Sokoine University of Agriculture

MSc Dissertation

Breeding Population Status and Habitat Preferences of White-Backed Vulture in Ikorongo-Grumeti Game Reserves and Ikona Wildlife Management Area in Western Serengeti

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May 2024

BREEDING POPULATION STATUS AND HABITAT PREFERENCES OF WHITE-BACKED VULTURE IN IKORONGO-GRUMETI GAME RESERVES AND IKONA WILDLIFE MANAGEMENT AREA IN WESTERN SERENGETI

Dissertation Submitted to Sokoine University of Agriculture in Partial Fulfilment of the Requirement for the Degree of Master of Science in Wildlife Management and Conservation

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May 2024

EXTENDED ABSTRACT

White -backed vulture (*Gyps africanus*) is a critically endangered species since its population has significantly declined across its geographical ranges. Apparently, vultures provide the ecosystem with cleaning service hence reduced risk of disease spread through their feeding behaviour. Studying the breeding population status and habitat of these species is of paramount importance in maintaining a healthy population as well as managing and mitigating threats to the species and their associated habitats.

This study aimed at investigating the breeding population status and the nesting preferences of the *Gyps africanus* population in the Ikorongo-Grumeti Game Reserves and Ikona wildlife management area (WMA) in the western corridor of the Greater Serengeti Ecosystem along the primary riparian zones (PRZ). The PRZ consisted of the Grumeti, Manchira, Nyabeu, Samisami, Rokare, Romoti, Maruru and Muyambi Rivers and their tributaries. This study used the project historical aerial surveys data gathered from 2013 to 2021 to produce breeding population trend and conducted ground nest surveys in 2021 and 2022 to estimate the breeding success and nesting preferences of the species. Due to the colonial nesting behaviour of the species, cluster analysis was used to group the nesting hotspots as well as remote sensing analysis to assess the tree cover in their breeding habitat.

According to results of the aerial survey data, the breeding population in the game reserves and their contiguous WMA has declined by 56%, and based on the active nest survey, breeding success was 29%, that being lower than the previously documented information for the Greater Serengeti Ecosystem which was 90%. Individuals were found nesting along the rivers in the Ficus and Vachellia and Senegalia woodlands. Majority of the nests (57%) were found on *Ficus sycomoras* whereas the least number (1%) was found on *Ficus indica*. The tree height was the most significant tree factor for nesting tree as observed from the generelized linear model. Nesting hotspots were classified as former or intermittent hotspots, and none were classified as persistent hotspots. Net tree loss was observed in all hotspot categories, and a 28% reduction in riparian tree cover was detected.

The study concludes that the *Gyps africanus* population in the area is declining, parallel with dropping breeding success and deteriorating breeding habitats. Seemingly, habitat loss/deterioration is a driving factor, which together with other potential threats to the population, need further investigation. Such baseline information is of critical importance to the management and conservation efforts of vultures and their overall breeding habitats in the ecosystem.

Keywords: White-Backed Vulture, breeding population, breeding success, tree cover, Greater Serengeti Ecosystem

IKISIRI KUU

Tumbusi mgongo mweupe (*Gyps africanus*) ni spishi iliyo hatarini kutoweka kutokana na idadi yake kupungua kwa kiasi kikubwa katika safu zake za kijiografia. Inavyoonekana, Tumbusi hutoa mfumo wa ikolojia na huduma ya kusafisha na hivyo kupunguza hatari ya magonjwa kuenea kupitia tabia zao za ulishaji. Kusoma hali ya idadi ya kuzaliana na makazi ya spishi hizi ni muhimu sana katika kudumisha afya ya mfumo ikolojia na vile vile kudhibiti na kupunguza matishio kwa spishi na makazi husika.

Utafiti huu ulilenga kuchunguza hali ya idadi ya kuzaliana na upendeleo wa eneo la kuzaliana kwa jamii ya Tumbusi mgongo mweupe katika Mapori ya Akiba ya Ikorongo-Grumeti na eneo la usimamizi wa wanyamapori la Ikona (WMA) katika ukanda wa magharibi wa Mfumo ikolojia wa Serengeti kando ya kanda za msingi za mto (PRZ). PRZ ilijumuisha Mito ya Grumeti, Manchira, Nyabeu, Samisami, Rokare, Romoti, Maruru na Muyambi na vijito vyake. Utafiti huu ulitumia data ya kihistoria ya uchunguzi wa angani iliyokusanywa kati ya 2013 na 2021 ili kutoa mwelekeo wa idadi ya kuzaliana kwa Tumbusi Mgongo mweupe, pia nilifanya tafiti za viota mnamo 2021 na 2022 ili kukadiria mafanikio ya kuzaliana na mapendeleo ya maeneo ya kuzaliana. Kwa sababu ya tabia ya kikoloni ya kuzalishia spishi, uchanganuzi wa nguzo ulitumiwa kupanga maeneo yenye viota pamoja na uchanganuzi wa hisi za mbali ili kutathmini idadi ya eneo lenye miti katika makazi yao ya kuzaliana.

Kwa mujibu wa matokeo ya takwimu za uchunguzi wa anga, idadi ya kuzaliana ya Tumbusi mgongo mweupe katika mapori ya akiba na WMA inayopakana nayo imepungua kwa 56%, na kulingana na utafiti wa viota, mafanikio ya uzalishaji yalikuwa 29%, ikiwa ni chini ya taarifa zilizoandikwa hapo awali katika mfumo wa Ikolojia wa Serengeti. Tumbusi mgongo mweupe walipatikana wakiwa na viota kando ya mito katika misitu ya Ficus na Acacia. Viota vingi (57%) vilipatikana kwenye *Ficus sycomoras* ambapo idadi ndogo zaidi (1%) ilipatikana kwenye *Ficus indica*. Urefu wa mti ulikuwa ndio

sababu kuu ya mti wa kuatamia kama inavyoonekana kutoka kwa uchambuzi wa takwimu. Sehemu kuu za viota ziliainishwa kuwa sehemu kuu za zamani au za mara kwa mara, na hakuna zilizoainishwa kama sehemu kuu zinazoendelea. Upotevu wa eneo lenye miti halisi ulizingatiwa katika kategoria zote za viota, na punguzo la 28% la eneo lenye miti ya mito liligunduliwa.
Utafiti unahitimisha kuwa idadi ya kuzaliana ya Tumbusi mgongo mweupe katika eneo hilo inapungua, sambamba na kushuka kwa ufanisi wa kuzaliana na kuzorota kwa makazi ya kuzaliana. Inaonekana, upotezaji wa makazi / kuzorota ni sababu ya kushuka kwa idadi hii, ambayo pamoja na vitisho vingine vinavyowezekana vinahitaji uchunguzi zaidi. Taarifa hizo za msingi ni muhimu sana kwa usimamizi na juhudi za uhifadhi wa Tumbusi mgongo mweupe na makazi yao ya jumla ya kuzaliana katika mfumo ikolojia.

Mane muhimu: Tumbusi mgongo mweupe, idadi ya kuzaliana, mafanikio ya kuzaliana, eneo la miti, mfumo ikolojia wa Serengeti

DECLARATION

I, Laizer Vainess Solomon, hereby declare to the Senate of Sokoine University of Sokoine University of neither Agriculture that this dissertation is my original work, and it was done within the period of registration and that it has neither been submitted nor being concurrently submitted in any other institution.

Laizer, Vainess Solomon (MSc. Wildlife Management Candidate)	Date

The above declaration is confirmed by

Prof. Shombe Hassan (Supervisor)	Date

Dr. Robert Byamungu (Supervisor)	Date

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ACKNOWLEDGMENTS

Foremost, I thank God for giving me the ability to be able to go through another academic process in my career journey, it is only by his unending Grace that I managed.

Secondly, I am grateful to the Sokoine University of Agriculture (SUA) for giving me an opportunity to pursue my MSc Wildlife management and Conservation. I would like to extend my sincere appreciation to the Grumeti Fund (GF) for Funding my studies and the research work during my course of my study 2020/2022. Deeply grateful to The Tanzania Wildlife Research Institute (TAWIRI), the Tanzanian Commission for Science and Technology (COSTECH) and the Tanzania Wildlife Authority (TAWA) for the research permits to undertake this study in the respective protected areas.

Thirdly, I am deeply thankful to my supervisors, Prof. Shombe N. Hassan and Dr. Robert M. Byamungu for their guidance and advice throughout my study program. To the Grumeti fund Research and Monitoring team Benson Benjamin, Exavery Kigosi, Noel Mbisse and Pete Goodman, my field assistant Mohamed Kitana and the RISE coordinator Victoria Mkessa for always being ready to assist with logistics and data collection in the Game Reserves and WMA. Also North Carolina Zoo associates in Tanzania for sharing their experiences and guiding me through the proper methodology and protocols for this research work

Finally, I would like to recognize the support from my family in pursuing my dreams as a researcher. My friends and classmates, for their unceasing encouragement through the MSc journey.

DEDICATION

“To RISE - Research and Innovation for the Serengeti Ecosystem”
Where my career vision was brought to existence.

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LIST OF ABBREVIATIONS AND ACRONYMS

AIC	Akaike Information Criterion
GSME	Greater Serengeti Mara Ecosystem
IGGR	Ikorongo-Grumeti Game Reserves
PRZ	Primary riparian zones
WBV	White-backed vulture
WMA	Wildlife Management Area

CHAPTER ONE

1.0 General Introduction

1.1 Background Information

Vultures are birds of prey with ability of locating carcasses from far distance and feeding on carcasses (Houston, 1971). Vultures play an important role in maintaining a healthy ecosystem through their lifestyle as obligate scavengers (Houston, 2001). Vultures provides an ecosystem with services such as cleaning organic wastes from the environment and acting as disease control agents hence reducing the spread of vectors of diseases in the environment (Markandya *et al*., 2013; Ogada *et al.,* 2012a,b). Vultures' ability to feed and digest carcasses is economically significant as it saves resources that would have been used to clean the environment (Chaudhary *et al.,* 2012). Vultures are also an important indicator of food to the ground scavengers that are limited in the range of movement and ability to locate the carcasses (Spiegel *et al*., 2013; Kane *et al*., 2014). Therefore, the presence of vultures in an ecosystem establishes a sustainable ecosystem and their removal may result to an ecosystem's health crisis through increased frequency of diseases outbreak (Prakash *et al.,* 2003).

Worldwide, there are 23 species of Vultures, 9 of which are found in Africa (Mundy *et al.,* 1992). Currently, 14 species out of 23 constituting to 61% of vulture species worldwide are threatened with extinction caused by unregulated human activities (Ogada *et al.,* 2016; Bamford *et al.,* 2009).Globally, vultures decline was reported since the 19th Century in Europe, North America followed by the catastrophic decline in Asia, where the Gyps vultures declined by more than 95% due to poisoning from the Veterinary drug (diclofenac), hence latter on banned by the government (Oaks *et al.,* 2004; *Gilbert et al*., 2006).

The continuous decline of vultures' population and their extinction has also been seen in Africa, where reports from West and East

Africa show that there has been a population decrease to over 95% of some of the species like the Hooded Vulture in West Africa, 42% of the overall population in Sudan and 70% in Kenya (Ogada and Keesing, 2010 ; Ogada *et al.,* 2016). Such declines are a result of land use changes (Herremans and Herremans-Tonnoeyr, 2000), human activities (Thiollay, 2006), intentional and unintentional poisoning (Henriques et al., 2020; Ogada, 2014), and unsustainable harvest of birds for trade due to cultural beliefs (Dabone *et al.,* 2023) Like elsewhere in Africa, Vultures are threatened by poisoning whether as unintentional victims through retaliatory poisoning linked with carnivore-livestock conflicts (Ogada, 2014), sentinel poisoning where vultures are poisoned through poisoning of carcasses for the purpose of eliminating them, and intentional poisoning due to belief-based use (McClure *et al.,* 2018; Thiollay, 2006) near boundaries of protected areas. Other threats include human activities (Zuberogoitia *et al.,* 2008), which may directly impact the breeding opportunities and success resulting in population decline of vulture (Gilbert *et al.,* 2006).

The most common vulture species as observed from carcasses in the Greater Serengeti Mara ecosystem is the White backed Vulture (*Gyps africanus*) (Kendall *et al.,* 2012). *Gyps africanus* is a critically endangered vulture species due to its declines across its ranges (Ogada *et al.,* 2016). The species *is* well known for its nesting habits in loose colonies with suitable nesting sites of tall trees in the riparian zones (Mundy *et al.,* 1992 ; Kendall and Bracebridge, 2022) and in some cases, aggregation of nests has been observed. The regional declines of the White- backed vulture population in Masai Mara, another part of the Greater Serengeti Mara ecosystem in Kenya (Virani *et al.,* 2011) are alarming, and prompts some conservational efforts for monitoring breeding populations.

1.2 Problem Statement and Justification

The Greater Serengeti Mara ecosystem (GRSME) is a home to vultures due to the presence of migrating ungulate (Kendall *et al.,* 2012; Houston, 1971), which are a source of food and suitable habitat for nesting in the lowlands with riparian forest in the savanna. Unfortunately, the White- backed vulture, one of the resident vulture species in the GRSME is declining across its range despite conservation efforts (Ogada *et al.,* 2012). Worse enough, limited information exists for some sub-populations such as the one in the western corridor of the GRSME whose baseline information dates to the studies by Houston (1971) and Houston (1974). Recent information about this sub-population describes a notable decline in the number of active nests as identified during aerial riparian surveys conducted in Grumeti -Ikorongo reserves, and Ikona wildlife management area (WMA) from 2013 to 2021 (Mbise and Goodman Pete, 2021). Apart from the reported studies, there have not been any other follow up studies to update existing or add new information as part of monitoring. The purpose of this study, therefore, was to investigate the White backed vulture breeding population in this corner of the GRSME in Tanzania, with attention to aspects of spatio-temporal population dynamics, breeding success, and habitat preferences and association.

1.3 Justification

The results of this study give information on the breeding population status of the White-backed vulture (*Gyps africanus)* in association to its breeding habitat preferences and tree cover. This information will help to advise the management on the conservation importance of the habitat as well as identify areas for further research in the ecosystem.

1.4 Objectives of the study

1.4.1 General objective

The main objective of the study was to investigate aspects of White-backed vulture’ breeding population status and nesting preferences

1.4.2 Specific objectives

1. To assess the breeding population trend of the critically endangered White-backed vulture (*Gyps africanus)*
2. To assess the breeding success of the critically endangered White-backed vulture (*Gyps africanus)*
3. To evaluate nesting preferences of the critically endangered White-backed vulture (*Gyps africanus)* with respect to tree characteristics
4. To evaluate whether changes in riparian tree cover influenced spatiotemporal trends of the critically endangered White-backed vulture (*Gyps africanus)* nesting patterns

1.5 Research Questions

1. What can be done to sustain the breeding population of the endangered Gyps africanus?
2. How is the critically endangered White-backed vulture breeding success measured?
3. What effect do tree attributes have on the nesting preferences of the critically endangered White-backed vulture?
4. Does the spatiotemporal trend of the critically endangered White-backed vulture nesting habits depend on changes in the cover of riparian trees?

1.6 Limitation of the Study

This study aimed at collection ground nest survey data from two breeding years: 2021 and 2022. In 2021 only two surveys were done due to heavy rains in the beginning of the year, which made it impossible to conduct the ground nest surveys.

The analysis of tree cover assumption was the tree preferred for nesting were the constitutes of the riparian forest, no account of tree availability was done for analysis. This limited how well the tree cover was associated with the nesting.

1.7 Dissertation Structure

This dissertation has been organized in publishable manuscript format following the Sokoine University of Agriculture (SUA) new guidelines and regulations of 2022, which include an extended abstract, a general introduction, chapters based on manuscripts, a general discussion, a general conclusion and recommendations, and appendices.

Chapter One presents the general introduction of the study, which includes background information about vultures and threats to vultures; problem statement and justification of the study; objectives; limitations of the study; and the dissertation structure and references.

Chapter Two presents the first manuscript titled "Breeding population status and sucess of the White backed vulture (*Gyps africanus*) in western Serengeti, northern Tanzania." This manuscript details the population trend and breeding success of the White backed vulture breeding population. This manuscript has been submitted to the journal of Ornithological Applications.

Chapter Three presents the second manuscript titled “Tree cover loss and nesting preferences of the White-backed vulture (*Gyps africanus*) in western Serengeti, northern Tanzania." This manuscript details the nesting preferences of the White-backed vulture (*Gyps africanus)* and tree cover classification of their breeding habitats as compared to the breeding population trend as analysed in the first manuscript.

The last chapter (Chapter Four) of this dissertation presents the general discussion and general conclusion and recommendations. This section includes major discussion of key findings from this study and conclusions and recommendation from each manuscript.

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CHAPTER TWO

Manuscript One

2.0 Breeding Population Status of the Critically Endangered White-backed Vulture (*Gyps africanus*) in Ikorongo-Grumeti Game Reserves and Ikona WMA in northern Tanzania.

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Abstract

The White-backed Vulture, *Gyps africanus,* is a critically endangered species as its population has declined across its range in Africa, largely due to poisoning. The western corridor of the Serengeti ecosystem in northern Tanzania hosts a significant number of individuals of this species. However, there is no published information on its breeding status**.** Here, we report findings from five riparian aerial nest surveys conducted from 2013 to2021 to establish population trend based on nest encounter rate, and ground nest surveys conducted in 2021 and 2022 to determine breeding success through examining nest status. This study covered the Ikorongo and Grumeti Game Reserves and the Ikona Wildlife Management Area. The nest encounter rate within the survey area was found to have declined from 0.99 nests km^{-2} in 2013 to 0.43 nests km^{-2}in 2021. On the other hand, breeding success was 29.4% significantly lower than reported in other ranges in the Greater Serengeti-Mara Ecosystem (GSME). Rigorous monitoring of this population is required towards establishing an in-depth understanding of the population breeding dynamics in the face of climate change and variability, and the key factors influencing its breeding success.

Key words: breeding success, population trend, white-backed vulture, Serengeti

2.1 Introduction

The White-backed vulture, *Gyps africanus* (hereafter WBV) is widely distributed in the savannahs of African countries with an estimated global population of 270,000 individuals (Mundy *et al.,* 1992). However, its population has been declining at alarming rate with an annual median estimate of 4.1%, that being within a range of 2.5-5.4% per year (Ogada *et al.,* 2012, Ogada *et al.,* 2016a). The species was assigned the critically endangered status in 2015 (Birdlife International, 2021). Since then, there has been increased effort devoted to monitoring WBV populations (Shaw *et al*., 2019), including identifying factors contributing to the decline (Kendall *et al.,* 2018) as well as breeding status and behaviour (Murn *et al*., 2017).

One of the major drivers of WBV declines is poisoning, whereby individuals are often the unintended victims of retaliatory killings of carnivores via the poisoning of livestock carcasses (Ogada, 2014;Richards, 2011). Vultures are also deliberately poisoned by poachers trying to avoid detection by rangers (Mateo-Tomás and López-Bao, 2020; Murn and Botha, 2018; Ogada *et al.,* 2016b) and for the use of their parts for cultural purposes (Henriques *et al.,* 2020; Ogada and Buij, 2011). Because WBVs travel large distances in search for food, their home ranges can extend beyond protected areas, thus exposing them to higher risks of poisoning (Kane *et al.,* 2022). On the other hand, human population increase around protected areas is associated with anthropogenic disturbance at nesting sites — contributing to reduced breeding success of local WBV (Bamford *et al.,* 2009; Zuberogoitia *et al.,* 2008).

In northern Tanzania, the Greater Serengeti-Mara Ecosystem (GSME) supports a diversity of vulture species, including WBV, Lappet-faced vulture (*Torgos tracheliotus*) and White-headed vulture (*Trigonoceps occipitalis)* (Pennycuick, 1976). Among these, WBV is the most common vulture species observed on carcasses in this ecosystem (Kendall *et al.,* 2012). However, recent declines of WBV have been reported in the Masai Mara (Kenya) and may be representative of declines throughout the ecosystem (Virani *et al.,* 2011, Kendall *et al.,* 2018). The GSME is an important ecosystem for the survival and recruitment of WBV due to the presence of large populations of migrating ungulates, which provide abundant primary food resources, particularly during breeding, and the presence of suitable breeding habitat (Houston, 1971; Houston, 1974).

WBVs prefer breeding in riparian forests adjacent to savannah habitats in the canopies of trees taller than 13 m, often in colonies or aggregated pairs (Mundy *et al.,* 1992; Virani et al., 2010; Monadjem & Garcelon, 2005; Bamford *et al.,* 2009). The nests are built with dry sticks, placed on the tree canopy or fork of trees. During breeding, WBV adults reduce their ranges and tend to mainly utilize habitat within protected areas (Kane *at al.,* 2022; Peters *et al.,* 2022). In the GSME, suitable WBV breeding habitat is primarily constrained to the protected extents of the high-altitude forests in the North along the Mara River and the low-altitude forests in the West along the Grumeti River and its tributaries (Reed *et al.,* 2009). Despite the consensus that the GSME contains critical WBV breeding habitat, little is known about the breeding population status. Houston (1974) conducted a study on the breeding biology of WBV and Ruppell's vulture with a focus on timing of the breeding season in relation to food supply variation in Serengeti National Park, but more recent data on species breeding biology and population in the GSME is lacking.

Monitoring population trends over time is a critical component of effective conservation science and provides essential insights to guide management decisions (Lindenmayer *et al.,* 2012). In the case of vultures, two common approaches to assess populations are the long-term monitoring of nest encounter rates and evaluations of breeding success via repeated breeding season nest surveys (Steenhof and Newton, 2007). The long-term monitoring of nest encounter rates can detect temporal variations in the breeding population, an important signal of the overall population trend (Lindenmayer *et al.,* 2012). Likewise, breeding season nest surveys provide important information on recruitment status and the breeding population dynamics of the focal species. Nest surveys are conducted by locating breeding pairs and observing breeding behaviour and fledgling success to derive breeding success estimates (Brown, 1990; Roche, 2006). Nest surveys can additionally lead to an improved understanding of factors influencing breeding success (Tapia and Zuberogoitia, 2018).

The purpose of this study was to investigate the WBV breeding population status in the Ikorongo and Grumeti Game Reserves and the Ikona WMA in western corridor of GSME, northern Tanzania. Specifically, we i) used aerial riparian survey data from 2013 to 2021 to examine the breeding population trends, and ii) conducted nest surveys to estimate breeding success. We discuss the conservation implications of our findings and identify areas for further research in this ecosystem.

2.2 Materials and Methods

2.2.1 Description of the study area

The study occurred in the western corridor of the GSME, specifically the Ikorongo-Grumeti Game Reserves and Ikona Wildlife Management Area (hereafter, IGGR). The study area covers 1,947 km^2 of the western corridor of the GSE in northern Tanzania (1^{o} 50$^{'}$ –2 ^{o} 12$^{'}$ S, 33 ^{o} 57$^{'}$ – 34 ^{o} 59$^{'}$ E). IGGR forms an important part of the GSME, separating Serengeti National Park from adjacent communities. Adjacent communities are densely settled relative to the rest of the communities in the ecosystem, and subsistence farming and livestock keeping are the primary livelihoods (Grumeti Fund., 2016). Within IGGR, photographic tourism and permitted hunting are the only allowable forms of human activity. We restricted our study to stretches of riparian forests that are suitable for WBV nesting in IGGR, which are largely constrained to within 500 m of each of the eight main rivers and their primary tributaries, herein defined as the primary riparian zone (PRZ) (Figure 2.1). These drainages were consistently sampled during all survey years and represent the core nesting habitat for WBV in the study area, containing more than 93% of all nests counted in each survey year. The PRZ contains the Grumeti, Manchira, Nyabeu, Samisami, Rokare, Romoti, Maruru and Muyambi Rivers, and several other smaller, unnamed drainages.

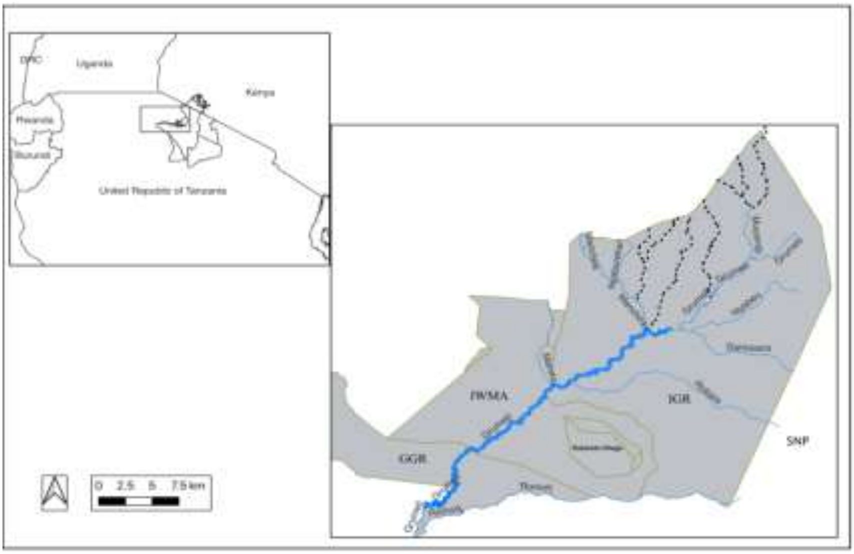

Figure 2.1: The study area included river drainages within Grumeti Game Reserve (GGR), Ikorongo Game Reserve (IGR) and the Ikona Wildlife Management Area (IWMA). Rivers and their associated Primary Riparian Zone (PRZ), which were surveyed during aerial riparian surveys conducted from 2013 to 2021, are shown in blue and black. Drainages in black are not named and were grouped collectively as ‘other’. The bolded section of the Grumeti River was the focal area for nest surveys conducted in 2021 and 2022. The Serengeti National Park (SNP) borders on the East and South. An inset map shows the location of the study area in the Greater Serengeti Mara Ecosystem (GSME) in northern Tanzania.

2.2.2 Data collection

2.2.2.1 Riparian aerial surveys (Goodman and Mbise, 2021)

Aerial surveys along the PRZ were conducted by the Grumeti Fund in 2013, 2014, 2016, 2019 and 2021. Drainages were surveyed via helicopter in July of each survey year when vulture nests are active. Survey flights were conducted by a pilot, data recorder and two rear observers at a maintained speed just above tree canopy height. All flights were conducted from 0 800 to 1 100hrs or from 1 500 to 1 730hrs. Nest locations were logged to the GPS by the data recorder and the nests were scored as either ‘not active’ (nests with no egg/adult/droppings), ‘active empty’ (nests with dropping/grass lining/adult beside nest), ‘active with egg’ (nest with an egg), or ‘active with chick’ (nest with a chick of any age). GPS logs of the survey path were recorded to account for the survey effort.

2.2.2.2 Nest Monitoring

Nest monitoring techniques were adapted from Fuller (1987) and a modified nest survey protocol from VulPro (Wolter *et al.,* 2020). Nest surveys were completed during the 2021 and 2022 breeding seasons on foot and provided information about breeding success that wasn’t possible to gather with single aerial surveys. We focused monitoring efforts along the Grumeti River, which contained the greatest concentration of nests during previous aerial riparian surveys. We used the nest locations identified in the 2019 riparian survey as the basis of our initial ground search of nests along the river. We searched a 500m buffer on either side of the river, and each active nest within this buffer zone was logged using a handheld GPS and assigned a unique ID number. We visited all active nests from 0900 to 1700 hours during three breeding stages as outlined below. In 2021, the surveys were restricted to one side of the river, but in 2022 the surveys were expanded to include both sides. We initially used binoculars to scan for nests at a particular locality, and then used it to inspect a nest under the nesting tree to determine nest status throughout all surveys. In the 2021 late-season survey and all 2022 surveys, we used a go-pro camera mounted on a telescoping pole to obtain photos and/or videos of the nest activities. We surveyed the nests in three phases that corresponded with the key breeding stages as adopted from (Leepile *et al.,* 2020).

1) 'Early-season' surveys: aimed to coincide with the incubation period between March and May. This survey was used to estimate the number of active nests in the sampling area. Nests were categorized as active if they had signs of fresh nest building, lining, or droppings, if adults were incubating, or if the nest contained an egg. In 2021, heavy rains during this period made nests inaccessible, leading to the termination of the exercise. Consequently, early surveys were only completed in the following year, 2022. Any active nests located during early season surveys were visited again during the subsequent mid-season surveys.
2) 'Mid-season' surveys: aimed to coincide with the early nestling stage or the late incubation period, typically observed in July. Nests containing an egg or chick of any age were determined to be active. Any active nests located during mid-season surveys were visited again during late-season surveys.
3) 'Late-season' surveys: aimed to coincide with the late nestling stage, defined by the age of the chick on the nest, observed in September prior to when chicks fully fledge and leave the nest. This survey was used to determine the breeding success of the previously active nests. If a chick had fully developed flight feathers, they were scored as ready to fly and hence a successful fledgling.

During surveys, active nests were considered to have failed if there was no sign of continued productivity in a nest, which was considered active in the preceding survey. In this regard, we considered nests to have failed if any of the following were observed: 1) An empty nest with no egg laid on a previously recorded active and lined nest, 2) Nests without signs of a hatched egg on a nest where a laid egg was previously recorded, 3) An empty nest with no chick where a chick was previously observed, and which would have not yet been ready to fledge based on the age of the chick, or 4) The disappearance of the nest from the nesting tree.

2.3 Data Analysis

2.3.1 Breeding population trends

Riparian aerial survey data were used to assess temporal trends in nest encounter rates as adopted from (Goodman and Mbise, 2021). We calculated the nest encounter rate in each survey year, which was derived by dividing the number of active nests by the total river distance surveyed. Variability in survey effort between years was accounted for by summing up the length of the river surveyed during each survey year. We examined the overall and per-drainage trend in nest encounter rate across survey years. Correlation between year and number of nests was done to see the trends of the overall nest encounter rate across all years.

2.3.2 Breeding success (Leepile *et al.,* 2020)

We calculated the area covered during the nest surveys by multiplying the length of the river surveyed by the width, which was the 500 m of buffer we searched for nests. In 2022, the area was doubled since the surveys involved searching both sides of the river. Nest monitoring data was used to determine breeding success. Overall, breeding success was defined as the proportion of active early-season nests that contained fledged chicks in the late-season survey. We calculated overall breeding success using only the 2022 nest survey data as an early season survey was not conducted in 2021, and the nest survey results from 2021 would likely overestimate breeding success and would not be comparable to other studies. To facilitate a comparison of breeding success between 2021 and 2022, we calculated the proportion of active nests from the mid-season survey that successfully fledged in the late season survey each year.

2.4 Results

2.4.1 Breeding population trends

The highest WBV total active nest encounter rate (0.99 nests/km) was observed in 2013 whereas the lowest (0.43 nests/km) was observed in 2021 (Table 2.1). This decline in the nest encounter rate trend was statistically significant at p value = 0.02 and characterised by a very strong negative correlation coefficient (-0.94). Temporal trends in nest encounter rates were most noticeable in the PRZ along the Grumeti and Manchira Rivers. Together, the two drainages accounted for 83% of active nests counted in 2013. By 2021, the yearly nest encounter rate along the two drainages had declined to the extent that jointly they accounted for 51% of active nests counted. Nests along the Grumeti declined by 63%, and along the Manchira by 68% (Table 2.1).

The active nest encounter rate increased across three drainages: the Samisami, Rokare, and Muyambi (Table 2.1). In 2013, these three drainages accounted for 4.7% of active nests, and the rate increased to 28% in 2021. Across all drainages, the 2021 nest encounter rate was highest along the Samisami (1.4 nests/km) and Muyambi (0.82 nests/km). Between 2013 and 2021 the active nest encounter rate along the Muyambi increased by 21% and increased along the Samisami by more than 600%. Despite the observed 600% increase in active nest encounter rate in Samisami by 2021, this comparison is likely to be skewed since the surveyed extent along the Samisami was greater in 2021 than 2013.

Table 2.1: **Aerial riparian census observations across all years, where n indicates the count of active nests along the specified drainage, e is the survey effort along the specified drainage measured as the length of the drainage surveyed, and Er is the nest encounter rate along the drainages (nest/km).**

Drainage	**2013**			**2014**			**2016**			**2019**			**2021**		
	n	*e*	*Er*	*n*	*e*	*Er*	*n*	*e*	*Er*	*n*	*e*	*Er*	*n*	*e*	*Er*
Grumeti	148	90.57	1.63	147	90.57	1.62	136	90.57	1.5	107	90.57	1.18	54	90.57	0.60
Manchira	28	17.07	1.64	17	17.07	1.00	24	16.29	1.47	9	17.07	0.53	9	17.07	0.53
Maruru	3	10.04	0.30	4	10.04	0.40	4	10.04	0.40	9	10.04	0.90	1	10.04	0.10
Muyambi	5	7.32	0.68	4	7.32	0.55	3	7.32	0.41	3	7.32	0.41	6	7.32	0.82
Nyabeu	2	13.74	0.15	3	13.74	0.22	3	11.16	0.27	1	13.74	0.07	0	13.74	0.00
Ramoti	-----	0.00	-----	4	35.36	0.11	4	35.36	0.11	4	35.36	0.11	0	36.39	0.00
Rokare	4	17.22	0.23	6	23.44	0.26	7	18.98	0.37	4	23.43	0.17	8	24.70	0.32
Samsami	1	5.32	0.19	8	14.24	0.56	11	14.24	0.77	14	14.24	0.98	20	14.24	1.40
Other	21	56.03	0.37	35	66.86	0.37	27	57.58	0.47	26	66.86	0.39	24	66.86	0.36
Total	**212**	**217.3**	**0.99**	**218**	**278.6**	**0.78**	**219**	**261.5**	**0.84**	**177**	**278.6**	**0.64**	**122**	**280.9**	**0.43**

2.4.2 Breeding success

In 2021, we searched 6.6 km² and in 2022 13.32 km² for nests. Despite the increased search effort, we located fewer active nests (n=34) during the 2022 early-season survey than during the 2021 mid-season survey (n=44), equivalent to a 23% deficit in active nests. Twenty-one nests were active in both 2021 and 2022. In 2022, 26% of the active nests located in the early season survey had no eggs laid by the mid-season survey, and 6% of the nests that had eggs failed by mid-survey while 44% of the active nests with eggs laid and young chicks by early season survey failed upon commencement of the drier season, which corresponds with the mid-survey (Table S1). In 2022, overall breeding success from the early to late season survey was 29.4%.

In the 2021 breeding season, 61.4% of active mid-season nests successfully fledged compared to 53.3% of active mid-season nests that successfully fledged in 2022. In the 2022 breeding season, 56% of nests failed between early and mid-season surveys, and 47% of the remaining active mid-season nests failed by the late-season survey. In 2021, the failure of active nests between mid and late-season surveys was 39%. We were unable to confirm the source of nest failures but noted that the timing of the greatest number of nest failures coincided with the mid-season survey in July, during the early dry season (Table 2.2).

Table 2.2: Breeding success derived from periodic nest surveys along the Grumeti River in 2021 and 2022 across Grumeti Game Reserve (GGR), Ikorongo Game Reserve (IGR), the Ikona WMA (IWMA), and in total. Overall breeding success was calculated as the percent of active nests found in the early survey period from which chicks fledged in the late season survey. Overall, breeding success was only determined for 2022 as no early season survey occurred in 2021 (NA). Mid to late season nest success was calculated to facilitate comparison of breeding success across survey years.

Year	**Management**	**Active nests by survey period**			**Breeding success**	
		Early	Mid	Late	Mid to late	Early to late
2021	GGR	NA	12	7	58.3	NA
	IWMA	NA	13	6	46.2	NA
	IGR	NA	19	14	73.7	NA
	Total	**NA**	**44**	**27**	**61.4**	**NA**
2022	GGR	11	6	3	50.0	27.3
	IWMA	14	5	3	60.0	21.4
	IGR	9	4	2	50.0	11.1
	Total	**34**	**15**	**8**	**53.3**	**29.4**

2.5 Discussion

The White-backed vulture is the most abundant vulture species in the African savannah, but populations are in decline across its range. We observed a 56% decline in the nest encounter rate in IGGR from 2013 to 2021, which corresponds with the observed regional population declines of WBV. In the Maasai Mara National Reserve, a 52% decline in the WBV population was observed across 27 years using road counts (Virani *et al.,* 2011). Elsewhere in Africa, declines were observed in WBV breeding colonies around Kimberly, South Africa, which was historically the main breeding site of WBV in southern Africa where nesting numbers declined by 26% in 13 years (Murn *et al.,* 2017). In Botswana, a 53% decline in active nests was observed over a decade (Leepile *et al.,* 2020). Given the nine-year duration of our survey period, the decline in the active nest encounter rates observed in our study is severe in comparison to those in the Mara and South Africa, but similar to those observed in Botswana. The main reason for the population declines in Botswana was poisoning, which killed vultures in the country and would lead to a 70.8% population decline by 2030 (Leepile *et al.,* 2020).

In a population where adults have high mortality rates, the number of adults responsible for breeding may be affected leading to fewer individuals breeding yearly, and ultimately, contributing to population declines (Monadjem *et al.,* 2012). The impact of adult survivorship is particularly influential for species such as WBV, which exhibit high parental care, investing 5.5 months of food provisioning post fledgling and raising only one offspring at a time (Mundy et al., 1992). The impact of poisoning on survivorship is well documented — for example declines in the Mara were attributed to poisoning events within and around the reserve (Plaza *et al.,* 2019; Virani *et al.,* 2011). We did not collect data on poisoning events in our study, thus the impact of poisoning is unknown, but likely to influence the observed trends given the wide-ranging behaviour of WBV. Further investigation into sources of adult mortality, such as poisoning, is required to better understand the influence of adult survivorship on observed declines in nest encounter rates in the western Serengeti.

We observed exceptionally low levels of breeding success in the western Serengeti. The overall breeding success for 2022 was 29.4%, which is much lower than is reported elsewhere across the continent. WBV breeding success in Africa is reported to be 55-57% (Mundy 1992**)**. In Kgalagadi Trans-Frontier Park (KTP), a 40% decline in breeding success was observed from 1988 to 1990 and was associated with drier weather conditions and suspected poisoning that led to a total failure in breeding attempts (Herholdt and Anderson, 2006). From our nest monitoring, we observed that active nests were most susceptible to failure after hatching. This aligns with findings by Maphalala *et al.* (2017) who showed that the survival of chicks is lowest after hatching because the adults spend less time on nests as the chick grows, which may attract predators or increase vulnerability of the chick during storms. We observed that, 26% of active nests did not show signs of an egg being laid. Failure to lay may result from having an incompetent mate (younger inexperienced birds) or mate loss (Leepile *et al.,* 2020). It is also possible that eggs were predated upon between surveys, making it difficult to definitively conclude that an egg was not laid. We were only able to determine overall breeding success from one breeding season, which may not fully represent the trends of breeding success in the area. To aid in understanding the low rates of breeding success that were observed, additional nest monitoring in the study area is required, as is an investigation of rainfall and weather patterns during the breeding season.

2.6 Conclusion

In conclusion, the decline of the WBV breeding population is a clear conservation issue in the western corridor of the GSME where WBV appear to be facing a multitude of threats. We observed an alarming decline in the breeding population trend as indicated by the nest encounter rate, as well as very low levels of breeding success. Expanding the riparian surveys and nest monitoring to include other potential WBV breeding areas in the GSME will provide improved clarity to the overall ecosystem-wide trends in the WBV breeding population. Undertaking more intensive nest surveys to determine the reasons for nest failures, as well as further investigation of poisoning in this ecosystem, would provide a greater understanding of the cause of declines in nesting rates and low breeding success. Determining other potential sources of mortalities such as use of vulture parts for traditional use and food availability are of paramount interest as well.

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CHAPTER THREE

Manuscript Two

3.0 Tree Cover Loss and Nesting Preferences of the Critically Endangered White-backed Vulture (*Gyps africanus*) in Ikorongo-Grumeti Game Reserves and Ikona WMA in Northern Tanzania.

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Abstract

The White-backed Vulture, *Gyps africanus,* is the most common vulture species in the savannah of the Serengeti ecosystem. The western corridor of the Serengeti ecosystem harbours a suitable habitat for the breeding of this species, but the information on how tree characteristics influence nesting preference and spatiotemporal trends in vulture nesting sites following tree loss are lacking. In this study we report findings from the aerial surveys conducted from 2013 to 2021 in the Ikorongo and Grumeti Game Reserves and the

Ikona Wildlife Management Area as well as the nesting preferences as observed in the ground nest surveys conducted from 2021 to 2022. The breeding population trend was found to have declined by 56% in the period of the aerial surveys. *Gyps africanus* preferred to nesting on the big trees along rivers as reported in other studies in the ecosystem, particularly the Ficus, Vachellia and Senegalia woodlands. From the hotspots analysis, nest sites were either intermittent or emerging, rather than persistent hotspot, suggesting a shift in nesting sites. The net tree cover loss was higher in all hotspots than tree cover gain in the nesting habitat and had declined by 29% in the aerial survey period, suggesting a possible habitat loss in the breeding habitat. More investigation on the causes of the habitat loss and low breeding success will benefit the conservation of these important areas.

Keywords: breeding population, nesting, White-backed vulture, tree loss.

3.1 Introduction

The White-backed vulture, *Gyps africanus* is the most common and widely distributed vulture species in the African savanna *(Mundy et al.*, 1992). This species has suffered declines across its range due to poisoning, trade for vulture parts and even habitat loss resulting in its conservation status to be critically endangered (Birdlife International, 2021). Studies about their breeding habitats and preferences in Greater Serengeti -Mara ecosystem (GSME) date back to 1970s whereby Houston (1974) reported stable breeding population and habitat. The Greater Serengeti -Mara ecosystem (GSME) comprises of suitable habitats for both foraging and breeding of the *Gyps africanus.* According to results from the Masai-Mara, Kenya, which forms part of GSME, the White-backed Vulture (*Gyps africanus*) shows a preference for nesting in Acacia woodland habitats rather than the open grassland habitats (Virani *et al.,* 2010).

Recently, the population declines of *Gyps africanus* have been associated with loss of suitable habitats for foraging and breeding (Monadjem and Garcelon, 2005). *Gyps africanus* prefer breeding in riparian forests adjacent to savannah habitats in the canopies of larger trees, often in colonies or aggregated pairs (Bamford *et al.,* 2008). The breeding season of this species in the Greater Serengeti Ecosystem is March/April to September/October, making these species spend more time in their breeding sites, consequently, habitat loss becoming a major conservation issue. Loss of these suitable habitats is also a major cause of population decline for this species in other parts of Africa (Chomba *et al.,* 2013). In the Western corridor of the Greater Serengeti Ecosystem, there are riparian forests along main rivers and their tributaries, which support the breeding of the *Gyps africanus* (Houston, 1974).

Despite the consensus that the western corridor in the Greater Serengeti Ecosystem contains critical WBV breeding habitat, little is known about their nesting preferences in relation to tree characteristics, as well as the status of tree cover in their breeding habitat. These conservation concerns have stimulated the nest surveys to investigate the nesting preferences as a tool to conserve and maintain the population. In other areas, statistical models developed using GIS data and aerial surveys have been used to predict the probability of nest presence based on habitat characteristics (Bamford *et al.,* 2009) hence ensuring limited threats to the breeding areas. The purpose of this study was to investigate the *Gyps africanus* nesting preferences and undertake tree cover assessment. Specifically, we i) assessed nesting preferences with respect to tree characteristics, and ii) evaluated whether changes in riparian tree cover influenced spatiotemporal trends in vulture nesting sites.

3.2 Materials and Methods

3.2.1 Study area and sites

The study was conducted in the western corridor of the GSME, specifically the Ikorongo-Grumeti Game Reserves and Ikona Wildlife Management Area (hereafter, IGGR) in Tanzania. The study area covers 1,947 km^2 of the western corridor of the GSE in northern Tanzania (1^{o} $50^{'}$ –2 ^{o} $12^{'}$ S, 33 ^{o} $57^{'}$ – 34 ^{o} $59^{'}$ E).IGGR forms an important part of the GSME, separating Serengeti National Park from adjacent communities. Adjacent areas are densely settled and subsistence farming and livestock keeping are the primary livelihood options (Grumeti Fund, 2016). Within IGGR, photographic tourism and permitted hunting are the only allowable forms of human activity.

We restricted our study to within 500 m stretches of riparian forests that are suitable for WBV nesting, herein defined as the primary riparian zone (PRZ) (Figure 3.1). The PRZ contains the Grumeti, Manchira, Nyabeu, Samisami, Rokare, Romoti, Maruru, and Muyambi Rivers, and several other smaller, unnamed drainages. These drainages were consistently sampled during all survey years and represent the core nesting habitat for WBV in the study area as it contains more than 93% of all nests counted in each survey year.

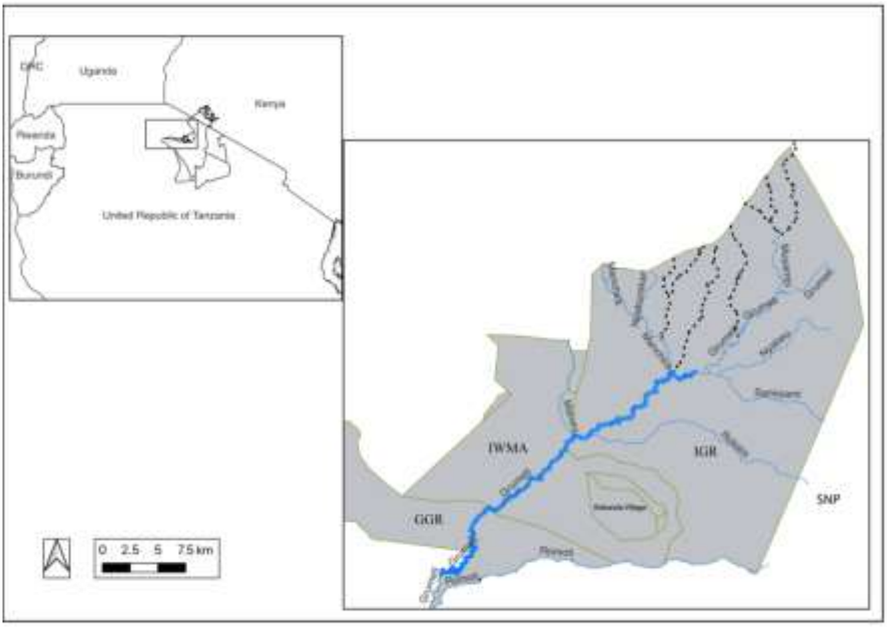

Figure 3.1: The study area consisted of river drainages within Grumeti Game Reserve (GGR), Ikorongo Game Reserve (IGR) and the Ikona Wildlife Management Area (IWMA). The rivers within the Primary Riparian Zone (PRZ) surveyed during aerial riparian surveys conducted from 2013 to2021, are shown in blue and black. Drainages in black are not named and were grouped collectively as 'other'. The bolded section of the Grumeti River was the focal area for nest surveys conducted in 2021 and 2022. The Serengeti National Park (SNP) borders on the East and South. An inset map shows the location of the study area.

3.3 Data Collection

3.3.1Riparian aerial surveys (Laizer *et al.,* 2023)

Aerial surveys along the PRZ were conducted by the Grumeti Fund in 2013, 2014, 2016, 2019, and 2021. Drainages were surveyed via helicopter in July of each survey year when vulture nests are active. Survey flights were conducted by a pilot, data recorder and two rear observers at a maintained speed just above tree canopy height. All flights were conducted from 0 800 to 1 100hrs or from 1 500 to 1 730hrs. Nest locations were logged to the GPS by the data recorder and the nests were scored as either ‘not active’ (nests with no egg/adult/droppings), ‘active empty’ (nests with dropping/grass lining/adult beside nest), ‘active with egg’ (nest with an egg), or ‘active with chick’ (nest with a chick of any age). GPS logs of the survey path were recorded to account for the survey effort.

3.3.2 Nesting tree characteristics

We recorded tree and nest characteristics that relate to WBV nesting preferences (Kendall *et al.,* 2018). Tree species, tree and nest heights (in meters, measured using the length of the telescoping pole), tree DBH (cm diameter at breast height, 1.3m from the ground), and tree canopy cover (using the Canopeo App as a percentage of light) were recorded at each active nest. We recorded the presence/absence of elephant damage to the nesting trees by checking for signs of debarking of the main stem, whether the tree was pushed over, or if branches were eaten off the tree. The tree characteristics were recorded only once for each nesting tree. We recorded the same characteristics for the nearest non-nesting tree within 20 m radius of the nesting tree to help explain differences between trees selected and not selected for nesting. The sampling of non-nesting tree species was restricted to the preferred species of nesting tree (as indicated by field observations) with a minimum height of 13 m, which was the minimum nesting tree height measured during the nest surveys. Non-nesting trees were selected in random directions using the point-centered quarter (PCQ) method to avoid repeating a tree (Majgaonkar *et al.,* 2018). Where present, up to four non-nesting trees were selected, one from each quarter of the cross. If a non-nesting tree was not present within 20 m, no tree was sampled. A total of 57 nesting trees and 43 non-nesting trees were sampled.

3.4 Data Analysis

3.4.1 Riparian aerial survey

Riparian aerial survey data were used to assess temporal trends in nest encounter rates as adopted from Goodman and Mbise (2021). We calculated the nest encounter rate in each survey year, which was derived by dividing the number of active nests by the total river distance surveyed. Variability in survey effort between years was accounted for by summing up the length of the river surveyed during each survey year. We examined the overall and per-drainage trend in nest encounter rate across survey years. Correlation between year and number of nests was done to see the trends of the overall nest encounter rate across all years as reported in Laizer *et al.* (2023).

3.4.2 Nesting preferences

A two-sample student t-test was used to test whether the measured characteristics of nesting trees significantly differed from the characteristics of non-nesting trees. We used generalized linear models (GLM) to further examine WBV nesting preferences (Kendall *et al.,* 2018). We examined whether tree DBH, tree height, tree species and tree canopy cover influenced the presence or absence of a nest on a tree. Prior to model construction, we confirmed that variables were not highly correlated ($r<0.7$). We used AICc model selection to distinguish among all possible sets of models describing the relationship between the aforementioned tree characteristics. All analyses were conducted using R version 4.2.2 and the stats package (R Core Team, 2022).

3.4.3 Tree cover classification

A decline in active nests was noted during the aerial riparian survey. We applied remote sensing techniques through the Google Earth Engine platform to better understand changes in tree cover during the riparian aerial survey timeframe, and whether tree loss was a contributing factor to the observed declines in nesting. We assumed that the loss of preferred nesting trees would be positively correlated with the loss of riparian forest cover, leading to the decline in the breeding population.

We created a forest/non-forest classification (Gorelick *et al.,* 2017) for the years 2012 and 2020 using Rapid Eye and Sentinel-2 satellite data, respectively, with a Random Forest model on the Google Earth Engine platform. Representative samples for forest (n=86) and non-forest (n=404) classes were obtained via the collection of ground truth points in 2017 and identification from high-resolution satellite imagery in 2022. We manually verified that the class of each sample point was the same in both classification years. To fully cover the study area, we used a composite of two Rapid Eye images obtained in January 2012 and we masked out clouds and shadows. We created a composite of Sentinel-2 surface reflectance (SR) imagery for June 1 to August 31, 2020 and used a cloud score of less than 10% per pixel to create a cloud mask. We used the following bands from Rapid Eye and Sentinel-2: blue, green, red, and near-infrared. Using these bands, we created the following indices: Normalized Difference Vegetation Index (NDVI), Green Normalized Difference Vegetation Index (GNDVI), Green Brown Vegetation Index (GBVI), Green Red Vegetation Index (GRVI), Enhanced Vegetation Index (EVI), and Green Chlorophyll Vegetation Index (GCVI). We withheld 20% of points in each class for testing and calculated the producer’s and user’s accuracy of the testing and training datasets.

To calculate the change in forest cover, we aggregated the 2012 classification (5 m) to match the spatial resolution of the Sentinel-2 classification (10 m). The majority class was used to assign the class of the aggregated pixels. We investigated the optimal option for resolving tiebreakers by creating two aggregated products, one in which the forest class was the default class assigned in the case of ties, and the other in which non-forest was the default. We calculated the accuracy of the two products relevant to the sample points; the forest default classification performed similarly (overall accuracy = 98.0%) to the non-forest default classification (overall accuracy = 97.6%). Classification errors in the forest default classification were evenly distributed among classes, while the non-forest default classification tended to under-predict forest. For this reason, we elected to use the forest-default classification.

We assessed the change in forest cover within the PRZ. Given that White-backed vulture preferentially nest near other White-backed vulture to form clusters (Mundy *et al.,* 1992), we identified areas in which nests were clustered using Getis-Ord Gi* statistic via the sfhotspot package in R (Ashby, 2022). We selected a cell size resolution of 100m and the nearest neighbour distance equal to the mean nearest neighbour distance between nests across all survey years (287m). We used the *hotspot_classify* function to examine spatiotemporal changes in nest clusters throughout the study timeframe (2013-2021) (Figure 3.2). This function assesses whether a cell is a hotspot or coldspot at each timestep (in our case, aerial survey year) and based on how the categorization of a given cell changes over time, classifies a cell according to whether it is an emergent, former, intermittent, or persistent cold/hotspot or no pattern area. Emergent hotspots are cells, which were classified as hotspots in 2019 or 2021, but had not been classified as hotspots in earlier surveys, whereas former hotspots are those that were not hotspots in 2019 or 2021, but were in at least one previous survey year. Persistent hotspots are cells which were consistently classified as hotspots (in at least 3 of 5 survey years), while intermittent hotspots were cells which had been classified as hotspots in one or two survey years, but were not considered emergent or former hotspots. No pattern areas were those which had never been classified as a hotspot. We calculated the percent tree cover in 2012 and 2020, and the net loss of tree cover from 2012 to 2020, by hotspot category and used the Kruskal-Wallis and posthoc Dunn test to determine whether tree cover significantly differed between categories.

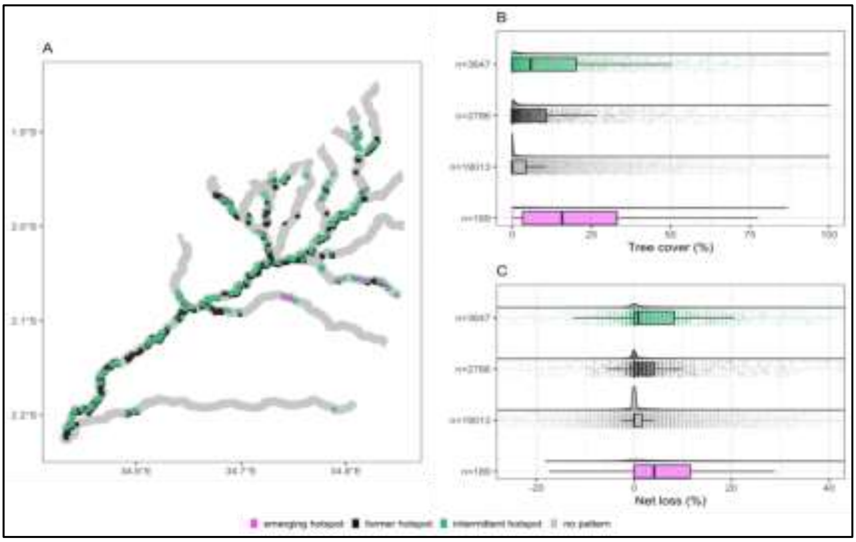

Figure 3.2: (A) Mapped nest hotspot results based on aerial riparian surveys flown from 2013 to 2021. Emerging hotspots are cells which were not classified as hotspots in early survey years (2013-2016), but were in recent survey years (2019-2021). Former hotspots are cells which were classified as hotspots in early survey years but were not in recent survey years. Intermittent hotspots are cells which were considered hotspots in one or two survey years but were not considered emergent or former hotspots. No pattern cells were not classified as hotspots across any survey year; (B) Tree cover (%) per cell by hotspot category in 2012; (C) Net tree cover loss from 2013 to2020 by hotspot category.

3.5 Results

3.5.1 Nesting Preferences

We recorded 57 nesting trees comprised of five tree species: *Ficus sycomoras* (n=33), *Senegalia polyacantha* (n=11), *Vachellia robusta* (n=10), *Trichillia sp*. (n=2) and *Ficus indica* (n=1). Nests were recorded at a mean height of 18.3 ± 6.1 m. We also recorded 43 non-nesting trees consisting of four tree species: *Senegalia polyacantha* (n=31), *Vachellia robusta* (n=6), *Ficus sycomoras* (n=5), and *Grewi*a sp (n=1). Elephant damage was not observed on any of the nesting or non-nesting trees.

Even though height of nesting trees (21.39 m) differed significantly (Welch's t=4.97, df =98, $p<0.05$) with height of non-nesting trees (17.59 m), no significant difference (Welch's t =1.58, df=98, $p >0.05$) in tree DBH was detected between nesting (1.14 cm) and non-nesting trees (0.97cm). Moreover, there was no significant difference in tree cover (Welch t=1.15, df =98, $p>0.05$) between nesting (13.27%) and non-nesting trees (9.58%). Modelling results indicated that the probability of nest presence was positively and significantly related to tree height. The best-performing model as indicated by AICc included tree height and tree species ($p = 0.0483$) (Table 3.1) are indicated by asterisks (*). Bolded are the two topmost models within Delta_AICc of 2.

Table 3.1: Results from Generalized Linear Model (GLM) on nesting preference of WBV along the Grumeti River. Models are arranged from best to least performing models based on the lowest Akaika's Information Criteria (AICc), the difference in AICc from the best and least performing models (Delta AICc) and AICc weights. Significant variables within each model with p <0.05

Model	AICc	Delta_AICc	AICcWt	P
Height* + species	**109.92**	**0.00**	**0.48**	**0.0483**
Species*	**111.97**	**2.05**	**0.17**	**<0.0001**
Height* + Canopy + species	112.17	2.25	0.15	0.0472
Canopy + species	114.21	4.29	0.06	>0.05
DBH + species	114.26	4.35	0.05	>0.05
DBH + height*+canopy + species	114.45	4.53	0.05	0.0446
Height*	116.12	6.21	0.02	<0.0001
Height* + canopy	118.13	8.22	0.01	<0.0001
DBH + height*	118.15	8.23	0.01	<0.0001
DBH + canopy +height*	120.08	10.17	0.00	<0.0001
DBH	138.13	28.21	0.00	0.126
Canopy	139.40	29.48	0.00	0.252
DBH + canopy	139.96	30.04	0.00	>0.05

3.5.2 Tree cover and nest hotspots

We observed a 28% reduction in riparian tree cover in the survey area between 2012 and 2020. Nesting hotspots were primarily identified along the Grumeti River, and nearly all hotspots were classified as former or intermittent (Fig. 3.2a). No areas were identified as persistent hotspots. The remaining drainages were largely classified as no-pattern areas. Emerging hotspots were rare and primarily restricted to Samisami, Rokare and Muyambi tributaries, rather than the Grumeti River, which is the main channel of this river system (Fig. 3.2b).

In 2012, tree cover (%) differed significantly and with moderate effect across hotspot categories (H=1675.7, df =3, p-value < 0.0001, $\varepsilon^2 = 0.07$) and was significantly greater ($p < 0.0001$) on average in emerging (21.2%) and intermittent (14.0%) hotspots than in former

(9.2%) or no pattern (5.9%) hotspots (Table 3.2). Similarly, in 2020 tree cover differed significantly and with moderate effect across hotspot categories (H=1136.2, df =3, p-value < 0.0001, ε^2 = 0.04) and was significantly ($p < 0.0001$) greater on average in emerging (14.7%) and intermittent (9.1%) hotspots than in former (5.9%) or no pattern (4.5%) hotspots (Table 3.2).

Net tree loss between 2012 and 2020 significantly differed across hotspot categories, although this relationship was weak (H=623.82, df =3, p-value < 0.0001, ε^2 = 0.02). On average, reductions in tree cover were observed across all hotspot categories (Fig. 3.2c). Net tree loss was significantly greater ($p < 0.0001$) on average in emerging (6.5%) and intermittent (4.9%) hotspots than in former (3.3%), and none were pattern hotspots (1.4%) hotspots (Table 3.2).

Table 3.2: Results of post hoc Dunn test from tree cover and Nest hotspot analysis showing pairwise comparisons of mean tree cover and mean tree loss across Emerging, Former, and Intermittent hotspot categories in 2012 and 2020. All p values are adjusted via a Bonferroni correction

	Year 2012			
	Emerging	Former	Intermittent	Mean tree cover (%)
Emerging	------	------	------	21.2
Former	p < 0.0001	-----	------	9.2
Intermittent	p < 0.0001	p < 0.0001	-----	14.0
No pattern	p < 0.0001	p < 0.0001	p < 0.0001	5.9
	Year 2020			
	Emerging	Former	Intermittent	Mean tree cover (%)
Emerging	------	------	------	14.7
Former	p < 0.0001	------	------	5.9
Intermittent	p = 0.06	p < 0.0001	------	9.1
No pattern	p < 0.0001	p < 0.0001	p < 0.0001	4.5
	Net loss from Year 2012 to Year 2020			
	Emerging	Former	Intermittent	Mean tree cover loss (%)
Emerging	------	------	------	6.5
Former	p < 0.0001	------	------	3.3
Intermittent	p = 0.002	p < 0.0001	------	4.9
No pattern	p < 0.0001	p < 0.0001	p < 0.0001	1.4

3.6 Discussion

In our study system, a 56% decline in nest encounter rate has been recorded previously (Laizer *et al.,* in prep). Similarly, the current study recorded spatial changes in active nest encounter rates in IGGR, suggesting that *Gyps africanus* nesting is changing too. Reduction in tree cover may be contributing to the observed changes in nesting site. We observed that vultures preferentially nested in areas of higher tree cover – in emerging and intermittent nest hotspots that were characterized by significantly greater tree cover than no-pattern areas. However, our findings suggest that these areas are at a risk of habitat loss and may become unsuitable for WBV nesting in the near future.

Establishing the drivers of tree cover loss was outside the scope of this study, but may be attributable to elephant browsing pressure (Morrison *et al.,* 2016;Vogel *et al.,* 2014).

Elephants are a primary driver in the changes of woody cover in the GSME (Kija *et al.,* 2020), and the overall Serengeti elephant population has increased by 16% between 1986 and 2020 (TAWIRI, 2020). Although we did not find evidence of elephant damage on suitable nesting trees, it is possible that over time elephants have reduced the number of trees available for nesting. Other reasons for tree loss include floods (Petit and Froend, 2001), fire (Pettit and Naiman, 2007), or lack of tree recruitment within the riparian zone (Kramer *et al.,* 2008). In IGGR, prescribed burns are used as a management tool and are unlikely to pose a threat to the riparian tree species. Nevertheless, wildfires emanating from human activities outside IGGR are common in the area, especially during late dry season (Hassan, 2010), and this could be causing tree death. However, further investigation into the drivers of tree loss is needed, as is a more on examination of tree availability.

With respect to nesting tree preference, we noted that our results align with previous works on the species. The WBV in our study system preferentially nested on *Ficus sycomorus* trees that were taller than 13 meters. The likely reason could be due to how tall the species can grow and how the branches are strong enough to hold the nest of a heavy, large bird (Chomba *et al.,* 2013;Virani *et al.,* 2010). However, a more extensive tree census across GSME would aid in resolving whether the availability of preferred nesting trees is a limiting factor for WBV nesting site preference in the study area — though other work suggests that this is unlikely (Kendall *et al.,* 2018).

To conclude, we can generalize that habitat loss is a driving factor for the observed declines of nest encounter rate in the western corridor of the GSME. Loss of tree cover affects the breeding population, and if not taken into consideration may lead to local extinction of the white-backed vultures in the system. In our tree cover analysis, we assumed that the composition of the riparian forest was mainly big trees, and these were the most preferred by White-backed vultures for nesting. This might have posed a constraint in understanding the influence of other factors such as tree availability. Therefore, an evaluation on the availability, sources of mortality, and recruitment of preferred nesting trees will aid in discerning the drivers behind decline in nest encounter rates, low levels of breeding success as well as habitat loss in the preferred nesting sites.

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CHAPTER FOUR

4.0 General Discussion, Conclusion and Recommendations

This study investigated the breeding population status and nesting preferences of the African White-backed vulture, *Gyps africanus* in IGGR in the western corridor of the GSME. The study found that the nest encounter rate had declined sharply in a period of nine years between the survey years. This decline matches the reported declines in other parts of Africa such as South Africa (Murn *et al.,* 2013), Botswana (Leepile *et al.,* 2020), and parts of West Africa (Ogada *et al.,* 2016). The major reason for declines in East Africa has been reported to be poisoning (Ogada *et al.,* 2016), a factor that this study did not investigate, but acknowledges the potentiality of this factor in influencing the the dynamics of the *Gyps africanus* population across its range where the threat exists for one reason or the other.

According to our tree cover survey results, vultures preferred to nest in areas with higher tree cover, and these areas suffered habitat loss. Therefore, a driving factor for the decreased nest encounter rates observed along the PRZ in the study area could be the decline in suitable breeding habitat especially the preferred trees. Since the *Gyps africanus* preferred big and tall trees along the river for nesting, our tree cover analysis assumed that the composition of the tree cover was the big trees that were preferred for nesting. The driver for the increased loss of the tree in IGGR was not well documented here but the effects of fire (Hassan, 2010) and elephant population increase (Vogel *et al.,* 2014) may not remain unvoiced.

Similarly, the breeding success was significantly very low compared to the rates that were reported previously in the ecosystem (Mundy *et al.,* 1992). However, the breeding success was only derived from a single year survey, and therefore, this is a fragile conclusion for IGGR. Most of the breeding attempts (egg laying) that happened during late breeding season never succeeded, suggesting that *Gyps* *africanus* is a species that depends on colonies for breeding as previously reported (Virani *et al.,* 2010). However, the nest failure drivers were not well studied because we did not have any camera traps to capture all the behaviors during breeding season.

Generally, there is a conservation concern on the breeding population of the *Gyps africanus* in the the western corridor of GSME. Speculated drivers for declines such as poisoning, trade of vulture parts and human disturbances should be well studied in the ecosystem together with habitat loss. This study’s findings suggests that a continued monitoring of the breeding population and expansion to other potential breeding areas will benefit the ecosystem and give information on the breeding population status as well as any changes in the behavior and habitats. Similarly, further investigation on tree availability as factor for nesting is essential in better understanding the tree loss aspect.

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APPENDIX

Appendix 1: Nest survey form

Observer		**Date**	
Nest ID		**Waypoint #**	
X		**Y**	
NEST CHARACTERISTICS			
Nest Status Simple	Nest fallen / destroyed	Tree fell	Active
	Inactive	Not found	Unable to confirm
Nest Status Detailed	Nest building	Complete and lined	Eggs
	Ad. Incubating	Eggs hatched	Chick age unkwn
	Chick ready to fly		
Nest Failure Details			
Nest location	AB	BL	
Species	AWB	WHV	HV
	LFV	RV	BE
	TE	OTH	
Other species			
Dist to nest		**Tree height**	
Nest total		**Tree species**	
Riparian zone	Yes	No	Unknown
Habitat	Forest	Woodland	Bushland
	Wooded grassland		
Elephant damage type	Bark stripping	Primary branch, not nest	Primary branch, nest
	Main stem, not nest	Main stem, nest	Tree pushed over
Elephant damage age	Recent	Old	
# Of photos taken		**Camera time**	
GoPro footage?	Yes / No	**GoPro time**	
DBH		**Canopy size**	
Notes			