al.*, 1997). With respect to abandonment, in the absence of large ungulates, wild or domesticated, grazed savannas can be overtaken by successional woody plants (Archer, 1995, Roques *et al.*, 2001), leading to loss of rangelands (Tobler *et al.*, 2003). On the other hand, as human and livestock pressures in these tropical grasslands increase, overuse can result in loss and fragmentation of habitat, alteration and quality of natural resources, changes in species composition, and even extinction of species (Wambuguh, 1998).

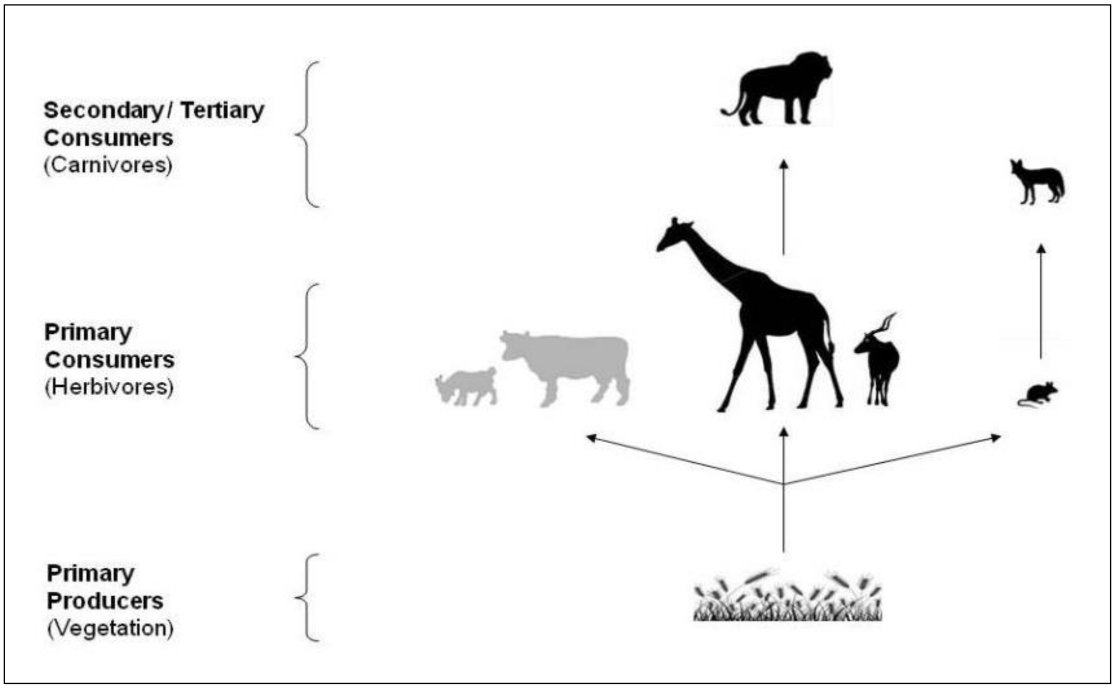


Figure 1.5. Conceptual model of trophic level interactions represented throughout study.

While there is no shortage of literature on interaction between livestock and large mammals, our scientific understanding of savanna food webs is still far from complete (Fox-Dobbs *et al.*, 2010; Miller *et al.*, 2012). Protection of many important grazers and browsers often requires fencing these herbivores into protected reserves. However, the majority of large mammal biodiversity in East Africa, with estimates usually between 70 – 80%, occurs in non-protected areas (Mbugua, 1986; Western, 2009; Ottichilo *et al.*, 2000; Georgiadis, 2007). Such unprotected landscapes overlap with livestock production at various levels. Locally, competition for grass between wild and domestic herbivores is common (Mizutani, 1999, De Leeuw, *et al.*, 2001; Augustine, 2004; Young, *et al.*, 2005; Georgiadis *et al.*, 2007; Georgiadis *et al.*, 2007b; Wambuguh, 2007; Sensenig, 2008; Odadi, *et al.*, 2007). Studies in Kenya addressing wild herbivores demonstrate complex trophic interactions with regard to abundance of small mammals (Keesing, 2000; Keesing and Crawford, 2001), birds (Ogada *et al.*, 2008), savanna trees (Goheen *et al.*, 2007), snakes (McCauley *et al.*, 2006), impala gazelle (*Aepyceros melampus*) (Augustine, 2004), leopard (*Panthera pardus*) (Mizutani and Jewel, 1998), lions (*Panthera leo*) (Ogada *et al.*, 2003; Frank, 2005; Woodroffe and Frank, 2005; Woodroffe *et al.*, 2007; MacLennan *et al.*, 2009), and wild dogs (*Lycaon pictus*) (Woodroffe *et al.*, 2005; Woodroffe, 2011). No clear predictions have emerged, however, with respect to how these species and other small carnivores will interact with livestock management to influence savanna biodiversity. Recent intensification of human activities, including agricultural development (Martens, 2013), livestock ranching (Heath, 2000), energy and other significant development (Odingo, 1971, Herrick *et al.*, 2012), has further partitioned savanna landscapes in ways that alter movement of wildlife (Georgiadis, 2011) and create additional human-wildlife conflict (Frank, *et al.*, 2011). To date, most human-wildlife conflicts have not been resolved in ways that sustain populations of large mammals (Ripple, 2015).

Continued high demand for land in Laikipia County is leading to further cultivation of previously uninhabited areas, while at the same time the county is experiencing an increase in wildlife tourism (Manyara and Jones, 2007; Laikipia Wildlife Forum, 2015). Changes over time in tourism in Laikipia show that after Kenya's independence in 1963, following a decline in tea and coffee agriculture, the government began to strategically invest in tourism (Akama 1999). From a wide variety of sectors, nature-based or ecotourism became dominant (Olindo, 1991), particularly in Laikipia where expanses of livestock rangelands were ideal locations for wildlife conservancies. Currently, Laikipia County harbors one of Kenya's fastest growing economies and human populations. At the same time, there is increasing demand for recreational wildlife safaris. The large urban center of Nanyuki, Laikipia's largest city, has seen substantial development, population growth, and expansion of tourism businesses. There are currently 371 registered safari and ecotourism operators in Kenya, of which 40 – 50 consistently maintain business in Laikipia County (KATO, 2014) in the form of small lodges, ranch houses, tented camps, camp sites, and adventure tours (LWF, 2015). Increased access to airfare, new hotels and safari lodges, and more numerous restaurants contribute to the success of wildlife conservancies. Yet, they also create higher levels of human-wildlife conflict.

With increased competition among wildlife, humans, and livestock for limited resources, many land owners are still experiencing only marginal benefits from cattle ranching (Kirigia *et al.*, 2007). Use of land outlined in the amended 2012 Kenya Constitution strongly supports wildlife conservation, but simultaneously maintains strong encouragement for livestock and agricultural development (Kenya National Council for Law Reporting, 2010). Kenya's constitution also maintains a policy in which all wildlife is deemed publicly owned, and therefore landowners are usually not allowed to directly manage wildlife on their property. Thus, Laikipia is unique among all counties in Kenya in that it sustains large mammal populations in landscapes with little to no national protection. All wildlife in Kenya is under the management authority of the Kenya Wildlife Service, presenting challenges for KWS to accommodate the high volume of human-wildlife conflict issues (Waithaka, 2012).

With a well-defined history of research efforts focusing on ecology and wildlife in East Africa (Talbot, 1965; Sinclair, 1985), we now see a growing body of research examining broad scale impacts of anthropogenic factors on its biodiversity (Petty, 2002; Wambuguh, 2007; Vanthomme *et al.*, 2013) with more work needed at the local and county levels. These smaller scales are where human-wildlife interactions occur and where management has unique opportunities to share resources across public and private property boundaries where the greatest gains can be made in solving human-wildlife conflicts. As conservation within Africa becomes increasingly management-driven and as Laikipia becomes more and more fragmented across multiple public and private land holdings, research must continue to address both the conservation of endangered species in non-protected areas and the impacts of human activity, including management itself, on species diversity within local protected and non-protected areas (Western, 2009). Effective conservation practice will require integrating best management practices at the local scale in combination with an understanding of large landscape ecological processes (Knight, 2011).

Land-use and the effects of disturbance in Laikipia County

Early research on habitat disturbance (e.g, Hutchinson 1953, hurricanes on coral reef systems: Horn, 1975; Huston, 1979; terrestrial forests: Connell, 1978, Sousa, 1984; Collins *et al.*, 1995) demonstrated that disturbance can promote diversity. Species diversity fluctuates over both short and long-term time scales in response to natural disturbances or environmental gradients throughout Kenya (Fjeldsaå and Lovett, 1997; Olff *et al.*, 2002; Tóth and Lyons, 2014), but the effects of anthropogenic disturbances are harder to predict. At lower levels of disturbance, as might be expected under normal grazing pressure from native African ungulates or low levels of livestock grazing (Rogers, 1993; Townsend and Scarsbrook, 1997), species diversity may increase (Rogers, 1993; Mackey and Currie, 2001) as herbivores alter competitive interactions between plants and promote nutrient cycling (Keirs *et al.*, 2010). Native herbivores, particularly large herbivores such as giraffe and elephant, can act as ecological engineers when altering habitat and promoting diversity in vegetation (Goheen *et al.*, 2010), leading eventually to a higher diversity of small mammals (Keesing, 1998; 2000). A study by Baum *et al.* (2007) showed that diversity of carnivores and their small mammal prey were most frequently recorded within an intermediate disturbance of shrub cover. At another extreme, intense livestock alteration of a landscape has severe negative effects on diversity (Mugatha, 2002; Young and Augustine, 2007). Likewise, an absence of herbivory may also change vegetation regimes and decrease landscape heterogeneity (Goheen *et al.*, 2007; Louthan *et al.*, 2013, 2014; Porensky *et al.*, 2013a). Preliminary data from degraded landscape studies suggest that intermediate disturbance patterns promote diversity in pastoral areas (Young, 2005), but that outcomes fluctuate depending on soil and rodent communities (Keesing, 2000; Keesing and Young, 2014). The impacts of disturbance are likely to vary across trophic levels (herbivores, small and mesopredators, and large carnivores) and to consequently alter food web structures (Prugh *et al.*, 2009). Below, I outline prior research on diversity and food web interactions for each of these groups.

Herbivores

Current research on large ungulate ecology demonstrates this guild of species are functionally important to large African landscapes, having direct contributions as ecological engineers (Keesing and Crawford, 2001), in stimulating and improving vegetative growth (Goheen and Palmer, 2010) and maintaining habitat mosaic (Kimuyu *et al.*, 2014; Pringle *et al.*, 2011), and in acting as drivers of ecological cascades (Keesing, 2000; Keesing and Young, 2014). Their indirect interactions with other species of both fauna and flora make them vital to sustaining habitat complexity that promotes high biodiversity (Wilson, 1992; McCauley *et al.*, 2008; Young, *et al.* 2013, 2014). Because indirect effects are often difficult to measure in real time, scientific research conducted in Laikipia in the past 20 years has provided a great deal more insight into the importance of these large mammals as they influence the abundance, distribution, and diversity of associated wildlife. This research also includes important emerging observations made on the interactions of native and domestic ungulates (Augustine, 2004; Augustine *et al.*, 2011; Keesing *et al.*, 2013; Odadi *et al.*, 2011; Porenski *et al.*, 2013;

Veblen and Young, 2010). Due to the timing of necessary research on the natural history and ecology of large herbivores, we are only now observing the start of long-term research on the impacts that anthropogenic factors have on the various roles that large ungulate herbivores have in large landscape conservation (Kartzinel, 2014). Of the many impacts that human activity and structures have on wildlife abundance, distribution, and diversity, it is management of the landscape that is of critical importance (Western and Strum, 1994; Kinnaird and O'Brien, 2012). Large herbivore species in Laikipia, Kenya (Appendix A) carry a significant role in the success of ecotourism (Laikipia Wildlife Forum, 2013; Republic of Kenya, 2013) and serve as effective ambassadors and flagship species in broader wildlife conservation of the region (Georgiadis, 2011; Sundaresan and Riginos, 2010; Tallis *et al.*, 2014). It is clear that wildlife conservancies in Laikipia, benefit from investing in large mammal conservation for both ecological and financial reasons and should recognize that managing a conservancy that includes livestock will increase the level of complexity and interaction. Large ungulate species are not functionally equivalent and differ in feeding strategy (Goheen *et al.*, 2007, 2010; Augustine, 2010), habitat preferences (Augustine, 2004; Fischhoff *et al.*, 2007), temporal activity (Georgiadis *et al.*, 2007a), and behavioral interactions with livestock (Denney, 1972; Augustine *et al.*, 2011; Odadi *et al.*, 2011; Porensky and Young, 2013).

The lesser known herbivores such as smaller mammals also play an important role in competitive interactions with livestock. Rodent abundance can increase in landscapes with suppressed predator populations (Hubbard, 1972). Diverse small mammal populations, on the other hand, may indicate a healthy ecosystem (Avenant, 2000;

Keesing and Crawford, 2001; Magige and Senzota, 2006). Small mammals represent more than one trophic level, as they are a highly omnivorous group of animals. Keesing (2000) and Young *et al.* (2013) describe a complex web of interactions in Kenya's savanna ecosystems (Appendix B), showing that small mammal communities have been previously underestimated as competitors for vegetation resources with both native and non-native ungulates (Keesing and Crawford, 2001; McCauley *et al.*, 2008). Therefore, these small herbivores could help predict ecological transition in disturbed areas.

Predators and mesopredators

In terrestrial environments, particularly in landscapes recovering from degradation, large herbivore populations are greatly affected by both resource availability and predation (Georgiadis, 2011; Romanach, *et al.*, 2011), but which predators are responsible for major changes is still unclear. Simple paradigms that place predator-prey interactions in hypothetical bottom-up versus top-down effects (Polis, 1990, 1999; Hunter and Price, 1992) are often inadequate as mechanisms for predicting changes in diverse, reticulate food webs. In particular, the effects of omnivory and the importance of intermediate, smaller predators (mesopredators) have been underestimated. These include not only predators eating herbivores, but predators eating vegetation and other predators. Such food chain complexity can lead to patterns of intraguild predation and trophic promiscuity (Hunter, 2009) that produce synergistic cascading effects such as mesopredator release (Crooks and Soulé, 1999).

Significant declines in carnivore population density and geographic range (Woodroffe, 2000) have been recorded in Laikipia County (Frank, 2005; Woodroffe *et* *al.*, 2005). Habitat requirements of carnivores place them in direct competition with humans and livestock (Frank, 1998; Treves and Karanth, 2003; Patterson *et al.*, 2004; Woodroffe *et al.*, 2005). The lion is one of the most directly affected species in this regard. Once widespread across Africa, Asia and Europe (Kuten and Anderson, 1980), lions are now limited to small and isolated populations in Africa and Asia (Hazza and Dolrenry, 2011; Patterson *et al.*, 2004; Frank, 2005). Lions and leopards are on the decline in East Africa and maintaining large, contiguous landscapes are vital to securing their survival (Frank *et al.*, 2009; Dolrenry *et al.*, 2014). Managing to preserve declining lion and leopard populations can be successful, as these species rebound quickly given the necessary space and protection (Hunter, 1998; Stuart-Hill and Grossman, 1993).

Apex predators throughout Laikipia (Appendix C) have not been well studied on most properties within this region. Reduced numbers of apex predators on Eland Downs and portions of ADC Mutara, combined with removal of livestock on ADC Mutara, may encourage release of medium-sized predators that compensate for the lack of top-down suppression (Rogers and Caro, 1998; Crooks and Soulé, 1999). Mesopredator release cannot be measured without knowledge of a prey base or primary productivity. The mesopredator is normally not a species of conservation concern, since the cascading effect of the loss of apex predators causes population declines in the prey or vegetation regime, with changes in mesopredators going unnoticed. A study by Rogers and Caro (1998) revealed that declines in song sparrows (*Melospiza melodia*) because of declines in top carnivores were the result of an increase in nest-destroying mesopredators. A similar study by Crooks and Soulé (1999) observed mesopredator release from the absence of coyotes, which caused a trophic cascade release of smaller carnivores, which in turn decimated bird populations.

Mesopredators on the Laikipia landscape include over 15 species of medium-sized carnivores such as black-backed jackals (*Canis mesomelas*), bat-eared fox (*Otocyon megalotis*), servals (*Felis serval*), and various genera of mongoose in the Family Herpestidae (Appendix D). Removal of an apex predator can initiate a trophic cascade in which smaller carnivores compensate for the absence of the superior predator by increasing in numbers and diversity (Soulé, *et al.*, 2003). This ultimately has a negative effect on small birds and mammals by the mesopredator, as shown through research on the interaction of coyotes and raccoon (*Procyon lotor*), upon sparrows and passerine birds (Rogers and Caro, 1998), and through interactions of coyotes and domestic cats (*Felis catus*) (Crooks and Soulé, 1999). Small carnivores are also quite vulnerable to the effects of habitat fragmentation, habitat alteration and the inevitable outcomes resulting from human-dominated landscapes. The small carnivore community can have profound effects on ecosystem function as they quickly respond to decreasing apex predators (Prugh *et al.*, 2009) or a variety of predator-prey interactions (Rockwood, 2015). Diverse carnivore communities also help to enhance food web complexity (Prugh *et al.*, 2009) if species are highly omnivorous and may not only compete for and share prey, but also feed on each other (Finke and Denno, 2004). Carnivores may act as both predators and competitors producing a factor of interference competition known as intraguild predation (Polis, 1990). Literature shows that in some cases, predators return before lower trophic levels respond, as they may be searching larger areas for prey and may be highly mobile

(Fedriani and Fuller, 2000). The probability of detection for camera trapping of mesopredators is far less than that of larger wildlife (MacKenie *et al.*, 2003; Rovero *et al.*, 2014), making the data difficult to address research questions aimed at small carnivore abundance. Small carnivores, though recorded for purposes of species richness in this study, are not a focus of research.

Coexistence of wildlife with livestock

The dynamic results of human alterations on food webs within this landscape are not well studied. However, our understanding of trophic dynamics is increasing, with animal behavior (Fishoff, *et al.*, 2007), population dynamics, and human impacts (Ripple and Beschta 2011) more integrated in research and theory. It is clear that food webs are indeed more complex than most studies reveal, and where anthropogenic factors are involved, we should incorporate cascade effects into assessments of ecological resiliency as habitats recover from degradation (Donihue *et al.*, 2013). Resiliency of natural landscapes in Laikipia will depend not only on ecosystem function Pringle *et al.*, 2010), but on the private land owners working together and recognizing the benefits of a large, intact landscape for both ecotourism and cattle ranching.

The semi-arid regions of East Africa generally have low rainfall, poor soils, and high evaporation rates that limit agricultural productivity, providing a niche for pastoral lifestyles (Wambuguh, 2007). Livestock may act in the same trophic and ecosystem engineering manner as wild herbivores, and the compatibility of native African wildlife with simultaneous livestock production (Figure 1.5) has been studied for more than 15 years (Augustine, 2004; De Leeuw *et al.*, 2000; Gadd, 2005; Georgiadis *et al.*, 2007; Mizuntani and Jewell, 1998; Mizutani, 1999; Odadi *et al.*, 2007; Young *et al.*, 2005; Young and Augustine, 2007;). These ecosystems have historically been used for production of domestic livestock (Augustine, 2003; Cole 1986; Walker and Noy-Mier, 1982), contributing to food production of the greater Sub-Saharan Africa (Jahnke, 1982). Many studies show negative effects of introducing livestock to natural systems (Pringle *et al.*, 2014; Georgiadis, 2007; Lamprey, 1983; Sinclair and Fryxell, 1985), while others claim positive contributions, such as benefits from added soil nutrients and promotion of diversifying habitat and suppressing vegetative encroachment (Augustine, 2003; Bennet, 2003; Young *et al.*, 1995; Bergstrom, 2013).

Figure 1.6. Image of an integrated Boran cattle herd visiting a bore hole among reticulated giraffe and common zebra on the Ol Pejeta Conservancy.

1.4 Landscape, field sites, and research design

Laikipia is one of 47 counties in the former Rift Valley Province of Kenya (Figure 1.1). It lies on the equator between the Aberdare Mountains and Mt. Kenya covering 9,700 km^2 East of the Great Rift Valley at 0° 17'S – 0° 45'N latitude and 36° 10'E – 37° 3'E longitude. This highland plateau of rolling hills sits at an elevation range between 1,700 – 2,000 m above sea level, with its west and southern boundary facing the

Aberdare mountain range (Taylor *et al.*, 2005), its southeastern corner toward Mt. Kenya (5,199 m), and transitions into the Samburu region toward the north. As one moves away from Mt. Kenya and toward Laikipia in a northwestern direction, a precipitation gradient services agriculture in the higher elevations closer to the base of Mt. Kenya's and, at lower elevations, transitions to the more arid environments in northern Laikipia County dominated by cattle ranching. Laikipia contains two major rivers, the Ewaso Narok and Ewaso Nyiro (often spelled "Ng'iro"), with a number of small tributaries that originate from the Aberdares and Mt. Kenya. These two rivers are a vital supply of water for humans, wildlife and livestock throughout Laikipia. Other sources come from aquifer fed springs, boreholes, and dams, but the northern range of the County tends to be solely dependent on the Ewaso Nyiro River (Taiti, 1992; Thouless, 1995). Temperatures have a mean annual range of 16 – 20°C (Odingo, 1971) and produce extremely arid conditions in northern Laikipia. Precipitation occurs mostly in two seasons per year (April to June and October to December) and deliver 'long rains' and 'short rains' respectively with a third season of extremely dry conditions in between (Graham, 2006). Rain can be quite variable in Laikipia with long droughts that will completely dry up major sections of riverine habitat. Despite occasional heavy rains, water can be diminished by the intense solar radiation that causes evaporation to exceed the rainfall (Odingo, 1971; Wiesman, 1994; Chamain-Jammes *et al.*, 2006). The last major drought in Laikipia County was in 2009, followed by heavy volumes of rain that have very likely contributed to an increase in wildlife population numbers.

Laikipia's human population of over 320,000 people (Kenya National Bureau of Statistics, 2014) is concentrated in urban centers and in the southern regions of the county. Current land-use in Laikipia is dominated by large-scale commercial ranching that has decreased from 57% to 42% between 1998 and 2006, followed by an increase in small-scale farming of 26% to 37%, with communally owned ranches at 8% of the landscape (Kohler, 1987; Graham, 2006).

Laikipia is largely comprised of lands of private, communal, and government ownership. It does not contain any national protection, but despite this, is well known for supporting high wildlife biodiversity and has the highest populations of endangered species in the country, including the greatest numbers of large mammals (Laikipia Wildlife Forum, 2009). Current research shows fluctuations in these wildlife populations, which are mostly declining throughout Kenya, but are relatively stable within Laikipia (Laikipia Wildlife Forum, 2009; Didier *et al.*, 2011). As a result of increased tourism and interests in biodiversity conservation, Laikipia is home to a growing number of organizations that work to protect its natural heritage and support local community participation, namely the African Wildlife Foundation (AWF), the Ol Pejeta Conservancy, Mpala Research Centre, Laikipia Wildlife Forum (LWF), LEWA Wildlife Conservancy (LEWA), and the Northern Rangelands Trust (NRT). Such interest and support brought numerous organizations together in 2006 to form the Ewaso Landscape Planning workshop (Didier *et al.*, 2011) that resulted in initiating a planning process for the conservation of the region. The combination of biodiversity and local support provides an ideal environment to explore conservation research in a human-dominated landscape.

Wildlife conservancies

The selection of the focal ranches for this study originated from discussions with staff members of the AWF in 2009, who had expressed a strong interest in learning more about the ecological value of the Eland Downs Ranch. It became evident that to learn more about Eland Downs required learning about its neighbors, the Segera Ranch and the ADC Mutara Ranch. To further add value to the landscape, agreements designed to better connect the ADC Mutara conservation area and the Ol Pejeta Conservancy are currently being developed to serve potential rhino conservation. Wildlife conservancies such as the Ol Pejeta Conservancy and the Mpala Research Centre have helped to encourage other landowners to endorse the benefits of biodiversity and have acted as a model for the importance and role of private land ownership in large landscape conservation. Management from each ranch provided access, guards, logistical support, communication with local community members, and assistance with transportation. Cumulatively, the following five ranches are considered the primary stakeholders of the Central Laikipia Collaboration (Figure 1.7):

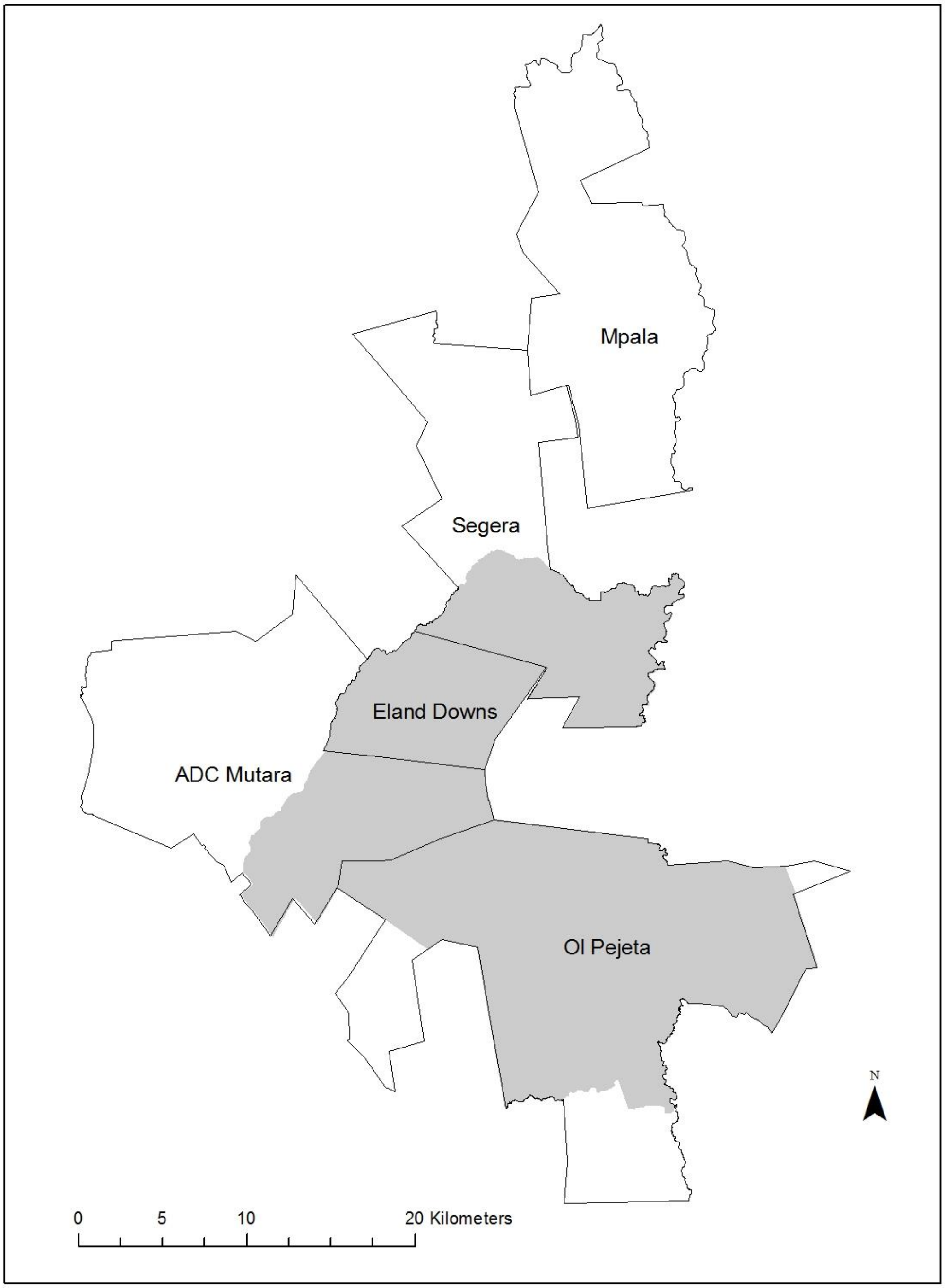


Figure 1.7. Portions of property managed for conservation sampled by camera trapping (grey) within the context of greater land ownership per ranch.

Mpala Ranch / Mpala Research Centre

The Mpala Ranch is a 50,000 acre unfenced property that maintains both cattle management, research and conservation activity. It is overseen by the Mpala Wildlife Foundation and formed the Mpala Research Trust, which then provides support toward the activities of the Mpala Research Centre, a world-renowned ecological research science facility. Unique to all of Kenya and East Africa, the Mpala Research Centre (MRC) is the conservation science engine for Laikipia County. It generates numerous peer-reviewed science literature, maintains important data for the region, and has developed a diverse portfolio of tourism and community programs. The MRC maintains accommodations for researchers, including research labs, housing, storage, lecture rooms, commons dining facilities, as well as housing and a primary school for staff and their children. It receives numerous visiting scientists and scholars from around the world. Additionally, it conducts education and outreach to communities throughout Laikipia County, provides medical services to locals, and promotes Kenya nationals to become future conservation leaders of the landscape. From private land ownership to the formation of a foundation and trust, this successful facility has substantive benefits to academia, Kenya, and the local landscape.

Segera Ranch

The Segera Ranch is a 50,000 acre property north and adjacent to the Eland Downs Ranch and south of Mpala. In 2005, the Segera Ranch transitioned from a predominantly cattle ranching operation to one that supports wildlife and community conservation. In 2009, the Zeitz Foundation of Germany established headquarters on the ranch and took over conservation management of the property. The 22,000 acre portion of Segera Ranch adjacent to Eland Downs had higher research value and was more accessible than the remaining northern half of the ranch. With added constraint to sampling efforts, all field study resources for studying Segera Ranch were focused in this area.

Management of Segera Ranch facilitated access to the property in 2013 and provided ranch guards for logistical support during all field survey and camera trapping efforts. In exchange for access and gathering of data on Segera Ranch, guards were trained in camera trapping use, field set design, and monitoring.

Eland Downs Ranch

In 2009 the African Wildlife Foundation (AWF) acquired Eland Downs ranch, a 17,500 acre (7,100 ha) property once owned by former President of Kenya, Daniel Toroitich arap Moi. At that time, the vegetation on Eland Downs had been heavily damaged over many years from extensive livestock grazing. A March 2009 baseline survey by the Mpala Research Centre (Kinnaird and O'Brien, 2009) reported a highly degraded landscape, low levels of native large mammals, and an overgrazed but existing mosaic of nutritionally valuable grasses, shrubs, and trees. It was acquired by the AWF for the purposes of connecting the larger landscape and for its geographic value as a potential bottleneck among the conservancy ranches in central Laikipia. In 2011, its ownership was then transferred from AWF to the KWS under consideration of it being proposed as an addition to Kenya's national park system. KWS has since managed the property for wildlife while allowing neighboring communities continued access to the Ewaso Nyiro River for their cattle herds. KWS will continue to limit livestock grazing on the property, but will make it accessible to the community for watering cattle during the day. Eland Downs has a long history in Laikipia as a focal point for heated land tenure disputes. Both AWF and The Nature Conservancy (TNC) have a strong record of working with multiple stakeholders on landscape level conservation management, and both had identified Eland Downs as highly important for conservation of the region and saw the successful transfer of ownership to KWS. Protection plans for Eland Downs go beyond support for wildlife, to include a number of benefits to the local community, as outlined in AWF's former Samburu Heartlands conservation strategy (AWF, 2010).

In Laikipia, many pastoralists perceived that native ungulates compete with their livestock for vegetation resources (Mizutani, 1999; Georgiadis *et al.*, 2003), and at times have responded by displacing or destroying these herbivores from their grazing areas (Heath, 2000; Riginos and Herrick, 2010). Overuse of forage is clearly a concern for both livestock and wild fauna, and is similar to other group ranches throughout Kenya. Large herbivore populations were found to be severely depleted on Eland Downs (Kinnaird and O'Brien, 2009), with results showing Thompson's gazelle (*Gazella thomsonii*) and Grant's gazelle (*Gazella grantii*) as relatively common, with an absence of other savannah and bush country species. Larger ungulates, such as Eland (*Tragelaphus oryx*), Greater Kudu (*Tragelaphus strepsiceros*), Waterbuck (*Kobus defassa*) and Hartebeest (*Alcelaphus buselaphus*) were not encountered during the Mpala survey, thought they did observe an occasional zebra (*Equus burchelli*). The absence of large ungulates, common on adjacent properties, was suggested to be a result of poaching or from displacement by livestock (Kinnaird and O'Brien, 2009). The number of livestock was estimated at 3,500 for a property of only 17,500 acres (69 km^2), an extremely high density given that nearby ranches of 50,000 acres (194 km^2) maintain less than 2,000 head of livestock. Wildlife transect data recorded during this study from 2012 and camera trapping data in 2013 reveal a return of many of these large herbivores, with high traffic of elephant, reticulated giraffe (*Giraffa camelopardalis reticulata*), and other large mammals. In addition, species usually poached in such human-dominated areas, such as the warthog (*Phacochoerus africanus*), were also observed during these studies. Vegetation biomass and ground cover surveyed in 2009 increased from previous levels, indicating that the habitat on Eland Downs ranch is in recovery. Eland Downs has additional recovery potential, if nearby landscapes which have transitioned in similar fashion are indicative of patterns for the region. The Ol Pejeta Conservancy and conservation areas of ADC Mutara were strictly cattle ranches at one time, but now contain high numbers of carnivores and herbivores (Wahungu, 2010), having reduced the livestock densities to provide for the growing populations of native wildlife over many years. Across Eland Downs, the vegetation and its structure have been significantly altered due to overgrazing and charcoal burning, a problem which has persisted in much of Laikipia County, Kenya (Okello et al, 2001). Species of grasses, forbs, and other vegetative ground cover were found to be poorly represented (Figure 1.8) in the Mpala baseline survey (2009). Land cover revealed substantial bare soil, with a lack of litter and highly browsed shrubs, indicative of intense grazing (Kinnaird and O'Brien (2009). The dominant Acacia species on the property (*A. drepanolobium*) is high in nutrition for livestock and wildlife (Okello *et al.*, 2007; Brody *et al.*, 2010), and are commonly found on the black cotton soils of Eland Downs.

Degraded landscapes can possess high value for conservation, mostly with regard to their ability to recover and contribute to nearby or adjacent conservation areas, as is the case with Eland Downs. However, the value of the habitat will be dependent on the

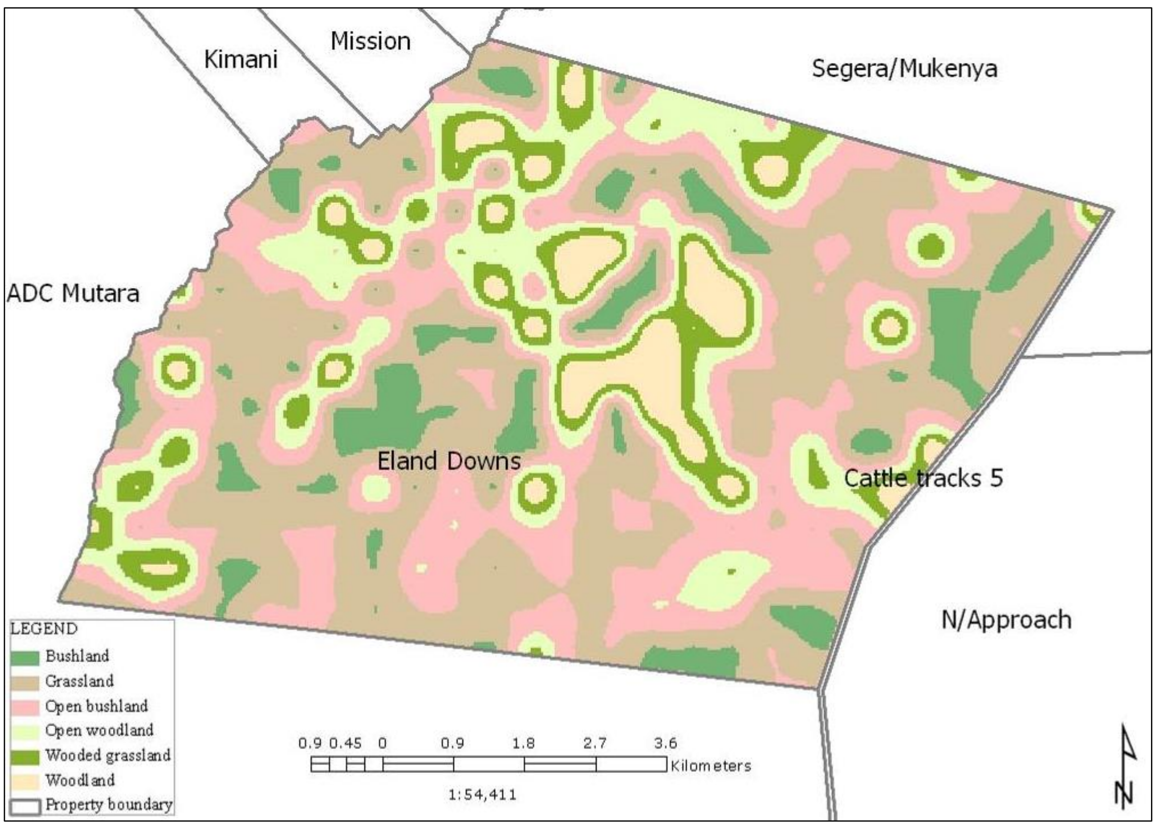


Figure 1.8. Land-use classification of Eland Downs established by the Mpala Research Center (Map courtesy of AWF; Kinnaird and O'Brien, 2009).

ecological influence and condition of surrounding habitats and management that supports conservation efforts. The timing of management changes on Eland Downs provided a unique opportunity for research that reflects priorities set by AWF and the KWS. Changes on Eland Downs from 2009 to 2013 will supply AWF with information they can use in evaluating similarly degraded properties in Kenya for future acquisition.

ADC Mutara Ranch

The ADC Mutara Ranch is a 63,000 acre government owned property primarily used for cattle and agriculture development. In 2007, the Ol Pejeta Conservancy, the African Wildlife Foundation and ADC Mutara began to establish a partnership to rehabilitate a 20,000 acre portion of the ranch for wildlife conservation known as the Mutara Conservancy (Figure 1.9) (Van Eden *et al.*, 2014).

The main objective of the conservation area is to provide for tourism revenue as well as allowing the movement of wildlife, in particular migratory elephant that travel between Laikipia and the more northeastern Samburu County. The conservation area of ADC borders the Ol Pejeta Conservancy and is therefore an ideal public-private sector partnership investment for future expansion of black rhino conservation. In 2014 the Jambo Cheser hotels & Resorts leased the 20,000 acre conservation area to build the Jambo Laikipia Tented Camp that includes 15 luxury tents. The Mutara ranch and the Ol Pejeta Conservancy signed a memo of understanding to allow guests on ADC to access the safari benefits of OPC.

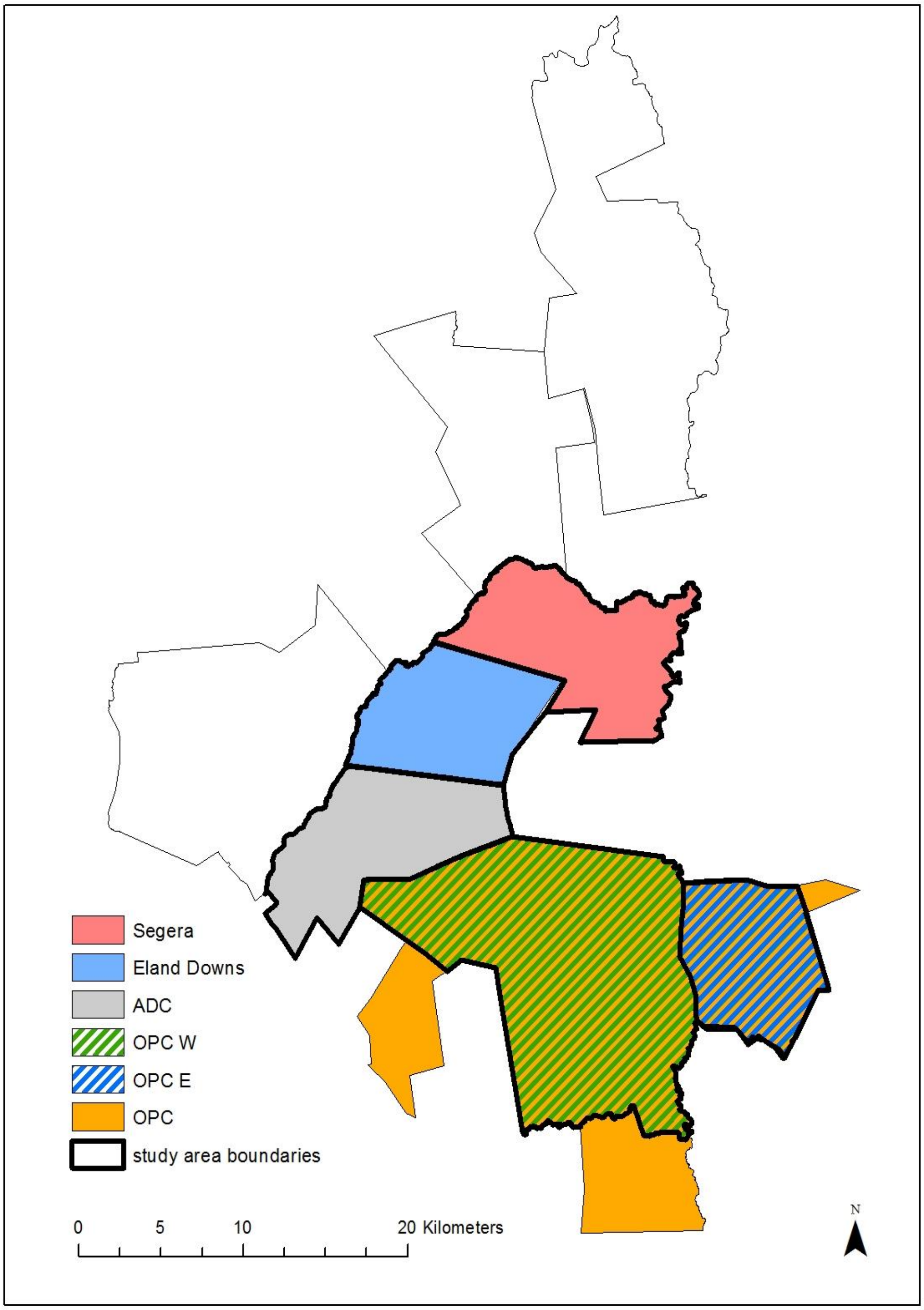


Figure 1.9. Areas sampled in this study (bold) delineated from four adjacent ranches in Laikipia County, Kenya. From North to South:, Segera, Eland Downs, ADC Mutara, and Ol Pejeta (East and West).

The Ol Pejeta Conservancy

The Ol Pejeta Conservancy was formerly known as the 25,000 acre Sweetwater's Game Reserve, referred to as "OPC East" in this study. Through wise investments over many years, OPC obtained a neighboring swath of property of 65,000 acres and today the conservancy is a combined 90,000 acres hosting ecotourism and tented-camp safaris, Boran and Ankole cattle operations, wildlife conservation activity, a research center, as well as significant community-related programs (education, medical, water resources). With strong and reliable infrastructure and high wildlife abundance, OPC is one of the most visited wildlife tourism destinations in all of Laikipia. It represents the southern anchor of the Central Laikipia Collaboration and is the only conservancy to maintain a population of black rhino.

The OPC landscape is unique and contains a large wetland and riverine environment, in addition to the more traditional Acacia-grassland ecosystem consistent with other ranches in this study. Ol Pejeta, with its history of managing the Sweetwaters Game Reserve, can be viewed as two adjacent ranches separated by the Ewaso Nyiro River. Given the management history, size of the entire ranch, and curiosity about the likely difference between the eastern and western halves, I decided to sample them independently. The results of this study will therefore include information on Ol Pejeta (the entire ranch combined, OPC), Ol Pejeta East (OPC E) and Ol Pejeta West (OPC W). In addition, there are three areas of the Ol Pejeta Conservancy that could not be sampled, a small section in the northeast under tourism/housing development, a southern portion dedicated to wheat production, and a western portion dedicated to the majority of the cattle operation.

The combination of all areas sampled in this study and across all ranches is over 129,500 acres of property or 524 km^2. The logistics of sampling across four ranches presented numerous challenges, but were eventually overcome by the generous support and guidance of NGO facilitation, ranch management, and supportive guards and researchers on site.

Research design

Initial site visits to Laikipia beginning in 2009 provided contact with ranch managers and allowed for general environmental observations on each ranch. These helped to establish a new and broad level habitat classification system and for planning and feasibility of the study. Preliminary site visits were followed by building a geographic information system (GIS) to randomize and establish a minimum of 80 geo-referenced points (stations) where vegetation and camera trapping surveys could be conducted across the entire landscape and specific to grassland, Acacia forest, Euclea forest, mixed Acacia-Euclea forest, and riverine habitat. Habitat classification was established through line transect vegetation surveys to record grass, tree, and shrub species diversity, percent cover and height, and dominance and canopy cover. Following these surveys, cameras were deployed at each of the same locations over a 3-month period to simultaneously record wildlife across all four properties. In addition, cameras were placed on the only existing wildlife corridors (gaps in fencing) connecting ADC Mutara and Ol Pejeta Conservancy ranches.

New tools for conservation managers in a public/private collaboration

One should not assume that land managers in close proximity to one another or on adjacent properties have the necessary collaborative tools to manage for the greater landscape. Managers often lack access to tools and resources that would otherwise provide the overview necessary to make decisions for long term conservation planning. As a product of this study, we bring together important data and information into an on-line management tool freely accessible to all managers in this community. Using Esri (Environmental Systems Research Institute) ArcGIS On-line and Story Map™ technology, I have designed and made available interactive and on-line maps (Figure 1.10) of all ranches, habitat, wildlife distribution, water resources, infrastructure, and special features across the landscape. The functionality of these tools include panning and zoom, interchangeable layers and base maps, measuring tools for distance and area, printing options, and most importantly a suite of sharing tools for email and/or social media that can benefit internal management or provide for public outreach and communication. Providing unique geographic information within a dynamic mapping interface will supply managers with new options for communication and landscape planning.

Much of the early research on large African mammals focused primarily on natural history, population dynamics, or studies of particular wildlife species. Current wildlife and rangeland management requires a greater understanding of large-scale ecological systems, multi-species interactions, and the effects of anthropogenic factors. With advances in technology for monitoring and assessments of biodiversity, managers can make more informed decisions toward large landscape conservation planning. In this study I am joining three important and influential variables: habitat, ranch management, and the diversity, distribution, and abundance of large mammals. The results of a GIS detailing habitat classification, combined with camera-trapping data across four wildlife conservation areas has been incorporated into the development of a unique on-line mapping tool for current and future landowners in Laikipia, Kenya.

Figure 1.10. Story Map™ geographic tools designed to explore camera trapping imagery, interact with various layers of information (habitat, anthropogenic structures, and locations of camera traps).

CHAPTER 2. Determining habitat preference and estimating diversity, distribution and abundance of large mammals.

2.1 Introduction

Data previously available from the focal ranches were incomplete and somewhat dated. These ranches are of various sizes, managed independently, and contain a mosaic of highly impacted habitat from different historic uses. It was therefore necessary to reestablish these data, build a comprehensive overview of the landscape, and provide descriptive statistics on habitat, landscape, and wildlife. Therefore, the ability to assess habitat preference and estimate diversity, distribution, and abundance of large mammals in this study required considerable preliminary research. These included developing GIS layers for standard spatial analyses, generating maps, georeferencing key landscape features, establishing vegetation transect surveys, and positioning deployment of camera traps.

These data allowed me to address the following preliminary questions and hypotheses. First, I hypothesized that ranch size was correlated to species richness, assuming that larger ranches would have higher estimated diversity. Specifically, the largest of ranch on this landscape, the Ol Pejeta Conservancy of 90,000 acres (364 km^2), was hypothesized to maintain greater and more complex habitat and harbor larger numbers of large mammals. Conversely, the smallest ranch in this system, the Eland Downs Ranch of 17,500 acres (71 km^2), was predicted to maintain the lowest estimates of species richness given its size and historic land-use of intense livestock grazing.

Given that this habitat distribution in landscape has been highly altered and directed by humans for over a century, I tested the habitat preference of selected trophic guilds (grazers and browsers). I hypothesized that grazers would be more abundant in grasslands and that browsers would be more abundant in Acacia habitat. Because carnivore habitat preferences are largely unknown, I included exploratory analysis of this functional group, with the tentative predication that carnivores will be most diverse in riverine habitat. I deliberately excluded analysis of data for grazer-browsers, omnivores, and insectivores due to the overlap of habitat use and that their elusive nature will likely lead to a lower detection probability with camera trapping surveys, as compared to mammals with a much larger body mass.

Natural history and classification of habitat

Kenya's Ewaso Ecosystem of nearly 56,000 km2 is rich in large mammal diversity, attracts tourists and researchers, and contributes to the country's economy. Laikipia's species diversity is a direct result of the mosaic of habitat spread throughout this generally cool, dry climate of the central high plains. The landscape is comprised of savanna grasslands intermittent with Acacia woodland and Euclea bushland forests. One of the limiting factors for vegetation is the well-known "black cotton" soil, a fine volcanic substrate often becoming brittle and dry without precipitation but quickly saturates into a low porous soil of poor drainage during the rains.

The area sampled in this study between the Ol Pejeta Conservancy and the southern portion of Segera Ranch is a contiguous landscape dominated by the whistling thorn Acacia (*A. drepanolobium*) and open grasslands, with intermittent densities of *Euclea* forests near major rivers and drainage basins. The Ewaso Nyiro River that travels through all ranches in this study provides the region with its only source of natural water. The wildlife in this ecosystem tends to be habitat specific with preference for habitat often influenced by factors such as foraging availability and diet (Rubenstein, 2011), shelter from exposure to weather and predation (Wahungu, 2010), as well as a variety of behavioral aspects driven by their immediate environment (Odadi, 2009).

Five generalized habitat types (Acacia, Euclea, Grassland, Mixed, and Riverine) were chosen for this study based on prior habitat classification systems from the Ol Pejeta Conservancy, Moi University, and the Mpala Research Centre (Table 2.1). In addition, local knowledge and supplemental vegetation surveys helped to derive these classifications. For purposes of the research, I simplified all existing habitat classifications into the following five categories.

Table 2.1. General classification of habitat on properties during study.

Habitat Classification	Description
Grassland	Dominance of grasses, very few trees or shrubs
Acacia	Dominance of Acacia trees
Euclea	Dominance of Euclea bush
Mixed Acacia-Euclea	Overlap of Acacia and Euclea habitat of equal mix
Riverine	Wetland and marsh habitat

Acacia

Acacia forests were combined from prior classification of "closed *A. drepanolobium* woodland," "open *A. Drepanolobium* woodland," "closed woodland," and modified "open woodland." This grouped classification is comprised of tree-dominated areas which may contain several species of *Acacia* trees and the occasional *Boscia* sp. tree. Canopy cover and tree densities are relatively high. These Acacia forests are the primary food source of the black rhino and are commonly known as the most important tree species supporting wildlife diversity in Laikipia (Wahungu, 2010). Acacia are often threatened by encroachment from *E. divinorum*, elephant damage, and compaction of soil from overabundance of livestock. Seasonality (dry vs. rainy season) has a considerable impact on tree morphology as seen in Figure 2.3. Forage availability, shade, and seed availability are quickly affected by precipitation. *Acacia drepanolobium* was reclassified to *Vachellia drepanolobium* in 2013 (Kyalangalilwa, *et al.*, 2013). For purposes of consistency with all prior literature and the use of Acacia as a habitat type, I continue to use *Acacia drepanolobium* in this study.

Figure 2.1. Contrasting foliage for *Acacia drepanolobium* between the dry (left) and rainy (right) seasons.

Euclea

Euclea classification was combined from “Euclea bushland”, “*Euclea divinorum”*, a modified “open bushland”, and “bushland” and represents a fairly monotypic habitat usually consisting of only a few species of bush dominated by the evergreen *E. divinorum*. The Ol Pejeta Conservancy has recorded Euclea encroachment in both grasslands and Acacia forest habitats. Euclea remains green and foliated during times of drought and is often selected for shade by various species of wildlife. The structure of Euclea is such that it protects and physically supports other shrubs and vines growing under and within it, which often then form large, dense vegetative masses. During droughts, wildlife will feed on Euclea as a last resort, but it is not a preferred food source.

Grassland

Grasslands were classified from "*Themida triandra* open grassland," "Oxalis-Eragrostris-*Pennisetum* grasslands," "Eragrostis-Digitaria-Chloris grasslands," "open plains grasslands," and "grasslands." This classification consists of pure grassland of any species of grasses with virtually no tree cover. The boundaries of a grassland are discernable through satellite imagery (Figure 2.2), which made delineation of habitat fairly easy. Savanna grasslands are the quintessential habitat known for harboring herds of large mammals such as buffalo, elephant, and wildebeest (Douglas-Hamilton, 1987), whose distributions and impact are often affected by presence or absence of predators (Schmitz, 2008). Grasslands throughout the study area are dominated by the red oat grass, *Themida triandra*. These grasslands are high in productivity, and very quickly transform to nutrient-rich, green foliage with increased precipitation and are capable of sustaining large herds of large mammals. It is for this reason that a reduction in cattle on Eland Downs will most likely result in a rapid increase in visitation of large mammals from neighboring lands. Fire plays an important role in maintaining grassland ecosystems, but the removal of fire as a planned management tool in Laikipia has created challenges from woody shrub encroachment (Gregory *et al.*, 2010). Ecological engineers such as elephant can help reduce tree and shrub encroachment on grasslands as they often push trees over and pull young trees and shrubs out of the ground (Porensky and Veblen, 2012). Their prior population numbers were a concern for wildlife managers who are now seeing an increase in the Laikipia elephant population.

Riverine

Considering the close proximity of all existing riverine classifications to actual rivers and river basins, riverine habitat in this study was combined from "*A. xanthophloea* open habitat", "riverine grasslands", "marsh", and "wetlands". With the Ewaso Nyiro River running through all ranches, riverine habitat was made possible as a shared classification. It makes up the lowest percentage of habitat per ranch, but is crucial to the survival of fauna and flora species throughout the landscape. The very large *Acacia xanthophloea* trees, also known as the yellow fever tree, are found almost exclusively in this habitat and are the preferred resting tree for olive baboons, leopards, and numerous species of birds. Their height and wide-spreading canopies are easy markers to spot landscape river beds and active drainage basins.

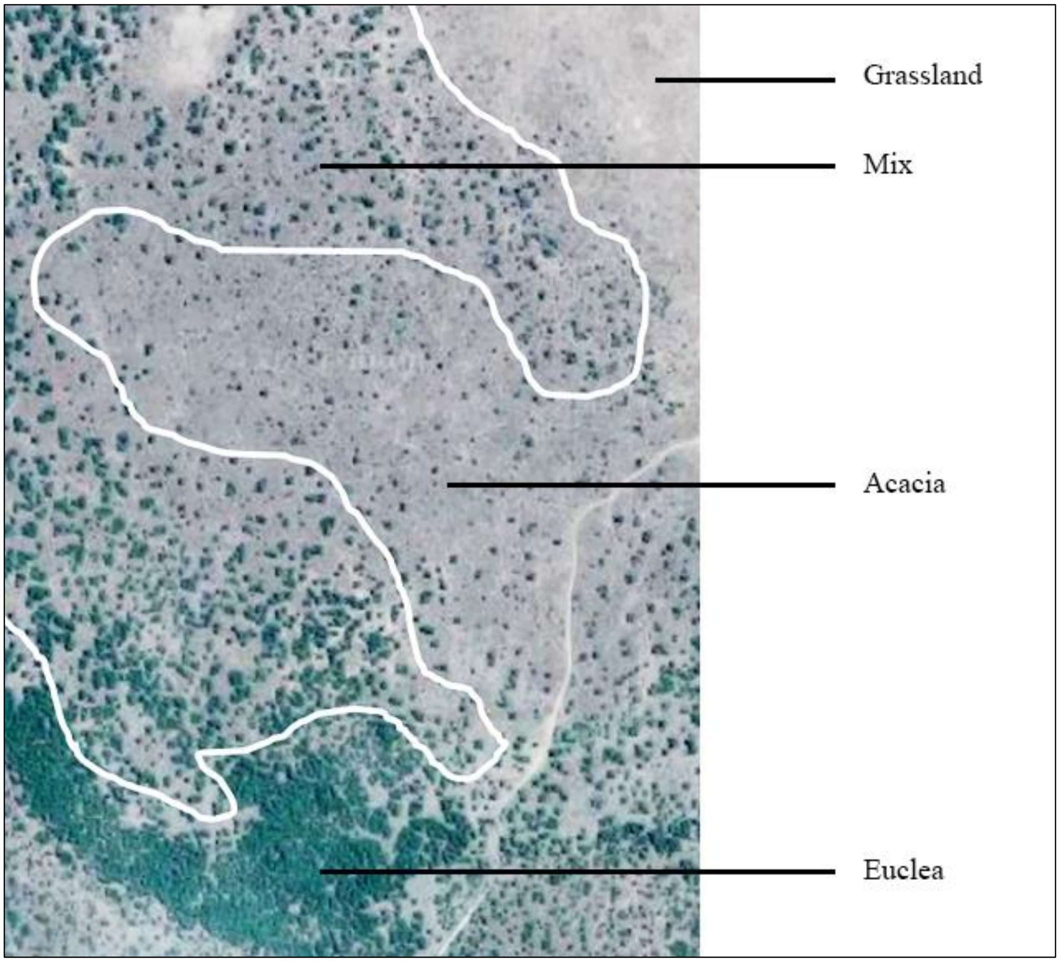


Figure 2.2. Example of a satellite image with classification of habitat. The white line establishes delineation of "Mixed" Acacia/Euclea habitat from surrounding landscape.

Mix

The Acacia-Euclea "mix" habitat was uniquely established as a category in this study for the sole purpose of addressing the need for black rhino to have access to both browse availability and density of vegetation (Lush *et al.*, 2015). These two characteristics are captured in this mixed classification. In addition, it is known among the Ol Pejeta Conservancy guards and wildlife research staff that black rhino are often seen in this mixed habitat (Mulama, 2012). Here, they are able to both browse on *A. drepanolobium* and rest in the seclusion and shade provided by *E. divinorum*. The intersection of Acacia and Euclea is a transition zone between these two habitats and is visually discernable. I classified mixed habitat as one featuring a relatively equal amount of *A. drepanolobium* and *E. divinorum*. Since Ol Pejeta is the only ranch in this system containing black rhino, a benefit to this classification is that it is projected on the greater landscape and might help provide information for future expansion of black rhino into neighboring properties.

The collection of the five distinct habitat classifications (Figure 2.3) simplifies numerous previously recorded classification systems for purposes of correlating habitat with trophic guild occupancy and species diversity.

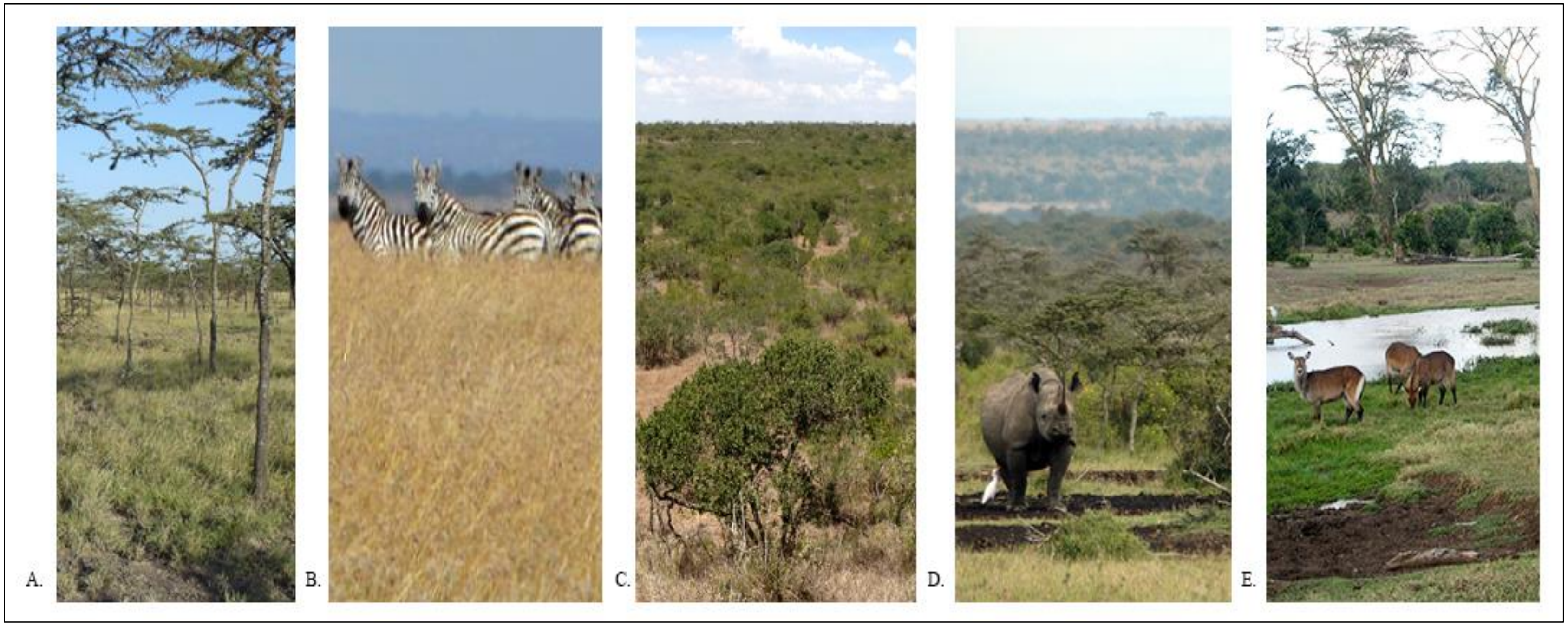


Figure 2.3. Generalized habitat classification for this study made up of Acacia (A), Grassland (B), Euclea (C.) Mixed (D.) and Riverine (E.).

Functional roles of large mammals in Laikipia

The large mammals of Laikipia County, Kenya are important to its ecology, tourism, and contribution to tropical savanna ecosystem science. This group of animals is a driving force in landscape conservation, requiring large range and distributions while promoting the need for corridors and connectivity.

Pressures from habitat alteration, fragmentation, human encroachment, and livestock management have influenced the distribution and diversity of Laikipia's large mammals. Historic wildlife migration routes have changed while new structures such as fencing and roads have affected species distribution and abundance. Knowledge of the factors determining these changes in native ungulates is important to conservation (Soule *et al.*, 2003; Sankaran *et al.*, 2013), especially when conservancies are in a contiguous fashion, adjacent to one another and with unique opportunities to share resources.

A strong example of a species highly affected by anthropogenic factors in Laikipia is the African elephant (Osborn and Parker, 2003). Elephants are often problematic among cattle in competing for grazing, and in agricultural areas where crop raiding has led to high economic loss and human casualties (Odadi *et al.*, 2007; Graham and Ochieng, 2008). They were virtually extirpated from Laikipia prior to the 1970s. Their absence likely contributed to the rise in woody species encroachment on grasslands and equally (Franz *et al.*, 2010), the lack of microhabitat from a decreased amount of fallen trees and disturbance to the top soil (Franz *et al.*, 2011). Elephant maintained historic migration routes between Laikipia and the Samburu region to the northeast

(Figure 2.4). Their return can be attributed to higher tolerance of sharing the landscape with humans and from the transformation of many cattle ranches to wildlife conservancies. Artificial water holes once built exclusively for cattle, are now shared with native wildlife on many private ranches. Today, elephants are still in conflict with humans in the form of damage to fences (Graham *et al.*, 2009; Graham and Ochieng, 2010) and continual crop raiding (Sitati and Walpole, 2006). Opening the landscape to a larger collection of natural, and in some cases unnatural (water holes), resources will provide the growing elephant population in Laikipia with alternatives to conflict. Other species of grazers commonly found in Laikipia grasslands include Cape buffalo and zebra, which at times are also in conflict with humans by causing casualties and as a foraging competitor with cattle.

Browsers, such as the giraffe, are lucrative species on a conservancy and serve important functions in ecology and tourism. Foraging behavior of the reticulated giraffe was recently discovered to have a synergistic effect on the survival of *A. drepanolobium* through research on the impact to various colonizing ant species (Palmer and Brody, 2012). Its removal as a large mammal could result in the death of Acacia trees at a landscape scale (Maclean *et al.*, 2011a, 2011b). Such research efforts should encourage us to be concerned with unknown ecological impacts from species removal. The conservation landscape, which was sampled during this research, includes other important browsers such as the endangered Jackson's hartebeest and the well-known and highly endangered black rhino. As the only native rhino species to East Africa, the black rhino are smaller and non-overlapping with the grazing white rhino.

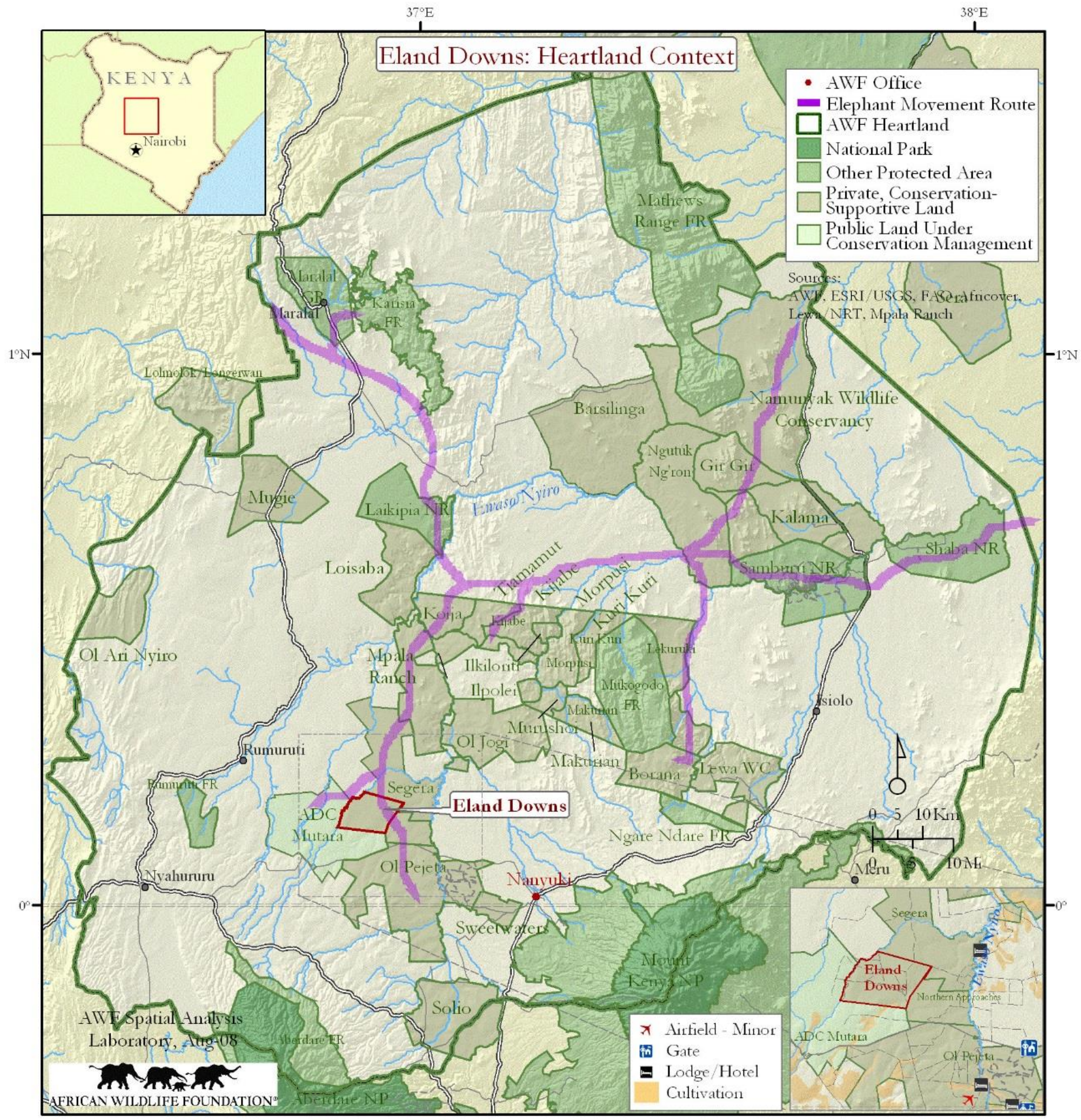


Figure 2.4. Locations of historic elephant corridors (purple) in the context of greater landscape (map courtesy of AWF).

Ironically, it was fencing that saved the black rhino from increased poaching in the 1980s, which nearly brought them to extinction. High security operations involving strong and often electrified fencing with additional field guards were required to protect the remaining populations in Kenya. This can be seen today on both the Ol Pejeta and Lewa Conservancies. In addition, Ol Pejeta has increased security further and maintains three guards per individual rhino. With a current population of 107 animals, this is a substantive effort to protect this highly endangered species.

Studies addressing a cumulative effect from joining multiple properties under conservation status into a whole system are still uncommon. Research in the past five years has brought this important concept to the forefront of landscape level conservation in Laikipia (Kinnaird and O'Brien, 2012). The geographic distribution of the ranches in this study provide such an opportunity, and will add to the pool of useful information for local land managers.

A key factor that has impacted and limited the assessments of habitat for large, wild ungulates is the challenge of estimating density and relative abundance (Funston *et al.*, 2010). This challenge usually comes from poor or low detection rates from direct sightings. There are also problems when trying to extrapolate from indirect evidence of wildlife presence (dung and tracks, tourism observations). This is where camera trapping comes in as a strategic and optimal tool. It has proven useful for biological inventories, detecting elusive wildlife, while at the same time (in Laikipia) providing added security measures.

2.2 Methods

This study required a strict and sequential approach to field work. Reconnaissance trips were undertaken in 2008 and 2009 to determine feasibility and logistics. These included meeting management and administration staff for each of the ranches in question, determining requirements for access, and building on-site logistical support. Secondly, the necessary affiliate status was secured through Dr. Geoffrey Wahungu of Moi University, now the head of NEMA (National Environmental Management Authority). Research permits and necessary administrative work was submitted, a plan was developed to begin field access in 2010, and my research base was established at the Ol Pejeta Conservancy. Baseline GIS development began in 2009 with the first field survey conducted in 2010 on ADC Mutara. Field surveys to delineate habitat on each ranch continued through 2013, at which point camera deployment commenced. Considerable logistical support was administered through the African Wildlife Foundation offices of Nairobi and Washington, DC, in addition to support from George Mason University (GMU) and the Smithsonian Institution (SI).

Development of a geographic information system (GIS)

Prior to any field studies, a preliminary GIS investment was made to explore and obtain available vector-based and remotely sensed data to assemble maps needed for this research. The initial GIS and Remote Sensing (RS) support came through the African Wildlife Foundation (AWF) in providing general administrative data (boundaries, roads, rivers, etc.) and remotely sensed images of Laikipia County, Kenya using SPOT 2010 imagery. This effort was followed by a remote sensing training grant through the American Association of Geographer's (AAG) SERVIR program providing access to Kenya's Regional Center for Mapping of Resources for Development (RCMRD) in Nairobi as per agreement under an AAG/NSF grant (# 0934063). Three months of training was provided at the RCMRD, which included remote sensing analysis and access to their imagery database for cloudless images of Laikipia County between the years of 2009 and 2013. In addition, an imagery grant was received in 2010 from the GeoEye Foundation covering the majority of the study area. Data derived from these sources were combined with support from the African Wildlife Fund (AWF) GIS Lab in Nairobi and Washington DC, and the Smithsonian Institution's GIS resources.

Minor image processing utilizing ENVI™ (Environmental Vegetation Index) software (EXCEL/VIS, 2015) helped identify habitat prior to field ground-truthing surveys and to assist with identifying long term vegetation monitoring plots (Figure 2.5).

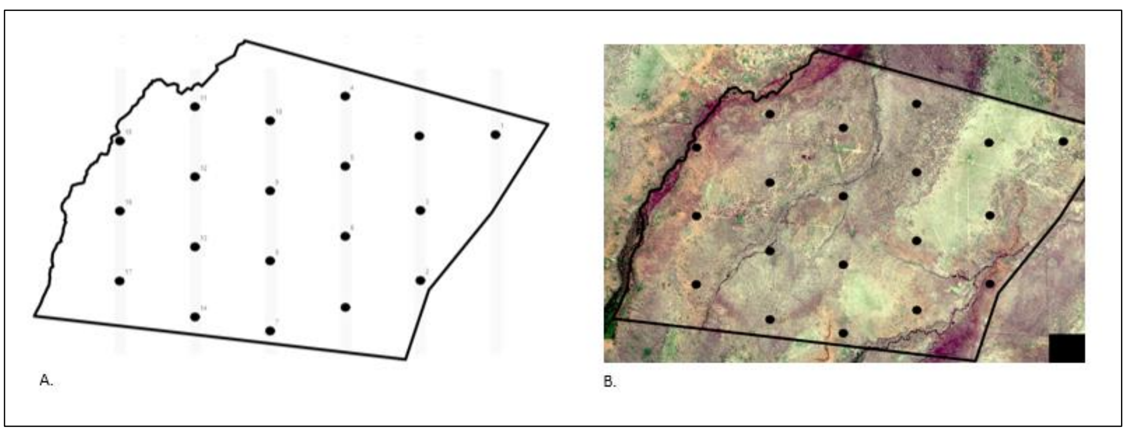


Figure 2.5. Locations of long-term vegetation survey stations established on Eland Downs.

The use of Arc GIS 10.3.1, ArcGIS Pro™ Advanced (Esri, 2015), Google Earth Pro (Google, 2015), and Earthpoint.us (Clark, 2015) online measuring tools were incorporated into all GIS use throughout this dissertation. ArcGIS technology was adopted to establish habitat classification and delineation through overlay on Esri satellite in addition to identifying anthropogenic structures such as roads, bore holes, and human settlement areas.

Vegetation field surveys

The vegetation surveys conducted throughout the study combined methodologies used during the baseline survey of Eland Downs by the Mpala Research Centre in 2009 (Kinnaird and O'Brien, 2009), surveys conducted by Moi University on ADC Mutara and Ol Pejeta in 2010, surveys by the Kenya Wildlife Service team on Eland Downs and ADC Mutara in 2012 and 2013, and from existing and new surveys on the Ol Pejeta Conservancy and Segera Ranch in 2013.

High resolution SPOT (CNES, 2015) and 2010 Google imagery were first used to determine major habitat types and to pre-determine suitable sampling locations prior to assigning random camera trapping locations Vegetation surveys were conducted at the site of each camera location. General imagery analysis helped to confirm important landscape features and eliminate bias during ground-truthing. The Eland Downs ranch had limited information on GIS habitat delineation and the ADC Mutara Ranch had no prior GIS work conducted. These two ranches required thorough ground-truthing. Segera Ranch and the Ol Pejeta Ranch had some available GIS information, which was modified and corrected with imagery analysis as vegetation edges and human structures had changed.

Fifteen stratified vegetative transect surveys, averaging 10-12km in length and 1km apart were conducted between February – March seasons over a 3-year period on ADC Mutara and Eland Downs ranches. These transects covered an estimated 160km^2 of landscape, with an additional 370km^2 by vehicle. Surveys were conducted on foot for Eland Downs and ADC Mutara ranches, and by Toyota Land Cruiser and Suzuki Martuti vehicles on Ol Pejeta and Segera ranches (Figure 2.6). All surveys were conducted between the hours of 0530 and 1700, as these long transect took all day to walk from one side of a ranch to the other. Surveys on ADC Mutara and Eland Downs were conducted with consistent teams made up of students, field technicians, and KWS park rangers and science staff. Access and logistical assistance was provided by ranch managers and gate attendants.

Figure 2.6. A group photo of the Kenya Wildlife Service transect team in 2013 (A.) and while conducting field vegetation surveys on the Eland Downs Ranch (B.).

The primary purpose of the vegetation transect exercises was to secure ground-truthing for habitat classification, for conducting radial analyses for tree diversity and percent cover (30m radius), estimates of shrubs and grasses (15m radius), and grass cover and height (1m quadrats) (Figure 2.7) at the location of each camera.

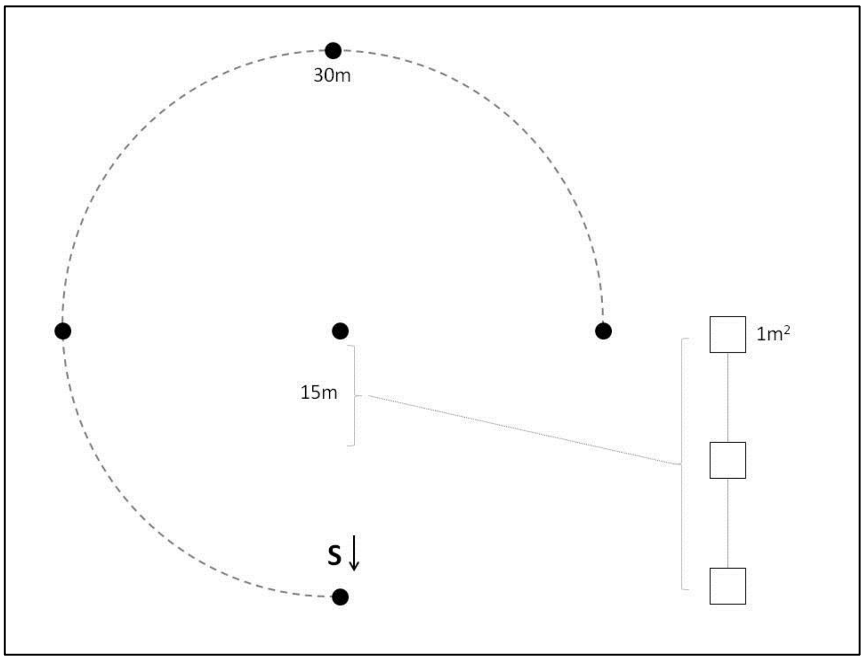


Figure 2.7. The design of vegetation surveys, including a 30m linear transect utilizing three $1m^2$ quadrats to estimate grass species, height and cover, and a 30m radial survey for estimating composition of trees and shrubs and to establish tree height and percent cover of trees.

Each survey location was isolated through use of a GPS Garmin 60cx, and measurements were collected using 50m field measuring tapes, a 1 x 1m metal frame, and a 1-meter wooden ruler. The metal frame was placed at 3 intervals (0, 7.5m and 15m) along a 15m transect to estimate grass species, cover, and maximum height. Upon location of the predetermined GPS coordinate, a 30m line transect was also placed on the ground consistently in a north-south direction. The team split into four equidistant positions creating a circle with a radius of 30m. We were then able to very efficiently record the number of trees and shrub diversity per quarter, with one person assigned the role of estimating tree cover. Subsequently, the 30m transect was reduced to 15m where % cover of grass and trees were recorded (Figure 2.9B). Vegetation survey data were not performed for use in statistical analysis, but was conducted to build a quick reference of vegetative species and provide a guide toward general condition and ground-truthing at the site of each camera location.

Camera trapping surveys

Two infrared camera trapping models were employed during this study: Spypoint BF7™ and Reconyx HyperFire ™ (Figure 2.8). Spypoint cameras were selected for their black infrared LED (Light Emitting Diode) lights, making them nearly impossible to see at night by the human eye. They were strategically assigned to Segera and Eland Downs ranches out of concern for theft, since both ranches have high visitation of neighboring community members with their livestock that would likely encounter the cameras during cattle grazing periods. Reconyx HyperFire™ cameras utilize a standard red LED, and these cameras were placed on Ol Pejeta and ADC Mutara ranches that maintain high security and virtually no visitation by community pastoralists.

Figure 2.8. A camera-trapped image of a giraffe (*Giraffa camelopardalis*) walking through grassland habitat captured with a Reconyx™ Hyperfire camera. Embedded metadata on the image reveal the date, time, and series (upper left), lunar cycle and temperature (upper right), and camera identification number (lower left). Remaining metadata can be extracted with image software.

Camera traps were randomly assigned to pre-established habitat classification polygons (Figure 2.9) using a GIS. Three individual habitat polygons of the same habitat classification were randomly selected per ranch, and within each of those polygons a 50m grid was overlaid to further randomize the location of a cell where a camera-trap would be deployed. Once the cell was identified, a quick visual assessment of the area was undertaken by vehicle, since many of these locations require brief periods of hiking. These locations were selected in 2012, and once the site was deemed suitable, the exact location of the camera was chosen the following year (2013). Cameras were set and positioned by locking them to the nearest and most appropriate tree within a 50m radius and a GPS point recorded as the camera's location.

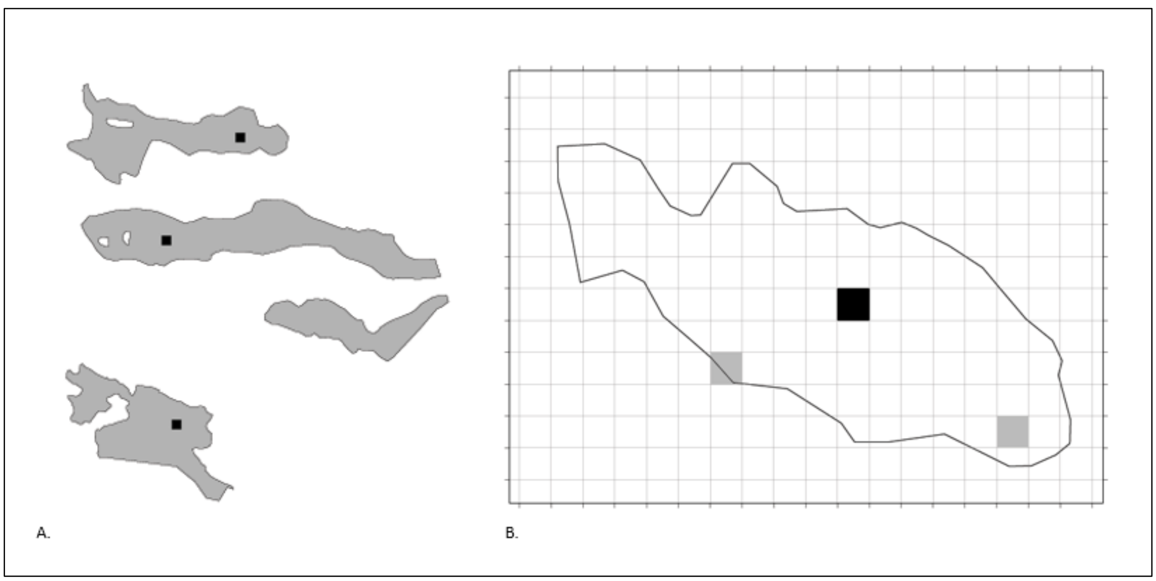


Figure 2.9. The process of using a GIS to randomize habitat polygons (A) followed by camera locations within each polygon (B) using a 50m grid system per habitat per ranch.

A maximum of 83 independent camera stations were in operation during this study including cameras that failed, were relocated, discontinued in use, and of which one was stolen. A minimum of 60 cameras (3 per each of 5 habitats per ranch) were in use for continuous coverage during the study for a period of 60 – 90 days from January to March, 2013. Camera-taps were randomly assigned throughout the four ranches within their designated conservation areas (129,500 acres, 524km^2) (Figure 2.10) in each of the five classifications of habitat per ranch. Each habitat per ranch was sampled by a minimum of three independent camera-traps, though additional cameras were used when available. Camera trapping logistics across this landscape were complicated with maintaining camera batteries, monitoring camera safety and condition, replacing memory cards and attending to any maintenance issues that arose. Traveling to monitor and deploy cameras was the primary challenge in this study, and I resorted to hiring teams of ranch guards and support staff to supply much of the monitoring effort so that the focus of efforts could be spent on priority camera maintenance and data recovery.

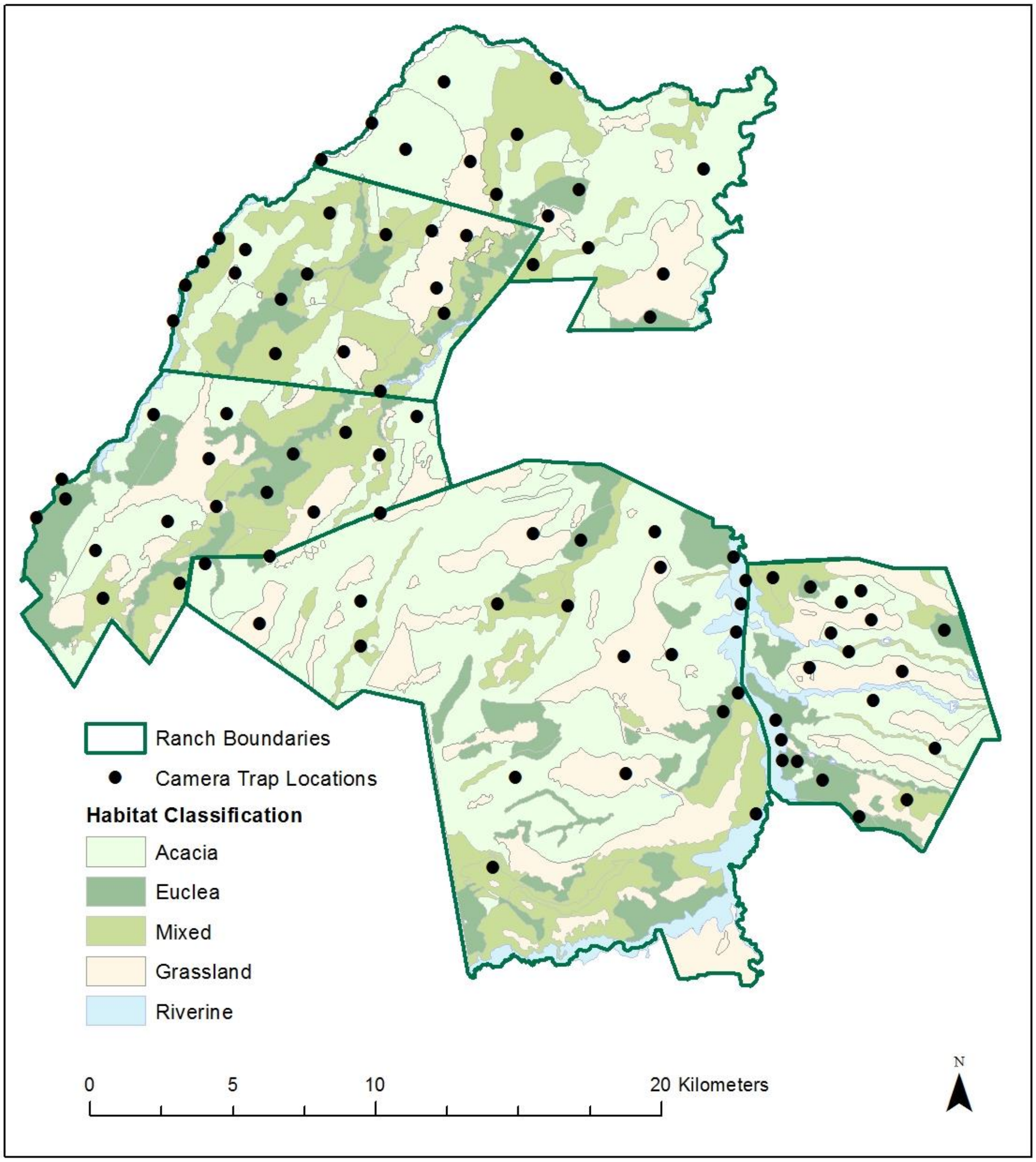


Figure 2.10. Locations of camera traps overlaid onto habitat classification.

Camera traps were generously provided from three institutions; George Mason University (GMU), the Smithsonian Institution (SI), and the Ol Pejeta Conservancy (OPC). With the availability and number of cameras on-site, camera-trapping deployment experimental design was customized to account for damage, card failure, and unknown variables that often present themselves in the field. The act of quickly moving, replacing, and redeploying any failing cameras throughout the entire 69 km^2 field study contributed to reducing pseudo-replication and other camera-trapping bias (Hurlbert, 1984; Tobler, *et al.*, 2008).

Cameras were set at a height of 0.4m - 0.6m above ground to provide opportunity to document mammals of low height, such as the white-tailed mongoose (*Ichneumia albicauda*), while also capturing taller and larger mammals such as impala (*Aepyceros melampus*) (Figure 2.11). Cameras were continuously run 24 hours each day of deployment with identical sensitivity settings and other programmable features being consistent. The external protective casing of each camera was numbered for easy monitoring of their condition by ranch guards during their daily security routine, and all memory cards per camera were marked for identification to the camera they were assigned. Marking memory cards is a secondary backup to programming each camera to identify all images it takes. This identification number is seen on every image within the lower left frame of the photograph. Data were very quickly collected in the field by replacing camera memory cards with new ones and by utilizing a mobile external hard drive so that images from a memory card could be downloaded, the card could be reformatted, and then inserted back into the camera.

Once downloaded, all imagery from each camera was carefully organized into a system of partitioned folders per ranch and per habitat. Images that captured camera installation activity were removed, and the remaining collection of photos were run through Picture Information Extractor™ (P.I.E) (Picmeta Systems, 2014) to generate metadata exported into Microsoft Excel™ spreadsheets, which were later used for cataloging species per image. Each individual image was visually examined for its content. Image processing first began with separating images that had wildlife from those that did not. Furthermore, images with target wildlife were separated from those without. All camera images are watermarked through automated camera settings to display their assigned camera ID number and useful metadata such as date, time, lunar cycle, and temperature. Watermarking each image through programming the camera was vital during the data download process, in which the collection of memory cards in the field can often lead to confusion as to which camera it came images originally come from.

Figure 2.11. Installation of camera traps on *Acacia drepanolobium* trees in Laikipia, Kenya as part of instruction during a study abroad program (A) and as part of research and training with security staff on Segera Ranch (B).

There were a disproportionate number of images, sometimes greater than 50% per camera that did not contain wildlife. This is largely due to physically mounting cameras to *A. drepanolobium* trees that sway slightly in windy conditions which then will trigger the cameras to take photos without wildlife present. Some camera trapping studies have adopted a system of mounting cameras to manufactured metal poles that are driven into the ground for more direct and easy placement. Though there are benefits to adopting camera housing units in the field, such as exact placement of cameras, shading from sunlight, and general camera protection, there are also significant drawbacks to the bias it can cause in the wildlife image data. This method was considered but ultimately decided against in this study, since iron poles tend to carry scent of where they are stored (near humans) and have been observed to be deliberately manipulated by wildlife. Elephant are known to pull them out of the ground, drag them for long distances causing damage to the cameras and housing unit, and dropping them in locations difficult to locate. By using Acacia trees as a base, cameras were better camouflaged and secured to native and familiar structures. There are other instances where wildlife interact with the foreign object (the housing unit), which can lead to camera damage, a repositioning of the camera, impact to battery life, and complete loss of the camera. Cameras from an independent study on Ol Pejeta were installed immediately outside a striped hyena den to document the traffic in and out of the den. Though the effort was successful and generated a large number of images of striped hyena activity, the cameras were ultimately destroyed by the animals chewing on them. In a more recent case, a black rhino came upon a dual camera set up on Ol Pejeta, and was photo-documented attacking the housing unit (Figure 2.12).

Figure 2.12. Camera trapped image, courtesy of Ol Pejeta Conservancy, displaying a black rhino attacking a camera-trap unit (image captured by twin camera setup) that was positioned on a metal pole. In this image, the pole and camera housing unit are welded together, with the camera-trap housed in this unit being flung outward from the impact. This image captured an additionally rare event, which is to witness a black rhino with both front hooves off of the ground.

Cameras were positioned to face either north or south to avoid direct exposure to the rising and setting sun. Sunlight directly entering the lens of a camera trap will increase the camera's internal temperature and quickly decrease the life of the batteries. Camera traps were locked to trees by using Master Python™ adjustable trail camera cable locks to avoid removal or accidental damage by wildlife and to discourage theft by humans. Once fastened to a tree, cameras often required some finesse to level the cameras and tighten their grip on the tree by tucking either folded grass or twigs behind the camera body.

Cameras were additionally set at the wildlife corridors between the Ol Pejeta Conservancy and the ADC Mutara Ranch (see Figure 2.13). Photographic images were observed from most expected wildlife as well as many human image captures. Set cameras were continually investigated by local community members as they traveled

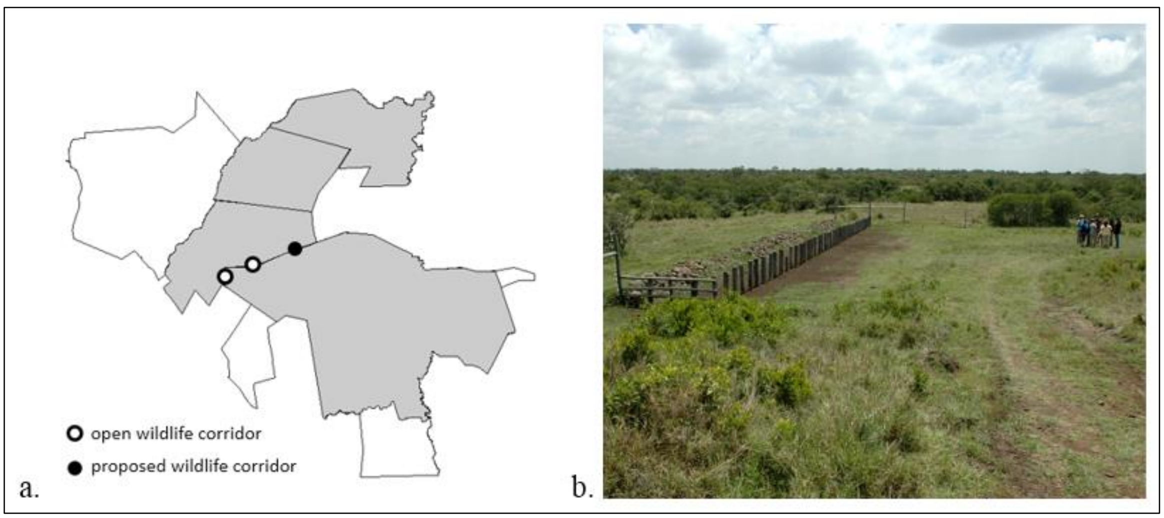


Figure 2.13. Locations of open and proposed wildlife corridors (a.) with image of the 50m gaps in electric fencing on the border of ADC Mutara Ranch and the Ol Pejeta Conservancy Ranches. Poles at 1m height and .5 m apart are used to prevent rhino access but allow movement of all other wildlife, including elephant.

across the landscape with their cattle. Both young and old members of the community would spend a considerable amount of time carefully examining the camera and often times prodding it with large sticks. Others in the community were made aware of the camera surveys through ranch guards, and found it entertaining to leave humorous self-images to be later discovered. Reviewing and recording imagery was labor intensive, given the high number of non-target images as a result of windblown grasses or when cameras were attached to small trees that moved during periods of high wind.

Abundance and diversity estimation

In large landscapes the requirements to estimate abundance can become expensive and logistically impossible. Portions of populations or targeting indicator species often become more appropriate and manageable. Similarly, it can be challenging for camera trapped data to capture abundance since the lack of detection is not necessarily an indicator that the species is absent (Bailey *et al.*, 2007). Recently there has been an increase in the number of studies that use occupancy as a surrogate for abundance (MacKenzie and Nichols, 2004; Tobler, *et al.*, 2015; Rovero *et al.*, 2014; Ahumada *et al.*, 2013; Burton *et al.*, 2012; Rovero and Marshall, 2009; Bowkette *et al.*, 2007), including those adopting a single season model (MacKenzie *et al.*, 2002; Tyre *et al.*, 2003; MacKenzie *et al.*, 2007). These studies have provided the guidance and statistical means to use the camera trapping rate, occupancy and or presence absence data as a reliable surrogate for abundance.

The software program EstimateS (Colwell, 2013) was used to analyze photographic data and derive species richness estimates based on the number of individuals captured and the frequency of capture.   These software programs allow for customization, depending on the research question (Silver *et al.*, 2004). Microsoft Excel™ (Microsoft, 2013) was used for standard statistical needs, such as t-tests, regression, correlations, and goodness of fit tests (chi-square). Camera trapping experts from SCBI were consulted prior to and post use operation.

To compute species richness values, I utilized EstimateS version 9 (Colwell, 2013), a free software application drawing on sampling data to assess and compare diversity and composition of species assemblages. One of the many benefits to using EstimateS is that it computes a variety of biodiversity statistics, including estimators of species richness (i.e. Chao 2) (Chao *et al.*, 2000), diversity indices, and will generate rarefaction and extrapolation values. Rarefaction and extrapolation values are important for visualizing the accumulation curve data as it reaches its asymptote. The rarefaction is a process within the EstimateS model of resampling randomly selected individuals (i.e. species), without replacement, in the data set until all samples have been referenced. The extrapolation feature allows one to visualize the expected richness values over a given sampling size beyond what the data maintain. In this case, I often extrapolated data with an average of 200 – 300 samples (individual animals or camera events) to a maximum value of 500. It can often be useful to extrapolate well beyond what might be necessary so as to visualize where the rarefaction curve meets the asymptote, and then revisit the extrapolation value to cut back on any excess.

To use camera trapping data for richness estimation, I converted grouped occurrences of a single species per camera to a single value of 1 or 0 (presence / absence) for all samples. These data are then uploaded to EstimateS as a sample-based incidence of one set of replicated data (camera replicates). Default settings, as recommended by the Smithsonian Conservation Biology Institute (SCBI, 2015), were used and included a maximum of 100 randomizations per calculation (without replacement), estimating every individual sample as opposed to evenly spaced points, selecting options to estimate Chao 2 richness values (Chao, 1984; Hortal, *et al.*, 2006), and to use the recommended upper limit for rare or infrequent species at a value of 10. Using these values allowed the final estimations for species richness to match the number of observed species from the field data. As a general measure of whether I sampled the community of wildlife species, I used the species accumulation curves with total camera trap days to determine if collection of data lasted long enough to capture the total number of species recorded on the landscape. In using EstimateS, statistical significance is inferred from a lack of overlap between accumulation curve confidence intervals (Colwell, 2012). Payton *et al.* (2003) and SCBI research staff (2015) explain that using an 84% confidence interval is ideal and has been developed to take into account an alpha of .05. The difference must be corrected from EstimateS output, which is a standard 95% CI. I recalculated all defaults to 84% prior to graphing data.

2.3 Results

Camera trapping effort and deployment

Camera trapping deployment resulted in a total of 154,669 images of which 31,611 contained wildlife and 27,266 contained targeted wildlife (Table 2.2). A total of 50 targeted species of wildlife, including humans, were recorded across the landscape (Figure 2.14; Appendix A). Various other species of birds, reptiles, and invertebrates were captured through imagery but were not considered in any analysis. Rodents were largely removed from the data, with the exception of one case referencing a high relative abundance of rodent images on Segera Ranch. Though there is a focus toward mammals in this study, two species of large birds, the kori bustard (*Arteotis kori*) and the ostrich (*Struthio camelus*) are included in diversity estimations due to their size and association in this wildlife community.

Table 2.2. Descriptive statistics representing sample size, camera deployment, and total species richness by guild and habitat.

Image sample size	Segera	Eland Downs	ADC Mutara	Ol Pejeta	OPC-W	OPC-E
Total images	36,123	41,018	37,466	40,062	10868	29,194
Total wildlife	3,136	3,348	3,294	21,833	7389	14444
Total target wildlife	2,139	1,220	2,721	21,186	7239	13947
	67%	36%	83%	97%	98%	97%
Total grouped wildlife*	291	278	379	903	386	517
Camera effort						
Cameras deployed	15	17	19	34	19	15
Camera nights	191	203	265	567	272	258
Total species (50)	28	30	32	36	27	31
Species per guild						
Grazer (7)	7	4	3	7	6	6
Grazer / Browser (9)	8	8	7	7	6	6
Browser (11)	10	8	8	9	4	9
Omnivore (4)	3	3	3	2	2	2
Carnivore (14)	14	12	8	10	8	7
other / insectivore (4)	4	4	3	3	2	2
Species per habitat						
Acacia (33)	9	15	19	19	8	14
Euclea (30	13	10	10	23	15	19
Grassland (32)	10	16	15	23	18	16
Mix (26)	14	13	8	17	13	10
Riverine (31)	17	10	19	19	17	15

* Grouped wildlife are the total number of individual members of a single species recorded by multiple camera events (i.e. herd of herbivores passing in front of a camera).

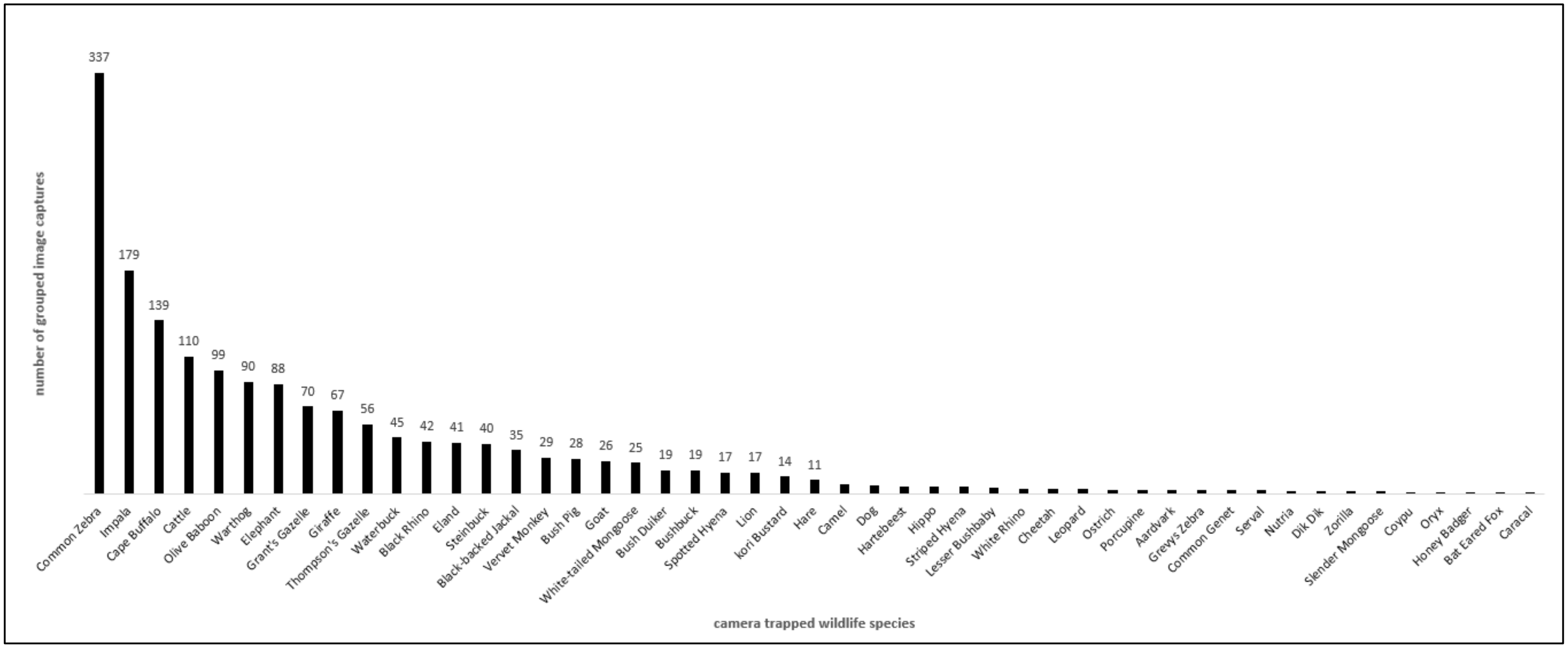


Figure 2.14. Number of grouped image captures for each species recorded on landscape.

There are species in the dataset that appear to be unique to each ranch, but are very well known to occur across the entire landscape. I have also observed these species where they were not found in camera-trapping efforts. One of the more distinct differences between two ranches occurred on the Ol Pejeta Conservancy. For purposes of history, geographic features (hydrology), and management, the Ol Pejeta conservancy was sampled as two independent ranches (East and West). There was a suspicion that the differences between these two areas might reveal important results on species diversity and abundance. Ol Pejeta West uniquely contained the kori bustard, caracal (*Caracal caracal*), leopard (*Panthera pardus*), slender mongoose (*Helogale parvula*), porcupine (*Hystrix cristata*), and domestic dog (*Canis lupus familiaris*). Of these species, the kori bustard is more prone to occupy large open grasslands and open Acacia forest, which are more plentiful on the western side of Ol Pejeta. Additionally, the domestic dog is likely the result of guard ownership near the OPC and ADC border. Ol Pejeta East uniquely contained the zorilla (*Ictonyx striatus*), lion (*Panthera leo*), coypu (*Myocastor coypus*), oryx (*Oryx beisa*), Grant's gazelle (*Nanger granti*), Guenther's dikdik (*Madoqua kirkii*), Jackson's hartebeest (*Alcephalus buselaphus jacksoni*), the scrub hare (*Lepus saxitilis*), the (*Sylvicapra grimmia*), and the lesser bushbaby (*Galago senegalensis*). The unique wetland ecosystem on Ol Pejeta East explains the presence of the nutria (coypu), but what is of interest is the complete lack of Grant's gazelles on Ol Pejeta West. There are few Jackson's hartebeest inside the Ol Pejeta conservancy (< 19) and they have remained in one large herd on OPC East. All other unique species are commonly observed on both sides of the conservancy, and are likely ranked due to their size and solitary nature.

Six species of grouped wildlife were detected > 90 times; the common zebra (337), impala (179), Cape buffalo (139), cattle (110), the olive baboon (99), and the warthog (90). There were five rare species which had only one recorded event each; coypu, oryx, honey badger, bat-eared fox, and the caracal. There were no significant differences between total species richness per ranch.

Vegetation surveys

Vegetation surveys were conducted to estimate the general condition of the habitat at the locations of each camera trap prior to installation. Three surveys were conducted per camera trap location for a total of 45 surveys per ranch (9 per habitat) or a grand total of 180 samples from Ol Pejeta to Eland Downs (Appendix II). Segera Ranch was not sampled for vegetation consistently, as estimates were taken at the time of camera installation at only 1 sample per location, which was a result of logistics and time availability in the season. The result of all surveys, combined with prior data, ground-truthing, and satellite imagery overlay corrections, produced a comprehensive habitat classification map of all four ranches (Figure 2.15).

Land cover habitat across all ranches is dominated by Acacia (39%), followed by Grassland (25%), Mixed (21%), Euclea (12%), and Riverine forest (3%) (Table 2.3). Stratified patterns of vegetation can be observed parallel to river basins, such as in Ol Pejeta East and on Eland Downs and ADC Mutara with more prominent mosaic patterns in larger areas such as Ol Pejeta West and Segera. ADC Mutara and Eland Downs are dominated by dense Mixed habitat at coverage of 43% and 29% respectively, while Ol

Pejeta and Segera are dominated by Acacia at 39% and 60% respectively. Sudden breaks in habitat connectivity occur on some ranch boundaries, such as on the border of Eland Downs and ADC Mutara, abruptly changing from Acacia to Mixed habitat, as well as small portions of the border between ADC Mutara and Ol Pejeta West. These are likely the result of the temporal influence of fencing on native and non-native herbivore foraging and movement. It is also important to note that most of these boundaries between ranches contain some form of service road which would also influence habitat connectivity and composition.

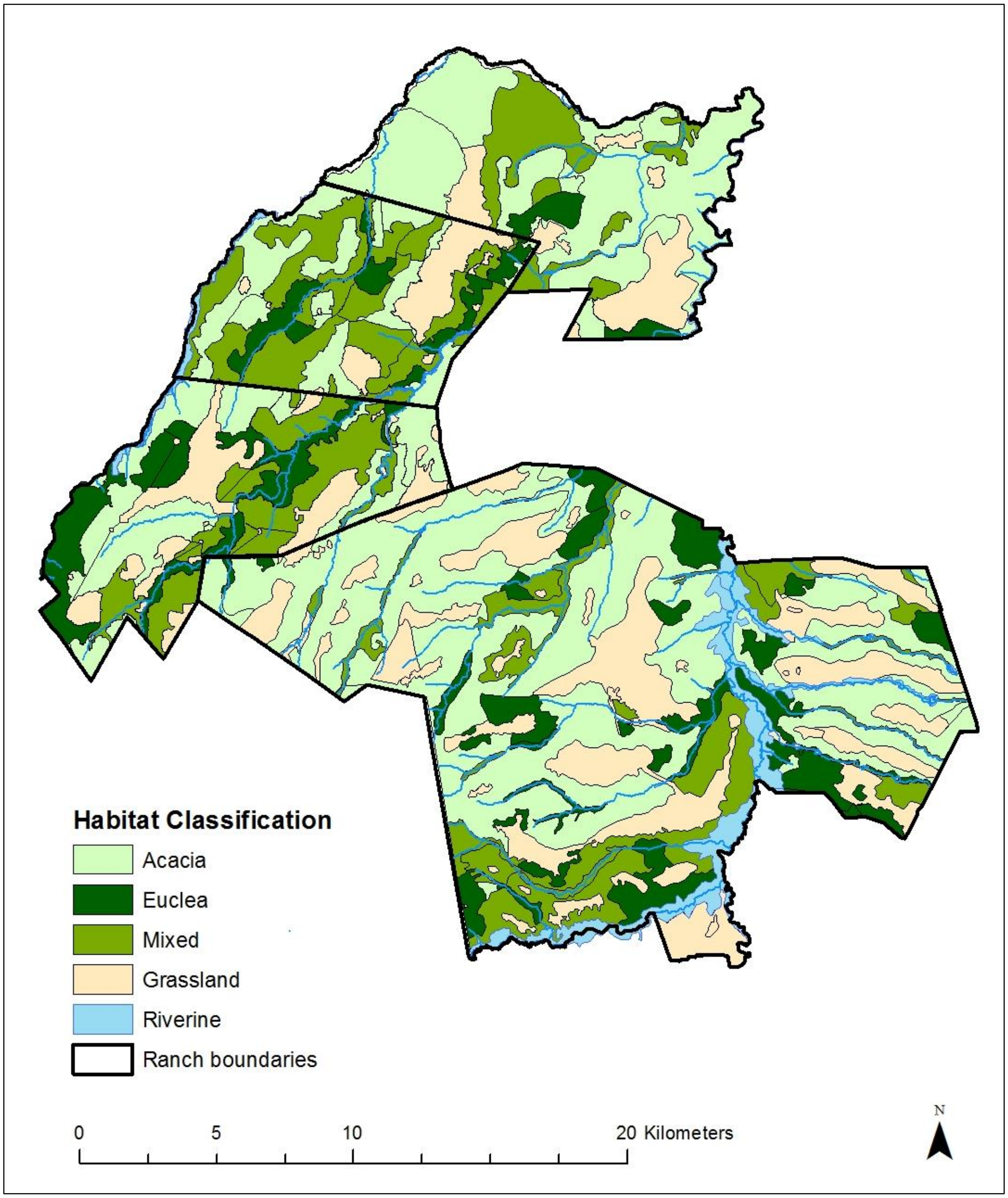


Figure 2.15. Habitat classification map of study area featuring five generalized habitat types.

Table 2.3. Summar of habitat acreage per ranch.

Ranch	Total Area Sampled (acres)	Habitat Classification					Total Area Sampled (Km^2)
		Acacia	**Euclea**	**Grassland**	**Mixed**	**Riverine**	
Segera	22,000.00	13,317.00	908.00	3,193.00	3,888.00	694.00	89.0
% of total ranch		60.53	4.13	14.51	17.67	3.15	
Eland Downs	17,500.00	5,110.00	2,132.00	2,186.00	7,594.00	478.00	70.8
% of total ranch		29.20	12.18	12.49	43.39	2.73	
ADC Mutara	20,000.00	4,115.00	4,501.00	5,350.00	5,798.00	236.00	80.9
% of total ranch		20.58	22.51	26.75	28.99	1.18	
Ol Pejeta West	55,000.00	22,206.00	5,924.00	16,928.00	8,083.00	1,859.00	222.6
% of total ranch		40.37	10.77	30.78	14.70	3.38	
Ol Pejeta SW	15,000.00	5,743.00	2,208.00	4,435.00	1,738.00	876.00	60.7
% of total ranch		38.29	14.72	29.57	11.59	5.84	
OPC total	70,100.00	27,989.37	8,142.77	21,393.78	9,835.70	2,738.38	283.7
% of total ranch		39.93	11.62	30.52	14.03	3.91	
Total habitat	129,500.00	**50,491.00**	**15,673.00**	**32,092.00**	**27,101.00**	**4,143.00**	524.1
% coverage		38.99	12.10	24.78	20.93	3.20	

Averages are fairly uniform across the landscape, with the exception of several observations for the two more important habitat classifications for large mammals (Acacia and grasslands). First, the proportional percent of *Acacia drepanolobium* trees on Segera ranch is higher than all other ranches. This might explain its high elephant visitation. There is also a shift in the dominant tree species in riverine environments from ADC Mutara to Ol Pejeta, which is dominated by *Euclea divinorum* and transitions to *Acacia xanthophloea* upon reaching Ol Pejeta. The large *A. xanthophloea* trees on Ol

Pejeta are indicative of the constant availability of water, and explains the very high relative abundance of olive baboons (25.58) which nest in the trees at night, compared to other ranches.

Considering the sample bias, grass height on Segera ranch was significantly higher than all other ranches. Speculation from the local community indicates this is due to a lower density of native grazing ungulates. In addition, it has also been suggested that this grass is older and of lower nutritional value and therefore avoided. One solution for removing poor nutrient quality grass is to consider hyper-grazing the area with cattle (Kinnaird, 2012). The dominant grass species across the landscape is consistently *Themida triandra*, with more frequent encounters of *Pennisetum sp.* on ADC Mutara and Ol Pejeta (Table 2.4).

Table 2.4. Summary of estimates for average percent cover and height of grass and trees.

		grass	tree
Segera	% cover	57.624	17.6
	height	52.662	362.6
Eland Downs	% cover	48.89	14.4
	height	36.256	345.4
ADC Mutara	% cover	58.34	25
	height	36.82	461.6
OPC W	% cover	50.366	26.8
	height	27.802	452.2
OPC E	% cover	52.86	34.8
	height	40.576	798.6

Diversity of large mammals

The results of EstimateS diversity index for Chao2 were calculated from all ranches combined (Figure 2.16) at 47.17 species from a total of 1,818 individuals. The rarefaction mean was 47 (0.62 SD) with an 84% confidence interval. The accumulation curve reached it asymptote prior to extrapolation and shows the conservation areas sampled in this study had been sampled sufficiently.

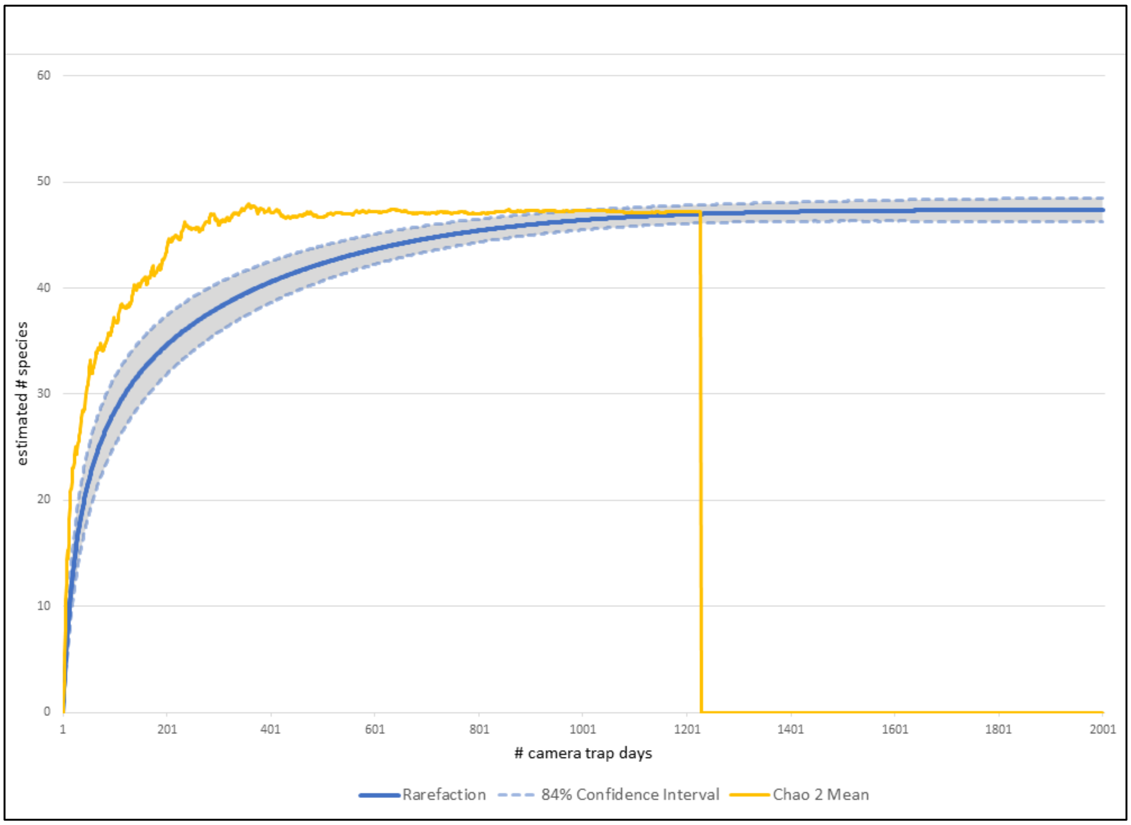


Figure 2.16. Species accumulation curve (blue) for target species detected by camera-trapping from combined ranches featuring a 84% confidence interval (grey). The Chao2 diversity index (orange) estimate is shown with data extrapolation occurring at camera trap day 1226.

Camera traps had detected nearly all wildlife species recorded on the landscape (n = 49) with the exception of rare species eliminated for low abundance in the EstimateS model. Species accumulation curves per ranch indicate that only two of the 4 ranches in this landscape were significantly different from one another (OPC combined and Eland Downs). Though there are high levels of diversity on all ranches, EstimateS modeling includes species with detection above 10 events per species during the trapping period. An overlap of confidence intervals indicates no significant difference, with the exception of Ol Pejeta and Eland Downs (Figure 2.17, 2.18).

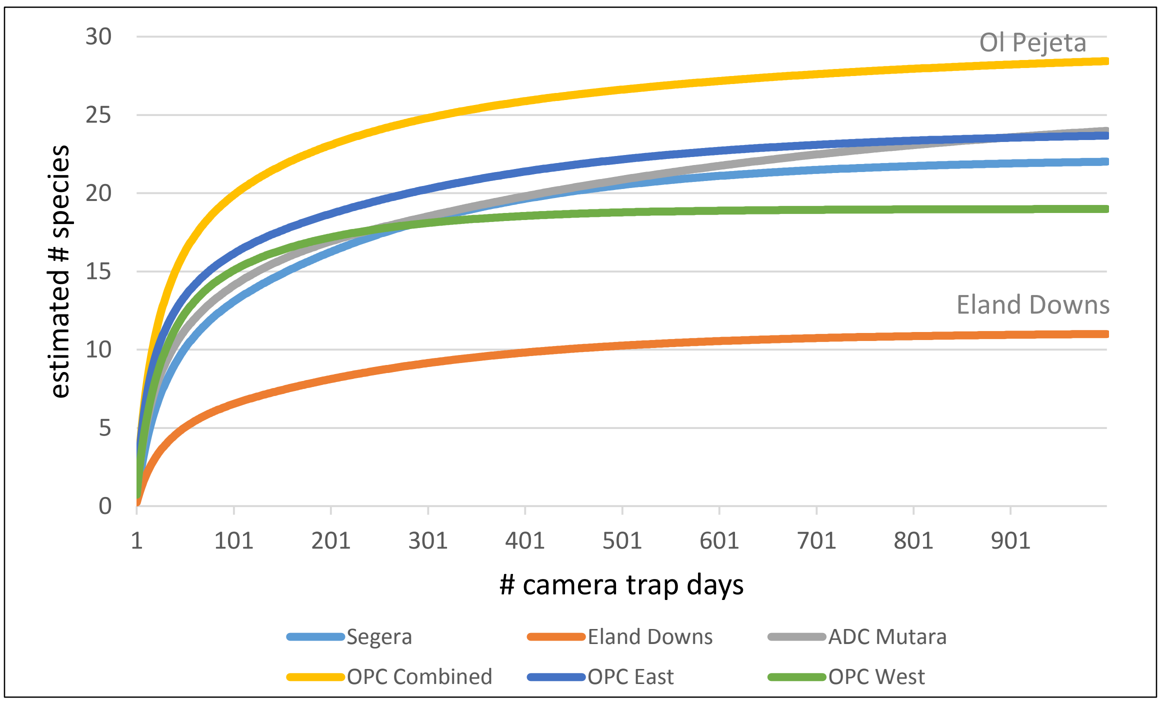


Figure 2.17. Species accumulation curves per ranch.

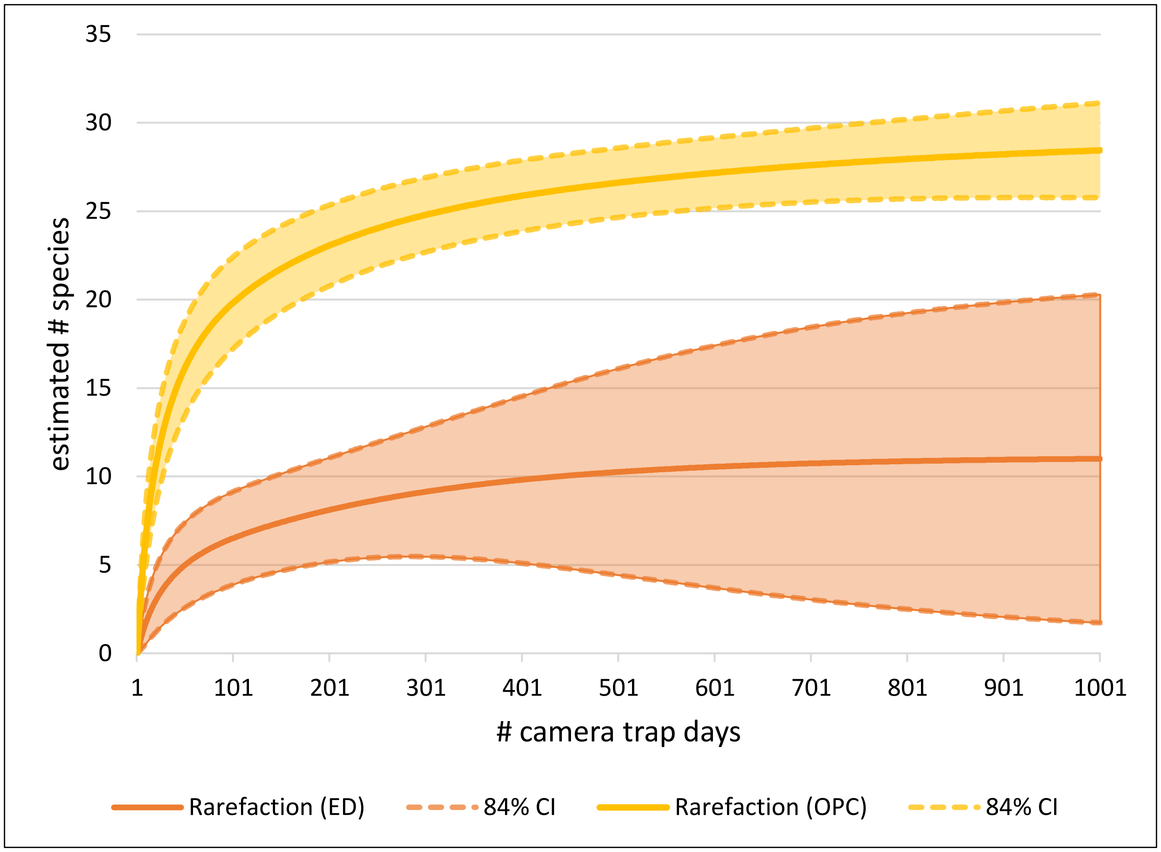


Figure 2.18. Significantly different, non-overlapping confidence intervals (α .05; 84%) for the Ol Pejeta Conservancy (yellow) and Eland Downs (orange).

This result supports the hypothesis that Eland Downs was significantly lower in diversity than the combined Ol Pejeta Conservancy, but was not significantly different from any other ranches, including the independent halves of Ol Pejeta. Eland Downs has a similar total richness value (30) to other ranches, but the frequency of visitation per species was considerably low in comparison.

Due to the sample size, confidence intervals among all accumulation curves for species richness per habitat were not significantly different, though they appear so graphically (Figure 2.19). Diversity among ranches was also compared using a general linear model (PROC GLM; SAS Institute 2011). I used a square-root transformation on the data to meet assumptions of normality prior to analysis, but the mean and standard error of the mean (SEM) presented in the results are of non-transformed data. Diversity differed among ranches ($F = 8.15$, df = 4, $P < 0.001$) primarily because diversity was relatively low at ED and higher at all other ranches (Table 2.5).

Table 2.5. Mean (SEM) diversity across five ranches in Kenya. Means with different letters are significantly different (Tukey's means separation test, $P < 0.05$).

Ranch	**Mean (SEM)**
Segera	5.2 (0.6)a
Eland Downs	3.0 (0.6)b
OPC East	6.8 (0.5)a
OPC West	6.3 (0.5)a
ADC Mutara	6.4 (0.6)a

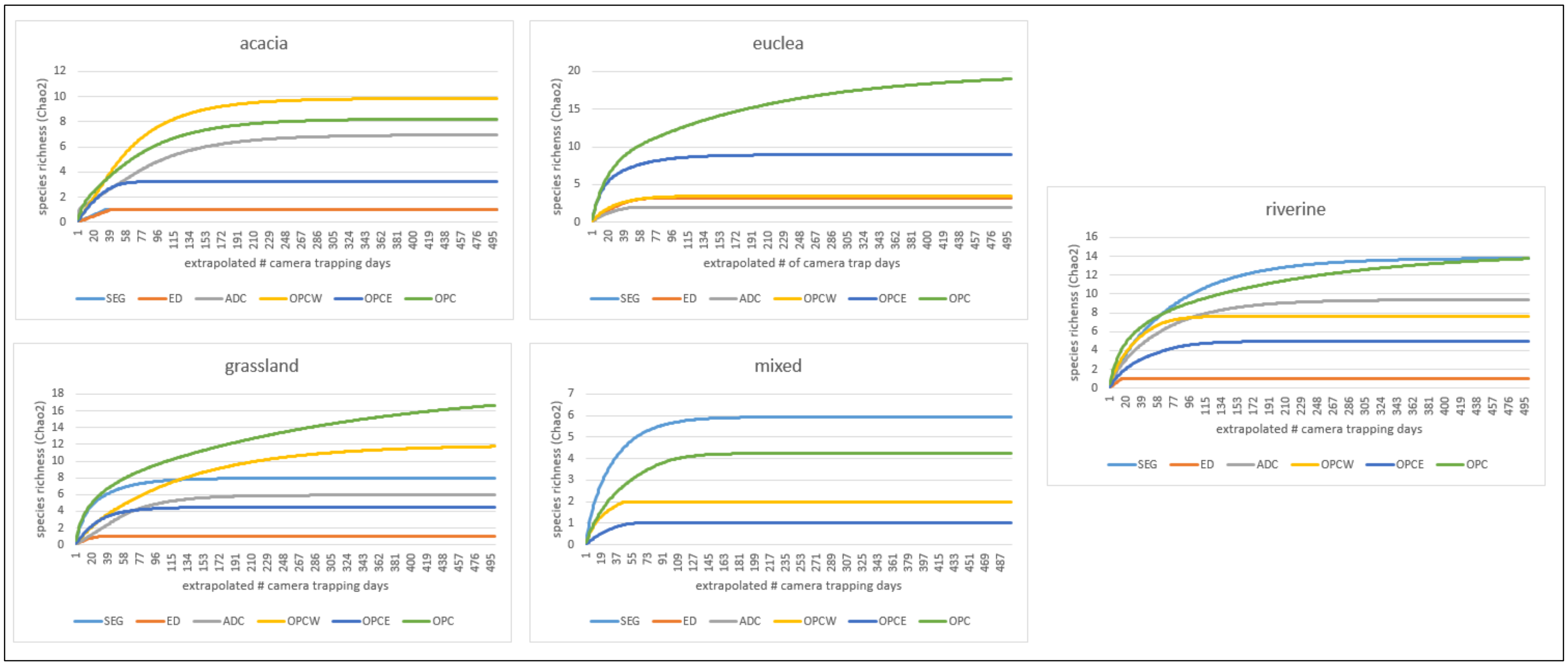


Figure 2.19. Habitat specific species accumulation curves per ranch.

Habitat preference, abundance and distribution

The majority of grouped species image captures were recorded in riverine habitat (n=484; 26%), followed by grassland (n=386; 21%), Acacia (n=345; 19%), mix (n=320; 17%), and Euclea (n=316; 17%) (Table 2.6). These estimates should not be misinterpreted from the actual sample of images taken, which are influenced by the effect of wind, herd size of animals passing through, etc. With regard to images by ranch, the majority of grouped species image captures were highest on OPC East (517; 28%), followed by OPC West (386; 21%), ADC Mutara (379; 21%), and both Segera and Eland Downs at (278; 15%).

There was a rapid accumulation of species detected through camera trap sampling in the first 200 days with a majority having been detected by 400 days (>40 species, roughly 80%). The camera trapping rate was defined as the ratio of samples collected independently to the number of camera trap days multiplied by 100. This is also identified as the relative abundance index (RAI). Camera trap days is defined as a 24 hour period during which cameras are in the field and operating until they are retrieved from service. To determine camera trapped events as being independent, 30-minute intervals were assigned to same species captures (modified from Bowkett *et al.*, 2007), as many species, such as the Cape buffalo and impala, forage in large herds and often in widespread patterns. Consecutive photos of the same species are not considered independent and were therefore grouped into a single species event. Each grouped species event is considered independent for that camera. The date and time of the first camera capture was assigned to each group event.

Table 2.6. Summary of relative abundances per 100 camera trap nights for the top 20 species per ranch.

Segera		Eland Downs		ADC Mutara		Ol Pejeta West		Ol Pejeta East		OPC cumulative	
Cattle	29.41	Common zebra	26.60	Common zebra	42.26	Common zebra	36.40	Impala	36.43	Common zebra	27.92
Human	19.79	Human	21.81	Buffalo	13.58	Impala	13.97	Olive baboon	25.58	Impala	24.91
Elephant	17.65	Elephant	14.89	Impala	9.43	Buffalo	12.13	Buffalo	25.19	Buffalo	18.49
Common zebra	17.11	Goat	12.77	Cattle	7.92	Cattle	10.66	Common zebra	18.99	Olive baboon	14.91
Rodent	8.02	Shoats	8.51	Reticulated giraffe	6.79	Warthog	10.29	Warthog	14.34	Warthog	12.26
Impala	7.49	Thompson's Gazelle	6.91	Grant's Gazelle	6.04	Thompson's Gazelle	9.93	Grant's Gazelle	13.95	Black Rhino	7.92
Vervet Monkey	6.42	Grant's gazelle	5.85	Human	6.04	Eland	5.88	Waterbuck	13.95	Waterbuck	7.74
Shoats	5.88	Steinbuck	5.32	Steinbuck	6.04	Reticulated giraffe	5.51	Black Rhino	12.02	Grant's Gazelle	6.79
Grant's Gazelle	5.35	Jackal	4.79	Olive baboon	5.66	Elephant	5.15	Reticulated giraffe	8.14	Reticulated giraffe	6.79
Steinbuck	5.35	Reticulated giraffe	4.79	Vervet Monkey	5.28	Bushbuck	4.78	Lion	5.81	Thompson's Gazelle	6.04
Eland	4.28	Impala	4.26	Warthog	5.28	Olive baboon	4.78	Jackal	3.49	Cattle	5.66
Warthog	4.28	Camel	3.72	Thompson's Gazelle	4.15	Black Rhino	4.04	Eland	3.10	Eland	4.53
Dog	3.21	Bush duiker	3.19	Eland	3.40	Jackal	3.68	White-tailed Mongoose	3.10	Elephant	3.96
Buffalo	2.67	Kori Bustard	3.19	Bush Duiker	3.02	White-tailed Mongoose	2.94	Elephant	2.71	Jackal	3.58
Jackal	2.67	Spotted Hyena	3.19	Elephant	2.64	Hippopatamus	1.84	Jackson's Hartebeest	2.33	White-tailed Mongoose	3.02
Reticulated giraffe	2.67	White-tailed Mongoose	2.66	Bush Baby	1.89	Waterbuck	1.84	Thompson's Gazelle	1.94	Bushbuck	2.83
Spotted Hyena	2.14	Cattle	2.13	Kori Bustard	1.89	Spotted Hyena	1.47	Cheetah	1.55	Lion	2.83
Bush Duiker	1.60	Olive baboon	2.13	Scrub Hare	1.89	Porcupine	1.10	Bush Duiker	0.78	Hippopatamus	1.13
Grevy's zebra	1.07	Warthog	2.13	Bushbuck	1.51	White Rhino	1.10	Bushbuck	0.78	Jackson's Hartebeest	1.13
Kori Bustard	1.07	Vervet Monkey	1.60	Striped Hyena	1.51	Genet	0.74	Human	0.78	Spotted Hyena	1.13

The highest abundance rate for all guilds was grazer (n=52), followed by grazer/browser (n=38), browser (n=29), omnivore (n=25), carnivore (n=11), and other (n=11) (Figure 2.20).

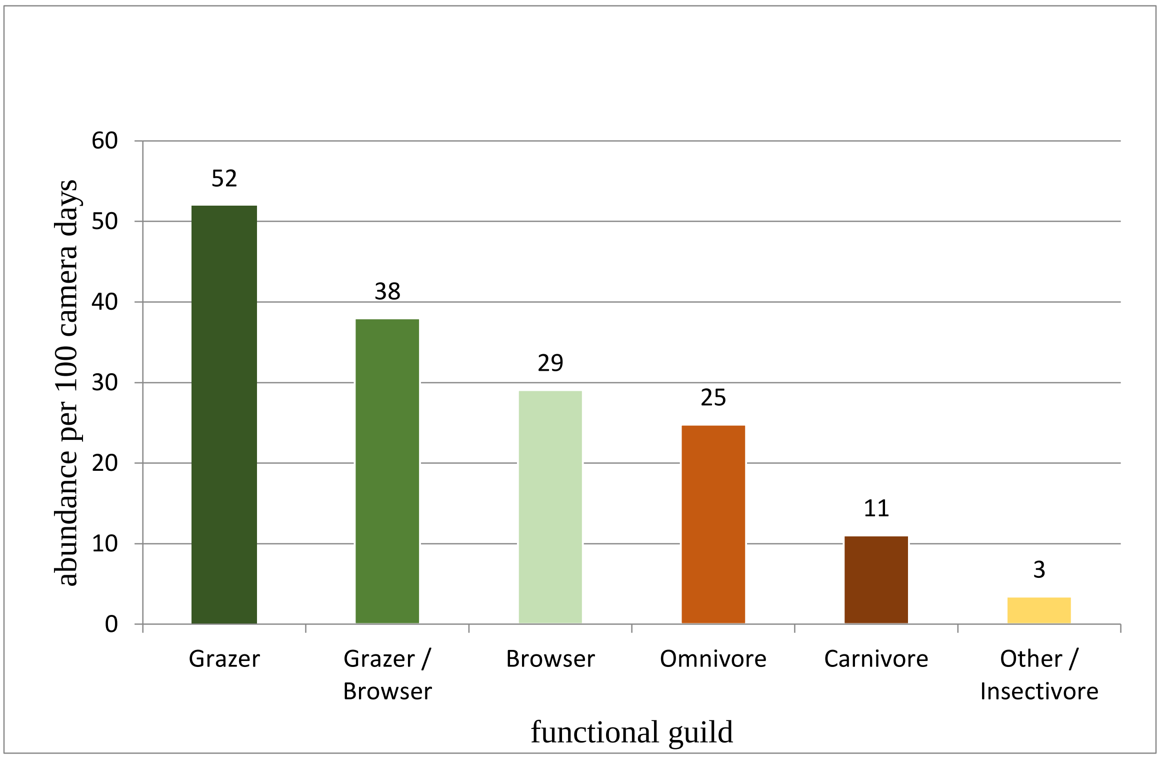


Figure 2.20. Relative abundances of camera detections per guild for combined ranches.

Non-parametric tests using Chi square analysis and total observed values were performed for each guild per habitat (Table 2.7). An overall Chi-square analysis reveals preference for habitat among all guilds (X^2 = 249.675; df = 20; p-value <.005), and each habitat was tested independently, also revealing preference for habitat per guild (Table

2.8) consistently with p-values < .005. This can further be visualized through relative abundance per guild in habitat seen in Figure 2.21. Riverine habitat is unique in having similar abundance values per guild, with the exception of carnivores and browsers.

Table 2.7. Summary of Chi-square values for preference of habitat by guild.

	Acacia	Euclea	Grassland	Mixed	Riverine
Chi-square	224.27	185.608	182.85	188.838	177.091
P-value	< .05	< .05	< .05	< .05	< .05
degrees of freedom	5	5	5	5	5

Table 2.8. Summary of estimates of camera detections and relative abundances per guild by habitat classification and per ranch.

	Guild	Habita Classification				
		Acacia	Euclea	Grassland	Mix	Riverine
detection	Grazer	148	90	127	120	123
	Browser	80	18	117	43	82
	Grazer/Browser	44	124	69	92	115
	Carnivore	16	27	33	20	33
	Omnivore	46	40	28	45	131
	Other	11	17	12	0	0
	TOTAL	345	316	386	320	484
relative abundance	Grazer	64.35	39.30	53.36	52.86	50.00
	Browser	34.78	7.86	49.16	18.94	33.33
	Grazer/Browser	19.13	54.15	28.99	40.53	46.75
	Carnivore	6.96	11.79	13.87	8.81	13.41
	Omnivore	20.00	17.47	11.76	19.82	53.25
	Other	4.78	7.42	5.04	0.00	0.00

	Guild	Ranch					
		SEG	ED	ADC	OCPW	OPCE	OPC
detection	Grazer	94	58	169	170	117	287
	Browser	31	45	69	58	137	195
	Grazer/Browser	79	93	71	86	115	201
	Carnivore	22	25	14	27	41	68
	Omnivore	48	49	45	43	105	148
	Other	4	8	11	2	2	4
	TOTAL	278	278	379	386	517	903
relative abundance	Grazer	50.27	30.85	63.77	62.50	45.35	54.15
	Browser	16.58	23.94	26.04	21.32	53.10	36.79
	Grazer/Browser	42.25	49.47	26.79	31.62	44.57	37.92
	Carnivore	11.76	13.30	5.28	9.93	15.89	12.83
	Omnivore	25.67	26.06	16.98	15.81	40.70	27.92
	Other	2.14	4.26	4.15	0.74	0.78	0.75

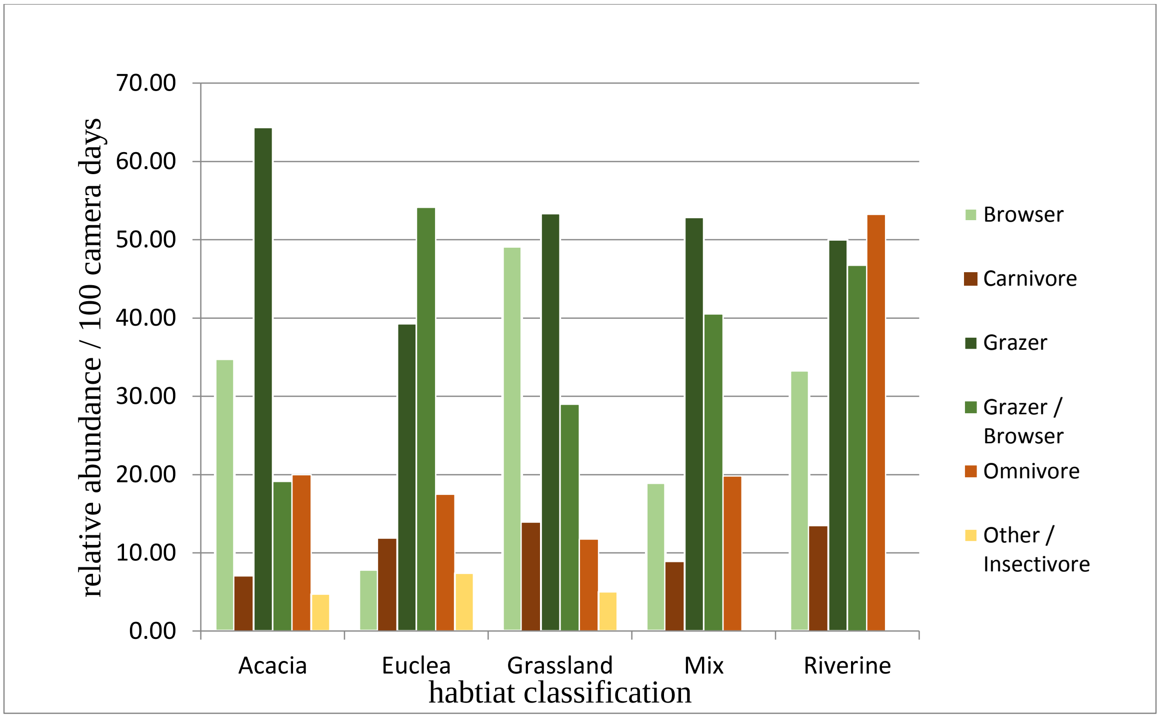


Figure 2.21. Relative abundances of camera detections per guild by habitat classification.

Trophic guild habitat preference.

For modeling occupancy of browsers, the model with the greatest support was a negative correlation with Euclea habitat (Coefficient -3.035, ± 0.9919). The highest ranking model for grazers was proximity to nearest river + proximity to nearest road + proximity to nearest human activity. For carnivores, the highest ranking model was proximity to riverine habitat (25.0426 ± 125953.99; insignificant). Though insignificant, it is the highest ranking model with an unknown predictor variable tied to occupancy of carnivores (likely prey abundance) (Table 2.9).

Table 2.9. Best fit AIC models that explain differences in detection and occupancy across the landscape. Five explanatory variables are usd in the models: proximity to nearest bore hole, proximity to nearest human activity, proximity to nearest river, proximity to nearest road, and cattle as predictor variables.

Model variables	K	AIC	ΔAIC	w_i	model liklihood
Occupancy modeling for detection					
Browsers					
Ψ (.) p (effort)	3	354.8	0.00	0.6098	1.0000
Ψ (.) p (effort + camera)	4	356.8	2.00	0.2243	0.3679
Ψ (.) p (.)	2	358	3.23	0.1213	0.1989
Ψ (.) p (camera)	3	360	5.23	0.0446	0.0732
Carnivores					
Ψ (.) p (.)	2	318	0.00	0.5091	1.0000
Ψ (.) p (effort)	3	319.7	1.66	0.2220	0.4360
Ψ (.) p (camera)	3	320.2.00	2.00	0.1873	0.3679
Ψ (.) p (camera + effort)	4	321.7	3.66	0.0817	0.1604
Grazers					
Ψ (.) p (effort)	3	316.2	0.00	0.7288	1.0000
Ψ (.) p (effort + camera)	4	318.2	2.00	0.2681	0.3679
Ψ (.) p (.)	2	327.8	11.58	0.0022	0.0031
Ψ (.) p (camera)	3	329.8	13.58	0.0008	0.0011
Occupancy modeling for guild					
Browsers					
Ψ (Euclea) p (effort)	3	342.4	0.00	0.6098	1.0000
Ψ (Grassland) p (effort)	4	351.5	9.18	0.2243	0.3679
Ψ (.) p (effort)	2	354.8	12.41	0.1213	0.1989
Carnivores					
Ψ (Riverine) p (.)	2	313.1	0.00	0.5091	1.0000
Ψ (Riverine + Acacia) p (.)	3	313.9	0.82	0.2220	0.4360
Ψ (Acacia) p (.)	3	316.7	3.62	0.1873	0.3679
Ψ (Bore hole) p (.)	4	317.1	3.97	0.0817	0.1604
Grazers					
Ψ (River + Road + Human) p (effort)	3	303.5	0.00	0.7288	1.0000
Ψ (River + Road) p (effort)	4	306.9	3.32	0.2681	0.3679
Ψ (River + Human) p(*Effort*)	2	310.1	6.59	0.0022	0.0031
Ψ (.) p (camera)	3	313.3	9.79	0.0008	0.0011

Occupancy modeling by trophic guild

I derived the dataset for the large mammal community by first filtering all image records per guild. Three particular trophic guilds were isolated for further analysis: grazers, browsers, and carnivores. Remaining identified guilds, “grazer-browser” and “other” (insectivores), were discarded as their representative data were more ubiquitous across habitats and throughout the landscape. I then derived a set of spatial environmental covariates that would most likely explain the presence or absence of guild. I calculated the following response variables to test against trophic guild detection probability: (1) proximity to the nearest bore hole and/or dam, (2) proximity to the nearest large river, (3) proximity to the nearest human activity area, (4) cattle herd sizes found on ranches, (5) condition of fencing, and (6) habitat. Fencing condition and cattle were found to be strongly correlated and were both represented numerically in the model through cattle herd size. The distance to the nearest covariate was measured using two methods, ArcGIS and Google Earth Pro. Google Earth Pro resulted in higher value data as the base imagery was more updated and of higher resolution than the standard Esri imagery with ArcGIS. This allowed the ability to pinpoint small, structural features on a landscape, such as bore holes (Figure 2.22).

Figure 2.22. Google Earth 2015 satellite imagery featuring a bore hole at a cross section of roads (dark dot on lower right of image), with old boma sites (faded lighter colored circles in center and center right of image) on the Ol Pejeta Conservancy.

Akaike Information Criterion (AIC) was used to rank all the candidate models and their Akaike weights (w_i), I used occupancy as the trophic guild-specific state variable of abundance to assess differences between selected trophic guilds and to determine the covariates of both occupancy and detection probability for these guilds. I modelled both estimated occupancy (Ψ) and detection probability (p) with and without the covariates.

Six models were developed to approach correlations of detection and occupancy by the response variable, guild. Encounter histories were created for analysis in the program PRESENCE by collapsing daily camera trapping records into 5-6 day blocks to establish a single encounter occasion, with a total of five encounter occasions for analysis (MacKenzie *et al.* 2006, Kelly *et al.* 2008). Site predictor variables of habitat (covariates) were deemed important for occupancy and were added to the modeling: Acacia, Euclea, Mix, Grassland, and Riverine. To eliminate the possibility of camera type bias, camera type (Reconyx vs. Spypoint) was also added to the modeling efforts. Probability of detection was derived from the encounter occasions of each species. If a species was detected in an encounter occasion, the probability of detecting the same species in a subsequent event was represented as 1. And additional survey covariate "effort" was the number of days pooled into each encounter occasion to produce an effort of surveying for that encounter history.

I used a single season, two species co-occurrence algorithm in program PRESENCE v.6.1 to estimate the probability of each species' occupancy (Ψ) and the detection probability (p) (Rovero *et al.*, 2014). The detection probability is the ratio of how more or less likely the species are to co-occur at a site. A detection probability value of 1 implies the species co-occur at a site independent of each other. A value less than one implies the species are less likely to occur at the site than if distributed independently (avoidance) and a value greater than one implies the species are more likely to occur at the site than if distributed independently (co-occurrence) (MacKenzie *et al.* 2004, 2006).

Due to the abundance of candidate models for AIC, each variable was first individually inserted into the model. The individual models within 2 delta AIC of the top model were ran with all possible combinations, and the resulting top model was selected for interpretation.

2.4 Discussion

Camera trapping has been found to be both cost effective and reliable (Rovero and Marshall, 2009) and was the chosen method to assess animal abundance through Laikipia's difficult terrain, remoteness, and where a variety of alternative methods were unfeasible. This method has also been demonstrated in recent literature that help to explain species richness and the diversity, distribution, and abundance of wildlife across landscapes affected by humans (Tobler *et al.*, 2008; Ahumada *et al.*, 2011; Kinnaird and O'Brien, 2012; and Tobler *et al.* 2015). I am confident in the use of camera trapping for these purposes. In the absence of typical line transect surveys to estimate densities of large mammals, I adopted camera trapping rates as an index of abundance (Rovero and Marshall, 2009). These data were necessary in order to develop general linear modeling to determine habitat preference for trophic guilds as well as associations of species diversity affected by human made structures. Statistical results from effort and number of camera deployments demonstrated that the study areas was sufficiently sampled (Figure 2.16).

General observations were first made in the field prior to formulating hypotheses. Including that most species were actually found throughout each habitat and among all ranches. When examining this through the data, there was no significant difference of species richness between properties in the study or among habitats. That both the field observations and camera trapping efforts are complimentary could be explained by constant animal movement between one habitat and another, which would not be reflective of habitat preference. The close proximity and mosaic of habitats delineated in the study could be a factor to explain the lack of difference in diversity.

When using occupancy as a surrogate for abundance, there were significant differences suggesting preference for two of three guilds analyzed (grazers and browsers). Grazers dominated Acacia habitat with both grazers and browsers dominating grasslands. These two habitats are almost always adjacent to one another and share a gradient through their transition. (Figure 2.23). I suspect the significant absence of browsers in Euclea habitat expressed in both chi-square and GLM modeling are a result of animal behavior to avoid areas of poor visibility and where the threat of predation is more likely (Fischoff *et al.*, 2007).

For ecotourism purposes, the most valuable habitats are Acacia and grasslands that attract large herbivores, particularly the big five (elephant, buffalo, rhino, lion, and leopard). These are also habitat classifications that make spotting wildlife relatively easy and are often ones to support low-maintenance infrastructure. Ol Pejeta and Segera Ranches are the only two of the four that contain a majority of habitat in Acacia and grassland with significant infrastructure, with ADC and Eland Downs serving future roles as highly diverse habitat with higher percentages of Mixed and Euclea classification supportive of wildlife corridor structure.

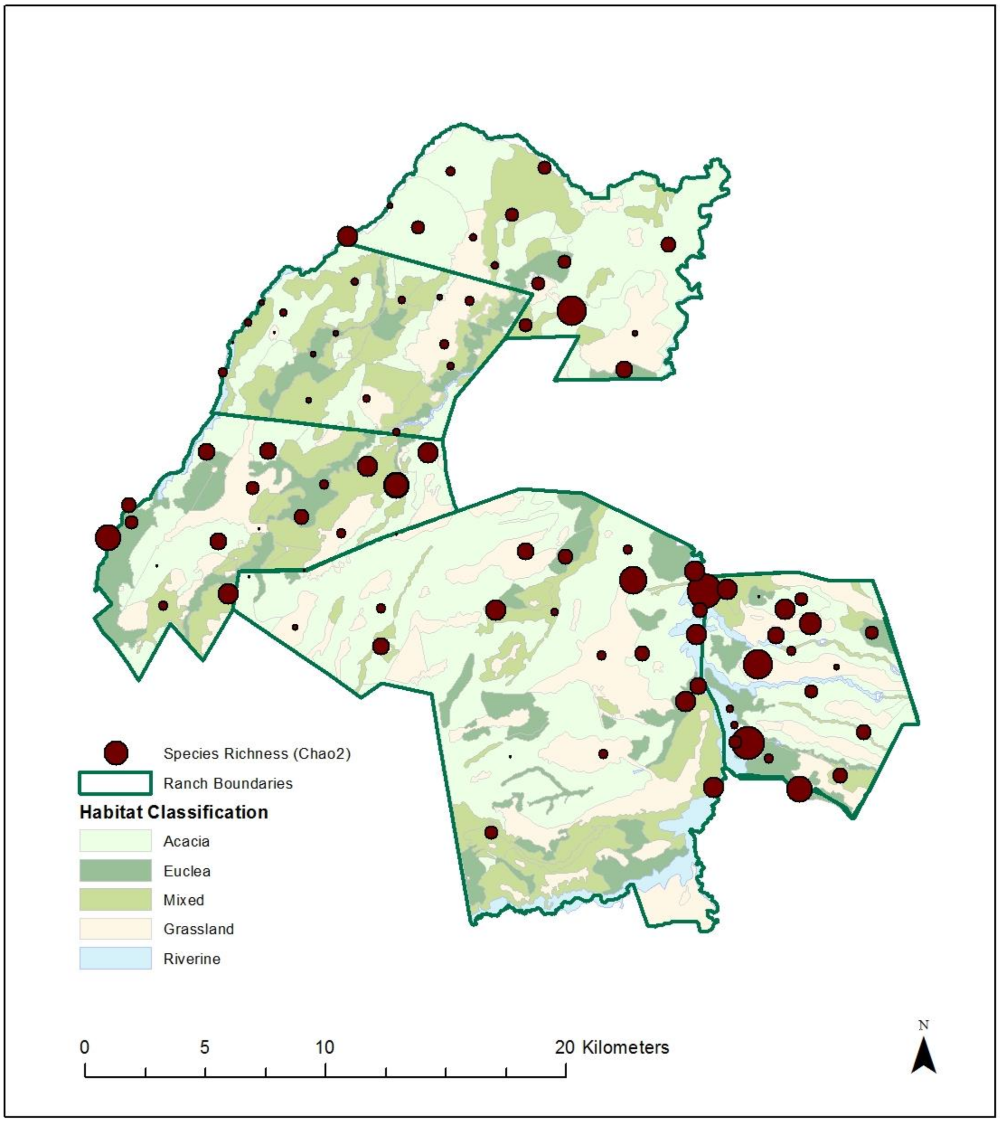


Figure 2.23. Species richness (Chao2) values per individual camera (n=83). Values range in size between 1 (smallest) through 18 (largest), with values at zero not displayed.

Mapping diversity across the entire landscape shows a congregation of higher Chao 2 values in close proximity to the Ewaso Nyiro River bisecting the Ol Pejeta conservancy. AIC models ranked natural rivers higher than artificial waterpoints when explaining species richness, which follows prior research from de Leeuw *et al.* (2001). ADC Mutara appears to maintain a more uniform spread of richness values, mostly among its Euclea and Mixed habitats indicative of water drainage. The topography on ADC is more diverse with various drainage basins and dried river beds developed through extreme rainfall events. Future conservation planning from connectivity of ADC to Ol Pejeta will compliment wildlife on either side given the high quality of Acacia habitat on ADC with abundant water resources on Ol Pejeta. This combination secures higher populations of large grazers and browsers such as the elephant and reticulated giraffe, though research is needed to address the issue regarding artificial waterpoints and attracting mostly water-dependent species. ADC Mutara also faces two minor issues, a threat of the spread of prickly pear cactus (*Opuntia spp*.) observed during 2010 transect surveys and infrastructure needs for successfully sustaining safari and ecotourism venues.

Results of the hypothesis test addressing ranch size rejected the null that there is no correlation between ranch size and level of species richness. In this study, larger ranches are more likely to contain more habitat availability and can support greater species diversity. Conservancy size and proximity to other habitats are also important criteria in this regard and for future land acquisition. Results of hypothesis testing of guild habitat preference across all ranches also rejected the null in two of three tests. Grazers and browsers more frequently occur in grassland and Acacia habitat, as expected. Carnivore occupancy was not correlated to any one specific habitat.

CHAPTER 3. Anthropogenic impacts on diversity and distribution of large mammals in Laikipia County, Kenya.

3.1 Introduction

Context of managed water resources in Laikipia

On a global scale, demand for water has continued to grow due to increased human populations, widespread urbanization, failing infrastructure for water transportation and catchments, and impacts from climate change. Laikipia County, Kenya faces significant challenges in this regard with its semi-arid conditions, illegal abstractions at catchments, licensing problems, water treatment costs, human-wildlife conflicts, and a growing human population. The Water Ministry of Kenya has developed a Laikipia County Water Conservation Master Plan (LWF, 2014) to tackle the many challenges in water quality and access. This endeavor will rely on the Water Act of 2002 which created the Water Resource Management Authority (WRMA), providing for people and wildlife to access water from the Ewaso Nyiro River. What will be most difficult is to address the issue of equality, as there are many marginalized communities in Kenya at a significant disadvantage for access to water (Gichuki *et al.*, 1998). The Master Plan will likely require some intervention to balance the need for water with

Laikipia's communities as well as large landscape conservancies managing water for wildlife.

One of the most important and limiting natural resources for wildlife and local people is surface water availability (Western, 1975; Petty, 2002; Redfern *et al.*, 2003; Mwakiwa *et al.*, 2013). Wildlife and human settlements tend to center around sources of water (de Leew, 2001), where daily access must be met when including livestock requirements. With such an shared resource, it becomes highly important for landowners to work together, particularly in times of drought and or when natural events compromise access to water (IPSI, 2014). It is usually the private land owners who can afford water infrastructure for wildlife (Mwakiwa, *et al.*, 2013) and will have the ability to build water pumping stations that bring water to the surface from wells or from nearby rivers.

Conservancies in Laikipia quickly benefit from increased water holes through tourism revenue. Water holes make charismatic wildlife more visible to tourists (Okello, 2008), are usually where wildlife congregate (Western, 1975; Owen-Smith, 1996), thus an assumption is made that the more water holes a conservancy has the better for wildlife diversity, abundance, and distribution (Mabunda *et al.*, 2008). With radial gradients of wildlife occurring around artificial water points (Pickup *et al.*, 1998), overgrazing of vegetation at water points can create degradation of the larger landscape as well as damage soils (Ludwig *et al.*, 1997). De Leeuw *et al.* (2001) examined the diversity and distribution of wildlife in relation to livestock and permanent water points, concluding that livestock occupancy was associated with permanent water holes, where native wildlife were found further away from artificial water points. Studies on short term response to artificial water points show an increase in diversity, whereas long-term impacts demonstrate lower biodiversity (Thrash *et al.*, 1991, Harrington *et al.*, 1999). An increase in artificial water points will attract particular species that could impact vegetation (Harrington *et al.*, 1999), have behavioral influence on community wildlife (Martin, 1983), and may result in increasing highly water-dependent species (grazers), such as elephant, zebra, and buffalo (Collinson, 1983). Mwakiwa *et al.* (2013) demonstrates a pattern in Kruger National Park resulting in a drop in roan antelope (Martin, 1983) over a period of seven years when there was a significant increase in the number of artificial water points. The water points attracted larger grazers, particularly during periods of drought, which outcompeted smaller antelope species.

These water holes are often installed on tops of hills and in the center of grassland habitat more appropriate for access to herds of cattle (Figure 3.1). Considering this is not a natural location for water to be found, it would be important for management to better understand the impact of anthropogenic water holes on diversity, abundance and distribution of wildlife in Laikipia.


Figure 3.1. Example of a borehole chamber with a drainage basin located on the Ol Pejeta Conservancy.

The Ol Pejeta Conservancy is known for maintaining a large number (n > 80) of water holes (bore holes) which were originally to provided for grazing of cattle. The number of water holes mapped (Figure 3.2) are limited to active water holes and several were grouped for their close proximity to one another. Ultimately, I map 22 independent artificial waterpoints on the Ol Pejeta Conservancy.

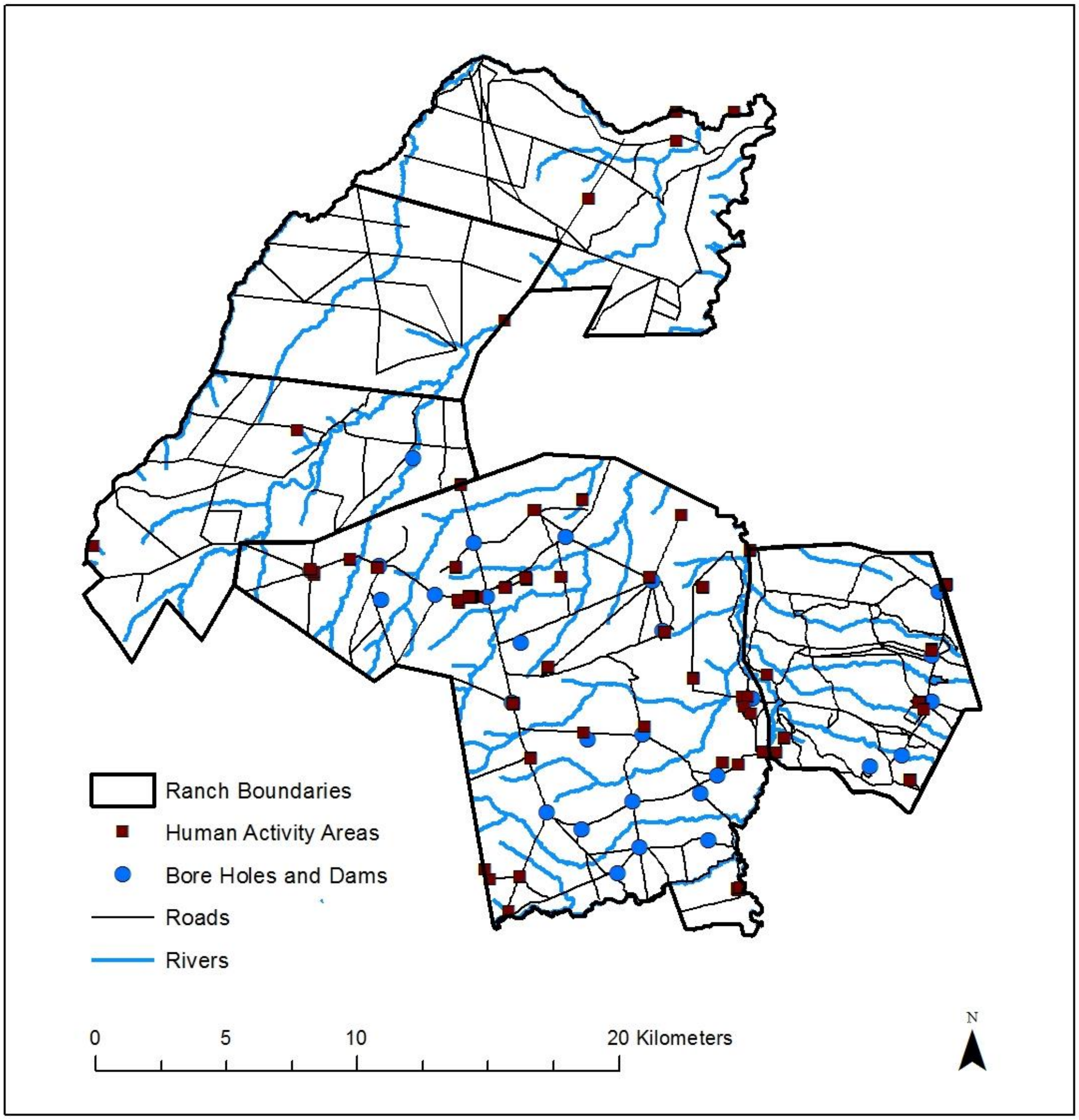


Figure 3.2. Locations of human activity areas, bore holes and dams, roads, and rivers in the conservation areas sampled during this study.

Roads and fencing infrastructure in Laikipia County

Roads are known to be significant sources of habitat fragmentation (Laurance *et al.*, 2004), increasing edge effects (Williams-Linera, 1990; Porensky, 2011, 2012), enabling pervasion of invasive species (Gooseman, 1997), impacting gene flow across landscapes (Riley *et al.*, 2006), creating barriers for migration and localized animal movement (Forman, 2003; van der Ree *et al.*, 2007; Taylor and Goldingay, 2004, 2009), and impacting highly dispersing wildlife (Elliot *et al.*, 2014). The impact of roads on wildlife has been documented as early as 1925 (Stoner, 1925) and has been increasingly studied in the past 15 years, including research on culverts (McDonald and St. Clair, 2004), canopy bridges (Gregory *et al.*, 2014), and large under and overpass structures that attempt to provide access for movement (Ng *et al.*, 2004). As an emerging field within ecology, we see growth in this areas of science through the establishment of the International Conference on Transportation and Nature (ICOET), the Road Ecology Center at the University of California, Davis, and the Western Transportation Institute at the University of Montana. The majority of impact studies have been in North America and Europe (Fahrig and Rytwinski 2009), with few quality studies in Africa (Laurance *et al.*, 2006; Vanthomme, *et al.*, 2012, Vanthomme *et al.*, 2015). There are very few studies that focus on the effects of roads within national parks or protected areas (Ament, *et al.*, 2007) (Figure 3.3).

Today, there are an estimated 1.2 billion vehicles (Green Car Reports, 2015) traveling over 11 million miles (18 million km) of roads, worldwide. With growing wildlife-vehicle collisions (KNBS, 2009), and an increase in road development in natural

Figure 3.3. An example of a raised, gravel road in Amboseli National Park, Kenya.

areas, it is imperative that transportation and infrastructure be managed to minimize impacts to wildlife. This is also important inside protected areas, where numerous roads and networked off-road trails are less likely to be considered a major influence on wildlife. Kenya's 160,000 miles (258,000 km) of roads largely include dirt roads, back country roads, and numerous unpaved roads that bisect parks and protected areas (Figure 3.4). Very recently, Kenya received great criticism for authorizing Chinese-funded development of a railway to cut through the famous Nairobi National Park.

Roads inside protected areas are physical characteristics of the landscape and may also have an impact on diversity, distribution, and abundance of wildlife species as would similar roads outside protected areas. The conservation of wildlife populations within a matrix of road infrastructure is an ecological challenge requiring further research. The choice of wildlife management technique will ultimately depend on the location, resources, and condition of the needs to preserve the species. In this study, I incorporate the measure of distance to nearest major road to each camera trapping unit as a response variable in estimating impact to species richness.

Figure 3.4. A low impact road inside the El Karama Conservancy (left) and a larger service / tourist roads on the Ol Pejeta Conservancy in Laikpia, Kenya.

Protected areas in Kenya are also quite often accompanied by fences (Clevenger *et al.*, 2001; Creel *et al.*, 2013), as this becomes a compulsory management technique in keeping biodiversity from dispersing out of the boundaries of a park and into conflict with humans (Figure 3.5). Ironically, the need for a fence to either keep people out or wildlife in impedes the very connectivity that landscape ecology requires (Woodroffe and Ginsberg, 1998). Fencing undermines ecosystem function leaving conservationists with the challenge of continually maintaining or restoring landscape connectivity (Curtin, 2015).

Large mammals, particularly carnivores, are among the groups of wildlife heavily impacted by anthropogenic factors such as roads and fencing and are observed to be on a global decline (Blaum *et al.*, 2007; Estes, *et al.*, 2011). Large carnivores can travel great distances and require large home ranges, inevitably bringing them into contact with humans. One of the most studied of these species in Africa is the lion (Loveridge *et al.*, 2010), where it has been largely suggested that fencing may be the best option for their long-term survival (Packer, *et al.*, 2013). Durant *et al.* (2015), Woodroffe *et al.* (2014) and Creel *et al.*, (2013) argue that populations of lions in fenced systems are often small, managed above carrying capacity, and view fencing as a last resort. Additionally, the high economic and ecological costs of fencing would otherwise be balanced out by the benefits of a large landscape approach (Woodroffe *et al.*, 2014), with empirical evidence needed on dispersal prior to management implications (Elliot *et al.*, 2014). With other large mammals whose populations are more density-dependent, such as the elephant and buffalo, fencing can reduce their population numbers and alter interaction at the community level, potentially leading to an "ecological meltdown" (Terborgh, *et al.*, 2001).

Figure 3.5. Variations of fencing and corridor technique used in Laikipia County. Above example on private propety near the Mpala Research Center, using sharp, protruding single wires hanging outward as a deterrent. Image below represents a wildlife corridor on the Ol Pejeta Conservancy designed to prevent rhino from exiting the conservancy, but allowing all other wildlife to pass.

Cattle management in Laikipia and the preservation of the Boran

Livestock production in Kenya is responsible for nearly ten percent of gross domestic product (World Bank, 2015). Stakeholders in this industry are made up of a variety of large and small scale operations from the local small-scale ranching to highly commercial levels. Beef production in Kenya has been gradually on the decline due to a reduced per capita red meat consumption (ILRI, 2015), overall economic decline, a decrease in the exportation of live animals, and the persistence of diseases such as Rift Valley Fever (RVF) throughout semi-arid regions of Africa (Aklilu, 2002; DePuy *et al.*, 2014). In semi-arid counties, such as Laikipia, pastoralists can often lack the resources, reaction time, and market opportunities necessary to recover from catastrophic natural events (floods, droughts, disease outbreaks) (Homann, 2004). It becomes increasingly important that wildlife conservancies that breed cattle consider the requirements of the community and find ways to integrate benefits that can be shared across a landscape. Pastoralists not only invest in cattle breeding for the obvious economic benefits, but for cultural and historic significance (Haile-Mariam and J. Philipsson, 1998; Lane 2013). To successfully maintain these sources of revenue and cultural ties in breeding cattle, pastoralists must consider management issues addressing disease, nutrient quality of native grass, and behavioral aspects of cattle that might reduce predation (Gifford-Gonzalez, 1998).

There are over 250 recorded breeds of cattle worldwide, usually classified into two species of the genus, *Bos*. The Zebu (*Bos indicus*) is found in Southeast Asia and Africa and the Taurine (*Bos taurus*) come from European origins. The Boran (*Bos* *indicus*), is a breed of cattle derived from the Zebu variety originally bred by the Borana Oromo people of Ethiopia (Figure 3.6). This breed is widespread throughout East Africa, but most popularly known in Kenya at an estimated head count over 550,000 (ILRI, 2015). The largest pure bred herd of over 6,000 animals can be found on the Ol Pejeta Conservancy. The Boran is a traditional beef cow with a humped back, much like that of the Brahman cattle in India. Growing awareness of declining genetic diversity and cultural heritage of the commonly known Boran breed of cattle in Kenya has led to the establishment of various conservation programs such as that found on the Ol Pejeta Conservancy (OPC, 2015; Haile-Mariam, 1998). Conservation goals to increase the population of purebred Boran include reducing monoculture operations, reducing crossbreeding, promoting the ecological utility of the Boran breeds in conservancies, and building strong connections with local pastoralists through integrated cattle management.

Figure 3.6. A Boran bull photographed in Euclea forest on the Ol Pejeta Conservancy.

Boran are known for their hardiness, ability to tolerate low quality forage, and their natural anti-predator herd instincts. For these reasons, the Boran adapts well to arid and semi-arid climates with lower precipitation such as that found in highlands of Laikipia, Kenya. The genetics of the Boran are unique to breeders in East Africa, with the Ol Pejeta Conservancy having placed particular importance on careful and meticulous breeding management. The Boran on Ol Pejeta maintain the most valuable genetic diversity of all Boran in East Africa (Boran Cattle Breeders Society, 2015). Additionally, their cultural significance is an investment in the variety of stakeholders in the Laikipia area that help achieve breeding objectives. An economic asset to local beef producers and beef markets in Kenya, the Boran is often the financial stability for many wildlife conservancies in Kenya (NRT, 2007; Zander, 2008).

The Boran are recognized in Kenya's beef and livestock industry as one of the most productive and profitable animals to manage and breed (ILRI, 2015). They maintain a series of characteristics that make them highly sought after, including a walking aptitude, drought resistance, temperament, extreme weather tolerance, herd instincts, advanced digestion, and disease resistance (KALRO, 2015; Rewe *et al.*, 2006; Riley *et al.*, 2001). According to the Boran Cattle Breeders Society (2015), Boran can walk long distances withstanding drought conditions and consume nutrient poor grasses during intervals of stress. Calves learn to walk very quickly after birth and can keep up with the herd, minimizing calf vulnerability to predation. Increased sweat glands, durable and thick skin, and a unique oily and UV reflective coat in Boran allow them to graze for longer periods of time in the day than other breeds of cattle (OPC, 2015).

These features also help reduce tick and fly infestations. A low metabolic rate is known to contribute to assist in digestive processes and reduces their susceptibility to illness following severe climate conditions.

Boran calves can reach sexual maturity at 18 months, with cows capable of reproducing for more than 10 years (Beal *et al.*, 1990). Lampkin and Lampkin (1960) had originally attributed their high fertility to low body weight over the course of the suckling period. A study by Nicholson and Sayers (1987) show that calving and birth rates were unaffected by water scarcity. Parental care is high in Boran females (Riley *et al.*, 2001), contributing to offspring success. The low temperament of the bulls allow them to be in closer proximity to one another in herds and in bomas, reducing much work on the cattle managers and guards.

Ol Pejeta has worked with the Northern Rangelands Trust (NRT) in prior years through a program called "linking livestock market with wildlife conservation" (NRT, 2007), offering opportunities to local communities to improve their livelihoods. The program helps locals by offering opportunity to earn income through livestock sales, access to potable water for their families, learning grazing methods, reducing landscape pressure with cattle management (AWF, 2011). Pastoralists can improve their current practice with new training and resource access. Ol Pejeta also makes connections to integrated cattle management in local schools through teacher education, tours to OPC, and sponsoring events for community children. One of the more unique aspects of the Ol Pejeta management is integrating Boran cattle in small herds (<100) among native wildlife (Figure 3.7) through a system of mobile bomas (small, temporary enclosures for cattle) (LPFN, 2015). Traditional bomas are made of thorny vegetation which has to be cut and piled to form a barrier for livestock from predation. Mobile bomas are designed for quick installation and are made of detachable fencing material (Figure 3.7), which are far more successful in keeping out predators, particularly hyena which can push through traditional boma material. Livestock guards are encouraged to distribute cattle to areas of poor quality (taller) grasses, which invigorates new growth while depositing nutrients. Native wildlife can then feed on improved grass and grasslands and Acacia forests have reduced pressure from encroachment.

Figure 3.7. Mobile boma structures in use by the Ol Pejeta Conservancy. Photos courtesy of Landscapes for People, Food, and Nature (2015).

The Mpala Research Center has been investigating management of bomas as a tool for landscape conservation through dissertation research (Porensky, 2012; Porensky and Young, 2013) and development of a management guide (Porensky *et al.*, 2011) that contains instructions for monitoring the effects of bomas across the landscape. Lessons learned address quality of grass, length of boma use, adaptation of bomas to their landscape features, and sharing of bomas and monitoring of data. Such management is used to support wildlife conservation in the context of large landscape management. The Ol Pejeta Conservancy has been demonstrating the benefits to grasslands of integrated wildlife-livestock management for over 10 years and very recently adopted the mobile boma system. The mobility of smaller, more focused cattle herds can help to break up the hardpan soil while depositing nutrients. The visible effects of improvement to the landscape have been widely recognized by Ol Pejeta management and a focus of proposed academic research.

Areas of human activity in conservancies

Human settlements have long been known to alter ecological processes (Theobald, *et al.*, 2000, 2005). Important natural areas quickly diminish as human population and occupation of landscapes expand. The year 2008 marked an important point in time when more than half of all people on Earth lived in urban areas (UNPF, 2015). Since then, urbanization has been increasing on a global scale with profound effects on the planet's ecological systems (Luck and Wu, 2002), usually to a point where environmental decline is irreversible. This trend is usually seen in developing countries where the majority of negative impacts are from urbanization (Langpap and Wu, 2008).

In 2002, the Wildlife Conservation Society published a report entitled "The Human Footprint" that demonstrated an impact of humans on landscape throughout the world (Sanderson, 2002). It revealed a staggering image of land-use change on planet Earth (Figure 3.8), with alarming development in Africa (Hansen *et al.*, 2005).

According to the report updated in 2004, there are few areas, if any, in all of Kenya that have not been affected by humans. A failure to learn from prior research and modify human behavior will surely result in further degradation of landscapes ultimately impacting human health (Dannenberg, *et al.*, 2003).

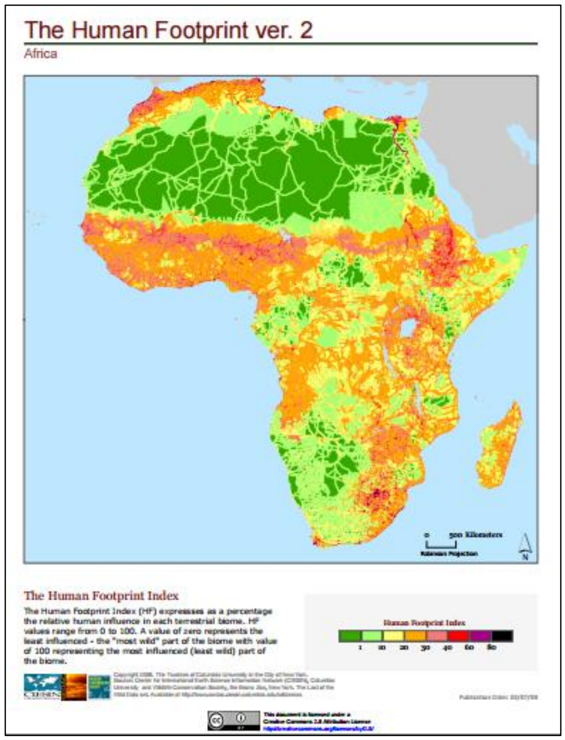


Figure 3.8. Image of human Footprint map from the Wildlife Conservation Society (WCS) covering Africa. Green are less impacted, whereas red to black are highly impacted.

The network of road and railway development has had a long and significant impact on Kenya's growing population centers, with Nanyuki as one of Kenya's fastest growing cities. Adjacent to the Ol Pejeta Conservancy, Nanyuki is now home to numerous conservation organizations and safari companies benefitting from access to the distribution and collection of conservancies. In these rural areas, small towns and villages are dotted throughout the landscape (Figure 3.9) and have been a permanent part of Laikipia for many decades.

Figure 3.9. Human activity areas include research and/or safari bandas (huts) (above), new housing development in cooperation with conservancy (center) and staff housing inside a conservancy (below).

Within the Ol Pejeta conservancy, there are numerous structures to house humans, both permanently and for seasonal and/or periodic use. Some examples of human activity areas are offices and administrative housing, research housing, staff housing, automotive and cattle facilities, storage, tented camp safaris, and other smaller structures with significant human activity.

3.2 Methods

Geographic Information Systems and camera trapping

Similar to vegetation surveys conducted during this research, GIS was used to establish accurate maps of anthropogenic structures across the landscape, namely roads, human activity areas, fencing, and bore holes and dams. The initial GIS and Remote Sensing (RS) support came through the African Wildlife Foundation (AWF) who provided general administrative data (boundaries, roads, rivers, etc.) and remotely sensed images of Laikipia County, Kenya using SPOT 2010 imagery. This effort was followed by a remote sensing training grant through the American Association of Geographer's (AAG) SERVIR program providing access to Kenya's Regional Center for Mapping of Resources for Development (RCMRD) in Nairobi as per agreement under an AAG/NSF grant (# 0934063). Three months of training was provided at the RCMRD, which included remote sensing analysis and access to their imagery database for cloudless images of Laikipia County between the years of 2009 and 2013. In addition, an imagery grant was received in 2010 from the GeoEye Foundation covering the majority of the study area. Data derived from of these sources were combined with support from the African Wildlife Fund (AWF) GIS Lab in Nairobi and Washington DC, and the Smithsonian Institution's GIS resources.

Minor image processing utilizing ENVI™ (Environmental Vegetation Index) software (EXCEL/VIS, 2015) helped identify habitat prior to field ground-truthing surveys and to assist with identifying long term vegetation monitoring plots. The use of Arc GIS 10.3.1, ArcGIS Pro™ Advanced (Esri, 2015), Google Earth Pro (Google, 2015), and Earthpoint.us (Clark, 2015) online measuring tools were incorporated into all GIS use throughout this dissertation. Through ArcGIS Spatial Analyst in Microsoft Excel, I use data from individual cameras in proximity to selected predictor variables, without the recognition of ranch boundaries on a landscape. All data were uploaded for processing through a general linear model (GLM) via Statistical Analysis Software (SAS, 2012).

Occupancy modeling by individual camera across the landscape

Two occupancy modeling approaches were used in this study. The first analysis focused on species richness where data were at the individual camera level across the landscape, with general liner modeling run using SAS/STAT™ software analysis (SAS, 2012). The second set of models were designed to focus on detection (presence/absence) data for trophic guilds, where grouped camera data were utilized and models run using PRESENCE v6.1 software (PRESENCE™, 2015).

This analysis was conducted by focusing exclusively on the individual camera, which provides a different perspective of the landscape than that of presence/absence data. I determined which variables are most likely to be attributed to species richness throughout the landscape by using Akaike's Information Criterion adjusted for small sample sizes (AIC_c), and Akaike's Weights, calculated from the residual sum of squares (RSS) from linear regressions. I used species richness as the response variable and, similar to trophic guild occupancy modeling, I used proximity to the nearest bore hole, proximity to nearest river, proximity to the nearest human settlement, proximity to the nearest major road, and condition of fencing as predictor variables. Conversions were needed for to provide fencing as a numeric ranking for analysis with low fencing=0, standard fencing=1, and maximum fencing=2. Density of cattle was estimated per ranch, but was ultimate removed from the model as it was highly correlated to fencing.

This approach identifies the most parsimonious models from proposed candidate models. All combinations of the predictor variables resulted in 31 candidate models. To improve normality, I used square root transformations on the data prior to analysis. The ΔAIC_c value for each model is the relative level of empirical support compared to the model with the highest support (Anderson 2008; Arnold 2010) and a value between 0 – 2 indicates strong support of the model. The weight of each model varies from 0 (no support) to 1 (complete support) and is the probability that it is the best model. The evidence ratio (Δ_i) is a quantitative measure of the strength of a model compared to the best model. Adjusted R^2 values indicate how well each model fits the data set and its relative efficacy as a tool for prediction. The relative importance of each predictor variable varies from 0 (no support) to 1 (complete support) and is calculated by summing the weight of each model in which the variable appears (Anderson 2008). Regression coefficients [± standard error (SE)] indicate whether predictor variables are positively or negatively associated with animal diversity.

3.3 Results

Anthropogenic impacts on species richness

Two models identifying differences in animal diversity across the landscape had strong empirical support relative to the other models in the candidate set ($\Delta AIC_c < 2$; Table 3.2). The adjusted R^2 values are somewhat low for the models, which means they do not encapsulate all factors influencing diversity. The AIC_c model that had the greatest support indicated that diversity was negatively associated with proximity to the nearest human activity areas and rivers (i.e., less diversity further from human activity and rivers), and positively associated with fencing (i.e., more diversity when there is more fencing). The second model with the highest support was similar, but also indicated a positive association with roads (i.e., more diversity farther from roads).

Table 3.1. Best fit AIC_c models from 31 candidate models that explain differences in animal diversity across the landscape. Five explanatory variables are used in the models: proximity to the nearest bore hole, proximity to the nearest human activity area, proximity to the nearest river, proximity to the nearest road, and fencing as predictor variables. The regression coefficient ± SE is given in parentheses for each variable. *K* =

number of parameters in the model, ΔAIC_c = relative level of empirical support compared to the model with the highest support; w_i = Akaike weights.

Model variables	K	ΔAIC_c	w_i	adj. R^2
Ψ (.) p (Human + river + fencing)	5	0	0.28	0.28
Ψ (.) p (Human + river + fencing + road)	6	1.39	0.14	0.27
Ψ (.) p (River + human)	4	3.01	0.062	0.24
Ψ (.) p (Human + fencing)	4	3.04	0.061	0.24
Ψ (.) p (Bore hole + river + human)	5	3.12	0.058	0.25
Ψ (.) p (Human + road + fencing)	5	3.59	0.046	0.25
Ψ (.) p (Bore hole + human + fencing)	5	4.22	0.033	0.26

The most important predictor variable (i.e., in most models that carried weight) was proximity to human activity areas (Σw_i = 0.77). Fencing and rivers also may be moderately important predictor variables because these variables carried a moderate weight (Σw_i = 0.66 and 0.62, respectively). Proximity to roads and bore holes are less likely to be important predictor variables because they carried lower weights (Σw_i = 0.29 and 0.16, respectively).

The consideration in selecting representative AIC models is largely contextual, based on the gradient of outcomes from multiple candidate models, and within the purview of the researcher for selection. For multiple candidate models, the AIC model with the lowest score, preferably at zero, is the preferred model. Models resulting in $\Delta AIC > 10$ do not contain any support and should be disregarded from any further consideration (Barnham and Anderson, 1998). Recognition of supported AIC models is commonly ≤ 2 ΔAIC (Barnham and Anderson, 1998; Anderson, 2010) and often seen in ecological research (SCBI, 2015; Richardson, *et al.*, 2011; 2013; 2014), and at times greater than 2 ΔAIC (Richardson and Hanks, 2009). In this study, I recognize supported models within 4 ΔAIC given that several of my candidate models fall within this range and contain values in close proximity to one another. I list all candidate AIC models in tables 3.1 and 3.2 containing values up to 10 and 4 ΔAIC respectively for the sole purpose of visualizing the distribution of increasingly weaker models and their values as they deviate from the preferred range.

3.4 Discussion

Surface water, be it from rivers or artificial waterpoints, will remain a crucial and constraining resource for large herbivore populations in Laikipia County, Kenya. Virtually all waterpoints in the study site were originally installed to accommodate the needs of livestock management with the added benefit of access for native wildlife. Artificial waterpoints clearly bring in the big game species which could boost ecotourism, but careful consideration should be addressed at the large landscape scale among all members of the Central Laikipia Collaboration (CLC) to view artificial waterpoints in the context of its impact to native herbivore diversity.

An important observation made in the field was that the locations of artificial waterpoints were almost exclusively within grassland habitat and frequently located at the top of small hills. Artificial water points contain a holding cistern where water is pumped and then fed to an outlet at its base (Figure 3.1). The attraction of wildlife to this structure could skew toward water dependent species, particularly large grazers like elephant, zebra, and buffalo (Collinson, 1983; Harrington *et al.*, 1999). Over long periods of time, such visitation could alter vegetation regimes (de Leeuw *et al.*, 2001) and inadvertently affect the ability of the landscape to support a more diverse community of native ungulates. Modeling results show that artificial waterpoints had no correlation to species richness (Figure 3.10), but were correlated to rivers. It is recommended that a future study concentrate on the distribution of artificial waterpoints and their impact on species diversity.

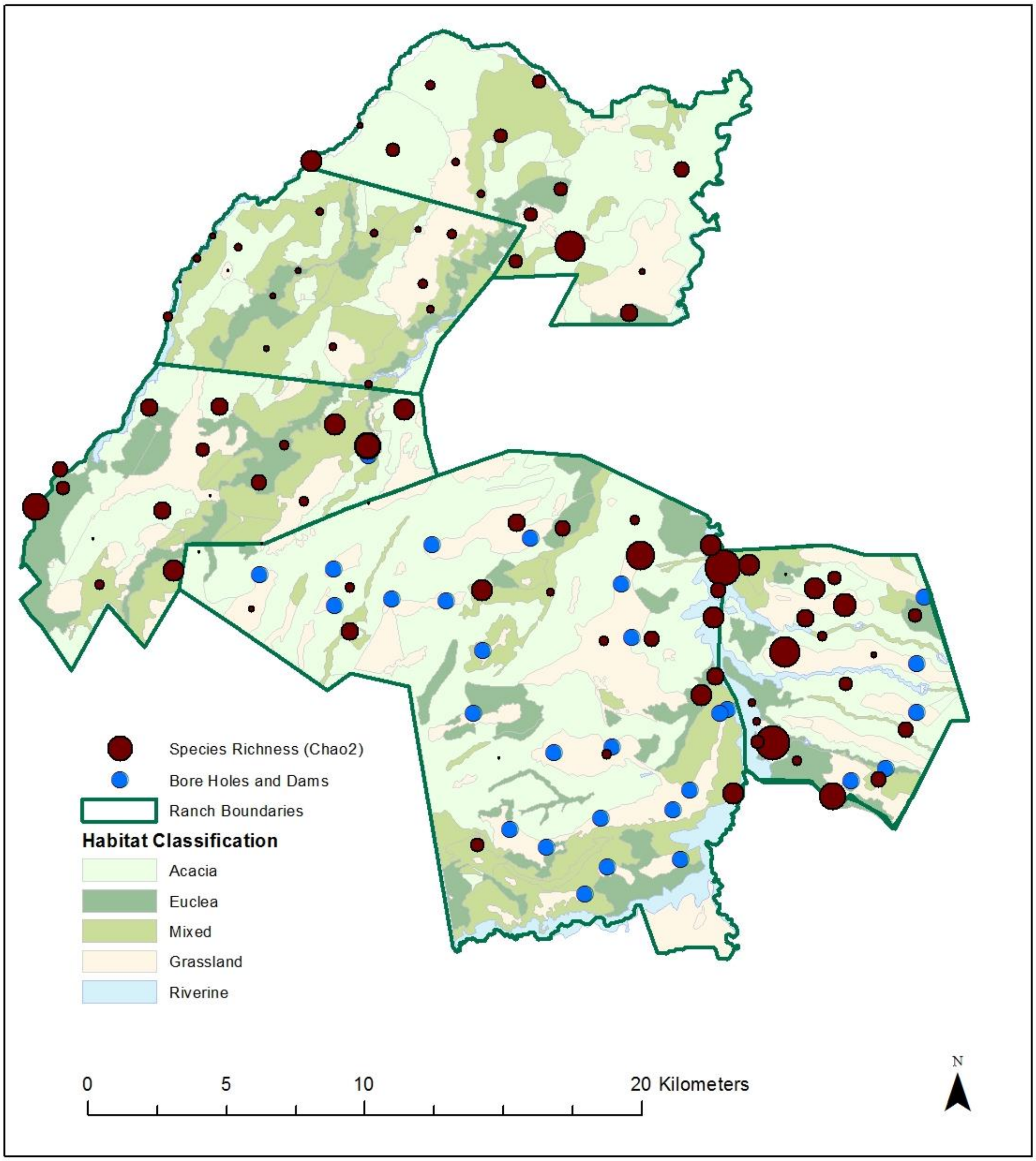


Figure 3.10. Species richenss (Chao2) values (maroon) for camera traps arranged in size from richness of 1 (small) to 18 (large) overlaid with the distribution of artificial waterpoints (blue).

Artificial waterpoints were not correlated to increased richness values on the Ol Pejeta Conservancy. High richness values occurednear the Ewaso Nyiro River. However, ADC Mutara ranch does show a distribution of higher richness values more evenly spread out throughout the ranch which has only one artificial waterpoint (a small dam). With the high frequency of large grazers observed at waterpoints, particularly in OPC West, it is possible that these grazers have occupied the waterpoints to such a degree to have behaviorally displaced other native ungulates, thus reducing overall diversity. Mwakiwa *et al.* (2013) found indirect effects on herbivore species from the presence of water-dependent species attracted to artificial waterpoints in South Africa. The implications are important when including the strong economic incentive of a ranch managers to increase tourism by increasing artificial waterpoints. The desire to photograph Africa's "big five" may indeed encourage waterpoint infrastructure, but inadvertently decrease overall ungulate diversity.

Models reflecting a correlation of access to water to species richness was only supported by the data when combined with rivers, humans, and roads. I suspect the association with rivers and numerous tented safari camps may have played a role in this correlation, as well as the practice of bringing cattle (which come with humans) to the river in small mobile bomas across the landscape. Roads are often associated with rivers, given their use for safari and ecotourism as well as monitoring of wildlife by staff.

With regard to overall diversity between ranches, Ol Pejeta Conservancy (as a whole ranch) had significant'y higher diversity than Eland Downs Ranch. Throughout the entire Central Laikipia Collaboration (CLC), these two ranches have been on the polar ends of both size and resource availability. Eland Downs is the smallest of all the ranches are 17,500 acres and heavily impacted by cattle grazing, illegal visitation by humans with cattle, and generally in very poor condition from intense grazing abuse prior to 2009 (Kinnaird and O'Brien, 2009). Eland Downs lacks fencing that might otherwise prevent the visitation by cattle, whereas Ol Pejeta has the tallest and strongest electrified fencing around its entire perimeter (Figure 3.11). With its unique wetland habitat, it will be challenging for any adjacent conservancy to compete with the size and diversity of habitat within Ol Pejeta. It should be noted that Eland Downs had no significant difference in total richness with any other property in the study, suggesting careful management of its habitat and water resources could enable its wildlife occupancy rates to rise to that of adjacent ranches.

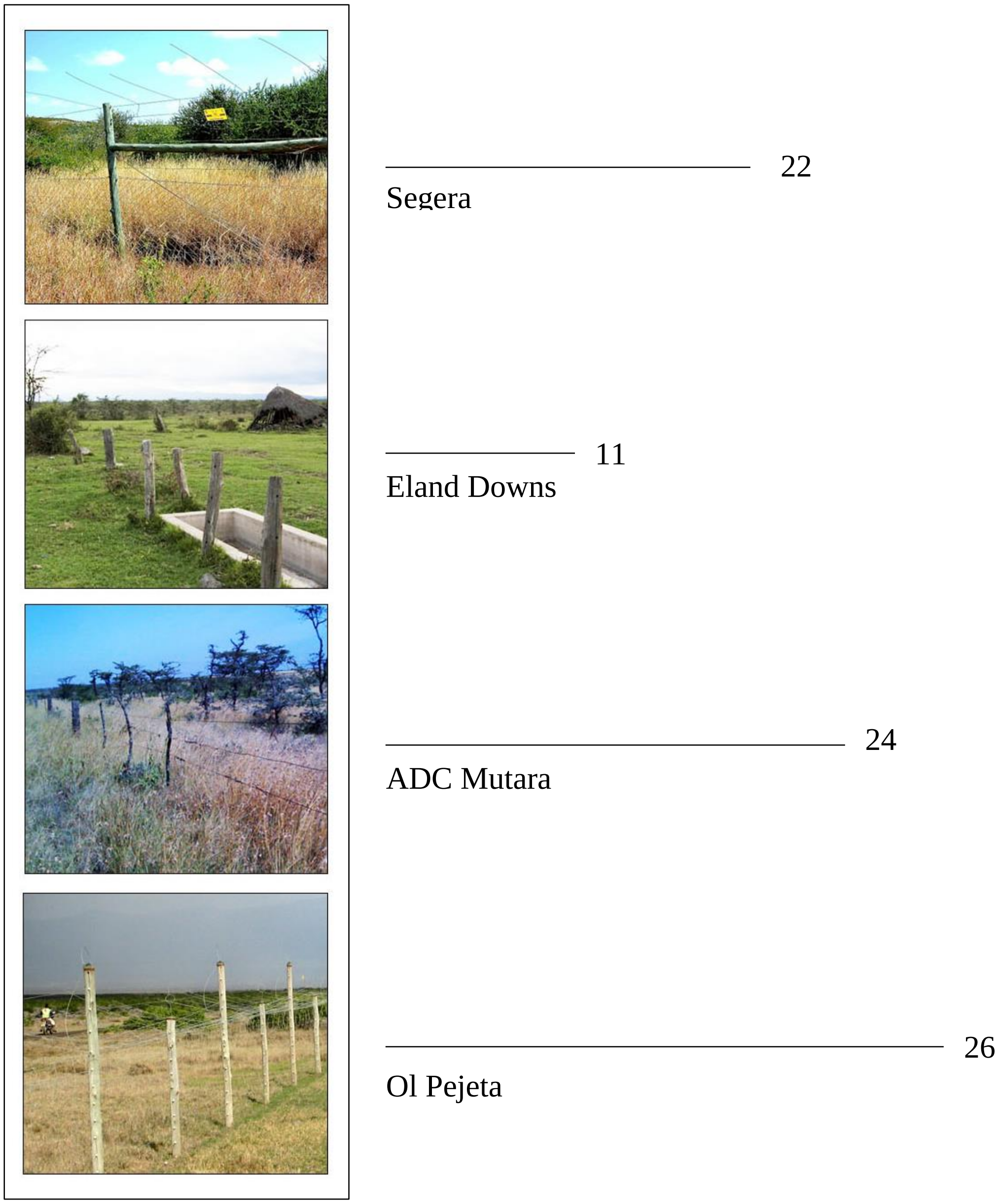


Figure 3.11. Variety of fencing types throughout study site with associated scale bar of average species richness values (Chao2): Segera (22) high fencing, Eland Downs (11) poor fencing, ADC Mutara (24) moderate fencing, and Ol Pejeta Conservancy (26) high fencing.

Throughout Laikipia, fencing as it applies to wildlife conservation, is usually installed to keep wildlife from dispersing (Woodroffe, 2014). Wildlife are better protected with fencing (Packer, *et al.*, 2013), but not at the large landscape level (Creel *et al.*, 2013), where fencing serves to fragment habitat. Kinnaird and O'Brien (2012) found that fencing on ranches and conservancies in northern Laikipia were semipermeable, where inconsistency of perimeter fencing allowed movement of wildlife. In this study, I make the assumption that ranches with poor to no fencing in Laikipia often resulted in both dispersal of native wildlife and illegal access of cattle into that property. With cattle come their human rotation managers resulting in an association between native wildlife, humans, and cattle. Figure 3.11 shows examples of fencing used throughout the study area, with high, electrified fencing installed at the Ol Pejeta Conservancy and virtually no fencing at all on Eland Downs Ranch. Fencing condition matched the prediction for increased species richness.

Availability of artificial waterpoints increased diversity only in combination with other predictor variables; data show no correlation of richness with artificial waterpoints alone. Richness was more closely related to natural rivers. Other factors such as roads and human activity were not correlated to diversity, but were included in higher ranking models. These data lean toward a positive correlation of humans and roads to diversity, likely a result of purposeful placement of tented camp safari locations in spots advantageous for wildlife viewing and the usage of natural rivers by humans and their cattle. Lastly, modeling indicated that fencing impacts wildlife diversity. Ranches with more intense and secure fencing had higher richness.

CHAPTER 4. Strategic use of monitoring and mapping technologies for public-private partnerships supporting wildlife conservation in Laikipia, Kenya.

Sustainably managing biological diversity within conservation landscapes has become increasingly challenging as economic, sociological, and human-wildlife conflict further interact. The relevance and importance of biodiversity in these settings has not diminished over time, if anything it has become more critical to human survival as our own populations continue to increase and negatively impact landscapes (Pringle *et al.*, 2010; Ripple *et al.*, 2015). Unfortunately, the value of nature and utilization of biodiversity is still primarily recognized through direct, consumptive use and global market values. Ecological systems, ecosystem services, cultural connectivity and complex cycles that provide these resources are still without proper financial measure, though the principles for conserving them have been outlined with great effort (Mangel, *et al.*, 1996).

Anthropocentric and utilitarian approaches to managing biodiversity have shifted to the private landownership level as properties collectively make up intact landscapes and are often managed in isolation with little consideration for their surrounding communities or the greater ecology (Western *et al.*, 1994). It can be observed in Kenya that wildlife conservancies working alongside local and pastoral communities are capable of hosting ecotourism in highly modified human landscapes (Homewood *et al.*, 2012). Managing such landscapes is often without the benefit of lessons learned from pastoralist's interaction with nature. Additionally, while our understanding of the ecology of human-modified landscapes is still quite poor compared to areas immediately surrounding national parks (Knight and Cowling, 2007). Through Laikipia's community-based wildlife conservation efforts, there is a renewed interest in creating economic opportunities for locals within the public-private conservation framework (LWF, 2015). These opportunities not only involve inclusion of locals to wildlife conservation effort, but also seek opportunity to use relevant technologies that allow for managing wildlife conservation in a human dominated landscape (Ochieng, 2015).

In this chapter, I review public-private partnerships in Laikipia, explore the application of GIS technologies for land management, introduce camera trapping as a necessary and standard tool for wildlife monitoring, and propose on-line technology with mapping application development.

4.1 Public-private conservation partnerships in Laikipia County, Kenya.

The Laikipia and Ewaso wilderness area of Kenya comprises a variety of private and public lands, local communities, conservation areas, and rangelands (Didier *et al.*, 2011a, 2011b). The entire Laikipia County lacks national protection of the environment, with the exception of the Eland Downs ranch now under ownership of the Kenya Wildlife Service (KWS). The methods in which Kenya's pre-colonial communities have historically interacted with large landscapes is not a part of today's partitioned and highly managed use of wildlife conservation areas (Krameri-Mbote, 1999; Manyara and Jones, 2009). Privatization brought fencing and intense infrastructure to a more localized approach at land management without much recognition of greater ecological systems. Though Kenya maintains 22 national parks, 28 national reserves, and 5 national sanctuaries (KWS, 2015), it has been estimated that greater than 70% of wildlife throughout Kenya are located outside of these protected areas (Chape *et al.*, 2005; Western, 2009) as seen through imagery of Kenya's Human Footprint Index (HFI) (Figure 4.1). Conservation and wildlife management in various communal and privately owned lands is a key element to the sustainability of Kenya's wild and living resources (Gitahi and Fitzgerald, 2011). Private lands now engage in shared resource agreements, conservation easements, and conservation enterprises, which combine biodiversity conservation with the added benefits of tourism revenue (Carter *et al.*, 2008). Soon after Kenya banned hunting of wildlife in the late 1970s, areas capable of sustaining large herds of native herbivores quickly benefitted from growing tourism. Pro-wildlife private lands with high populations of native wildlife became income-sustainable, found new ways to maintain cattle integration (OPC, 2015) and work with the larger group ranching efforts with more local communities and the patchwork of private and protected land parcels (AWF, 2015). Both private ranches and grouped ranching efforts often join to form collaborative conservancies where resources are shared and spread among partners. At the large landscape scale, conservancies and private property become effective wildlife corridors. Outside of its largest and fastest growing town, Nanyuki, Laikipia's human population is relatively low. This low rural population surrounded by high pro-wildlife private land ownership allow more opportunity for effective biodiversity conservation and sustainable tourism.

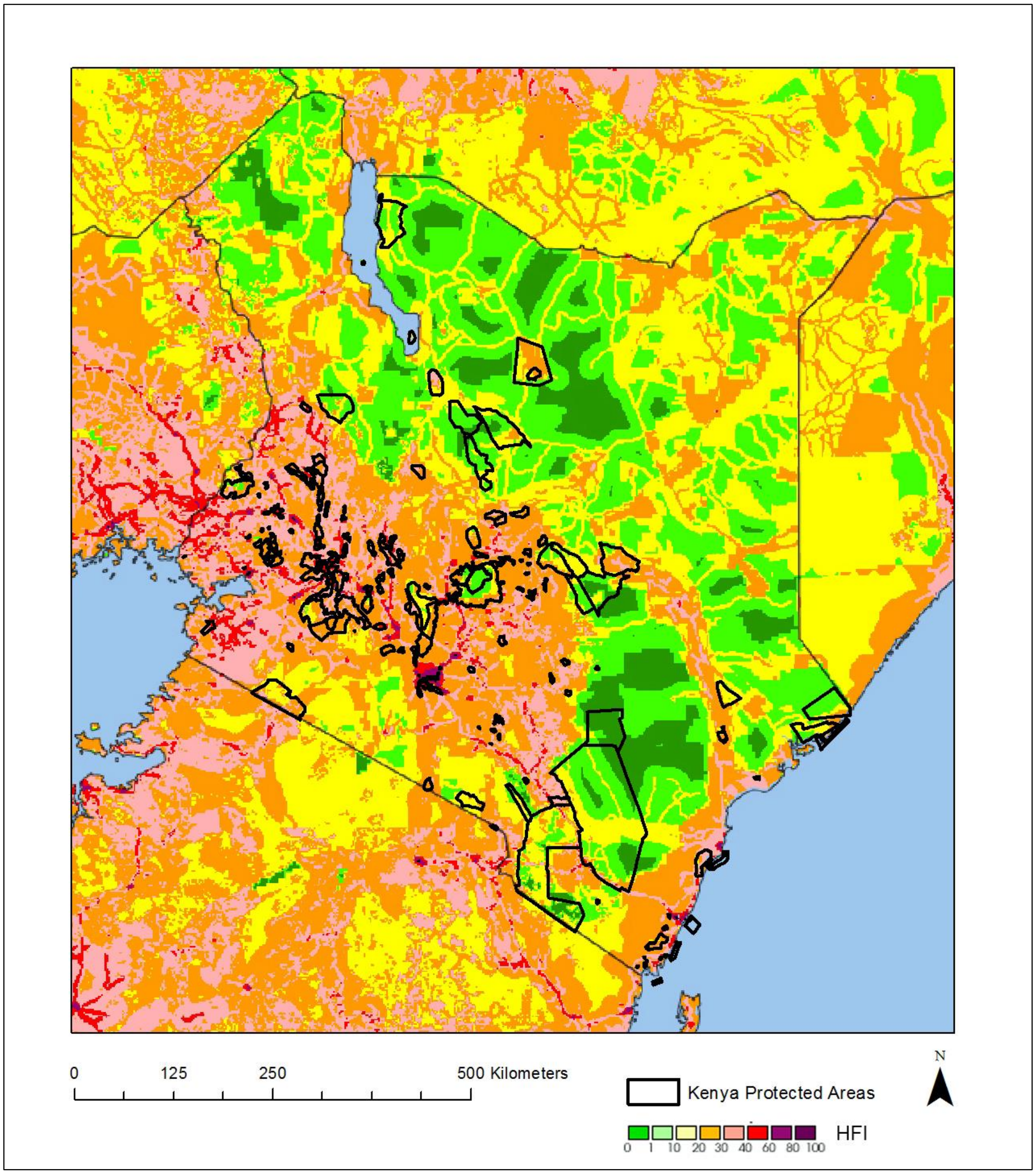


Figure 4.1. Georeferenced base image of Kenya's Human Footprint Index (HFI) of relative human influence in each terrestrial biome (Woolmer *et al.*, 2008), overlaid with Kenya's protected areas boundaries. A value of 0 (dark green) represents the least influenced (most wild) part of a biome with a value of 100 (dark red) representing the most influenced (least wild) part of a biome.

For over 25 years, populations of wildlife have been closely monitored throughout all regions of Kenya by the Ministry of Environment and Natural Resources' (MENR) Department of Range Surveys and Remote Sensing (DRSRS) (MENR, 2015). Results have shown a rather alarming and steady decline of large mammals throughout the entire country, which can be representative of the impact of conservation (Georgiadis, 2007). Wildlife and natural resources are under pressure as a result of the growth in human populations and land sub-division activities (LWF, 2015). Inevitably, human-wildlife conflict has become an increasingly heated issue as land-use changes are more incompatible with the populations of wildlife. Kenya's amended constitution contains provisions for a healthier balance for humans and wildlife (Kenya National Council for Law Reporting, 2015), as the former command-control approach in conservation to increase environmental protection has been ineffective (Holling and Meffe, 1996). Though many community-based conservation efforts in East Africa are well underway, tropical biodiversity as a whole continues to diminish, which may call for ecologists and conservationists to address the problem more directly and not to rely too heavily on indirect interventions (du Toit *et al.*, 2004).

The viability of wildlife in non-protected landscapes will be a better determinant of their persistence throughout the savannas of East Africa than in national parks or reserves (Western, 1989 and Hutton *et al.*, 2005; Veblen, 2012; Ford *et al.*, 2014). This is largely due to the ecological needs of large mammals that tend to depend on the geography and resources of various small protected areas that might be accessible. More so than in protected areas, the conservation efforts that are required to sustain large populations of wildlife require very active management practices. For many, this in turn will be dependent on wildlife interaction among human communities (with or without livestock), and how human activity may influence the greater ecological process involved (Georgiadis *et al.*, 2007b).

There are numerous group ranches in Laikipia County, communally owned by pastoral families as well as large-scale commercial ranches where the livestock are managed at moderate to high densities. Research show that pro-wildlife properties (owned by wildlife supporters) tend to have low density and small-scale operations, which favor sustaining large populations of large mammals (Kinnaird and O'Brien, 2012). The opposite can be seen in more transitional properties, owned by individuals who tolerate or actively discourage wildlife. To approach the complex issue of understanding land owners and how best to maintain the integrity of an ecosystem, active community-based conservation initiatives have been in put place throughout Laikipia. Georgiadis *et al.* (2003, 2007) and Woodroffe and Frank (2005) have conducted numerous case studies specifically for large mammal conservation.

Robust studies from Georgiadis *et al.* (2007a and 2007b) involved the modeling of ungulate populations throug time and showed that the abundance of herbivores in the Laikipia system varied greatly with land-use type and that the populations were limited by rainfall (cattle and plains zebra *Equus burchelli*) or by other factors such as density (plains zebra and giraffe) (Coe *et al.*, 1976; Goheen *et al.*, 2012). Overall a pattern emerges for the Laikipia's wildlife population numbers. Aerial surveys by the DRSRS reveal diminishing populations of wildlife in Laikipia over a period of less than 10 years (Georgiadis *et al.*, 2011), but more recent observations of wildlife (Kinnaird, 2012) and an increase in camera trapped events along corridors between ADC Mutara Ranch and the Ol Pejeta Conservancy (Figure 4.2) show a sharp rise not only in corridor use, but in the overall numbers of large mammals using those corridors.

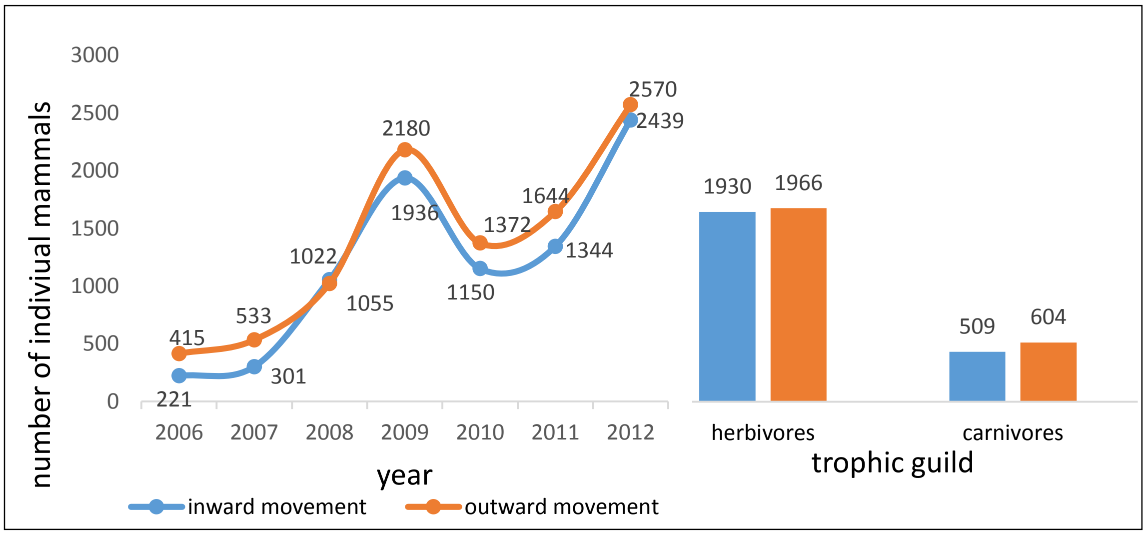


Figure 4.2. Estimated numbers of individual wildlife utilizing two corridors between the ADC Mutara Ranch and the Ol Pejeta Conservancy (left), and the difference in numbers between herbivores and carnivores during 2012 activity. Graphs courtesy of OPC (2015)

Perhaps a direct contribution to these changing numbers is the tolerance of sharing the landscape with livestock. The impetus for sustaining group ranches and communal lands might not be solely for income generated through cattle and shoat (sheep and goats) operations. Sustaining a way of life, via land-use, might be more important than we initially presume, and this could have direct contributions to conservation. This can also be seen in the Southwestern US through a professionally connected group ranching system known as the Malpai Borderlands Group (MBG, 2015), where numerous stakeholders have collectively managed cattle integration with wildlife to preserve their way of life as a primary incentive above income generated through cattle operations.

There are two main land uses throughout Laikipia. Group ranching by pastoralists have the larger land holdings and contain large numbers of livestock, therefore have less wildlife. Communal ranches are smaller operations and contain more wildlife species and more wildlife diversity. Livestock management is generally the same in both categories, where there are no paddocks and limited fencing, allowing for the livestock to roam under watch. Additional research conducted by Sanderson *et al.* (2002) is of particular interest, where strategies for landscape species requirements in mixed land-use practices have been examined. In addition, the Laikipia Wildlife Forum (LWF, 2015) and the Lewa Wildlife Conservancy (LEWA, 2015) have substantial investment toward landscape-based community conservation. LWF is a regional effort to manage wildlife through engaging land-owners and land users, offering additional opportunity for research and training. Programs of LWF now involve community conservation, wildlife management, tourism, education, and security.

Aside from all the increasing negative attention the human dimensions of these conservation projects create, sound and systematic means for planning for community-based conservation need to remain an integral part of the Laikipia County. The historical truth is that livestock and wildlife can share the landscape at low livestock densities (ILRI, 2015), however the coexistence is not without conflict. Increases in cattle generally come at the expense of wildlife numbers. Balancing wildlife and livestock requires a clear understanding of both the positive and negative aspects of their interaction.

The Laikipia County is a good example of how motivated individuals can engender support and sympathy for local wildlife and find ways to connect to their economies so that there are clear and direct benefits to communities. Laikipia has attracted a number of impressive research scientists and conservation leaders, many of whom are using Laikipia conservation as a model toward similar efforts in other regions of tropical savanna habitat.

4.2 Utilization of geographic information systems and satellite technologies for landscape management.

There is relatively no debate within the conservation community that humans have a strong and largely negative impact on global ecosystems and ecological processes. We have long known about our high biodiversity consumption rates (Pauly and Christensen, 1995), appropriation of primary productivity (Vitousek, *et al.*, 1986), continued population increase and utilization of natural resources (McNeil, 2000; Wilson, 2002). Regardless of the plethora of information measuring human impacts and the call for solutions, it has been equally challenging to realize change at the political level (Soule and Terborgh, 1999). Though GIS and remote sensing have been part of monitoring and measuring land-use changes for decades (Skole and Tucker, 1993; Ramankutty and Foley, 1999), the development of verifiable maps conveying large scale changes to Earth were only first introduced in 2000 (Loveland *et al.*, 2000). Since then, the number of mapping technologies specializing in the impact of anthropogenic change on conservation planning have increased exponentially (Horning *et al.*, 2010; Rose *et al.*, 2014). With the launching of new satellites, such as the Landsat 8 from the European Space Agency (ESA), this is an ideal time to action new data from higher resolution sources into conservation planning.

A substantive effort to specifically measure the ecological footprint of humans was initiated by Sanderson *et al.* in 2002 under the leadership of the Wildlife Conservation Society (WCS). It recommends acknowledgment of the human footprint is a first step to be followed up by a commitment to preserve remaining wild places relatively untouched by humans. By using the human footprint maps, Earth's landscapes can be reviewed for landscape conservation value. It would then be up to decision makers, scientists, and various stakeholders to make a case for landscape protection. Version 2.0 of The Human Footprint is accessible on-line (Woolmer *et al.*, 2008) and reveals continued human influence across Kenya (Figures 4.3, 4.4).

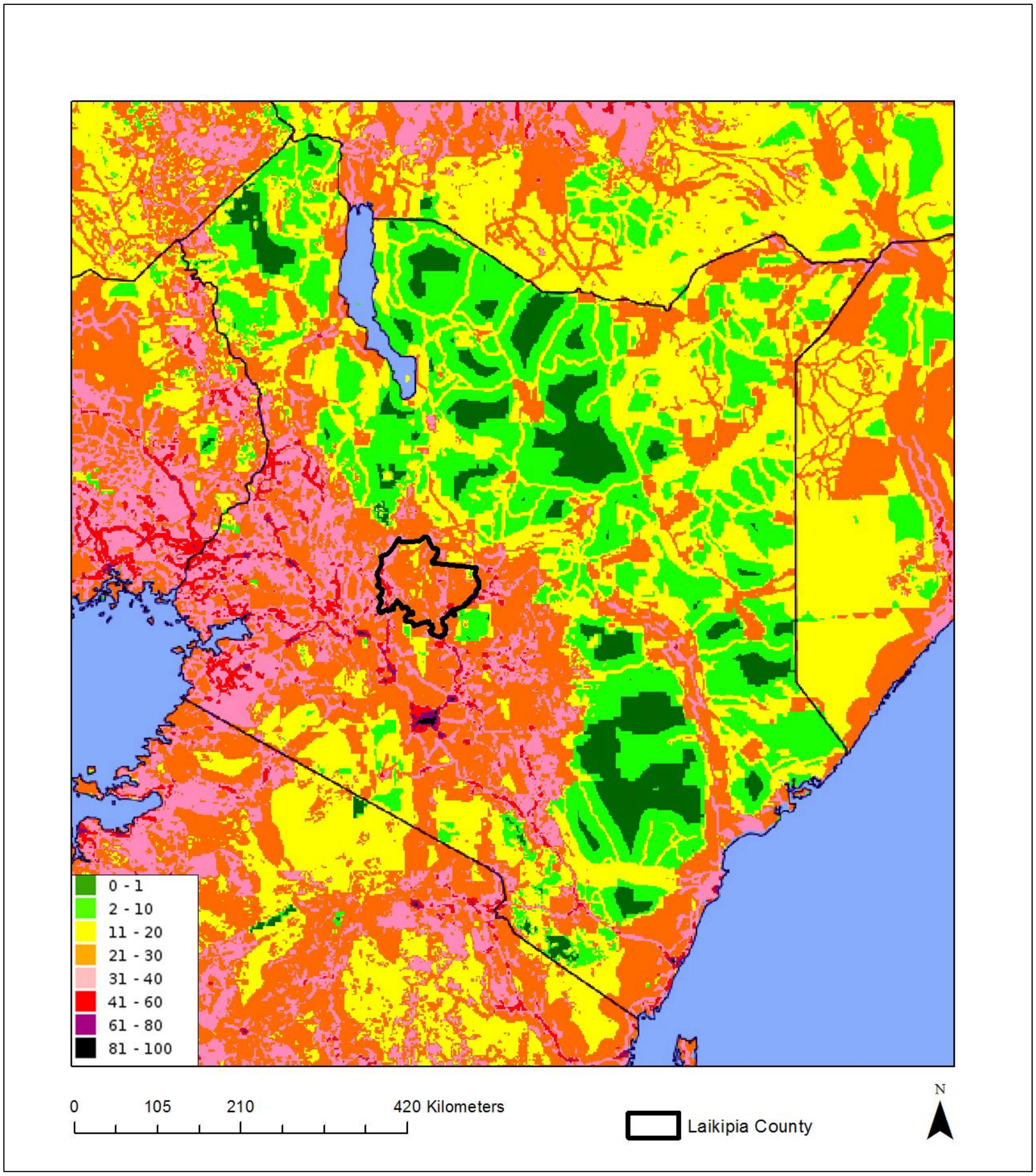


Figure 4.3. Georeferenced base image of the The Human Footprint overlaid with the administrative boundary of Kenya using the Human Footprint Index (HFI) as determined by updated research from Sanderson *et al.* (2008) of the relative human influence in each terrestrial biome. A value of 0 represents the least influenced (most wild) part of a biome with a value of 100 representing the most influenced (least wild) part of a biome.

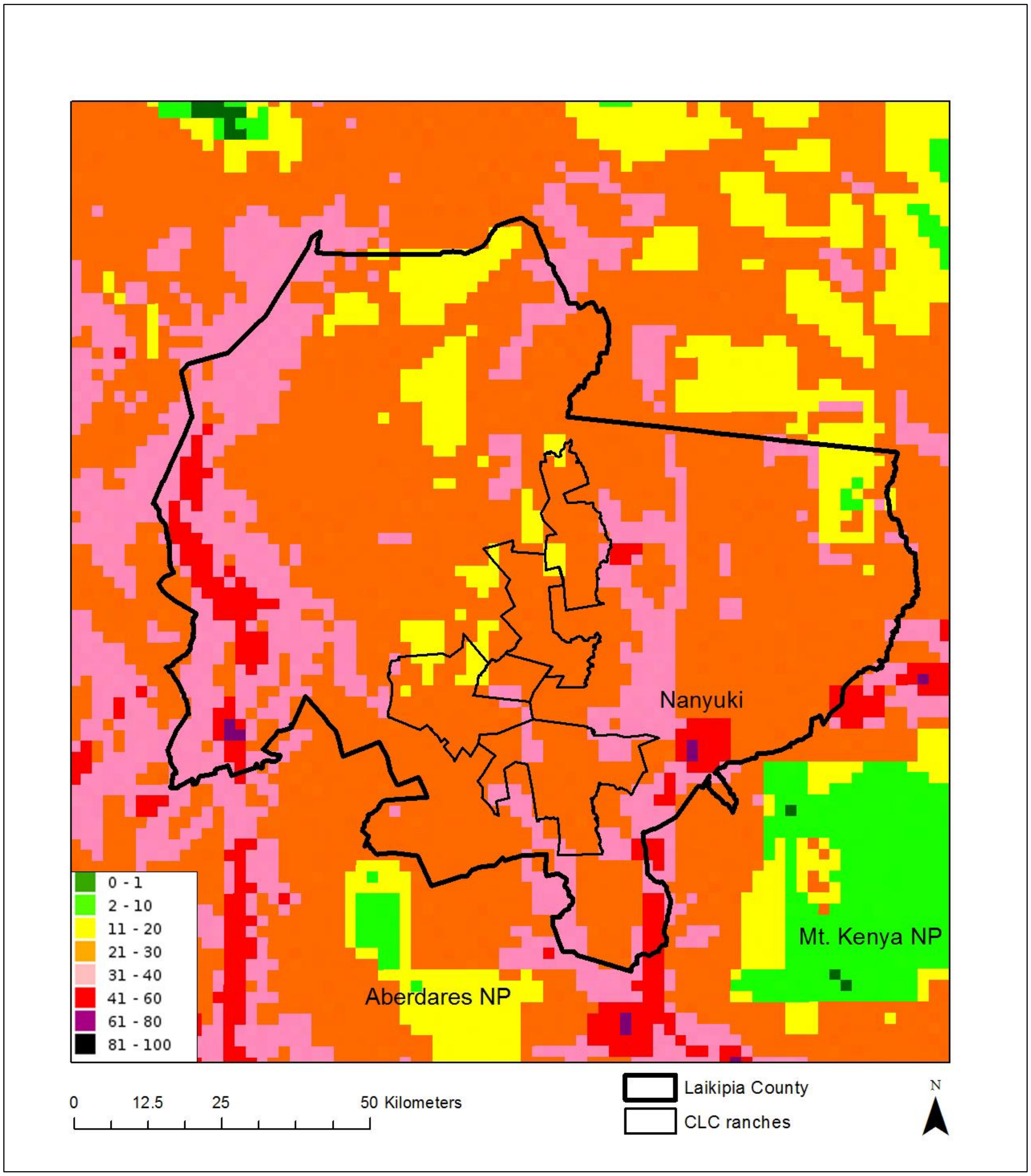


Figure 4.4. Georeferenced base image of the The Human Footprint overlaid with administrative boundaries of Laikipia County, Kenya and the CLC ranches using the Human Footprint Index (HFI) as determined by updated research from Sanderson *et al.* (2008) of the relative human influence in each terrestrial biome. A value of 0 represents the least influenced (most wild) part of a biome with a value of 100 representing the most influenced (least wild) part of a biome.

New geographic technologies are providing creative tools and resources for conservation science that make monitoring, measuring, and assessing the condition of biodiversity possible in ways and at scales never seen before. There are new methods for monitoring global fires (GFW, 2015), global climate change (USGS, 2015), changes in net primary productivity (Ardo, 2015), and planetary reflectance of solar radiation (Abtew and Malesse, 2015). We also have the ability to measure surface change in three dimensions, with tools such as lidar (Hansen, 2015) that provide estimation of total above ground biomass. At the organismal level, we have new geography-based tools to radio-track elephant across long distances using seismic sensors (Annia and Sangaiah, 2015), newly published research on using geolocators fitted onto small migratory birds (Hallworth *et al.*, 2015), a much improved system to estimate gene flow through large scale wildlife corridors via faecal analysis (Dutta *et al.*, 2013; Sharma *et al.*, 2013) and continued use of stable-carbon isotope analyses that contribute to large scale animal movement data (Reudink *et al.*, 2014).

There are two main ways in which new GIS technologies are helping to change conservation science. First, it is providing a level of detail about our planet we have not experienced before, from microscopic and molecular techniques to global scale remotely sensed data analyses at high resolution. Secondly, the amount of information now available to students, researchers, and citizens is unprecedented, with much of it freely accessible on-line. For conservation, this is a careful balance of having the tools, technologies, and data for research while at the cost that these same technologies and information can be used for poaching wildlife and causing harm to populations.

Collaborative efforts in sharing new technologies between property owners is a key to a sustainable future for wildlife conservation in Laikipia County, Kenya. Linking remote sensing and GIS resources with field data per ranch will improve conservation planning requirements across the greater landscape.

4.2 Utilization of camera trapping for monitoring and estimating wildlife abundance.

Camera trapping is a useful and standard method of gathering wildlife data through its ease of use and non-intrusive field application. An associated benefit is its ability to collect several layers of metadata simultaneously, such as time, date, and temperature. Many research scientists and students of landscape ecology and species-specific conservation biology have concluded that camera trapping is a reliable and compulsory field technique. In studying wildlife, camera-traps provide a non-intrusive method of capturing images, with added benefit of being ideal for documenting rare and/or elusive species (Rovero and Deluca, 2007; Kolowski and Alonso, 2010) or as a compulsory management tools for monitoring wildlife over time (O'Brien *et al.*, 2010). Popularized earlier by numerous studies of tigers (Karanth, 1995; Karanth and Nichols, 1998, 2002), camera traps are now commonplace for detecting endangered species (Sanderson and Trolle, 2005), calculating abundance and density estimations (O'Brien *et* *al.*, 2003; Rovero and Marshall, 2009; O'Brien and Kinnaird, 2011), conducting habitat association analyses (Bowkett *et al.*, 2008), establishing global camera networks (Ahumada *et al.*, 2011), and can act as an alarm system for livestock theft (Bauer *et al.*, 2005; O'Brien 2010).

Cameras eliminate the need for physical capture or handling of the target animals, can operate without maintenance for long periods of time, are digital, weather proof, and offer sensitivity settings for specific conditions. Due to ease in use and ability to gather photographic evidence, camera-trapping studies have also become common in a wide range of landscape-level research (Giman *et al.*, 2007; Rowcliff *et al.*, 2008; Trolle *et al.*, 2008; Kolowski and Alonso, 2010; Li et al, 2010) and data from these studies have now been collectively applied to survey studies that enable comparative analysis between design and analysis methodologies (Sanderson, 2004; Kelly and Holub, 2008; Tobler, *et al.*, 2008). The past 15 years of camera-trapping studies is reflected well in the significant increase in the number of published camera-trapping research efforts (Rowcliffe and Carbone, 2008; Sundaresan *et al.*, 2011; Rovero *et al.*, 2014).

Though its use today is widespread and relatively common, it is the design, application, and the associated software that is continually evolving. Many camera trapping studies focus on land management (parks, preserves, private landholdings) through use of multi-year data collected at several sites and at various time intervals (Kinnaird and O'Brien, 2012). These data are then compiled into one single database for use in science investigation. The unique quality of this research study is that multiple wildlife conservancies were camera trapped simultaneously with a systematic approach to assigning the location of cameras to specific habitat classifications shared throughout the landscape. This allows us to not only compare individual parcels of land to one another, but to look at the sub-ranch level for comparisons to understand the impact of habitat and anthropogenic influence between and among ranches.

Results of camera trapping effort are provided in Chapter 2 and 3 of this dissertation. During the study period, there were two rare images captured, one of a melanistic serval and another of a leopard cub (Figure 4.5). The importance of photodocumenting biodiversity through camera trapping has added value beyond species identification. Presence and absence studies, particularly of small carnivore species, have strong implications for management (Kays *et al.*, 2015). There were also species encountered in the field which were not recorded by camera traps, such as the wild dog (*Lycaon pictus*), and species such as the Besia oryx (*Oryx besia*) only recorded on one ranch but observed on all ranches while in the field.

Figure 4.5. Above image: Image capture of a melanistic serval in Acacia habitat on the Segera Ranch (left) as compared to a clearer image (right) recently photographed by Leslie Daniels (AWF, 2014). Below image: Image capture of a leopard cub in a dried river bed on the Eland Downs ranch (left) and further magnified (right).

The Ol Pejeta Conservancy (OPC) has long maintained park guards to monitor the condition and well-being of its increasing rhino population. OPC has also implemented the use of camera trapping as a directed technology, with a focus on photo-documenting its inventory of black rhino. OPC has hired photographers, reached out to volunteers, and has on occasion looked to camera-trapping to confirm a visual condition of each of the black rhinos. One of several valuable images from this study that supported OPCs effort at photo-documenting black rhino reveals a female with her calf (Figure 4.6), with the condition and growth of the calf previously unknown to the OPC rangers at that time. As seen in the image, the female rhino is ear notched for identification, which has not yet been implemented on the calf. Today, OPC has invested in a large array of camera traps as a reliable method of documentation and archival of their wildlife.

Figure 4.6. Image capture of a female black rhino with calf in riverine habitat on the Ol Pejeta Conservancy.

An added benefit to use of camera trapping on a wildlife conservancy in Laikipia, is that it may supply management with documentation of illegal access to the property by poachers. Camera trapping was successfully tested for its use in detecting poachers in Southeast Asia (Steinmetz *et al.*, 2014), and may similarly provide documentation on individuals who can later be identified by members of the local community (Figure 4.7). Since information on monitoring of park boundaries travels quickly in such a community, it will also be of great value for people to be aware that camera-trapping is in use and

Figure 4.7. Camera trapped image of a local community member with cattle illegally accessing the Segera Ranch.

may result in personal detection during any illegal access into the park. Steinmetz *et al.*, (2014) indicate that community outreach resulted in a sharp decline in poaching and that there was no correlations to poaching and park guard patrols.

4.3 Integration of on-line mapping technology for landscape level management.

Soon after the release of on-line mapping technologies in the early 1990s, there was a rapid increase in the number of map-based businesses such as vehicle navigation systems and basic static maps within websites during the dot com boom of 2002 (Appendix K). The use of on-line mapping made its way into academia and research in early 2000 for purposes of data referencing and repository, national atlas development, the integration of satellite technologies, and embracing an open source architecture with the GIS community to stimulate creativity. The most important achievements for on-line mapping in landscape conservation came when ESRI (now, Esri®) launched ArcIMS (Internet Map Server) 3.0 in 2000, the first publicly available on-line map service to work with Esri GIS software. From this point forward, users of ArcGIS could finally transition their resulting work into interactive maps on websites, blogs, or independently on-line. This was soon followed by the necessity to store the growing memory requirements for map information on-line, which gave way to the development of the Geography Network™, a reliable internet-based repository of geographic data that was sharable, accessible on-line, and encouraged community contributions. Google Maps® and Google Earth® began to dominate on-line mapping by the year 2005.

For wildlife conservation, on-line mapping achieved greater success after 2009 when ArcGIS Online™ was developed, a global public library for all realms of geospatial mapping where layers of information can be shared on-line and combined with user's personal data. The ability to mashup NGO community maps and source data made it far more likely to get accurate information as well as prevent redundancy. In the GIS community, one of the most concerning issues when taking on large GIS projects is to unknowingly replicate work already completed or in progress. Having access to and knowing about real-time GIS projects is crucial in the development stages, which is what ArcGIS Online provides. Equally important was the rise and development of the Society for Conservation GIS (SCGIS, 2015), which quickly expanded in the early 2000s and provided a unique forum for the international GIS community working in the natural sciences. With on-line technology from SCGIS, users are quite easily capable of discussing any GIS related topic with the benefit of access to information and data.

In the last five years, Esri developed its latest suite of on-line mapping tools called Story Maps™. Story Maps have added value to the sharing of geographic data by encouraging the creation of a geographic narrative that is accompanied by maps. In this case, Story Maps helps tell a story with a large selection of templates and editing tools to customize the how that story is visualized (Figure 4.8).

Figure 4.8. One example template of an Esri Story Map (image courtesy of Esri®), featuring a central, interactive map with tab options for additional maps (themes) and the ability to add text, video, and images to the story line (left hand side). The central map is highly interactive with full zoom capabilities and options to embed web material.

The combination of GIS, Esri Story Maps and remote sensing technologies with camera trapping data can provide the type of geospatial narrative needed for land managers to more effectively visualize their conservation planning. The general operation of most landscape-level conservation work is inherently geospatial. Protected areas mapping, establishing wildlife corridors, solving boundary disputes, and future land acquisition are a few examples of practical geospatial needs for conservation property owners and practitioners.

Esri Story maps and relevance to conservation planning in Laikipia, Kenya

To address the requirements of creating a Story Map for the needs of land managers in Laikipia, it will be important to review seven common questions necessary to address development of on-line mapping applications. 1. What is the geographic scope of the conservation landscape? For this study, I will choose to focus on the four ranches from which I have been conducting camera-trapping and vegetation surveys (Segera, Eland Downs, ADC Mutara, and Ol Pejeta) (Figure 1.8). 2. What will be the story? There are many directions in story development for conservation in Laikipia, but I will choose to focus on the chapters of my research and first tell the story of the ranches, habitat, and general anthropogenic structures across the landscape (roads, fences, water holes, human activity areas). It is important to showcase what data is available to the ranch managers and to use the Story Map technology to engage their interest to interact with the map on-line and to promote the general notion of sharing resources and information landscape-wide. From there, it will be up to the ranch managers to decide how best to portray the research data and information on-line, should they choose to do so. Some data are of sensitive nature and should be handled carefully. 3. What will be the cost to develop this Story Map? For purposes of this research, I will be using a personal account in the development of beta-version on-line maps for Laikipia. All conservancies in Laikipia can take advantage of the Esri Not-for-profit ArcGIS Online account status, which waives costs of software. A membership provides full access to ArcGIS Online and a full download of ArcGIS Pro software for typical GIS needs. 4. Where do the maps get stored in a Story Map? Since data ownership and sharing can be a strong concern for GIS enthusiasts, it is important that all stakeholders realize that all mapping data are hosted on-line through a repository known as the ArcGIS Online cloud. These data can be made available for sharing or kept private, it is entirely up to the user. The security of hosting authoritative, trusted data through Esri is important, as Story Maps does not require any server use from the client. All data and tools are on-line, with no need to download software to interact with Story Maps. 5. Can a map be made that is designed to share internally? Yes, it is an option to build, develop, and test proprietary data into on-line maps. By using email addresses, sharing with stakeholders is easy through members of an organization or any other stakeholders in Laikipia. There may be concerns for data that reveal locations of species, such as the black rhino, that will warrant internal-use only. These features are important in designing Story Maps for internal audiences for purposes of training, access, and information, and also for the general public to share the geography, infrastructure data, and wildlife distribution maps as a means to promote ecotourism. 6. How can the Story Map be promoted through communications? All Story Maps contain abbreviated web links, making sharing quite easy. All Story Maps provide html code that will allow a web designer to embed maps or display them from link in existing pages. In addition, all Story Maps are configured to work with all smart devices, so they will scale as necessary. Iphones and iPads are frequently used for tourists on the go, and this is an environment in which this level of accessibility is ideal. 7. How are Story Maps administered and managed? Through on-line accounts, Story Maps are easily administered where access, features, and tools are managed for an unlimited numbers of users. Esri offers full on-line technical support, in addition to the Esri Kenya office located in Nairobi.

In this study, I use the data gathered during research to build several Esri Story Maps to portray the landscape, wildlife, and anthropomorphic structures of the combined ranch system. These maps will remain in beta-test version and shared primarily with the Ol Pejeta Conservancy for their review. Feedback from OPC will help guide the process of how best to use the maps which largely feature substantial camera trapping data from OPC and the ADC Mutara Ranch.

The application of mapping technology to assist in supporting black rhino management on the ADC Mutara Conservation Area

Despite numerous attempts to ward off poaching and reduce alteration and loss of habitat, the world's black rhino (*Diceros bicornis michaeli*) population continues to decline both in size and range. Poaching has remained the primary cause of the decline (Walpole *et al.*, 2001; Rice and Jones 2006), though efforts to increase their numbers are focused on secure areas with available habitat (Lush et al, 2015). The population estimates for Africa were once in the hundreds of thousands nearly a century ago, but dropped to an alarming 398 individuals in 1991 (STR, 2015; Thuo *et al.*, 2015), a 97.6% decline from the 1960s (WWF, 2015). Successful conservation efforts to restore black rhino populations have enabled it to exceed 5,000 in 2015 (Save the Rhino Fund, 2015), with an increase in Kenya's population from 381 individuals in 1987 to a current estimate of over 650 (NRT, 2015). The species is listed as critically endangered by the International Union for the Conservation of Nature (IUCN), with the largest populations in Kenya residing on the Ol Pejeta and Lewa Conservancies, with smaller populations in national parks and sanctuaries.

Black rhinos are known to occupy a variety of habitat types in their current range (Dinerstein, 2011). Ecological modeling to further understand habitat requirements of the black rhino in Kenya have been studied (Lush *et al.*, 2015), and have identified that quality of habitat and browse availability are just as critical. For a proper browse availability study, a 3-dimensional perspective of vegetation in addition to its quality would be necessary. Lush *et al.* (2015) found that black rhino preferred *Scutia myrtina* trees (rare on OPC), followed by *Euclea divinorum* and *Acacia drepanolobium* respectively. In addition, black rhino were found to occupy mostly Acacia and Euclea habitat, which is similar to prior research efforts in Kenya (Rice and Jones, 2006).

The current population of black rhino on the Ol Pejeta Conservancy still remains just above 100 individuals, with consideration of the 20,000 acre ADC Mutara Conservation Area for possible expansion. Browse availability on OPC supporting black rhino has reached its carrying capacity (Mulama, 2013) and any increase in population numbers will need to be addressed through increasing habitat. Through vegetation surveys and GIS, I estimate a combined addition of 14,414 acres of Acacia (4,115 acres), Euclea (4,501 acres), and Mixed (5,798 acres) habitat that will be important for supporting black rhino on ADC (Table 2.3). The location and spatial arrangement of these habitat is made available through the on-line mapping tool that I have developed specifically for this purpose (Figure 4.10).

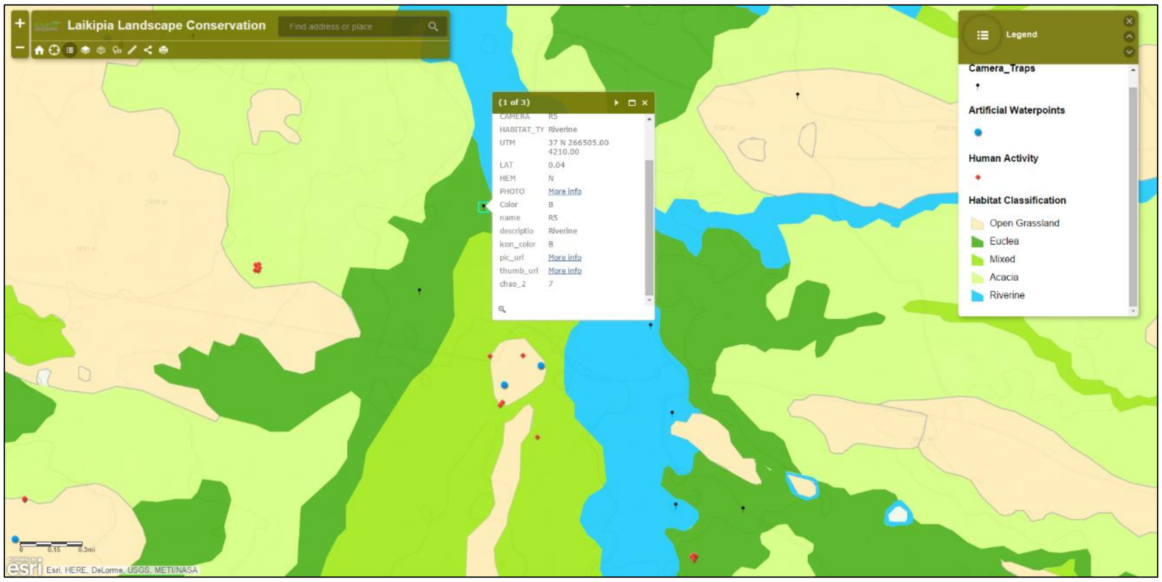


Figure 4.9. Example of beta-version of Story Map using the "basic" map template offering a detailed toolbar that includes a home button, user location on map, layer selection, base layer selection, details text box, measuring tools, sharing tools, and print options.

There are two beta versions of the on-line mapping application available for review by the Ol Pejeta Conservancy. First, I produced a general landscape map that features the administrative boundaries of the four ranches in this study along with locations of the camera traps and links to photos from each camera (Figure 1.9). It is the intention of this tool to demonstrate that future technology will allow instant uploading of images from camera traps to a centralized database to be viewed at any time. This would give land managers a real-time perspective of images captured, not just of wildlife but of any illegal human and cattle access. This interactive map is meant for a general perusal of the landscape and a quick overview of administrative boundaries.

The second mapping application I developed (Figure 4.10) will feature far more technical tools for exploring the landscape in more detail. This will be developed under a "basic" Story Map template (Esri, 2015b) that has been customized for these data. This application features a toolbar with several important functions tailored for a land manager (Table 4.1).

Table 4.1. Overview of basic tools available on the beta-version interactive mapping application adopting a customized Esri basic template.

Tool	Function
Home button	Will bring the user back to the default screen
Position location	Indicates the location of the user in relation to the map
Layers	Allows the user to select or deselect layers
Base maps	Offers a selection of base map layers
Overview map	Provides a pop up overview map of larger landscape
Measuring tools	Tools for measuring distance, points, and area
Sharing	Sharing options by URL, social media, or map copy
Print	Printing options at various scales

The effectiveness of close public-private partnerships among conservation areas in Laikipia will continue to lead to its success. With primary examples from the Laikipia Wildlife Forum and the African Wildlife Foundation, the sharing of information and resources among land owners has shown to be increasing with positive results. New tools visualized through on-line mapping is a key to a sustainable future for Laikipia's wildlife conservation efforts. Easily accessbile and available to share, these tools can assist in training, workshops, tourism communications, and general communication between ranch managers.

Appendix A

List of herbivorous mammal species recorded among all properties in the study

Common name by family	Genus and species
PROBOSCIDEA	
African Bush Elephant	*Loxodonta africana*
PERISSODACTYLA	
White rhinoceros*	*Ceratotherium simum*
Black rhinoceros	*Diceros bicornis*
Grevy's zebra	*Equus grevyi*
Common zebra	*Equus quagga*
ARTIODACTYLA	
Impala	*Aepyceros melampus*
Jackson's hartebeest	*Alcelaphus buselaphus jacksoni*
Thompson's gazelle	*Eudorcas thomsonii*
Giraffe	*Giraffa camelopardalis*
Hippopotamus	*Hippopotamus amphibius*
Waterbuck	*Kobus ellipsiprymnus*
Guenther's dik-dik	*Madoqua kirkii*
Grant's gazelle	*Nanger granti*
Beisa oryx	*Oryx beisa*
Common warthog	*Phacochoerus africanus*
Bushpig	*Potamochoerus larvatus*
Steinbuck	*Raphicerus campestris*
Bush duiker	*Sylvicapra grimmia*
Cape buffalo	*Syncerus caffer*
Eland	*Taurotragus oryx*
Bushbuck	*Tragelaphus sylvaticus*

* not native to Kenya.

Appendix B

List of small mammal (rodent) species recorded at both the Ol Pejeta Conservancy (Ol Pejeta Conservancy, 2012) and the Mpala Ranch (Young, 2010).

Name	**Genus and species**
Spiny mouse	*Acomys sp.*
Rock mouse	*Aethomys hindei*
Grass rat	*Arvicanthus nairobea*
Water rat	*Dasymys sp.*
Climbing mouse	*Dendromus melanotis*
Woodland mouse	*Grammomys dolichurus*
Dormouse	*Graphiurus murinus*
Porcupine	*Hystrix galeata*
Multimammate mouse	*Mastomys sp.*
Nutria / coypu*	*Myocastor coypus*
Pygmy mouse	*Mus musculoides*
Tree squirrel	*Paraxerus ochraceus medici*
Pouched mouse	*Saccostomus mearnsi*
Gerbil	*Tatera sp.*
Side-striped ground squirrel	*Xerus erythropus*

* *not native to Kenya*

Appendix C

List of large carnivore species observed at both the Ol Pejeta Conservancy (Ol Pejeta Conservancy, 2013) and the Mpala Ranch (Mpala Research Centre, 2010).

Common name	**Genus and species**
Cheetah	*Acinonyx jubatus*
Spotted hyena	*Crocuta*
Striped hyena	*Hyaena*
Wild dog	*Lycaon pictus*
Leopard	*Panthera pardus*
Aardwolf	*Proteles cristatus*

Appendix D

List of small carnivore species observed and recorded at both the Ol Pejeta Conservancy (Ol Pejeta Conservancy, 2012) and the Mpala Ranch (Mpala Research Centre, 2010).

Common name	**Genus and species**
Cape clawless otter	*Aonyx capensis*
Water mongoose	*Atilax paludinosus*
Marsh mongoose	*Atilax paludinosus*
Side-striped jackal	*Canis adustus*
Black-backed jackal	*Canis mesomelas*
Civet	*Civetta*
Caracal	*Felis caracal*
Wild Cat	*Felis lybica*
Serval	*Felis serval*
Small-spotted genet	*Genetta*
Large-spotted genet	*Genetta tirgrina*
Dwarf mongoose	*Helogale parvula*
Slender mongoose	*Herpested (Galerella) sanguineus*
White-tailed mongoose	*Ichneumia albicauda*
Zorilla	*Ictonyx striatus*
Ratel / Honey Badger	*Mellivora capensis*
Bat-eared fox	*Otocyon megalotis*

Appendix E

List of wildlife species per guild.

CARNIVORES	
Bat-eared Fox	*Otocyon megalotis*
Black-backed Jackal	*Canis mesomelas*
Caracal	*Caracal*
Cheetah	*Acinonyx jubatus*
Common genet	*Genetta*
Honey Badger	*Mellivora capensis*
Leopard	*Panthera pardus*
Lion	*Panthera leo*
Serval	*Leptailurus serval*
Slender Mongoose	*Helogale parvula*
Spotted Hyena	*Crocuta*
Striped Hyena	*Hyaena*
White-tailed Mongoose	*Ichneumia albicauda*
Zorilla	*Ictonyx striatus*
GRAZERS	
Cape Buffalo	*Syncerus caffer*
Common Zebra	*Equus quagga*
Coypu (nutria)	*Myocastor coypus*
Grevy's Zebra	*Equus grevyi*
Hippopotamus	*Hippopotamus amphibius*
White Rhinoceros	*Ceratotherium simum*
BROWSERS	
Beisa Oryx	*Oryx beisa*
Black Rhinoceros	*Diceros bicornis*
Camel	*Camelus sp.*
Giraffe	*Giraffa camelopardalis*
Grant's Gazelle	*Nanger granti*
Guenther's Dik-dik	*Madoqua kirkii*
Jackson's Hartebeest	*Alcelaphus buselaphus jacksoni*
Scrub Hare	*Lepus saxatilis*

Thompson's Gazelle	*Eudorcas thomsonii*
Vervet Monkey	*Chlorocebus pygerythrus*
Waterbuck	*Kobus ellipsiprymnus*

GRAZER/BROWSER

Bush Duiker	*Sylvicapra grimmia*
Bushbuck	*Tragelaphus sylvaticus*
Cattle	*Bos sp.*
Crested Porcupine	*Hystrix cristata*
Eland	*Taurotragus oryx*
Elephant	*Loxodonta africana*
Goat	*Capra sp.*
Impala	*Aepyceros melampus*
Sheep	*Ovis sp.*
Steinbuck	*Raphicerus campestris*

OMNIVORE

Bushpig	*Potamochoerus larvatus*
Common Warthog	*Phacochoerus africanus*
Olive Baboon	*Papio anubis*
Dog	*Canis lupus familiaris*

OTHER / INSECTIVORE

Aardvark	*Orycteropus afer*
Kori bustard	*Ardeotis kori*
Lesser Bushbaby	*Galago senegalensis*
Ostrich	*Struthio camelus*

Appendix F

Vegetation species list

Trees	Shrubs	Grasses
Acacia drepanolobium	*Abutilon sp.*	*Aristida congesta*
Acacia mellifera	*Acalypha crenata*	*Aristida kenyensis*
Acacia nilotica	*Aerva lantana*	*Bothriochloa insculpta*
Acacia xanthophloea	*Asparagus africana*	*Brachiaria lachnantha*
Boscia sp.	*Asparagus racemosa*	*Chloris plectostachyum*
Euclea divinorum	*Aspilia pluriseta*	*Cymbopogon sp.*
	Balanites glabra	*Cynodon dactylon*
	Caesalpinia decapetala	*Digitaria scalarum*
	Carissa edulis	*Eragrostis chalcantha*
	Commelina sp.	*Eragrostis superba*
	Dyschoriste radicans	*Eragrsotis tenuifolia*
	Erythrococca bogensis	*Hypachne schimperi*
	Grewia similis	*Lintonia nutans*
	Gymnphocarpus semilunatus	*Microchloa caffra*
	Hibiscus sp.	*Microchloa kunthii*
	Indigofera bogdanii	*Panicam maximum*
	Indigofera brevicalyx	*Pennisetum mezianum*
	Indigofera schimperi	*Pennisetum stramenium*
	Lycium shawii	*Rhynchelytrum roseum*
	Maerua triphylla	*Setaria sphacelata*
	Maytenus senegalensis	*Sporobolis africanana*
	Maytenus senegalensis	*Sporobolus discosporus*
	Pseudognaphalium sp.	*Sporobolus pyramidalis*
	Psidia punculata	*Themida triandra*
	Psilotrichum schimperi	
	Rhamnus staddo	
	Rhinacanthus ndorensis	

Rhus natalensis
Rhus vulgaris
Rhynchosia holstii
Scutia myrtina
Sericocomopsis pallida
Sida cuneifolia
Sida Schimperiana
Solanum indicum
Teclea simplicifolia

Appendix G

Vegetation survey statistics

Ranch		habtiat calssification				
		Acacia	**Euclea**	**Grasslands**	**Mix**	**Riverine**
SEG	Grass	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*
	% cover	72.99	61.77	75.02	41.45	36.89
	height	56.44	66.17	73.62	41.99	25.09
	Tree	*A. drep*	*A. drep*	*A. drep*	*A. drep*	*A. drep*
	% cover	12	45	0	24	7
	height	478	484	0	450	401
ED	Grass	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*
	% cover	62.01	59.33	53.12	30.83	39.16
	height	31.73	40.81	20.8	53.61	34.33
	Tree	*A. drep*	*A. drep*	*A. drep*	*A. drep*	*A. drep*
	% cover	9	40	0	20	3
	height	427	430	0	420	450
ADC	Grass	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*
	% cover	73.85	62.11	70.9	43.83	41.01
	height	42.87	42.88	34.13	30.83	33.39
	Tree	*A. drep*	*A. drep*	*A. drep*	*A. drep*	*E. divinorum*
	% cover	45	49	2	24	5
	height	480	473	400	440	515
OPCW	Grass	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*
	% cover	66.81	53.39	45.21	41.09	45.33
	height	29.81	65.01	29.71	31.92	32.56
	Tree	*A. drep*	*A. drep*	*A. drep*	*A. drep*	*E. divinorum*
	% cover	22	45	3	34	30
	height	476	434	415	476	460
OPCE	Grass	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*	*T. triandra*
	% cover	78.13	49.55	57	48.19	31.43
	height	34.89	63.22	37.99	33.87	32.91
	Tree	*A. drep*	*A. drep*	*A. drep*	*A. drep*	*A. Xanth*
	% cover	19	75	2	53	25
	height	438	480	411	477	2187

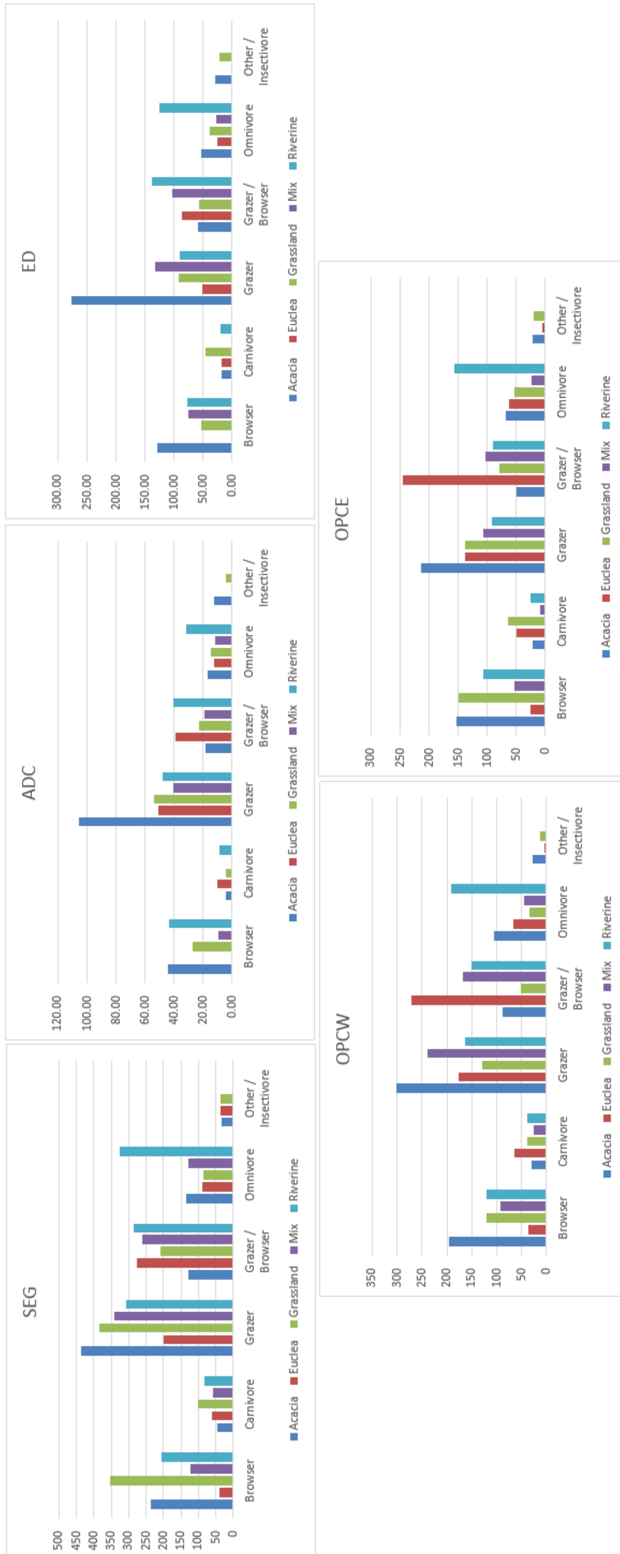
Appendix H

Summary of top relative abundance estimates of species per ranch

SEGERA		ELAND DOWNS		ADC MUTARA		Ol Pejeta W		Ol Pejeta E		OPC TOTAL	
Cattle	29.41	Common zebra	26.60	Common zebra	42.26	Common zebra	36.40	Impala	36.43	Common zebra	27.92
Human	19.79	Human	21.81	Buffalo	13.58	Impala	13.97	Olive baboon	25.58	Impala	24.91
Elephant	17.65	Elephant	14.89	Impala	9.43	Buffalo	12.13	Buffalo	25.19	Buffalo	18.49
Common zebra	17.11	Goat	12.77	Cattle	7.92	Cattle	10.66	Common zebra	18.99	Olive baboon	14.91
Rodent	8.02	Shoats	8.51	Reticulated giraffe	6.79	Warthog	10.29	Warthog	14.34	Warthog	12.26
Impala	7.49	Thompson's Gazelle	6.91	Grant's Gazelle	6.04	Thompson's Gazelle	9.93	Grant's Gazelle	13.95	Black Rhino	7.92
Vervet Monkey	6.42	Grant's gazelle	5.85	Human	6.04	Eland	5.88	Waterbuck	13.95	Waterbuck	7.74
Shoats	5.88	Steinbuck	5.32	Steinbuck	6.04	Reticulated giraffe	5.51	Black Rhino	12.02	Grant's Gazelle	6.79
Grant's Gazelle	5.35	Jackal	4.79	Olive baboon	5.66	Elephant	5.15	Reticulated giraffe	8.14	Reticulated giraffe	6.79
Steinbuck	5.35	Reticulated giraffe	4.79	Vervet Monkey	5.28	Bushbuck	4.78	Lion	5.81	Thompson's Gazelle	6.04
Eland	4.28	Impala	4.26	Warthog	5.28	Olive baboon	4.78	Jackal	3.49	Cattle	5.66
Warthog	4.28	Camel	3.72	Thompson's Gazel	4.15	Black Rhino	4.04	Eland	3.10	Eland	4.53
Dog	3.21	Bush duiker	3.19	Eland	3.40	Jackal	3.68	White-tailed Mong	3.10	Elephant	3.96
Buffalo	2.67	Kori Bustard	3.19	Bush Duiker	3.02	White-tailed Mongoo	2.94	Elephant	2.71	Jackal	3.58
Jackal	2.67	Spotted Hyena	3.19	Elephant	2.64	Hippopatamus	1.84	Jackson's Hartebee	2.33	White-tailed Mongoose	3.02
Reticulated giraffe	2.67	White-tailed Mongoos	2.66	Bush Baby	1.89	Waterbuck	1.84	Thompson's Gazell	1.94	Bushbuck	2.83
Spotted Hyena	2.14	Cattle	2.13	Kori Bustard	1.89	Spotted Hyena	1.47	Cheetah	1.55	Lion	2.83
Bush Duiker	1.60	Olive baboon	2.13	Scrub Hare	1.89	Porcupine	1.10	Bush Duiker	0.78	Hippopatamus	1.13
Grevy's zebra	1.07	Warthog	2.13	Bushbuck	1.51	White Rhino	1.10	Bushbuck	0.78	Jackson's Hartebeest	1.13
Kori Bustard	1.07	Vervet Monkey	1.60	Striped Hyena	1.51	Genet	0.74	Human	0.78	Spotted Hyena	1.13
Olive baboon	1.07	Buffalo	1.06	Waterbuck	1.13	Human	0.74	Scrub Hare	0.78	Cheetah	0.75
Scrub hare	1.07	Nutria	1.06	White-tailed Mon	1.13	Leopard	0.74	Spotted Hyena	0.78	Human	0.75
Serval	1.07	Ostrich	1.06	Goat	0.75	Steinbuck	0.74	Steinbuck	0.78	Steinbuck	0.75
Striped Hyena	1.07	Scrub Hare	1.06	Jackal	0.75	Aardvark	0.37	Aardvark	0.39	White Rhino	0.75
Bushpig	0.53	Bat-eared fox	0.53	Aardvark	0.38	Grevy's zebra	0.37	Cattle	0.39	Genet	0.57
Guenther's dikdik	0.53	Eland	0.53	Camel	0.38	Kori Bustard	0.37	Genet	0.39	Porcupine	0.57
Leopard	0.53	Honey badger	0.53	Dog	0.38	Slender Mongoose	0.37	Hippopatamus	0.39	Aardvark	0.38
Lion	0.53	Leopard	0.53	Guenther's dikdik	0.38			Ostrich	0.39	Bush Duiker	0.38
Waterbuck	0.53	Striped Hyena	0.53	Lion	0.38			Striped Hyena	0.39	Leopard	0.38
White-tailed Mongoos	0.53	Zorilla	0.53	Serval	0.38			White Rhino	0.39	Scrub Hare	0.38
				Slender Mongoos	0.38			Zorilla	0.39	Grevy's zebra	0.19
				Spotted Hyena	0.38					Kori Bustard	0.19
										Ostrich	0.19
										Slender Mongoose	0.19
										Striped Hyena	0.19
										Zorilla	0.19

Appendix I

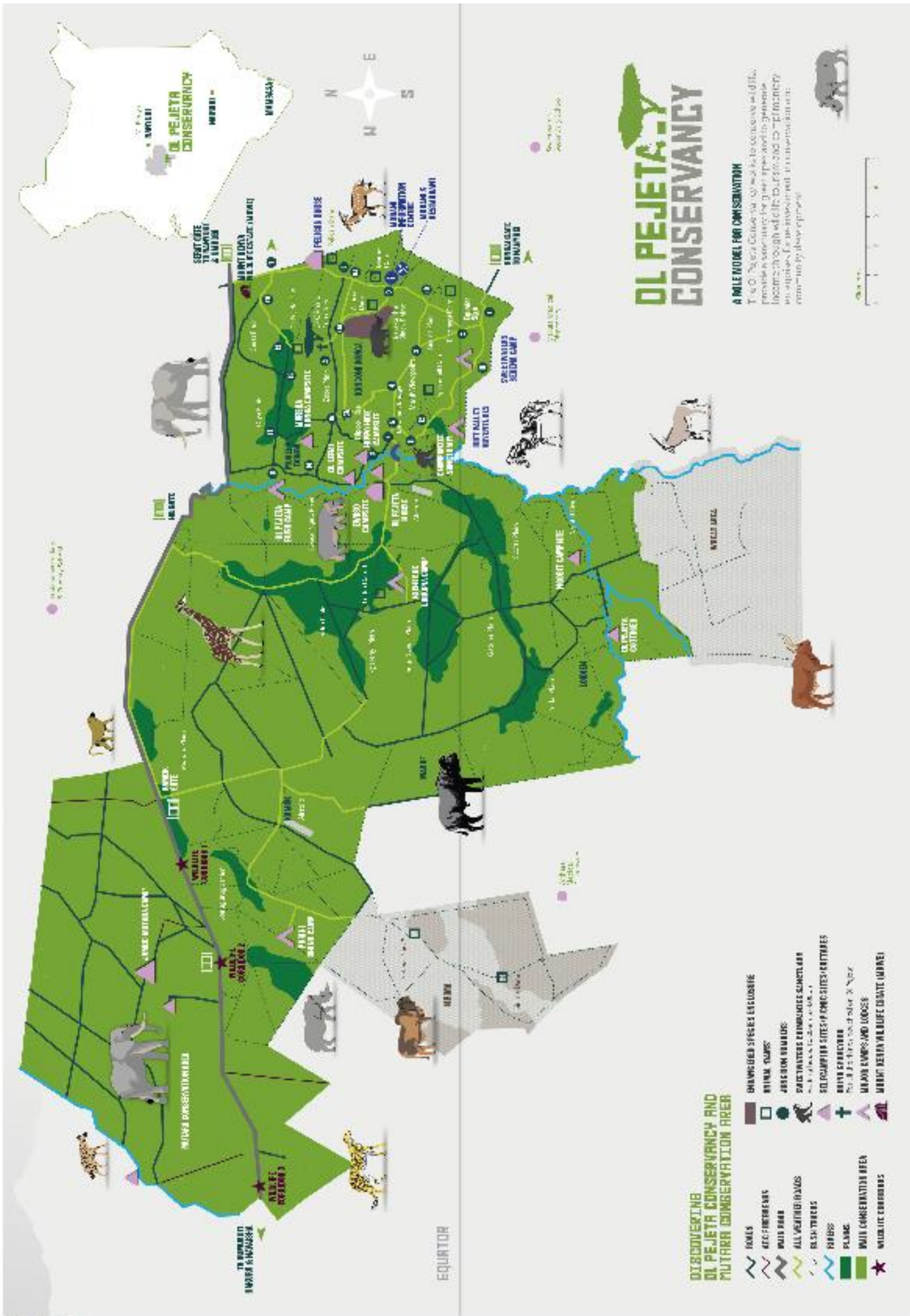
Relative abundance of species per guild and habitat by ranch.



ED
300.00
250.00
200.00
150.00
100.00
50.00
0.00
Browser
Carnivore
Grazer
Grazer / Browser
Omnivore
Other / Insectivore
Acacia
Euclea
Grassland
Mix
Riverine
ADC
120.00
100.00
80.00
60.00
40.00
20.00
0.00
SEG
500
450
400
350
300
250
200
150
100
50
0
OPCE
350
300
250
200
150
100
50
0
OPCW

Appendix J

Map of the Ol Pejeta Conservancy, including the ADC Mutara conservation area.


OL PEJETA CONSERVANCY
DISCOVERING
OL PEJETA CONSERVANCY AND
MUTARA CONSERVATION AREA
EQUATOR

Appendix K

Development timeline of notable achievements in on-line mapping

Date	Activity
1969	An internet transaction is successfully tested between University of California and Stanford
1985	TeleAtlas is developed from the Netherlands
1986	First navigation system for automobiles
1989	First websites go online; Internet is "born"
	Development of the World Wide Web (WWW)
1993	Xerox PARC Map Viewer established; first mapserver
1994	First on-line atlas developed: The National Atlas of Canada
1995	First interactive, on-line mapping: The Gazetteer for Scotland.
1996	Founding of MapQuest. First location service for addresses and navigation.
1997	USGS is mandated to develop the US Online National Atlas Initiative
	University of Minnesota MapServer 1.0 designed to deliver remotely sensed data
1998	USGS, Microsoft, and Hewlett Packard launch US Terraserver
	ESRI launches MapObjects, as a first entry to the on-line mapping world
2000	ESRI launches ArcIMS 3.0 (Internet Map Server)
	ESRI develops the Geography Network for sharing data and services
2002	Dot com boom; MicroSoft launches MapPoint
2004	First dragable maps come on-line
2005	Mapping API becomes freely available on-line; Google surges

	Google Maps and Google Earth developed
2006	MapQuest free API becomes available
2007	Google Maps / Google Earth rise
2008	Google base data becomes available; Streetview
2009	ESRI ArcGIS.com is developed to produce a global on-line mapping library
	Google surpasses MapQuest
2010	On-line mapping becomes a known science; development sharing begins
2011	ESRI and Google dominate on-line mapping
	Start of Esri Story Maps

Literature Citations

Abtew, W. and A. M. Melesse. 2015. Landscape changes impact on regional hydrology and climate. *In*. Landscape Dynamics, Soils, and Hydrological Processes in Varied Climates. Springer Geography. Pp:31-50.

African Wildlife Foundation. 2010. Commemorative 2010 Annual report. Pp26.

African Wildlife Foundation. 2011. Cattle and economics management program. Accessed from website: http://www.awf.org/economic/livestock-management

Ahumada, J. A., J. Hurtado and D. Lizcano. 2013. Monitoring the status and trends of tropical forest terrestrial vertebrate communities from camera trap data: a tool for conservation. Public Library of Science (PLoS ONE). 8:373707.

Ahumada, J. A., C. E. F. Silva, K. Gajapersad, C. Hallam, J. Hurtado, E. Martin, A. McWilliam, B. Mugerwa, T. O'Brien, F. Rovero, D. Shell, W. R. Spironello, N. Winari, and S. J. Andelman. 2011. Community structure and diversity of tropical forest mammals: data from a global camera trap network. Philosophical Transactions of the Royal Society B. 366:2703-2711.

Akama, J. 1999. The evolution of tourism in Kenya. Journal of Sustainable Tourism. 7(1):6-25.

Aklilu, Y., Irungu, P, and A. Reda. 2002. An audit of the livestock marketing status in Kenya, Ethiopia and Sudan. Community based animal health and participatory epidemiology unit (CAPRE) PACE Program. OAU_IBAR. Pp.1-85.

Ament, R. J., A. P. Clevenger, O. Yu, and A. R. Hardy. 2007. An assessment of road impact to wildlife populations inside national parks. Western Transportation Institute College of Engineering Montana State University WTI-MSU Technical Report REP-07-01.

Annia, J. S. and A. K. Sangaiahb. 2015. Elephant tracking with seismic sensors: a technical perceptive review. Jurnal Teknologi. 74(1):193–203.

Anderson, D. R. 2008. Model based inference in the life sciences: a primer on evidence. Springer, New York.

Archer, S. 1995. Tree-grass dynamics in a Prosopis – thornscrub savanna parkland: reconstructing the past and predicting the future. Ecoscience. 2:83-99.

Ardo, J. 2015. Comparison between remote sensing and a dynamic vegetation model for estimating terrestrial primary production of Africa. Carbon Balance and Management. 10(8):1-15.

Arnold, T. W. 2010. Uninformative parameters and model selection using Akaike's Information Criterion. Journal of Wildlife Management. 76(6):1175-1178.

Augustine, D. J. 2003. Long term, livestock-mediated redistribution of nitrogen and phosphorous in an east African savanna. Journal of Applied Ecology. 40:137-149.

Augustine, D. J. 2004. Influence of cattle management on habitat selection by impala on central Kenyan rangeland. Journal of Wildlife Management. 68(4):916-923.

Augustine, D. J. 2010. Response of native ungulates to drought in semi-arid Kenyan rangeland. African journal of ecology. 48(4):1009-1020.

Augustine, D. J., T. P. Young, K. E. Veblen, C. Riginos and J. R. Goheen. 2011. Pathways for positive cattle-wildlife interactions in semi-arid rangelands. Smithsonian Contributions to Zoology; Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem. 55-71.

Avenant, N. L. 2000. Small mammal community characteristics as indicators of ecological disturbance in the Willem Pretorious Nature Reserve, Free State, South Africa. South African Journal of Wildlife Research. 30:26-30.

Bailey, L. L., J. E. Hines, J. D. Nichols, and D. I. MacKenzie. 2007. Sampling design trade-offs in occupancy studies with imperfect detection: examples and software. Ecological Applications. 17(1):281-290.

Baker, J. P., D. W. Hulse, S. V. Gregory, D. White, J. V. Sickle, P. A. Berger, D. Dole, and N. H. Schumaker. 2004. Alternative futures for the Williamette River Basin, Oregon. Ecological Applications. 14(2):313-324.

Barnes, J. 2013. The ghosts of happy valley. Aurum Press Limited. London.

Bauer, J. W.. K. A. Logan, L. L. Sweanor, and W. M. Boyce. 2005. Scavenging behavior in puma. The Southwestern Naturalist. 50:466-471.

Beal, W. E., D. R. Notter and R. M. Akers. 1990. Techniques for estimation of milk yield in beef cows and relationship of milk yield to calf weight gain and postpartum reproduction. Journal of Animal Science. 68:937-947

Bennett, A. F. 2003. Linkages in the Landscape: The Role of Corridors and Connectivity in Wildlife Conservation. Gland, Switzerland: IUCN.

Bergstrom, B. J. 2013. Would East African savanna rodents inhibit woody encroachment? Evidence from stable isotopes and microhistological analysis of feces. Journal of Mammalogy. 94:436-447.

Blaum, N., E. Rossmanith, M. Schwager, and F. Jeltsch. 2007. Response of mammalian carnivores to land use in arid savanna rangelands. Basic and Applied Ecology. 8(6):552-564.

Booth, A. D., K. Szpakowska, E. Pschikova and K. Griffin. 2015. Structure of an ancient Egyptian tomb inferred from ground-penetrating radar imaging of deflected overburden horizons. Archaeological Prospection. 22(1):33-44.

Boran Cattle Breeders Society. 2015. http://www.borankenya.org/

Bowkett, A. E., F. Rovero, and A. R. Marshall. 2007. The use of camera-trap data to model habitat use by antelope species in the Udzungwa Mountain forests, Tanzania. African Journal of Ecology. 46(4):1-9.

Brody, A. K., T. M. Palmer, K. Fox-Dobbs, and D. F. Doak. 2010. Termites, vertebrate herbivores, and the fruiting success of *Acacia drepanolobium*. Ecology. 91:399-407.

Burnham, K. P. and D. R. Anderson. 1998. Model selection and multimodal inference: a practical information-theoretic approach. 2nd Edition. Springer. Ft. Collins, Co.

Burton, A. C., M. K. Sam, C. Balangtaa and J. S. Brashares. 2012. Hierarchical multi-species modeling of carnivore responses to hunting, habitat and pretty in a West African protected area. Public Library of Science (PLoS ONE). 7:e38007.

Carter, E., W. M. Adams and J. Hutton. 2008. Private protected areas: management regimes, tenure arrangements and protected area categorization in East Africa. Oryx. 42:177-186.

Chamain-Jammes, S., H. Fritz, and F. Murindagomos. 2006. Spatial patterns of NDVI-rainfall relationship at the seasonal and inter-annual time scales in an African savanna. International Journal of Remote Sensing. 27(23):5185-5200.

Chan, K. M. A., M. R. Shaw, D. R. Cameron, E. C. Underwood, G. C. Dailey. 2006. Conservation Planning for Ecosystem Services. Public Library of Science (PLoS ONE) Biology. 4(11):2138-2152.

Chandler, M., D. P. Bebber, S. Castro, M. D. Lowman, P. Muoria, N. Oguge, and D. I Rubenstein. 2012. International citizen science: making the local global. Frontiers in Ecology and the Environment 10: 328-331.

Chao, A. 1984. Non-parametric estimation of the number of classes in a population. Scandanavian journal of Statistics. 11:265-270.

Chao, A. W. Hwang, Y. Chen, and C. Kuo. 2000. Estimating the number of shared species in two communities. Statistica Sinica. 10:227-246.

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

Clark, B. 2015. Earthpiont.us. Online measuring tools. Accessed from website: www.earthpoint.us

Clevenger, P. A., B. Chruszcz and K. E. Gunson. 2001. Highway mitigation fencing reduces wildlife-vehicle collisions. Wildlife Society Bulletin. 29(2):646-653.

CNES. 2015. Centre national d'études spatiales (CNES). SPOT Satellite Imagery. https://cnes.fr/fr

Coe, M. J., D. H. M. Cummings, and J. Phillipson. 1976. Biomass and production of large African herbivores in relation to rainfall and primary production. Oecologia. 22:341-354.

Cole, M. M. 1986. The Savannas. Biogeography and geobotany. Academic Press, London.

Collins, S. L., S. M. Glenn, and D. J. Gibbons. 1995. Experimental analysis of intermediate disturbance and initial floristic composition: decoupling cause and effect. Ecology. 76:486-492.

Collinson, R. F. H. 1983. Management for improved veld conditions and production on game ranches. Proceedings for the 4th Hiabisa Soil Conservation Symposium.

Colwell, R. K. 2013. EstimateS. User Manual. Accessed from website: http://viceroy.eeb.uconn.edu/estimates/

Connell, J. H. 1978. Diversity in tropical rainforests and coral reefs. Science. 199(4335):1302-1310.

Constanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. The value of the world's ecosystem services and natural capital. Nature. 387:253-260.

Creel, S., B. Dantzer, W. Goymann, and D. Rubenstein. 2013. The ecology of stress: effects of the social environment. Functional Ecology. 27: 66-80.

Creel, S. M. S. Becker, S. M. Durant, J. M'Soka, W. Matandiko, A. J. Dickman, D. Christianson, *et al.*, 2013. Conserving large populations of lions – the argument for fences has holes. Ecology Letters. 16(11):1413.

Crooks K. R. and M. E. Soule. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature. 400:563-566.

Curtin, C. G. 2015. The science of open spaces. Theory and practice for conserving large complex systems. Island Press, Washington, D.C.

Dannenberg, A. L., R. J. Jackson, H. Frumkin, R. A. Schieber, M. Pratt, and C. Kochtitzky. 2003. The Impact of Community Design and Land-Use Choices on Public Health: A Scientific Research Agenda. American Journal of Public Health. 93:1500-1508.

De Leeuw, J., M. N. Waweru, O. O. Okello, M. Maloba, P. Nguru, M. Y. Said, H. M. Aligula, I. M. A. Heitkonig, and R. S. Reid. 2001. Distribution and diversity of wildlife in Northern Kenya in relation to livestock and permanent water points. Biological Conservation. 100:297-306.

Denney, R. N. 1972. Relationships of wildlife to livestock on some developed ranches on the Laikipia Plateau, Kenya. Journal of Range Management. 25:415-425.

DePuy, W., V. Benka, A. Massey, S. Deem, M. Kinnaird, T. O'Brien, S. Wanyoike, J. Njoka, B. Butt, J. Foufopoulos, J. Eisenberg, and R. Hardin. 2014. Q fever risk across a dynamic, heterogeneous landscape in central Kenya. 11(3):429-433.

Didier, K. A., A. Cotterill, I. Douglas-Hamilton, L. Frank, N. J. Georgiadis, M. Graham, F. Ihwagi, J. King, D. Malleret-King, D. I. Rubenstein, D. Wilkie, and R. Woodroffe. 2011. Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem. *In*: Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) 632:95-105. Smithsonian Institution Scholarly Press.

Didier, K.A., A. Cotterill, I. Douglas-Hamilton, L. Frank, N.J. Georgiadis, M. Graham, F. Ihwagi, J. King, D. Malleret-King, D. Rubenstein, D. Wilkie & R. Woodroffe. 2011. Landscape-scale conservation planning of the Ewaso Nyiro: a model for land-use planning in Kenya? *In.* Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) 632:105-123. Smithsonian Institution Scholarly Press.

Dinerstein, E. 2011. Family Rhinocerotidea (Rhinoceroses). *In.* Handbook of mammals of the world, Volume 2: Hoofed Mammals. (Eds.) D. E . Wilson and R. A. Mittermeier. Lynx, New York, NY.

Dixit, A., S. Levin, and D. I. Rubenstein. 2012. Reciprocal insurance among Kenyan pastoralists. DARPA. Theoretical Ecology. 6(2):173-187.

Dolrenry, S., J. Stenglein, L. Hazzah, R. S. Lutz, and L. Frank. 2014. Metapopulation approach to African lion (Panthera leo) conservation. Public Library of Science (PLoS ONE). 9(2):e88081.

Donihue, C. M., L. M. Porensky, J. Foufopoulus, C. Riginos, and R. M. Pringle. 2013. Glade cascades: indirect legacy effects of pastoralism enhance the abundance and spatial structuring of arboreal fauna. Ecology. 94:827-837.

Douglas-Hamilton, I. 1987. African elephants: population trends and their causes. Oryx 21:11-24.

Du Toit, J. T., B. H. Walker, and B. M. Campbell. 2004. Conserving tropical nature: current challenges for ecologists. Trends in ecology and evolution. 19(1):12-17.

Durant, S. M., M. S. Becker, S. Creel, S. Bashir, A. J. Dickman, R. C. Beudels-Jamar, L. Lichtenfeld, R. Hilborn, J. Wall, G. Wittemyer, *et al.*, 2015. Developing fencing policies for dryland ecosystems. Journal of Applied Ecology. 52(3):544.

Dutta, T., S. Sharma, J. Maldonado, T. Wood, H. S. Panwar, and J. Seidensticker. Gene flow and demographic history of leopards (Panthera pardus) in the central Indian highlands. 2013. Evolutionary Applications. 6(6):949-59.

Elliot, N. B., S. A. Cushman, D. W. Macdonald, A. J. Loveridge, and N. Pettorelli. 2014 The devil is in the dispersers: predictions of landscape connectivity change with demography. Journal of Applied Ecology. 51(5)1169.

Esri. 2015. Environmental Systems Research Institute (Esri©). ArcGIS 10.3.1.

Estes, J. A., J. Terborgh, J. S. Brashares, M. E. Power, J. Berger, W. J. Bond, et. al., 2011. Trophic downgrading of planet earth. Science. 333:301-306.

EXCEL/VIS. 2015. Excelis Visual Information Solutions (EXCEL/VIS©). http://exelisvis.com/

Fahrig, L., and T. Rytwinski. 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology and Society. 14:21.

Fedriani, J. M., T. K. Fuller, R. M. Sauvajot, and E. C. York. 2000. Competition and intraguild predation among three sympatric carnivores. Oecologia. 125:258-270.

Finke, D. L. and R. F. Denno. 2004. Predator diversity dampens trophic cascades. Nature. 429: 407-410.

Fischhoff, I. R., S. R. Sundaresan, J. Cordingley, and D. I. Rubenstein. 2007. Habitat use and movements of plains zebra (*Equus burchelli*) in response to predation danger from lions. Behavioral Ecology. 18:725-729.

Fjeldsa, J. and J. C. Lovett. 1997. Geographical patterns of old and young species in African forest biota: the significance of specific montane areas as evolutionary centres. Biodiversity Conservation. 6:325–346.

Ford, A. T., J. R. Goheen, T. O. Otieno, L. Bidner, L. A. Isbell, T. M. Palmer, D. Ward, R. Woodroffe, and R. M. Pringle. 2014. Large carnivores make savanna tree communities less thorny. Science. 346(6207):346-349.

Forman, R. T., D. Sperling, J. A. Bissonette, A. P. Clevenger, C. D. Cutshall, V. H. Dale, L. Fahrig, R. France, C. R. Goldman, K. Heanue, J. A. Jones, F. J. Swanson, T. Turrentine, and T. C. Winter. 2003. Road ecology: science and solutions. Island Press, Washington, D.C., USA.

Fox-Dobbs, K., Doak, D. F., Brody, A. K., Palmer, T. M. 2010. Termites create spatial structure and govern ecosystem function by affecting N2 fixation in an East African savanna. Ecology. 91:1296-1307.

Frank, L. 1998. Living with lions: carnivore conservation and livestock in Laikipia District, Kenya. L. G. Report published by DAI for USAID. 63pp.

Frank, L. 2005. People and predators in Laikipia District, Kenya. *In.* Woodroffe, R., S. Thirgood, and A. Rabinowitz. (Eds). People and Wildlife: Conflict or Coexistence? Cambridge University Press, London.

Frank, L. 2011. Living with Lions: Lessons from Laikipia. *In.* Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) 632:73-84.

Frank, L., A. Cotterill, S. Ekwanga, S. Maclennan, L. Hazzah, S. Dolreny, S. Blackburn, A. Howard, and B. Eldridge. 2009. Living with lions annual report.

Frank, L.G., Cotterill, A., Dolrenry, S. Hazzah, L. 2011. The role of carbofuran in the decline of lions and other carnivores in Kenya. In Carbofuran and Wildlife Poisoning: Global Perspectives and Forensic Approaches. Ed. by N. Richards. John Wiley & Sons Ltd., Chichester, pp.70-74.

Franz, T. E., Caylor, K. K., Nordbotten, J. M., Rodriguez-Iturbe, I., and M. A. Celia. 2010. An ecohydrological approach to predicting regional woody species distribution patterns in dryland ecosystems. Advances in Water Resources. 33:215-230.

Franz, T. E., E. G. King, K. K Caylor, and D. A. Robinson. 2011. Coupling vegetation organization patterns to soil resource heterogeneity in a central Kenyan dryland using geophysical imagery. Water Resources Research. 47:W07531.

Funston, P. J., Frank, L., Stephens, T., Davidson, Z., Loveridge, A., Macdonald, D. M., Durant, S., Packer, C., Mosser, A. and S. M. Ferreira. 2010. Substrate and species constraints on the use of track incidences to estimate African large carnivore abundance. Journal of Zoology. 281:56-65.

Gadd, M. 2005. Conservation outside of parks: attitudes of local people in Laikipia, Kenya. Environmental Conservation. 32(1):50-63.

Georgiadis, N. J. 2011. Conserving wildlife in African landscapes: Kenya's Ewaso ecosystem. Smithsonian Contributions to Zoology. 632:vi-123.

Georgiadis, N.J. 2011. Introduction: Conserving wildlife in Kenya’s Ewaso landscape. *In.* Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya’s Ewaso Ecosystem, N. Georgiadis, (Ed.) 632:1-10.

Georgiadis, N. J., M. Hack, and K. Turpin. 2003. The influence of rainfall on zebra population dynamics: implications for management. Journal of Applied Ecology. 40:125-136.

Georgiadis, N. J., J. G. N. Olwero, G. Ojwang, and S. S. Romañach. 2007a. Savanna herbivore dynamics in a livestock-dominated landscape: I. Dependence on land use, rainfall, density, and time. Biological Conservation. 137:461-471.

Georgiadis, N. J., J. G. N. Olwero, G. Ojwang, and S. S. Romañach. 2007b. Savanna herbivore dynamics in a livestock-dominated landscape: II. Ecological, conservation, and management implications of predator restoration. Biological Conservation. 137:473-483.

Georgiadis, N.J., J.G.N. Olwero, Osundwa and G. Aike. 2011. Reassessing aerial sample surveys for wildlife monitoring, conservation, and management. *In.* Smithsonian

Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) 632:31-42.

Gichuki, F.N., Liniger, H.P., MacMillan, L.C., Schwilch, G. and J. K. Gikonyo. 1998. Scarce Water: Exploring Resource Availability, Use and Improved Management. Eastern and Southern Africa Geographical Journal. Special Number, 8: 15-28.

Gifford-Gonzalez, D. 1998. Early pastoralists in East Africa: ecological and social dimensions. Journal of Anthropological Archeology. 17:166-200.

Giman, B. R. Steubing, N. Megum, W. J. McShea, and C. M. Stewart. 2007. A camera trapping inventory for mammals in a mixed use planted forest in Sarawak. The Raffles Bulletin of Zoology. 55(1):209-215.

Gitahi, N. and K.H. Fitzgerald. 2011. Conserving wildlife on private land: the legal framework for landownership and new tools for conservation. *In.* Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, ed.). 632:95-104.

Global Forest Watch (GFW). 2015. GFW interactive map. Accessed online at: http://www.globalforestwatch.org/

Goheen, J. R. and T. M. Palmer. 2010. Defensive plant-ants stabilize megaherbivore-driven landscape change in an African savanna. Current Biology. 19:1768-1772.

Goheen, J. R., T. P. Young, F. Keesing, and T. M. Palmer. 2007. Consequences of herbivory by native ungulates for the reproduction of a savanna tree. Journal of Ecology. 95:129-138.

Goheen, J. R., T. M. Palmer, F. Keesing, C. Riginos, and T. P. Young. 2010. Large herbivores facilitate savanna tree establishment via diverse and indirect pathways. Journal of Animal Ecology. 79: 372-382.

Goheen, J. R., T.M. Palmer, G. K. Charles, K. M. Helfen, S. N. Kinyua, J. E. Maclean, H. S. Young and R. M. Pringle. 2012. Piecewise disassembly of a large-herbivore community across a rainfall gradient: The UHURU experiment. Public Library of Science (PLoS ONE). 8(2): e55192.

Good, S. P. and K. K. Caylor. 2011. Climatological determinants of woody cover in Africa. Proceedings of the National Academy of Sciences. 108:4902-4907.

Google. Map data ©2015 Google Earth Pro. Accessed from website: https://www.google.com/work/earthmaps/earthpro.html

Gooseman, M. 1997. Internal fragmentation: the effects of roads, highways and powerline clearings on movement and mortality of rainforest vertebrates. Pp 241-255. *In.* W. F. Laurance and R. O. Bierregaard Jr., (Eds). Tropical forest remnants: ecology, management, and conservation of fragmented communities. University of Chicago Press, Chicago.

Grace, J. J. San Jose, P. Meir, H. S. Miranda, R. A. Montes. 2006. Productivity and carbon fluxes of tropical savannas. Journal of Biogeography. 33:387-400.

Graham, M. 2006. Coexistence in a land-use mosaic. Ph. D. Dissertation, Cambridge University. Cambridge, United Kingdom.

Graham, M. D. and T. Ochieng. 2008. Uptake and performance of farm-based measures for reducing crop-raiding by elephants (Loxodanta Africana) among smallholder farms in Laikipia District, Kenya. Oryx. 42:76–82.

Graham, M. D. and T. Ochieng. 2010. The use of electrified fences to mitigate human-elephant conflict: experiences from the Laikipia Plateanu in northern Kenya. *In.* Ferguson, K. and J. Hanks (Eds.) Fencing Impacts: A review of the environmental, social and economic impacts of game and veterinary fencing in Africa with particular reference to the Great Limpopo and Kavango-Zambezi Transfrontier Conservation Areas. Pretoria: Mammal Research Institute. pp 185- 192

Graham, M. D., N. Gichohi, F. Kamau, G. Aike, B. Craig, I. Douglas-Hamilton, and W. M. Adams. 2009. The Use of Electrified Fences to Reduce Human Elephant Conflict: A Case Study of the Ol Pejeta Conservancy, Laikipia District, Kenya. Laikipia Elephant Project Working Paper No. 1, Nanyuki, Kenya.

Grant, C. C., R. bengis, D. Balfour, M. Peel, W. Davies-Mostert, H. Killian, R. Little, I. Smith, M. Garai, M. Henley, B. Anthony, and P. Kartley. 2008. Controlling the distribution of elephants. *In.* Scholes, R. J. and K. G. Mennell (Eds.), Assessment of South African elephant management. Witwatersrand University Press, Johanesburg. Pp. 329-369.

Green Car Reports. 2015. http://www.greencarreports.com/

Gregory, N. C., R. L. Sensenig, and D. S. Wilcove. 2010. Effects of controlled fire and livestock grazing on bird communities in East African savannas. Conservation Biology. 24:1606-1616.

Gregory, T., R. Farah Carrasco, J. Deichmann, J. Kolowski, and A. Alfonso. 2014. Arboreal camera trapping: taking a proven method to new heights. Methods in Ecology and Evolution. 5(5):443-451.

Grootenhuis, J. G. and H. H. T. Prins. 2000. Wildlife utilization: a justified option for sustainable land use in African savannas. *In*: Prins, H. H. T., J. G. Grootenhuis, and T. T. Dolan (Eds.). Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, Boston. Pp. 382-469.

Haile-Mariam, M. and M. B. Philipsson. 1998. Boran – indigenous Affican cattle with potential. Currents (Sweden) No.17/18. Pp. 38-43.

Hallworth, M. T., T. S. Sillett, S. J. Van Wilgenburg, K. A. Hobson and P. P. Marra. 2015. Migratory connectivity of a Neotropical migratory songbird revealed using archival light-level geolocators. Ecological Applications. 25(2): 336-347.

Hansen, A. J., R. L. Knight, J. M. Marzluff, S. Powell, K. Brown, and P. H. Gude. 2005. Effects of Exurban Development on Biodiversity: Patterns, Mechanisms, and Research Needs. Ecological Applications. 15(6):1893-1905.

Hansen, C. F. 2015. Lidar remote sensing of forest canopy structure: an assessment of the accuracy of Lidar and its relationship to higher trophic levels. A doctoral dissertation. Accessed through the University of Vermont: http://scholarworks.uvm.edu/graddis/356

Harrington, R., N. Owen-Smith, P. Viljoen, H. C. Biggs, D. R. Mason, and P. Funston. 1999. Establishing the causes of the roan antelope decline in Kruger National Park, South Africa. Biological Conservation. 90:69-78.

Hazza, L. and S. Dolrenry. 2011. Living with lions. Annual report.

Heath, B. 2000. Ranching: an economic yardstick. *In*: Prins, H. H. T., J. G. Grootenhuis, T. T. Dolan (Eds.). Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, Boston. Pp. 21-33.

Herrick, J. E., S. Andrews, G. Baldi, B. T. Bestelmeyer, J. Brown, J. Davies, M. Duniway, K. M. Havstad, D. Peters, J. Quinton, C. Riginos, P. Shaver, D. Steinaker, and S. Twomlow. 2012. Revolutionary land use change in the 21st century: is (rangeland) science relevant? Rangeland Ecology and Management. 65:590-598.

Holling, C. S. and G. K. Meffe. 1996. On command-and-control, and the pathology of natural resource management. Conservation Biology. 10:328-337.

Homann, S. 2004. Indigenous knowledge of Boran pastoralists in natural resource management: a case study from Southern Ethiopia. Doctoral dissertation: Justus Liebig University, Giessen. Pp. 1-249.

Homewood, K. M., P. C. Trench, and D. Brockington. 2012. Pastoralist livelihoods and wildlife revenues in East Africa: a case for coexistence? Pastoralism: Research, Policy and Practice. 2(19):2-23.

Horn, H. 1975. Markovian properties of forest succession. Pages. *In.* Ecology and Evolution of Communities. Harvard University Press, Massachusetts, USA. Pp.196-211.

Horning, N., J. Robinson, E. Sterling, W. Turner, and S. Spector. 2010. Remote sensing for ecology and conservation. Oxford University Press, Oxford. Pp. 496.

Hortal, J., P. A. V. Borges, and C. Gaspar. 2006. Evaluating the performance of species richness estimators: sensitivity to sample grain size. Journal of Animal Ecology. 75:274-287.

Hubbard C. 1972. Observations on the life history and behavior of some small rodents from Tanzania. Zoologica Africana. 7: 419-449.

Hunter, L. T. B. 1998. The behavioural ecology of reintroduced lions and cheetahs in the Phinda Resource Reserve, KwaZuluNatal, South Africa. PhD thesis. University of Pretoria.

Hunter, M.D. 2009. Trophic promiscuity, intraguild predation, and the problem of omnivores. Agricultural and Forest Entomology. 11: 125-131.

Hunter, M. D., and P. W. Price. 1992. Playing chutes and ladders: heterogeneity and the relative roles of bottom-up top-down forces in natural communities. Ecology. 73:724-732.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs. 54(2):187-211.

Huston, M. A. 1979. A general hypothesis of species diversity. American Naturalist. 113:81-101.

Hutchinson, E. 1953. The concept of pattern in ecology. Proceedings of the Academies of National of Sciences. 105:1-12.

Hutton, J., W. M. Adams, and J. C. Murombedzi. 2005. Planning for success: Serengeti lions seek prey accessibility rather than abundance. Journal of Animal Ecology. 74:559-566.

ILRI. 2015. International Livestock Research Institute. Accessed from website: https://www.ilri.org/

IPSI. 2014. International Partnership for the Satoyama Initiative (IPSI). Water Conservation Strategy for Laikipia County 2014 – 2018: A collaborative activity by KENWEB and LWF for 2013-2014.

IUCN. 2015. Species Red List: International Union for the Conservation of Nature.

Jahnke, H. E. 1982. Livestock production systems and livestock development in tropical Africa. Kieler Publications. Uppsala University.

Karanth, K. U., and M. E. Sunquist. 1995. Prey selection by tiger, leopard and dhole in tropical forests. Journal of Animal Ecology. 64:439–450.

Karanth, U and J. D. Nichols. 1998. Estimation of tiger densities in India using photographic captures and recaptures. Ecology. 79(8):2852-2862.

Karanth, K. U., N. S. Kumar, and J. D. Nichols. 2002. Field surveys: estimating absolute densities of tigers using capture– recapture sampling. *In*. K. U. Karanth and J. D. Nichols (Eds.). Monitoring tigers and their prey: A manual for researchers, managers and conservationists in tropical Asia. Centre for Wildlife Studies, Bangalore, India. Pp: 139–152

Kartzinel, T. R., J. R. Goheen, G. K. Charles, E. Defranco, J. E. Maclean, T. Otieno, T. M. Palmer, and R. M. Pringle. 2014. Plant and small mammal responses to large-herbivore exclusion in a semi-arid African savanna: the first five years of the UHURU experiment. Ecology. 95:787.

KATO. 2015. The Kenya Association of Tour Operators. Accessed from website: http://www.katokenya.org/

Kays, R. R. Costello, T. Forrester, M. C. Baker, A. W. Parsons, E. L. Kalies, G. Hess, J. J. Millspaugh, and W. McShea. 2015. Journal of Mammalogy. 95(5):981-987.

Kaye-Zwiebel, E. and E. King. 2014. Kenyan pastoralist societies in transition: varying perceptions of the value of ecosystem services. Ecology and Society. 19(3):17.

Keesing, F. 1998. Impacts of ungulates on the demography and diversity of small mammals in central Kenya. Oecologia. 116:381-389.

Keesing, F. 2000. Cryptic consumers and the ecology of an African savanna. BioScience. 50(3):205-215.

Keesing, F. and T. Crawford. 2001. Impacts of density and large mammals on space use by the pouched mouse (*Saccostomus mearnsi*) in central Kenya. Journal of Tropical Ecology. 17:465-472.

Keesing, F., B. F. Allan, T. P. Young, and R. S. Ostfeld. 2013. Effects of wildlife and cattle on tick abundance in central Kenya. Ecological Applications. 23:1410–1418.

Keesing, F. and T. P. Young. 2014. Cascading consequences of the loss of large mammals in an African savanna. Bioscience. 64:487-405.

Kelly, M. J. 2008. Design, evaluate, and refine: camera trap studies for elusive species. Animal Conservation. 11: 182-184.

Kelly, M. J. and E. L. Holub. 2008. Camera trapping of carnivores: trap success among camera types and across species, and habitat selection by species, on Salt Pond Mountain, Giles County, Virginia. Northeastern Naturalist. 15(2):249-262.

Kenya Agricultural and Livestock Research Organization. 2015. http://www.kalro.org/

Kenya National Bureau of Statistics. 2009. Current census information. Accessed from website: http://www.cbs.go.ke/

Kenya National Bureau of Statistics. 2014. Kenya facts and figures. Pp:87. Accessed from website: http://www.knbs.or.ke/index.php?option=com_phocadownload&view=category&id=20&Itemid=595

Kenya National Council for Law Reporting. 2010. The Constitution of Kenya, the Official Law Reports of the Republic of Kenya.

Kenya Wildlife Service. 2007. Conservation and management strategy for the black rhino and management guidelines for the white rhino in Kenya (2007-2011). Kenya Wildlife Service, Nairobi.

Kiers, E. T., T. M. Palmer, A. R. Ives, J. Bruno, and J. L. Bronstein. 2010. Mutualisms in a changing world: an evolutionary perspective. Ecology Letters. 13:1459-1474.

Kimuyu, D.K., R.L. Sensenig, C. Riginos, K.E. Veblen and T.P. Young. 2014. Wild and domestic browsers and grazers reduce fuels, fire temperatures, and Acacia ant mortality in an African savanna. Ecological Applications. 24(4): 741-749.

Kinnaird, M. F. 2012. Director, Mpala Research Centre (MRC), Personal communication.

Kinnaird, M. F. and T. G. O'Brien. 2009. Baseline survey of Eland Downs: report to the African Wildlife Foundation.

Kinnaird, M. F. and T. G. O'Brien. 2012. Effects of Private-Land Use, Livestock Management, and Human Tolerance on Diversity, Distribution, and Abundance of Large African Mammals. Conservation Biology. 26(6):1026-1039.

Kirigia, A., J. T. Njoka, P. D Kinyua and T. P. Young. 2013. Characterizations of livestock manure marketing chain and the annual income contribution of manure trade in Mukogodo, Laikipia, Kenya. African Journal of Agricultural Research. 8:5864-5871.

Knight, A. T., R. M. Cowling, M. Rouget, A. Balmford, A. T. Lombard and B. M. Campbell. 2007. Knowing but not doing: selecting priority conservation areas and the research-implementation gap. Conservation Biology. 22(3):610-617.

Knight, M. 2011. African Rhino Specialist Group report. Pachyderm. 49:6-15.

Kohler, T. 1987. Land use in Transition. Aspects and problems of small scale farming in a new environment: the example of Laikipia District, Kenya. Geographica Bernensia, African Study Series A5.

Kolowski J. M, and A. Alonso. 2010. Density and activity patterns of ocelots (Leopardus pardalis) in northern Peru and the impact of oil exploration activities. Biological Conservation. 143: 917–25.

Krameri-Mbote, A. P. G. 1999. Property rights and biodiversity management in Kenya – the case of land tenure and wildlife management (Doctoral Dissertation). Stanford University. UMI microform accession order: 9924587.

Kurten and E. Anderson. 1980. Pleistocene Mammals of North America. Columbia University Press, New York. 422 pp.

Kyalangalilwa, B., J. S. Boatwright, B. H. Daru, O. Maurin, and M. Van der Bank. 2013. Phylogenetic position and revised classification of *Acacia s.l.* (Fabaceae: Mimosoideae) in Africa, including new combinations in *Vachellia* and *Senegalia*. Botanical Journal of the Linnean Society. 172:500-523.

Laikipia Wildlife Forum (LWF). 2007. Conservation in action: conserving wildlife outside national parks. LWF Newsletter. January, 2006.

Laikipia Wildlife Forum (LWF). 2008. Conservation in action: conserving wildlife outside national parks. LWF Newsletter. January, 2007.

Laikipia Wildlife Forum (LWF). 2009. Laikipia Wildlife Forum Newsletter 2009.

Laikipia Wildlife Forum (LWF). 2011. Laikipia-A Natural History Guide. Laikipia Wildlife Forum, Nanyuki, Kenya.

Laikipia Wildlife Forum (LWF). 2013. Wildlife conservation strategy for Laikipia County 2012-2030.

Laikipia Wildlife Forum (LWF). 2014. Water conservation strategy for Laikipia County 2014-2018. Pp100.

Laikipia Wildlife Forum (LWF). 2015. Conservation Enterprise program. Accessed from website: http://www.laikipia.org/portfolio-item/conservation-enterprise/

Lampkin, K. and G. Lampkin. 1960. Studies on the production of beef from zebu cattle is East Africa: milk production in suckled cows and its effect on calf growth. Journal of Agricultural Science. 55:233-239.

Lamprey, H. F. 1983. Pastoralism yesterday and today: the overgrazing problem. *In*. Bourliere, F. (Ed.), Tropical Savannas: Ecosystems of the World, Vol. 13. Elsevier, Amsterdam, The Netherlands. Pp. 643-666.

Landscapes for People, Food, and Nature. 2015. A recent visit to Ol Pejeta Ranch in Laikipia, Kenya, highlighted the value of using mobile bomas, or enclosures, in a mixed wildlife and livestock ranching system. Accessed from website: http://bit.ly/1S9HVJ3

Lane, P. J. 2011. An outline of the later Holocene archaeology and precolonial history of the Ewaso Basin, Kenya. *In.* Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) Pp: 632:11-30.

Lane, P. J. 2013. Trajectories to pastoralism in northern and central Kenya: an overview of the archaeological and environmental evidence. *In.* M. Bollig and Hans-Peter Wotzka (Eds.), Pastoralism in Africa: Past, Present and Future. Oxford: Berghahn. Pp. 104-144

Langpap, C. and J. Wu. 2008. Predicting the Effect of Land-Use Policies on Wildlife Habitat Abundance. Canadian Journal of Agricultural Economics. 56:195-217.

Laurance, S. G., P. C. Stoupher, and W. F. Laurance. 2004. Effects of road clearings on movement patterns of understory rainforest birds in central Amazonia. Conservation Biology. 18(4):1099-1199.

Laurance, W. F., B. M. Croes, L. Tchignoumba, S. A. Lahm, A. Alonso, M. E. Lee, P. Campbell, and C. Ondzeano. 2006. Impacts of roads and hunting on central African rainforest mammals. Conservation Biology. 20:1251–1261.

Louthan, A. M., D. F. Doak, J. R. Goheen, T. M. Palmer and R. M. Pringle. 2013. Climatic stress mediates the impacts of herbivory on plant population structure and components of individual fitness. Journal of Ecology. 101:1074-1083.

Louthan, A. M., D. F. Doak, J. R. Goheen, T. M. Palmer, and R. M. Pringle. 2014. Mechanisms of plant–plant interactions: concealment from herbivores is more important than abiotic-stress mediation in an African savannah. Proceedings of the Royal Society B. 281:20132647.

Loveland, T. R., B. C. Reed, J. F. Brown, D. O. Ohlen, J. Zhu, L. Yang, and J. W. Merchant. 2000. Development of a global land cover characteristics database and IGBP DISCover from 1-km AVHRR data. International Journal of Remote Sensing. 21:1303-1330.

Loveridge, A. J., S. W. Wang, L. G. Frank and J. Seidensticker. 2010. People and wild felids: conservation of cats and management of conflicts. *In.* The Biology and Conservation of Wild Felids. Ed. by Macdonald, D. and Loveridge, A. Oxford University Press.

Luck, M., and J. Wu. 2002. A gradient analysis of urban landscape pattern: a case study from the Phoenix metropolitan region, Arizona, USA. Landscape Ecology. 17:327-339.

Ludwig, J., D. Tongaway, D. Freudenberger, J. Noble, and K. Hodgekinson. 1997. Landscape ecology, function and management: principles from Australia's rangelands. CSIRO, Australia.

Lush, L., M. Mulama, and M. Jones. 2015. Predicting the habitat usage of African black rhinoceros (Diceros bicornis) using random forest models. African Journal of Ecology. 53(3):1-9.

Mabunda, D., D. J. Pienaar, and J. Verhoef. 2003. The Kruger national park: a century of management and research. *In.* du Toit, J. T., K. H. Rogers, and H. C. Biggs (Eds). The Kruger experiences. Ecology and management of savanna heterogeneity. Island Press, Washington. Pp:3021.

MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology. 83(8):2248-2255.

MacKenzie, D. I., J. D. Nichols, J. E. Hines, M. G. Knutson and A. B. Franklin. 2003. Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. Ecology. 84(8):2200-2207.

MacKenzie, D. I. and J. D. Nichols. 2004. Occupancy as a surrogate for abundance estimation. Animal Biodiversity and Conservation. 27(1):461-467.

MackKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Academic Press, Elsevier. Burlington, MA.

MacKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2007. Occupancy estimation and modeling. Academic Press. Elsevier, Inc.

Mackey, R. L. and D. J. Currie. 2001. The diversity-disturbance relationship: is it generally strong and peaked? Ecology. 83(12):3479-3492.

Maclean, J. E., J. R. Goheen, D. F. Doak, T. M. Palmer, and T. P. Young. 2011a. Cryptic herbivores mediate the strength and form of ungulate impacts on a savanna tree. Ecology. 92:1626-1636.

Maclean, J. E., J. R. Goheen, T. M. Palmer, and T. P. Young. 2011b. Small mammals limit tree population growth in an African savanna. Ecology. 92:1626–1636.

MacLennan, S. D., R. J. Groom, D. W. Macdonald, and L. G. Frank. 2009. Evaluation of a compensation scheme to bring about pastoralist tolerance of lions. Biological Conservation. 142(11):2419-2427.

Magige, F. and R. Senzota. 2006. Abundance and diversity of rodents at the human-wildlife interface in Western Serengeti, Tanzania. African Journal of Ecology. 44:371-378.

Malpai Borderlands Group (MBG), 2015. The formation and history of the Malpai Borderlands Group. Accessed online at: http://www.malpaiborderlandsgroup.org/?section=26

Manyara, J. and E. Jones. 2007. Community-based tourism enterprises development in Kenya: an exploration of their potential as avenues of poverty reduction. Journal of Sustainable Tourism. (15):6-628-644.

Martin, R. 1983. Species report for roan, sable, and Tseessebe in support of the trans-boundary mammal project of the ministry of environment and tourism, Namibia. University of Pretoria, Pretoria.

Martins, D. J. 2013. People, Plants and Pollinators: Uniting Conservation, Food Security, and Sustainable Agriculture in East Africa. Conservation Biology: Voices from the Tropics. *In.* First Edition. Navjot S. Sodhi, Luke Gibson, and Peter H. Raven. (Eds.) John Wiley & Sons, Ltd. Published 2013 by John Wiley & Sons, Ltd.

Mbugua, S. W., 1986. Monitoring livestock and wildlife in Kenya. *In*. Hansen, R.M., Woie, B.M., Child, R.D. (Eds.), Range Development and Research in Kenya. Winrock International Institute for Agricultural Development, Morrilton, AZ, pp. 413–430.

McCauley, D. J., F. Keesing, T. P. Young, B. F. Allen, and R. M. Pringle. 2006. Indirect effects of large herbivores on snakes in an African savanna. Ecology. 87(10):2657-2663.

McCauley, D. J., F. Keesing, T. Young, and K. Dittmar. 2008. Effects of the removal of large herbivores on fleas of small mammals. Journal of Vector Ecology. 33(2):263-268.

McDonald, W. R., and C. C. St. Clair. 2004. Elements that promote highway crossing structure use by small mammals in Banff National Park. Journal of Applied Ecology. 41:82–93.

McNeil. J. R. 2000. Something new under the sun: an environmental history of the twentieth century world. New York. W. W. Norton.

Menaut, J. C. 1983. The vegetation of African savannas. *In.* F. Bourliere (Ed.), Tropical savannas, Elselvier, Amsterdam.

Microsoft. 2013. Microsoft Excel© (Microsoft Office 2013). Redmond, Washington..

Miller, G. R., J. M. Cable, A. K. McDonald, B. Bond, T. E. Franz, L. Wang, S. Gou, A. P. Tyler, C. B. Zou and R. L. Scott. 2012. Understanding ecohydrological connectivity in savannas: a system dynamics modelling approach. Ecohydrology. 5:200–220.

Ministry of Environment and Natural Resources (MENR). 2015. Kenya government Department of Range Surveys and Remote Sensing (DRSRS) Service Charter. Accessed on June, 2014. http://www.environment.go.ke/?p=52

Mistry, J. 2000. Savannas. Progress in Physical Geography. 24:601–608.

Mizutani, F. 1999. Biomass density of wild and domestic herbivores and carrying capacity on a working ranch in Laikipia District, Kenya. African Journal of Ecology. 37:226-240.

Mizutani, F. and P. A. Jewel. 1998. Home-range and movements of leopards (*Panthera pardus*) on a livestock ranch in Kenya. Journal of the Zoological Society of London. 244:269-286.

Muchiru, A. N., D. J. Western and R. S. Reid. 2008. The role of abandoned pastoral settlements in the dynamics of African large herbivore communities. Journal of Arid Environments. 72:940-952.

Muchiru, A. N., D. J. Western and R. S. Reid. 2009. The impact of abandoned pastoral settlements on plant and nutrient succession in an African savanna ecosystem. Journal of Arid Environments. 73:322-331.

Mugatha, S. M. 2002. Influence of land use patterns on diversity, distribution and abundance of small mammals in Gachoka Division, Mbeere District, Kenya. LUCID Working Paper Series Number 8. International Livestock Research Institute.

Mulama, M. 2012. Personal communication; Chief Science Officer for the Ol Pejeta Conservancy.

Mwakiwa, E., W. F. de Boer, J. W. Hearne, R. Slotow, F. can Langevelde, M. Peel, C. C. Grant, Y. Pretorius, J. D. Stigter, A. K. Skidmore, I. M. Keitkong, H. J. de Knegt, E. M. Kohi, N. Knowx, and H. H. Prins. Optimization of wildlife management in a large game reserve through waterpoints manipulation: a bio-economic analysis. Journal of Environmental Management. 114:352-361.

Ng, S. J., J. W. Dole, R. M. Sauvajot, P. D. Seth, and T. J. Valone. 2004. Use of highway undercrossings by wildlife in southern California. Biological Conservation. 115:499–507.

National Rangelands Trust (NRT). 2015. Critically endangered black rhino reintroduced to Samburu ranges 25 years since the last individual was poached in the area. Accessed online at: http://www.nrt-kenya.org/press-release-rhino-return-to-samburu/

Nicholson, M. J. and A. R. Sayers. 1978. Relationships between body weight, condition score and heart girth changes in Boran cattle. Tropical Animal Health and Production. 19(2)115-120.

Northern Rangelands Trust (NRT). 2007a. NRT 3rd quarterly report to Globe Foundation and St Louis Zoo Friends Association. Linking Livestock Markets to Wildlife Conservation. 13p.

Northen Rangelands Trust (NRT). 2007b. Accessed from website: http://www.nrt-kenya.org/linking-livestock-markets-to-wildlife-conservation-2/

O'Brien, T. G. 2013. Research scientist, Mpala Research Center (MRC), personal communication.

O'Brien, T. G. 2010. Wildlife picture index and biodiversity monitoring: issues and future directions. Animal Conservation. 13:350-352.

O'Brien, T. G., M. F. Kinnaird, and H. T. Wibisono. 2003. Crouching tiger, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. Animal Conservation. 6:131-139.

O'Brien, T. G., J. E. M. Baillie, J.E.M., L. Krueger, and M. Cuke. 2010. The Wildlife Picture Index: monitoring top trophic levels. Animal Conservation. 13: 335-343.

O'Brien, T. G. and M. F. Kinnaird. 2011. Density estimation of sympatric carnivores using spatially explicit capture—recapture methods and standard trapping grid. Ecological Applications. 21(8):2908-2916.

Ochieng, A. 2015. Staff member, Anthony Ochieng, Laikipia Wildlife Forum (LWF). Personal communication.

Odadi, W. O., T. P. Young, and J. B. Okeyo-Owuor. 2007. Effects on wildlife on cattle diets in Laikipia rangeland, Kenya. Rangeland Ecology and Management. 60:179-185.

Odadi, W. O., J. B. Okeyo-Owuor, T. P. Young. 2009. Behavioral responses of cattle to shared foraging with wild herbivores in an East African rangeland. Applied Animal Behavioral Science. 116(2-4):120-125.

Odadi, W. O., A. S. Abdulrazak, M. M. Karachi and T. P. Young. 2011. African wild ungulates compete with, or facilitate, cattle depending on season. Science. 333:1753-1755.

Odadi, W. O. O., M. Jain, S. E. Van Wieren, H. H. T. Prins and D. I. Rubenstein. 2011. Facilitation between bovids and equids in an African savanna. Evolutionary Ecology Research. 13:237-252.

Odingo, R. S. 1971. The Kenya Highlands: Land-use and agriculture development. East African Publishing, Nairobi.

Ogada, M., R. Woodroffe, N. O. Oguge, and L. G. Frank. 2003. Limiting depredation by African carnivores: the role of livestock husbandry. Conservation Biology. 17(6):1521-1530.

Ogada, D. L., M. E. Gadd, R. S. Ostfeld, T. P. Young, and F. Keesing. 2008. Impacts of large herbivorous mammals on bird diversity and abundance in an African savanna Oecologia. 156(2):387-97.

Okello, B. D., T. G. O'Connor, and T. P. Young. 2001. Growth, biomass estimates, and charcoal production of *Acacia drepanolobium* in Laikipia, Kenya. Forest Ecology and Management. 142:143-153.

Okello, B. D., T. P. Young, C. Riginos, D. Kelly, and T. G. O'Connor. 2007. Short-term survival and long-term mortality of *Acacia drepanolobium* after a controlled burn. African Journal of Ecology. 46:395-401.

Okello, M. M., S. G. Manka and D. E. D'Amour. 2008. The relative importance of large mammal species for tourism in Amboseli National Park, Kenya. Tourism Management. 29:751-760.

Olff, H., M. E. Ritchie, and H. H. T. Prins. 2002. Global environmental controls of diversity in large herbivores. Nature. 415:901–904

Olindo, P. 1991. The old man of nature tourism. In. Whelan, nature tourism. Island Press. Washington, D.C. Pp: 23-28.

Ol Pejeta Conservancy (OPC). 2010. Personal communication; OPC staff.

Ol Pejeta Conservancy (OPC). 2013. Species lists obtained at the Ol Pejeta Wildlife and Research Office; internal management documents.

Ol Pejeta Conservancy (OPC). 2015. General information Accessed from website: http://www.olpejetaconservancy.org/

Olson, D. M., E. Dinerstein, E. D. Wikramanayake, N. D. Burgess, G. V. N. Powell, E. C. Underwood, J. A. D'amico, I. Itoua, H. E. Strand, J. C. Morrison, C. J. Loucks, T. F. Allnutt, T. H. Ricketts, Y. Kura, J. F. Lamoreux, W. W. Wettengel, P. Hedao, and K. R. Kassem. 2001. Terrestrial ecoregions of the world: A new map of life on Earth. Bioscience. 51:933–938.

Osborn, F. and G. Parker. 2003. Towards an integrated approach for reducing the conflict between elephants and people: a review of current research. Oryx. 37(1):80-84.

Ottichilo, W. K., J. Leeuw, A. Skidmore, H. H. T. Prins, and M. Y. Said. 2000a. Population trends of non-migratory wildlife in Masai Mara ecosystem, Kenya. African Journal of Ecology. 38(3):202-216.

Ottichilo, W.K., J. Grunblatt, M. Y. Said, and P. W. Wargute. 2000b. Wildlife and livestock trends in the Kenya rangeland. *In*. Prins, H.H.T., Grootenhuis, J.G., Dolan, T.T. (Eds.), Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, Boston, Pp. 203–218.

Owen-Smith, N. 1996. Ecological guidelines for waterpoints in extensive protected areas. South African Journal of Wildlife Research. 26:107-112.

Packer, C., A. Loveridge, S. Canney, T. Caro, S.T. Garnett, M. Pfeifer, K.K. Zander, A. Swanson, D. MacNulty, G. Balme, H. Bauer, C.M. Begg, K.S. Begg, S. Bhalla, C. Bissett, T. Bodasing, H. Brink, A. Burger, A.C. Burton, B. Clegg, *et al*. 2013. Conserving large carnivores: Dollars and fence. Ecology Letters. DOI: 10.1111/ele.12091

Palmer, T. M. and A. K. Brody. 2012. Enough is enough: The effects of symbiotic ant abundance on herbivory, growth and reproduction in an African Acacia. Ecology. 94: 683-691.

Patterson, B. D., S. M. Kasiki, E. Selempo and R. W. Kays. 2004. Livestock predation by lions (*Panthera leo*) and other carnivores on ranches neighboring Tsavo National Parks, Kenya. Biological Conservation. 119:507-516.

Payton, M. E., M. H. Greenstone and N. Schenker. 2003. Overlapping confidence intervals or standard error intervals: What do they mean in terms of statistical significance? Journal of Insect Science. 3:34:1-6.

Pauly, D. and V. Christensen. 1995. Primary production required to sustain global fisheries. Nature. 374:255-257.

Petty, J. 2002. People, livelihoods and collective action in biodiversity management. *In.* Biodiversity, Sustainability and Human Communities: protecting beyond the protected (Eds.) T. O'Riordan and S. Stoll-Kleeman. Cambridge University Press, Cambridge. Pp. 61-86.

Pickup, G., G. N. Bastin and V. H. Chewings. 1998. Identifying trends in land degredation in non-equilibrium rangelands. Journal of Applied Ecology. 35:365-377.

Picmeta Systems. 2014. Picture Information ExtractorTM. Accessed online: http://www.picmeta.com/

Polis, G. A. 1990. The biology of scorpions. Stanford University Press, Stanford, Ca. 247-293.

Polis, G. A. 1999. Why are parts of the world green? Multiple factors control productivity and the distribution of biomass. Oikos. 86:3-15.

Porensky, L. M. 2011. When edges meet: interacting edge effects in an African savanna. Journal of Ecology. 99:923-934.

Porensky, L. H. 2012. Edge effect interactions in an African savanna. Ph.D. Dissertation; University of California, Davis. Pp: 152.

Porensky, L. M. and K. E. Veblen. 2012. Grasses and large herbivores reinforce landscape heterogeneity by excluding trees from ecosystem hotspots. Oecologia. 168:749-759.

Porensky, L. M. and T. P. Young. 2013. Edge-effects interactions in fragmented and patchy landscapes. Conservation Biology. 27(3):509-519.

Porensky, L. C Riginos, J. Balachowski, and J. Belnap. 2011. Guide to monitoring boma sites. Version 3. 28p.

Porensky, L. M., S. E. Wittman, C. Riginos, and T. P. Young. 2013. Herbivory and drought interact to enhance spatial patterning and diversity in a savanna understory. Oecologia 173:591-602.

Porensky, L. M., S. F. Bucher, K. E. Veblen, A. C. Treydte, and T. P. Young. 2013. Megaherbivores and cattle alter edge effects around nutrient hotspots in an East African savanna. Journal of Arid Environments. 96:55-63.

Pringle, R. M., D. F. Doak, A. K. Brody, R. Jocque´, and T. M. Palmer. 2010. Spatial pattern enhances ecosystem functioning in an African savanna. Public Library of Science (PLoS ONE) Biology. 8:1-12.

Pringle, R. M., T. M. Palmer, J. R. Goheen, D. J. McCauley, and F. Keesing. 2010. Ecological importance of large herbivores in the Ewaso Ecosystem. Smithsonian Contribution to Zoology. 632:43-53.

Pringle, R. M., T. M. Palmer, D. J. McCauley, and F. Keesing. 2011. Ecological importance of large herbivores in the Ewaso ecosystem. *In.* Smithsonian Contributions to Zoology (Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) Pp:632:43-54.

Pringle, R. M., J. R. Goheen, T. M. Palmer, G. K. Charles, E. Defranco, R. Hohbein, A. T. Ford, and C. Tarnita. 2014. Low functional redundancy among mammalian browsers in regulating an encroaching shrub (Solanum campylacanthum) in African savanna. Proceedings of the Royal Society B. 281:20140390.

Prugh, L., C. J. Stoner, C. W. Epps, W. T. Bean, W. J. Ripple, A. S. Laliberte and J. S. Brachares. 2009. The rise of the mesopredator. Bioscience. 59(9):779-791.

Ramankutty, N. and J. A. Foley. 1999. Estimating historical changes in global land cover: croplands from 1700 to 1992. Global Biogeochemical Cycles. 13:997-1027.

Redfern, J. V., R. Grant, H. Biggs, W. M. Getz. 2003. Surface water constraints on herbivores foraging in the Kruger National Park, South Africa. Ecology. 84:2093-2107.

Republic of Kenya. 2013. National Tourism Strategy: Department of Tourism, Ministry of East Africa, Commerce and Tourism.

Reudink, M. W., S. L. Van Wilgenburg, L. S. Steele, A. G. Pillar, P. P. Marra and A. E. Mckellar. Patterns of migratory connectivity in Vaux's Swifts at a northern migratory roost: A multi-isotope approach. The Condor. 117(4): Pp. 670-682.

Rewe, T, D. Indetie, J. M. K. Ojango, and A. K. Kahi. 2006. Economic values for production and functional traits and assessments of their influence on genetic improvement in the Boran cattle in Kenya. Journal of Animal Breeding Genetics. 123(1):23-36.

Richardson, M. L.and L. M. Hanks. 2009. Partitioning of niches among four species of orb-weaving spiders in a grassland habitat. Community and Ecosystem Ecology. 38(3): 651-656.

Richardson, M. L. and L. M. Hanks. 2011. Differences in spatial distribution, morphology, and communities of herbivorous insects among three cytoptypes of *Solidago altissima* (Asteraceae). American Journal of Botany. 98(10):1595-1601.

Richardson, M. L., M. L. J. Watson and C. L. Peterson. 2013. Influence of community structure on the spatial distribution of critically endangered *Dicerandra immaculata* var. *immaculata* (Lamiaceae) at wild, introduced, and extirpated locations in Florida scrub. Plant Ecology. 214:443-453.

Richardson, M. L., J. Rynear and C. L. Peterson. 2014. Microhabitat of critically endangered *Lupinus aridorum* (Fabaceae) at wild and introduced locations in Florida scrub. Plant Ecology. 215:399-410.

Riginos, C. and J. E. Herrick. 2010. Monitoring Rangeland Health: A Guide for Pastoralists and Other Land Managers in Eastern Africa, Version II. Nairobi, Kenya: ELMT-USAID/East Africa.

Riley, D. G., J. O. Sanders, R. E. Knutson and D. K. Lunt. 2001. Comparison of F1 Bos indicus / Hereford cows in central Texas I. Reproductive, maternal and size traits. Journal of Animal Science. 79:1431-1438.

Riley, S. P. D., J. P. Pollinger, R M. Sauvajot, E. C. York, C. Bromley, T. K. Fuller, and R. K. Wayne. 2006. A southern California freeway is a physical and social barrier to gene flow in carnivores. Molecular Ecology. 15:1733–1741.

Ripple, W. J. and R. L. Beschta. 2003. Wolf reintroduction, predation risk and cottonwood recovery in Yellowstone National Park. Forest Ecology and Management. 184:299–313

Ripple, W. J., T. M. Newsome, C. W. Wolf, R. Dirzo, K. T. Everatt, M. Galetti, M. W. Hayward, G. I. H. Kerley, T. Levi, P. A. Lindsey, D. W. Macdonald, Y. Malhi, L. E. Painter, C. J. Sandom, J. Terborgh, and B. van Valkenburgh. 2015. Collapse of the world's largest herbivores. Science Advances. 1:E1400103.

Rockwood, L. 2015. Introduction to Population Ecology. Blackwell Publishing. Oxford.

Rogers, C. S. 1993. Hurricanes and coral reefs: the intermediate disturbance hypothesis revisited. Coral Reefs. 12:127-137.

Rogers, C. M. and M. J. Caro. 1998. Song sparrows, top carnivores and nest predation: a test of the mesopredator release hypothesis. Oecologia. 116:227-233.

Romañach, S. S., P. A. Lindsey and R. Woodroffe. 2011. Attitudes toward predators and options for their conservation in the Ewaso Ecosystem. *In.* Smithsonian Contributions to Zoology. Conserving Wildlife in African Landscapes: Kenya's Ewaso Ecosystem, N. Georgiadis, (Ed.) Pp: 632:85-94.

Roques, K. G., T. G. O'Connor, and A. R. Watkinson. 2001. Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. Journal of Applied Ecology. 38(2):268-280.

Rose, R. A., B. Dirck, J. R. Eastman, G. Geller, S. Goetz, L. Guild, H. Hamilton, M. Hansen, R. headley, J. Hewson, *et al.*, 2014. Ten ways remote sensing can contribute to conservation. Conservation Biology. 29(2):350-359.

Rovero, F. and D. W. Deluca. 2007. Checklist of mammals of the Udzungwa Mountains of Tanzania mammalia. 71:47-55.

Rovero, F. and A. R. Marshall. 2009. Camera trapping photographic rate as an index of density in forest ungulates. Journal of Applied Ecology. 46:1011-1017.

Rovero, F., E. Martin, M. Rosa, J. A. Ahumada, and D. Spitale. 2014. Estimating species richness and modelling habitat preferences of tropical forest mammals from camera trap data. Public Library of Science (PLoS ONE). 9(7):1-12.

Rovero, F., F. Zimmerman, D. Berzi, and P. Meek. 2014. Which camera trap type and how many do I need? A review of camera features and study designs for a range of wildlife research applications. Hystrix, the Italian Journal of Mammalogy. 24(2):148-156.

Rowcliffe, J. M. and C. Carbone. 2008. Surveys using camera traps: are we looking to a brighter future? Animal Conservation. 11:185-186.

Rubenstein, D. R. 2011. Spatiotemporal environmental variation, risk aversion, and the evolution of cooperative breeding as a bet-hedging strategy. Proceedings of the National Academy of Sciences. 28(2):10816-10822.

Sanderson J. G., 2004. Camera phototraping monitoring protocol. The tropical ecology, assessment and monitoring (team) initiative. TEAM report.

Sanderson, J. G. and M. Trolle. 2005. Monitoring elusive animals. Animal Science. 93:148-155.

Sanderson E. W., M. Jaiteh, M. A. Levy, K. H. Redford, A. V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. Bioscience. 52:891-904.

Sankaran, M., J. Ratnam, and N. P. Hanan. 2004. Tree-grass coexistence in savannas revisited – insights from an examination of assumptions and mechanisms invoked in existing models. Ecology Letters. 7:480-490.

Sankaran, S., D. J. Augustine and J. Ratnam. 2013. Native ungulates of diverse body sizes collectively regulate long-term woody plant demography and structure of a semi-arid savanna. Journal of Ecology. 101:1389-1399.

SAS. 2012. SAS® 9.0 Procedures Guide. Cary, NC: Statistical Analysis Software. SAS Institute Inc. ©

STR (Save The Rhino). 2015. Africa's rhino face worst poaching crisis in decades. Accessed through: http://bit.ly/1NmlqfZ

Schmitz, O. J. 2008. Effects of predator hunting mode on grassland ecosystem function. Ecology. 88:2415-2426.

Sensenig, R. L. 2008. Fire ecology in Laikipia, Kenya. Booklet distributed to Laikipia landowners, managers, and stakeholders. Pp:79.

Sharma, S., T. Dutta, J. Maldonado, T. Wood, H. S. Panwar, and J. Seidensticker. 2013. Forest corridors maintain historical gene flow in a tiger metapopulation in the highlands of central India. Proceedings of the Royal Society B. 280: 20131506.

Silver, W. L., L. M. Kueppers, A. E. Lugo, R. Ostertag and V. Matzek. 2004. Carbon sequestration and plant community dynamics following reforestation of tropical pasture. Ecological Applications. 14:1115–1127.

Sinclair, A. R. E., and J. M. Fryxell. 1985. The Sahel of Africa: Ecology of a Disaster. Canadian Journal of Zoology. 63:987-94.

Sitati, N. W. and M. Walpole. 2006. Assessing farm-based measure for mitigating human-elephant conflict in Transmara District, Kenya. Oryx. 40:279–286.

Skole, D. and C. Tucker. 1993. Tropical deforestation and habitat fragmentation in the Amazon - satellite data from 1978 to 1988. Science. 260:1905-1910.

Smithsonian Conservation Biology Institute (SCBI). 2015. Center for Conservation and Sustainability. Personal Communication.

Soulé, M. E., J. A. Estes, J. Berger, and C. Martinez del Rio. 2003. Ecological effectiveness: conservation goals for interactive species. Conservation Biology. 17(5):1238-1250.

Sousa, W. P. 1984. The role of disturbance in natural communities. Annual Review of Ecology and Systematics. 15:353-391

Steinmetz, R., S. Srirattanaporn, J. Mor-Tip and N. Seuaturien. 2014. Can community outreach alleviate poaching pressure and recover wildlife in South-East Asian protected areas? Journal of Applied Ecology. 51(6):1469-1478.

Stoner, D. 1925. The toll of the automobile. Science. 61:56-57.

Stuart-Hill, G. and D. Grossman. 1993. Parks, profits and professionalism: lion return to Pilanesberg. African Wildlife. 47:267.

Sundaresan, S., and C. Riginos. 2010. Lessons learned from biodiversity conservation in the private lands of Laikipia, Kenya. Great Plains Research. 20:17-28.

Sundaresan, S. R., C. Riginos, and E. S. Abelson. 2011. Management and analysis of camera trap data: alternative approaches (response to Harris *et al.* 2010). Bulletin of the Ecological Society of America. 92:188-195.

Taiti, S. W. 1992. The Vegetation of Laikipia – Mt. Kenya. Laikipia Research Programme, Laikipia, Kenya.

Talbot, L. 1965. A survey of past and present wildlife research in East Africa. African Journal of Ecology. 3(1):61-85.

Talbot, L. 1986. Demographic factors in resource depletion and environmental degradation in East African rangeland. Population and Development Review. 12(3): 441-451.

Talbot, L. and P. Olindo. 1990. Kenya: The Maasai Mara and Amboseli reserves. *In.* A. Kiss (Ed.). Living with wildlife: wildlife resource management with local participation in Africa. World Bank Technical Paper No. 130. Africa Technical Department Series. Washington, D.C. Pp. 67.

Tallis, H., J. Lubchenko and 238 co-signatories. 2014. A call for inclusive conservation. Nature. 515(7525): 27-28.

Taylor, B. D., and R. L. Goldingay. 2004. Wildlife road-kills on three major roads in north-eastern NSW. Wildlife Research. 31:83–91.

Taylor, B. D., and R. Goldingay. 2009. Can road-crossing structures improve population viability of an urban gliding mammal? Ecology and Society. 14(2):13.

Taylor, D., P. J. Lane, V. Muiruri, A. Ruttledge, R. Gaj McKeever, T. Nolan, P. Kenny and R. Goodhue. 2005. Mid-to late-Holocene vegetation dynamics on the Laikipia Plateau, Kenya. The Holocene. 15(6):837-846.

Terborgh, J., L. Lopez, P. Nunez, M. Rao, G. Shahabuddin, G. Orihuela, M. Riveros, R. Ascanio, G. H. Adler, T. D. Lambert, and L. Balbas. 2001. Ecological meltdown in predator-free forest fragments. Science. 294(5548):1923-1926.

Theobald, D. M. 2000. Fragmentation by inholdings and exurban development. Pages 155– 174 in R. L. Knight, F. W. Smith, S. W. Buskirk, W. H. Romme, and W. L. Baker. (Ed.) Forest fragmentation in the southern Rocky Mountains. University Press of Colorado, Boulder, Colorado, U.S.A.

Theobald, D. M, T. Spies, J. Kline, B. Maxwell, N. T. Hobbs, and V. H. Dale. 2005. Ecological Support for Rural Land-Use Planning. Ecological Applications. 15(6): 1906-1914.

Thouless, C. 1995. Conflict between humans and elephants on private land in northern Kenya. Oryx. 28(2):119-127.

Thrash, I., P. J. Nel, G. K. Theron. And J. D. P. Bothma. 1991. The impact of the provision of water for game on the woody vegetation around a dam in the Kruger National Park. Koedoe. 34:131–148.

Tobler, M. W., R. Cochard, and P. J. Edwards. 2003. The impact of cattle ranching on large-scale vegetation patterns in a coastal savanna in Tanzania. Journal of Applied Ecology. 40(3):430-444.

Tobler, M., S. Carrillo-Percastegui, R. Leite Pitman, R. Mares and G. Powell. 2008. An evaluation of camera traps for inventorying large-and medium-sized terrestrial rainforest mammals. Animal Conservation. 11:169-178.

Tobler, M. W., A. Zuniga-Hartley, S. E. Carrillo-Percastegui, and G. V. N. Powell. 2015. Spatiotemporal hierarchical modelling of species richness and occupancy using camera trap data. Journal of Applied Ecology. 52:413-421.

Toth, A. B., S. K. Lyons, and A. K. Behrensmeyer. 2014. A century of change in Kenya's mammal communities: increased richness and decreased uniqueness in six protected areas. Public Library of Science (PLoS ONE). 9(4): e93092.

Townsend, C. R. and M. R. Scarsbrook. 1997. The intermediate disturbance hypothesis, refugia, and biodiversity in streams. Limnology and Oceanography. 42(5):938-949.

Treves, A. and K. U. Karanth. 2003. Human-carnivore conflict and perspectives on carnivore management worldwide. Conservation Biology. 17(6):1491-1499.

Trolle, M., A. J. A. J. Noss, J. L. P. Cordeiro, and L. F. B. Oliveira. 2008. Brazilian tapir density in the Pantanal: a comparison of systematic camera trapping and line-transect surveys. Biotropica. 40:2011-217.

Tyre, A. J., B. Tenhumberg, S. A. Field, D. Niejalke, K. Parris, and H. P. Possingham. 2003. Improving precision and reducing bias in biological surveys: estimating false-negative error rates. Ecological Applications. 13:1790–1801.

United Nations Population Fund (UNPF). 2007. State of world population 2007: Unleashing the Potential of Urban Growth. Accessed online: http://www.unfpa.org/.

United States Geological Survey (USGS). 2015. Global climate change mapping. Accessed on: http://www.usgs.gov/climate_landuse/.

Van der Ree, R., E. van der Grift, N. Gulle, K. Holland, C. Mata, and F. Suarez. 2007. Overcoming the barrier effect of roads—how effective are mitigation structures? An international review of the use and effectiveness of underpasses and overpasses designed to increase the permeability of roads for wildlife. *In.* C. L. Irwin, D.

Nelson, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation, Little Rock, Arkansas, USA. Pp: 423–432.

Van Eden, M., E. Ellis and C. Koleski. 2014. 2014 survey to determine local community perceptions of the West Laikipia fence, adjoining protected areas, and the Kenya Wildlife Act Compensation Plan. A report by the Laikipia Wildlife Forum and Colorado State University. Pp27.

Vanthomme, H. J. Kolowski, L. Korte, and A. Alfonso. 2013. Distribution of a community of mammals in relation to roads and other human disturbances in Gabon, Central Africa. Conservation Biology. 27(2):281-291.

Vanthomme, H. J. Kolowski, B. S. Nzamba, and A. Alfonso. 2015. Hypothesis driven and field-validated method to prioritize fragmentation mitigation efforts in road projects. Ecological Applications. 25(7):2035-2046.

Veblen, K. E. 2012. Savanna glade hotspots: plant community development and synergy with large herbivores. Journal of Arid Environments. 78:119-127

Veblen, K. E., and T. P. Young. 2010. Contrasting effects of cattle and wildlife on the vegetation development of a savanna landscape mosaic. Journal of Ecology. 98: 993-1001.

Vitousek, P. M., P. R. Ehrlich, A. H. Ehrlich, and P. A. Matson. 1986. Human appropriation of the products of photosynthesis. Bioscience. 36:368-373.

Wahungu G. M, L. K. Mureu, P. G. Macharia. 2010. Variability in survival and mortality of *Acacia drepanolobium* Sjostedt following prescribed burning at Olpejeta Conservancy, Kenya. African Journal of Ecology. 48:744-750.

Walker, B. H. and I. Noy-Mier. 1982. Aspects of the stability and resilience of savanna ecosystems. *In*. Huntley, B. J., Walker, B. H. (Eds.), Ecology of Tropical Savannas. Springer, Berlin, Germany. Pp. 556-590.

Wambuguh, O. 1998. Local communities and wildlife: a spatial analysis of human-wildlife interactions in Laikipia District, Kenya. Ph.D. Dissertation. University of California, Berkeley.

Wambuguh, O. 2007. Interactions between humans and wildlife: Landowner experiences regarding wildlife damage, ownership and benefits in Laikipia District, Kenya. Conservation and Society. 5(3):408-428.

Waithaka, J. 2012. The Kenya Wildlife Service in the 21st Century: protecting globally significant areas and resources. *In.* The George Wright Forum. 29(1):21-118.

Wildlife Conservation Society (WCS), Center for International Earth Science Information Network (CIESIN), Columbia University. 2005. Last of the Wild Project, Version 2, 2005 (LWP-2): Global Human Footprint Database (Geographic). Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). http://dx.doi.org/10.7927/H4M61H5F. Accessed 11/27/2015.

Western, D. 1975. Water availability and its influence on the structure and dynamics of a savannah large mammal community. East African Wildlife Journal. 13:265-286.

Western, D. 1989. Conservation without parks: wildlife in the rural landscape. *In*. Western, D. and M. C. Pearl (Eds.), Conservation for the Twenty-first Century. Oxford University Press, Oxford. Pp: 158-165.

Western, D., M. Wright and S. Strumm (Eds). 1994. Natural Connections: perspectives in Community-based Conservation, Island Press, Washington DC.

Western, D., S. Russell, and I. Cuthill. 2009. The status of wildlife in protected areas compared to non-protected areas of Kenya. Public Library of Science (PLoS ONE). 4(7):e6140.

Williams-Linera, G. 1990. Vegetation structure and environmental conditions of forest edges in Panama. Journal of Ecology. 78:356-373.

Wilson, E.O. 1992. The Diversity of Life, Harvard University Press, Cambridge, MA.

Woolmer, G., S. C. Trombulak, J. C. Ray, P. J. Doran, M. G. Anderson, R. F. Baldwin, A. Morgan, and E. W. Sanderson. 2008. Resclaing the human footprint: a tool for conservation planning at an ecoregional scale. Plandscape and Urban Planning. 87:42-53.

Woodroffe, R. 2011. Demography of a recovering African wild dog (Lycaon pictus) population. Journal of Mammalogy. Journal of Mammalogy. 92(2):305-315.

Woodroffe, R. and L. Frank. 2005. Lethal control of African lions (Panthera leo): local and regional population impacts. Animal Conservation. 8:91-98.

Woodroffe, R. and Ginsberg, J. R. 1998. Edge effects and the extinction of populations inside protected areas. Science. 280:2126-2128.

Woodroffe, R., P. Lindsey, S. Romañach, A. Stein, and S. M. K. ole Ranah. 2005. Livestock predation by endangered African wild dogs (Lycaon pictus) in northern Kenya. Biological Conservation. 124:225-234.

Woodroffe, R., Frank, L. Lindsey, P., ole Ranah, S., Romanach, S. 2007. Livestock husbandry as a tool for carnivore conservation in Africa's rangelands: a case-control study. Biodiversity and Conservation. 16:1245-1260.

Woodroffe, R. H. Simon, and S. M. Durant. 2014. To fence or not to fence. Science. 344:46-48.

World Bank. 2015. General information. Accessed online: http://www.worldbank.org/ke

WWF (World Wildlife Fund). 2015. Few rhinos survive outside of national parks and preserves. Accessed online at: http://www.worldwildlife.org/species/rhino

Young, H. S. 2012. Post doctoral researcher, Mpala Research Center (MRC), personal communication.

Young, M. and M. Carr. 2015. Assessment of habitat representation across a network of marine protected areas with implications for the spatial design of monitoring. Public Library of Science (PLoS ONE). 10(3): e0116200.

Young, H. S., D. J. McCauley, K. M. Helgen, E. Otarola-Castillo, J. R. Goheen, T. M. Palmer, R. M. Pringle, T. P. Young and R. Dirzo. 2013. Effects of mammalian herbivore declines on plant communities: Observations and experiments in an African savanna. Journal of Ecology. 101:1030-1041.

Young, H. S., R. Dirzo, K. M. Helgen, D. J. McCauley, S. Billeter, M. Kosoy, L. Osikowicz, T. P. Young, and K. Dittmar. 2014. Declines in large wildlife increase landscape-level prevalence of rodent-borne disease. Proceedings of the National Academy of Sciences. 111:7036-7041.

Young, H. S., D. J. McCauley, R. Dirzo, J. R. Goheen, B. Agwanda, E. O. Castillo, A. Ferguson, S. N. Kinyua, M. McDonough, T. M. Palmer, R. M. Pringle, T. P. Young, and K. M. Helgen. 2015. Context-dependent effects of large mammals declines on small mammal communities in central Kenya. Ecological Applications. 25:348-360.

Young, T. P. and D. J. Augustine. 2007. Interspecific variation in the effects of large mammalian herbivores on reproduction in Acacia species in Laikipia, Kenya. Biotropica. 39:559-561.

Young, T. P., N. Partridge and A. Macrae. 1995. Long-term glades in Acacia bushland and their edge effects in Laikipia, Kenya. Ecological Applications. 5(1):87-108.

Young, T. P., T. M. Palmer, and M. E. Gadd. 2005. Competition and compensation among cattle, zebras, and elephants in a semi-arid savanna in Laikpia, Kenya. Biological Conservation. 122:351-359.

Zander, K. 2008. Modelling the value of the Borana cattle in Ethiopia. An approach to justify its conservation. Ecological Economics. 68:34-45.

BIOGRAPHY

Ryan G. Valdez received his Bachelor of Science degree from Texas A&M University, Wildlife & Fisheries Sciences in 1993, followed by a Master of Forest Science from Yale University's School of Forestry and Environmental Science in 1996. Ryan currently serves as senior manager for conservation science and policy at the National Parks Conservation Association (NPCA). Prior to this, he worked for the Smithsonian Institution's National Zoological Park for nearly 10 years before pursuing academic research under a Ph.D fellowship with the Smithsonian Conservation Biology Institute (SCBI). Ryan has worked on various conservation programs throughout the Americas and Africa and has a strong interest in geospatial sciences. Born in Barbados, Ryan has lived most of his life in the US, grew up in the Rio Grande Valley of TX, and currently resides in Washington, DC. He serves as an academic and career mentor for various youth and diversity groups.