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Occupancy Analysis and Density Estimation of Kori Bustards (*Ardeotis kori*) and Helmeted Guineafowl (*Numida meleagris*) for Use in Landscape Conservation Planning in the Northern Tuli Game Reserve, Botswana

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OCCUPANCY ANALYSIS AND DENSITY ESTIMATION OF KORI BUSTARDS (*ARDEOTIS KORI*) AND HELMETED GUINEAFOWL (*NUMIDA MELEAGRIS*) FOR USE IN LANDSCAPE CONSERVATION PLANNING IN THE NORTHERN TULI GAME RESERVE, BOTSWANA

by

Kathryn R. McCollum

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Under the Supervision of John P. Carroll and Larkin A. Powell

Lincoln, Nebraska

December, 2015

OCCUPANCY ANALYSIS AND DENSITY ESTIMATION OF KORI BUSTARDS (*ARDEOTIS KORI*) AND HELMETED GUINEAFOWL (*NUMIDA MELEAGRIS*) FOR USE IN LANDSCAPE CONSERVATION PLANNING IN THE NORTHERN TULI GAME RESERVE, BOTSWANA

Kathryn R. McCollum, M.S.

University of Nebraska, 2015

Advisors: John P. Carroll and Larkin A. Powell

For understudied species, more informed conservation planning and decision-making on both the local and landscape levels may be attained through the use of occupancy and abundance estimations. Here, we focus on two iconic bird species in eastern Botswana, kori bustards (*Ardeotis kori*) and helmeted guineafowl (*Numida meleagris*). The overall goal of this project was to better understand the hierarchy of factors that influence occupancy (ψ) and density of kori bustard and helmeted guineafowl populations within the Northern Tuli Game Reserve and how these factors may interact to affect landscape conservation and usage. We performed distance sampling for both species over two field seasons throughout the Northern Tuli Game Reserve, Botswana during June 2014-July 2014 and May 2015-July 2015. We found that kori bustard probability of occupancy was influenced by open canopies (ψ^{2014}_{open}=0.373, SE±0.086; ψ^{2015}_{open}=0.392, SE±0.061) when compared to closed canopies (ψ^{2014}_{closed}=0.000, SE±0.000; ψ^{2015}_{closed}=0.000, SE±0.000). Kori bustard densities were highest in 2014 in areas of sparse vegetation at higher elevations with 5.02 individuals/km^2 (95%

confidence interval: 1.04 – 24.2 individuals) and lowest in areas of dense vegetation at upper elevation with 0.02 individuals/km^2 (95% confidence interval: 0.005 – 0.140 individuals). In 2015 highest densities were found in areas of sparse vegetation at lower elevations with 2.20 individuals/km^2 (95% confidence interval: 1.73 – 2.80 individuals) and lowest in areas of sparse vegetation at upper elevations with 0.130 individuals/km^2 (95% confidence interval: 0.071 – 0.239 individuals). Helmeted guineafowl occupancy was most influenced by dense vegetation (ψ^{2014}_{dense} =0.800, SE±0.103; ψ^{2015}_{dense} =0.752, SE±0.116) and closed canopy (ψ^{2014}_{closed}=0.857, SE±0.132; ψ^{2015}_{closed}=0.755, SE±0.181), with some influence by lower elevations (ψ^{2014}_{lower}=0.514, SE±0.084; ψ^{2015}_{lower}=0.637, SE±0.082) when compared to sparse vegetation (ψ^{2014}_{sparse}=0.405, SE±0.065; ψ^{2015}_{sparse}=0.436, SE±0.067), open canopy (ψ^{2014}_{open}=0.448, SE±0.061; ψ^{2015}_{open}=0.477, SE±0.064) and upper elevations (ψ^{2014}_{upper}=0.462, SE±0.082; ψ^{2015}_{upper}=0.367, SE±0.082). In 2014, helmeted guineafowl were found at highest densities in areas of sparse vegetation at lower elevations with 828 individuals/km^2 (95% confidence interval: 564 – 1217 individuals) and lowest densities in areas of sparse vegetation at upper elevations 49.1 individuals/km^2 (95% confidence interval: 30.9 – 78.1 individuals). In 2015, helmeted guineafowl were found at highest densities in areas of dense vegetation at higher elevations with 2,085 individuals/km^2 (95% confidence interval: 905 – 4803) and at lowest densities in areas of sparse vegetation at upper elevations with 38.9 individuals/km^2 (95% confidence interval: 23.81 – 63.81 individuals). By determining which habitat and landscape factors influence kori bustard and helmeted guineafowl density and occupancy we will be able to make more informed decisions to aid in the conservation of both species and species that utilize the same types of habitats and resources. We discuss how using these data for landscape conservation planning could have a positive impact on the future of the study site and surrounding area. Habitat-specific information may identify risks during landscape conservation planning within the range of the kori bustard and helmeted guineafowl.

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TABLE OF CONTENTS

LIST OF TABLES

CHAPTER 1

CHAPTER 2

LIST OF FIGURES

Chapter 1 – Occupancy Analysis and Density Estimation of Kori Bustards (*Ardeotis kori*) in the Northern Tuli Game Reserve, Botswana

Abstract

For understudied species, more informed conservation planning and decision-making on both local and landscape levels may be achieved through the use of occupancy and abundance analyses. The kori bustard (*Ardeotis kori*) is an iconic species in Botswana which can serve a role as a flagship species for conservation action. We used distance sampling over two field seasons throughout the Northern Tuli Game Reserve, Botswana during June-July 2014 and May-July 2015. We found that kori bustard probability of occupancy was influenced by open canopies (ψ^{2014}_{open}=0.373, SE±0.086; ψ^{2015}_{open}=0.392, SE±0.061) when compared closed canopies (ψ^{2014}_{closed}=0.000, SE±0.00; ψ^{2015}_{closed}=0.000, SE±0.00). Kori bustard densities were highest in 2014 in areas of sparse vegetation at higher elevations with 5.02 /km^2 (95% confidence interval: 1.04 – 24.2 individuals) and lowest in areas of dense vegetation at upper elevation with 0.02 individuals/km^2 (95% confidence interval: 0.005 – 0.140 individuals). In 2015 highest densities were found in areas of sparse vegetation at lower elevations with 2.20 individuals/km^2 (95% confidence interval: 1.73 – 2.80 individuals) and lowest in areas of sparse vegetation at upper elevations with 0.1307 individuals/km^2 (95% confidence interval: 0.071 – 0.239 individuals). The determination of which habitat and landscape factors influence kori bustard density and occupancy provides some tools necessary to develop more effective management plans. Kori bustards and other low-density species that utilize the similar sparse types of habitats and resources could benefit from landscape scale conservation efforts. Our study confirms to biologists and land managers that protection of sparse

vegetation and open canopy areas are of higher importance to kori bustard conservation than protection of dense vegetation and closed canopy areas. On a broader scale, open canopy areas may be at risk to conversion to rowcrop agriculture as demands for food increase. Habitat-specific information may identify risks during landscape conservation planning within the range of the kori bustard.

Introduction

Habitat degradation and fragmentation are two of the major current threats to biodiversity and conservation (Fischer and Lindenmayer 2007, Jetz et al. 2007). Because of human encroachment, many species have to respond to the loss of habitat and loss of access to critical resources caused by habitat fragmentation. Causes of habitat fragmentation from anthropogenic forces include myriad of issues ranging from urbanization to global climate change. As the consequences of habitat fragmentation are better understood, biologists can contribute to the preparation of more effective conservation plans, policies, and regulations.

The intensification of agricultural systems to produce products for human consumption, bioenergy, and livestock consumption is not a new problem, but is increasing in scale in the Middle East and Sub-Saharan Africa (Lotze-Campen et al. 2010). Agricultural expansion has caused an increase in the occurrence of human-wildlife conflicts. The consequences to wildlife depend in large part on whether the agricultural land is privately or publicly owned (Kinnaird and O'Brien 2012). The borders between these land types are often arbitrary to wildlife movement, yet representative of contrasting rule sets which affect conservation of wildlife.

An example of the public and private land interface can be found in and around the Northern Tuli Game Reserve (Selier 2008; -22.115909, 29.090403), a 720 km^2 wildlife reserve located in the northeastern corner of Botswana (Snyman 2010; Figure 1.2). North of the reserve in Zimbabwe, the Tuli Circle (-21.973388, 29.135202) is managed by Zimbabwean National Parks and Wildlife Department and hunting is still allowed in this region (Figure 1.2). East of the park are privately owned hunting farms and to the west and south are farming areas used for agricultural crops and communal lands used for grazing goats and cattle (Selier 2008; Figure 1.2). This environment is unique in its combination of agricultural land and wild, undeveloped areas, providing ideal conditions for researchers to study the effects anthropogenic changes on the landscape have on local species, such as the kori bustard (*Ardeotis kori*) in the area (Selier 2008).

Kori bustards are large birds in the Family Otididae native to eastern and southern Africa (Johnsgard 1991, Senyatso 2011; Figure 1.1). Kori bustards have the distinction of being, by some measures, the heaviest flying bird species in the world (Liebenberg 2000). The southern subspecies of the kori bustard (*Ardeotis kori kori*) is found throughout Zimbabwe, Botswana, southern Angola, Namibia, South Africa, and southern Mozambique (Johnsgard 1991), and is one of the national birds of Botswana. The other kori bustard subspecies, *Ardeotis kori struthiunculus*, is found in eastern Africa from Ethiopia south to central Tanzania and Lake Victoria (Johnsgard 1991; Figure 1.1).

Habitat degradation and fragmentation have led to a reduction in the number of this once very common bird, especially outside major game reserves (Herremans 1998, Sinclair et al. 2002). Senyatso et al. (2013) determined that the range of the kori bustard has decreased by 8% in southern Africa since the early 1900s, and that number of individuals within the range has greatly decreased over this time. The species is categorized as near threatened on the IUCN Red List due to the current rapid decrease in populations (Birdlife International 2013). The South African red data book classifies kori bustards as vulnerable (Brooke 1984). Kori bustards' low tolerance to human activity and their low reproduction rates during dry years have compounded these already declining population trends (Herremans 1998, Osborne and Osborne 2001, Lichtenberg and Hallager 2008). Other threats to species survival include collisions with powerlines

(Martin and Shaw 2010, Shaw et al. 2010), poaching, and predation (Senyatso et al. 2013). Kori bustards receive varying levels of protection throughout their range, with full protection in reserves and no regulation in other areas (Senyatso 2011, Figure 1.2). Kori bustards have the potential to be classified as an umbrella species for the habitat types that they utilize.

To better understand the threats to conservation of the kori bustard and the development of suitable management strategies, basic population information is needed about this understudied species. We performed occupancy analyses and abundance estimations with data collected through distance sampling. Through abundance estimations we were able to estimate the density of kori bustards in the study area, as well as which habitat factors influence the density of kori bustards. Through occupancy analysis, we were able to determine which habitat types have higher occupancy rates within the study area, as well as which factors are influential in kori bustard occupancy (MacKenzie and Nichols 2004). Although they provide somewhat similar information, using both techniques provide a more complete understanding of both population size and range within the study area (MacKenzie and Nichols 2004). The kori bustard was chosen for this study because of its potential to be an umbrella species for the area, meaning that conservation of environmental factors that benefit the study species will also benefit multiple other species that utilize the same landscape features. The kori bustard has cultural significance with the local people (Low 2011) and is beneficial for the local ecotourism industry by attracting bird watchers to the area.

The determination of which environmental covariates have major influence on kori bustard habitat usage will allow for improved conservation of the species, as well as other species that rely on and utilize similar habitats. The goal of this study was to better understand the hierarchy of factors that influence abundance and occupancy of kori bustard populations within the Northern Tuli Game Reserve and how landscape conservation efforts and usage affect these factors. The specific objectives of this study are:

1. Determine the effects of different habitat types on the presence of kori bustards on the landscape.
2. Determine the variation in density of kori bustards in specific habitat types on the landscape.

We will address these questions using surveys to estimate probability of occupancy and density for each habitat type sampled. These surveys will aid in the understanding of habitat usage by kori bustards throughout the study site and surrounding area.

Methods and Analysis

Study Area

Botswana is a 581,730 km^2 landlocked country located in southern Africa (Senyatso 2011; Figure 1.2). The Northern Tuli Game Reserve (-22.115909, 29.090403) is a 720 km^2 unfenced protected area located in the northeastern corner of Botswana (Snyman 2010, Forssman 2013; Figure 1.2). The Northern Tuli Game Reserve was established as a nature reserve in the mid 1960’s when landholders combined their areas into one large reserve as part of a conservation effort (Snyman 2010; Figure 1.2, 1.3). Previous to the reserve’s formation, much of the land was used for rowcrop agriculture and grazing livestock (Selier 2008). The Northern Tuli Game reserve is now used for ecotourism and research purposes and has three ecotourism lodges in the areas surrounding it (Snyman 2010). Little to no habitat management is performed in the area, which allows for natural habitat development and change. The two largest contributors to habitat change in the past few years is thought to be flooding and the increasing elephant populations. Flooding influences local seed banks, and in recent years has led to the introduction of different plant species near the rivers. Elephant populations aid in the sustainment of sparse vegetation and open canopy areas through feeding and movement (O'Connor et al. 2007).

The southern park boundary follows the Limpopo River, serving as the Botswana-South Africa border. The eastern park boundary follows the Shashe River, serving as the Botswana-Zimbabwe border. The western boundary is marked by a foot-and-mouth disease fence and the southwestern boundary is marked by a fence along the Motloutse River. The northern boundary follows along the Tuli Circle (-21.973388, 29.135202) in Zimbabwe, which is a managed hunting concession. Animal movements between the Tuli Circle and the reserve are unrestricted (Snyman 2010; Figure 1.2, 1.3). A ban on commercial wildlife hunting was put into place beginning in January 2014, prohibiting any commercial take of wildlife within the country (Government of Botswana 2014). Effects of the hunting ban on wildlife populations is unknown as the regulation has only recently come into effect. Still, poaching is a common issue in the country and surrounding area, affecting all types of wildlife (Senyatso 2011).

The landscape consists of sandstone and basalt ridges overlooking alluvial floodplains, small rivers, and drainage lines (Forssman 2013). These rivers and drainages flow into the Limpopo, Shashe, and Motloutse rivers during the wet season and form small watering holes during the dry season (Snyman 2010). Multiple habitat types exist within the Northern Tuli Game Reserve, providing an opportunity to compare which landscape features affect kori bustard occupancy and density (Figure 1.3). Habitats within the Northern Tuli Game Reserve are into five categories (A. Snyman, pers. comm., August 2014) based on vegetation density, water, and canopy cover: 1) Bare Soil, which contains open canopy, little to no vegetation, and no water; 2) Sparse Vegetation, which contains open canopy, little to moderate vegetation, and no water; 3) Grassy/Woody, which contains mixed open and closed canopy, moderate vegetation, and no water; 4) Dense Vegetation/Woodland, which contains closed canopy, dense vegetation, and no water; and 5) Water, which contains open canopy, no vegetation, and water (Figure 1.3).

Study Species

Kori bustards are large and conspicuous birds that are sexually dimorphic in terms of body size (Raihani et al. 2006). The height of an adult kori is usually 1.2-1.5 meters (Liebenberg 2000). Adult males typically weigh 13.5-19 kg and females 4.5-6.4 kg. The adult male kori's wingspan typically measures 740-761mm and an adult female wingspan measures 600-635mm (Johnsgard 1991). Kori bustards are open grassland and open woodland species that are generally found in flat landscapes with medium to heavy grass cover in an area with some rocky outcrops (Johnsgard 1991, Osborne and Osborne 2001). Individuals are usually found alone or in pairs and generally thought of as a low-density species, but where food is readily available they can become gregarious (Liebenberg 2000, Senyatso 2011). Kori bustards are opportunistic omnivores that have been noted to eat insects, small vertebrates such as lizards and rodents, as well as leaves, buds, and sap from plants (Johnsgard 1991). Kori bustards are often found in areas that have recently been burned, most likely due to the abundance of new grasses budding as well as the increased access to insects and other animals that have been exposed by the burning (Senyatso 2011). Kori bustards can also be found around herds of ungulates, feeding on the insects that are being disturbed by the other animals' movements.

Occupancy Analysis

Presence-absence analyses are useful because they take into account the detection probability of the species of interest (MacKenzie et al. 2003). Presence-absence studies involve sampling multiple sites over a short period of time (MacKenzie et al. 2003).The goal is to estimate the proportion of sampling units containing animals, as opposed to abundance estimates which estimate the number of animals within a particular sampling area (Royle and Nichols 2003). The estimation of presence is useful for rare and elusive species in which surveys may contain many samples of zero (MacKenzie et al. 2003).

We performed line transect sampling along ten transects throughout the study area (Figure 1.4). The sample area includes two regions, EcoTraining and Central (Figure 1.3, 1.4; Table 1.2). The EcoTraining region consists of six transects located near the EcoTraining camp within the Northern Tuli Game Reserve. The Central region consists of four transects in the middle of the reserve area. The Central transects are closer to many of the tourism lodges in the Northern Tuli Game Reserve, and therefore have more tourists and vehicles compared to the EcoTraining transects. Transects ranged in length from 1.48 km to 14.45 km and were placed within the two regions (Table 1.2). Transects were routed during 2014 along pre-existing roads, following regulations of the reserve to have as little impact on the surrounding environment and landscape as possible. Routes were set out to include all habitat types that exist within the reserve so sampling would be representative of area. The sample area for this study includes samples of all habitat types (Figure 1.3, 1.4). The amount of each habitat type sampled was kept close-to proportional to the amount of each habitat type in the entire Northern Tuli Game Reserve (Table 1.1).

Transects were driven every one to three days at varying times of the day ranging from 06:45CAT (sunrise) to 17:30CAT (sunset) to prevent time bias on data. Data were collected by or in the presence of the primary researcher (Kathryn McCollum) as well as student volunteers from the University of Nebraska and University of Georgia. As each transect was driven, we recorded the number of kori bustard detected. At each detection a GPS point was created using a handheld GPS unit (Garmin 60CSX) and recorded with a unique individual ID. The distance of the first sighted individual from the transect was determined using a handheld Nikon Monarch laser rangefinder and was noted. The number of individuals was recorded, as well as other observations including cloud cover, time of day, transect number, habitat type, and which side of the transect the individuals were on. Any other notable points about the sighting were also recorded, such as if the individuals were flying, near watering holes, or near large trees.

We used program PRESENCE (MacKenzie et al. 2002, 2003) to obtain occupancy and detection probabilities for the four previously classified habitat types. We used a single-season occupancy model developed by MacKenzie et al. (2002) to account for incomplete detection of kori bustard in our data. Every completed survey of a section of transect was considered a unique occupancy occasion, giving us 11-26 occasions for each transect section. Transects in the Ecotraining region were surveyed more often than those in the Central region due to logisitical constraints. However, the same habitat types were sampled in both regions so the difference in repetition between the two regions should have little to no impact on the data. A kori bustard was counted as detected if it was observed during the completion of a transect. Counts for each section were converted to binary data for the occupancy analysis, with a “1” representing detection and a “0” representing no detection. To avoid double counting, the same transect was not surveyed multiple times on the same day during the same time period. For example, if transect one was surveyed at 07:00CAT, it would not be surveyed again until past noon.

We ran five models for both years to determine which covariates affected kori bustard occupancy, with ψ representing probability of occupancy and p representing probability of detection: ψ (canopy)p(.) to assess the effect of canopy; ψ (elevation)p(.) to assess impact of upper or lower elevation; ψ (vegetation)p(.) to determine effect of sparse or dense vegetation; ψ (.)p (.) as a control model; and ψ (.)p (t) to determine if time had an influence on occupancy. A model was determined to be influential if it had a ΔAIC < 2 (MacKenzie et al. 2002). A goodness-of-fit test was conducted for the global model to assess the fit of the models.

Abundance Estimation

Distance surveys are used to determine the population size or density of a species within a pre-determined area using either transect or point sampling (Anderson et al. 1983). Line transect sampling involves randomly placing transects throughout the study area, then following these transects and recording all sightings of the target species as well as their horizontal distance from transect. Detection can include actual sightings as well as detection by other means such as vocalizations or tracks, but the observer must be able to determine a perpendicular distance from the transect for the detection to be recorded (Buckland et al. 2001). To get a general estimate of how much area would need to be sampled in order to have lower CV values, Buckland et al. (2001) provide the following equation, with CV_t representing the desired CV value, D representing density estimate, L_0 representing the total line length, n_0 representing the number of individuals, b representing a constant 3, and L representing individual transect length:

$$L = \left(\frac{b}{\left(CV_t(D)\right)^2}\right)\left(\frac{L_0}{n_0}\right)$$

Data collected during the transect sampling for the abundance estimation was used for this analysis. We utilized program DISTANCE (Buckland et al. 2001) to analyze the transect data to determine a density estimation for the population of kori bustards within the Northern Tuli Game Reserve. Data were estimated separately by year. Transects were split into approximate 1000 meter sections to provide more detailed habitat classifications, then into four habitat categories with the use of ArcGIS (version 10.3.1) by vegetation density and elevation (Table 1.5, Table 1.6). Areas labeled as Dense Vegetation/Woody and Grassy/Woody were considered “dense” and areas labeled Sparse Vegetation and Bare Soil were considered “sparse” (Figure 1.3). Areas at elevations higher than 540.0 meters were considered “upper” elevation and areas below this point were considered “lower” elevation. This delineation point was chosen arbitrarily as it was the median point of the range of elevations encountered throughout the ten transects. We then used a global analysis to test which model best fit each category. Four estimators were used to determine the model of best fit for each habitat type: uniform, half-normal, hazard-rate, and negative exponential. Models were evaluated by program DISTANCE using Akaike’s Information Criterion (AIC) (Buckland et al. 2001). The model with the lowest AIC score and fewest parameters (K) was considered the best fit. Models were also evaluated using the Kolmogorov-Smirnov goodness-of-fit test, with models having P>0.05 considered well-fitted to the data (Buckland et al. 2001).

Results

O*ccupancy Analysis*

Kori bustards were detected on 31 occasions over 18 transect sections in 2014 and on 129 occasions over 25 transect sections in 2015 within the Northern Tuli Game Reserve. The naïve occupancy, or the proportion of sites where kori bustards were detected, were low for both years (2014: 0.34, 2015: 0.25). Detection as a factor of time did not describe the occupancy probability of kori bustards in either field season (w^{2014}_{AIC}=0.00, w^{2015}_{AIC}=0.00). Therefore, no habitat covariate models with time as a survey-specific factor were incorporated. Occupancy of kori bustard was strongly associated with canopy openness in both field seasons (Table 1.3). We found that kori bustard probability of occupancy varied between open (ψ^{2014}_{open}=0.373, SE± 0.086; ψ^{2015}_{open}=0.392, SE± 0.061) and closed (ψ^{2014}_{closed}=0.000, SE± 0.00; ψ^{2015}_{closed}=0.000, SE± 0.00) canopies. All other models had a ΔAIC>2, and therefore much less descriptive of variation in occupancy probability than the top model (Table 1.3).

Abundance Estimation

We recorded 34 observations of kori bustards by sampling 987.121 km of transect in 2014 and 208 observations of kori bustards by sampling 1133.548 km of transect in 2015 (Table 1.5, Figure 1.5). If we use the equation provided by Buckland et al. (2001) with our density estimate for areas of sparse vegetation at upper elevations in 2014, to obtain a CV value of 10%, we would need to sample 4,752 km of sparse upper elevation habitat type. For our study area this would not be possible as there are only approximately 200 km of sparse upper habitat available to be surveyed. The greatest relative abundance for 2014 was in areas categorized as dense vegetation and lower elevation and for 2015 was in areas categorized as sparse vegetation and lower elevation

(Table 1.5). Densities of kori bustards throughout the region ranged from 0.02 - 5.02/km^2 throughout both field seasons (Table 1.6, Figure 1.6). Kori bustard densities were highest in 2014 in areas of sparse vegetation at higher elevations with 5.02 individuals/km^2 (95% confidence interval: 1.04 – 24.2 individuals) and lowest in areas of dense vegetation at upper elevation with 0.02 individuals/km^2 (95% confidence interval: 0.005 – 0.140 individuals). In 2015 highest densities were found in areas of sparse vegetation at lower elevations with 2.20 individuals/km^2 (95% confidence interval: 1.73 – 2.80 individuals) and lowest in areas of sparse vegetation at upper elevations with 0.130 individuals/km^2 (95% confidence interval: 0.071 – 0.239 individuals).

In our study, the most common cluster size was of two kori bustards per detection in 2014 and 2015 (Table 1.5). We found that vegetation had an impact on detection, with areas of sparse vegetation having higher density estimations than areas of dense vegetation (Table 1.6, Figure 1.6). Elevation had less of an impact on density, as shown by the upper elevation having the higher estimation in 2014 and the lower elevation having the higher estimation in 2015 (Table 1.6).

Discussion

Occupancy Analysis

Our occupancy analysis results suggest kori bustard presence is most influenced by canopy openness. In both field seasons, the canopy covariate had an effect on occupancy rates of kori bustards, suggesting they use less-forested areas with fewer tall trees to areas with more trees and closed canopies. Areas with open canopy had the greatest occupancy estimates (ψ^{2014}_{open} = 0.37, SE±0.086) and 2015 (ψ^{2015}_{open} = 0.39, SE±0.061; Table 1.4). Areas with closed canopies had no detections of kori bustards in either field season (Table 1.4). Lower occupancy levels are expected for kori bustards, as they are noted to be a widespread species (Johnsgard 1991, Liebenberg 2001, Senyatso 2011).

The choice of open canopy and open landscape environments may be for multiple reasons, including easier maneuverability or favored foraging (Johnsgard 1991). Kori bustards are by some measures the heaviest flying birds in the world (Liebenberg 2000), and therefore require more room for take-off and landing than other avian species. Areas of open canopies allow individuals the space to take flight and to land with less risk of injury than areas of more dense vegetation. Food accessibility is another major resource that would influence kori bustard presence in one habitat type over another (Johnsgard 1991). Areas of sparse vegetation may be more plentiful in the amount of food sources available to kori bustards, which would in turn increase presence in those areas.

None of the other covariates included in our analyses play an important role in kori bustard occupancy in our study area, as none of the other covariates have ΔAIC < 2. In 2015 the elevation covariate has a ΔAIC of 2.87 (Table 1.3), with lower elevation having a higher occupancy estimate ($\psi = 0.46$) suggesting that elevation may have a bigger role than shown in this study.

Abundance Estimation

The results from the density estimations suggest kori bustards are most abundant in areas of sparse vegetation at lower elevations. In both field seasons, the habitats with the highest estimated density of kori bustards were those classified as having sparse vegetation, which suggests the habitat type used more often by the kori bustard is thinner understory cover (Table 1.6). Less thick understory could be used over thicker understory due to a need for space in order to take flight. Sparse vegetation could also be used by the prey species, such as rodents, lizards, and insects, which utilize the particular vegetation found in less dense areas.

The overall low density of kori bustards throughout the study area is a characteristic that has been previously observed in other populations of this species (Liebenberg 2000, Senyatso 2011). Although there were fewer detections in 2014 than 2015, we were still able to obtain valuable density estimates for 2014. The difference in number of detections between the two field seasons raises the question of what other factors not included in this study may be influencing kori bustard large-scale distribution. To improve the rigor of density estimates it would be useful to sample even more area than what was available in this study area. Through the use of the previously introduced equation provided by Buckland et al. (2001) in a pilot study, future long-term studies could acquire more detections and gain more insight into habitat use by kori bustards in similar habitat types.

Implications

We were able to determine that the kori bustard population in this study used areas of open canopies and sparse vegetation when compared with areas of closed canopies and dense vegetation through the use of both presence and abundance estimations. The usefulness of the utilization of tools to understand the presence-abundance relationship of species has been recognized by many, as shown in the paper done by Gaston (1999). One of the implications of presence-abundance relationships addressed by Gaston (1999) is in relation to species conservation, in which species with low abundance and low presence may be at higher risk for extinction than species with greater abundance and presence. We found that kori bustards in the Northern Tuli Game Reserve have low presence with probability of occupancy below 0.50 in both years of surveys (Table 1.4), and low abundance with density estimations at 5.02 individuals/km^2 in 2014 and 2.20 individuals/km^2 in 2015. Biologists can use the information that kori bustards may be more at-risk to extirpation and extinction due to low presence and abundance as reasoning to conserve more of the currently available sparse vegetation and open canopy habitat and continue research on other kori bustard populations.

There was a noted difference in detections of kori bustard between the two field seasons, with only 34 detections in 2014 and 129 detections in 2015. The changes between the years seemed to rely on rainfall, with 2014 representing an average year for timing of rainfall and 2015 having a late rain at the end of the wet season. The later rain in 2015 caused vegetation to persist late into the fall and winter, which could have allowed for longer foraging opportunities for kori bustards. Kori bustard movement is highly influenced by food availability (Johnsgard 1991, Senyatso 2011), which could be the reason that more detections occurred in 2015 than 2014.

The habitat types typically occupied by kori bustards are also the types of habitat typically utilized for cattle grazing throughout African savannas (Börner et al. 2007, Lukomska et al. 2014). With cattle as one of the top agricultural exports of the country of Botswana (Bahta 2015), this conflict could become a bigger issue as space becomes more limited and land use change occurs. As addressed by Senyatso (2011), the two largest impacts on wildlife caused by cattle grazing in this environment are that of bush encroachment and additional competition for resources. Although it is an issue that has not been studied in the Northern Tuli Game Reserve, bush encroachment is a major issue in other parts of the kori bustard’s range for both livestock and wildlife (Senyatso 2011, Börner et al. 2007). Kori bustards have been shown to use sparse vegetation with open canopy over dense vegetation with closed canopy (Table 1.3, 1.4, 1.5, 1.6; Figure 1.6), which are habitat types that would decrease as bush encroachment increases.

Sub-Saharan Africa contains some of the most unused cropland in the world (Jenkins 2003), and as demands for food increase with the increasing human populations, so will the pressures to utilize all available lands for rowcrop agriculture. The easiest lands to convert to rowcrop agriculture would be those with already open canopies and sparse vegetation, which are the same habitat types shown to be used by kori bustards. Kori bustards are known to have low tolerance for human activity, and would most likely avoid areas of agriculture and human development instead of adapting to the change in habitat (Herremans 1998, Osborne and Osborne 2001, Lichtenberg and Hallager 2008).

Land managers can benefit from knowledge on kori bustard habitat usage through an ecotourism perspective. Conservation of sparse vegetation and open canopy areas are helpful in the preservation of not only kori bustards, which are a species of interest to bird watchers around the world, but to other charismatic megafauna such as elephants and lions (Selier 2008, Snyman 2010). Having these species within a reserve will sustain and possibly increase ecotourism in the area, which will allow for more funding for conservation and protection of all species found there.

Summary

Our study illustrates some of the habitats affecting the space utilization of an understudied species, the kori bustard, in a landscape consisting of a matrix of land uses. Occupancy of kori bustards was influenced by canopy, with open canopy used more than closed canopy (ψ^{2014}_{open}= 0.373, SE±0.086; ψ^{2014}_{closed}=0.000, SE±0.000; ψ^{2015}_{open}= 0.392, SE±0.061; ψ^{2015}_{closed}=0.000, SE±0.000). Kori bustards were found at higher densities in areas of sparse vegetation in upper elevations (5.02 individuals/km^2, 95% confidence interval: 1.041 – 24.24). Kori bustards were recently identified by the IUCN as ‘near threatened’ status due to loss of habitat and population decline (Birdlife International 2013), so any new information on kori bustards will be helpful in the development of future conservation and management plans for the species and its habitats. To support kori bustard populations in both presence and abundance, emphasis should be placed on the preservation of open canopy areas with sparse vegetation throughout their range which can be accomplished through the intentional conservation by landowners of these habitat types and the avoidance of conversion of land use to agricultural fields or livestock grazing. Conservation of the kori bustard will be beneficial not only for the ecosystem, but for ecotourism as well because of the appeal of this iconic species to bird watchers and the simultaneous conservation of other species which utilize the same habitat types.

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Tables

Table 1.1. Proportion of habitat types sampled compared to overall amount of habitat type determined from vegetation layers in ArcGIS (version 10.3.1) within the Northern Tuli Game Reserve, Botswana.

Habitat Type	Area Sampled (km^2)	Area Available (km^2)	Proportion Sampled
Bare Soil	7.25	89.2	0.0812
Sparse Vegetation	43.26	410.1	0.1054
Grassy/Woody	11.75	169.9	0.0691
Dense Vegetation /Woodland	10.05	52.1	0.1928

Table 1.2. Location, length and brief habitat description of transects used for the kori bustard research project from June – July 2014 and May – July 2015 in the Northern Tuli Game Reserve, Botswana. Locations representative of two regions sampled within the reserve, with EcoTraining defined as area around the EcoTraining camp and Central defined as area in the inner part of the reserve.

Transect	Location	Length (km)	Brief Habitat Description
T1	EcoTraining	4.80	Croton forest, basalt ridges, sandstone ridges, floodplain, open grassland, acacia thicket
T2	EcoTraining	5.18	Marsh/floodplain, basalt ridges, sandstone ridges, sage plains, open grassland, acacia thicket
T3	EcoTraining	1.48	Sandstone ridges, mopane thicket
T4	EcoTraining	5.71	Sandstone ridges, floodplain, croton forest, open grassland, acacia thicket
T5	EcoTraining	3.88	Sandstone ridges, acacia thicket, open grassland
T6	EcoTraining	3.88	Open grassland, mopane thicket
T7	Central	8.45	Appleleaf forest, open grassland, acacia thicket, croton forest
T8	Central	9.01	Croton forest, open grassland, acacia thicket, mopane thicket

T9	Central	14.45	Mopane thicket, open grassland, sandstone ridges, basalt ridges, riverbed, croton forest
T10	Central	8.14	Croton forest, riverbed, mopane thicket, sandstone ridges, basalt ridges

Table 1.3. Occupancy (ψ) and detection (p) model selection results for kori bustard (*Ardeotis kori*) in the Northern Tuli Game Reserve, Botswana over 2 field seasons during June - July 2014 and May - July 2015. K represents number of parameters. ΔAIC represents difference between model and best-fitting model (model with lowest AIC).

Model	Year	K	AIC[1]	ΔAIC	AIC weight
ψ (canopy) p (.)	2014	3	265.54	0.00	0.59
ψ (.)p (.)	2014	2	267.64	2.10	0.20
ψ (elevation) p (.)	2014	3	268.85	3.31	0.11
ψ (vegetation) p (.)	2014	3	269.37	3.83	0.08
ψ (.)p(t)	2014	21	292.74	27.20	0.00
ψ (canopy) p (.)	2015	3	656.59	0.00	0.71
ψ (elevation) p (.)	2015	3	659.46	2.87	0.17
ψ (.) p (.)	2015	2	660.88	4.29	0.08
ψ (vegetation) p (.)	2015	3	662.86	6.27	0.03
ψ (.) p(t)	2015	27	693.64	37.05	0.00

[1] Akaike's Information Criterion

Table 1.4. Kori bustard (*Ardeotis kori*) occupancy (ψ) estimates, standard errors (SE) and 95% confidence intervals (CI) for habitat covariates of occupancy models from two field seasons, June 2014 – July 2014 and May 2015 – July 2105 in the Northern Tuli Game Reserve, Botswana.

Covariate	Year	Ψ	SE	95% CI	
Canopy	2014				
Open		0.3738	0.0861	0.2250	0.5511
Closed		0.0000	0.0000	Not estimable	Not estimable
Vegetation	2014				
Sparse		0.3582	0.0895	0.2065	0.5448
Dense		0.2714	0.1435	0.0824	0.6070
Elevation	2014				
Upper		0.2727	0.0991	0.1233	0.4997
Lower		0.3947	0.1060	0.2146	0.6089
Canopy	2015				
Open		0.3921	0.0616	0.2799	0.5169
Closed		0.0000	0.0000	Not estimable	Not estimable
Vegetation	2015				
Sparse		0.3579	0.0645	0.2433	0.4914
Dense		0.3417	0.1248	0.1489	0.6063
Elevation	2015				

Upper	0.2505	0.0727	0.1353	0.4165
Lower	0.4608	0.0849	0.3043	0.6254

Table 1.5. Relative abundance for kori bustards (*Ardeotis kori*) in each habitat type surveyed June-July 2014 and May-July 2015 within the Northern Tuli Game Reserve, Botswana.

Year	Vegetation	Elevation	Model Selection	Number of Observations	Number of Samples	Effort (km)	Relative Abundance
2014	Dense	Lower	Uniform	5	118	102.740	0.04867
2014	Dense	Upper	Half-normal	1	103	102.795	0.009730
2014	Sparse	Lower	Hazard-rate	19	460	422.320	0.04499
2014	Sparse	Upper	Hazard-rate	9	374	359.266	0.02505
2015	Dense	Lower	Uniform	12	148	130.137	0.09221
2015	Dense	Upper	Uniform	2	89	88.821	0.02252

2015	Sparse	Lower	Uniform	178	592	539.318	0.3300
2015	Sparse	Upper	Uniform	16	390	375.272	0.04264

Table 1.6. Density estimates (individuals/km^2) for kori bustards (*Ardeotis kori*) in each habitat type surveyed June-July 2014 and May-July 2015 within the Northern Tuli Game Reserve, Botswana. N/A cells had sample sizes too small to calculate GOF p values.

Year	Vegetation	Elevation	Density (birds/km^2)	95% CI	GOF P value	%CV
2014	Dense	Lower	0.1896	0.0780 – 0.4608	0.1797	47.07
2014	Dense	Upper	0.0270	0.0051 – 0.1408	N/A	100
2014	Sparse	Lower	0.6459	0.2533 – 1.654	0.6050	49.24
2014	Sparse	Upper	5.025	1.041 – 24.24	N/A	81.37
2015	Dense	Lower	0.6506	0.3399 – 1.245	0.8557	33.78
2015	Dense	Upper	0.3247	0.0785 – 1.342	1.0000	77.82
2015	Sparse	Lower	2.204	1.734 – 2.802	0.5152	12.27
2015	Sparse	Upper	0.1307	0.0713 – 0.2396	0.7384	31.56

Table 1.7. Density of clusters estimates (individuals/km^2) and mean cluster size for kori bustards (*Ardeotis kori*) in each habitat type surveyed June-July 2014 and May-July 2015 within the Northern Tuli Game Reserve, Botswana.

Year	Vegetation	Elevation	Density of Clusters Estimation (95% CI)	%CV	Expected Cluster Size (95% CI)	Mean Cluster Size (95% CI)
2014	Dense	Lower	0.1580 (0.0686 – 0.3636)	44.02	1.2000 (1.0000 – 1.9001)	1.2000 (1.0000 – 1.9001)
2014	Dense	Upper	0.0270 (0.0051 – 0.1408)	100	1.0000	1.0000
2014	Sparse	Lower	0.3646 (0.1532 – 0.8681)	44.87	1.7711 (1.1599 – 2.7044)	2.2632 (1.4081 – 3.6375)
2014	Sparse	Upper	2.225 (0.4711 – 10.51)	77.46	2.2584 (1.2640 – 4.0349)	1.8889 (1.2980 – 2.7487)
2015	Dense	Lower	0.3718 (0.1999 – 0.6914)	32.18	1.7500 (1.3973 – 2.1918)	1.7500 (1.3973 – 2.1918)

2015	Dense	Upper	0.2165 (0.0614 – 0.7630)	70.82	1.5000 (1.0000 – 92.741)	1.5000 (1.0000 – 92.741)
2015	Sparse	Lower	1.336 (1.063 – 1.679)	11.69	1.6495 (1.5328 – 1.7751)	1.6966 (1.5556 – 1.8505)
2015	Sparse	Upper	0.0996 (0.0557 – 0.1781)	30.21	1.3125 (1.0811 – 1.5934)	1.3125 (1.0811 – 1.5934)

Figures

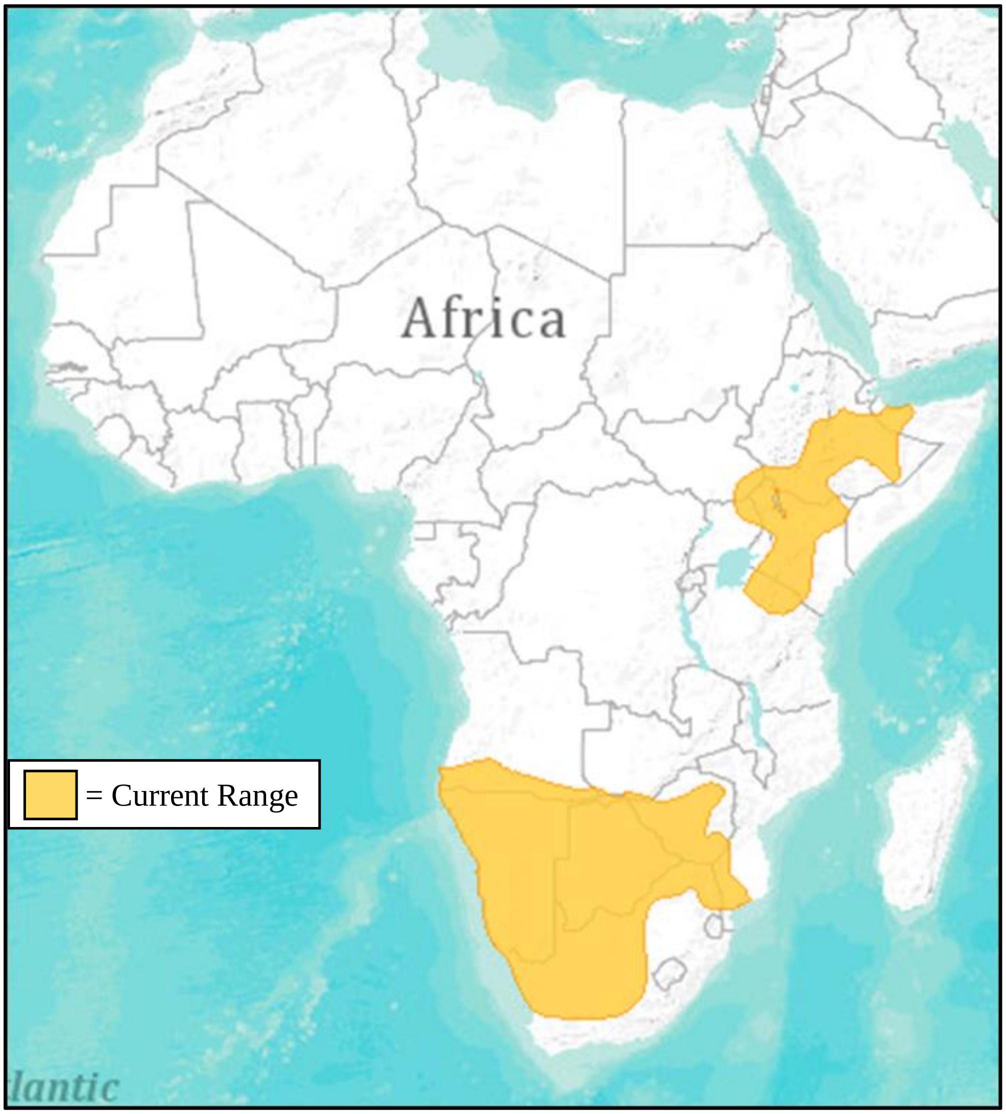

Figure 1.1. Range of kori bustard (*Ardeotis kori)* as of 2013 including both subspecies (Birdlife International 2013). *Ardeotis kori struthiunculus* found in the Northeastern area, *Ardeotis kori kori* found in the southern Africa portion of range.

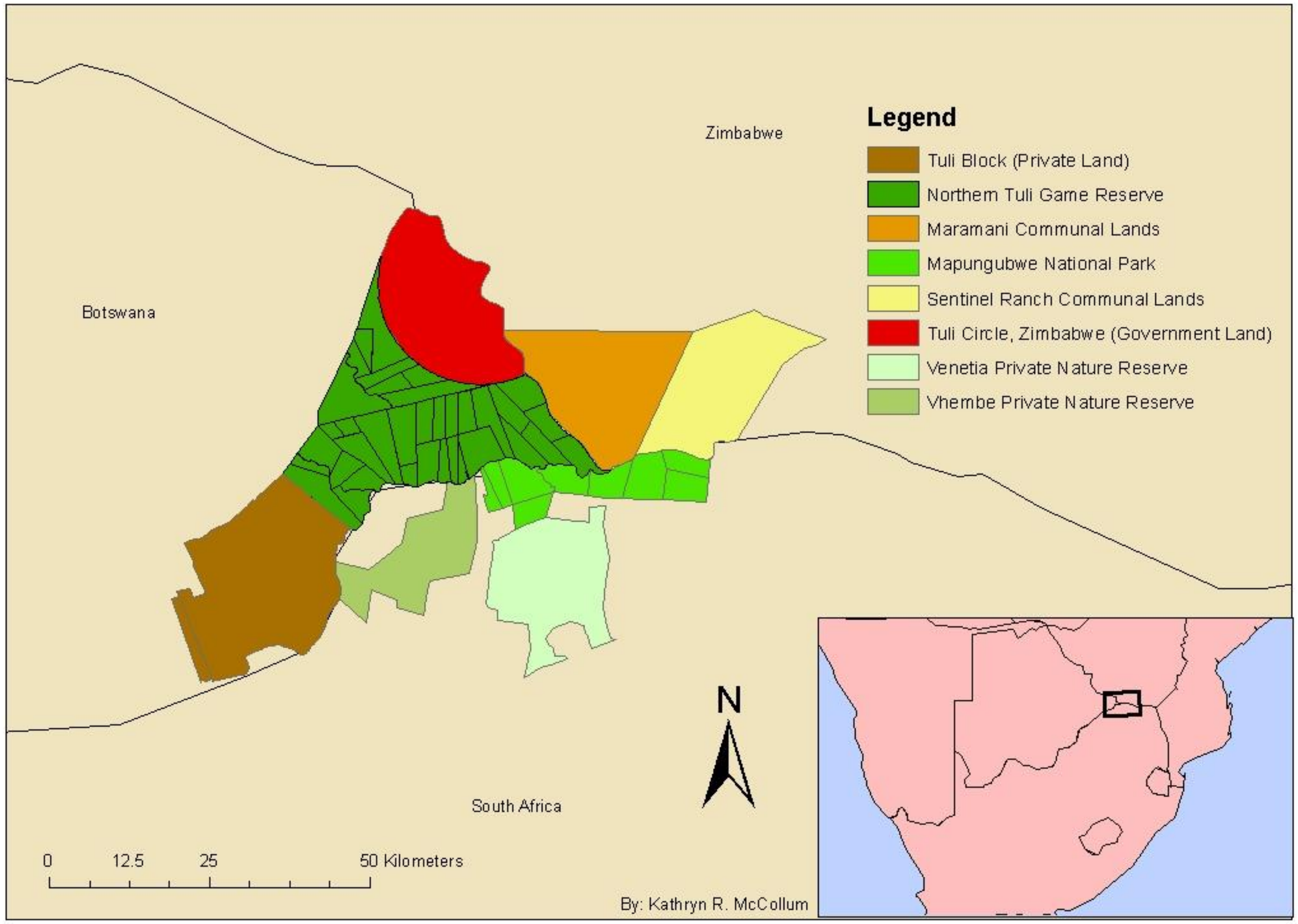

Figure 1.2. Ownership of property for the Northern Tuli Game Reserve and surrounding areas in southern Africa. Southern Africa represented in inset with study site and surrounding area enlarged from the black square. Ownership boundaries are representative of different wildlife conservation regulations.

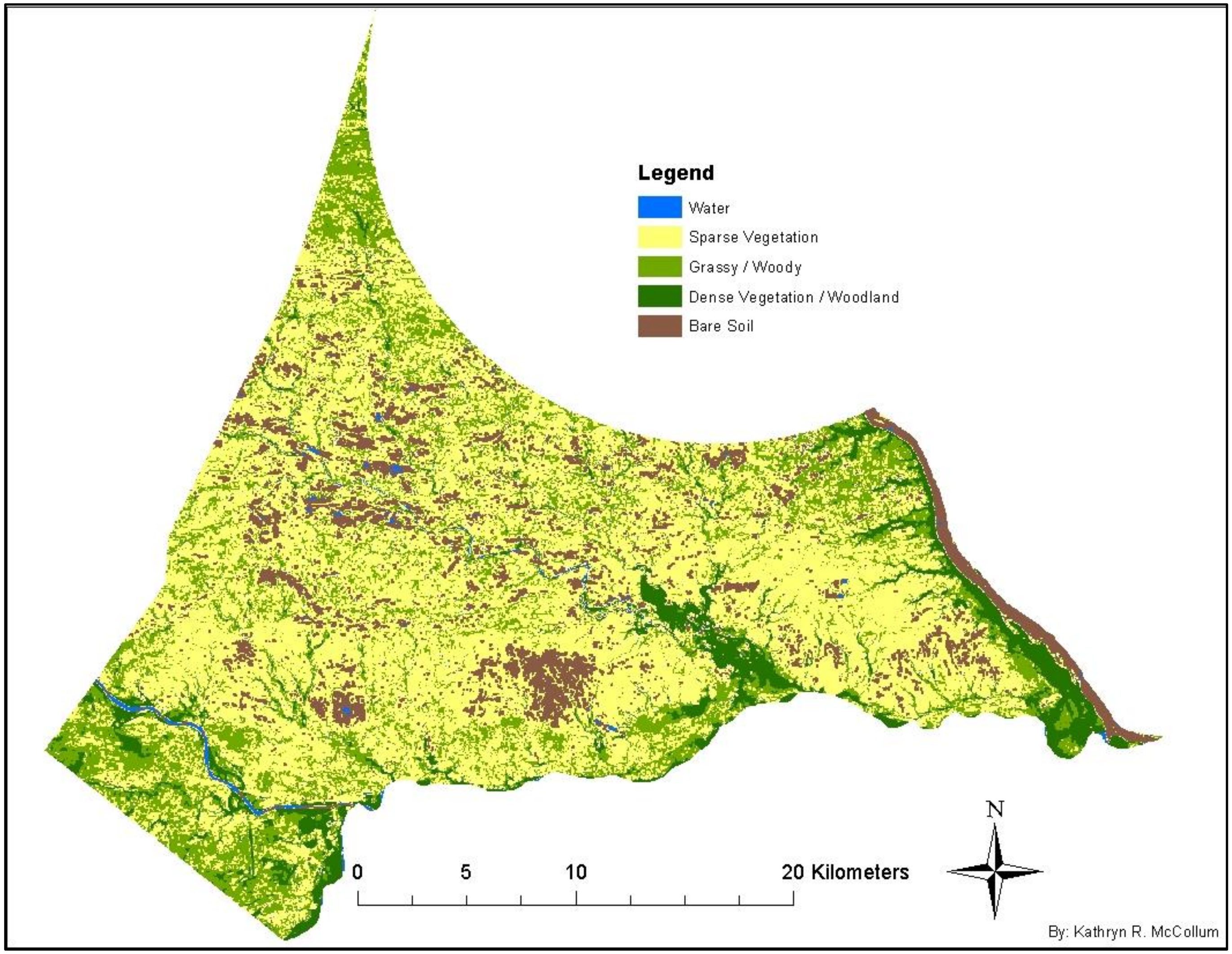

Figure 1.3. General habitat types found throughout the Northern Tuli Game Reserve, Botswana surveyed for the kori bustard research project from June-July 2014 and May-July 2015.

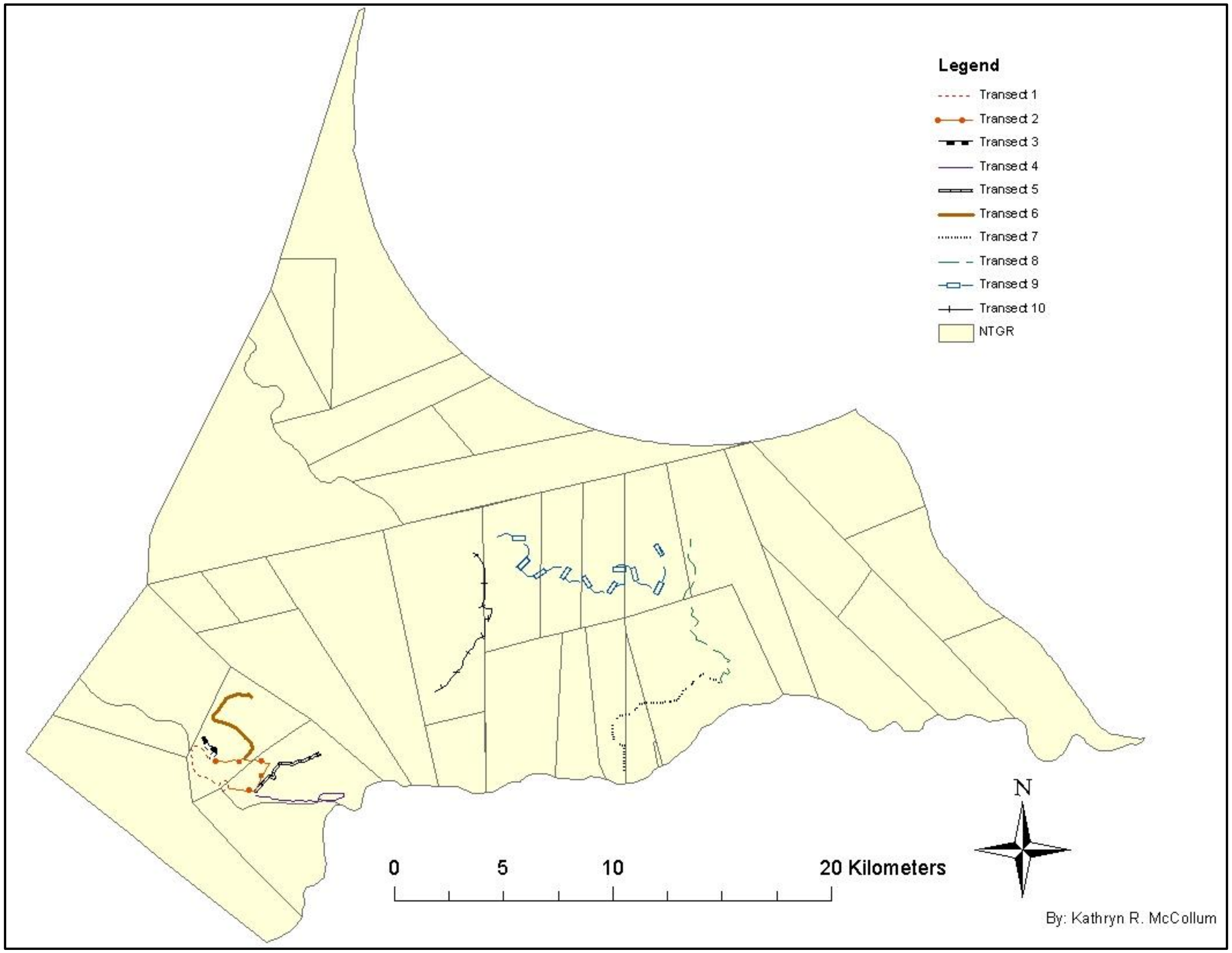

Figure 1.4. Layout of study transects for kori bustard research project within the Northern Tuli Game Reserve, Botswana from June – July 2014 and May – July 2015.

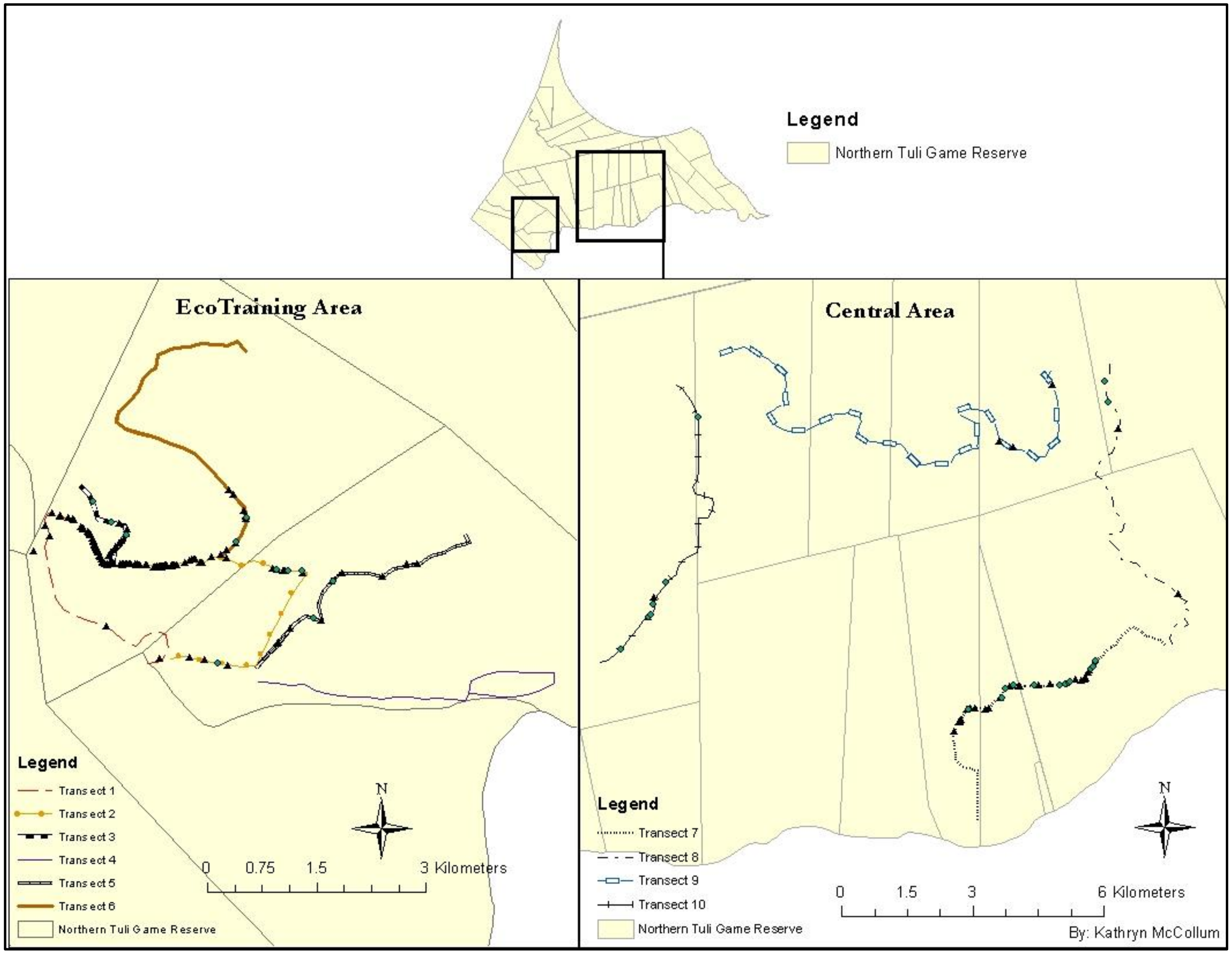

Figure 1.5. Kori bustard detection points throughout the Northern Tuli Game Reserve, Botswana from June-July 2014 and May – July 2015. Two specified areas, EcoTraining and Central, were only regions sampled.

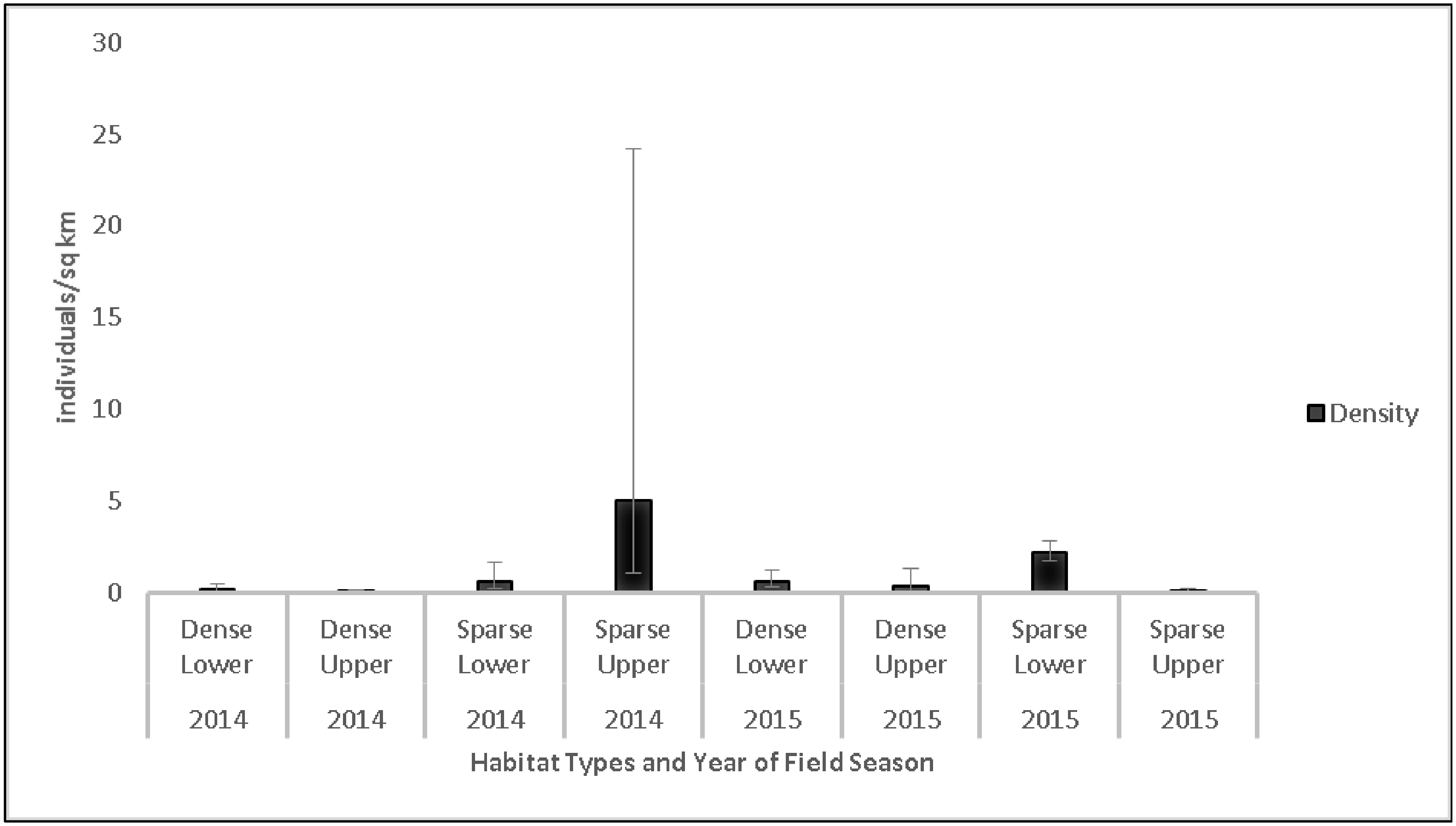

Figure 1.6. Density estimates for kori bustards in different habitat types throughout the Northern Tuli Game Reserve, Botswana for two field seasons from June – July 2014 and May – July 2015.

*Chapter 2 – Occupancy Analysis and Density Estimation of Helmeted Guineafowl (**Numida meleagris**) in the Northern Tuli Game Reserve, Botswana*

Abstract

Anthropogenic fragmentation of habitat throughout southern Africa has impacted habitat and resource availability for multiple wildlife species. Here, we focus on a prevalent bird species in eastern Botswana, helmeted guineafowl (*Numida meleagris*). Helmeted guineafowl are one of the most common upland gamebirds in Sub-Saharan Africa, but little is known on their habitat utilization. The goal of this project is to better understand the hierarchy of factors that influence occupancy (ψ) and density of helmeted guineafowl populations within the Northern Tuli Game Reserve and how landscape conservation and usage affect these factors. We used occupancy and abundance analyses to determine baseline data for this understudied species, as well as more informed conservation planning and decision-making on both the local and landscape levels. We performed distance sampling over two field seasons throughout the Northern Tuli Game Reserve, Botswana during June-July 2014 and May-July 2015. Helmeted guineafowl occupancy was most influenced by dense vegetation (ψ^{2014}_{dense} =0.800, SE±0.103; ψ^{2015}_{dense} =0.752, SE±0.116) and closed canopy (ψ^{2014}_{closed}=0.857, SE±0.132; ψ^{2015}_{closed}=0.755, SE±0.181), with some influence by lower elevations (ψ^{2014}_{lower}=0.514, SE±0.084; ψ^{2015}_{lower}=0.637, SE±0.082) when compared to sparse vegetation (ψ^{2014}_{sparse}=0.405, SE±0.065; ψ^{2015}_{sparse}=0.436, SE±0.067), open canopy (ψ^{2014}_{open}=0.448, SE±0.061; ψ^{2015}_{open}=0.477, SE±0.064) and upper elevations (ψ^{2014}_{upper}=0.462, SE±0.082; ψ^{2015}_{upper}=0.367, SE±0.082). In 2014, helmeted guineafowl were found at highest densities in areas of

sparse vegetation at lower elevations with 828 individuals/km^2 (95% confidence interval: 564 – 1217 individuals) and lowest densities in areas of sparse vegetation at upper elevations 49.1 individuals/km^2 (95% confidence interval: 30.9 – 78.1 individuals). In 2015, helmeted guineafowl were found at highest densities in areas of dense vegetation at higher elevations with 2,085 individuals/km^2 (95% confidence interval: 905 – 4803 individuals) and at lowest densities in areas of sparse vegetation at upper elevations with 38.9 individuals/km^2 (95% confidence interval: 23.81 – 63.81 individuals). The determination of which habitat and landscape factors influence helmeted guineafowl density and occupancy provides some tools necessary to develop more effective management plans. Helmeted guineafowl and other generalist species that utilize similar dense types of habitats and resources could benefit from landscape scale conservation efforts. Species that predate on helmeted guineafowl could benefit from the protection of dense vegetation and closed canopies vicariously because of the population stability of the prey species, which would allow the ecosystem to support larger numbers of predator populations. Our study confirms to biologists and land managers that protection of dense vegetation and closed canopy areas are of higher importance to helmeted guineafowl conservation than protection of sparse vegetation and open canopy areas. On a broader scale, closed canopy areas may be at risk of conversion to rowcrop agriculture through bush encroachment removal plans as demands for food increase. Habitat-specific information may identify risks during landscape conservation planning within the range of the helmeted guineafowl.

Introduction

Habitat degradation and fragmentation are two of the most significant current threats to biodiversity and conservation (Fischer and Lindenmayer 2007, Jetz et al. 2007). Human populations continue to exponentially increase, most notably in developing countries like those found throughout Sub-Saharan Africa (Lotze-Campen et al. 2010). Population increases come with a higher demand for space, food, and other natural resources for human consumption. Intensification of agricultural systems to produce products for human consumption, bioenergy, and livestock consumption is not a new problem, but is increasing in scale in the Middle East and Sub-Saharan Africa (Lotze-Campen et al. 2010). Expansion of agriculture has led an increase in the occurrence of human-wildlife conflict and the consequences to wildlife are depend in large part on whether the land is private or publicly owned (Kinnaird and O'Brien 2012).

An example of the public and private land interface can be found in and around the Northern Tuli Game Reserve (Selier 2008; -22.115909, 29.090403), a 720km^2 wildlife reserve located in the northeastern corner of Botswana (Snyman 2010, Figure 2.2). Unique in its combination of farmland and wild, undeveloped areas, the Northern Tuli Game Reserve provides ideal conditions for researchers to study the effects changes in landscape have on species (Selier 2008). North of the reserve in Zimbabwe, the Tuli Circle (-21.973388, 29.135202) is managed by Zimbabwean National Parks and Wildlife Department and hunting is still allowed. Privately owned hunting farms border the reserve to the east, and to the west and south are farming areas and communal lands used for gazing goats and cattle (Selier 2008, Figure 2.1).

The utilization of avian species as a food source by small, medium and large carnivores such as jackal, leopards, and hyenas is known to occur (Hayward et al. 2006, van der Merwe et al. 2009, Van de Ven et al. 2013). Helmeted guineafowl (*Numida meleagris*) are one of the avian species noted to be preyed upon. However, the effect of the presence of helmeted guineafowl on the ability of an ecosystem to support many predators is poorly understood. Many predator species, such as lions, leopards, and hyenas, have been declining throughout sub-Saharan Africa, partially due to lack of access to food sources (Snyman 2010, van de Merwe et al. 2009, Van de Ven et al. 2013). An ample population of guineafowl could potentially aid in the alleviation of this concern. Relatively accurate estimations of both the mesopredator and helmeted guineafowl population sizes and densities within the available habitat types are a first step towards gaining a better understanding of this relationship. Density and biomass estimations of helmeted guineafowl have been performed in other areas of southern Africa (Monadjem 2002, Malan and Benn 1999), but no studies have been conducted about helmeted guineafowl in the variety of landscapes that make up the Northern Tuli Game Reserve and the area surrounding it in eastern Botswana. Density estimations of helmeted guineafowl in different habitat types could be utilized to explain differences between small and medium predator population sizes across the landscape.

Helmeted guineafowl seem to flourish in areas with a mosaic of open landscape types, such as those found in areas with mixed agricultural fields and natural scrubland savannas (Malan and Benn 1999, Ratcliffe and Crowe 2001). In this aspect, helmeted guineafowl have benefited from some human impact on the environment, which has added some of the necessary habitat features such as open fallow land and small crop fields to new locations as well as the capture and relocation of some wild birds (Little et al. 2000). However, the extensive amount of agricultural fields and crop farming have caused a noted decline in the population sizes of this species, especially in eastern South Africa (Little et al. 2000, Ratcliffe and Crowe 2001). The cause of the decline is attributed to the lack of weeds, arthropods, cover and suitable nesting areas in crop-heavy landscapes. Collisions with powerlines and hybridization with domesticated guineafowl also have negative impacts on helmeted guineafowl populations (Walker et al. 2004, Shaw et al. 2010).

To better understand the issues related to helmeted guineafowl populations and the development of suitable management strategies, basic population information is needed about this species in multiple habitat types. The Northern Tuli Game Reserve is a good representation of the mosaic of landscapes that has become more typical as human encroachment occurs. Therefore, studying helmeted guineafowl populations in this area will allow for more informed decision making for conservation planning. We performed both occupancy analyses and abundance estimations with data collected through sampling. Using occupancy analysis, we determined which habitat types have higher occupancy rates within the study area, as well as which factors are influential in helmeted guineafowl occupancy (MacKenzie and Nichols 2004). Using abundance estimations we estimated the density of helmeted guineafowl throughout the study area, as well as which habitat factors influence the density of helmeted guineafowl. Although they provide somewhat similar information, using both techniques provides a more complete understanding of both population size and range within the study area (MacKenzie and Nichols 2004). Both population size and species range are important to know for conservation of helmeted guineafowl as well as other species which rely upon the helmeted guineafowl and the resources it utilizes. The protection of habitats which support high populations of helmeted guineafowl could be beneficial to land managers and the ecotourism industry indirectly by providing sustenance for more predator species in the study area, leading to overall higher diversity and abundance in the predator communities.

The determination of which environmental covariates have major influence on helmeted guineafowl habitat usage will allow for improved conservation of the species, as well as other species that rely on similar habitats or utilize helmeted guineafowl as a food resource. The goal of this study was to better understand the hierarchy of factors that influence abundance and occupancy of helmeted guineafowl populations within the Northern Tuli Game Reserve and how landscape conservation efforts and usage affect these factors. The specific objectives of this study are:

3. Determine the effects of different habitat types on the presence of helmeted guineafowl on the landscape.
4. Determine the variation in density of helmeted guineafowl in specific habitat types on the landscape.

We will address these questions using surveys to estimate probability of occupancy and density for each habitat type sampled. These surveys will aid in the understanding of habitat usage by helmeted guineafowl throughout the study site and surrounding area.

Methods and Analysis

Study Area

Botswana is a 581,730 km^2 landlocked country located in southern Africa (Senyatso 2011; Figure 2.2). The Northern Tuli Game Reserve (-22.115909, 29.090403) is a 720 km^2 unfenced protected area located in the northeastern corner of Botswana (Snyman 2010, Forssman 2013; Figure 2.2). The Northern Tuli Game Reserve was established as a nature reserve in the mid 1960's when landholders combined their areas into one large reserve as part of a conservation effort (Snyman 2010; Figure 2.2, 2.3). Previous to the reserve's formation, much of the land was used for rowcrop agriculture and grazing livestock (Selier 2008). The Northern Tuli Game reserve is now used for ecotourism and research purposes and has three ecotourism lodges in the areas surrounding it (Snyman 2010). Little to no habitat management is performed in the area, which allows for natural habitat development and change. The two largest contributors to habitat change in the past few years is thought to be flooding and the increasing elephant populations. Flooding influences local seed banks, and in recent years has led to the introduction of different plant species near the rivers. Elephant populations aid in the sustainment of sparse vegetation and open canopy areas through feeding and movement (O'Connor et al. 2007).

The southern park boundary follows the Limpopo River, serving as the Botswana-South Africa border. The eastern park boundary follows the Shashe River, serving as the Botswana-Zimbabwe border. The western boundary is marked by a foot-and-mouth disease fence and the southwestern boundary is marked by a fence along the Motloutse River. The northern boundary follows along the Tuli Circle (-21.973388, 29.135202) in Zimbabwe, which is a managed hunting concession. Animal movements between the Tuli

Circle and the reserve are unrestricted (Snyman 2010; Figure 2.2, 2.3). A ban on commercial wildlife hunting was put into place beginning in January 2014, prohibiting any commercial take of wildlife within the country (Government of Botswana 2014). Effects of the hunting ban on wildlife populations is unknown as the regulation has only recently come into effect. Still, poaching is a common issue in the country and surrounding area, affecting all types of wildlife (Senyatso 2011).

The landscape consists of sandstone and basalt ridges overlooking alluvial floodplains, small rivers, and drainage lines (Forssman 2013). These rivers and drainages flow into the Limpopo, Shashe, and Motloutse rivers during the wet season and form small watering holes during the dry season (Snyman 2010). Multiple habitat types exist within the Northern Tuli Game Reserve, providing an opportunity to compare which landscape features affect helmeted guineafowl occupancy and density (Figure 2.3). Habitats within the Northern Tuli Game Reserve are into five categories (A. Snyman, pers. comm., August 2014) based on vegetation density, water, and canopy cover: 1) Bare Soil, which contains open canopy, little to no vegetation, and no water; 2) Sparse Vegetation, which contains open canopy, little to moderate vegetation, and no water; 3) Grassy/Woody, which contains mixed open and closed canopy, moderate vegetation, and no water; 4) Dense Vegetation/Woodland, which contains closed canopy, dense vegetation, and no water; and 5) Water, which contains open canopy, no vegetation, and water (Figure 2.3).

Study Species

Helmeted guineafowl are a prominent avian species found throughout open-country habitats other than desert and mountain systems in sub-Saharan Africa (Little et al. 2000,

Figure 2.2). Grey-bodied with white flecks and naked red and blue heads with bare casques (Sinclair et al. 2002), these unmistakable birds are Africa's most widespread upland gamebird, inhabiting any open land area with a drinking water source (Little et al. 2000). Helmeted guineafowl weigh an average of 1.5 kilograms with no obvious morphological differences between sexes although males tend to be slightly larger than females (Prinsloo et al. 2005). Helmeted guineafowl are opportunistic omnivores, consuming mostly seeds, bulbs, and stems during the nonbreeding season and invertebrates such as grasshoppers and termites during the breeding season when more protein is needed for mating and egg production (Little et al. 2000). Helmeted guineafowl are found in flocks ranging from 15 to 40 individuals in the nonbreeding season, and are sometimes found in gregarious groups numbering in the thousands around super abundant resources (Little et al. 2000, Ratcliffe and Crowe 2001). Flocks stay together at night in roosts, which can be used for many years.

Helmeted guineafowl (*Numida meleagris*) are one of the most common and recognizable upland gamebird species found throughout sub-Saharan Africa. Both the ease of recognition and commonness of the helmeted guineafowl attributed to their selection as a study species for the project. There is potential to classify the helmeted guineafowl as an umbrella species for the area, meaning that conservation of environmental factors that benefit the study species will also benefit multiple other species that utilize the same landscape features. The helmeted guineafowl is utilized by species as a food source, and could have a role in the ability of an ecosystem to support diverse predator communities (Hayward et al. 2006, van der Merwe et al. 2009, Van de Ven et al. 2013).

Occupancy Analysis

Presence-absence analyses are useful because they take into account the detection probability of the species of interest (MacKenzie et al. 2003). Presence-absence studies involve sampling multiple sites over a short period of time (MacKenzie et al. 2003).The goal is to estimate the proportion of sampling units containing animals, as opposed to abundance estimates which estimate the number of animals within a particular sampling area (Royle and Nichols 2003).

We performed line transect sampling along ten transects throughout the study area (Figure 2.4). The sample area includes two regions, EcoTraining and Central (Figure 2.3, 2.4; Table 2.2). The EcoTraining region consists of six transects located near the EcoTraining camp within the Northern Tuli Game Reserve. The Central region consists of four transects in the middle of the reserve area. The Central transects are closer to many of the tourism lodges in the Northern Tuli Game Reserve, and therefore have more tourists and vehicles compared to the EcoTraining transects. Transects ranged in length from 1.48 km to 14.45 km and were placed within the two regions (Table 2.2). Transects were routed during 2014 along pre-existing roads, following regulations of the reserve to have as little impact on the surrounding environment and landscape as possible. Routes were set out to include all habitat types that exist within the reserve so sampling would be representative of area. The sample area for this study includes samples of all habitat types (Figure 2.3, 2.4). The amount of each habitat type sampled was kept close-to proportional to the amount of each habitat type in the entire Northern Tuli Game Reserve (Table 2.1).

Transects were driven every one to three days at varying times of the day ranging from 06:45CAT (sunrise) to 17:30CAT (sunset) to prevent time bias on data. Data were collected by or in the presence of the primary researcher (Kathryn McCollum) as well as student volunteers from the University of Nebraska and University of Georgia. As each transect was driven, we recorded the number of helmeted guineafowl detected. At each detection a GPS point was created using a handheld GPS unit (Garmin 60CSX) and recorded with a unique individual ID. The distance of the first sighted individual from the transect was determined using a handheld Nikon Monarch laser rangefinder and was noted. The number of individuals was recorded, as well as other observations including cloud cover, time of day, transect number, habitat type, and which side of the transect the individuals were on. Any other notable points about the sighting were also recorded, such as if the individuals were flying, near watering holes, or near large trees.

We used program PRESENCE (MacKenzie et al. 2002, 2003) to obtain occupancy and detection probabilities for the four previously classified habitat types. We used a single-season occupancy model developed by MacKenzie et al. (2002) to account for incomplete detection of helmeted guineafowl in our data. Every completed survey of a section of transect was considered a unique occupancy occasion, giving us 11-26 occasions for each transect section. Transects in the Ecotraining region were surveyed more often than those in the Central region due to logisitical constraints. However, the same habitat types were sampled in both regions so the difference in repetition between the two regions should have little to no impact on the data. A helmeted guineafowl was counted as detected if it was observed during the completion of a transect. Counts for each section were converted to binary data for the occupancy analysis, with a “1” representing detection and a "0" representing no detection. To avoid double counting, the same transect was not surveyed multiple times on the same day during the same time period. For example, if transect one was surveyed at 07:00CAT, it would not be surveyed again until past noon.

We ran five models for both years to determine which covariates affected helmeted guineafowl occupancy, with ψ representing probability of occupancy and p representing probability of detection: ψ (canopy)p(.) to assess the effect of canopy; ψ (elevation)p(.) to assess impact of upper or lower elevation; ψ (vegetation)p(.) to determine effect of sparse or dense vegetation; ψ (.)p(.) as a control model; and ψ(.)p(t) to determine if time had an influence on occupancy. A model was determined to be influential if it had a ΔAIC < 2 (MacKenzie et al. 2002). A goodness-of-fit test was conducted for the global model to assess the fit of the models.

Abundance Estimation

Distance surveys are used to determine the population size or density of a species within a pre-determined area using either transect or point sampling (Anderson et al. 1983). Line transect sampling involves randomly placing transects throughout the study area, then following these transects and recording all sightings of the target species as well as their horizontal distance from transect. Detection can include actual sightings as well as detection by other means such as vocalizations or tracks, but the observer must be able to determine a perpendicular distance from the transect for the detection to be recorded (Buckland et al. 2001).

Data collected during the transect sampling for the abundance estimation was used for this analysis. We utilized program DISTANCE (Buckland et al. 2001) to analyze the transect data to determine a density estimation for the population of helmeted guineafowl within the Northern Tuli Game Reserve. Data were estimated separately by year. Transects were split into approximate 1000 meter sections to provide more detailed habitat classifications, then into four habitat categories with the use of ArcGIS (version 10.3.1) by vegetation density and elevation (Table 2.5, 2.6). Areas labeled as Dense Vegetation/Woody and Grassy/Woody were considered "dense" and areas labeled Sparse Vegetation and Bare Soil were considered "sparse" (Figure 2.3). Areas at elevations higher than 540.0 meters were considered "upper" elevation and areas below this point were considered "lower" elevation. This delineation point was chosen arbitrarily as it was the median point of the range of elevations encountered throughout the ten transects. We then used a global analysis to test which model best fit each category. Four estimators were used to determine the model of best fit for each habitat type: uniform, half-normal, hazard-rate, and negative exponential. Models were evaluated by program DISTANCE using Akaike's Information Criterion (AIC) (Buckland et al. 2001). The model with the lowest AIC score and fewest parameters (K) was considered the best fit. Models were also evaluated using the Kolmogorov-Smirnov goodness-of-fit test, with models having $P>0.05$ considered well-fitted to the data (Buckland et al. 2001).Models which failed the goodness-of-fit test were removed. Right truncation was used as suggested by Buckland et al. (2001) for the removal of outliers.

Results

Occupancy Analysis

Helmeted guineafowl were detected on 169 occasions over 35 transect sections in 2014 and on 147 occasions over 36 transect sections in 2015 within the Northern Tuli Game Reserve. Overall helmeted guineafowl occupancy was most influenced by dense vegetation (ψ^{2014}_{dense} = 0.80, SE ±0.10; ψ^{2015}_{dense} = 0.75, SE ±0.11). The naïve occupancy, or the proportion of sites where helmeted guineafowl were detected, was 0.48 for 2014 and 2015. Detection as a factor of timedid not describe variation in occupancy probability in either field season (w^{2014}_{AIC}=0.00, w^{2015}_{AIC}=0.00). Therefore, no habitat covariate models with time as a survey-specific factor were incorporated (Table 2.3). Probability of occupancy of helmeted guineafowl was strongly associated with dense vegetation in 2014 (ψ^{2014}_{dense}=0.800, SE ±0.103, w_{AIC}=0.72) when compared to sparse vegetation (ψ^{2014}_{sparse}=0.405, SE±0.065; Table 2.4). In 2015 probability of occupancy was influenced by lower elevation (ψ^{2015}_{lower}= 0.637, SE ±0.082, w_{AIC}=0.38) when compared to upper elevation (ψ^{2014}_{upper}=0.462, SE±0.082) and dense vegetation (ψ^{2015}_{dense}= 0.752, SE ± 0.116, w_{AIC}=0.31) when compared to sparse vegetation (ψ^{2015}_{sparse}=0.436, SE±0.067; Table 2.4). However, the null model, ψ(.)p(.), had a ΔAIC value of 1.02 and a w_{AIC} value of 0.22, which provides some evidence that occupancy does not very by vegetation density or elevation in 2015 (Table 2.3).

Abundance Estimation

We recorded 435 observations of helmeted guineafowl by sampling 986.1 km of transect in 2014 and 315 observations of helmeted guineafowl by sampling 1133 km of transect in 2015 (Table 2.5, Figure 2.5). The highest relative abundance (number of individuals/ km^2) for 2014 was in areas of sparse vegetation at lower elevations with 15.37 individuals/km^2 and the lowest relative abundance was in areas of sparse vegetation at upper elevations with 5.107 individuals/km^2 (Table 2.5). In 2015 highest relative abundance was in areas of dense vegetation at upper elevations with 10.68 individuals/km^2 and lowest relative abundance was in areas of sparse vegetation and upper elevation with 3.911 individuals/km^2 (Table 2.5). Densities of helmeted guineafowl ranged from 38.98 individuals/km^2 to 2,085 individuals/km^2 throughout both 2014 and 2015 (Table 2.6, Figure 2.6). In 2014, helmeted guineafowl were found at highest densities in areas of sparse vegetation at lower elevations with 828 individuals/km^2 (95% confidence interval: 564 – 1217 individuals) and lowest densities in areas of sparse vegetation at upper elevations 49.1 individuals/km^2 (95% confidence interval: 30.9 – 78.1 individuals; Table 2.6). In 2015, helmeted guineafowl were found at highest densities in areas of dense vegetation at higher elevations with 2,085 individuals/km^2 (95% confidence interval: 905 – 4803 individuals) and at lowest densities in areas of sparse vegetation at upper elevations with 38.9 individuals/km^2 (95% confidence interval: 23.81 – 63.81 individuals; Table 2.6). Cluster sizes ranged from 15.92 individuals/cluster (95% confidence interval: 11.82 – 21.43 individuals) to 26.75 individuals/cluster (95% confidence interval: 22.00 – 32.53 individuals) in 2014 and 18.58 individuals/cluster (95% confidence interval: 15.23 – 22.65 individuals) to 34. 96 individuals/cluster (95% confidence interval: 26.17 – 46.70 individuals) in 2015 (Table 2.7).

Discussion

Occupancy Analysis

Helmeted guineafowl habitat use is most influenced by vegetation density. The vegetation model was the highest ranking of the models tested for 2014 and the second highest for 2015 (Table 2.3), suggesting helmeted guineafowl use areas with more dense understory vegetation more than areas with thinner understory vegetation. Areas with dense vegetation had the highest occupancy in both years (ψ^{2014}_{dense} = 0.800, SE±0.103; ψ^{2015}_{dense}= 0.755, SE±0.116) when compared to areas with sparse vegetation (ψ^{2014}_{sparse}=0.405, SE±0.065; ψ^{2015}_{sparse}=0.436, SE±0.067; Table 2.4). The choice of dense vegetation over sparse vegetation could be for many reasons, including adequate shelter, protection from predators, and food sources (Little et al. 2000, Ratcliffe and Crowe 2001, van Niekerk 2013). Helmeted guineafowl are small enough in stature to use dense vegetation as cover when avoiding predators (van Niekerk 2002). Areas of dense vegetation may be more plentiful in food resources for helmeted guineafowl, which would have an effect on their occupancy (Ratcliffe and Crowe 2001).

Elevation was the highest ranking model for 2015, but was one of the lowest ranking models for 2014 (Table 2.3). The difference in occupancy between higher and lower elevation is larger in 2015 than 2014, which would somewhat explain the elevation model’s higher ranking (Table 2.4).There could also be a difference in vegetation availability between the two years, which would have an impact on helmeted guineafowl presence (Little et al. 2000).

Abundance Estimation

Helmeted guineafowl were most abundant in areas of dense vegetation at upper elevations (Table 2.4, 2.5; Figure 2.6). In 2014 and 2015, the areas with the highest density estimates were classified as dense vegetation, suggesting that helmeted guineafowl use habitat with thicker understory cover more than habitat with thinner understory cover. Thicker understory cover could be higher in density for multiple reasons, including concealment from predators, food sources, and shelter from extreme temperatures (Little et al. 2000, van Niekerk 2013, 2002). Helmeted guineafowl have disruptive patterning on their feathers (Little et al. 2000) which could be an evolutionary adaption that allows them to remain unseen by predators under the shade of thicker understory. Helmeted guineafowl are omnivorous and have been documented feeding on many types of plant material and insects (Little et al. 2000, Ratcliffe and Crowe 2001, van Niekerk 2013), which could be more abundant in areas of thicker vegetation. Different types of vegetation would support different insect communities, both of which could affect the ability of a habitat to support helmeted guineafowl populations.

I found that areas of higher elevation had greater densities of helmeted guineafowl when compared to areas of lower elevation, but only when dense vegetation was also part of the habitat type (Table 2.6). Areas of upper elevation with sparse vegetation had the lowest abundance estimates in both field seasons, with 49.14 individuals/km^2 (95% confidence interval: 30.91 – 78.12 individuals) in 2014 and 38.98 individuals/km^2 (95% confidence interval: 23.81 – 63.81 individuals) in 2015 (Table 2.6). These results suggest that dense vegetation plays a key role in helmeted guineafowl abundance.

Large density estimates for helmeted guineafowl in dense habitat types (2015: 2,085 individuals/km^2, 95% confidence interval: 905.6 – 4803 individuals) at our study site support previous observations of group sizes throughout southern Africa in areas of suitable habitat (Table 2.4, Table 2.5, Figure 2.6; Little et al. 2000, Ratcliffe and Crowe 2001) and allowed for robust analysis of which habitat variables influence helmeted guineafowl density. High population abundance is one of the reasons helmeted guineafowl are thought to have an impact on predator populations in the areas they inhabit. The biomass provided by helmeted guineafowl in an environment can be quite significant (Monadjem 2002). As a resilient and adaptive species, this could mean that as other prey species decrease in number, helmeted guineafowl could continue to help support the predator population. Gaps remain in the knowledge of predator species' utilization of helmeted guineafowl within the Northern Tuli Game Reserve, such as which specific species may benefit most from helmeted guineafowl as a food source. Research could also continue to determine how helmeted guineafowl range impacts and relates to predator species presence and abundance. Our study serves as a baseline for helmeted guineafowl populations in the area by describing their use of habitats consisting of dense vegetation and closed canopies instead of sparse vegetation and open canopy areas.

Implications

We were able to determine that the helmeted guineafowl population in this study used areas of closed canopies and dense vegetation when compared with areas of open canopies and sparse vegetation through the use of both presence and abundance estimations. The usefulness of the utilization of tools to understand the presence-abundance relationship of species has been recognized by many, as shown in the paper done by Gaston (1999). One of the implications of presence-abundance relationships addressed by Gaston (1999) is in relation to population monitoring, in which species presence and abundance were used over time to observe changes within populations and communities. We found that helmeted guineafowl populations in the Northern Tuli Game Reserve have relatively high probabilities of occupancy and density estimates throughout the study area (Table 2.4). Biologists can use the information that helmeted guineafowl are in greater presence and abundance in areas of dense vegetation for future management plans throughout the area by incorporating dense vegetation at upper elevations into conservation areas.

Notable changes between the survey years included a difference in the timing of rainfall, with 2014 representing an average year for schedule of rainfall and 2015 having a late rain at the end of the wet season. The later rain in 2015 caused vegetation to persist late into the fall and winter, which could have allowed for longer foraging opportunities as well as more cover for helmeted guineafowl.

The habitat types shown to be used by helmeted guineafowl are similar to the habitat types that increase with bush encroachment. Bush encroachment is an issue for many groups of pastoralists throughout sub-Saharan Africa (Senyatso 2011). Helmeted guineafowl are fairly adaptable to habitat changes and can survive in a mosaic landscape (Ratcliffe and Crowe 2001), therefore bush encroachment would not have as much of an impact on their populations as other local species. Muntifering et al. (2006) states that bush encroachment has led to an increase in certain predators in some areas, which is thought to be associated with an increase in prey species availability. Helmeted guineafowl are utilized as prey by multiple predator species (Hayward et al. 2006, van der Merwe et al. 2009, Van de Ven et al. 2013), so increases in helmeted guineafowl populations could lead to increases in certain predator species. Although it is an issue that has not been studied in the Northern Tuli Game Reserve, bush encroachment is a major issue in other parts of the kori bustard's range for both livestock and wildlife (Senyatso 2011, Börner et al. 2007).

Agricultural expansion poses a threat to helmeted guineafowl populations as well. Sub-Saharan Africa contains some of the most unused cropland in the world (Jenkins 2003), and as demands for food increase with the increasing human populations, so will the pressures to utilize all available lands for rowcrop agriculture. Although helmeted guineafowl have been shown to benefit in some ways from increased access to agricultural fields, they are negatively impacted by an overabundance of cropland due to the detrimental losses of arthropods and weeds (Ratcliffe and Crowe 2001). Absence of dense vegetation leaves helmeted guineafowl lacking for nesting space as well as cover from predators.

Land managers can benefit from knowledge on helmeted guineafowl habitat usage through an ecotourism perspective. Conservation of dense vegetation at upper elevation areas are helpful in the preservation of not only helmeted guineafowl, but to other charismatic megafauna that utilize them as a potential food source such as jackals, leopards and lions (van de Merwe et al. 2009, Van de Ven et al. 2013). Having these species within a reserve will sustain and possibly increase ecotourism in the area, which will allow for more funding for conservation and protection of all species found there.

Summary

Our study illustrates some of the habitat types affecting the space utilization of helmeted guineafowl in a landscape made up of a mosaic of land uses. Occupancy of helmeted guineafowl was influenced by vegetation, with dense vegetation used more than sparse vegetation (ψ^{2014}_{dense}= 0.800, SE±0.103; ψ^{2014}_{sparse}=0.405, SE±0.065; ψ^{2015}_{dense}= 0.752, SE±0.116; ψ^{2015}_{sparse}=0.436, SE±0.067). We found that helmeted guineafowl were found at higher densities in areas of dense vegetation at upper elevations (2085 individuals/km^2, 95% confidence interval: 905.6 – 4803).Helmeted guineafowl are common throughout the study area, and it is through this common-ness that their importance to the ecosystem is found. As a species, helmeted guineafowl have the potential to increase an ecosystems ability to support larger predator communities through the amount of biomass they provide. Helmeted guineafowl population numbers are large in certain habitat types, which could be beneficial to many small and mesopredator species. Our work shows that vegetation density and elevation both influence helmeted guineafowl abundance and probability of occupancy. Conservation of helmeted guineafowl will be beneficial not only for the ecosystem, but for ecotourism as well because of the simultaneous conservation of other charismatic megafauna which utilize the same habitat types. To maintain helmeted guineafowl populations in both presence and abundance, emphasis should be placed on the preservation of sparse vegetation areas throughout their range which can be accomplished through the intentional conservation by landowners of these habitat types and the avoidance of conversion of land use to agricultural fields.

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Tables

Table 2.1. Proportion of habitat types sampled compared to overall amount of habitat type determined from vegetation layers in ArcGIS (version 10.3.1) within the Northern Tuli Game Reserve, Botswana.

Habitat Type	Area Sampled (km^2)	Area Available (km^2)	Proportion Sampled
Bare Soil	7.25	89.2	0.0812
Sparse Vegetation	43.26	410.1	0.1054
Grassy/Woody	11.75	169.9	0.0691
Dense Vegetation /Woodland	10.05	52.1	0.1928

Table 2.2. Location, length and brief habitat description of transects used for the helmeted guineafowl research project from June – July 2014 and May – July 2015 in the Northern Tuli Game Reserve, Botswana. Locations representative of two regions sampled within the reserve, with EcoTraining defined as area around the EcoTraining camp and Central defined as area in the inner part of the reserve.

Transect	Location	Length (km)	Brief Habitat Description
T1	EcoTraining	4.80	Croton forest, basalt ridges, sandstone ridges, floodplain, open grassland, acacia thicket
T2	EcoTraining	5.18	Marsh/floodplain, basalt ridges, sandstone ridges, sage plains, open grassland, acacia thicket
T3	EcoTraining	1.48	Sandstone ridges, mopane thicket
T4	EcoTraining	5.71	Sandstone ridges, floodplain, croton forest, open grassland, acacia thicket
T5	EcoTraining	3.88	Sandstone ridges, acacia thicket, open grassland
T6	EcoTraining	3.88	Open grassland, mopane thicket
T7	Central	8.45	Appleleaf forest, open grassland, acacia thicket, croton forest
T8	Central	9.01	Croton forest, open grassland, acacia thicket, mopane thicket

T9	Central	14.45	Mopane thicket, open grassland, sandstone ridges, basalt ridges, riverbed, croton forest
T10	Central	8.14	Croton forest, riverbed, mopane thicket, sandstone ridges, basalt ridges

Table 2.3. Occupancy (ψ) and detection (p) model selection results for helmeted guineafowl (*Numida meleagris*) in the Northern Tuli Game Reserve, Botswana over 2 field seasons during June-July 2014 and May 2015-July 2015. K represents number of parameters. ΔAIC represents difference between model and best-fitting model (model with lowest AIC).

Model	Year	K	AIC[1]	ΔAIC	AIC weight
ψ(vegetation)p(.)	2014	4	746.71	0.00	0.72
ψ(canopy)p(.)	2014	4	749.88	3.17	0.14
ψ(.)p(.)	2014	2	750.50	3.79	0.10
ψ (elevation)p(.)	2014	4	754.30	7.59	0.01
ψ(.)p(t)	2014	21	771.06	24.35	0.00
ψ (elevation)p(.)	2015	4	791.20	0.00	0.38
ψ(vegetation)p(.)	2015	4	791.58	0.38	0.31
ψ(.)p(.)	2015	2	792.22	1.02	0.22
ψ (canopy)p(.)	2015	4	794.42	3.22	0.07
ψ(.)p(t)	2015	27	807.44	16.24	0.00

[1]Akaike's Information Criterion

Table 2.4. Helmeted guineafowl (*Numida meleagris*) occupancy (ψ) estimates, standard errors (SE) and 95% confidence intervals (CI) for habitat covariates of occupancy models from two field seasons, June 2014 – July 2014 and May 2015 – July 2105 in the Northern Tuli Game Reserve, Botswana.

Covariate	Year	ψ	SE	95% CI	
Canopy	2014				
Open		0.4482	0.0619	0.3320	0.5702
Closed		0.8574	0.1323	0.4189	0.9805
Vegetation	2014				
Sparse		0.4054	0.0653	0.2862	0.5369
Dense		0.8004	0.1033	0.5302	0.9344
Elevation	2014				
Upper		0.4628	0.0825	0.3101	0.6229
Lower		0.5148	0.0846	0.3533	0.6732
Canopy	2015				
Open		0.4775	0.0641	0.3558	0.6019
Closed		0.7558	0.1810	0.3118	0.9548
Vegetation	2015				
Sparse		0.4364	0.0679	0.3107	0.5708
Dense		0.7529	0.1169	0.4707	0.9126
Elevation	2015				
Upper		0.3676	0.0822	0.2252	0.5376
Lower		0.6378	0.0828	0.4659	0.7805

Table 2.5. Relative abundance (individuals/km^2) for helmeted guineafowl (*Numida meleagris*) in each habitat type surveyed June-July 2014 and May-July 2015 within the Northern Tuli Game Reserve, Botswana.

Year	Vegetation	Elevation	Model Selection	Number of Observations	Number of Individuals	Number of Samples	Effort (km)	Relative Abundance
2014	Dense	Lower	Half-normal	48	837	118	102.74	8.146
2014	Dense	Upper	Negative Exponential	39	621	103	102.79	6.041
2014	Sparse	Lower	Hazard Rate	242	6478	459	421.33	15.37
2014	Sparse	Upper	Hazard Rate	106	1835	374	359.26	5.107
2015	Dense	Lower	Hazard Rate	46	1297	148	130.13	9.966
2015	Dense	Upper	Negative Exponential	27	949	89	88.821	10.68

2015	Sparse	Lower	Hazard Rate	163	5517	592	539.31	10.22
2015	Sparse	Upper	Negative Exponential	79	1468	390	375.27	3.911

Table 2.6. Density estimates (individuals/km^2) for helmeted guineafowl (*Numida meleagris*) in each habitat type surveyed June-July 2014 and May-July 2015 within the Northern Tuli Game Reserve, Botswana.

Year	Vegetation	Elevation	Density of Animals Estimation (birds/km^2)	95% CI	GOF P value	%CV
2014	Dense	Lower	64.29	35.78 – 115.5	0.7035	30.39
2014	Dense	Upper	156.25	86.53 – 282.1	0.32308	30.62
2014	Sparse	Lower	828.61	564.1 – 1217	0.09203	19.78
2014	Sparse	Upper	49.145	30.91 – 78.12	0.70819	23.94
2015	Dense	Lower	223.41	119.7 – 416.9	0.41350	32.45
2015	Dense	Upper	2085.8	905.6 – 4803	0.49428	44.03
2015	Sparse	Lower	660.45	404.1 – 1079	0.4403	25.41

2015	Sparse	Upper	38.984	23.81 – 63.81	0.31189	25.49

Table 2.7. Density of cluster estimates (individuals/km^2) and mean cluster size for helmeted guineafowl (*Numida meleagris*) in each habitat type surveyed June-July 2014 and May-July 2015 within the Northern Tuli Game Reserve, Botswana.

Year	Vegetation	Elevation	Density of Clusters Estimation (95% CI)	%CV	Expected Cluster Size (95% CI)	Mean Cluster Size (95% CI)
2014	Dense	Lower	3.37 (2.04 – 5.58)	25.87	19.045 (13.847 – 26.193)	17.438 (14.110 – 21.550)
2014	Dense	Upper	8.8828 (5.3068 – 14.869)	26.50	17.590 (12.913 – 23.960)	15.923 (11.829 – 21.434)
2014	Sparse	Lower	20.395 (14.255 – 29.180	18.39	40.628 (35.211 – 46.877)	26.756 (22.007 – 32.531)
2014	Sparse	Upper	2.4444 (1.6078 – 3.7164)	21.56	17.311 (14.753 – 20.313)	20.105 (16.372 – 24.690)
2015	Dense	Lower	4.8302 (2.8751 – 8.1146)	26.74	46.253 (32.033 – 66.786)	28.196 (20.387 – 38.995)

2015	Dense	Upper	36.559 (17.249 – 77.487)	39.16	57.053 (37.861 – 85.975)	34.963 (26.175 – 46.702)
2015	Sparse	Lower	10.142 (6.435 – 15.983)	23.46	65.120 (53.725 – 78.931)	33.847 (28.123 – 40.735)
2015	Sparse	Upper	1.8730 (1.2086 – 2.9025)	22.57	20.814 (16.456 – 26.325)	18.582 (15.239 – 22.659)

Figures

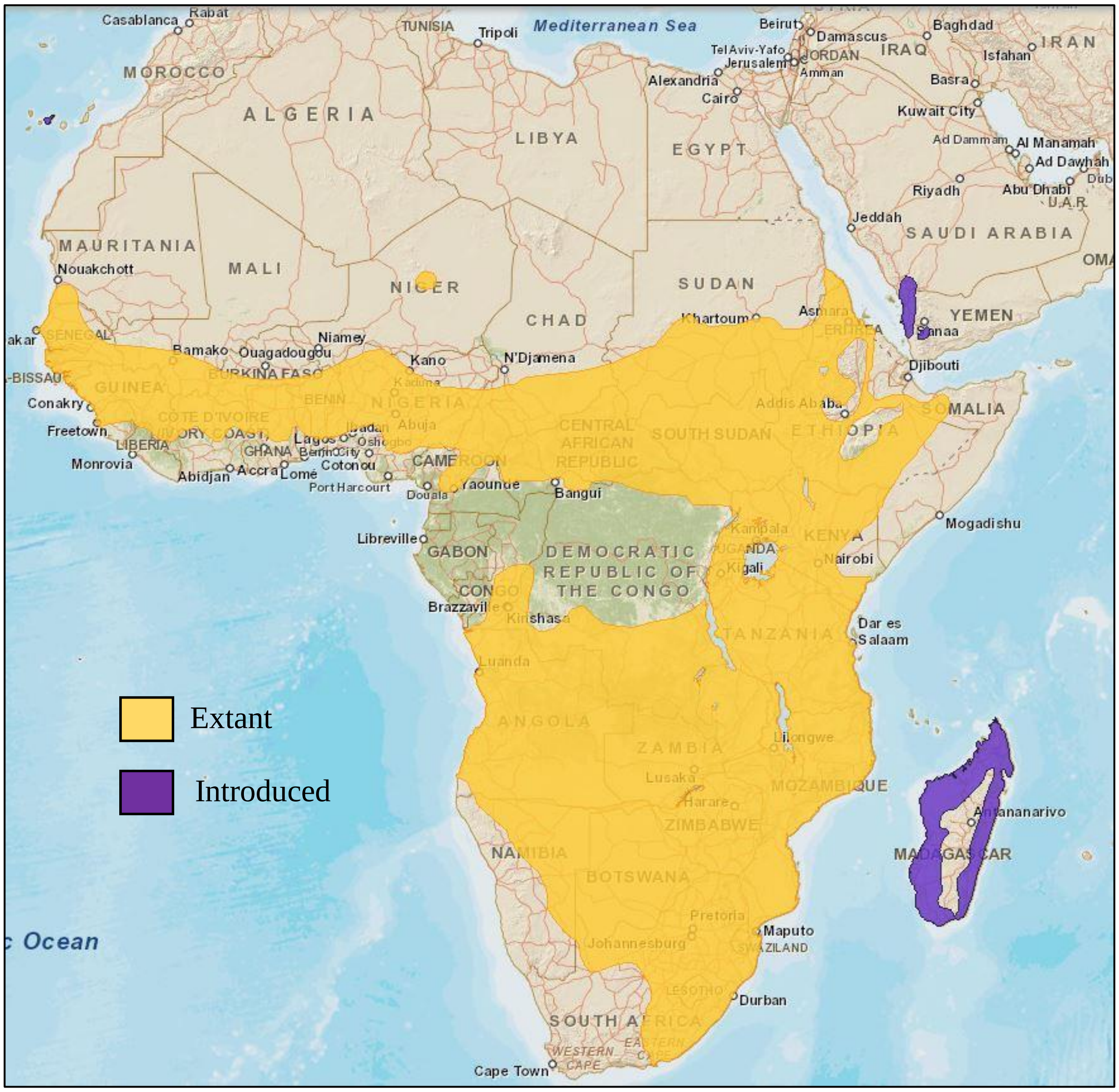

Figure 2.1. Range of helmeted guineafowl (*Numida meleagris*) as of 2014 within the African continent and surrounding area (Birdlife International and NatureServe 2014).Purple areas representative of locations of helmeted guineafowl introduction into habitat, yellow representative of native range of helmeted guineafowl.

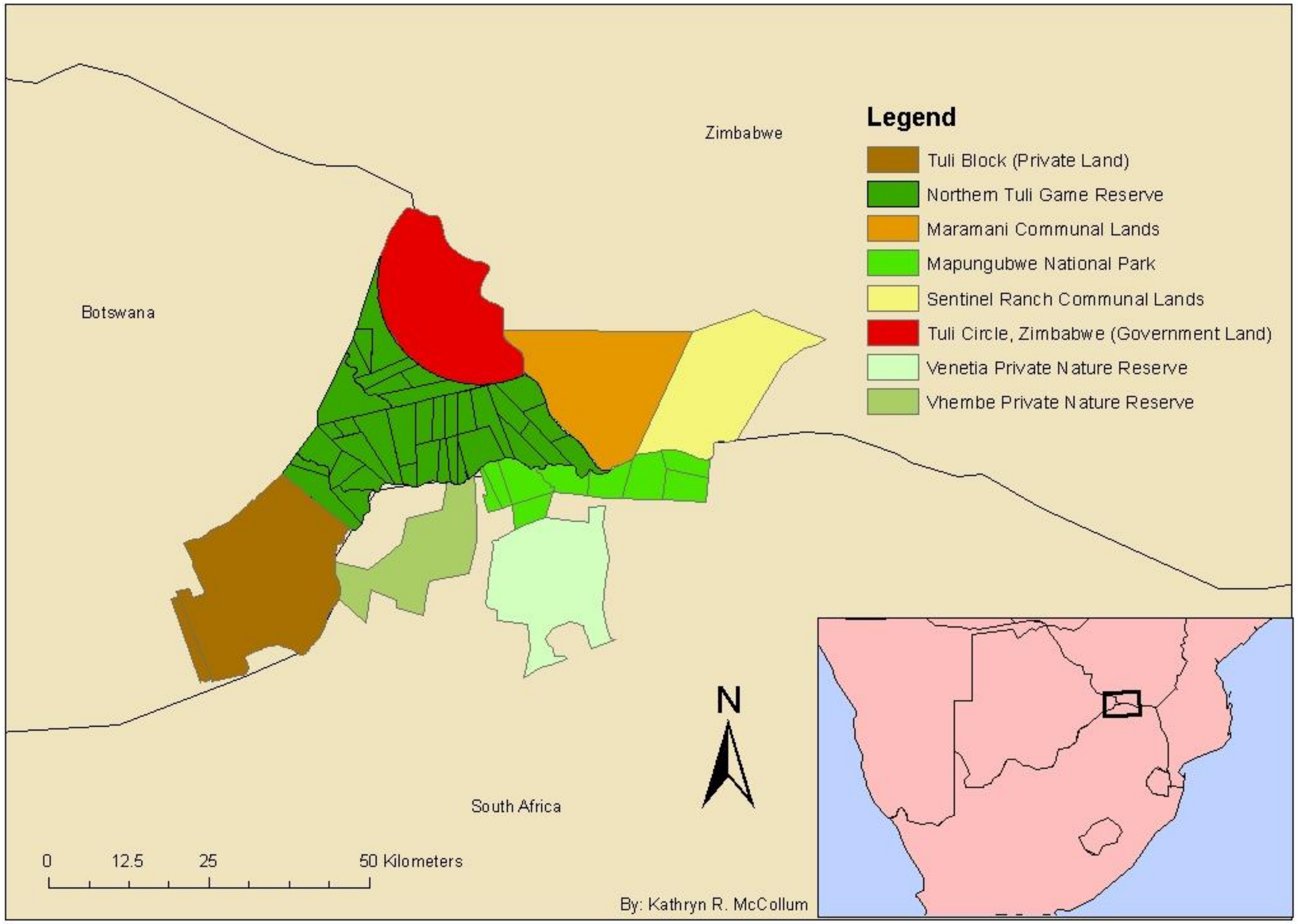

Figure 2.2. Ownership of property for the Northern Tuli Game Reserve and surrounding areas in southern Africa. Southern Africa represented in inset with study site and surrounding area enlarged from the black square. Ownership boundaries are representative of different wildlife conservation regulations

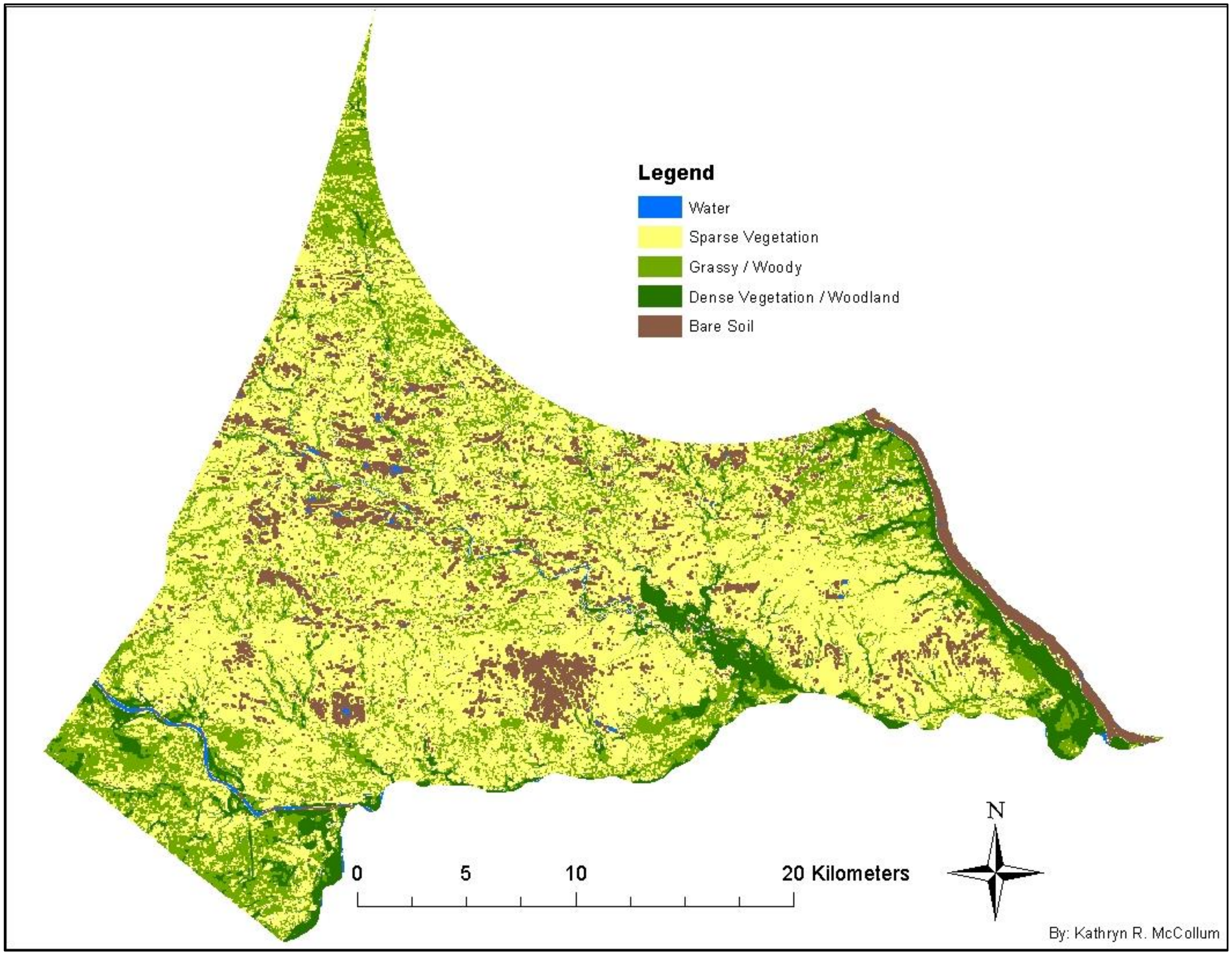

Figure 2.3. General habitat types found throughout the Northern Tuli Game Reserve, Botswana surveyed for the helmeted guineafowl research project from June-July 2014 and May-July 2015.

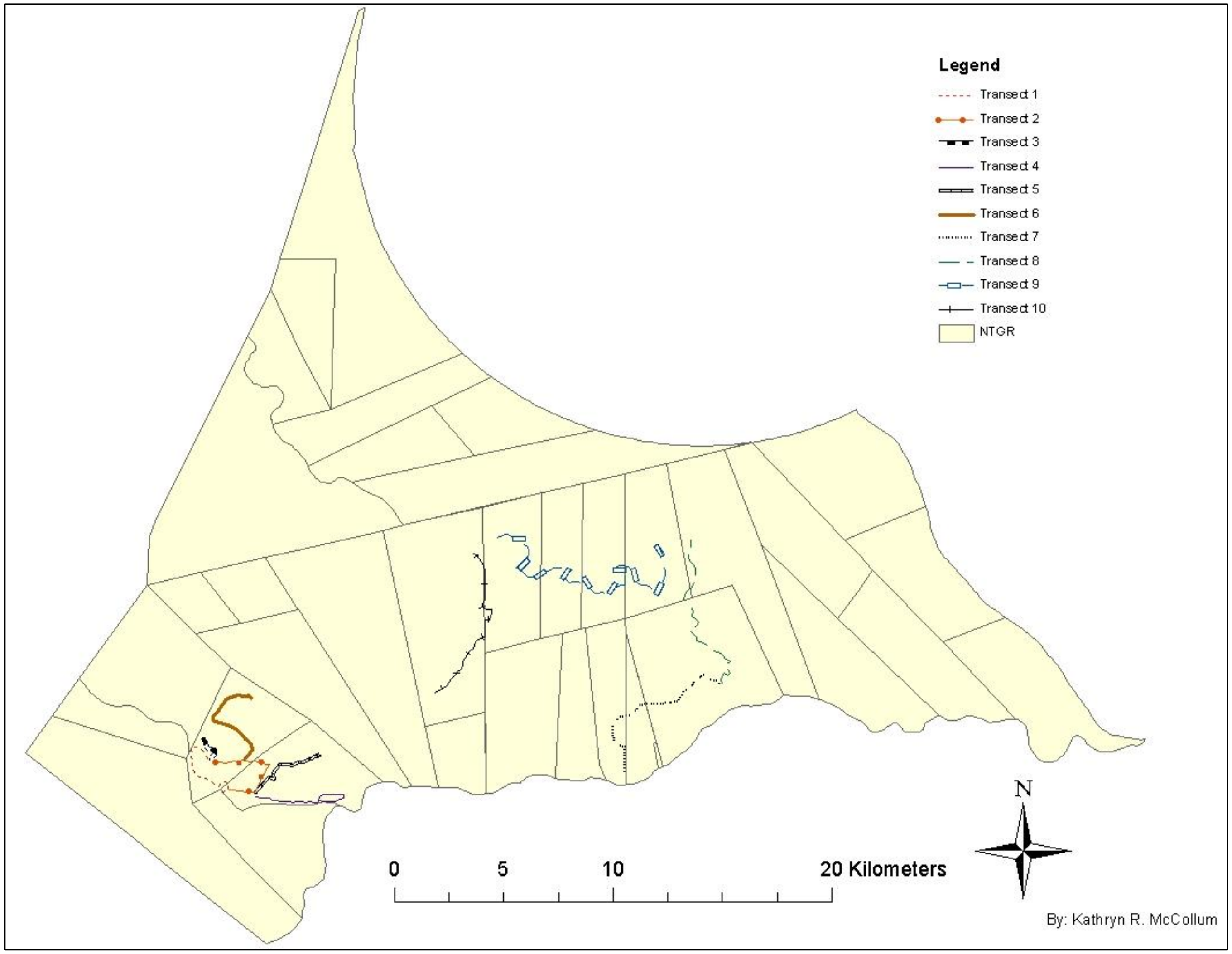

Figure 2.4. Layout of study transects for helmeted guineafowl research project within the Northern Tuli Game Reserve, Botswana from June – July 2014 and May – July 2015

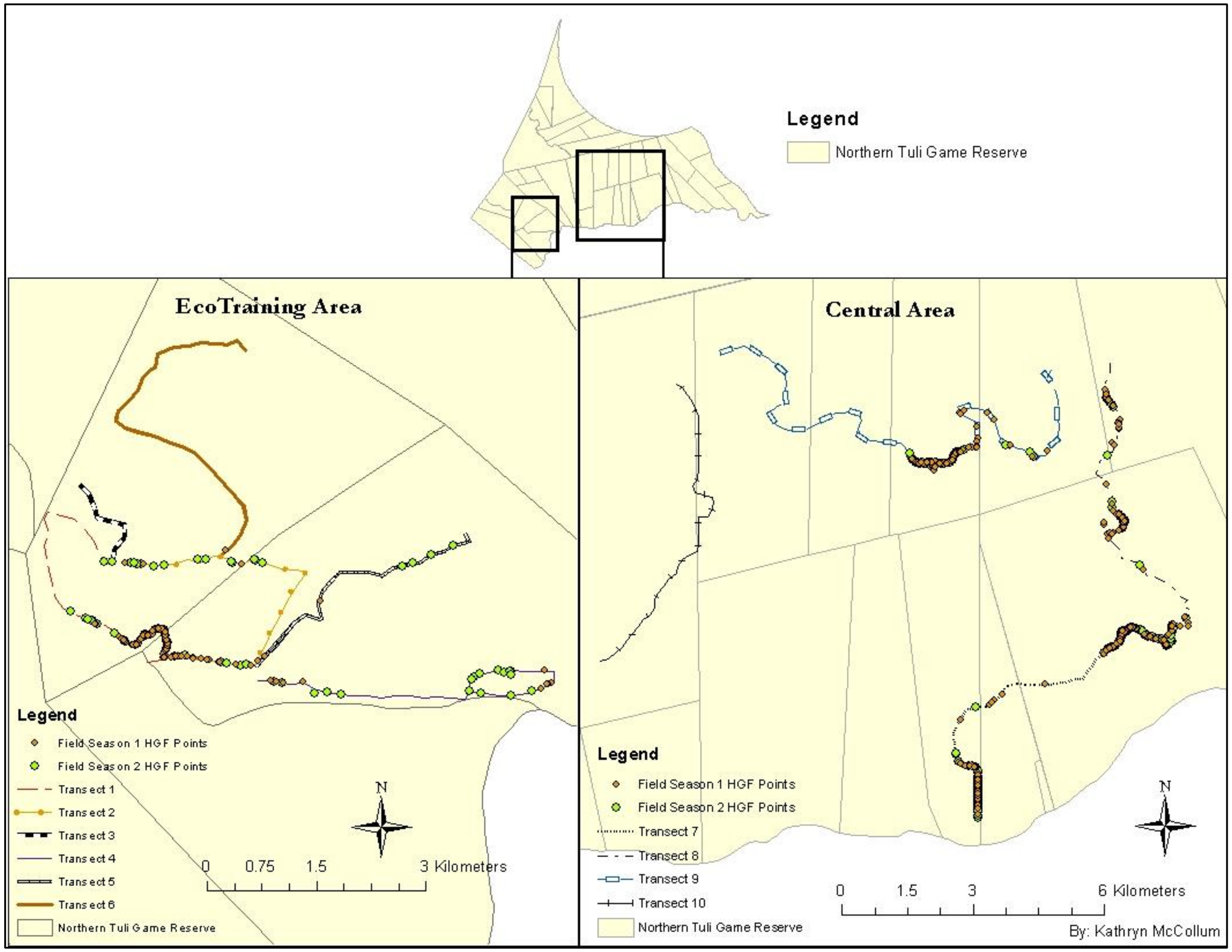

Figure 2.5. Helmeted guineafowl detection points throughout the Northern Tuli Game Reserve, Botswana from June-July 2014 and May – July 2015. Two specified areas, EcoTraining and Central, were only regions sampled

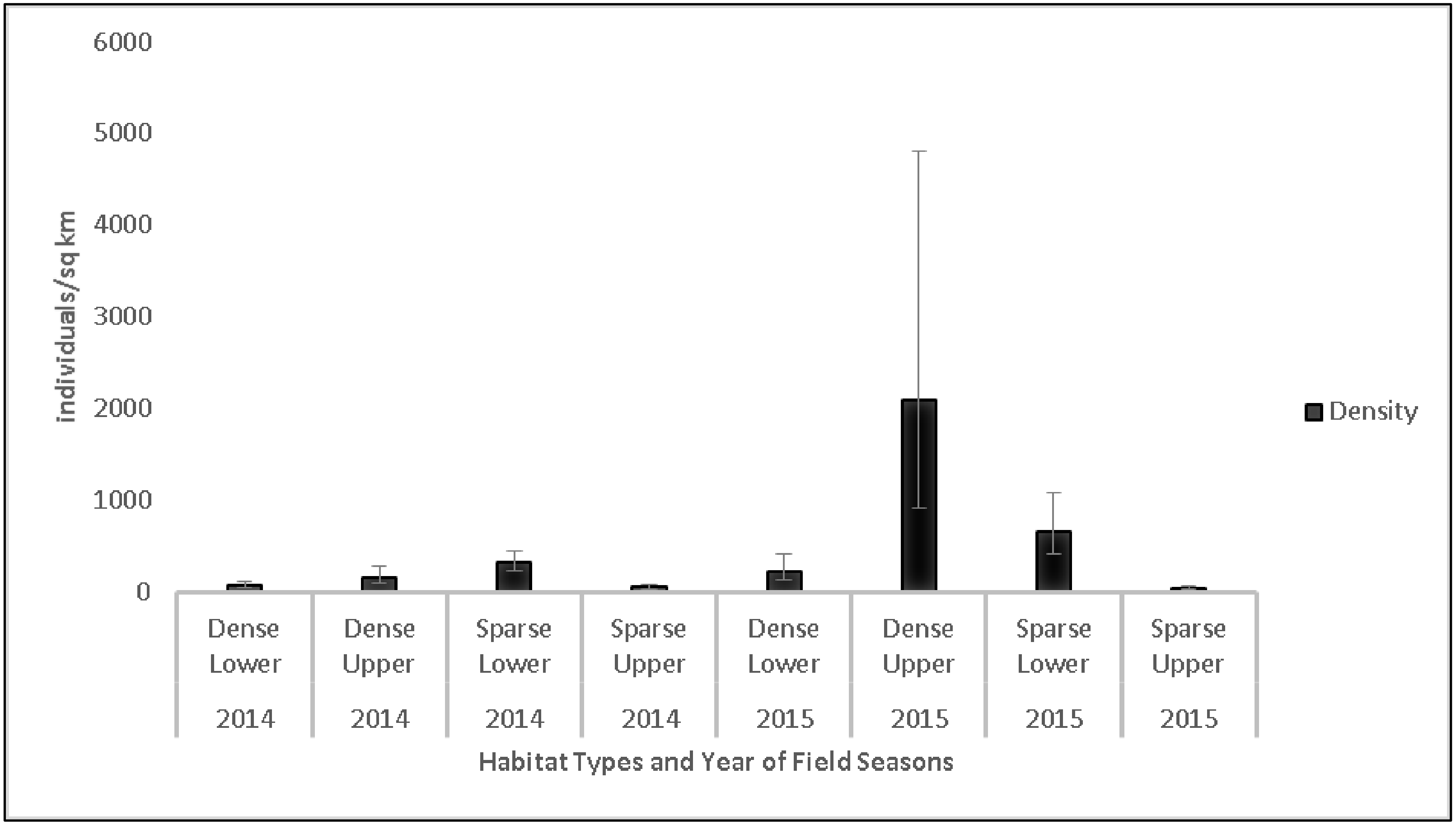

Figure 2.6. Density estimates for helmeted guineafowl in different habitat types throughout the Northern Tuli Game Reserve, Botswana for two field seasons from June – July 2014 and May – July 2015

Chapter 3 – Implications of Landscape Conservation Planning on Private and Public Lands in Southern Africa

Abstract

As human populations continue to increase around the world, land use change is inevitable. Landscape conservation planning is one useful strategy to limit possible negative impacts to wildlife and take advantage of new opportunities created by changes in land use. Here we address current challenges to conservation throughout sub-Saharan Africa and possible options for alleviating some of the human impact currently being experienced by numerous species throughout the continent. We conclude by focusing on a particular area in southern Africa, the Northern Tuli Game Reserve, and view how conservation planning on a landscape scale could positively impact many species throughout the reserve and the surrounding areas. We suggest concentrating on improving the connectivity of reserves in future landscape conservation plans through the preservation of key habitat types that aid in the conservation of important as well as conspicuous species.

Introduction

The largest cause of ecological changes is anthropogenic effects, specifically human-caused habitat degradation and fragmentation (Fischer and Lindenmayer 2007). Balmford (2001) showed that areas of higher human populations and intensive agricultural practices are often located near areas of higher biodiversity and species richness, which makes the importance of understanding and mitigating human impact on the surrounding environment even more substantial. As human populations increase, the negative effects accompanying this change including pollution, climate change, habitat fragmentation and habitat destruction will increase as well (Jetz et al. 2007, Jenkins 2003, Pimm and Raven 2000).

To counteract the issue of habitat fragmentation, efforts must be put towards conservation on a landscape scale through the incorporation of a matrix of land uses in a way that not only benefits wildlife, but does not negatively affect humans (Sanderson et al. 2002). Landscape conservation planning is essential to habitat connectivity in areas currently or on the verge of becoming fragmented (Saura and Pascuak-Hortal 2007). Examples of programs utilizing landscape conservation planning include a wide range of conservation minded groups such as the Wildlife Conservation Society, which proposed a "landscape species" method of conservation planning (Sanderson et al. 2002).

The objective of this chapter is to showcase the positive impact landscape conservation planning could have on fragmented habitats by increasing habitat connectivity and health in sub-Saharan Africa using the Northern Tuli Game Reserve as a case study.

Challenges

Increasing human population sizes are not a new issue to Africa, but over recent decades the rate of growth has increased and the human population is predicted to rise to approximately 8.9 billion by 2050 (Cohen 2003). With this increase in population size comes an increase in demand for resources (Cohen 2003, Lotze-Campen et al. 2010), resulting in land use changes from natural landscapes into agricultural use (Jenkins 2003, Lotze-Campen et al. 2010). As technology advances, so does the ability to convert new areas for agricultural development. Over half of the currently unused suitable cropland in the world is found in South America and sub-Saharan Africa, which indicates that if human population sizes continue to increase, the pressure of agriculture will continue to intensify in these areas (Jenkins 2003) with associated pressure on resident wildlife populations. This brings us to the core issue of land use change, which is the struggle to find balance between environmental conservation and human needs for land and space.

To better understand and cope with these changing landscapes throughout the continent of Africa, general views need to evolve from the ideas of “old Africa”, with its sweeping open savannas and large untouched jungles, to “new Africa”, which contains much more of a mosaic of land types. Habitat fragmentation resulting from increasing agricultural pressure has become common throughout sub-Saharan Africa. Consequently, the conservation focus has changed from trying to create large new preserves full of untouched habitat to connecting reserves already in existence through corridors and other environmental pathways. If African landscapes continue to become less connected over time, problems for wildlife conservation could be seen across species, landscapes, and ecosystems.

Case Study: Northern Tuli Game Reserve

Here we will emphasize the impact that beneficial conservation planning could have on the landscape scale in and around the Northern Tuli Game Reserve in eastern Botswana. We focus on two unique and important bird species found in the region, the kori bustard (*Ardeotis kori*) and helmeted guineafowl (*Numida meleagris*), and how this planning could positively impact both species, as well as other more conspicuous species which utilize the same resources and habitat types such as African lions (*Panthera leo*) and African elephants (*Loxodonta spp*).

The Northern Tuli Game Reserve (-22.115909, 29.090403) is a 720 km^2 unfenced wildlife reserve located in eastern Botswana (Figure 3.1). Established as a nature reserve in the mid 1960's when landholders combined their areas into one large reserve as part of a conservation effort (Snyman 2010), previously much of the land was used for rowcrop agriculture and grazing livestock (Selier 2008). The Northern Tuli Game Reserve is an association of landowners who make decisions regarding reserve land use and access. Considerations are given towards environmental conservation with an emphasis on ecotourism and research throughout the area, which has three ecotourism lodges (Snyman 2010). Little to no habitat management is performed in the area, which allows for natural habitat development and change. The two largest contributors to habitat change in the past few years is thought to be flooding and the increasing elephant populations. Flooding influences local seed banks, and in recent years has led to the introduction of different plant species near the rivers. Elephant populations aid in the sustainment of sparse vegetation and open canopy areas through feeding and movement (O'Connor et al. 2007).

The northern boundary follows along the Tuli Circle (-21.973388, 29.135202) in Zimbabwe, which is a managed hunting concession. Animal movements between the Tuli Circle and the reserve are unrestricted (Snyman 2010, Figure 3.1). A ban on commercial wildlife hunting was put into place beginning in January 2014, prohibiting any commercial take of wildlife within the country (Government of Botswana 2014). Effects of the hunting ban on wildlife populations is unknown as the regulation has only recently come into effect. Still, poaching is a common issue in the country and surrounding area, affecting all types of wildlife (Senyatso 2011).

Helmeted guineafowl and kori bustards are two important species found in eastern Botswana. Helmeted guineafowl are one of the most common upland gamebirds in sub-Saharan Africa, able to inhabit any open land area access to water (Little et al. 2000). The helmeted guineafowl common prey item of mesopredators, and plays a role in the ability of an ecosystem to support diverse predator communities (Hayward et al. 2006, van der Merwe et al. 2009, Van de Ven et al. 2013). Kori bustards are also common in and around the Northern Tuli Game Reserve, but their populations have been on a noted decline over the past few years (Herremans 1998, Sinclair et al. 2002). In a recent study done by Senyatso et al. (2013), it was determined that the kori bustard’s species range has decreased by 8% in southern Africa since the early 1900s, and that number of individuals within the range has greatly decreased over this time. The decline of the kori bustard is representative of a decline in available habitat through habitat fragmentation and degradation.

Vegetation and cover both have influence of kori bustard and helmeted guineafowl density and occupancy (McCollum 2015), interestingly the two species seem to use the opposite habitat type of each other, even though they can be seen in the same locations. Kori bustards were found at higher densities in areas of sparse vegetation, whereas helmeted guineafowl were seen at higher densities in areas of dense vegetation. Vegetation and canopy were shown to have an effect on occupancy as well, with more kori bustards present in areas of sparse vegetation with open canopy and more helmeted guineafowl present in areas of dense vegetation with closed canopy. These factors can be used for landscape conservation planning for both species through the protection of both habitat types, which will aid in the conservation of other important species that utilize the same areas. For example, the same sparse vegetation and open canopy habitat used by kori bustards is utilized by more high profile species such as elephants (Selier 2008), hyenas (Cooper et al. 1999), and lions (Snyman 2010). Elephants play a key role in ecotourism throughout the Northern Tuli Game Reserve, which has a large elephant population. The preservation of these species will aid in the sustainment of ecotourism as a feasible business industry, which in turn contributes to conservation of the ecosystem as a whole. A similar case can be modeled for the dense vegetation habitat used by helmeted guineafowl, which could serve as shelter for other prey species.

The matrix of habitats surrounding the Northern Tuli Game Reserve is one that for the moment is stable in its land use (Figure 3.1). Currently, landowners in the reserve have a focus towards conservation, which is supported through the business of ecotourism. However, motivations of landowners could change over time, especially if demand for agricultural lands increases in the coming years. To ensure the longest benefit of ecotourism in the area, landscape conservation is a crucial piece of planning that can incorporate the preservation of habitat types utilized by conspicuous wildlife such as elephants as well as species of concern like the kori bustard.

Summary

As we look to the future for land use mitigation options such as the creation of more corridors and reserves, it will be important to keep in mind which habitat factors are most influential in important species environments. Decisions for which habitat types to incorporate into new conservation planning are dependent on which species are trying to be conserved. For kori bustards, a species listed as near-vulnerable on the IUCN Red List, this information will be pertinent in creating landscape corridors to continue to allow for population connectivity.

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Figure

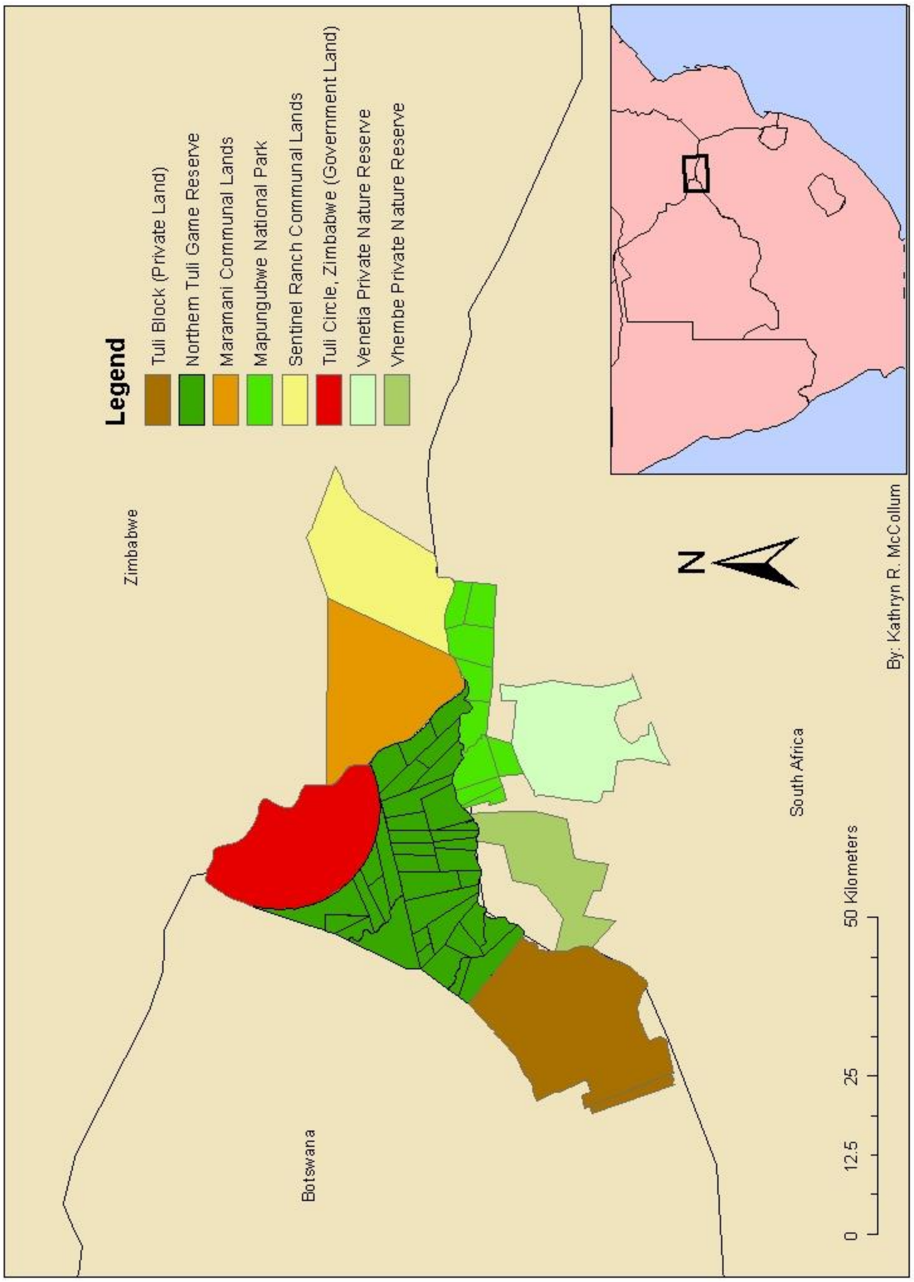

Figure 3.1. Ownership of property for the Northern Tuli Game Reserve and surrounding areas in southern Africa. Southern Africa represented in inset with study site and surrounding area enlarged from the black square. Ownership boundaries are representative of different wildlife conservation regulations