Assessing translocation effects on an African elephant (*Loxodonta africana*) source population: demographics, landscape use and response to drones in Majete Wildlife Reserve, Malawi.

by

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Thesis presented in partial fulfilment of the requirements for the degree of
Master of Science

at
Stellenbosch University
Department of Conservation Ecology & Entomology, Faculty of AgriSciences

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March 2020

Declaration

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Wesley Hartmann

March 2020

Abstract

Translocations are increasingly common as a wildlife management strategy to reintroduce species that have undergone a local extirpation or to reinforce populations that have become isolated. Translocation effects have been well documented on the moved animals, however, much less is known about the effects these large-scale anthropogenic events have on source populations. The African elephant (*Loxodonta africana*) is a keystone species which plays a pivotal role in the ecosystems it inhabits. With their wide-ranging impacts, understanding how elephants utilize the space available to them, or how anthropogenic events such as translocations, could influence these patterns, is vital for the effective management of elephant populations. An additional human-associated disturbance on wildlife populations, although on a much smaller scale, is the use of technology for observational purposes. Generally, these technologies have enabled an unobtrusive means by which to detect and observe wildlife (i.e. remote-sensing camera traps), but increasingly the wildlife sciences are using unmanned aerial vehicles, more commonly known as drones, which are not unobtrusive. A great number of review papers have summarized and emphasized the drone's capabilities in the wildlife sciences, but few have investigated the effect drones have on animals themselves. This study reviews all published translocation events that occurred on the African continent between the years 2000 and 2019 in an attempt to determine the factors that influence the success of translocations in an African context, and investigates how a source elephant population responded demographically and spatially to a large translocation event in Majete Wildlife Reserve where (n=154) 42% of elephants were removed and 70% of adult females. The study also investigated how this population responded behaviourally, post translocation, to the approach and presence of a drone.

The demographic status of the Majete elephants was assessed via a combination of aerial survey data and individual identification techniques. Since the translocation, the population has increased from an estimated 200 individuals to 232 over a two-year period. A sex ratio of 5:2 male to female was found for adult elephants (older than 10 years) and a population growth rate of 7% per annum was estimated. The current growth rate is likely due to conception prior to the translocation event and is expected to decrease due to the extreme adult male bias in the current population. The removal of herds, primarily from one region within the reserve, significantly influenced the diversity of use of artificial water points by elephant herds. Herds historically only found in peripheral regions of the reserve were sighted more frequently in the areas other elephant herds had been removed from. While the population tolerated drone use reasonably well, increasing approach speeds and an approach angle of 90° (as opposed to 45°), were found to have significant negative effects on the likelihood of a successful drone approach towards elephants, regardless of sex and herd/group size. No flight or environmental variables were found to significantly influence the success of a sustained drone flight, however the outcome of the preceding approach was found to significantly influence success (GLZ, Estimate = 2.39497, $p < 0.0001$) i.e. a successful approach was more likely to result in a successful presence flight.

Opsomming

Die gebruik van translokasies as ‘n metode van wildbestuur word al hoe meer algemeen benut om spesies wat plaaslik uitgesterf het terug te plaas in hul natuurlike omgewing, of om dierebevolkings wat drasties geïsoleer geword het weer te versterk. Die impak van translokasie op hervestigde individue is goed gedokumenteer, maar minder kennis bestaan oor die impak van hierdie grootskaalse antropogeniese gebeurtenisse op bronbevolkings. Die Afrika-olifant (*Loxodonta africana*) is ‘n hoeksteen-spesie wat ‘n sleutelrol vervul in die ekosisteme waarin dit voorkom. As gevolg van hul alomvattende impak op die omgewing is dit noodsaaklik vir die effektiewe bestuur van olifantbevolkings om te verstaan hoe olifante beskikbare ruimte benut, en hoe antropogeniese gebeure soos translokasies hierdie patrone kan beïnvloed. Die gebruik van tegnologie vir waarnemingsdoeleindes is ‘n bykomende mens-geassosieerde steuring op wildbevolkings, hoewel op ‘n veel kleiner skaal. Oor die algemeen het hierdie tegnologieë grotendeels op ‘n onopsigtelike wyse plaasgevind en sodane die opsporing van dierelewe moontlik gemaak (d.w.s. deur die gebruik van afgeleë kamera-lokvalle), maar wildnavorsing maak toenemend gebruik van onbemande lugvoertuie, meer algemeen bekend as hommeltuie oftewel ‘*drones*’, wat nie onopvallend is nie. ‘n Groot aantal hersieningsartikels het alreeds die funksies van die hommeltuig in wildnavorsing opgesom en beklemtoon, maar enkele het al die effek wat hommeltuie op die dier self het ondersoek. Dié studie hersien alle gepubliseerde translokasie gebeure wat tussen die jare 2000 en 2019 op die Afrika-vasteland plaasgevind het, in ‘n poging om die faktore wat die sukses van translokasies in die Afrika-konteks bepaal te identifiseer, en ondersoek die demografiese en ruimtelike reaksie van ‘n olifant-bronbevolking op ‘n grootskaalse translokasiegebeurtenis in Majete wildreservaat waar 42% (n=154) van die olifante geherlokeer is, insluitende 70% van die volwasse wyfies. Die gedragsreaksie van hierdie bevolking op die benadering en teenwoordigheid van ‘n hommeltuig ná translokasie word ook in die studie ondersoek.

Die demografiese status van die Majete-olifante was geevalueer deur middel van ‘n kombinasie van lugopnames en individuele identifikasietegnieke. Sedert die translokasie het die bevolking oor ‘n periode van twee jaar van na raming 200 individue tot 232 toegeneem. ‘n Manlike tot vroulike geslagsverhouding van 5:2 was gevind vir volwasse olifante (ouer as 10 jaar), en ‘n bevolkingsgroeikoers van 7% per jaar word geskat. Die huidige groeikoers is waarskynlik te danke aan bevrugting voor die translokasiegebeurtenis, en sal na verwagting afneem as gevolg van die groot aantal volwasse manlike olifante in die huidige bevolking. Die verwydering van kuddes, hoofsaaklik van ‘n enkele streek in die reservaat, het die diversiteit in die benutting van kunsmatige waterpunte deur olifantkuddes beduidend beïnvloed. Kudde wat voorheen slegs in die perifere streke van die reservaat aangetref is, was meer gereeld waargeneem in die gebiede waarvandaan die ander olifantkuddes verwyder is. Alhoewel die bevolking die gebruik van die hommeltuig redelik goed verdra het, was daar gevind dat ‘n toenemende benaderingsnelheid en ‘n benaderingshoek van 90° (in teenstelling met 45°) ‘n beduidende negatiewe uitwerking op die waarskynlikheid van ‘n suksesvolle hommeltuig-benadering tot olifante gehad het, ongeag van die geslag en kuddegrootte. Daar was gevind dat geen vlug- of omgewingsveranderlikes die sukses van ‘n volgehoue hommeltuig-vlug beduidend beïnvloed het nie, maar die uitkoms van die voorafgaande benadering het wel die sukses daarvan beduidend beïnvloed (GLZ, Estimate = 2.39497, $p < 0.0001$), d.w.s. dat ‘n suksesvolle benadering meer waarskynlik was om in ‘n suksesvolle teenwoordigheidsvlug te ontaard.

This thesis is dedicated to my parents; your continued support and endless love is what has made this possible. Thank you.

Acknowledgements

I wish to express my sincere gratitude and appreciation to the following persons and institutions:

- Dr Alison Leslie, my supervisor. Without you this amazing opportunity would never have been possible.
- Dr Vicki Fishlock my Co-Supervisor, for your tireless patience, support and advice.
- Dr Craig Tambling for your endless help and support.
- The EarthWatch Institute for funding this research and all the volunteers that I had the privilege of meeting and working with.
- African Parks and Majete Wildlife Reserve, for letting me call a very special place on this planet 'home' for 14-months.
- DJI, the D1Store and Ilenna Copely for sponsoring the drone.
- Craig Hay and John Adendorf, (Reserve Managers of MWR), for your support and assistance throughout my time in MWR.
- Gervaz Tamala (Fields Operation Manager), Tizola Moyo (Head of Anti-Poaching in MWR) and Martin Awazi (Head of Law Enforcement in MWR), for your assistance and encouragement throughout my time in MWR.
- All the dedicated and brave scouts of MWR, special mention to Blessings Chikumbu, for keeping me safe throughout my time in MWR
- The various staff members within MWR, special mention to Mr Mlilo
- Mr Kapitapita of the Malawian Immigration Office
- To Sally Reece and Fleur Visser for your assistance in and out of the field, and for largely ensuring I remained sane during the tough months.
- My fellow students for all your assistance with viewing and proof reading and emotional support
- Rhiannon Jones, for more than you'll know

Preface

This master's thesis is composed of six chapters. Chapter One is an introduction to the subject matter, providing background information about the importance of having accurate demographic information regarding elephant populations, the impact large-scale anthropogenic events (i.e. translocations) can have on the source population, and the advancement in technology and its potential uses in the conservation sector, through highlighting the use of drones for observation data capturing. Additionally, the hypotheses, the goals of the study, the objectives, significance of the research and the assumptions and limitations of the study are all included in Chapter One. Chapters Two, Three, Four and Five have been compiled as stand-alone manuscripts to enable publication in peer-reviewed journals. As a result, there is some repetition between chapters. Chapter Six serves as both a discussion chapter for the entire thesis as well as a management recommendation document for Majete Wildlife Reserve, Malawi, and African Parks Majete (Pty) Ltd. Thus, the literature review is not discussed in Chapter Six.

Chapter 1	**Thesis Introduction and Outline**
Chapter 2	**Literature Review** A Review of African Wildlife Translocations in a Global Context.
Chapter 3	**Research results** The demographics of the African elephant (*Loxodonta africana*) population in Majete Wildlife Reserve, Malawi, two-years post translocation.
Chapter 4	**Research results** Artificial Waterpoint Use of a Source African elephant (*Loxodonta africana*) population in Majete Wildlife Reserve, Malawi, two-years post translocation.
Chapter 5	**Research results** First guidelines and ethical protocol for surveying African elephants (*Loxodonta africana*) using a drone.
Chapter 6	**General discussion, conclusions and Management Recommendations**

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Chapter One

Thesis Introduction and Outline

1.1. Introduction

Monitoring a species' demographic responses to ecological pressures and anthropogenic influences are of key importance for effective conservation strategies. However, this is often limited by insufficient data concerning the life history strategies of species across multiple biological systems (Stockwell *et al.* 2003; Owen-Smith *et al.* 2005). Comprehensive data informing the physical landscapes of species, such as demographics and spatial use, are often lacking and only generated from a single population (Gaillard *et al.* 1998; Gaillard *et al.* 2000). Even less is known about the social landscapes upon which species operate and function, yet these are equally important in understanding the behaviour and ecological strategies of individual animals, which have collective consequences for population-level behaviour. Successful management and conservation still depends on having a much more complete understanding of species-specific responses to a range of influences across a wide array of biological circumstances.

The African elephant (*Loxodonta africana*) (herein referred to as elephant/s) is a species whose demographics (Moss 2001; Kioko *et al.* 2013; Wittemyer *et al.* 2013; Jones *et al.* 2018), behaviour (Douglas-Hamilton 1973; Adams & Berg 1980) and ecology (Douglas-Hamilton 1973; van Aarde *et al.* 2008) have been studied in great detail. This is partly due to elephants being a keystone species and thus having the ability to drastically alter the ecosystems they occupy (Power *et al.* 1996). The ecological system components elephant influence, includes canopy cover (Dublin *et al.* 1990), species diversity and distribution (Pringle 2008), and seed dispersal (Spanbauer & Adler 2015). The location, resource availability, level of confinement of a population, and population density are all factors that can affect the manner in which elephants impact their environments (Skarpe *et al.* 2004; Kerley & Landman 2006; Chamaille-Jammes *et al.* 2008; Loarie *et al.* 2009) and elephants are capable of local habitat destruction while simultaneously producing and enriching many ecological benefits to the same environment (Gough and Kerley 2006; Morrison *et al.* 2016). Demographic data for elephants have theoretical value as an example of a large, long-lived, extremely social species with large effects on the environment, and has practical value for conservation strategies. Consequently, demographics of elephant populations are vital in understanding what the potential factors are that could contribute to elephant population effects on their respective environments.

Despite elephant populations in Central, East and West Africa declining (Thouless *et al.* 2016), elephant populations in protected areas in southern African states are growing rapidly (van Aarde & Jackson 2007; Maciejewski & Kerley 2014). Such rapid growth rate has been a major concern for many reserve managers as this often translates into highly negative impacts on the local environments due to vegetation damage and alteration by elephant populations in high densities (Owen-Smith 1996; Whyte *et al.* 2003; Guldemond & van Aarde 2007*;* Harris *et al.* 2008). Solutions including culling, translocation, and contraception have been proposed and trialled, but all have their respective downfalls (van Aarde *et al.* 1999;

Pimm & van Aarde 2001). Harris *et al.* (2008) astutely point out the 'obvious' paradox of using unnatural means to control a natural population and state that a 'simple' solution is to merely provide elephant populations with more space, but this is fundamentally compromised by a lack of available land. The immigration of people into communal lands adjacent to and often surrounding protected areas has resulted in a continuous loss of habitats for elephants (Cumming & Lynam 1997) and many other species (Ogutu *et al.* 2011). Protected areas continue to become increasingly fragmented and isolated, with a cascade of negative consequences (Newmark 2008; Ogutu *et al.* 2011).

Human-mediated movement of animals is not a new phenomenon, and due to increasing extinction (Ehrlich & Holdren 1971, as cited in Griffith *et al.* 1989; Ceballos & Ehrlich 2002; Baillie *et al.* 2004; Cardillo *et al.* 2006; IUCN 2018), reduction of biodiversity, and overcrowding of protected habitats (Wilson 1988, as cited in Griffith *et al.* 1989; Pinter-Wollman 2012), translocation programs have become important in wildlife management, both for conservation and other purposes (Griffith *et al*. 1989; Kleiman 1989; Stanley Price 1991; Wolf *et al.* 1998). Conservation translocations serve primarily as a management tool in the efforts to prevent the complete loss and depletion of imperilled species and populations, often utilizing captive-bred animals to either supplement or re-establish wild populations (Bangs & Fritts 1996; Shute *et al.* 2005; Faria *et al.* 2008; Fitzgerald *et al.* 2015). Bubac *et al.* (2019) conducted an extensive review of translocations across the globe with a specific focus on post-release monitoring. Despite rising threats to endemic species inhabiting Africa as a region (Myers *et al.* 2000; Sodhi *et al.* 2010), the degree of geographic bias in documenting and publishing translocation events found in previous studies (Fischer & Lindenmayer 2000; Seddon *et al.* 2014) has not lessened over the past two decades (Bubac *et al.* 2019), with the majority of translocation studies recorded in North America, Europe and Oceania. Additionally, sink populations are often the areas of interest (Skarpe *et al.* 2004; Millspaugh *et al.* 2006; IUCN, 2013; Briers-Louw 2019), with little or no understanding of how translocation events affect source populations. Understanding the impact that translocations have on the demographics and movement of source populations is important, as it allows for the determination of best translocation practices (i.e. how many individuals to remove and from where within an area).

Despite the obvious benefits of successful translocations, they inevitably disturb animal populations. Disrupted elephant populations typically experience two specific effects that may impact on social function: 1) the initial/immediate trauma that may accompany the disruptive event, and 2) the loss of opportunities for interacting with older, often more experienced members of a group that could act as appropriate role models or sources of knowledge (McComb *et al.* 2001; McAuliffe & Whitehead 2005; Greve *et al.* 2009; McComb *et al.* 2011). Initial trauma may involve the surviving members witnessing the death of group members, and it is now evident that early life social trauma may have profound effects on physiological development and adult behaviour patterns (Bradshaw *et al.* 2005; Lupien *et al.* 2009; Lukas *et al.* 2011). Both predictive (Bradshaw & Schore 2007; Lupien *et al.* 2009; Lukas *et al.* 2011) and experimental studies (Shannon *et al.* 2013) show that such traumatic events have subtle effects on learning, in particular interfering with the capacity to appropriately gauge adequate responses to both social and environmental stimuli. Millspaugh *et al.* (2006) demonstrated that translocation events were an immediate, acute stressor on individual elephants. Management may therefore play a role in minimizing external stressors during translocations and avoiding additional stressors post-translocation, so as to avoid potentially stressing individuals to the extent they incur harmful biological costs (e.g. impaired immune system) (Moberg 2000). Additional stress, which may not be avoidable, includes the translocation experience itself where animals are left in unfamiliar environments and/or with unfamiliar conspecifics, and warrants investigation, particularly for elephants where learning where to go and the cultural transmission of this and other knowledge occurs over decades (McComb *et al.* 2001; McComb *et al.* 2011).

Being deprived of opportunities to interact with appropriate role models affects knowledge acquisition and decision making (Shannon *et al*. 2013). This is particularly relevant in elephants where older individuals play pivotal leadership roles and manage decision-making in the context of both social and ecological threats (McComb *et al.* 2001; McAuliffe *et al.* 2005; McComb *et al.* 2011). In the absence of these older, experienced individuals, younger group members may be presented with fewer opportunities to learn the most apt response in dangerous situations (McComb *et al.* 2001; McComb *et al.* 2011; Thorton & Clutton-Brock 2011; Van Schaik & Burkart 2011). Additionally, elephant source populations may suffer stress if the removal of key individuals disrupt well defined social hierarchies and relationships (Gobush *et al.* 2008; Silk *et al.* 2010; IUCN, 2013). Atypical behavioural patterns that have arisen from socially disruptive events have the potential to be transmitted between generations, ultimately persisting in the long term (Shannon *et al.* 2013).

The development and use of new technologies have greatly aided the wildlife sciences, especially for species that are elusive, wide-ranging, sensitive to anthropogenic disturbances, and/or dangerous to approach, or who inhabit habitats which may be extensive, remote, challenging and in some cases impossible to access via foot on the ground level (Chabot & Bird 2015). One new technology currently rapidly gaining popularity, most likely due to interest in the commercial sector, is of an aerial nature. Known variously as unmanned aircraft systems (UAS), unmanned aerial vehicles (UAVs), remotely piloted aircraft systems or, most popularly drones, this burgeoning sector promises to offer further assistance to conservation efforts and the wildlife sciences. The popularity of drone use among wildlife biologists, ecologists and conservationists is clear from the many review papers investigating the applications and proliferation of drones in remote sensing, natural resource sciences and ecology (Watts *et al*. 2012; Anderson & Gaston 2013; Colomina & Molina 2014; Shahbazi *et al.* 2014; Whitehead & Hugenholtz 2014; Whitehead *et al.* 2014; Pajares 2015). Chabot and Bird (2015) conducted an extensive review of the applications of drones in wildlife management in which they highlighted optical surveying and observation of animals, uses in autonomous wildlife telemetry tracking, habitat research and monitoring as well as a review of the broader potential for UAVs. While the capabilities and potential practical uses of drones in the field of wildlife and conservation biology has been investigated thoroughly, their effects on the subjects of the studies (i.e. the animals themselves) has been investigated to a much lesser extent. Previous drone related research has been conducted to investigate the use of drones in mitigating human-elephant conflict (Hahn *et al.* 2016) and for general elephant population survey purposes (Vermeulen *et al.* 2013), but no protocol or ethical recommendations have been investigated and determined. With their sensitivity to anthropogenic impacts there is a clear need for guidelines as to how to conduct drone research focusing on elephants. The protocols developed in this study can be used to minimise stresses when utilizing drones for elephant related research.

In 2017, Majete Wildlife Reserve (MWR) in Malawi aided one of the largest elephant translocations in history. As a result, a great opportunity arose to investigate the demographic response of a remnant elephant population (source population) to such a large-scale change. Data such as population size, sex ratio and age structure provide a baseline for future population growth calculations. Additionally, changes in range use by the remaining elephants can be assessed by quantifying how they use regions "emptied" by the translocation. Quantifying how these elephants respond to drones allows development of ethical and appropriate flight protocols to ensure minimising further disturbance and stress to this population. The guidelines developed in this study can equally be applied to other populations where drone disturbance is of concern.

1.2. Research Question

The research question for this thesis consists primarily of two parts:

1) How has the translocation event of 2017 (i.e. the removal of 154 elephants out of MWR) impacted the demographics of MWRs' remaining elephant population as well as the source population's movement and utilization of areas throughout the reserve?
2) What is the best (i.e. least stressful flight protocol for an elephant) way to fly an unmanned aerial vehicle (aka a drone) around elephants for observational purposes?

1.3. Research Statement

This study presents the findings of the demographic changes to MWRs' elephant population and their utilization of certain areas throughout the reserve, two years after a major translocation event. An assessment was conducted on the population structure, including age class structure and sex ratio for 2018/2019.

An ethical flight protocol for drones was developed by determining which flight- (i.e. speed, angle of approach, starting altitude) and environmental-factors (i.e. total number of individuals in a group, if calves were present, season, weather, etc.) influenced elephant behaviour and affected their response to the drone. Data gathered in this study aims to assist in the management of MWRs' elephant population as well as inform conservationists about the potential impacts translocation events have on source populations. Furthermore, the ethical flight protocol developed in this study can be followed by researcher and filmmaker alike, to ensure the least amount of disturbance to the elephants they are wishing to observe aerially.

1.4. Hypotheses

1.4.1. Null Hypotheses

a) Two years post translocation, MWRs' elephant population has not increased, the sex ratio is not skewed and there is no change in age class structure.
b) The areas from which elephants were removed have remained empty and have not been filled by elephants from the surrounding areas and intensity of usage of waterholes has not changed.
c) Neither flight- nor environmental-factors will influence how elephants respond to the approach and presence of a drone.

1.4.2. Alternative Hypotheses

a) Two years post translocation, MWRs' elephant population is increasing: the sex ratio is severely skewed towards males (as the translocation only removed adult females and dependent offspring), adults (small, medium and large) form the largest proportion of the population, whereas calves and infants form the smallest.
b) The areas where elephants were predominantly removed from MWR, initially experienced a decrease in use, however the areas are now being used more by herds that were previously excluded, and the subsequent intensity of usage of artificial water points would also have decreased initially but has subsequently increased.
c) Flight-factors will influence how elephants respond to the approach and presence of a drone. Specifically, slow flight speeds and high starting altitudes will result in the least disturbance amongst the droned elephants. Additionally, the presence of calves and infants will make herds respond more sensitively to the approach and presence of a drone.

1.5. Research Goals

The main research goal is to assess the effects of translocation events on a remaining source population of African elephants. This study is intended to provide the management of MWR with detailed demographic data of their current elephant population. Additionally, knowledge of how large-scale translocation events impact source populations is provided and aims to inform conservationists prior to future translocations. Furthermore, the drone flight protocol that was developed aims to allow for the ethical observation/filming (whether for scientific or entertainment purposes) of elephants.

1.6. Research Objectives and Research Questions

The first objective was to determine how the demographics of MWRs' elephant population has changed two years post removal of 154 individuals.

a) What is the current elephant population size in MWR?
b) What is the population's sex ratio and age/size class structure? (number of males and females, number of adults, juveniles and calves)
c) How many elephant herds are remaining in the reserve?

The second objective was to determine whether the areas from which elephants were removed have been filled by herds previously excluded from these regions in the reserve, and if the removal of herds has resulted in a change in the intensity of usage of certain artificial water points.

a) What was the diversity of use (i.e. number of different herds visiting specific regions) of the three broad regions within MWR pre-translocation?
b) What is the diversity of use (i.e. number of different herds visiting specific regions) of the three broad regions within MWR post-translocation?
c) If herds have moved into the areas with less elephant density due to the translocation event, how long did it take before movement into these areas occurred?
d) Has the intensity of usage of artificial water points changed over time?

The third objective was to identify factors (both manipulated and environmental) that influence how elephants respond to the approach and presence of a drone.

a) How do approach speed, angle of approach and starting altitude influence elephant responses to the approach of a drone?
b) How do environmental factors, such as temperature, wind, the presence of calves and infants, and total number of individuals, influence elephant responses to the approach of a drone?
c) How do flight speed and flight pattern (fixed altitude vs. varied altitude) influence elephant responses to the sustained presence of a drone?
d) How do environmental factors, such as temperature, wind, the presence of calves and infants and, total number of individuals, influence elephant responses to the sustained presence of a drone?

1.7. Significance of Research

As the network of protected areas across the African continent grows (IUCN and UNEP-WCMC 2016), so too does the isolation of such areas, which is often due to anthropogenic landscape alterations (Olivier *et al.* 2009). In highly fragmented systems the likelihood of recruits locating and colonizing unoccupied areas is low (Craig *et al.* 2012). Ultimately, the long-term capacity of such areas to maintain viable populations of wildlife species is threatened by anthropogenic activities both within and outside of reserves (Newmark 2008).

With many populations confined to a fraction of their former ranges, reduced individual survival rates and increased extinction risks are possible consequences (Ceballos & Ehrlich 2002; Cardillo *et al.* 2006). As a result, translocation programs have become important in wildlife management (Griffith *et al.* 1989; Kleiman 1989; Stanley Price 1991; Wolf *et al.* 1998), however, such translocation events are profoundly affecting many free-ranging populations of these highly social mammals (Caswell *et al.* 1999; Campbell *et al.* 2008; Bouché *et al.* 2011) with a direct impact on the species' social structure (Shannon *et al.* 2013). Yet an in-depth and detailed understanding of precisely how these "anthropogenic disturbances" effect pivotal aspects of social function is lacking (Shannon *et al.* 2013). Specifically, elephant source populations may suffer stress if the removal of key individuals disrupt well defined social hierarchies and relationships (Gobush *et al.* 2008; Silk *et al.* 2010; IUCN, 2013). Additionally, source population demographics are rarely studied post-translocation and yet they provide an opportunity to fill a knowledge gap of demographic responses to anthropogenic influences.

Understanding not only how populations respond demographically to the removal of large numbers of individuals, but how their utilization of space changes is a key insight for the management of areas practicing translocations. It enables informed decision-making about not only how many, but the demographic break-down of the population removed as well as from which areas animals should be captured.

As for the final research chapter of this study, the development and use of new technologies have greatly aided the wildlife sciences. One new technology that is currently gaining popularity at a rapid rate are drones, which promise to offer further assistance to conservation efforts and the wildlife sciences. Their popularity among wildlife biologists, ecologists and conservationists is evident due to the many review papers investigating the applications and proliferation of drones in remote sensing, natural resource sciences and ecology (Watts *et al*. 2012; Anderson & Gaston 2013; Colomina & Molina 2014; Shahbazi *et al.* 2014; Whitehead & Hugenholtz 2014; Whitehead *et al.* 2014; Pajares 2015). While the capabilities and potential practical uses of drones in the field of wildlife and conservation biology has been investigated thoroughly, their effects on the subjects of the studies (i.e. the animals) has been investigated to a much lesser extent.

If we as conservation practitioners and biologists wish to use drones for observational studies, the influence of a drone on the subject's behaviour should first be determined. In an attempt to quantify how elephants respond to the approach and presence of a drone, an ethical flight protocol was created to ensure the least amount of disturbance possible for the elephants in question; thus enabling accurate data collection regarding behaviour when utilizing these relatively new aerial tools.

1.8. Scope of Limitations

In this project five main limitations were identified and considered.

First, African elephants are a long-lived species (Wittemyer *et al*. 2013) with the longest mammalian reproductive life as well as gestation period but a slow rate of reproduction (Moss 2001; Wittemyer *et al*. 2013). Since the Majete translocation event occurred only two years ago (2017), it was not possible to determine complete life-history information or detailed demographic status for the full population. However, a cross section of MWRs' elephant population's status was obtained for the years 2018/2019, providing valuable demographic information for the future management of MWRs' elephant population and for conservationists to consider the impact of translocations on source populations. Overall accessibility to and visibility of the Majete elephant population was a major challenge. Majete Wildlife Reserve's road network is fairly limited, leaving vast areas of the reserve inaccessible, or only passable by 4x4 vehicle. The condition of the roads within MWR were severely limiting during the wet season (December – April). Daily or even weekly transects of all the roads in the reserve was not possible. Visibility was greatly limited by the dense woodland vegetation that dominates the reserve, with a few small savanna areas that are located in the south. Furthermore, many elephants within these difficult to access areas were not habituated towards cars or humans and could not be observed for long enough to collect reliable demographic data. Camera traps were therefore used as a non-invasive means to collect demographic data within these regions.

Third, little to no demographic information was available from when elephants were first reintroduced into MWR (2006) until 2016, when the first detailed demographic study took place within the reserve (Forrer 2017). Findings from the current study could therefore not be compared with regards to how the population has changed over time, referring specifically to sex ratios and age groups, but simply between the 2016 and the 2018 survey.

Fourth, much of the data for Chapter Four utilized images captured by camera traps that were set-up as part of long-term monitoring within the reserve. There were periods of time when cameras were either simply not out in the field at waterholes or had been placed but not checked for lengthy periods of time, due to logistical challenges.

Fifth, and finally, external behavioural indicators can only provide so much information about the perception of stressors, and stress hormone analysis might provide a more detailed picture of the extent and duration of stress. This study acts as a starting point for the investigation into how elephants respond to drones, but further studies should be conducted investigating an elephant's physiological responses (heart rate, cortisol levels) to a drone.

1.9. Assumptions

a) Individual elephants that were captured on camera in a continuous sequence of photographs in a short period of time, between 10 to 60 minutes, were assumed to be from the same herd.
b) Once an extended amount of time, longer than 60 minutes, had elapsed between photographs, individual elephants were assumed to be from a different herd.
c) Bulls of the age categories, small, medium and large adults, were assumed to have left their original family herds (size classes were derived from the following literature: Moss 1996; Whitehouse & Hall-Martin 2000; Moss 2001). Bulls tend to leave their family herds between the ages of 11 and 13 years (Moss 1996; Whitehouse & Hall-Martin 2000; Moss 2001, range 8-18 years (VF pers. comm.). If an individual was captured on photograph with the same family herd multiple times, the individual was assumed to still belong to the herd. However, if an individual male estimated to be a small adult appeared alone or with different family herds in photographs, the individual's demographic information was noted and was assumed to range alone.
d) If more than one family herd was recorded while feeding or at waterholes, it was recorded and noted for potential family links between the two herds that may be part of a larger family grouping.
e) The outward observable behaviour of the elephants recorded served as a proxy for how the elephants responded to the drone.

1.10. Brief Chapter Overviews

This master's thesis is composed of six chapters. Chapter One is an introduction to the subject matter providing background information on the importance of accurate demographic information for elephant populations, the impact large-scale anthropogenic events (i.e. translocations) can have on the source population, and the rise and potential use of drones in the conservation sector. Chapter One includes the hypotheses, study goals and objectives, significance of the research and the assumptions and limitations of the study. Chapters Two, Three, Four and Five have been compiled as stand-alone manuscripts to enable publication in peer-reviewed journals. As a result, there is some repetition between chapters. Chapter Six serves as both a discussion chapter for the thesis as well as a management recommendation document for African Parks Majete (Pty) Ltd. Thus, the literature review is not discussed in Chapter Six.

Chapter Two is a literature review that provides information on the state of translocations across the African continent. Bubac *et al.* (2019) noted that despite rising threats to endemic species inhabiting Africa as a region (Myers *et al.* 2000; Sodhi *et al.* 2010) there has been no redress of the geographic bias in reporting or documenting translocation events found in previous studies (Fischer & Lindenmayer 2000; Seddon *et al.* 2014). Chapter Two aimed to address the noted geographic bias found throughout the translocation literature across four decades, with a specific focus on the post-release monitoring of translocated populations of African fauna.

Chapter Three describes how the population structure and demographics of MWRs' elephants have changed two years post translocation of 154 individuals out of the reserve. The resulting population age structure and sex ratio for 2018/2019 is discussed, alongside the likely potential impacts of the removal of so many individuals on the source population in terms of possible challenges in recruitment and social disruption.

Chapter Four investigates if the removal of individuals from specific areas within MWR has changed the intensity at which certain artificial water points are used by breeding herds, and allowed herds previously limited to other areas to move into areas vacated as a result of the translocation. Once again, the impacts of such a large anthropogenic event are discussed and notes of caution are made for future translocations as to how families and herds are removed.

Chapter Five describes the best flight protocols around elephants when using an unmanned aerial vehicle (aka drone), with two separate parts of the flight considered: the approach, and the sustained presence of a drone. Manipulated variables such as flight speed, angle of approach, starting altitude and general flight pattern were investigated for their influence on elephant behaviour. Environmental factors, including but not limited to, the number of individuals in a herd and if infants and calves were present, were also investigated. This chapter serves as a starting platform for future studies to investigate other elephant populations or even different taxonomic groups and attempts to quantify the possible influence of drones.

Chapter Six summarises the main research findings of this thesis in terms of population structure, utilization of areas within MWR and behavioural responses to the approach and presence of a drone. Additionally, recommendations are provided for the future management of MWR's elephants and a flight protocol is provided for drone users world-wide.

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Chapter Two

A Review of African Wildlife Translocations in a Global Context.

2.1. Abstract

Fischer and Lindenmayer (2000) conducted an extensive review of over 170 animal relocation cases over the preceding two decades. In 2019, Bubac *et al.* conducted a similar review, but focused on the trends in post-release monitoring (PRM) within the translocation literature. Despite rising threats to endemic species inhabiting Africa as a region, the degree of geographic bias in documenting and publishing translocation events has not decreased over the past two decades, with the majority of translocation studies recorded in North America, Europe and Oceania. One hundred and seventeen translocation events were reviewed in the current study with the majority of reviewed events taking place in South Africa (63). Geographic trends in the number of translocation events could not be explained by a country's gross national income (GNI) per capita or megafauna conservation index. Other possible explanatory variables may be the number of tertiary institutions within a country which may result in a higher number of documentations and subsequent publications of translocations, or a country's reliance on its wildlife tourism industry. Most events included some form of PRM (87) but did not result in higher rates of success where PRM did occur. Only one study and one review paper reported the costs of the translocation, and despite our increased knowledge in the field of reintroduction biology, success rates were found to decrease in more recent times. Further investigation is required into the translocation realm across the African continent and a simple, user-friendly open-access database is recommended so that reserve managers can easily record translocation events.

2.2. Introduction

An extensive network of protected areas has been created across the African continent over the last century (Chape *et al.* 2005). From 2005 and 2016, there was an increase of 66% in protected areas globally with an average growth rate of 4.4% per year (IUCN and UNEP-WCMC 2016). Within the same time frame, the African continent saw an increase of 36% in protected areas (WCMC 2016). By 2016 there were more than 1100 national parks in sub-Saharan Africa, and that number is expected to grow (WCMC 2016). Such areas play a pivotal role in the protection and conservation of many ecosystems and thus wildlife species, and encouragingly, the total protected-area coverage in Africa has doubled since 1970, and now includes 3.06 million km^2 of both terrestrial and marine habitats (WCMC 2016). Concurrently, many ecosystems have experienced significant changes due to habitat loss, fragmentation, climate alterations and land-use intensification (Watson & Watson 2015). Whilst the increase in number of protected areas across the continent is encouraging, these areas are becoming ever more fragmented and isolated.

Where habitats are contiguous and functional, connectivity is high, and animals are able to eventually locate and occupy suitable and previously vacant areas (Huxel & Hastings 1999). Conversely, in highly fragmented systems the likelihood of recruits locating and colonizing unoccupied areas is low (Craig *et al.* 2012). Ultimately, the long-term capacity of such areas to maintain viable populations of wildlife species is threatened by anthropogenic activities both within and outside of protected areas (Newmark 2008). Many wildlife populations across the world have become fragmented due to anthropogenic landscape alteration, both for agricultural and conservation purposes (Olivier *et al.* 2009). Protected areas become increasingly isolated by restricting animal movements and dispersal through habitat loss, fence and road development, over-hunting and the spread of diseases (Newmark 2008). This results in islands of protected areas within the ever-expanding human-dominated matrix. With many populations confined to a fraction of their former ranges, reduced individual survival rates and increased extinction risks are possible consequences (Ceballos & Ehrlich 2002; Cardillo *et al.* 2006).

Human-mediated movement of animals is not a new phenomenon, although historic motivations were generally for nutritional, recreational and aesthetic purposes (Griffith *et al.* 1898; Seddon 2010). Modern human-mediated movements, i.e. translocations, have become important in wildlife management (Griffith *et al*. 1989; Kleiman 1989; Stanley Price 1991; Wolf *et al.* 1998) as increasingly fragmented landscapes become increasingly vulnerable to extinction risks (Ehrlich & Holdren 1971, as cited in Griffith *et al.* 1989; Ceballos & Ehrlich 2002; Baillie *et al.* 2004; Cardillo *et al.* 2006; IUCN 2018), a reduction of biodiversity and overcrowding of overly small protected habitats (Wilson 1988, as cited in Griffith *et al.* 1989; Pinter-Wollman 2012).

Conservation translocations serve primarily as a management tool in the efforts to prevent the complete loss and depletion of imperilled species and populations, often utilizing captive-bred animals to either supplement or re-establish wild populations (Bangs & Fritts 1996; Shute *et al.* 2005; Faria *et al.* 2008; Fitzgerald *et al.* 2015). However, several historical studies suggest that many relocations, spanning a broad range of taxa, are unsuccessful (Griffith *et al.* 1989; Kleinman 1989; Dodd & Seigel 1991; Short *et al.* 1992; Wolf *et al.* 1998; Fischer & Lindenmayer 2000; Guy *et al.* 2015; Jenkins *et al.* 2015), as well as carrying significant costs (e.g. Kleiman et al., 1991; Lindburg 1992; Rahbek 1993, Weise *et al.* 2014; Boast *et al.* 2015). Naturally, the suspicion that translocations may not always be effective led to an increasing interest in determining which factors influenced the success of animal relocations. In an extensive review, Fischer and Lindenmayer (2000) describe previous studies (Griffith *et al.* 1989; Wolf *et al.* 1998; Reading *et al.* 1997) which utilized mailing surveys to wildlife managers in an attempt to detect the ecological and anthropogenic factors influencing relocation success. However, all these studies used data not readily available to most wildlife managers, many of whom typically rely on published material (Fischer & Lindenmayer 2000). Many studies were also limited by factors including (1) geographical limitations and addressing only ecological or anthropogenic issues (Griffith *et al.* 1989; Wolf *et al.* 1998; Reading *et al*. 1997), (2) a sore lack of structured and systematic empirical approaches (Kleiman 1989; Stanley Price 1991), and/or, (3) restrictions to specific taxonomic groups (Dodd & Seigel 1991; Stanley Price 1991; Short *et al.* 1992).

Fischer and Lindenmayer's review radically altered this approach; they assessed over 170 animal relocation case studies, as well as theoretical papers that were all published in major journals throughout the world across two decades (1980 – 2000). Fischer and Lindenmayer (2000) made several statements which must also be made in this review: a) by using published material, the sample of case studies used may be several years older than assessments made from mailing surveys, and b) a large number of the results of animal relocations are never recorded, let alone published, and thus the sample is likely to represent only a subset of all relocations conducted. However, the analysis and summary of the abundant published case studies on translocations enabled these authors to create a useful starting point for wildlife managers and conservation biologists. Finally, it must be noted that by reviewing published literature, there will be a bias towards successful relocation efforts which is also true for the method of using mailing surveys (Reading *et al.* 1997; Fischer & Lindenmayer 2000).

In the effort to improve future translocation outcomes, an evaluation of when and why translocation attempts failed was required (Sutherland *et al*. 2010). Emphasising the importance of a scientific approach when planning and conducting translocations enabled the development of an adaptive framework for such a purpose (Armstrong & Seddon 2008). Since Fischer and Lindenmayer's review the number of publications and case studies documenting conservation translocations has risen steeply (Seddon *et al*. 2012; Brichieri-Colombi & Moehrenschlager 2016; Bubac *et al.* 2019), and although a 24.3% increase in the number of animals reintroduced from 1998 to 2005 was reported (Seddon *et al.* 2007), it is likely that these values under-represent the true number of conservation translocations attempted. Poor documentation in the current literature, inaccessibility to the general public, early failed translocation events, and/or publication biases are all likely reasons as to why the actual number of translocation events conducted is difficult to determine (Fischer & Lindenmayer 2000; Bubac *et al.* 2019). Aiding the rise in reported translocation events was the formal recognition of the field of reintroduction biology in the mid-1990s, which followed an increased effort in research, including the specific testing of hypotheses and monitoring of reintroduction projects (Serena 1995; Seddon *et al*. 2007; Taylor *et al*. 2017). However, many studies are still not up to par, being descriptive and comprising of opportunistic evaluations (Seddon *et al.* 2007; Taylor *et al.* 2017).

Bajomi *et al.* (2010) highlighted the value of review papers for translocations as they noted the sheer number of publications, as well as the large range of publication sources, could make it challenging for conservationists to keep up with the accelerated growth of published resources. Historically reviews have assessed the success of translocations in the context of a single group of organisms (e.g., Cochran-Biederman *et al*. 2014; Miller *et al*. 2014), derived conclusions regarding both current and future trends of reintroduction biology (Seddon *et al*. 2007; Armstrong & Seddon 2008), gauged biases across taxa (Fischer & Lindenmayer 2000; Seddon *et al*. 2005; Bajomi *et al*. 2010), provided an overview of reintroduction methodologies (Seddon *et al*. 2014), evaluated the likelihood of translocation success in the context of a given species' conservation status (Brichieri-Colombi & Moehrenschlager 2016), and reviewed trends in post-release monitoring and how PRM length is associated with translocation failure (Bubac *et al.* 2019). This body of review literature has highlighted a few trends, for example an extreme under-representation of invertebrates in the translocation literature where most case studies/projects involve mammals or birds (Fischer & Lindenmayer 2000; Seddon *et al.* 2005). A second example is a continuation of the geographic bias Fischer and Lindenmayer noted, with most reintroductions occurring in developed areas including Europe, North America and Oceania (Seddon *et al*. 2014; Brichieri-Colombi & Moehrenschlager 2016; Bubac *et al.* 2019). Despite rising threats to endemic species inhabiting Africa as a region (Myers *et al.* 2000; Sodhi *et al.* 2010), this degree of geographic bias in publishing translocation events as found in previous studies, has not lessened over the past two decades.

Conducting post-release monitoring (PRM) following a translocation event ensures acceptable population establishment, growth, and viability (Armstrong & Seddon, 2008), which are metrics often used to evaluate whether a translocation has been successful (Wolf *et al*. 1998; Bubac *et al.* 2019). Insufficient monitoring can lead to a translocation event being incorrectly labelled as 'successful' (Fischer & Lindenmayer, 2000), jeopardizing continued management support and resources for the translocated system (Bubac *et al.* 2019). PRM is therefore a key issue regarding translocations and their subsequent success or failure yet is poorly followed up across Africa. Species with slow life histories require longer PRM (Bubac *et al.* 2019), presenting extra challenges to resource-limited wildlife managers. Bubac *et al.* (2019) recommend a minimum PRM period of four years after a translocation, as most failures occurred within this time frame. Specifically, African elephants (*Loxodonta africana*) are labelled as a 'High Risk Translocation Group' (Bubac *et al.* 2019), yet virtually no PRM occurs in elephant translocation efforts. Elephant translocations are frequently responses to perceived management issues such as human-wildlife conflict (HWC), and the limited financial means (and associated conservation or academic resources) further compounds the issue of PRM.

By reviewing translocations across Africa, this chapter aimed to address geographic bias but also attempted to evaluate the primary reasons for translocations on the African continent and to attempt to decipher why reporting of African translocations is so low. Additionally, the status of PRM on the African continent was investigated and its effect on the scoring of translocation success was determined. It further aims to highlight the lack of reporting/monitoring of translocation source populations.

2.3. Methods

2.3.1. Definitions of terms

In this paper, slightly different definitions were used than those by Fischer and Lindenmayer (2000) due to the IUCN/SSC updating its definitions in 2013. Fischer and Lindenmayer defined the term 'relocation' as "any intentional movement by humans of an animal or a population of animals from one location to another". This was done to create a neutral overarching term in the hopes to avoid the confusion which other terms may cause. One such example of a confusing term, is "translocation", as it is used as both an overarching term (e.g. Griffith *et al.*

1989) and as a specific type of relocation, i.e. the "capture and transfer of free-ranging animals from one part of their historic geographic range to another" (Kleinman 1989). However, the latest definition given by the IUCN/SSC (2013) of the term 'translocation' is "the human-mediated movement of living organisms from one area, with release in another", which is very similar to the definition of Fischer and Lindenmayer's 'relocation'. Thus, the term 'translocation' shall be used throughout this paper as an overarching term. Table 2.1 provides a comparison between the terminology used by Fischer and Lindenmayer (2000) and that used throughout this paper.

Both the historical definition and colloquial use of the term translocation are very broad; 'translocations' may involve moving living organisms from the wild or from captive environments, they may be accidental (stowaways) or intentional, and the latter translocations can be conducted due to a wide variety of motivations (e.g. reducing/increasing population size, for animal welfare, for political, commercial or recreational interests, or for conservation purposes) (IUCN/SSC 2013). Of primary focus here is the *conservation translocation*, which the IUCN defines as "the intentional movement and release of a living organism where the primary objective is a conservation benefit". The aim of these translocations is generally to improve the conservation status of the focal species on a local or global scale, and/or restore natural processes or ecosystem functions. It is important to note whether the focal species is being released within or outside of its indigenous range, thus conservation translocations can either be *population restorations*, which involve releasing the organism within its indigenous range, or *conservation introductions*, which comprises of the movement and release of an organism outside of its indigenous range (IUCN/SSC 2013).

Population restorations comprise of two primary activities: 1) *reinforcement,* which is the purposeful movement and release of an organism into an existing population of conspecifics, and 2) *reintroduction*, which is the deliberate movement and release of an organism inside its indigenous range from which it has gone locally extinct (IUCN/SSC 2013). The main aims of *reinforcements* are to improve population viability, which may be achieved by increasing genetic diversity and population size, or by increasing the representation of particular demographic groups. *Reintroductions* simply aim to re-establish a viable population of the focal species within its indigenous range (IUCN/SSC 2013). The goal of the conservationist and relevant parties therefore enables for a more specific definition of translocation.

The remainder of this paper will deal exclusively with reintroductions, conservation translocations and reinforcements, following Fischer and Lindenmayer (2000).

Table 2.1. A conversion of the terms and definitions used by Fischer and Lindenmayer (2000) in their review of animal translocations based on the IUCN's definitions stated in 1996 to the terms and definitions used in this paper based on the IUCN/SCC's latest definitions (2013).

Fischer and Lindenmayer Terms	IUCN (1996)/Fischer & Lindenmayer (2000) Definitions	Term used throughout this Paper	IUCN/SSC (2013) Definitions
Relocation	any intentional movement by humans of an animal or a population of animals from one location to another	**Translocation**	the human-mediated movement of living organisms from one area, with release in another
Introduction	an attempt to establish a species, for the purpose of conservation, outside its recorded distribution but within an appropriate habitat and ecogeographical area.	**Conservation Introduction**	the intentional movement and release of an organism outside its indigenous range
Re-introduction	an attempt to establish a species in an area which was once part of its historical range, but from which it has been extirpated or become extinct.	**Reintroduction**	the intentional movement and release of an organism inside its indigenous range from which it has disappeared.
Translocation	the deliberate and mediated movement of wild individuals or populations from one part of their range to another	**Conservation Translocation**	the intentional movement and release of a wild living organism from one part of their range to another where the primary objective is a conservation benefit
Supplementation	the addition of individuals to an existing population of conspecifics	**Reinforcement**	the intentional movement and release of an organism into an existing population of conspecifics.

2.3.2. Literature Search

Seven major international journals were explicitly searched, as well as opportunistically collecting articles from other journals (See Table 2.2). The majority of journals selected were the same as those reviewed by Fischer and Lindenmayer (2000) as to allow for the best means of comparison. Articles and case studies that were published between January 2000 and October 2019 were reviewed. Utilizing many of the journals own 'Advanced Search' functions, the following was entered in order to find appropriate articles and case studies: "(translocate* OR relocate* OR introduct* OR reintroduct* OR supplementat* OR reinforcement) AND Africa".

Table 2.2. A summary of the journals that were scrutinized, and the types of translocation studies and organisms relocated.

Journal	No. of articles	No. of case studies	No. of reintroductions	No. of translocations	No. of reinforcements	No. of mammals	No. of birds	No. of others
Wildlife Man.	1	1	0	4	9	14	0	0
Biol. Cons.	2	1	1	2	0	3	3	0
Afric. Wild. Res.	8	0	4	9	3	15	0	0
Oryx	3	0	26	11	11	47	0	0
Int. Zoo Yb.	0	1	4	2	1	5	2	1
Biod. & Cons.	4	2	5	7	1	11	1	1
Others	9	1	7	9	1	15	0	0
Total	27	6	47	44	26	109	6	2

2.3.3. Creation of a database and analysis of data

A database of all appropriate literature was created using MS Excel, and the following questions asked of each case study:

(1) What taxonomic group was examined?

(2) Was the reason for the original decline explicitly stated?

(3) If so, was it successfully addressed before the relocation took place?

(4) In which part of Africa did the relocation take place?

(5) In what year was the case study apparently first reported in the literature? The aim of this question was to detect broad trends in the frequency and success of relocations through time.

(6) Was the only objective to conserve the species released?

(7) What type of translocation was undertaken (using the definitions in Table 1)?

(8) Did the individuals released originate from a captive or wild population?

(9) What was the cost of the translocation?

(10) Were predators absent in the release area?

(11) Were any supportive measures taken? These were defined as soft release, predator control, provision of food or shelter, habitat modification, special veterinary care, etc. Studies that somehow pre-conditioned animals prior to release or gradually exposed individuals to their new environment were defined as "soft releases". "Hard-releases" were assumed if no mention of pre-conditioning or extensive boma holding periods occurred.

(12) Did any form of PRM take place, and if so, what type and for how long?

(13) Was the relocation a success?

Arguably, one of the most difficult aspects of this review, as noted by Fischer and Lindenmayer (2000), was defining what success means in the translocation of animals. Similar success metrics will be used as those employed in the latter authors' review. A reintroduction was considered a success if it resulted in a self-sustaining population (Griffith *et al.* 1989; Fischer & Lindenmayer 2000). Doff and Sigel (1991) note that this benchmark generally takes a long time, resulting in many of the reintroduction case studies in Fischer and Lindenmayer's review to be classified as "unknown". Determining whether success had been achieved in a conservation translocation was similarly difficult to accomplish as it was dependent on the objective of the specific translocation, which in many cases was not stated. Fischer and Lindenmayer only defined conservation translocations as "failures" if "they clearly did not meet the objectives stated in that specific paper". Likewise, assessing the success of reinforcements was tricky and rarely determined due to the large variance in objectives. It should be noted, that by using the aforementioned definition of failure, Fischer and Lindenmayer may have unintentionally biased against failure. Instead, in the cases where the success of a translocation was not able to be determined, it would be 'more fair' to simply say "not stated", as was done in this review. By doing so, a large gap in proper reporting of both objectives and success metrics in translocation publications may be eliminated. This lack of proper reporting may be driven by the need to publish relatively quickly post-operation, as well as insufficient funding for PRM programmes, but then lacks the necessary follow up to properly evaluate operations.

What the following review found was largely influenced by what the authors of the reviewed articles and case studies decided to publish. As a result (a) only broad trends were identified from the information obtained, and (b) often the set of answers attained was largely incomplete despite the simple questions asked. No published articles or case studies documenting translocations on the African continent from 2015 – 2019 were detected in the literature search. A study was conducting post-release monitoring of a translocated individual/group of individuals during this period (e.g. Briers-Louw *et al.* 2019), but in all cases the monitored animals were translocated before 2015.

2.3.4. Reporting of results

Due to the descriptive nature of the data collected, only tables and histograms were utilized in the analysis. Previous studies employed more refined statistical techniques (e.g. regression analysis; Griffith *et al.* 1989; Wolf *et al.* 1998), but these seemed inappropriate given the relatively sparse amount of detailed information available from the majority of articles, and because these techniques have inadvertently misguided decision makers. Statistical models may over-estimate the chances of success when they are extrapolated to different environmental conditions (Fischer & Lindenmayer 2000). Thus, the goal of this paper was not to test for statistical significance of the findings, but rather to locate and highlight some of the most predominant trends in the current literature.

2.4. Results

2.4.1. General trends

The database consisted of 117 translocation events from 27 journal articles, 6 case studies and 4 review papers (Table 2.2). Reintroductions were the most common type of translocation event across Africa (47/117), closely followed by conservation translocations (44/117), with reinforcements occurring the least (26/117). Unlike previous reviews (Fischer & Lindenmayer 2000; Bubac *et al.* 2019), where an almost even split between the number of bird and mammal translocations was recorded, an extreme bias was found with mammals accounting for 93% of all translocation studies. Of the 117 translocations, 85 (74%) were for conservation purposes, while 28 (23%) were conducted to reduce or attempt to resolve HWC, with the remaining 4 (3%) not stating the purpose of the translocation.

2.4.2. Costs

Costs were reported in only one study and one review paper. Boast *et al.* (2015) calculated that the total cost to translocate and monitor one problem cheetah within Namibia was US$7110, with the bulk of the costs ($4340) contributing toward the monitoring aspects (GPS satellite collar and data retrieval). The authors stated that the cost of a single problem-cheetah translocation could compensate for at least 12 head of livestock (using figures from Fontúrbel & Simonetti 2011). In a review investigating translocations as a way to reduce human-carnivore conflict in a global context, Fontúrbel and Simonetti (2011) estimated the translocation cost per individual to be $3756, which they state is a sum equivalent to compensate for up to 30 head of livestock. They concluded that translocation of problem carnivores is a costly and mostly ineffective conservation practice, and alternative strategies such as livestock compensation and supporting best herding practices are a better way forward. Boast *et al.* (2015) concluded that translocations draw on already limited resources of many conservation organizations and state wildlife departments, in addition to diverting personnel and equipment away from other conflict mitigation activities, which is of particular concern in the African context where funding is an extremely limiting resource.

2.4.3. Translocation success through time

No published studies were identified from 2015 – 2019, and the preceding 15 years were divided into five-year time periods for the analysis. Of the 117 translocation events, 68 (58%) were classified as successful, 40 (34%) as unsuccessful and 9 (8%) as unknown, due to either the authors stating the success of the translocation was unknown at the time of publication, or the outcome simply not being stated. The success of more recent studies was more likely to be unknown than earlier studies and likely accounts for the decrease in successful translocation outcomes over the years (Table 2.3).

Table 2.3. Translocation success through time.

Period	No. of Studies	No. of Successes	No. of Failures	No. Unknown
2000-2004	67	44 (66%)	21 (31%)	2 (3%)
2005-2009	37	18 (49%)	15 (41%)	4 (11%)
2010-2014	13	6 (46%)	4 (31%)	3 (23%)
Total	117			

2.4.4. Translocations across the continent

South Africa was the African country with the greatest number of translocation events reported occurring over the last two decades (n = 63 [54%]), with Morocco experiencing only one (1%) (Figure 2.1), with only eight of the 54 African states reporting wildlife translocations. The proportion of each type of translocation event varied between each country (Figure 2.1). Most countries had only a single translocation type reported, e.g. Malawi and Tanzania, and South Africa was the only country where translocations consisted of all three types.

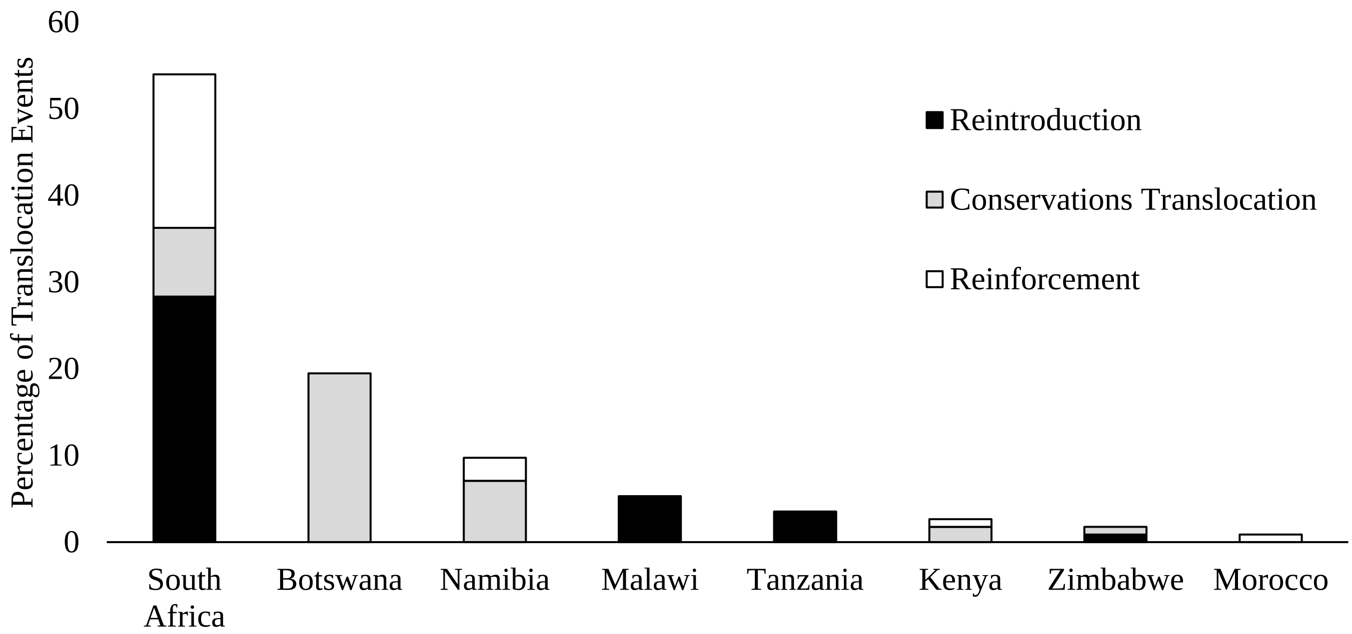

Figure 2.1. The types of translocations that occurred across the African continent 2000-2014 broken down by the country in which they occurred.

2.4.5. Translocation Success in relation to source population (Captive vs Wild)

Of the 117 translocations, the majority (n= 98) involved a wild source population (Table 2.4). The success rate for translocations from a wild source population was relatively similar to that from a captive source population (59% and 58% respectively), however, the substantial difference in the number of studies examined between the two different types of source populations should be kept in mind.

2.4.6. Translocation success in relation to supportive measures

Table 2.4. The success of a translocation in relation to the animal source population

Source of translocated population	No. of Studies	No. of Success	No. of Failure	Unknown
Wild	98	58 (59%)	36 (37%)	4 (4%)
Captive	12	7 (58%)	3 (25%)	2 (17%)
Unknown	7	3 (43%)	1 (14%)	3 (43%)

Of the translocations investigated, only two provided some form of supportive measure (i.e. habitat modification, supplemental food, provision of shelter), to the translocated individuals, and therefore no comparisons can be justified.

2.4.7. Translocation success in relation to removing the cause of decline

Only a slight majority of translocation events (66; 56%) clearly stated the initial cause of decline of the translocated animal, as well as successfully removing or mitigating the cause. Within this subset (Known and Removed), the highest percentage translocation success was identified (74%) along with the second lowest

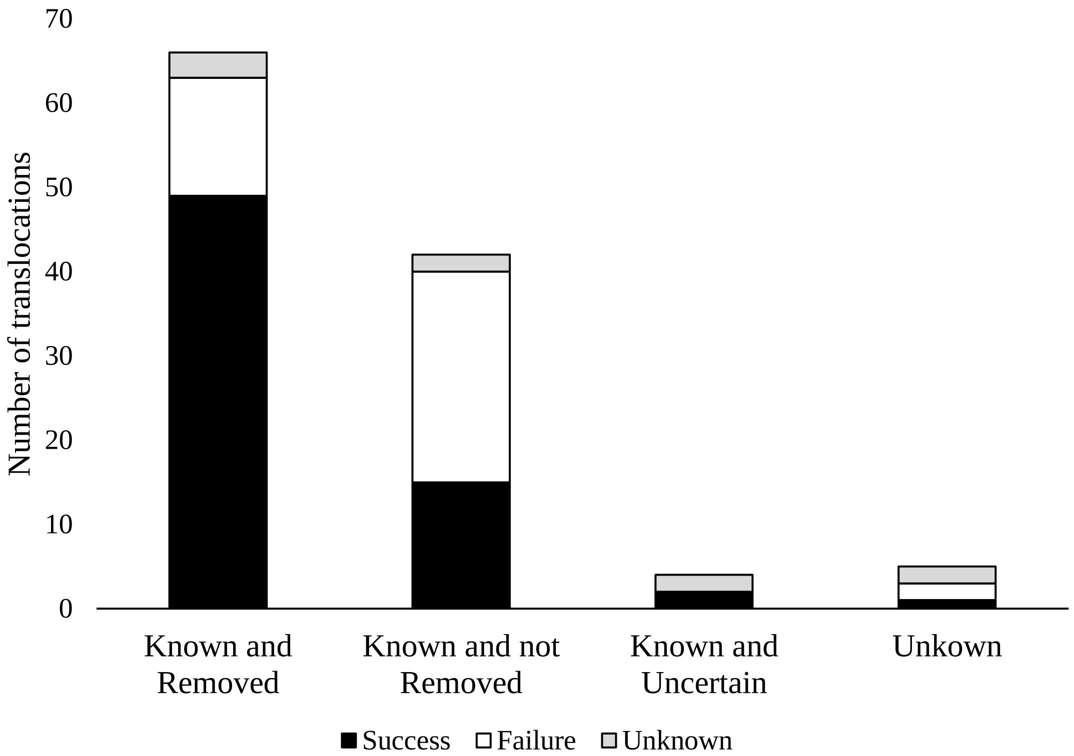

Figure 2.2. Effect of knowing and effectively removing the initial cause of decline on translocation success. “Known and Uncertain” refers to cases where the initial cause of decline was known, but it was uncertain whether this was effectively removed.

rate of translocation failure (21%); second to the “Known and Uncertain” group, which only consisted of four translocation events (Figure 2.2). “Known and not Removed” was found to have the highest percentage of failed translocations (60%) followed by the “Unknown” group (40%).

2.4.8. Post-Release Monitoring (PRM) across the African continent

The majority of translocation events recorded included some form of PRM (87 events, 74%), with 21% not including any PRM. For the remaining five percent of translocation events it could not be determined if PRM was carried out. All countries recorded included at least one translocation event where PRM occurred (Figure 2.4). Both translocation with and without PRM had relatively similar rates of success (61% and 63%, respectively) (Figure 2.3). The mean length of time of PRM instances was 24 months, with the shortest being one month and the longest still on-going (>200 months).

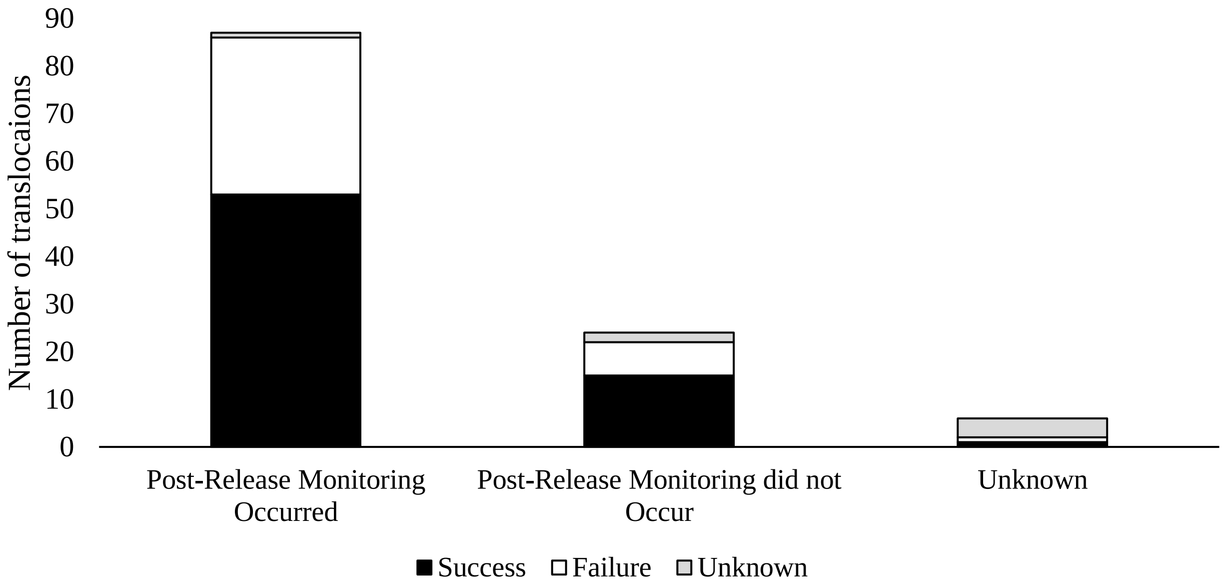

Figure 2.3. The breakdown of outcomes (i.e. success, failure or unknown) when separated into studies that conducted PRM or not.

2.4.9. Translocation Events and the Monitoring of Source Populations

None of the studies investigated involved monitoring the source populations from which the translocated animals were taken. The closest involved a study which investigated whether the pre-translocated animals would survive in their new environment (i.e. the hunting ability of captive-born tigers, Fàbregas *et al.* 2012). All other studies solely looked at the performance and outcome of the translocated animals.

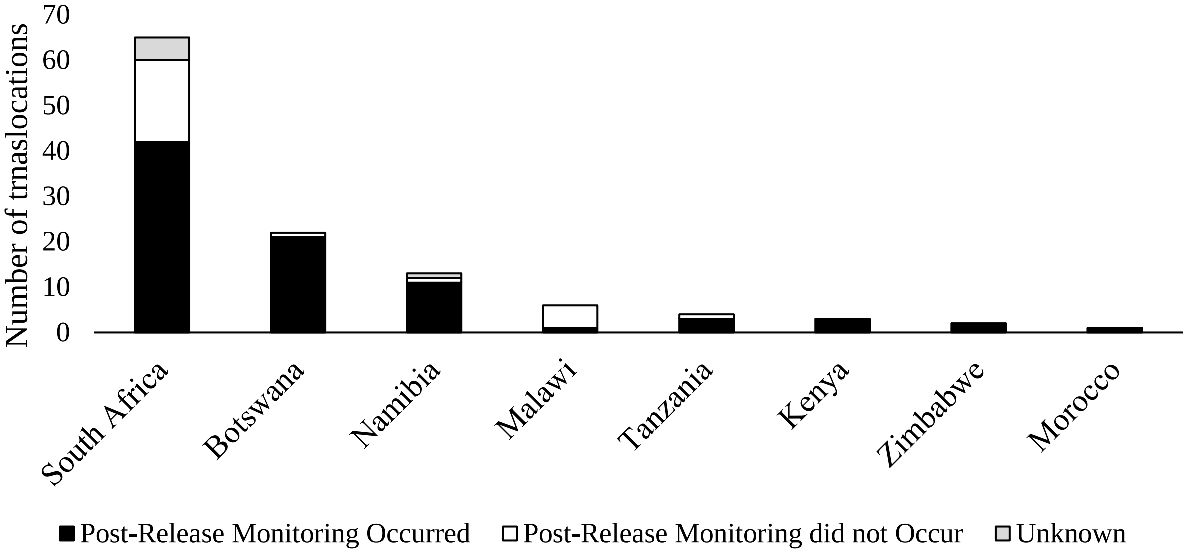

Figure 2.4. Where post-release monitoring occurred after translocations across the African continent.

2.4.10. Why translocations occurred where they did

In an attempt to investigate the large disparity in translocation events between African countries, the total number of translocations conducted in a country was compared with its relative GNI/capita (Gross National Income per capita, The World Bank) and Megafauna Conservation Index (MCI) (Lindsey *et al.* 2017) (Figure 2.5). While South Africa had the highest number of translocation events (65), it scored the lowest MCI (60/100). Conversely, many countries which had significantly fewer translocations had extremely high MCI values (e.g. Namibia, n = 13, MCI = 99/100; Malawi, n = 6, MCI = 91; Zimbabwe, n = 2, MCI = 96/100).

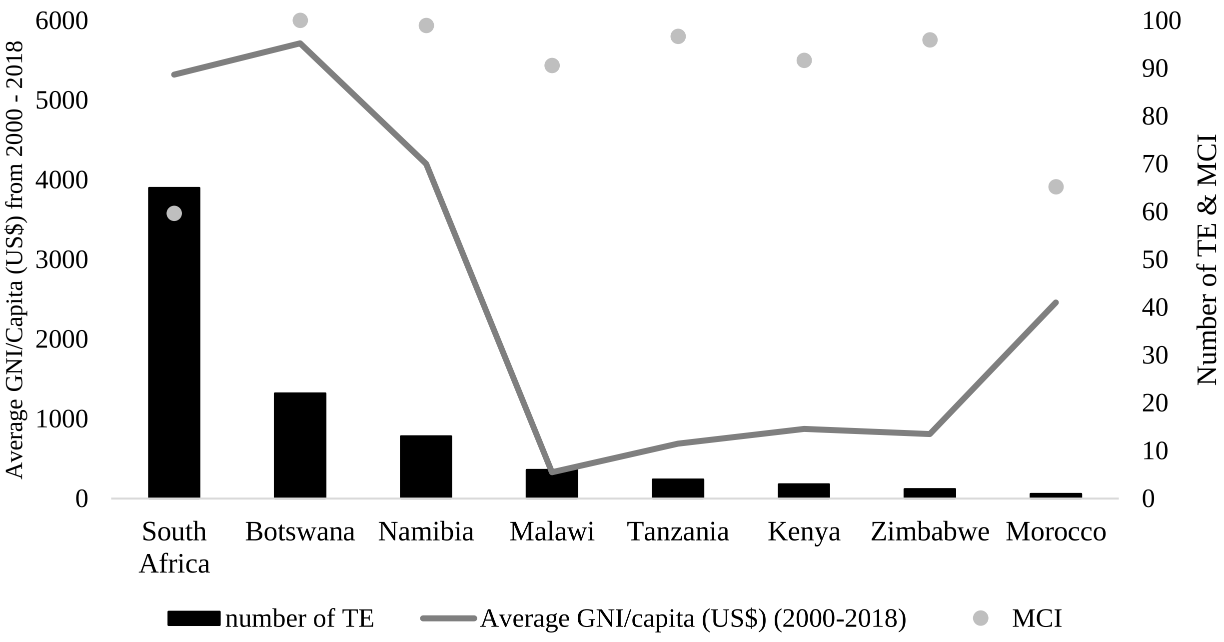

Figure 2.5. The number of Translocation Events (TE)[ł] that occurred in each country in comparison to their average GNI per capita (calculated from 2000 – 2018, data from the World Bank) [*] as well as their Megafauna Conservation Index (MCI) [ł **]. ł - y-Axis on the right-hand side. * - y-Axis on the left-hand side. ** - As calculated by Lindsey *et al.* (2017).

Similarly, a country's GNI/capita (averaged from 2000-2018) was not a good predictor of the number of translocation events in a country, demonstrated by Morocco, which only had one recorded translocation, yet had the fourth highest average GNI/capita during that time period.

2.5. Discussion

2.5.1. General Trends and Costs

The trend of reintroduction programmes being the most common translocation type over the last two decades across the African continent is not surprising, as the number of unprotected wilderness areas across the African continent is decreasing (Chardonnet 2019). Areas now coming under protection, where animals historically occurred but were extirpated by poaching, have seen an increase in the number of reintroduction events. It is predicted, that as the rate of new protected areas decreases, so too will the number of reintroduction events, while the number of conservation translocations and reinforcements are expected to increase.

Despite the recommendation almost two decades ago (Fischer & Lindenmayer 2000) the vast majority of published studies do not report translocation costs. In both cases where costs were reported, it was determined that a more cost-effective alternative to a translocation might simply be replacing the livestock killed by the problem animal (Fontúrbel & Simonetti 2011; Boast *et al.* 2015). It was concluded that the translocation of problem carnivores is a costly and mostly ineffective conservation practice, and alternative strategies such as livestock compensation and best herding practices are a better way forward (Fontúrbel & Simonetti 2011). Furthermore, although the costs were not explicitly stated, Weise *et al.* (2015) provided supporting information that a lone brown hyaena may be translocated successfully but warned that this should not be the standard management approach, as the costs were equivalent to those incurred by replacing killed livestock. Summarizing their study, Boast *et al.* (2015) state that translocations draw on the already limited resources of many conservation organizations and state wildlife departments in addition to diverting personnel and equipment away from other conflict mitigation activities, which is of particular concern in the African context where funding is an extremely limiting resource. The high costs associated with translocations is only another reason to ensure the maximum likelihood of success when conducting a translocation.

2.5.2. Translocation for Conservation

Consistent with what was found almost two decades ago by Fischer & Lindenmayer (2000), the majority of translocation events were for a conservation purpose (74%), with 23% conducted in an attempt to reduce/resolve HWC and the remaining 3% were without an explicit explanation. The proportion of HWC translocations can only be expected to grow as a growing human population, with the subsequent conservation of land for anthropogenic use, increases HWC across the continent (Hutton & Leader-Williams 2003; Madden 2004). Human-wildlife conflict may be species specific, such as human-elephant conflict (HEC), or guild specific, for example human-carnivore conflict (HCC). However, generally the successful resolution for all HWC is such that the levels of antagonistic interactions between humans and wildlife decreases to some level acceptable to humans. For HCC, success would entail the reduction of conflict (i.e. the killing of livestock) at the source site. Of the reviewed translocations that were conducted for the purpose of reducing HWC, only nine (32%) were successful. Often, animals returned to the source site of translocation (e.g. Pinter-Wollman 2009, Boast *et al.* 2015), resulting in failure to achieve the specific aim. These findings are consistent with previous studies and reviews (Linnell *et al.* 1997; Fischer & Lindenmayer 2000). However, it should be noted that studies in which animals did not return to the translocation source site, explicitly stated the distance moved (e.g. Pinter-Wollman *et al.* 2009, Weise *et al.* 2015a) and should be consulted prior to future translocation events.

Historically, translocations to resolve HEC entailed moving elephants from areas with high levels of HEC to areas with minimal conflict, but that could sustain a larger elephant population (Pinter-Wollman 2012). Resultantly, elephant numbers at the source site were reduced, as well as increasing the size of suitable habitat for both the source and the translocated elephant populations (Pinter-Wollman 2012). Worryingly, Pinter-Wollman (2012) stated that to the author's knowledge, there were no published accounts of whether translocations as a management strategy achieved the goals both economically and in terms of reducing HEC. The findings of this study are consistent with Pinter-Wollman's (2012) remarks. With only two published studies regarding translocating elephants as a means to reduce HEC, more needs to be done in terms of general reporting. Additionally, how translocation events effect elephant behaviour is reported even less, with only one study focusing on this issue in a translocated elephant population (Pinter-Wollman 2009). No studies were found that investigated how translocations influenced the source populations at all (i.e. demographically, behaviourally or ecologically).

Future studies should not only document the number of animals moved, and the resulting survival rates of the population, but how both the source- and sink-populations respond to translocations as a whole.

2.5.2.1. Taxonomic Groups

The vast majority of translocation events recorded across the African continent involved mammals. It has previously been suggested that this may indicate translocations are an appropriate conservation strategy for mammals (Fischer & Lindenmayer 2000). Mammals only account for 0.25% of species globally (Rahbeck 1993), and thus receive extremely disproportionate levels of conservation energy. Globally, and also more recently, the disproportionate levels of mammalian translocations have been justified by the lack of public interest, and therefore ultimately the lack of funding, for small and/or uncharismatic vertebrates, as well as species that have been researched thoroughly (Seddon *et al*. 2005; Martín-López *et al*. 2009; Sitas *et al*. 2009; Fleming & Bateman 2016; Bubac *et al.* 2019). This appears to be the case across the African continent, with the tourism industry of many countries, in particular those heavily reliant on wildlife, largely dependent on the persistence of charismatic mega-fauna. Although not detected in this review, the number of fish, herpetofauna and invertebrate studies in general have increased in more recent years, suggesting a growing awareness of and investment in these taxa in conservation exploration globally (Bubac *et al.* 2019).

2.5.2.2. Translocation Success Across time and Country

Surprisingly, rates of translocation success across the African continent appear to have decreased in the periods examined, although the failure rates across time periods have remained the same. This is likely accounted for by the increase in cases where the outcome of the translocation was unknown. Although monitoring and reporting methods should be improving over time, these findings are consistent with the current literature, as Bubac *et al.* (2019) found no increase in success rates of translocations globally despite the increase in knowledge and technology investment in the field. In exploring investment, this review suggested that GNI/capita and MCI of a country are not good explanatory variables for the number of translocation events reported across Africa. Other likely explanations include the number of tertiary academic institutions within each country, or links to foreign research investment possibly leading to better recording, reporting and publishing of translocations, or the significance of wildlife tourism contributions to economies

2.5.2.3. Translocation Success and Post-Release Monitoring

Encouragingly, the majority of translocations recorded included some form of PRM, reflecting increasingly widespread recognition amongst practitioners (Ewen & Armstrong 2007). Although most studies reviewed included PRM, this did not equate to higher rates of successful translocation events than those that did not employ any PRM. In fact, the success rates for where PRM did and did not occur were roughly equal. The majority of translocation failures occur within the first four years of the event, and it has been suggested that this period may be of critical importance to PRM efforts (Bubac *et al.* 2019). It is acknowledged that longer lived species, such as the elephant, will require significantly longer periods of PRM to evaluate success or failure, than species that mature at a faster rate. As the general measure of success in translocations is the presence of a self-sustaining population (Fischer & Lindenmayer 2000; Seddon *et al.* 2012; Bubac *et al.* 2019), it must be acknowledged that the time scale for the assessment of success is important (Seddon *et al.* 2014). The length of PRM that took place varied greatly, with many of the short PRM terminated by the death of the animal, resulting in the termination of monitoring (Tavecchia *et al.* 2009). These short PRM efforts could explain the relatively low median PRM length of 24 months found in this review. It has been recommended that PRM occurs for a minimum of four years (Bubac *et al.* 2019).

2.5.3. Strategies to Improve Relocations as a Conservation Tool

Translocations have become one of the main tools in the ecologist's toolbox when wanting to restore or reinforce animal populations. The ability to attract large amounts of publicity (Dodd & Seigel 1991), promote conservation education (Rahbek 1993; Wiese *et al.* 1996; Seddon *et al.* 2012), and raise funds (Snyder *et al.* 1996; Wiese *et al.* 1996) make translocations a very appealing option for the conservation of biodiversity. However, the concerning trend of decreasing success rates in more recent times asks many tough questions of the effectiveness of translocations. Fischer and Lindenmayer (2000) identified five broad areas of concerns, and they are re-addressed below, with an additional concern.

2.5.3.1. Concern No. 1: rigorous checking for the appropriateness of the approach

A positive trend identified was the number of studies investigating specific ecological aspects that influenced the success of the translocated organisms (e.g. Linklater *et al.* 2011; Jenkins *et al.* 2015; Briers-Louw *et al.* 2019). The accumulation of such studies in addition to the already large database of broad reviews which provide useful summaries about which factors influence translocation success (Griffith *et al.* 1989; Wolf *et al.* 1996; Dodd & Seigel 1991; Fischer & Lindenmayer 2000; Seddon *et al.* 2012; Bubac *et al.* 2019) should be extensively studied prior to the formation of a translocation event. As HWC is widespread across the African continent, the influence of translocations on the communities surrounding the areas into which the animals will be released should also be evaluated (Cunningham 1996; Nolet & Rosell 1998). Finally, the question of whether there are more effective alternatives to achieve population restoration success given the financial resources available should be contemplated. This is particularly relevant for a continent with many developing countries and limited financial resources.

2.5.3.2. Concern No. 2: clearer definitions of success

Issues with the definition of "success" with regards to translocations have been identified historically (Fischer & Lindenmayer 2000). Ecologically, there are many small steps which, if achieved, could each be considered a success. For example, the adoption of a burrow that had been prepared in a new environment by a translocated tortoise was considered successful (Lohoefener & Lohmeier 1986). However, it has generally been defined as the establishment of a self-sustaining population (Griffith *et al.* 1989; Fischer & Lindenmayer 2000), which can vary greatly between species, and thus the timescale over which success is evaluated is of great importance (Dodd & Seigel 1991; Seddon *et al.* 2014; Bubac *et al.* 2019). The best example of where success was defined appropriately was by Weise *et al.* (2015) where they explicitly stated three clear criteria which must be met in order for the translocation of stock-raiding leopards to be considered a success. The explicit stating of success criteria enables subsequent monitoring programs (see below) to constantly assess the outcomes of the translocation.

2.5.3.3. Concern No. 3: constant monitoring of success

Encouragingly, most studies included some form of PRM. Although the presence of a PRM protocol did not necessarily contribute to higher rates of translocation success, it does allow for the reasons to be identified and documented as to why the translocation may have failed or succeeded. For example, although the elephants translocated to reduce HEC in Kenya ultimately returned to the source site (Pinter-Wollman 2009), the study suggests that for elephants 160 km is not a large enough displacement to negate homing behaviour. The average length of PRM found in the current study was 24 months, although this is likely shorter than the true mean, as many PRM efforts were terminated upon the death of the monitored animal, which was usually caused by some form of anthropogenic behaviour (e.g. roadkill, poisoning, hunting). It has been recommended that PRM occurs for a minimum of four years, as it is during this time period when most translocations were found to fail (Bubac *et al.* 2019). The recommendations made two-decades ago by Fischer & Lindenmayer (2000) are once again made here: anyone responsible for conducting a translocation should design and implement long-term (minimum of four years) monitoring practices, where key ecological parameters, such as the number of animals, sex ratios, adult/juvenile ratios, population change, and a constant re-assessment of the threatening processes are recorded.

2.5.3.4. Concern No. 4: better financial accountability

Only one study that occurred in Africa reported the finances of the translocation event (Boast *et al.* 2015), although a review investigating the effectiveness of global problem-predator translocations provided an average cost incurred (Fontúrbel & Simonetti 2011). Boast *et al.* (2015) provided a breakdown of expenditures, enabling translocation planners to better understand the financial implications of translocations and the subsequent monitoring of stock-raiding cheetahs. Given the limited financial resources available for conservation, it is recommended that all future translocation programmes report their finances in the manner that Boast *et al* (2015) did, as it would be useful for the planning and fund-raising steps of other programmes. Additionally, it will aid in the determination of whether translocations are a cost-effective strategy or if alternatives, such as the financial reimbursement of raided livestock, should be considered.

2.5.3.5. Concern No. 5: publication of results

This may be one of the largest concerns for translocations in general, but specifically for those happening on the African continent. Numerous reviews have already stated the geographic bias in the literature, with North America, Europe and Oceania recording the greatest number of translocations (Fischer & Lindenmayer 2000; Bubac *et al.* 2019). The developing status of many countries on the African continent may be responsible for the low number of translocations documented and published. Financial constraints and the relatively low numbers of tertiary educational institutions may be contributing to the lack of African translocations found in the literature. This author is also of the same belief as Fischer and Lindenmayer (2000), who stated that the field of translocation biology needs to make the outcomes of translocations easily and readily available for future planners to learn from previous mistakes and successes. While the publication of translocations in peer-reviewed journals ensures to uphold the quality of science, it does somewhat impede the widespread accessibility that is needed (Stanley Price 1991; Minckley 1995). The development of an open-access database where translocation practitioners can upload the data relevant to the translocation would be the ideal solution. Such data would include, but not be limited to, the number of animals moved, the demographic breakdown of animals moved, country in which the animals were removed from or taken to, the translocation/wildlife movement company used, the cost of the translocation, the date of the translocation, and if the translocation was successful in its specific aims. The monitoring and subsequent determination of success can remain in the published literature as to ensure academic quality. The accumulation of such basic data would eventually allow for general trends to be highlighted, as well as enable wildlife managers and scientists to easily access the historic records of translocations. While this may be the ultimate end-goal, political differences, between countries and/or conservation organizations might hinder the project. However, an institution such as the IUCN who is familiar with wide-spread collaboration (both geographically and politically) would be an ideal partner to launch this initiative.

2.5.3.6. Concern No. 6: The monitoring of source populations

No studies that qualified for this review investigated the effects of a translocation event on the source population. This is of major concern: If we continue to rely on source populations for translocation events, yet cause deleterious effects to the source populations, the aim is pointless. Only one study which investigated the effects of translocations on source populations was found in a general Google Scholar search (Dimond & Armstrong 2007) (i.e. the search was not restricted to the African continent). The latter authors investigated the response of a source population of North Island Robins (*Petroica longipes*) to the removal of individuals and used these data to develop an adaptive harvesting model. The population was monitored after each translocation and the model was continuously updated. It is recommended that all source populations are monitored in a similar way. Changes in population structure and habitat use may indicate the large-scale, long-term effects that translocations may have on source populations.

2.5.4. Conclusion

Despite the popularity and vast potential of translocations, many are carried out in an ad hoc manner and are either not documented and/or published or monitored appropriately. Extensive pre-release assessments are required to ensure that each translocation has the best odds of succeeding. Such assessments include both a thorough review of all current literature regarding the translocation of the intended species to be moved, but also detailed demographic information and examination of potential release sites. Most importantly, the key factors that contributed to the initial cause of decline must be addressed. The definition of "success" must be tailored to each study and be explicitly stated prior to release. Post-release monitoring should be conducted for a minimum of four years, and the monitoring of source populations should not be neglected. Finally, the recording and reporting of translocation events should be made simple. An open-access database would provide scientists and wildlife managers the opportunity to easily record and review translocation events. Despite the potential political challenges in establishing data sharing on such a scale, the benefits would be well worth the investment for conservation practitioners across Africa.

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Chapter Three

The post translocation demographics of the African elephant (*Loxodonta africana*) in Majete Wildlife reserve, Malawi.

3.1. Abstract

As protected areas become increasing isolated, effective management strategies that maintain population viability are required. Translocations have been and continue to be a favoured strategy among many conservationists. While the impacts of translocations have been extensively documented on the moved group of animals, the source population is often neglected. Here, the demographics of the elephant population of Majete Wildlife Reserve, Malawi, were examined two years after 154 individuals were removed from the reserve. Adult elephants were found to make up the largest proportion of the population (64%), and an extreme adult-male bias was found in the sex ratio (5:2). This adult-male bias was not surprising, as the translocation intentionally only removed adult-females and dependent offspring. The average annual growth rate over the two-year period (2017 – 2019) was 7%, but this is likely due to conceptions that occurred pre-translocation and can be expected to decrease due to the adult-male bias found. Two-years post translocation appears to be too early to draw meaningful conclusions from the event's effects on the demographics of an elephant population, but it gives a good estimate of remnant population structure from which to base ongoing monitoring effort. It is recommended that every second year, and if possible annually, a detailed demographic survey such as the one carried out in this study, be conducted on Majete's elephant population to better understand the impacts of the 2017 translocation on the source population demographics.

3.2. Introduction

Detailed demographic data for wildlife species with high economic- and conservation-value are often sparse (Gaillard *et al.* 1998) and may be heavily biased towards single-population studies (Gaillard *et al.* 2000). Ultimately limited understanding of life history strategies across species and ecological systems limits our ability to evaluate demographic responses to anthropogenic pressures and ecological changes (Stockwell *et al.* 2003; Owen-Smith *et al.* 2005; Wittemyer *et al.* 2013).

The African elephant (*Loxodonta africana*) has been better studied than most other wildlife species, with individual based monitoring techniques or culling data generally providing the information used to determine elephant demographic variables (Moss *et al.* 2011; Wittemyer *et al.* 2013). This is no coincidence as the African elephant is a keystone species (Power *et al.* 1996) thus playing a significant role in the diversity of the habitat it occupies (Blake *et al.* 2009; Wittemyer *et al.* 2013). Elephants can induce dramatic changes to their surrounding environments with these impacts determined by several factors that include location, available resources, the confinement of a population and population density (Skarpe *et al*. 2004; Kerley & Landman 2006; Chamaille-Jammes *et al.* 2008; Loarie *et al.* 2009). It is therefore no understatement that the state of elephant populations is vital to the health and integrity of the ecosystem in which they exist (Laws

1970; Western & Maitumo 2004; Kerley & Landman 2006; Young *et al.* 2009). Additionally, a high economic value is associated with African elephants as they are a conservation flagship species and highly valued by the tourism industry (Douglas-Hamilton 1987; Owen-Smith *et al.* 2006).

Unfortunately, this flagship status has not ensured a ubiquitous conservation status for elephants across Africa, with complete population eradications in certain areas (Blanc *et al.* 2007). These collapses have further unbalanced regional and continental distributions so that a range state like Botswana now houses more than a quarter of the continent's elephants (Bouche *et al.* 2011). Most demographic studies have been conducted in protected areas with stable or increasing populations (Wittemyer *et al.* 2013). While these data provide important baselines, these data no longer reflect many of African elephant populations (Blanc *et al.* 2007; Bouche *et al.* 2011; Maisels *et al.* 2013), and comparative data on populations experiencing a wide array of pressures is sorely needed to determine differences in population status and response to human influences (Wittemyer *et al.* 2013).

Translocation events are a considerable anthropogenic influence exerted on wildlife populations (Chapter 2). Translocations are the movement of living organisms from one area, with release in another, via human-mediated means. Conservation Translocations are more specifically defined by the IUCN as "the intentional movement and release of a living organism where the primary objective is a conservation benefit: this will usually comprise improving the conservation status of the focal species locally or globally, and/or restoring natural ecosystem functions or processes." (IUCN, 2013). Conservation Translocations often take one of two forms: 1) Population Restoration, where the organism in transport is released within its indigenous range, or 2) Conservation Introduction, where the organism in transport is released outside of its indigenous range (IUCN, 2013). Population Restorations can further be divided into the following two activities: 1) Reinforcement, which is defined as the intentional movement and release of an organism into an existing population of conspecifics. Reinforcements generally aim to enhance the viability of a population by increasing genetic diversity, population size, or increasing specific demographic groups (e.g. females, males) or stages (e.g. adults, juveniles) (IUCN, 2013), 2) Reintroduction is the intentional movement and release of an organism inside its indigenous range from which it has disappeared. Reintroduction events aim to re-establish a viable population of the focal species (IUCN, 2013).

Detailed information on how sociality translates into fitness consequences and the role of standard social structure in arbitrating these effects are not well-known (Silk 2007; Silk *et al.* 2010). This is paramount in cognitively developed social mammals, such as the African elephant, where intricate social relationships in a complex social system develop over long life spans (McComb *et al.* 2011), and where cultural transmission of knowledge between generations occurs (McComb *et al.* 2001; McAuliffe & Whitehead 2005; McComb *et al.* 2011). Human activities, such as hunting/culling, translocations and habitat fragmentation are profoundly affecting many free-ranging populations of these highly social mammals (Caswell *et al.* 1999; Campbell *et al.* 2008; Bouché *et al.* 2011), and these activities impact directly on the species' social structure (Shannon *et al.* 2013). The frequency of translocation events for elephants has increased dramatically in recent years, likely due to modern public dissatisfaction with the historic practice of culling to remove large numbers of individuals, and as wildlife management authority investments in the skills and infrastructure (particularly vehicles) to achieve elephant translocations. Studies have demonstrated significant long-term effects of anthropogenic disruptions on both physiological stress levels and broad behavioural patterns (Gobush & Wasser 2008; Gobush *et al.* 2008; Jachowski *et al.* 2012; Tingvold *et al.* 2013). Anthropogenically disturbed elephant populations suffer impairment in key decision-making abilities, failing to distinguish between callers on the basis of social familiarity, despite the disturbance occurring decades before the study (Shannon *et al.* 2013), creating inter-generational effects. Yet systematic work to generate in-depth understanding of how these "anthropogenic disturbances" affect pivotal aspects of social function is lacking (Shannon *et al.* 2013).

Disrupted populations typically experience two specific effects that may impact on social function; 1) immediate trauma of the disruptive event, and 2) the loss of opportunities for interacting with older, often more experienced members of a group that act as appropriate role models or sources of knowledge (Slotow *et al.* 2000; McComb *et al.* 2001; McAuliffe & Whitehead 2005; Greve *et al.* 2009; McComb *et al.* 2011). Social trauma experienced early in life may result in adverse consequences and have profound effects on physiological development and adult behaviour patterns (Bradshaw *et al.* 2005; Lupien *et al.* 2009; Lukas *et al.* 2011). Early social trauma may be compounded by a lack of access to appropriate role models. For elephants older individuals play pivotal leadership roles and manage decision-making in the context of both social and ecological threats (McComb *et al.* 2001; McAuliffe *et al.* 2005; McComb *et al.* 2011). In the absence of these older, experienced individuals, younger group members may be presented with fewer opportunities to learn the most apt response in dangerous situations (McComb *et al.* 2001; McComb *et al.* 2011; Thorton & Clutton-Brock 2011; Van S Chaik & Burkart 2011). Furthermore, there is the potential for any atypical behavioural patterns that have arisen from socially disruptive events to be transmitted between the generations, ultimately persisting in the long term (Shannon *et al.* 2013).

A key case study with regards to the consequences of elephant disruption is that of the infamous young bulls in Pilansberg National Park (PNP), South Africa. The young males were orphaned due to culling practices in Kruger National Park, South Africa and translocated to PNP, where later in life, they displayed levels of hyper-aggression, which resulted in them killing 107 rhinoceroses over a 10-year period (Slotow *et al.* 2000; Slotow & Van Dyk 2001; Bradshaw & Schore 2007). While extreme, and well-publicised, other populations also show considerable impacts on individual immunity, reproductive success and cognitive success (Gobush & Wasser 2008; Shannon *et al.* 2013). Importantly, it has been both predicted (Bradshaw & Schore 2007; Lupien *et al.* 2009; Lukas *et al.* 2011) and demonstrated (Shannon *et al.* 2013) that such traumatic events have more subtle effects on learning, in particular interfering with the capacity to appropriately gauge adequate responses to both social and environmental stimuli. Millspaugh *et al.* (2006) demonstrated that translocation events were an immediate, acute stressor on individual elephants. Management may play a role in minimizing external stressors during translocations and avoiding additional stressors post-translocation, as to avoid potentially stressing the elephant to a level where it would incur a biological cost of some sort (e.g. impaired immune system), thus harming the animal (Moberg 2000). Additionally, the stress experienced by the translocated animals of being left in an unfamiliar environment with unfamiliar conspecifics should warrant investigation. This is crucial when regarding elephants, who's learning of social and ecological knowledge occurs over decades (McComb *et al.* 2001; McComb *et al.* 2011). Traditionally, sink populations are often monitored closely and the effects of translocation events on animal welfare studied intensely (Skarpe *et al.* 2004; Millspaugh *et al.* 2006; IUCN, 2013; Briers-Louw 2019). Meanwhile, the source populations for translocations rarely receive any attention.

Elephant populations may suffer stress if the removal of key individuals disrupts well-defined social hierarchies and relationships (Gobush *et al.* 2008; Silk *et al.* 2010; IUCN, 2013). Source population demographics are rarely studied post-translocation, despite the opportunity they provide to fill a knowledge gap of demographic responses to anthropogenic influences. Very few detailed elephant studies have been able to persist long-term (except for Samburu National Park: Douglas-Hamilton 1987; Addo Elephant National Park: Whitehouse & Hall-Martin 2000; Amboseli National Park: Moss 2001; Moss *et al.* 2011), but have provided source data and demonstrated the usefulness of the mark-recapture methods as a management tool to determine the demographic structure of both confined and free-ranging elephant populations (Whitehouse & Hall-Martin 2000; Morley & van Aarde 2007; Trimble *et al.* 2009). What is missing is more information about other elephant populations, under different ecological stressors.

Elephants were first reintroduced into MWR in 2006, and by 2010, a total of 213 individual elephants had been reintroduced from various parks in South Africa (pers. comms African Parks). Before 2016, only aerial counts had been used to survey the elephant population status and these suggested the population was increasing. Only in 2016 was the population's first detailed demographic survey conducted and it was confirmed that the population had increased dramatically, with a total of 380 elephants (Forrer 2017). In 2017, 154 elephants from Majete Wildlife Reserve [MWR] and 370 elephants from Liwonde National Park [LNP]) were reintroduced to Nkhotakota Wildlife Reserve (NWR) [all within Malawi] in a major translocation exercise, after poaching resulted in the extirpation of elephants in NWR. This mammoth move has relieved ecological pressures both within MWR and LNP and hopefully enables the elephant population within NWR to grow and stabilize. The removal of 154 elephants provided an opportunity to reassess the demographics of the elephant population within MWR. Although population sizes are useful, their explanatory powers are not limited because age structure plays a pivotal role in understanding how elephant populations are faring. For example, calf survival rates provide key insights to population recruitment levels. The current study investigated the population demographic structure of MWR's elephant population, one-year post translocation event, in an attempt to better understand how source elephant populations respond to large scale translocation events. The study aimed to identify if there was a possibility of serious medium to long-term effects of translocation events on elephant source populations, using individual identification techniques to determine population size, age and sex structure. Current demographic parameters (population size, sex ratio and age structure) were compared to those from initial reintroductions and the first detailed survey by Forrer (2017).

3.3. Methods

3.3.1 Study Area

The study took place in MWR, which is located at the southern tip of the Great Rift Valley in the lower Shire Valley region of southern Malawi (S15° 54’26.6”; E034°44’24.3”) (Figure 3.1). The reserve is 700 km^2 in size with two perennial rivers, the *Mkulumadzi* and the *Shire*, defining its northern and eastern boundaries, respectively. The altitude within the reserve varies greatly, with the western region containing steeply undulating hills disrupted by river valleys with the terrain flattening towards the Shire River. Two well defined seasons occur in Majete; the wet season (December to May) and the dry season (June to November). Annual precipitation varies within the reserve, with the eastern lowlands and the western highlands receive 680-800 mm and 700-1000 mm, respectively (Wienand 2013). Whilst water availability is dependent on seasonal rainfall, there are 10-artificial borehole-fed waterholes, as well as several naturally occurring perennial springs. Majete Wildlife Reserve is primarily dominated by multi-altitude woodlands, but also contains grassland areas. The four general classifications of vegetation type found in MWR are: 1) savanna (*Combretum* species, *Vachellia* species and *Panicum* species); 2) low altitude (205-280 m) mixed woodland (*Vachellia* species and *Steculia*); 3) medium altitude (230-410 m) mixed woodland (*Brachystegia boehmii*, Diiospyrus *kirkii* and *Combretum* species); and 4) high altitude (410-770 m) miombo woodland (*Brachystegia boehmii, Burkea africana* and *Pterocarpus*) (African Parks 2017).

In 1955, MWR was gazetted but poaching and poor management led to most of its large game being decimated by 2003 (Forrer 2017). As a result, a Public Private Partnership (PPP) agreement was made between African Parks, Majete (Pty) Ltd. and the Malawian Department of National Parks and Wildlife (DNPW) to initiate one of Africa’s greatest reintroduction programmes. Over 2550 individual animals, comprising of 14 different species, were reintroduced into MWR. Elephant (*Loxodonta africana*), black rhino (*Diceros bicornis*), buffalo (*Syncerus caffer*), sable (*Hippotragus niger*), hartebeest (*Alcelaphus buselaphus*), several other antelope species, as well as many predators were all included in the reintroduction programme.

Thanks to the good management techniques and sustainable strategies now in place at MWR, the successfully reintroduced wildlife flourished. Majete Wildlife Reserve’s elephant population was no exception. The dramatic increase in elephant population size led to a heavy impact on the vegetation within MWR (Staub *et al.* 2013; Weinand 2013; Forrer 2017). Due to this, and the recent acquisition of NWR which led to it being deemed a suitable sink location, during the months of June and July of 2017, 154 elephants were translocated from MWR to NWR, with the relative demographic information of many of the moved elephants recorded to the best of management’s ability.

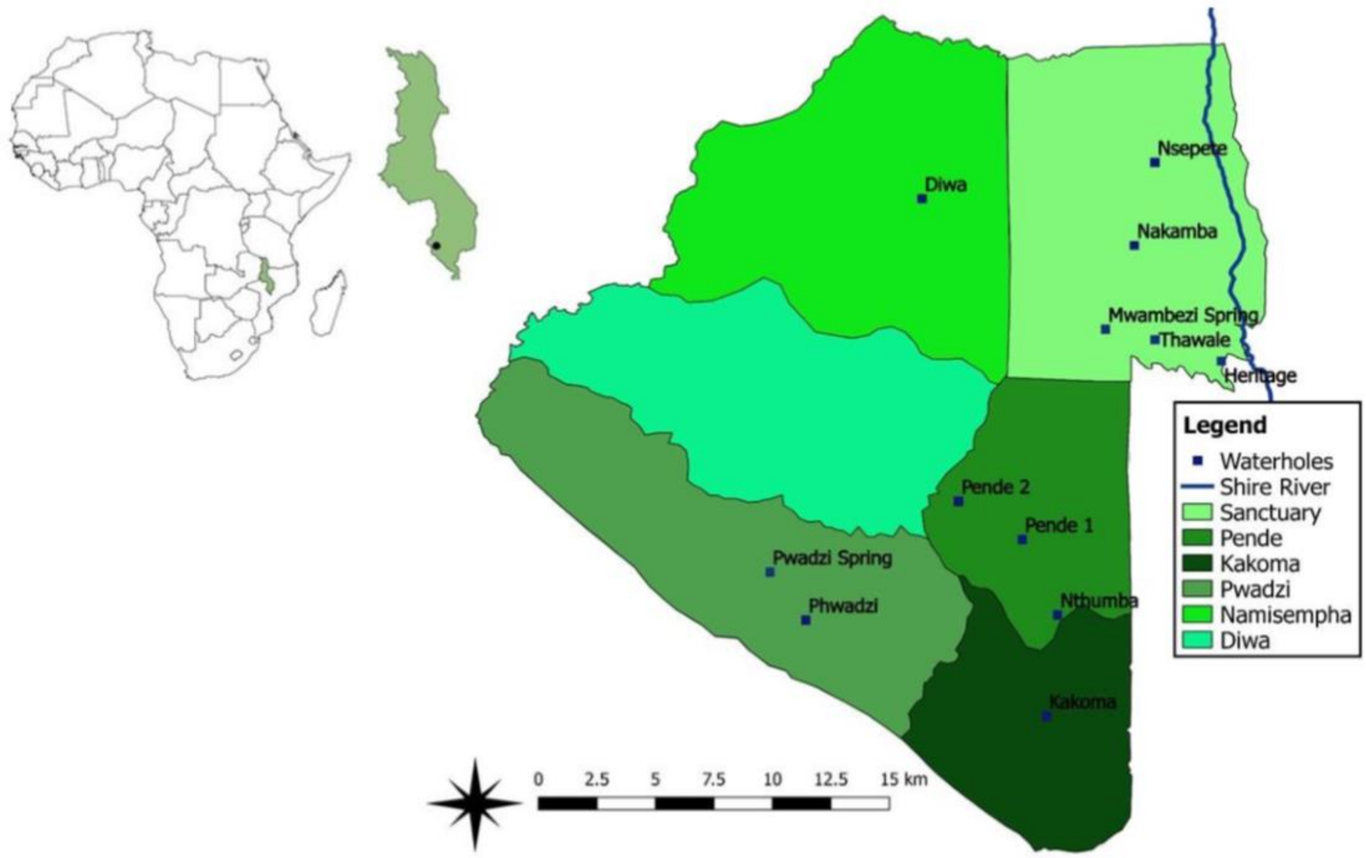

Figure 3.1. The location of Malawi on the African continent (green) and the position of Majete Wildlife Reserve within Malawi. The differing regions and locations of the perennial water sources that occur within the reserve are also shown. (Shapefiles per comms. African Parks (Pty) Ltd.)

3.3.2. Methods

The following methods largely followed those conducted by Forrer (2017) as this allowed for the best means of comparison between the two studies.

3.3.2.1. Defining the population size, age and sex structure

The majority of individual elephants had to be identified in order to determine the current size, age structure and sex ratio of the elephant population found within MWR. Techniques used to estimate the population size or densities of large mammals living in wooded areas are generally poorly developed, and methods such as aerial surveys and faecal counts are not appropriate for a reserve, such as MWR, where the majority of the vegetation is woodland (Caro 1999; Walsh *et al.* 2001; Whitehouse *et al.* 2001; Payne *et al.* 2003). Aerial survey methods often underestimate the number of individuals in relatively small populations (i.e. below 250 individuals) residing in dense habitats, and this error increases with an increasing population size (Whitehouse *et al.* 2001, Morely & van Aarde 2007). Therefore, mark-recapture methods were employed to study the demographics of the Majete elephant population through on-the-ground visuals and via the use of remote camera traps.

Methods of identification followed those outlined by Forrer (2017) and were used to identify and record an elephant's sex, age and other unique characteristics. Two primary methods were used to sample the MWR elephant population. The first was field observations which comprised of randomized distance sampling and waterhole counts. The second was the use of camera traps, which recorded images of individuals and herds, where cameras were placed along the road network in grids, as well as at various watering holes and springs. Field observations began in April 2018 and ended in April 2019. Waterhole and perennial spring camera traps form part of a larger, continuous monitoring program of MWR. Thus, these images are captured year-round. However, analysis of these images was conducted from January 2018 through April 2019. Camera traps placed along the grid network were recorded from July 2018 through April 2019.

a) **Individual Identification Techniques**

Elephants were photographed using a Nikon D7000 whenever observed in the field, and the total number of individuals and GPS location was marked for each encounter. Individuals were 'marked' by recording various characteristics and unique markings. Characteristics could include: age, sex, body size, ear shape, any notches or holes present in the ears, patterns of blood vessels in the ears, wrinkles on the face, tusk size and configuration, lumps or scars on the body and kinks or baldness of the tail. Elephants were then later positively identified as 'marked' or 'unmarked' using the database developed by Forrer (2017). When an unknown elephant was encountered, it was given an identification code, sexed and aged, and any unique characteristics recorded, after which it was then added to the 'known' population database.

Elephants grow throughout their lifetime, and individuals were aged according to body size using the following categories (Moss 1996):

Infant (0-0.9 years):	Infant's shoulder is taller than the breast level of the mother, reaching wrinkles above elbow.
Calf (1-4.9 years):	Top of calf's shoulder is above the mother's armpit, back is level with anal flap and reaches the lower quarter of the mother's ear. Tusks are 5-7 cm in length.
Juvenile (5-9.9 years):	The overall size of the elephant was about three quarters of an adult female. Tusks splayed and are 25-30 cm in length.
Small Adult Female (10-19.9 years):	The elephant is the same size as other adult females. The tusks now have an adult configuration (convergent, straight or asymmetrical). The elephant's body is square in shape compared to older females who are more rectangular.
Small Adult Males (10-19.9 years):	The elephant is the same size or slightly taller than an adult female. The head shape (sloping rather than angular) is more pronounced. The tusk circumference is thicker than that of females of similar age.

Medium Adult Female (20-34.9 years):	The elephant is taller than all adult females, and the head has begun to thicken (changing into an hourglass shape; wide at the eyes and at the base of the tusks).
Large Adult Female (>35 years):	The tusks are marginally thicker than those of a medium adult female. The elephant's back is lengthened, making the animal appear long. Older females are typically hollowed above the eyes and their ears are held lower.
Large Adult Males (>35 years):	The elephant is very big, often towering over the largest females. The overall body is heavy set with a thick neck. The circumference at the base of the tusks is greater than in younger males.

b) **Distance Sampling**

Drive transects were conducted along established roads throughout MWR. Due to the nature of the road network and logistical difficulties, sampling was conducted weekly within the area known as the Sanctuary and occurred monthly in all other areas. Sampling begun just before dawn and continued until after sunset. A scheduled break was taken during the hottest part of the day (~ 13h00-15h00) when elephants also rest. Majete Wildlife Reserve consists of dense vegetation, as a result, elephant visibility is generally less than 150 m, so a cruising speed of approximately 15 kilometers per hour was maintained when searching for elephants. Upon the initial sighting of an elephant with the naked eye and/or binoculars, the vehicle was stopped, and an observation period commenced; date, time of observation, GPS position of observer, total number of elephants seen, and number of males and females and the respective age classes of individuals were recorded. A "Quality of Count" index was used to exclude problematic observations whilst still maximizing the data used. Each observation was noted as being "Perfect", "Good Count", or "Ball Park". Such an index is useful in situations where elephant herds are large and/or are heading in a direction which the researcher cannot follow. Later, images were sorted and added to the database explained above.

c) **Waterhole Counts**

Ten borehole-fed artificial water points are found within MWR. Counts were conducted at four artificial water points in the Sanctuary region (Nsepete, Nakamba, Thawale and the Heritage) and at two artificial water points in the Pende region (Pende 1 and Ntumba), with counts being conducted from June 2018 to December 2018 (See Figure 1 for the location of each area within MWR). Each count lasted a period of 12 hours, starting between 05h00 and 06h00. In the Sanctuary, an additional two or three observers were stationed on a viewing platform or in an elevated hide. During such counts, the data collected included: weather conditions (cloud cover, temperature), total number of elephants sighted, time of observation of individual or herd, the sex and ages of each individual and several other behavioral observations (both intraspecific and interspecific interactions). Elephants were photographed and identified or added to the database as appropriate.

d) **Camera Trapping**

Two Cuddeback camera trap models were used: The X-Change Colour camera Model 1279 (CUDDEBACK, Wisconsin, USA) and the Professional Color Model 1347 (CUDDEBACK, Wisconsin, USA) were used between the years 2018 and 2019. Cameras were stationed at all 10 of the artificial water points in MWR as well as at two of the perennial springs (See Figure 3.1 for locations) and programmed to take one photo upon being triggered with a delay of 60 seconds between trigger events, and each camera was secured approximately 60 cm above the ground facing towards the artificial waterpoint. Cameras were serviced on average every 2-3 weeks which included changing of the SD cards as well as battery checks and replacements if needed. Elephant images captured via the camera traps were subjected to the standard identifying and sorting process.

e) **Aerial Census**

African Parks, Majete (Pty) Ltd. conducts aerial surveys approximately every two years for the purpose of coarse animal population estimates. Counts were conducted in 2010, 2012 and 2015, and the procedure for the count conducted in this study followed the standard methods for the area by African Parks; the aerial count was conducted in the late dry season (Oct 2018) and each count consisted of a pilot and 1-2 observers and lasted for three days. Counts were flown in transects with the calibrated strip width of each transect set at 500 m and the flight path oriented in an east to west direction. When an animal was sighted, the data recorded included GPS track log of all transects, game species and number of individuals, and any significant waypoints were also marked using a GPS. Aerial Census data were used simply as a rough estimate for population size, as concerns with this methodology has been outlined above.

3.3.2.2. Comparing current and pre-translocation population demographics: Statistical Analysis

As this study compared its findings to those of Forrer (2017) most of the statistics are descriptive in nature. Data captures were documented in MS Excel (MS Office, version 1910 [Build 12130.20272]) and STATISTICA (version 13. http://statistica.io, Dell), was used to analyze the data using standard descriptive statistics. Relationships between nominal variables were investigated with contingency tables and appropriate chi-square tests (likelihood ratio or Pearson chi-square). A p-value of $p < 0.05$ represented statistical significance in hypothesis testing. 95% confidence intervals were used to describe the estimation of unknown parameters.

3.4. Results

3.4.1. Current population size, age and sex structure

The 2018 aerial census recorded a total of 210 individual elephants throughout MWR and the current study identified a total of 232 individuals from January 2018 to May 2019 (Table 3.1); 35 different family herds and 102 solitary bulls (Appendix 1). The age structure of the population was primarily adult (64%: Table 3.1.), with slightly uneven numbers of juveniles in different age classes, as expected given elephant inter-birth intervals. Most adults were medium-sized (Figure 3.3). Sex ratios for elephants under the age of 10 years (i.e. infants, calves, juveniles) were relatively close to 1:1, although gender was not determined for 27 (31.0%) of the individuals. The sex ratio of adult elephants (between the years of 10-60) within MWR was heavily skewed towards males, with a ratio of 5:2 (Table 3.1; Figure 3.2).

Table 3.1. Age and sex structure of the elephant population in MWR in 2018 and early 2019 (post-translocation). The sex ratios and the proportion of each age class in the population are also shown. Sex for 27 infants, calves and juveniles was not determined.

Age Class		Total	Males	Females	Unknown Sex	Sex Ratio	Age Class Proportion of Population (%)
Infant	(<1 year)	17	3	3	11		7.2
Calf	(1-4.9 years)	43	15	13	15	29:31:27	18.2
Juvenile	(5-9.9 years)	27	11	15	1		11.4
Small Adult	(10-19.9 years)	54	42	12			229
Medium Adult	(20-34.9 years)	63	39	24		5:2	26.7
Large Adult	(>35 years)	32	19	13			13.6
Total		**236**	**129**	**80**	**27**		**100.0%**

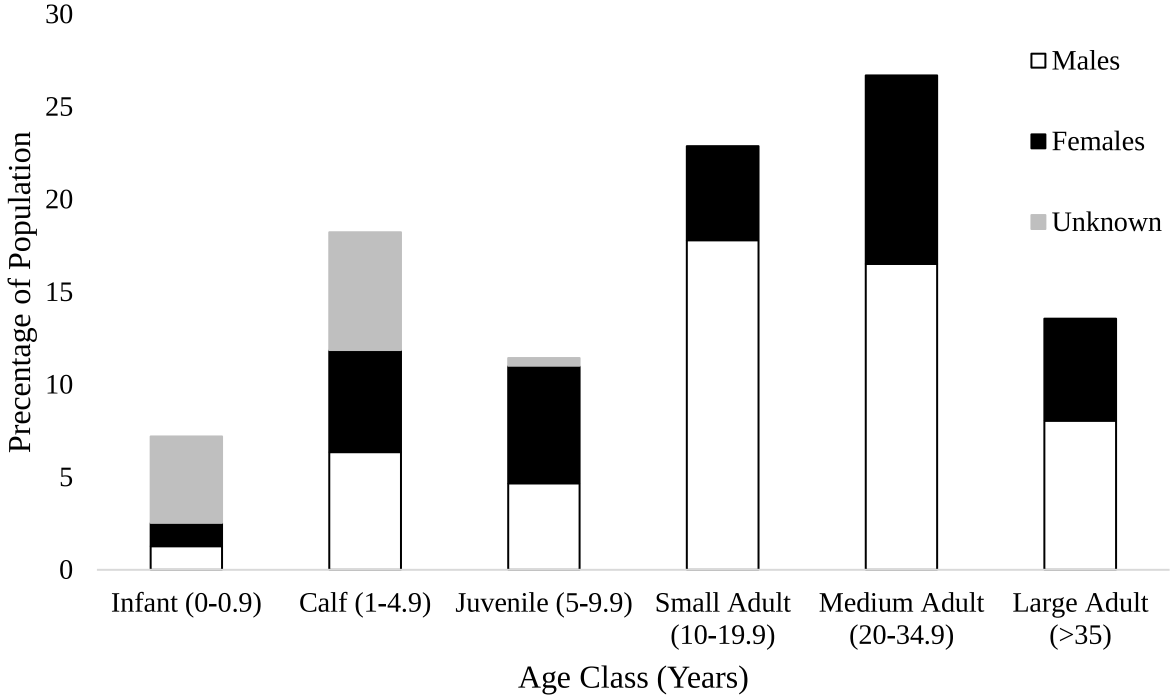

Figure 3.2. The proportional representation of each age class, broken down by sex, in the elephant population in MWR, Malawi, in the years 2018/2019.

3.4.2. Demographic changes pre- and post-translocation

After the initial reintroduction of elephants, the population increased substantially from around 65 animals to an estimated total of 389 individuals (Figure 3.3, Forrer 2017) and experienced minimal mortality, resulting in an annual population growth rate of 13.8% (Forrer 2017). During the time between the initial introduction (2006) and the last in-depth demographic survey (2016) a significant increase in the number of adults was recorded (males $p=0.04$, $r^2=0.8$; females $p=0.01$, $r^2=0.9$) was recorded (Forrer 2017). Unfortunately, no demographic data of the younger age groups (i.e. infants, calves and juveniles) was recorded during the 2006 and 2008 reintroductions.

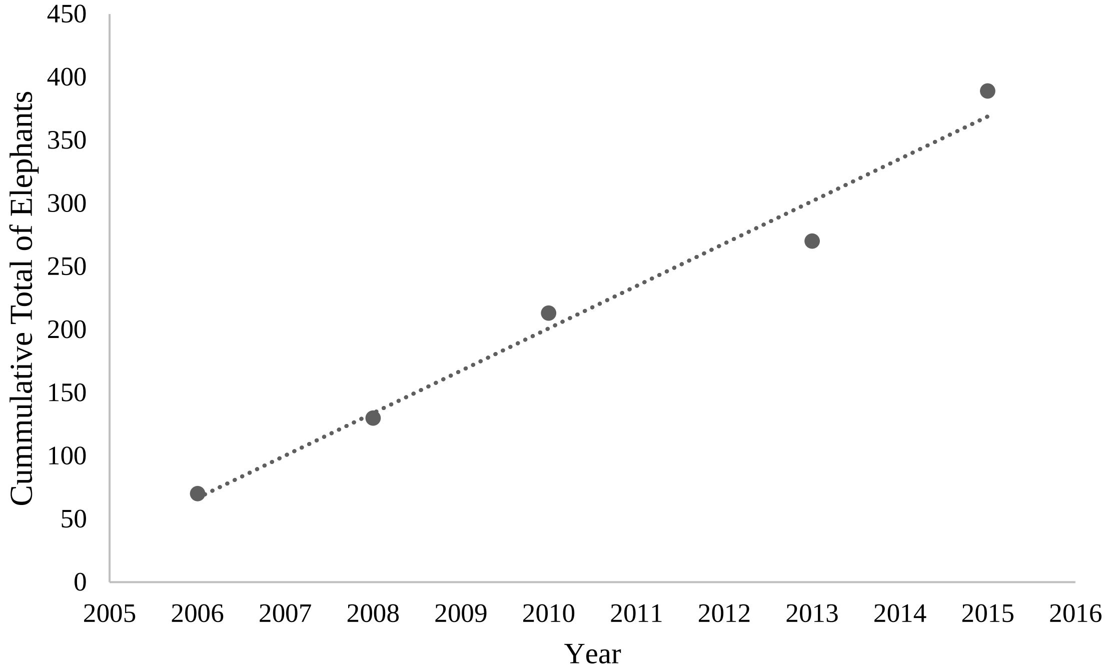

Figure 3.3. The cumulative total number of elephants in MWR from the start of the reintroduction process in 2006 until 2016 ($p=0.00007$, $r^2=0.9971$) (Forrer 2017).

The 2017 translocation event removed a total of 154 elephants, which should have resulted in there being around 235 individual elephants directly thereafter. However, several cases of double counting (when one elephant is identified and recorded as two separate individuals) were discovered by the author of this study, which resulted in an overestimation of the total population. It is more likely that there were around 360 elephants at the time of the translocation event, resulting in the remnant population comprising of approximately 200 individuals (Figure 3.4). The adjusted annual growth rate for the MWR elephant population between the conclusion of reintroductions (2010) and the 2015 aerial census is 11%.The current study estimates a total of 236 individual elephants (Table 3.1), representing an annual growth rate percentage of 7% since the translocation event.

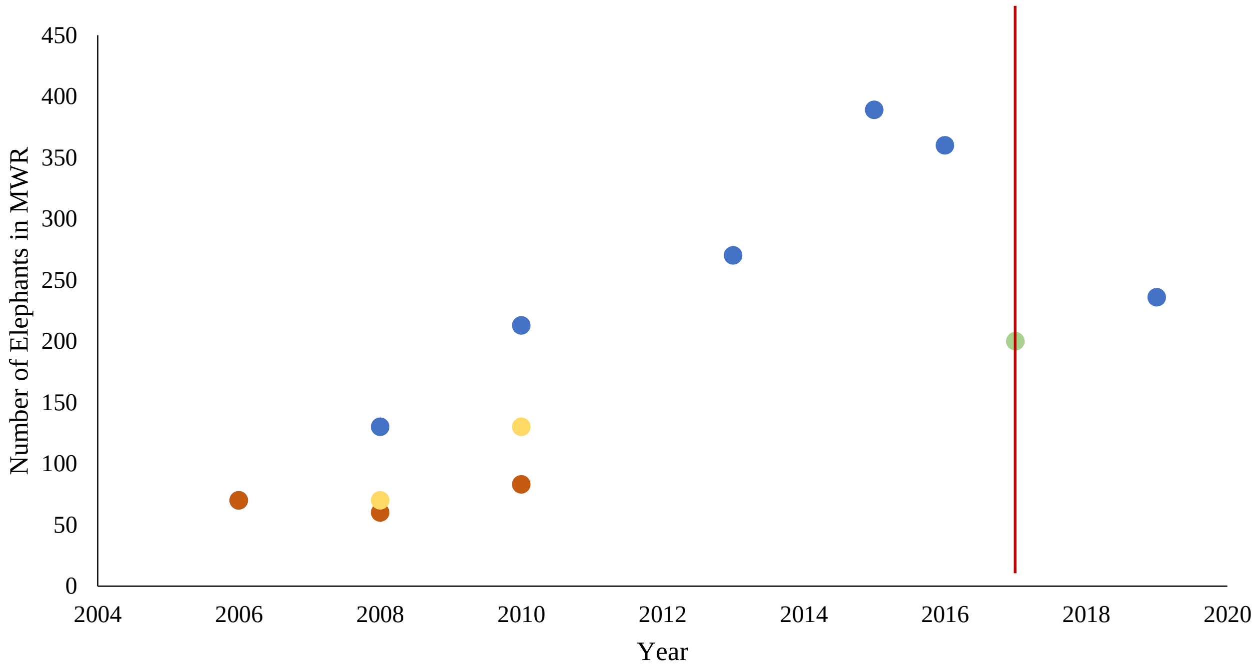

Figure 3.4. The cumulative total number of elephants in MWR since the start of the reintroduction process in 2006 until 2019 (blue dots). The 2016 population number has been corrected for cases of double counting. Orange dots represent the number of elephants introduced into MWR, and yellow dots represent the difference between the total number of elephants (blue dots) and the number of elephants reintroduced (orange dots). The redline represents the translocation event that occurred in 2017, and the green dot symbolises the total number of elephants remaining in the source population.

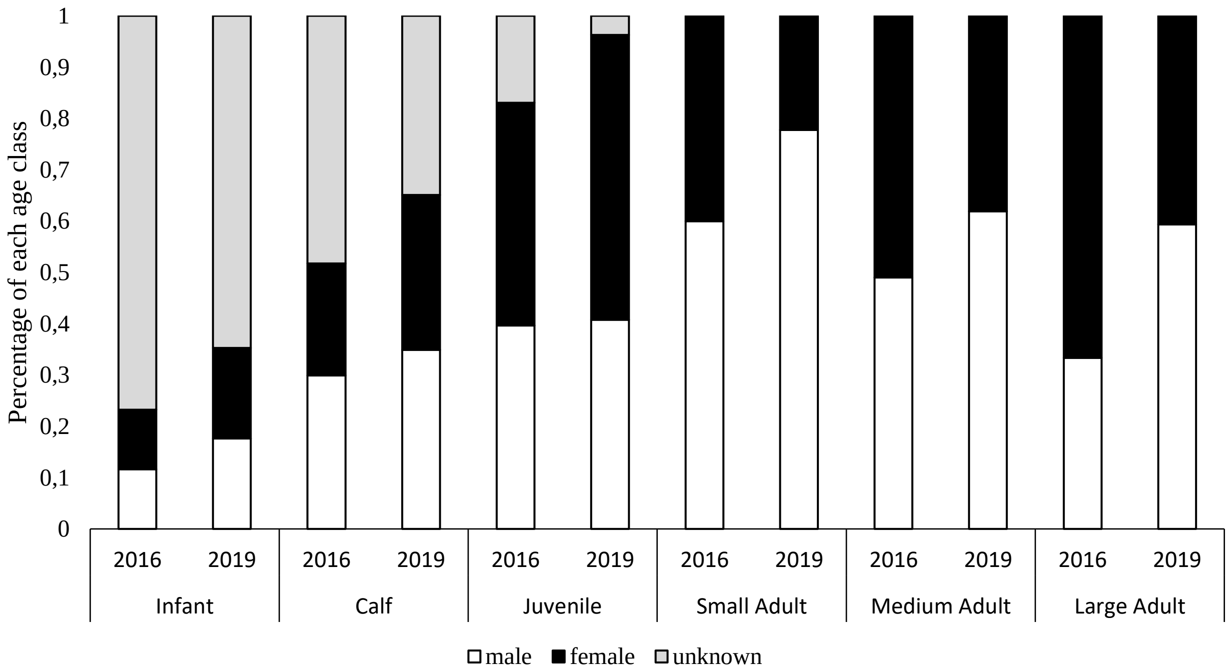

Figure 3.5. A comparison of the relative percentages of the age and sex structure of the elephant population in MWR between 2016 and 2019.

Figure 3.5 provides a comparison of the MWR elephant population's demographics a year before (2016) and two years after (2019) the translocation event. The extreme adult male-bias can be clearly seen, with a ratio of five adult males to two adult females.

3.5 Discussion

3.5.1 Population status, growth rate and effects of translocation

The 2018 aerial count conducted in MWR counted a total of 210 individual elephants whereas the current study was able to positively identify 232 animals. The likely reason for under-reporting by the aerial count was the unexpected rainfall in the middle of the dry season, resulting in a thicker canopy cover than usual. Aerial surveys often under sample when surveying woodland areas (Caro 1999, Walsh *et al*. 2001, Whitehouse *et al*. 2001, Payne *et al*. 2003), which is the dominant vegetation type within MWR (Weinand 2013).

The adult-male bias found in the sex ratio was expected, as the translocation event of 2017 intentionally only removed adult-females and dependent offspring. This decision was taken as African Parks had already procured enough males from LNP for the restocking of NWR. Under natural conditions, male African elephants experience very intense reproductive competition with other males (Poole 1989a, b; Poole & Moss 1989; Hollister-Smith *et al.* 2007). Since escalated contests can result in death, males spend decades learning complex social and behavioural strategies to manage their competitive risks, largely mediated by the phenomenon of musth. Musth males experience increased levels of testosterone and heightened sexual activity, and musth is an honest signal to contest access to females (Moss 1983; Hall-Martin & van der Walt 1984; Hall-Martin 1987; Poole 1987, 1989a, b, 1999). Escalated aggressive interactions most commonly involve at least one musth male (Hall-Martin 1987; Poole 1989a). While non-musth males are capable of breeding successfully (Poole 1989b), most matings are by musth males over 35 years of age, who are preferred by females as musth is a signal of male quality (Moss 1983; Poole 1989b). Since musth patterns are asymmetrical between males, it allows staggering of competition and allows males to outcompete older, larger males, especially those at the end of their musth period (Poole *et al.* 2011).

Given the complexities of male social dynamics that centre around male-male competition, the severe male bias in the sex ratio caused by the premediated artificial removal of only breeding herds during the 2017 translocation is of serious concern. The extreme lack of access to breeding age females now present in MWR may well intensify interactions between the breeding males. Large males (generally over the age of 35 years) guard females during mid-oestrus to ensure the best opportunities for reproduction, while small to medium males gain access only during early- and late-oestrus (Poole 1989b). The 2017 translocation has drastically altered not only the demographics of MWR's elephant population, but the subsequent social landscape for the remaining elephants. There was no bias in sex ratio between juveniles (elephants < 10 years old), which is consistent with other elephant populations (Whyte *et al.* 1998; Slotow *et al.* 2005; Gough & Kerley 2006). An aging demographic was identified with the largest proportion of elephants belonging to the 'adult' class, once again likely due to the removal of breeding herds only.

Majete's previous detailed demographic survey which concluded in 2016, positively identified 366 individual elephants, and in combination with the aerial count that year suggested that there was likely 420 (pers. comms African Parks) elephants in MWR at the time of the study (Forrer 2017). However, numerous cases of double counting were identified within the previous study, and it is likely there were significantly less elephants than previously thought. The total number of individuals prior to the translocation event of 2017 was probably closer to 360 individuals: This is almost 20% less than what management had thought when planning the details of translocation. As a result, approximately 200 elephants remained immediately post translocation and this has increased to 232 individuals two years later. During this time, in the absence of migration or poaching, the MWR elephant population is increasing at an estimated annual growth rate of 7%. It can be reasonably assumed that MWR's elephant population is operating under 'ideal' conditions, where mortality is extremely low and fecundity approaches the physiological maximum (i.e. where the calving interval is 22 months after the loss of a young calf or a pregnancy) (Calef 1998; Moss 2001). With abundant space and resources, as well as numerous perennial water sources and no poaching, it can be expected that MWR's elephant population will continue to increase.

The rate at which MWR's elephant population increases will be interesting to quantify. Although the translocation resulted in an overall lower elephant density and thus an increase in the size of suitable habitat for the remnant elephants (Pinter-Wollman 2012) and therefore an expected increase in reproductive rate, the extreme adult-male bias created by the removal of only adult-females and independent offspring is expected to slow the population growth rate considerably (Whyte *et al.* 1998). The current estimated annual growth rate of 7% calculated in this study is likely due to conceptions that occurred prior to the translocation, but rapid population increases after an event which dramatically decreased elephant numbers has been documented (Foley & Faust 2010). However, it is assumed that the annual population growth rate will slow considerably due to the sex ratio bias as well as the expected decrease in sexually mature female recruitment as evident by an aging elephant population. This, however, is not a negative outcome, as it allows MWR management additional time to organize and implement a sustainable, long-term plan for the elephant population. Additionally, MWR management should implement the continuous monitoring of its elephant population, as instantaneous counts such as conducted in this study may simply reflect short term fluctuations in reproductive rates which represent seasonal and cohort effects. Assessing juvenile survival will be a crucial factor in determining reliable population recruitment patterns.

3.5.2. Conclusion

Operating under ideal circumstances (i.e. abundant space and resources, numerous perennial water sources and the absence of heavy-poaching) the MWR elephant population has grown at an estimated annual rate of 7% since the removal of 154 individuals in 2017. The estimated population size of the population for 2018/2019 is approximately around 232 individuals. It is predicted that the population will continue to increase, but the rate at which that happens is not clear. Although the suitable habitat size for the remnant elephants has increased, an extreme adult-male bias is present and is a result of the translocation only removing adult-females and dependent offspring. The 2017-translocation has significantly decreased the MWR elephant population and it is desirable for the management of MWR to keep the growth rate and resulting numbers of elephants low.

A contraceptive management plan is an option, as this has been shown to be an effective strategy for the management of relatively small elephant populations (van Aarde & Jackson 2007). However, it is my belief that a contraceptive management strategy further disrupts an already extremely disrupted elephant population. Additionally, there is still plenty of unknown consequences of the use of contraceptives in regulating elephant populations. A much simple, far less invasive and disruptive management strategy would be to limit the number of AWPs located throughout MWR. There are currently 10 perennial AWPs, two well-known perennial springs and two perennial rivers in MWR. Such a large number of permanent water sources has enabled the elephant population to rapidly increase despite the harsh dry seasons the area experiences. The manual regulation, and closure of certain AWPs would limit the reproductive rates of the female elephants within MWR, thus slowing the population's growth rate. Contraceptive protocols could be instigated at a much later stage, should they ever prove necessary, but at present there seems no reason to assume that the sex bias against females, coupled with natural population regulation, could not result in a stable population. Regardless the need for any further intervention will remain uncertain without regular demographic monitoring to assess the state of the population.

Regular monitoring programs should be implemented to better understand not only the long-term effects of translocations on the demographics of source elephant populations, but they will provide valuable data useful to management, such as the average age females first give birth and the average calving intervals of females (Whitehouse & Hall-Martin 2000; Moss 2001; Moss *et al.* 2011) which are highly variable (e.g. 22-114 months in Amboseli, Kenya; Moss *et al.* 2011). Dimond and Armstrong (2007) investigated the response of a source population of North Island robins (*Petrocia longipes*) to translocation events. They were able to build a model which predicted the response of the population to repeated translocations and ensure the source population's demographic integrity remained intact. Such a study ensures that the source population remains viable for future translocations, as well as the general health of the animals in and ecosystem surrounding that source population. The development of such a model should be investigated for the elephant population within MWR. Finally, as MWR is a fenced reserve, long-term monitoring programs should include the recording of elephants' impacts on woody vegetation due to surface water availability, as previously recommended (Weinand 2013; Forrer 2017). Finally, the extreme male bias now present in MWR will undoubtedly change the social dynamics between adult male elephants. With relatively fewer breeding-age females available, intraspecific competition can be expected to intensify, which may have unforeseen negative consequences for the general behaviour of a large number of frustrated adult male elephants. With regular demographic and behavioural monitoring and vegetation impact assessments, management will have most, if not all the information required for a comprehensive, effective elephant management strategy for MWR.

3.6. Acknowledgements

I would like to thank the Earthwatch Institute for funding the Animals of Malawi Research Project, as well as this study. Additionally, I would like to thank all members of the MWR management-, mechanic- and law-enforcement-teams for their assistance with the collection of data. Finally, thank you to my research partners for your assistance in the field, Professor Daan Nel for help with statistics, and my supervisors for assistance with the write-up.

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Chapter Four

Intensity of usage of artificial waterpoints of African elephants (*Loxodonta africana*) in response to seasonal and anthropogenic drivers in Majete Wildlife Reserve, Malawi.

4.1. Abstract

Elephant movement patterns are largely influenced by the availability of surface water. Furthermore, herds segregate spatially based on dominance with more dominant herds having access to more suitable habitats. The anthropogenic-mediated removal of dominant herds from an area may therefore enable herds (likely of a subordinate status) that have been excluded to surrounding areas to now access and utilize these preferable areas. This study demonstrated that the removal of 154 elephants predominantly from a preferred habitat area significantly changed the intensity of use of certain artificial waterpoints. The diversity of herds at different artificial waterpoints did not to change significantly between months before and after the translocation event, which indicates that herds from surrounding areas filled the vacuum created by the removal of individuals predominantly from one region but the water resources available to them were not utilized at the same intensity. Large-scale anthropogenic events such as translocations can influence the distribution of herds throughout a reserve. This is especially true, when the translocation removes animals predominantly out of one area within a reserve. The effects of a highly localized removal of animals are not just restricted to elephant herds, but other taxa would likely have been influenced as well, and further investigation into a broader study is required.

4.2. Introduction

The patchy distribution of resources is a major driver of non-random space use within home ranges for most mammalian species (Harris *et al.* 1990; Grainger *et al.* 2005). However, other factors such as individual age (Cederlund & Sand 1994), sex (Owen-Smith 1988; Stokke 1999; Relyea *et al.* 2000), reproductive status (Bertrand *et al.* 1996), human population density and agriculture (Hoare & Du Toit 1999), and even research methodologies (White & Garrot 1990; Harris *et al.* 1990: Seamen & Powell 1996) have been found to account for differences in both the size and utilization of animal home ranges. Like other species, African elephants (*Loxodonta africana*) are non-randomly distributed across landscapes and habitats (*e.g.* Douglas-Hamilton 1972; Leuthold 1977; Owen-Smith 1988; Western & Lindsay 1984; Grainger *et al.* 2005; Ntumbi *et al.* 2005; van Aarde *et al.* 2008). Resource requirements differ both within and across elephant populations and play a major role in how landscapes are utilised, and some resources have disproportionate effects on elephant space use (White 1994; Grainger *et al.* 2005; Fishlock & Lee 2013).

The availability of surface water is a major determinant in elephant distribution patterns (*e.g.* Smit *et al.* 2007; Ntumbi *et al.* 2005; Harris *et al.* 2008) and understanding the water requirements of elephants is central to understanding their movements in space and time (van Aarde *et al.* 2008). Generally, females and breeding herds have smaller home ranges than those of male elephants (Grainger *et al.* 2005; Galanti *et al.* 2006; Thomas *et al.* 2012) because lactation generates higher water requirements amongst females than males (Gobush *et al.* 2008; Dunkin *et al.* 2013). Elephants with limited water access traverse great distances (Lindeque & Lindeque 1991), whereas those with access to year-round (generally artificial) water have small and stable home ranges (Whyte 2001). Seasonal home range changes also occur due to water availability – greater surface water in wet seasons allows elephants to move away from high-use dry season areas (Stokke & Du Toit 2002; De Beer *et al.* 2006; Chamaillé-Jammes *et al.* 2007; O'Connor *et al.* 2007; Jackson *et al.* 2008). Thus, elephant distribution is spatially and temporally variable and modified by access to water (van Aarde *et al.* 2008).

Understanding the potential ramifications of providing water to elephants is perhaps best understood in protected and/or fenced areas. The elephant-conservation conundrum is an interesting one; this keystone species has been eliminated from much of its recent geographical range (Barnes *et al.* 1999) yet locally successful populations may be considered too numerous (Whyte 2001). The debate as to how individual elephant populations should be managed ensues, and the likelihood of a 'one-solution-fits-all' outcome is low, because the geographic, economic, and political situations differ greatly, and ecological circumstances also vary (Mapaure & Campbell 2002; Baxter & Gertz 2005; Sankaran *et al.* 2008). However, what is universally true for all protected areas is that the provision of 'artificial water' (water made available through boreholes) and the erection of fences, both extremely common practices, have consequences for elephant numbers and behaviour (Whyte 2001; Loarie *et al.* 2009). Loarie *et al.* (2009) found a combination of fences and artificial waterholes caused elephant behaviour in the dry seasons to be increasingly similar to that displayed in the wet season, leading to potential vegetation overexploitation, both in areas previously only accessible in the wet season, and due to local pressure on resources near fences as wet season home ranges shrank and elephants 'bunched'.

Habitat selection occurs to obtain better resources (Western & Lindsay 1984). Notably, elephant herds spatially segregate based on dominance, (i.e. they are sensitive to intra-specific competition) with the more dominant herds having access to prime areas – i.e. those with preferred food sources and more water availability (Wittemyer *et al.* 2005; Wittemyer *et al.* 2007). Subordinate herds are excluded from these 'prime' areas, and generally occupy less-desirable areas, which generally contain fewer elephants (Wittemyer *et al.* 2005; Wittemyer *et al.* 2007; Young *et al.* 2009). As an example, the removal of elephants in the Kruger reduced local densities in some areas, as a result elephants from elsewhere in the park moved into these less occupied areas (van Aarde *et al.* 1999). The artificial removal of dominant herds may therefore allow subordinate herds to occupy 'prime', now less densely occupied habitats upon their removal.

While the removal of dominant herds may enable access to areas subordinate herds were previously excluded from, large-scale disruptions can also result in negative consequences for survivors. Disrupted populations typically experience two specific effects that may impact on social function; 1) immediate trauma of the disruptive event, and 2) the loss of opportunities for interacting with older, often more experienced members of a group that act as appropriate role models or sources of knowledge (McComb *et al.* 2001;

McComb *et al.* 2011). Social trauma experienced early in life have profound effects on physiological development and adult behaviour patterns (Bradshaw *et al.* 2005; Lupien *et al.* 2009; Lukas *et al.* 2011). This early social trauma may be compounded by a lack of access to appropriate role models. For elephants, older individuals play pivotal leadership roles and manage decision-making in the context of both social and ecological threats (McComb *et al.* 2001; McAuliffe *et al.* 2005; McComb *et al.* 2011). In the absence of these older, experienced individuals, younger group members may be presented with fewer opportunities to learn appropriate responses in dangerous situations (McComb *et al.* 2001; McComb *et al.* 2011; Thorton & Clutton-Brock 2011; Van S Chaik & Burkart 2011). There is also the potential for atypical behaviour patterns arising from socially disruptive events to be transmitted between the generations, and ultimately persisting in the long term (Shannon *et al.* 2013). Traditionally, sink populations are often monitored closely and the effects of translocation events on animal welfare studied intensely (Skarpe *et al.* 2004; Millspaugh *et al.* 2006; IUCN, 2013; Briers-Louw 2019). Meanwhile, the source populations for translocations rarely receive any attention. What is missing is more information about how elephant population movements through space and time respond to large-scale human induced stressors.

Translocation events are a considerable anthropogenic influence exerted on wildlife populations. Translocations are the movement of living organisms from one area, with release in another, via human-mediated means. Conservation Translocations are more specifically defined by the IUCN as "the intentional movement and release of a living organism where the primary objective is a conservation benefit: this will usually comprise improving the conservation status of the focal species locally or globally, and/or restoring natural ecosystem functions or processes." (IUCN, 2013).

Elephants were first reintroduced into Majete Wildlife Reserve (MWR), Malawi, in 2006 and by 2010, a total of 213 individual elephants had been reintroduced into MWR. Before 2016, only aerial counts had been used to survey the elephant population status and these suggested the population was increasing. 2016 saw the population's first detailed demographic survey and it was confirmed the population had increased dramatically, with a total of 380 elephants recorded in MWR (Forrer 2017). A major translocation event occurred in Malawi between June and July of 2017, with 154 elephants removed from MWR and reintroduced to Nkhotakota Wildlife Reserve (NWR). Little is known about the effects of such large disturbance events on the social aspects of elephant societies. The removal of 154 elephants from MWR provided an opportunity to assess how source populations respond spatially to translocation events. The 2017 translocation removed elephants primarily from the northern parts of MWR, an area with ample food resources, ease of access to many permanent water sources and extensive road networks, and good habitat for elephants (Harris *et al.* 2008), resulting in temporarily lower elephant densities within these areas.

The following study investigated the use of artificial waterpoints and movement between these waterpoints by Majete's elephant herds in the period 2016 – 2019. The aim was to determine if the 'vacuum' (decreased elephant density) in the high preference region created by the translocation has been filled i.e. are elephants from other areas of the reserve now utilizing the artificial waterpoints in this area. It is hypothesized that elephants from surrounding low preference regions have moved into the high preference area. This was tested by determining the intensity and diversity of usage of artificial waterpoints across three regions of MWR between 2016 and 2019. It is predicted that the intensity and diversity of usage of the high preference region should be the highest pre-translocation due to the suitability of the habitat for elephants. It was also predicted that a decrease in intensity and diversity of usage would be observed directly after translocation within the high preference region, followed by a return to pre-translocation levels. A decrease in the intensity and diversity of usage of the other low preference regions was expected post-translocation as it was predicted herds traditionally from these regions would move to the high preference area. Finally, the frequency of sightings of individual herds were quantified pre- and post-translocation to investigate which herds were moving between which regions.

4.3. Methods

4.3.1. Study Area

The study took place at Majete Wildlife Reserve (MWR), which is located at the southern tip of the Great Rift Valley in the lower Shire Valley region of southern Malawi (S15° 54’26.6”; E034°44’24.3”) (Figure 1). The reserve is 700 km^2 in size with two perennial rivers, the *Mkulumadzi* and the *Shire*, defining its northern and eastern boundaries, respectively. The altitude within the reserve varies greatly, with the western region containing steeply undulating hills disrupted by river valleys with the terrain flattening towards the Shire River. Two well defined seasons occur in Majete; the wet season (December to May) and the dry season (June to November). Annual precipitation varies within the reserve, with the eastern lowlands and the western highlands receive 680-800 mm and 700-1000 mm, respectively (Wienand 2013). Whilst water availability is dependent on seasonal rainfall, there are 10-artificial borehole-fed waterholes, as well as several naturally occurring perennial springs. Majete Wildlife Reserve is primarily dominated by multi-altitude woodlands, but also contains grassland areas. The four general classifications of vegetation type found in MWR are: 1) savanna (*Combretum* species, *Vachellia* species and *Panicum* species); 2) low altitude (205-280 m) mixed woodland (*Vachellia* species and *Steculia*); 3) medium altitude (230-410 m) mixed woodland (*Brachystegia boehmii*, Diiospyrus *kirkii* and *Combretum* species); and 4) high altitude (410-770 m) miombo woodland (*Brachystegia boehmii*, *Burkea africana* and *Pterocarpus*) (African Parks 2017).

In 1955, MWR was gazetted, but poaching and poor management lead to most of its large game being decimated by 2003 (Forrer 2017). As a result, a Public Private Partnership (PPP) agreement was by made between African Parks, Majete (Pty) Ltd. and the Malawian Department of National Parks and Wildlife (DNPW) to initiate one of Africa’s greatest reintroduction programmes. Over 2550 individual animals, comprising of 14 different species, were reintroduced into MWR. Elephant (*Loxodonta africana*), black rhino (*Diceros bicornis*), buffalo (*Syncerus caffer*), sable (*Hippotragus niger*), hartebeest (*Alcelaphus buselaphus*), several other antelope species, as well as many predators were all included in the reintroduction program.

During the months of June and July of 2017, 154 elephants were translocated from MWR to NWR, with the relative demographic information of many of the moved elephants recorded. This translocation has provided an opportunity to assess the usage response of artificial waterpoints by the elephant population within MWR to large-scale anthropogenic disturbances (i.e. translocations).

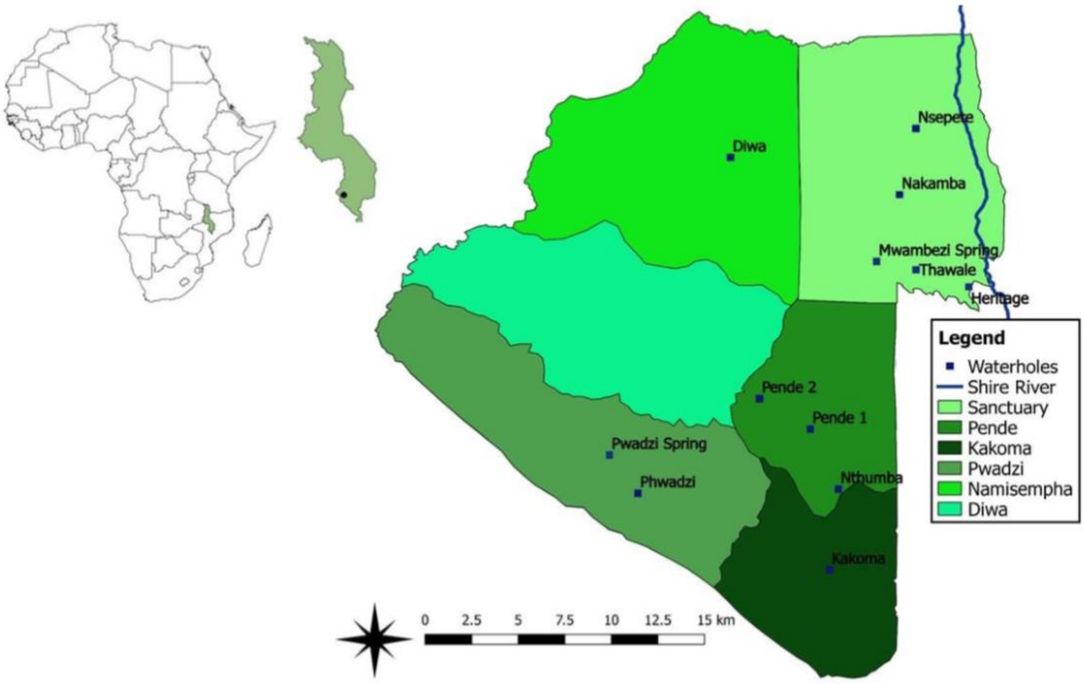


Figure 4.1. The location of Malawi on the African continent (green) and the position of Majete Wildlife Reserve within Malawi. The differing regions and locations of the perennial water sources that occur within the reserve are also shown. (Shapefiles per comms. African Parks (Pty) Ltd.)

4.3.2. Methods

4.3.2.1 Determining Artificial Water Point Usage by Elephants

Artificial water point(s) (AWP[s]) are defined as a man-made water hole which is filled via a solar-powered pump from an underground borehole water reservoir. There are 10 located throughout MWR (Figure 4.1) and they are all included in this study. Camera traps have been deployed at the AWPs since 2014, as part of a long-term monitoring program conducted by the Majete Wildlife Research Program. Images captured by the camera traps deployed at the AWPs within MWR between 2016 and 2019 were utilized to determine elephant use. Only female visits were considered in this analysis, since only two young adult males were removed during the translocation exercise.

Elephant use was characterised with two measures; the **Intensity of Usage (IOU)**, defined as the total number of independent sightings of all herds within a region for a given time period and the **Diversity of Usage (DOU)**, defined as the total number of different herds sighted at least once within a region for a given time period. These measures allow the number of herds that use a region to be determined. As an example: Herd-1 was independently sighted four times within Region X for Period Y and Herd-2 five times: IOU of Region X for Period Y is nine, DOU is two.

a) **Camera trap placement and specifications**

Two Cuddeback camera trap models were used: The X-Change Colour camera Model 1279 (CUDDEBACK, Wisconsin, USA) was used between the years 2018 and 2019, and the Professional Color Model 1347 (CUDDEBACK, Wisconsin, USA) was used between 2016 and 2017. Cameras were programmed to take one photo upon being triggered with a delay of 60 seconds between trigger events, and each camera was secured approximately 60 cm above the ground facing towards the artificial waterpoint. Cameras were serviced on average every 2-3 weeks which included changing of the SD cards as well as battery checks and replacements if needed.

b) **Identifying Herds from Camera Trap Sequences**

To permit analysis of camera trap images, the following assumptions were applied: 1) each image containing elephants (only family herds) was considered a single observation if no other photographs are taken; 2) if individuals were captured on camera in a continuous sequence of photographs in a short period of time (~30 minutes) on the same day, they were assumed to be from the same herd; 3) when family herds were confirmed by positively identifying at least two adult females in that herd with MWR elephant database, the sequence of photographs captured of the herd within 24 hours was considered to be a single observation for that particular herd; and 4) if a time period of greater than 60 minutes elapsed between photographs and new individuals were confirmed, it was assumed that they are from a different herd. Camera trap images were sorted and labeled using CAMELOT (GitLab version 1.5.5).

4.3.3. Statistical Analysis

Both the IOU and DOU were investigated at the regional level where regions were determined using general habitat types defined in previous studies (Weinand 2013; Forrer 2017; Geenen 2019). Three broad regions containing between two and four AWPS were assessed (Table 4.1). For descriptive observations on IOU and DOU, monthly IOU and DOU totals for each region (corrected for different effort levels) were converted to number of sightings per 100 camera trapping days to correct for differences in trapping lengths between AWPs.

All statistical tests were conducted in R (R Core Team 2018) and R-Studio (RStudio Team 2018, Version 1.2.1335).

Table 4.1. The three broad regions found within Majete wildlife Reserve and the corresponding artificial waterpoints (AWP) within each region

Region	Name of AWP	Total Number of AWPs
Sanctuary/Northern (Region 1) High preference	Nsepete Nakamba Thawale Heritage	4
Pende/ Southern (Region 2) Low preference	Pende2 Pende1 Nthumba Kakoma	4
Pwadzi/ Western (Region 3) Low preference	Diwa Pwadzi	2

4.3.3.1. Intensity of usage of AWPs

To assess regional changes in monthly IOU, the number of independent sightings for all breeding herds were summed by region and assessed using a moving window chi-square (where a chi-square test was done between consecutive months across the entire temporal spectrum), and significance corrected using a Bonferroni correction ($p < 0.05/39$) to account for multiple tests ($n = 39$ different chi-square tests).

A Pearson's correlation analysis was conducted between the chi-square values calculated above and the absolute change in percent of elephant IOU between months within each region. This was done as the chi-square value gives a relative indication of the degree of difference between regions between two consecutive months. Thus, if high chi-square value correlates with a high absolute change between intensity of use within a region then the movement of animals into or out of that region could be driving the change in the elephant IOU. The percentage change in IOU for each region was graphed with the chi-square value.

4.3.3.2. Diversity of usage of AWPs

A similar analysis as above was conducted on the DOU data. To assess regional changes in monthly DOU, the total number of different herds sighted at least once during that month were summed by region and changes were assessed using a moving window chi-square. Significance corrected using a Bonferroni correction ($p < 0.05/39$) to account for multiple tests. The percentage change in DOU for each region was graphed with the chi-square value. Since no monthly comparisons were statistically different (See Results), no assessment of within region DOU change was conducted.

4.3.3.3. Region herd composition before and after translocation

Frequency curves were developed for individual elephant herds pre- and post-translocation to investigate changes in proportion of time spent in each area by elephant herds between regions. To account for the difference in camera trapping efforts between regions and across the years, the total number of independent sightings of a herd each month was converted to the number of independent sightings per 100 camera trapping days and divided by the number of cameras in that region, and then summed per region. The responses were ordered by re-sight frequency, so herds sighted more often would occur closer to the y-axis than those sighted less frequently. Herds were categorised as either Sanctuary (Region 1), Pende (Region 2), or Pwadzi (Region 3) according to where the majority (largest proportion of sightings) pre-translocation sightings occurred, and categories were kept consistent for post-translocation curves. The pre- and post-translocation sighting frequencies of each herd was compared using descriptive statistics.

4.4. Results

From January 2016 to April 2019, 113 381 photos containing at least one elephant were identified. Of these, 83 209 were eligible for analysis (the other ~30 000 images were of male elephants). Sixty herds were identified across all three regions pre-translocation (Jan 2016 – June 2017) and 45 herds identified post-translocation (Jul 2017 – Apr 2019), and all regions had an increased frequency of captures during the dry period compared to the wet period (Figure 4.2).

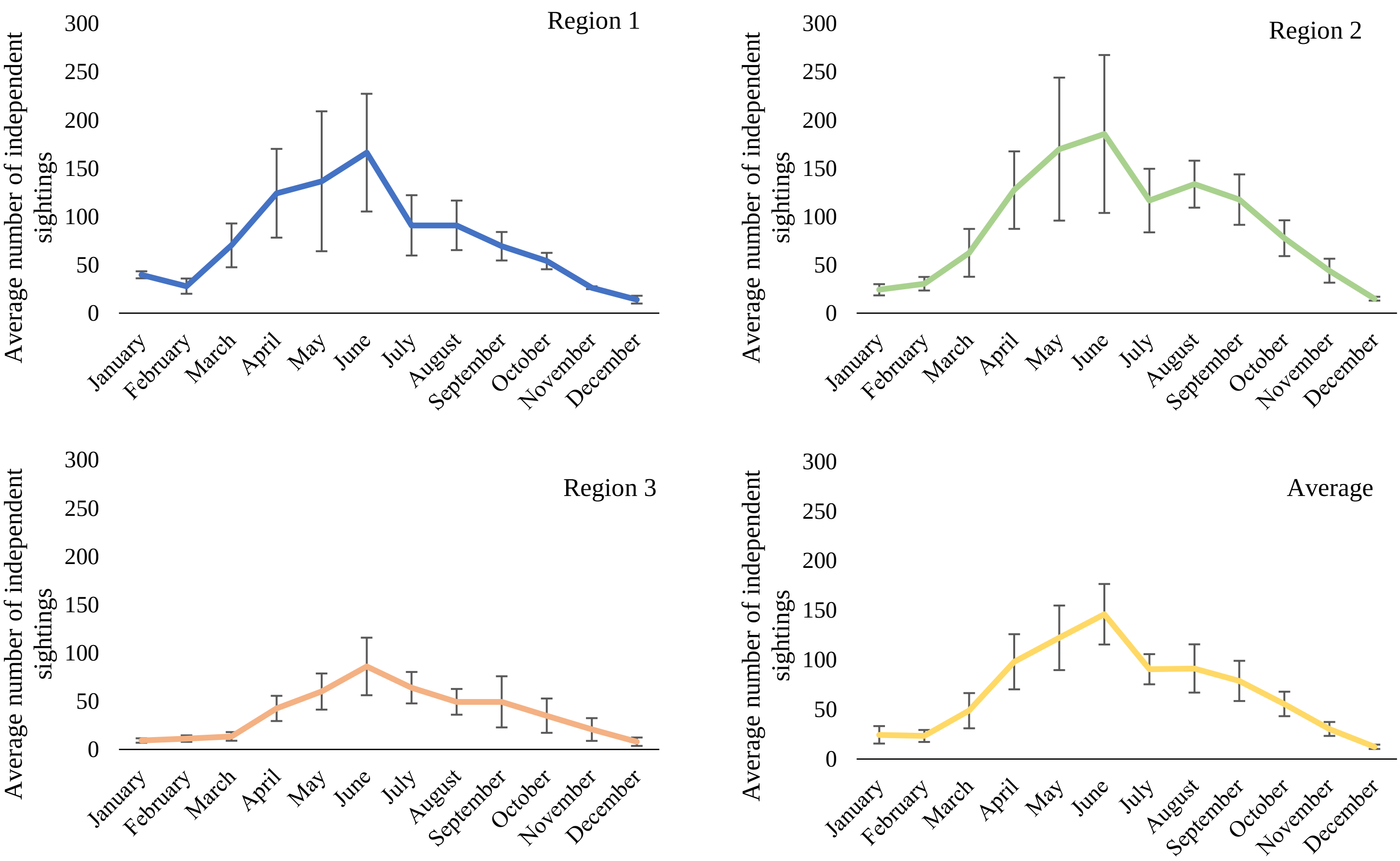


Figure 4.2. The average number of monthly sightings of elephant herds by region between January 2016 and April 2019. Error bars represent the standard error.

4.4.1. Intensity of Usage at AWPs

A strong seasonal influence was detected in the IOU, with AWPs utilized more during the dry months (May – November) as opposed to the wet months (December – April) (Figure 4.2, 4.3). The first half of the 2017 dry season saw the greatest IOU across all three regions, but there was a sharp decrease immediately after the translocation in regions one and two (Figure 4.4). The IOU of AWPs corrected by camera trapping effort during the following year's dry season (2018) was lower than similar periods in 2017 and 2016 (Figure 4.4). During the months recorded for 2019, region one experienced an increase in the IOU of herds utilizing the AWPs within its boundaries (Figure 4.4).

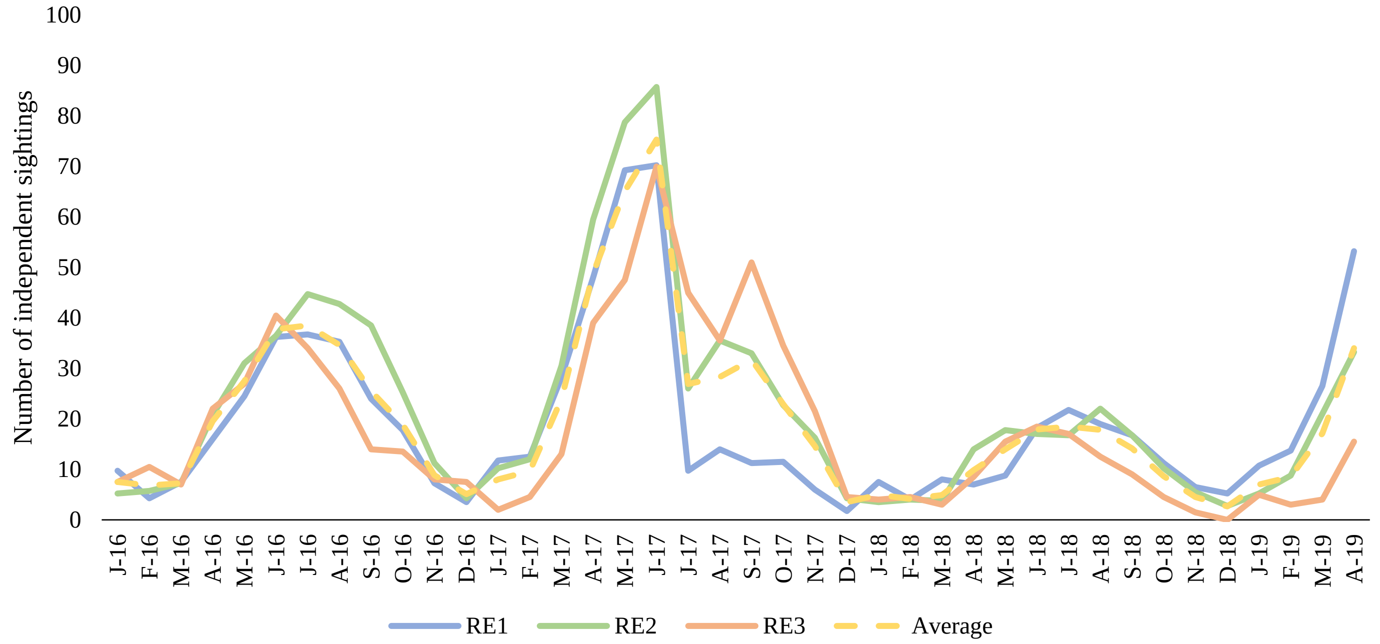


Figure 4.3. The total number of independent sightings of elephant herds per 100 camera trapping days in a region divided by the

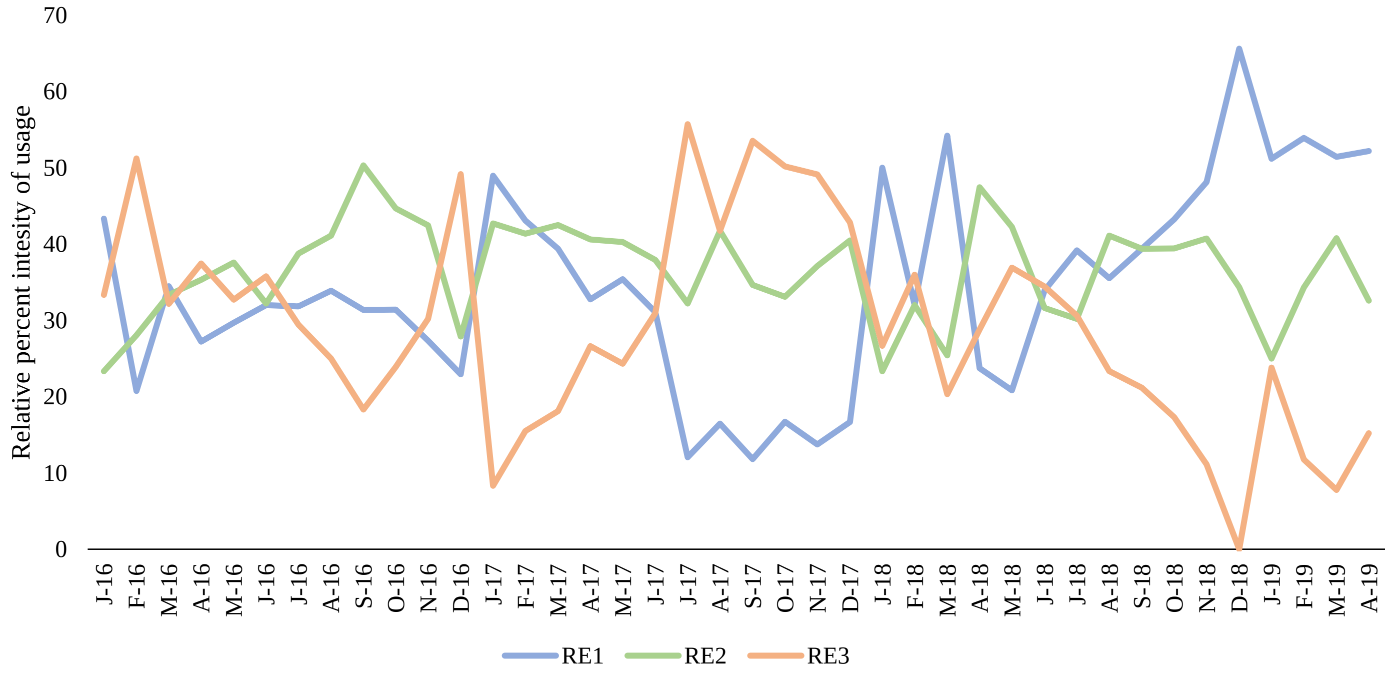


Figure 4.4. The intensity of usage corrected for camera trapping effort between the three regions within MWR across the years 2016 – 2019.

Significant changes in the IOU between consecutive months were detected at four separate time periods (T_{11}, Chi-square = 21.171, p = 0.00003; T_{17}, Chi-square = 54.25, $p < 0.00001$; T_{26}, Chi-square = 15.775, p = 0.00038; T_{35}, Chi-square = 8.869, p = 0.01186). Time periods T_{11}, T_{26} and T_{35} were all seasonal shifts in IOU, whereas the change detected in T_{17} (the largest change) was between the months of the translocation (June – July 2017) (Figure 4.5 – 4.6). Correspondingly, a steep decrease can be seen in the absolute change in percentage of IOU during this time in Region 1 (Figure 4.5).

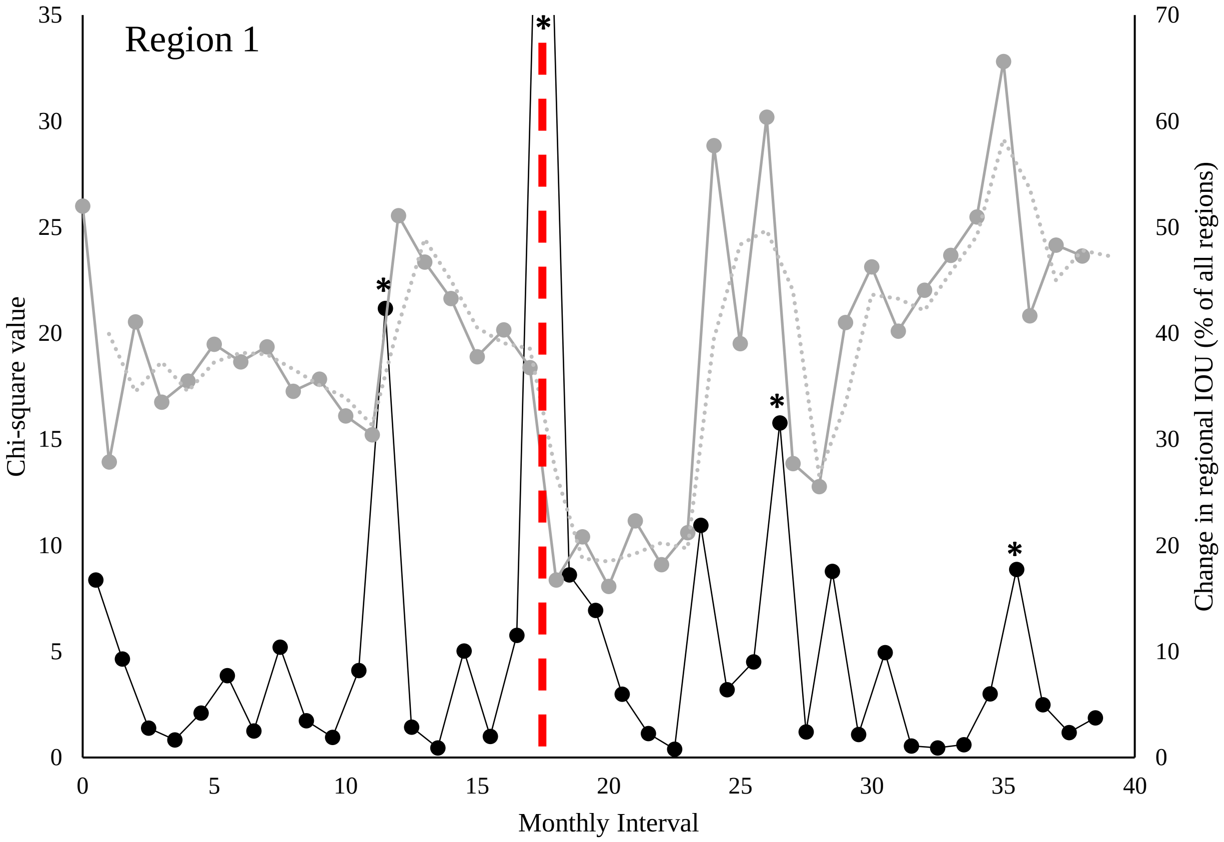


Figure 4.5. Moving-window analysis using a chi-square statistic (black line) to determine monthly IOU shifts in Region 1. The absolute change in percentage is represented by the grey solid line with moving average trendline, dashed grey line. The red dashed line represents the months of translocation (June and July 2017). Asterisks signify statistically significant changes in IOU within region. Significance levels were modified using a Bonferroni correct ($p < 0.001$). Significant intervals occur at: 11, Dec 2016 – Jan 2017; 17, Jun – July 2017 (chi-square value = 54.25); 26, Mar - Apr 2017; 35, Dec 2018 – Jan 2019.

Region 2's absolute change in percentage of IOU remained relatively constant throughout the translocation period, but then sharply decreased at the end of 2017 (T_{24}, Figure 4.6). Region 3 saw a sudden increase in absolute change in percentage of IOU during the period of the translocation but then a subsequent decrease until the present (Figure 4.7). The change in IOU between regions was found to be caused primarily by the changes in IOU in Region 1 (Person's Correlation, $r = 0.39$, $p = 0.015$; Figure 4.8 Region 1) and Region 3 (Pearson's Correlation, $r = 0.63$, $p < 0.0001$; Figure 4.8 Region 3). In contrast, the changes in Region 2 were not related to the overall shift in the distribution of herds (Person's Correlation, $r = 0.062$, $p = 0.71$; Figure 4.8 Region 2).

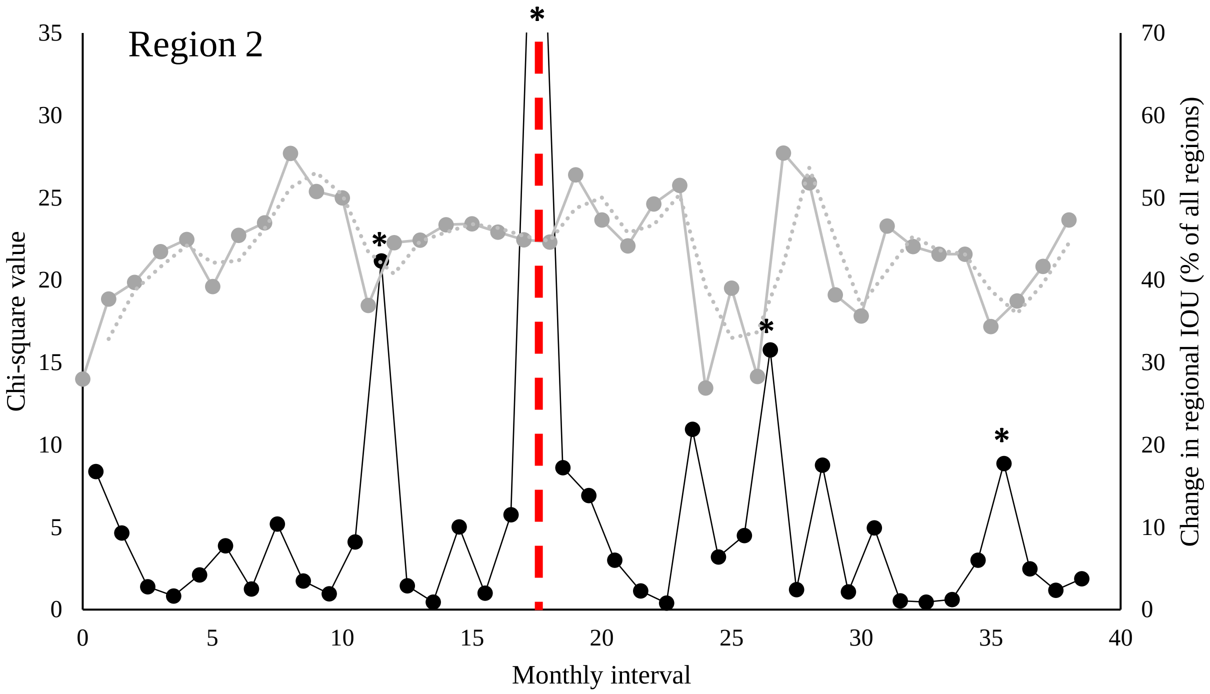


Figure 4.6. Moving-window analysis using a chi-square statistic (black line) to determine shifts in the monthly consecutive relative change in percentage intensity of use of elephant herds, in Region 2 in Majete Wildlife Reserve, Malawi. The absolute change in percentage is represented by the grey solid line with moving average trendline, dashed grey line. The red dashed line represents the months of translocation (June and July 2017). Asterisks signify statistically significant changes in IOU within region. Significance levels were modified using a Bonferroni correct ($p < 0.001$). Significant intervals occur at: 11, Dec 2016 – Jan 2017; 17, Jun – July 2017 (chi-square value = 54.25); 26, Mar - Apr 2017; 35, Dec 2018 – Jan 2019.

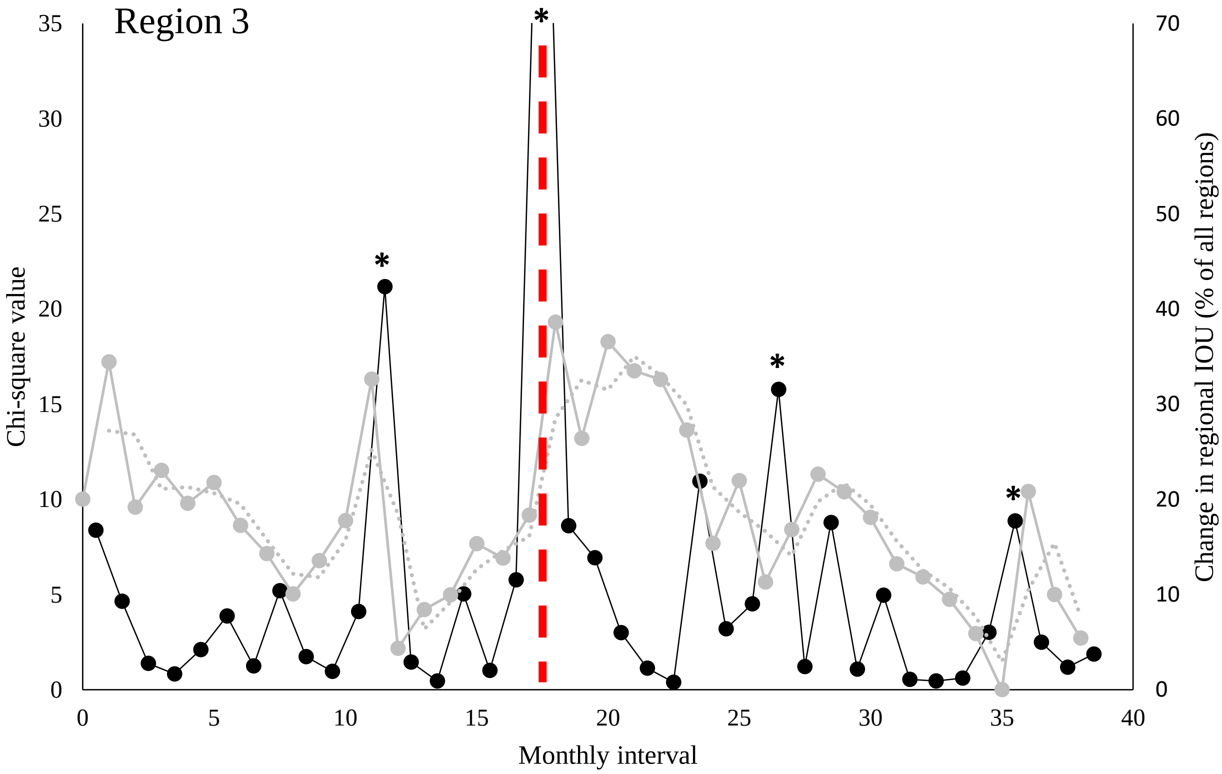
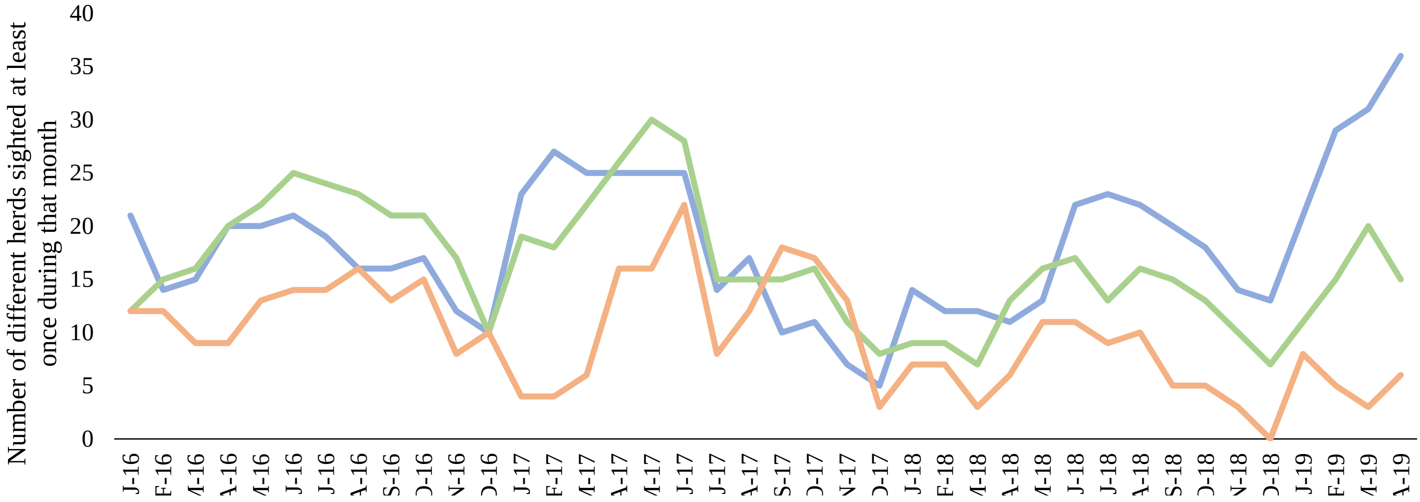


Figure 4.7. Moving-window analysis using a chi-square statistic (black line) to determine shifts in the monthly consecutive relative change in percentage diversity of use of elephant herds, in Region 3 in Majete Wildlife Reserve, Malawi. The absolute change in percentage is represented by the grey solid line with moving average trendline, dashed grey line. The red dashed line represents the months of translocation (June and July 2017). Asterisks signify statistically significant changes in IOU within region. Significance levels were modified using a Bonferroni correct ($p < 0.001$). Significant intervals occur at: 11, Dec 2016 – Jan 2017; 17, Jun – July 2017 (chi-square value = 54.25); 26, Mar - Apr 2017; 35, Dec 2018 – Jan 2019.

A strong seasonal influence on the DOU at AWPs was detected, with drier months having a greater number of different herds than wetter months when looking at both the uncorrected (Figure 4.9) and corrected (Figure 4.10) data. June 2017 saw the greatest number of individual herds sighted at least once across all three regions (75), whereas the following month (July 2017, the month directly after translocation) had a much lower value (37) (Figure 4.9).



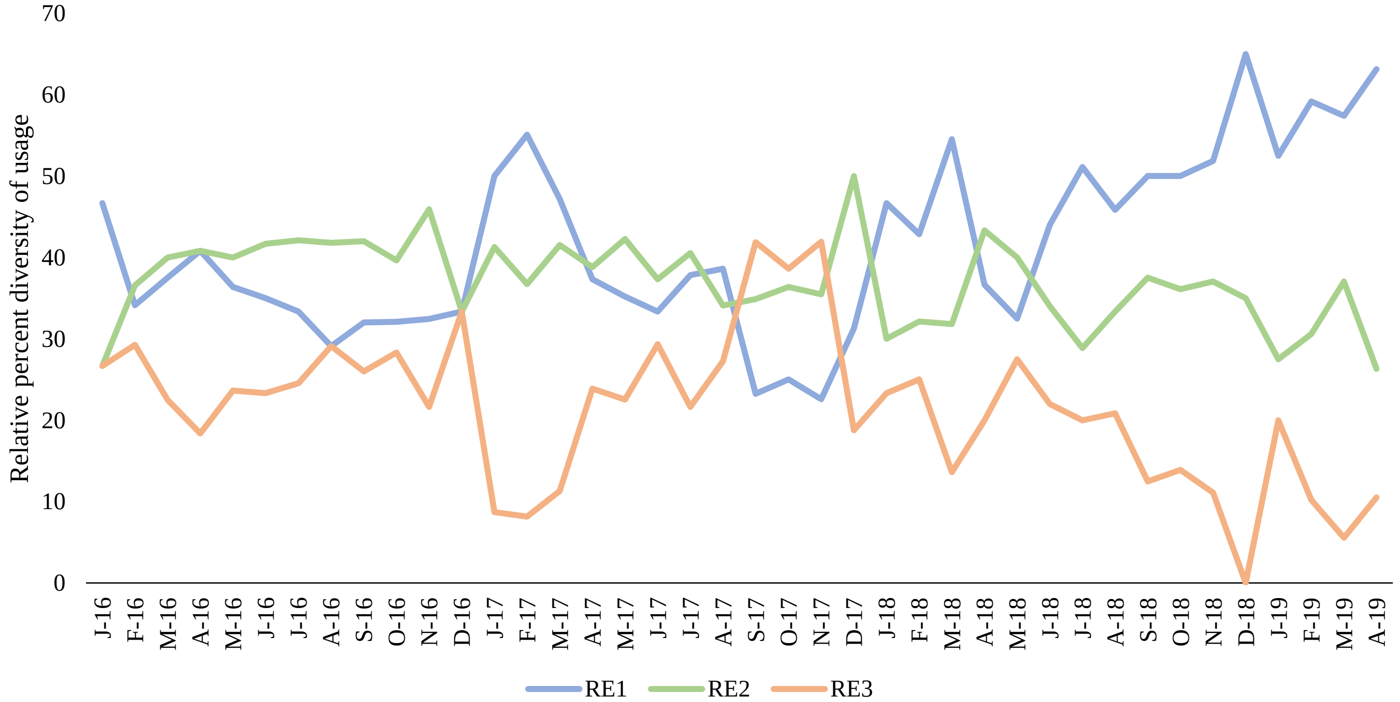


Figure 4.10. The diversity of usage corrected for camera trapping effort between the three regions within MWR across the years 2016 – 2019.

No significant changes in the DOU between consecutive months were detected between December 2016 and January 2017 (Figures 4.11 – 4.13). Although a decrease can be seen in the absolute change in percentage of DOU during the time of the translocation (between June and July 2017) in Region 1 (Figure 4.11). Region 2's absolute change in percentage of DOU remained relatively constant throughout the translocation period but saw an increase in December 2017 and a subsequent decrease in the following month

(Figure 4.12). Region 3 experienced an increase in absolute change in percentage of DOU during the period of the translocation but then a subsequent decrease until the present (Figure 4.13).

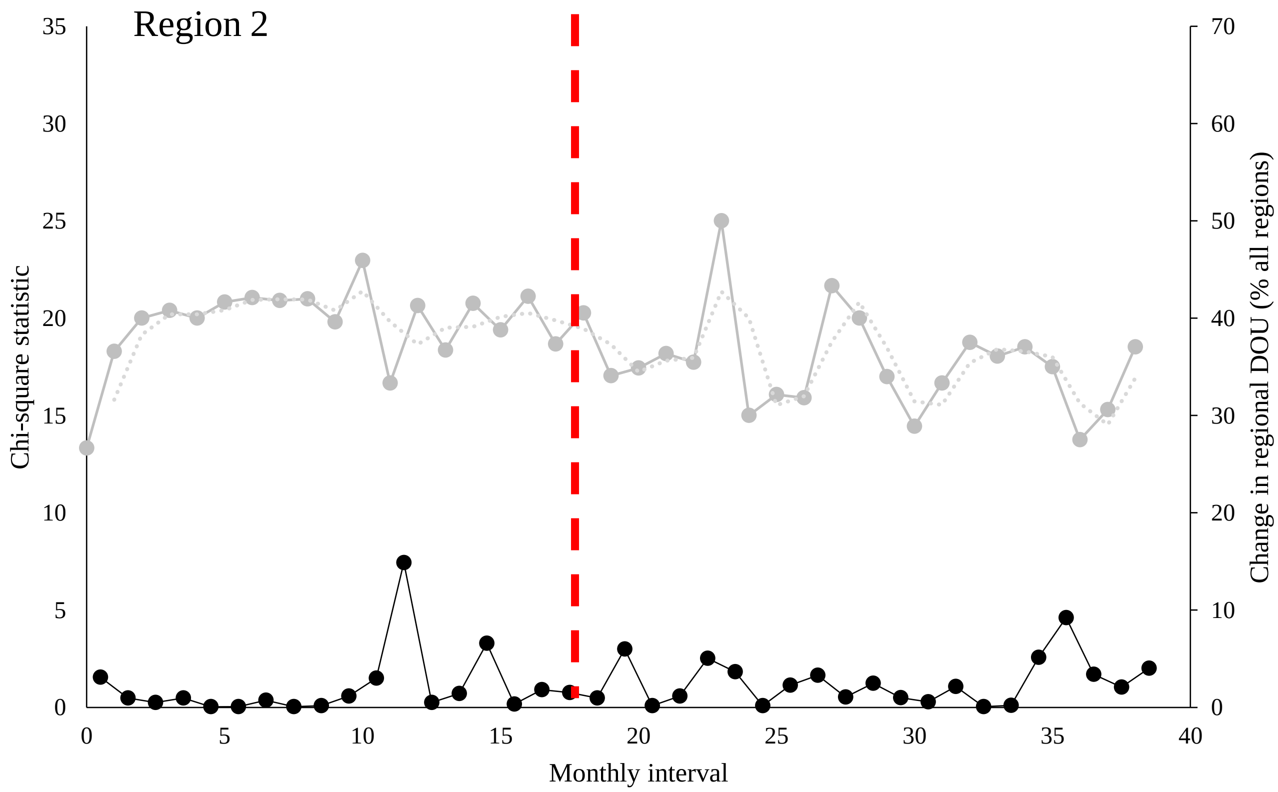
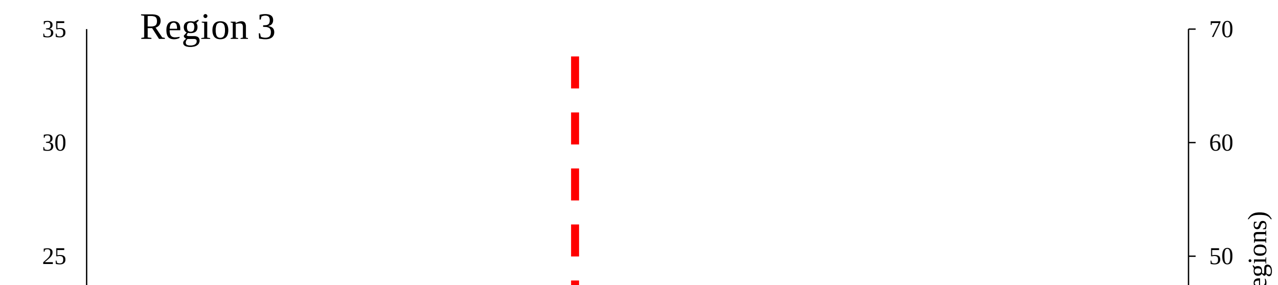


Figure 4.12. Moving-window analysis using a chi-square statistic (black line) to determine shifts in the monthly consecutive relative change in percentage diversity of use of elephant herds, in Region 2 in Majete Wildlife Reserve, Malawi. The absolute change in percentage DOU is represented by the grey solid line with moving average trendline, dashed grey line. The red dashed line represents the months of translocation (June and July 2017). Significance levels were modified using a Bonferroni correct ($p < 0.001$).



4.4.3. Region herd composition before and after translocation

In Figure 4.14 the curves on the left represent the number of sightings a specific herd was seen in Region 1, 2 and 3, respectively (Figure 4.14). As herd region labels were determined by the region a herd was sighted most frequently in between January 2016 and June 2017 (pre-translocation), it can clearly be seen that herds identified as Sanctuary herds were sighted more frequently in Region 1 (the Sanctuary) than Pende and Pwadzi herds were (Figure 4.14). The same pattern can be seen for Pende herds in Region 2 and Pwadzi herds in Region 3 (Figure 4.14). The R1 POST panel shows Pwadzi- and Pende-herds sighted more often within Region 1 (Figure 4.14) than in the R1 PRE panel.

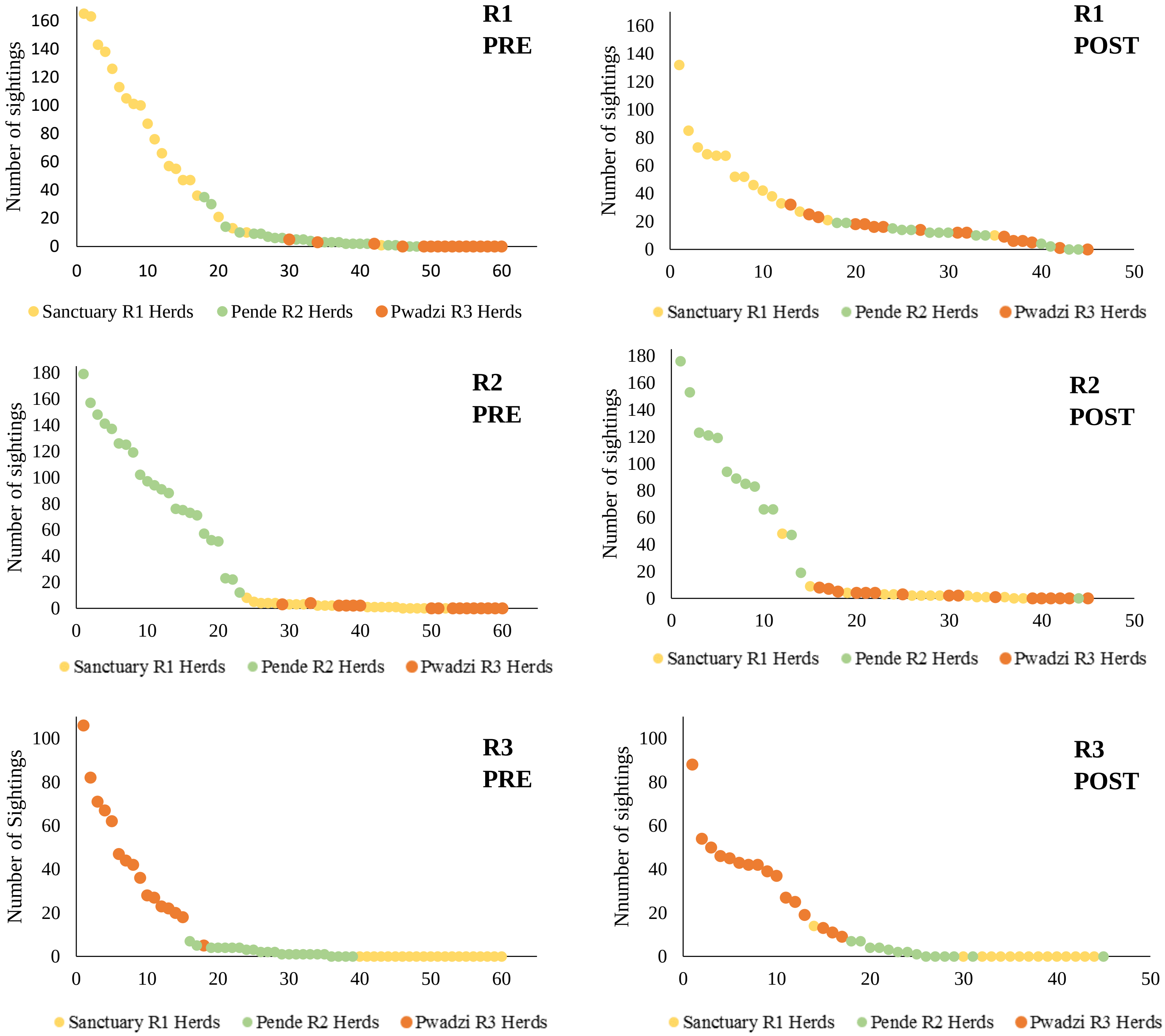


Figure 4.14. Frequency curves of the number of sightings of individual elephant herds sighted in a specific region. Panels labelled 'PRE' represent data collected between January 2016 and June 2017 (i.e. before the translocation) with those labelled 'POST' comprising of data from July 2017 to April 2019 (i.e. after the translocation had occurred).

Fourteen out of the 16 Region 3 herds were sighted at least 10% more frequently in Region 1 after the translocation (Appendix 3A). Only three Region 3 herds were sighted at least 10% more in Region 2 after the translocation. The same 14 herds that were sighted more frequently in Region 1 post-translocation, were sighted at least 10% less frequently in Region 3, their historic 'distributional range', after the translocation. Only one Region 2 herd (NTH6) was sighted more frequently in Region 1 after the translocation (Appendix 3B). As for Region 1 herds, three were sighted at least 10% less frequently in Region 1 after the translocation (Appendix 3C). Two of these herds (NAH6 and NAH14) were sighted at least 10% more frequently in Region 2, while NEH6 was sighted at least 10% more frequently in Region 3.

4.5. Discussion

Consistent with expectations, AWPs are an attractive force for elephants during dry seasons in MWR. However, the magnitude of the effects of surface water availability on the distribution of elephants has been questioned by one study which suggests that climate and physiological processes play a major part in determining the level of the dependence elephant have on water (Dunkin *et al.* 2013). Elephant distributions are also shaped by the landscapes they experience, especially with regards to dominance hierarchies. More dominant herds enjoy better access to areas with preferred food resources and ample water availability (Wittemyer *et al.* 1995; Wittemyer & Getz 2007; Wittemyer *et al.* 2007).The Sanctuary area (Region 1) in MWR is an area with abundant food resources, numerous perennial water sources, it lies in a lower altitude region, characterized by gentle slopes favoured by elephants (Nellemann *et al.* 2002; de Knegt *et al.* 2011), and is mainly covered by preferential vegetation types for elephants (low altitude mixed deciduous woodland, riverine associations and ridge-top mixed woodland) (Sherry 1989; Staub 2009; Weinand 2013). Since the Sanctuary is prime elephant habitat, resident herds were likely to be more dominant than those in the surrounding areas (Regions 2 and 3). The removal of herds primarily from the Sanctuary area during the 2017 translocation could have enabled an opportunity for traditionally excluded herds to move into better habitat. It should be noted that the amount of rainfall was not significantly different between the years studied (pers. comms. Majete Malaria Project) and thus changes detected can be ascribed to the effects of the translocation and not natural variation in rainfall between the years.

The 2017 translocation influenced the utilisation of resources by elephant herds throughout MWR especially as the translocation removed animals predominantly from a single area. Changes in IOU across the study period were attributable to changes in Regions 1 (the Sanctuary) and 3 (Pwadzi); even though herds were removed from Region 2 (Pende) there was no significant change detected in this area, likely due to the removal of only 2 herds. A change in the IOU was expected as less elephants would intuitively lead to a decrease in the utilization of areas and resources.

A DOU that has not changed significantly supports the theory that herds from surrounding regions have moved into different regions, as if this were not the case, we would have seen a significant decrease in the DOU in Region 1. Pwadzi herds (those traditionally seen in Region 3) have been sighted more frequently in Region 1 post-translocation 1, whereas Pende herds (traditionally found in Region 2) were not sighted more frequently in Region 1 compared to their pre-translocation sightings. Regions 2 and 3 remain relatively unchanged in terms of herd composition. However, two Region 1 herds and one Region 1 herd were sighted at least 10% more frequently in Region 2 and Region 3, respectively. These Sanctuary herds may have been scared out of the Sanctuary due to the disturbances of the translocation event (i.e. helicopters, large trucks, etc.).

The 2017 translocation of elephants out of MWR resulted in lower elephant densities in the Sanctuary area as well as the possible removal of many dominant herds. Subsequent lower densities and loss of potentially dominant herds resulted in herds (mostly likely subordinate) from the surrounding regions, mainly from the Pwadzi area (Region 3) to move into the Sanctuary area. Unfortunately, no dominance hierarchy existed on record prior to the translocation, and thus comparison between the new and old is not possible. However, the removal of key individuals disrupts well-defined social hierarchies and relationships and has been found to cause elephant populations to suffer stress (Gobush *et al.* 2008; Silk *et al.* 2010; IUCN, 2013). With the pre-translocation dominance hierarchy likely disturbed by the event, future research should aim to determine the new dominance hierarchy within MWR. Understanding the social landscape of the MWR elephant population will help inform future translocations as which herds to remove (through the investigation of the determined social network), as well as how translocations disrupt the social hierarchies of source elephant populations. This is important as the removal of the ‘wrong’ herds could result in unforeseen negative consequences. One such example, would likely occur by the removal of only, or mainly older/more dominant herds, as it has been found that young herds (those with younger matriarchs) under-reacted to the presence of key threats (McComb *et al.* 2011). With the majority of remnant herds now relatively young, they may be more susceptible to such threats.

Very little is known about how source populations respond to translocation events demographically, spatially, or socially. There are a multitude of studies investigating how the translocated animals respond to the event (examples: Letty *et al.* 2006; Teixeira *et al.* 2007; Witzenberger & Hochkirch 2008) with reviews on the subject focusing their attention on the success of the translocated animals with little to no mention of the state of the source population (Fischer & Lindenmayer 2000; Seddon *et al.* 2012; Bubac *et al.* 2019). To the author’s knowledge, only one study has been published which investigated developing an adaptive model to ensure the demographics of a population of New Zealand robins remained healthy as to allow for future ‘harvesting’ or translocations (Dimond & Armstrong 2007).

4.5.1 Conclusions

Having preferred forage and ample perennial water sources, both rivers and AWPs, (Weinand 2013), the Sanctuary can be considered a prime-elephant habitat and elephant herds of higher dominance status will most likely occupy this area (Wittemyer *et al.* 1995; Archie *et al.* 2006; Wittemyer & Getz 2007; Wittemyer *et al.* 2007). The artificial removal of elephant herds primarily from one region within MWR resulted in a significant decrease in the intensity of usage between different regions. The diversity of herds did not change, indicating that herds previously not seen in certain regions are now utilizing those areas. Management should be aware of the effects of such large-scale, highly localized disruptions, such as translocations, have on how elephants move through space and time, and bear in mind the findings of this study when planning future translocations.

4.6. Acknowledgements

I would like to thank the Earthwatch Institute for funding the Animals of Malawi Research Project, as well as this study. Thank you to the Earthwatch volunteers for your assistance with sorting through the thousands of camera trap images. Additionally, I would like to thank all members of the MWR management-, mechanic- and law-enforcement-teams for their assistance with the setting-up of cameras and subsequent data collection. Finally, thank you to my research partners for your assistance in the field, Craig Tambling for your immense guidance and patience, and my supervisors for assistance with the write-up.

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Chapter Five

First guidelines and ethical protocol for surveying African elephants (*Loxodonta africana*) using a drone.

5.1. Abstract

Unmanned aerial vehicles, commonly known as drones, are being increasingly used in management, conservation and ecological research. Numerous reviews on drones tout almost unlimited potential within the wildlife sciences as they open up inaccessible habitats. However, the influence of drones on the animals themselves is far less understood, and impact studies to construct protocols for best practices are urgently needed. The impact of approach speed, angle of approach and initial starting altitude were tested on the behavioural responses of African elephants (*Loxodonta africana*), along with sustained speed and flight pattern. Seventy nine approach flights and 70 presence flights were performed with a quadcopter drone. Approach speed and angle of approach were found to have a significant impact on how successful an approach flight was. Neither speed nor flight pattern had any measurable impact on an elephant's behaviour during a sustained flight. It is recommended that drones be launched at a distance of 100 m from an elephant/herd of elephants, ascending to a height of 50 m and using an approach speed of 2 m/s and an approach angle of 45° or less. This study aimed to provide a significant step towards the ethical use of drones in wildlife research. Future work is required to investigate the impacts of drones on other taxa. Physiological responses to drones would determine how accurately behavioural responses can be interpreted to determine the relationship between drone and subject animals.

5.2. Introduction

The study, monitoring, and management of animals as well as their habitats, is usually the focus of wildlife science (Chabot & Bird 2015). Despite the relatively simple goals of wildlife science, achieving these goals can be challenging as resources are often limited and wild animals tend to be elusive, wide-ranging, sensitive to anthropogenic disturbances, and/or dangerous to approach (Chabot & Bird 2015). Additionally, many target animals occupy habitats that are extensive, remote, challenging and perhaps impossible to access at ground-level.

New technologies have greatly aided the wildlife sciences in accessing difficult subjects in challenging habitats. Examples include motion triggered camera traps (O'Connell *et al.* 2011), aircraft (Fleming and Tracey 2008), remote sensing satellites (Kerr & Ostrovsky 2003), radar (Larkin 2005), thermal cameras (O'Neil *et al.* 2005), projectile-based animal capturing devices and chemical immobilization agents (Roffee *et al.* 2005; Schemnitz 2005), and a vast number of electronic animal tracking devices, as well as all of the accompanying software (Thomas *et al.* 2011). One new technology currently rapidly gaining popularity, most likely due to interest in the commercial sector, is of an aerial nature. Known variously as unmanned aircraft systems (UAS), unmanned aerial vehicles (UAV), remotely piloted aircraft systems or, mostly popularly drones, this burgeoning sector promises to offer further assistance to conservation efforts and the wildlife sciences. Popularity among wildlife biologists, ecologists and conservationists is clear from the many review papers investigating the applications and proliferation of drones in remote sensing, natural resource sciences and ecology (Watts *et al*. 2012; Anderson & Gaston 2013; Colomina & Molina 2014; Shahbazi *et al.* 2014; Whitehead & Hugenholtz 2014; Whitehead *et al.* 2014; Pajares 2015). Chabot and Bird (2015) conducted an extensive review of the applications of drones in wildlife management in which they highlighted optical surveying and observation of animals, uses in autonomous wildlife telemetry tracking, habitat research and monitoring as well as a review of the broader potential for UAVs. While the capabilities and potential practical uses of drones in the field of wildlife and conservation biology has been investigated thoroughly, their effects on the subjects of the studies (i.e. the animals themselves) has been investigated to a much lesser extent.

This study assessed the effects of drones on African elephants (*Loxodonta africana*). Previous drone related research has been conducted to investigate the use of drones in mitigating human-elephant conflict (Hahn *et al.* 2016) and for general elephant population survey purposes (Vermeulen *et al.* 2013), but no protocol or ethical recommendations have been investigated and determined. With their sensitivity to anthropogenic impacts, there is a clear need for guidelines as to how to conduct drone research focusing on elephants. The protocols developed in this study can be used to minimise stresses when utilizing drones for elephant related research.

The primary aim of this chapter was to determine how an unmanned aerial vehicle (hereafter referred to as a drone) should be operated when observing African elephants.

Specific objectives were to determine:

a) What the ideal approach speed and angle are when observing elephants aerially?
b) What the ideal sustained flight protocol is when observing elephants aerially?
c) How different elephant groups (breeding herds vs lone bulls vs bachelor herds) respond to the presence of a drone?
d) How the elephant populations of Majete Wildlife Reserve (MWR) and Liwonde National Park (LNP) differ in their responses to the presence of a drone?

It was hypothesized that: i) slower speeds and a less steep angle of approach would be the favoured approach protocol for droning African elephants as these patterns were successful with other taxa (Vas *et al.* 2015); ii) slower speeds were hypothesized to result in greater success rates for the sustained flight as well as a fixed flight altitude; iii) it was hypothesized that herds containing dependent offspring (i.e. breeding and mixed herds) would be more sensitive to the approach and presence of a drone (i.e. they would have lower rates of success regardless of approach and/or flight protocol); and iv), it was hypothesized that MWR's elephant population would be more sensitive to the approach and presence of a drone as LNP's elephants have had prior exposure to drones as well as extensive experience with helicopters.

5.3. Methods

5.3.1 Study Area

5.3.1.1. Majete Wildlife Reserve

The study took place in MWR, which is located at the southern tip of the Great Rift Valley in the lower Shire Valley region of southern Malawi (15.9364° S, 34.6414° E) (Figure 1). The reserve is 700km^2 in size with two perennial rivers, the *Mkulumadzi* and the *Shire*, determining/defining its northern and eastern boundaries, respectively.

The altitude within the reserve varies greatly, with the western region containing steeply undulating hills disrupted by river valleys with the terrain flattening towards the Shire River in the east. Two well defined seasons occur in Majete; the wet season (December to May) and the dry season (June to November). Annual precipitation varies within the reserve, with the eastern lowlands and the western highlands receive 680-800mm and 700-1000mm, respectively (Wienand 2013). Whilst water availability is dependent on seasonal rainfall, there are 10-artificial borehole-fed waterholes, as well as several naturally occurring perennial springs.

Majete Wildlife Reserve is dominated by multi-altitude woodlands, but also contains grassland areas. The four general classifications of vegetation type found in MWR are: 1) savanna (*Combretum* species, *Vachellia* species and *Panicum* species); 2) low altitude (205m-280m) mixed woodland (*Vachellia* species and *Steculia*); 3) medium altitude (230-410m) mixed woodland (*Brachystegia boehmii*, Diiospyrus *kirkii* and *Combretum* species); and 4) high altitude (410-770m) miombo woodland (*Brachystegia boehmii*, *Burkea africana* and *Pterocarpus*) (African Parks 2017).

In 1955, MWR was gazetted, but poaching and poor management led to most of its large game being decimated by 2003 (Forrer 2017). As a result, a Public Private Partnership (PPP) agreement was by made between African Parks, Majete (Pty) Ltd. and the Malawian Department of National Parks and Wildlife (DNPW) to initiate one of Africa’s greatest reintroduction programmes. Over 2550 individual animals, comprising of 14 different species, were reintroduced into MWR. Elephant (*Loxodonta africana*), black rhino (*Diceros bicornis*), buffalo (*Syncerus caffer*), sable (*Hippotragus niger*), hartebeest (*Alcelaphus buselaphus*), several other antelope species, as well as many predators were all included in the reintroduction program.

Since the reintroduction took place, wildlife has flourished. Majete Wildlife Reserve’s elephant population was no exception. The dramatic increase in elephant population size lead to a heavy impact on the vegetation within MWR. The combination of increased rates of vegetation damage, due to the rapidly expanding elephant population within MWR, and the desire to reintroduce elephants to Nkhotakota Wildlife Reserve (NWR) led to a decision to conduct a major translocation operation. During the months of June and July of 2017, 154 elephants were translocated out of MWR with the demographic details of each elephant removed recorded.

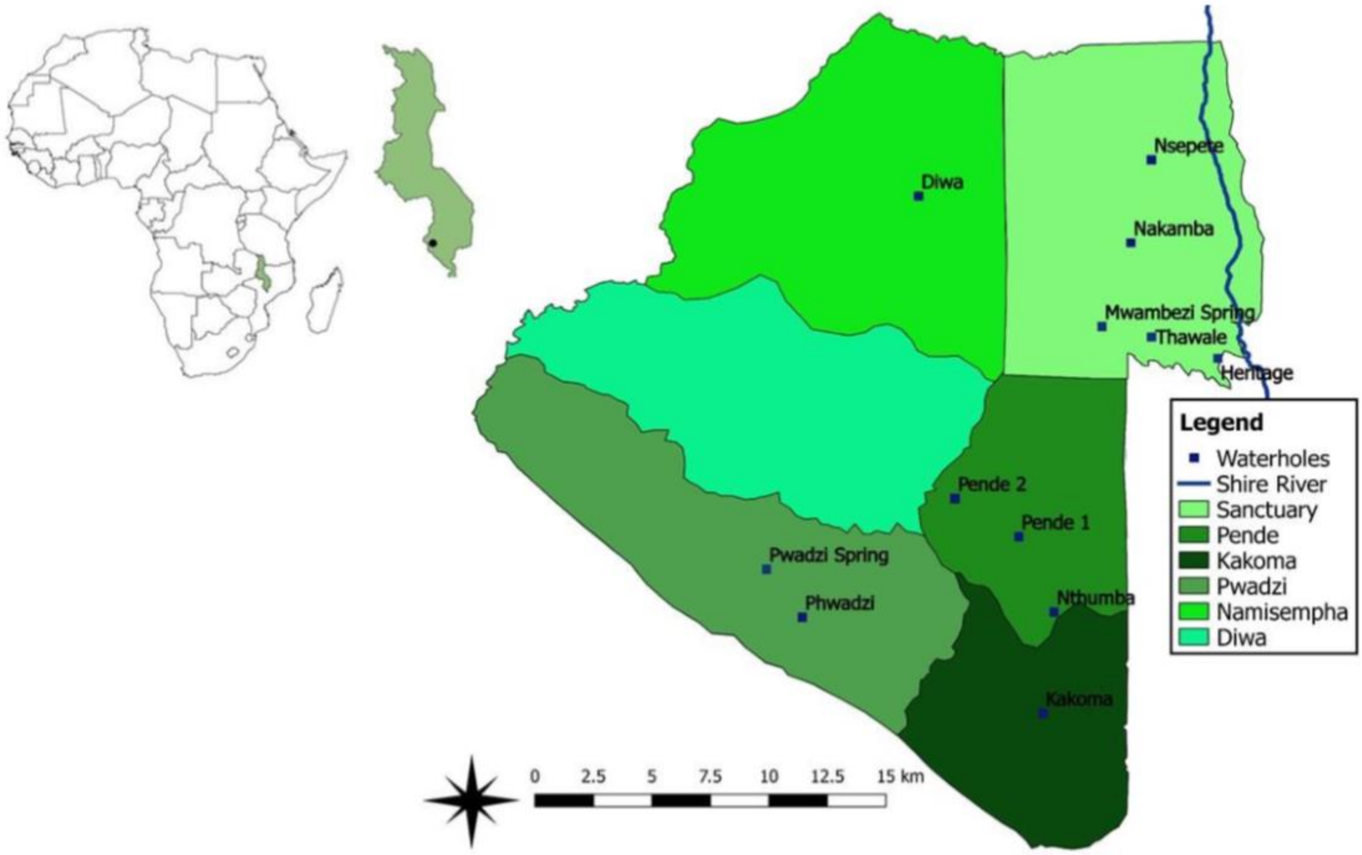

Figure 5.1. The location of Malawi on the African continent (green) and the position of Majete Wildlife Reserve within Malawi. The differing regions and locations of the perennial water sources that occur within the reserve are also shown. (Shapefiles per comms. African Parks (Pty) Ltd.)

5.3.1.2. Liwonde National Park

Liwonde National Park (LNP) is located in the Upper Shire Valley, which forms a part of the Great East African Rift Valley in southern Malawi (14.8441° S, 35.3466° E) (Figure 5.2). The park is 548km^2 in size and is generally flat except for three separate groups of hills. The dominant vegetation type in the park is *Colophospermum mopane* woodland, which occupies about 70% of the total area of the park. Other vegetation types include mixed woodland on the hills, floodplain, grassland and riverine forests/thickets, drought deciduous forest thickets and mixed woodlands on the hills, all of which occupy minor areas in the Park (Dudley 1994).

Liwonde's seasons differ slightly from those in MWR with its dry season from April to October and a rainy season from November to March. Annual rainfall ranges from 700mm to 1,400mm. There are four artificial water points which provide perennial water, located throughout the park.

Historically a sport hunting ground for European planters and administrators from 1920-1969 (Taylor 2002; Morris 2006), LNP was declared a controlled shooting area in 1962, which was updated to a game reserve in 1969 and finally gazetted into a National Park in 1973. Mangochi Forest Reserve was included within LNP's boundaries in 1977, and in 1978, LNP was formally opened to the public for game-viewing (Morris 2006).

In 2015, and in partnership with Malawi's DNPW, the African Parks Network assumed management of LNP. The new partnership saw an increase in financial contributions which allowed for the overhauling of law enforcement and the construction of a new perimeter fence along the boundary (Sievert *et al.* 2018). Increased management further led to the reduction in elephant and rhino poaching, the translocation of 1,329 animals for restocking other Malawian reserves, the return of five vulture species, the supplementation of the remnant black rhino population and, the reintroduction of lion (*Panthera leo*) and cheetah (*Acinonyx jubatus)* (African Parks 2018; Sievert & Reid 2018; Sievert *et al.* 2018). Liwonde National Park now supports a variety of mammals, of which the elephant and the hippopotamus (*Hippopotamus amphibius*) are the keystone species. Other common species include the waterbuck (*Kobus ellipsiprymnus*), sable antelope, impala (*Aepyceros melampus*), greater kudu (*Tragelaphus strepsiceros)* and warthog (*Phacochoerus africanus*). For a comprehensive description of the flora and plant communities, the reader is referred to Dudley (1994).

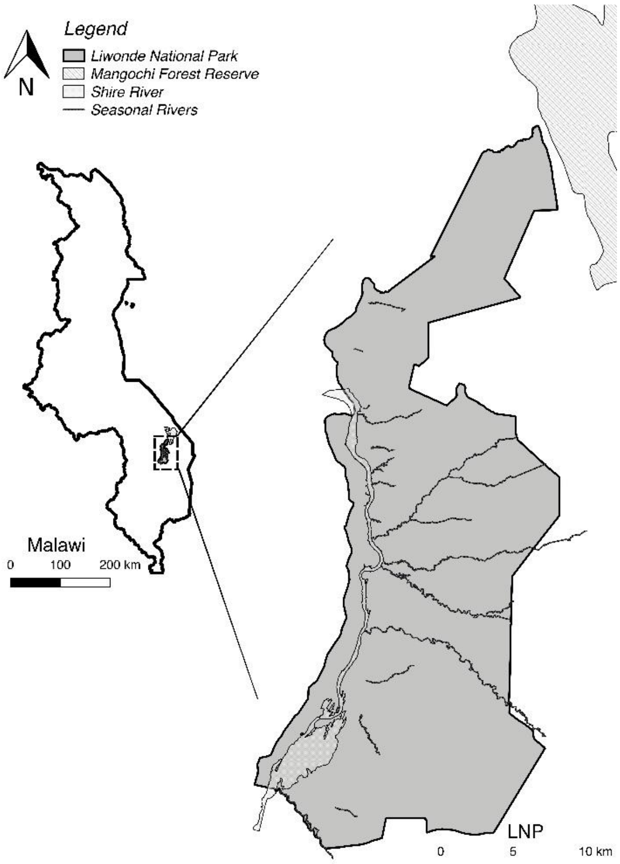

Figure 5.2. The location of Liwonde National Park within Malawi. The differing regions and locations of the perennial water sources that occur within the reserve are also shown. Image source: Olivia Sievert

5.3.2. Unmanned Aerial Vehicle (Drone)

A UAV quadcopter, known as the Mavic Pro Platinum (DJI, Shenzhen, China) was used in this study. This specific model was chosen for its compact design (easy to carry out in the field), ability to launch and land in most places (thanks to its quadcopter design) and noise reduction blades (4 dB quieter than the traditional DJI Mavic Pro). At the time of the study, September 2018, the Mavic Pro Platinum was the quietest commercial drone available. The drone is equipped with a GPS and internal measurement unit which both aid in determining the position and height of the drone. The drone is controlled via the use of a remote control

Table 5.1. Aircraft Specifications taken from the website of DJI (2018)

Aircraft	Supported Battery	LiPo 3S (3830mAh, 11.4V)
	Weight (Battery & Propellers Included)	734 g
	Hover Accuracy (Ready to Fly)	Vertical: +/- 0.1m, Horizontal: 0.3m
	Max Ascent / Decent Speed	5m/s Ascent, 3m/s Decent
	Max Flight Speed	65km/h
	Max Range from Remote	7km
	Max Flight Time (Single Battery)	27 Minutes
	Diagonal Motor-Motor Distance	335mm
Camera	Sensor	1/2.3” (CMOS), Effective pixels: 12.35 M (Total pixels: 12.71 M)
	Lens	FOV 78.8° 26mm (35mm format equivalent) f/2.2 Distortion < 1.5% Focus from o.5m to ∞
	Image Size	4000x3000
	Max Video Bitrate	60Mb/s

(DJI, Shenzen China) and has a maximum range of 7km. More information regarding the drone’s specifications can be found in Table 5.1.

5.3.3. Data Collection

Data collection in MWR occurred between September 2018 and April 2019, and in LNP between April and May 2019. The same methodology was applied within both reserves. As elephants range widely and unpredictably, the sampling scheme was opportunistic in nature.

When an elephant or herd was encountered, the drone was launched and flown towards it/them. The DJI flight software displays the appropriate metadata (altitude, distance, speed, etc.) and was captured by screen-recording the playback device (Apple iPhone 7+, Apple Inc., USA). The location of launch and landing was recorded as well as the ambient temperature, and strength and direction (up, down or cross) of the wind. Wind strength was determined by using the Beaufort Scale (Table 5.2). Flights were not conducted if the wind speed was greater than a ‘Strong breeze’ (Beaufort number 6, 10.8-13.8 $m.s^{-1}$), as this exceeds the drone’s wind tolerance capabilities. The drone was launched at a minimum distance of 100 m, as to ensure a safe enough distance from the elephant(s) should the launch induce an aggressive response.

There were two primary components to the data collection, 1) *approach*, and 2) *presence*. The *approach* methodology largely followed that conducted by Vas *et al.* (2015), with novel methodology utilized for the *presence* data collection.

Table 5.2. The Beaufort Scale was used to determine the strength of the wind (wind speed) during drone flights.

Wind Level	Description	Speed (m.s^{-1})
0	Calm - Smoke rises vertically	< 0.3
1	Light Air- Smoke drifts	0.3-1.5
2	Light Breeze - Wind felt on Face	1.6-3.3
3	Gentle Breeze - leaves & twigs constant motion	3.4-5.5
4	Moderate Breeze - Raises dust	5.6-7.9
5	Fresh Breeze- Small trees begin to sway	8-10.7

5.3.3.1. Drone Approach

From the launch site, the drone ascended vertically to either a height of 35, 50 or 100 m, and then proceeded to approach the elephant(s). The speed and angle of approach was varied in the following manner according to the specific approach pattern of the particular flight: (speed: 2, 4 and 6 m.s^{-1}; angle: 45° and 90° from the horizon-thus, the 90° trajectory involved flying directly over the elephant(s) before descending).

Each variable was given a 'shortcut' and a subsequent abbreviation (Table 5.3), and the total 18 possible flight patterns were each given a unique code depending on their unique variables (e.g. FAH = Fast [6 m.s^{-1}], Angle [45°], High [100 m], see Appendix 2A for complete list of approach variables and Appendix 2B for presence variables). Before the flight was initiated, a random approach protocol was chosen. This was done by assigning each approach protocol a number (1 – 18 [Appendix 2A]) and utilizing Excel to generate a random number between one and 18.

During each approach, the live on-screen video was recorded and analysed post-flight for elephant responses. Response data was determined using a standardized scoring system adapted from Langbauer *et al*. (1991), Poole (1999), O'Connell-Rodwell *et al*. (2006) and Soltis *et al*. (2014). The elephant's behaviour was scored for the display of one of five responses; 1) No Response; 2) Vigilance; 3) Agitated; 4) Flight; and 5) Aggressive (see Table 5.4 for descriptions). At one-minute intervals during each approach, the drone height from the ground and distance from the elephant were recorded (m), as well as the elephant's response.

Approaches were classed as "successful" if the drone was able to reach a distance of 30 m or closer to the elephant(s) without inducing a type 3, 4 or 5 response from at least one adult individual. Conversely, approaches were deemed "unsuccessful" if the elephant(s) displayed a type 3, 4 or 5 response from at least one adult individual before reaching the target distance of 30 m. Thirty meters was determined to be close enough as the Mavic Pro Platinum's 2x zoom capability enables high resolution observations from this distance. Flights were terminated if the elephants(s) displayed either a Flight (4) or an Aggressive (5) response at any point during the flight, which was done to minimize the stress experienced by the elephant(s).

5.3.3.2. Drone Presence

If the approach did not elicit a Type 4 or 5 response, the drone was kept in the air and the *presence* portion of the data collection commenced. Just as with the approach portion, the drone could be flown at speeds of 2, 4 or 6 $m.s^{-1}$. Additionally, the drone could be kept at a fixed height of 30 m (the 'fixed' flight protocol), or it could ascend/descend between 25 and 35 m (the 'varied' flight protocol). Before starting

the presence portion of the flight, a flight protocol was chosen at random using the same technique as described for the approach methodology (See Appendix 2B for complete list of flight pattern combinations and corresponding codes and numbers).

Table 5.3. The variables used throughout the *approach* and *presence* portions of data collection with their corresponding 'shortcuts' and abbreviations.

Portion of Data Collected	Variable		Shortcut	Abbreviations
Approach	Speed ($m.s^{-1}$)	2	Slow	S
		4	Medium	M
		6	Fast	F
	Angle of Approach (°)	45	Angle	A
		90	No Angle	N
	Initial Height (m)	35	Low	L
		50	Medium	M
		100	High	H
Presence	Speed ($m.s^{-1}$)	2	Slow	S
		4	Medium	M
		6	Fast	F
	Flight Pattern	Fixed Height	Fixed	F
		Varied Height	Varied	V

At one-minute intervals, the height from the ground and distance from the elephant were recorded, as well as the elephant's dominant reaction over the course of that minute. The same response categories as the approach methodology were used. Presence flights were deemed 'successful' when the drone's specific presence flight protocol did not elicit a type 3, 4 or 5 response. Once again, flights were terminated if the elephants(s) displayed either a Flight (4) or an Aggressive (5) response at any point during the flight, which was done to minimize the stress experienced by the elephant(s).

5.3.4. Statistical Analysis

Data was captured using MS Excel and analysed using STATISTICA 13 (*TIBCO Software Inc. (2017); http://statistica.io.*, Dell). Basic summary statistics were used to describe the variables of interest using 95% confidence intervals. Relationships between continuous variables and a nominal predictor variable were examined using one-way ANOVA or non-parametrically using the Mann-Whitney tests (for two groups) or Kruskal-Wallis tests (comparing more than two groups). Nominal variable relationships were investigated with contingency tables and likelihood ratio or Pearson's chi-square tests. Data were checked to meet assumptions of the GLZ. The results of GLZs were reported to investigate the influence of Environmental- and Flight-Variables (Table 5.5) on the success of flight approach or presence. A p-value of $p < 0.05$ represented statistical significance for all hypothesis testing. Model fits were reported via multiple measures (AICc, BIC and Nagelkerke's R2)

Analysis of the difference in presence flight lengths was checked for normality. The data violated normality assumptions, thus requiring the use of a non-parametric test, and a Kruskal-Wallis test was used.

Table 5.4. The various possible elephant responses recorded during both the *approach* and *presence* data collection.

Elephant Response Type	Code	Description
No Response	1	No visible sign of disturbance. Elephant(s) continue with what they were doing prior to the drone approaching/persisting.
Vigilance Response	2	Elephant(s) stop what it/they was/were doing. Head is turned towards the direction of th drone with ears slightly ajar and fixed. Trunk possibly extended towards the direction o the drone (attempting to smell the source of the noise i.e. the drone). Aware of the drone presence, but no visible signs of disturbance.
Agitated Response	3	Clearly aware of the drone's presence. Defensive behaviour: shielding of young, ears he completely out, slight vocalizations (small trumpets) and headshakes.
Aggressive Response	4	Elephant(s) actively fleeing in the opposite direction to the drone. Loud vocalizations possible (loud trumpeting).
		Loud vocalizations (trumpeting). Ears held completely out. Headshakes. Often stands

Table 5.5. Input variables used to investigate possible factors influencing drone approach and presence flights around elephants.

Variable	Environmental	Flight
Ambient Temperature	X	
Wind Speed (Beaufort Scale)	X	
Season (Wet/Dry)	X	
Weather (Overcast/Cloudy/Sunny)	X	
Population (MWR/LNP)	X	
Inf. & Calv. Present?	X	
Total # Individuals	X	
Flight Speed		X
Angle of Approach		X
Starting Altitude		X
Flight Pattern		X

5.4. Results

It was found that the population from which the elephant belonged had no significant effect on the outcome of an approach or presence flight and thus analysis was conducted only on the entire data set (MWR & LNP).

5.4.1. Approach Protocol Results

5.4.1.1. General Trends

A total 79 Approach flights were undertaken between September 2018 – April 2019, with 63 trials in MWR and 16 in LNP (Table 5.6). In both sites, lone bulls were droned the most. Average percent of successful approaches, regardless of protocol was 52% in MWR and 60% in LNP, albeit with a lower number of trials. Group type did not affect the likelihood of an approach being successful (GLZ, Estimate = 0.44879, $p = 0.324$, Table 5.7).

Table 5.6. A summary table displaying the breakdown in effort for Approach flights across Majete Wildlife Reserve and Liwonde National Park.

	Majete Wildlife Reserve			Liwonde National Park			Total		
Elephant Type	**Absolute # droned**	**% Droned**	**% Success**	**Absolute # droned**	**% Droned**	**% Success**	**Absolute # droned**	**% Droned**	**% Success**
Bachelor Herd	14	22	43	4	25	75	18	23	50
Breeding Herd	12	19	50	3	19	33	15	19	47
Lone Bull	25	40	56	6	38	67	31	39	58
Mixed Herd	12	19	58	3	19	67	15	19	60
Total	**63**	**100**	**52 (Av)**	**16**	**100**	**60 (Av)**	**79**	**100**	**54 (Av)**

When broken down by approach variable (speed, angle and starting altitude), slower speeds resulted in greater percentages of successful approaches (GLZ, Estimate = 0.90850, $p = 0.004$, Table 5.7; Figure 5.3). Likewise, an approach angle of 45° resulted in greater percentages of successful approaches (GLZ, Estimate = 0.04724, $p = 0.01458$, Table 5.7; Figure 5.3) than a 90° approach angle. However, starting altitude was found to have no significant influence on the success of an approach (GLZ, Estimate = 0,00554, $p = 0.718$, Table 5.7). No environmental factors were found to influence approach success significantly (Table 5.7).

Table 5.7. The results from the global GLZ run on variables influencing the success of an Approach. Variables with a significant effect are highlighted in red. Only the first 10 rows reported. AICc = 64.98, BIC = 83.70, Nagelkerke R^2 = 0.75.

Effect	Estimate	Standard Error	Lower CL 95%	Upper CL 95%	p
Intercept	-5.59199	4.163003	-13.7513	2.567345	0.179188
Temp	-0.06374	0.135311	-0.3289	0.201465	0.637593
Speed	0.9085	0.316199	0.2888	1.528239	0.004063
Angle	0.04724	0.019322	0.0094	0.085105	0.014498
Altitude	0.00554	0.015351	-0.0245	0.03563	0.718096
Calv. or Inf.	-0.38873	0.41792	-1.2078	0.430381	0.352295
Season	0.40891	0.402906	-0.3808	1.198595	0.310149
Weather	-0.35873	0.728948	-1.7874	1.070026	0.622677
Elephant Category	0.44879	0.335948	-1.2824	1.178826	0.323627
Population	0.1137	0.66327	-1.1863	1.413687	0.863889

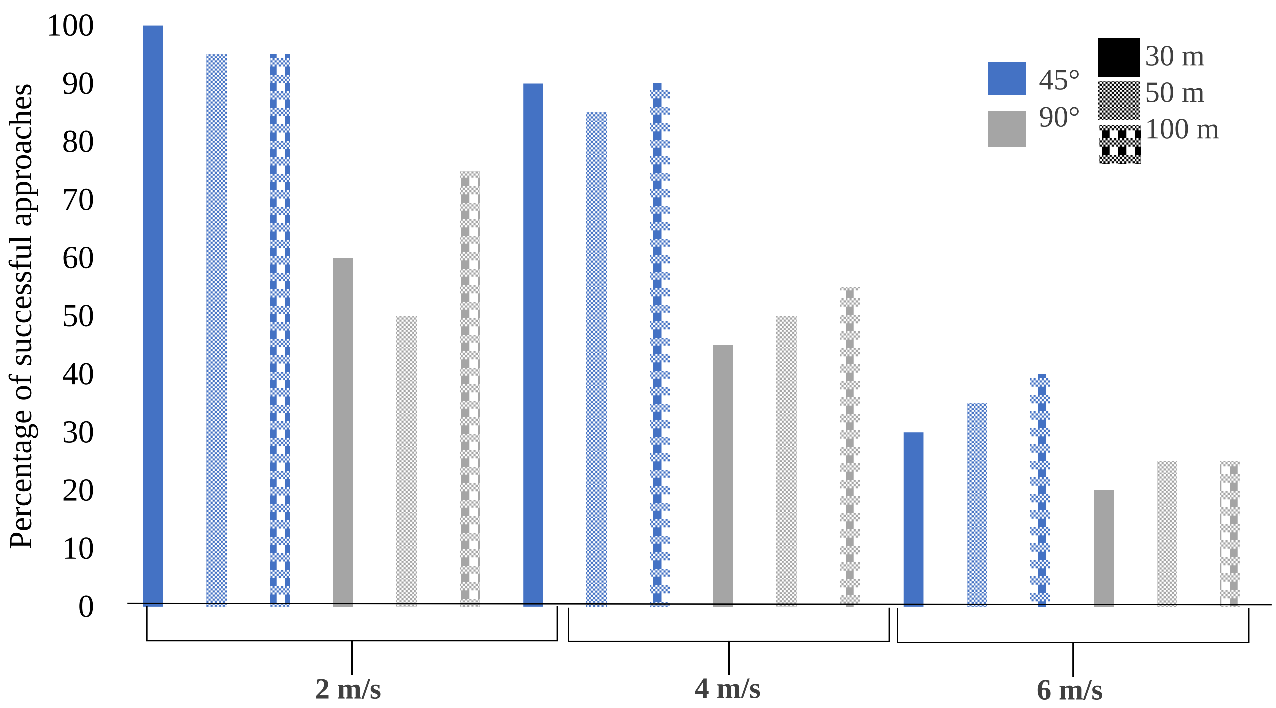

Figure 5.3. The percentage of successful approaches, regardless of elephant category, when broken down by Approach Speed (2, 4 or 6 m/s), Angle of Approach (45° or 90°), and Starting Altitude (30, 50 or 100 m).

5.4.2. Presence Protocol Results

5.4.2.1. General Trends

A total 70 Presence flights were conducted between September 2018 – April 2019, with 56 trials in MWR and 14 in LNP (Table 5.8). Again, lone bulls were droned the most. Average percent of successful presence flights, regardless of protocol was 56% in MWR and 70% in LNP, albeit with a lower number of trials. Once again, group type did not affect the likelihood of a presence flight being successful (GLZ, Estimate = 0.68829, p = 0.422, Table 5.9). Average presence flight length was 15 minutes. However, the flight length of presence flights with a preceding successful approach differed significantly from those with unsuccessful preceding approach flights (Kruskal-Wallis chi-squared = 42.369, p-value < 0.0001, Figure 5.4).

Table 5.8. A summary table displaying the breakdown in effort for Presence flights across Majete Wildlife Reserve and Liwonde National Park.

	Majete Wildlife Reserve			**Liwonde National Park**			**Total**		
Elephant Type	**Absolute # droned**	**% Droned**	**% Success**	**Absolute # droned**	**% Droned**	**% Success**	**Absolute # droned**	**% Droned**	**% Success**
Bachelor Herd	12	21	58	3	21	100	15	21	67
Breeding Herd	11	20	45	3	21	67	14	20	50
Lone Bull	22	39	55	5	36	80	27	39	59
Mixed Herd	11	20	64	3	21	67	14	20	64
Total	**56**	**100**	**55 (Av)**	**14**	**100**	**70 (Av)**	**70**	**100**	**60 (Av)**

No presence flight variable (speed and flight pattern [fixed vs varied]) was found to significantly influence the success of a sustained presence flight (Table 5.9; Figure 5.5). Likewise, no environmental factors were found to influence a sustained presence flight's success significantly (Table 5.9).

Notably, only the success of the preceding approach was found to influence the success of the following presence flight (GLZ, Estimate = 2.39497, p < 0.0001, Table 5.9).

Table 5.9. The results from the global GLZ run on variables influencing the success of a sustained presence. Variables with a significant effect are highlighted in red. Only the first 10 rows reported. AICc = 52.20, BIC = 67.83, Nagelkerke R^2 = 0.79.

Effect	**Estimate**	**Standard Error**	**Lower CL 95%**	**Upper CL 95%**	**p**
Intercept	2.96177	5.392610	-7.60755	13.53109	0.582849
Temp	-0.07591	0.171585	-0.41222	0.26039	0.658180
Speed	-0.01996	0.363246	-0.73191	0.69199	0.136298
Pattern	0.54753	0.484675	-0.4242	1.49747	0.258612
Succ. App.	2.39497	0.605801	1.20762	3.58232	0.000077
Calv. or Inf.	-1.16054	0.930763	-2.98480	0.66372	0.212444
Season	0.40891	0.402906	-0.3808	1.198595	0.310149
Weather	0.46393	0.438124	-0.39478	1.32264	0.289645
Elephant Category	0.68829	0.435648	-1.55824	1.068226	0.422127
Population	-0.34112	0.228979	-0.73191	0.69199	0.956174

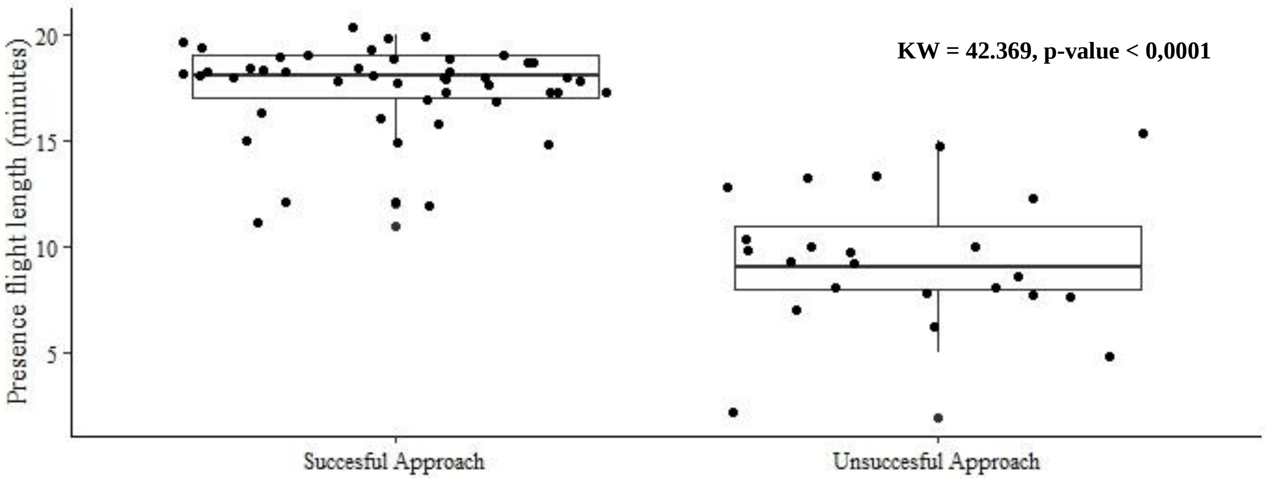

Figure 5.4. Box and whisker plot with a jitter plot overlay. A significant difference was detected between the two groups with significance levels measured at $p < 0.05$.

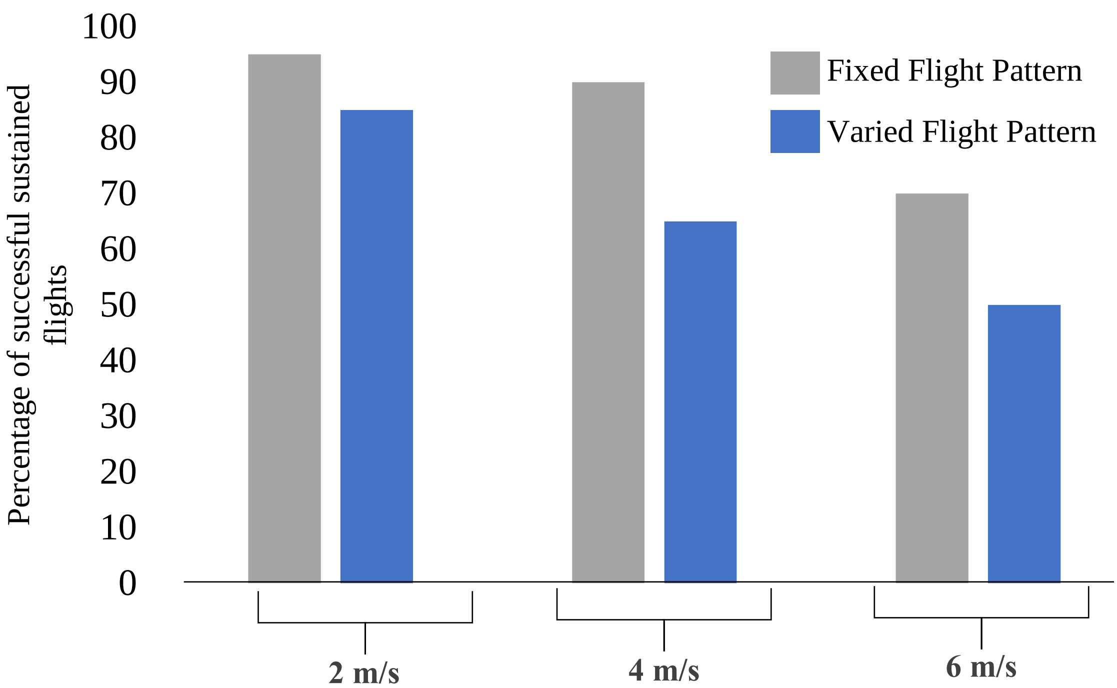

Figure 5.5. The percentage of successful presence flights, regardless of elephant category, when broken down by Flight Speed (2, 4 or 6 m/s) and Flight Pattern (Fixed or Varied).

5.4.2.2. Investigating within a Failed Approach

It was quite clear that if the preceding approach was a success it was likely that the commencing sustained flight was as well. However, we wanted to tease this apart further, to determine if there were any trends with the 'failed approach' flights that may aid drone pilots in ensuring a successful sustained flight. For this section, we solely investigated the data set as a whole (i.e. including both populations). Three predictor variables (Flight Speed, Flight Pattern and Elephant Category) were selected for investigation based on their relevance to possible elephant behavioural responses.

The generalized linear model, which only included the data where there had not been a preceding successful approach, indicated no predictor variables with a significant effect on the success of a sustained flight (Table 5.10).

Table 5.10. The results returned from the Failed Approach data set GLZ run on variables influencing the success of a sustained presence. AICc = 26.70, BIC = 30.50, Nagelkerke R^2 = 0.15.

Effect	Estimate	Standard Error	Lower CL 95%	Upper CL 95%	p
Intercept	4.852860	3.185993	-1.39157	11.09729	0.127712
Speed	-0.521698	0.591605	1.68122	0.63783	0.377866
Pattern	-0.628334	0.692862	-1.98632	0.72965	0.364477
Elephant Category	0.860604	0.703881	-0.51898	2.24019	0.221460

5.5. Discussion

The rapid spread of drone technology throughout the civil and scientific sector is encouraging. The timely and repeatable manner in which drones deliver HD picture and video footage of animals in often hard-to-reach places is a tool that every conservationist should have in their back pocket. And while the potential uses for this novel technology have been well documented (Jones *et al.* 2006; Koh & Wich 2012; Allan *et al.* 2015; Chabot & Bird 2015; Christie *et al.* 2016), an investigation into the potential impacts this particular method of collecting data may have on the study subjects themselves is required. A handful of previous studies have indicated the usefulness of drones for collecting observational data, and have quantified responses, for example with seals (Pomeroy & Connor 2015), birds (Vas *et al.* 2015) and bears (Ditmer *et al.* 2015). In this study, we quantified how elephants responded to various drone approach patterns and sustained drone flights. Although only a 54 and 60% success rate were achieved for the approach and presence flights, respectively, key insights into which factors influence elephant behavioural responses were obtained.

5.5.1. Approach Flights

Environmental variables, such as ambient temperature, weather, wind speed or season did not influence success rates of approach flights. Contrary to predictions, the category of elephant group targeted for a flight also had no effect on approach flight success, so females with calves and infants were not more sensitive to the approach of a drone. Nor did starting altitude affect approach success, even though it would allow for more time for elephants to assess whether the drone was a threat or not. Perhaps most surprisingly, there was no difference between populations, even though LNP elephants had high exposure to drones and helicopters and would be expected to show lower responses in general. This result might be interpreted with caution due to the low sample size in LNP.

Approach speed and angle significantly affected the success of an approach flight. The speed aspect makes logical sense, as the drone is quieter at slower speeds and may allow elephants additional time to identify (or attempt to determine) whether the drone is a threat or not. The reason a 45° angle of approach was preferable to a 90° of approach may very well follow similar logic. In the field, when elephants elicited an agitated response, they were observed to often turn towards the direction in which the drone was approaching. With a 45° angle of approach, elephants were observed to tilt their heads upwards and appeared to be looking directly at the drone, once again, determining whether it was a 'real' threat or not. However, with a 90° of approach, elephants were unable to look directly above them and thus the noise from an unknown source descending upon them intuitively seems to explain their discomfort with this approach angle.

5.5.2. Presence Flights

As for flight variables, very surprisingly neither pattern (a fixed or varied flight height) nor speed had a significant effect on success rates. It was assumed that slower speeds would yield greater rates of success as would a fixed flight height. A fixed flight height was assumed to be preferable to elephants as the pitch of the drone does not change much if kept at a constant height. This was indicated in Figure 5.6, as across the three speeds, a fixed flight pattern consistently yielded a higher percentage of successful sustained flight. Likewise, slower speeds resulted in higher percentages of successful sustained presence flights. However, for both variables there was no statistical difference.

The only variable that was found to have a significant effect on the success of a sustained presence flight was if the preceding approach flight had been successful or not. It was found that if an approach was successful, a sustained presence would almost always be too. In an attempt to tease this apart further, the data were split into two groups, one which had preceding successful approaches, and one that did not. However, no variable (environmental or flight) was found to influence success significantly.

5.5.3. Liwonde National Park

As only just over a week was spent in LNP, a limited amount of data were collected. Not all approach- or presence-flight patterns were conducted, and thus statistical analysis within and between LNP's and MWR's elephant populations was not possible. However, from the data collected, no clear trends were apparent in either the approach flights or presence flights. This indicates that there appears to be a critical amount/number of flights required for patterns to emerge. Somewhere between the number of flights conducted within MWR (63 approaches and 56 presences), where clear trends and significant differences were observed, and LNP (16 approaches and 14 presences) is the minimum amount of flights required for statistical validity.

Unlike in previous studies where the animals were easy to locate and almost always readily accessible (e.g. Vas *et al.* 2015), droning sessions took place opportunistically when elephants were found in the field. Thus, despite spending 14-months in the field collecting data, a relatively low number of observational flights (both approaches and presences) were conducted. Additionally, not all patterns were flown the same number of times, with some never being flown at all (the case for the LNP population). This may account for some of the variables not being significant in influencing elephant behavioural responses to the drone.

5.5.4. Conclusion

This study has clearly demonstrated that the speed and angle at which elephants are approached by a drone play a critical role in the elephants' ability to tolerate drones. Since presence flights depended only on the approach success, early exposure to drones determines the ability to stay with target subjects. Drone pilots, regardless of purpose (i.e. scientific or cinematic) need to be made aware of this and understand that speed and angle of approach will affect elephant responses, but once approaches are unsuccessful, mitigating the effects in order to remain with target elephants becomes difficult. Essentially, drone pilots should approach elephants with extreme care as reckless flying could result in agitating a population permanently thus inhibiting any meaningful future drone work to be conducted. Further highlighting the importance of carefully approaching elephants aerially is the difference between the average length of time of sustained flights, for flights that had a successful versus unsuccessful approach. Successful approaches allowed a presence flight to be around the target elephant(s) for an average of seven and a half minutes more than presence flights with an unsuccessful approach. Aggressive or reckless approaches not only create negative associations between the target elephant population and drones, but it ultimately inhibits long periods of observations and compromises the data/footage captured.

A major assumption of this study, as well as previous ones (Pomeroy & Connor 2015; Vas *et al.* 2015) is the use of observable behaviour as a proxy for the animals' response to drones. While this may serve as an acceptable model for the time being, greater efforts should be made to measure the physiological response of the animals as well. Ditmer *et al.* (2015) found that although bears showed little to no outward behavioural response to drones their heart rates increased significantly during drone flights. For elephants, the use of faecal samples to measure cortisol levels, as to measure stress levels of elephants (Foley *et al.* 2002; Viljoen *e al.* 2008; Ganswindt *et al.* 2010), would provide information as to whether the outward behavioural response elicited by a drone is a good indicator of its physiological response as well.

This study hopes to act as a platform from which future research can be built upon. It aimed to outline a methodology of how to quantify an animal's response to the approach and presence of a drone. While the focus of this study was elephants, the methodology employed can be easily manipulated to a wide variety of other wildlife species. If we wish to utilize new technology to aid in the efforts of conservation, at the very least we should understand how these technologies effect those that we are trying to conserve.

5.6. Ethical Clearance

This study was cleared ethically by the University of Stellenbosch's Research Ethics Committee: Animal Care and Use. Protocol number: ACU-2019-7822.

5.7. Acknowledgements

Thank you to the Earthwatch Institute for funding the Animals of Malawi Project and for funding this study. I would like to thank the Management Team of MWR for their support, and MWR's various staff members, who without, data collection would not have been possible. Additionally, I would like to thank DJI and the D1Store, Australia, and Ilenna Copley for the sponsorship and acquisition of the DJI Mavic Pro Platinum and accompanying spare parts used throughout this study. Finally, thank you to my research partners for your assistance in the field, Professor Daan Nel for help with statistics, and my supervisors for assistance with the write-up.

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Chapter Six

Research findings, conclusions and management recommendations for African Parks, Majete managerial staff.

6.1. Overview

It is anticipated that the elephant population within MWR will increase, as there is ample forage, abundant year-round water and reduced resource competition due to the removal of 154 individuals in 2017. However, future population growth rates are uncertain now that there is an extreme adult-male bias in the sex ratio, with significantly fewer females of breeding age. The first two research chapters of this study investigated the demographic and habitat use responses of an elephant population to a large-scale anthropogenic event. This was achieved through individual identification techniques, which allowed the accurate recording of information on the size/age-class structure and sex ratio of MWR's elephant population. Additionally, through the repeated identification of herds between artificial waterpoints across the years (2016-2019), it was determined that herds found predominantly in the south-western parts of MWR had migrated into the areas from which other individuals/herds were translocated from. Finally, the third research chapter quantified elephant responses to the various approach and flight patterns of a drone. By utilizing a standardized protocol, insights into which variables influence elephants' reactions to a drone (both its approach and sustained presence) were identified. The findings aimed to provide drone pilots, whether scientific or cinematic, with firm guidelines to fly responsibly around elephants, using the least stressful flight patterns available to them.

6.2. Research Findings

6.2.1. Chapter Three: The demographics of the African elephant population in MWR

- The most recent aerial count (Nov 2018) conducted in MWR counted a total of 210 individual elephants. However, aerial counts often tend to under-sample woodland habitats which dominate MWR's vegetation, and thus the actual number of elephants within MWR is most likely higher. This was confirmed by the fact that by the end of April 2019, this study was able to positively identify 236 individuals.
- As the elephants of MWR are a closed, non-dispersing population, there are minimal effects of immigration, emigration, human-elephant conflict and heavy poaching to consider. Since the removal of individuals in 2017, the population has grown at an estimated annual rate of 7%.
- The age structure of MWR's elephant population was determined to be as follows: 7% infants (< 1 year), 18% calves (1 – 4,9 years), 11% juveniles (5 – 9,9 years), 23% small-adults (10 – 19,9 years), 27% medium-adults (21 – 34,9 years) and 14% large-adults (>35 years).
- The sex ratio of individuals below the age of 10 was roughly 1:1:1 (male:female:unknown), and the sex ratio of adults was heavily skewed towards males with a 5:2 ratio (males:females).

- Twenty cases of double counting were found in MWR's previous demographic survey, and so at the time of translocation there were almost certainly fewer elephants than previously estimated. The translocation therefore may have had a larger than anticipated effect on the MWR elephant population.

6.2.2. Chapter Four: The use of artificial waterpoints of the remaining African elephant population post translocation

6.2.2.1 Intensity of Usage

- The intensity of use (IOU), i.e. the rate at which elephants used different artificial water points, varied across regions within MWR between 2016 and 2019.
- The IOU fluctuated with seasons across all three regions, with the all regions experiencing greatest IOU during drier months.
- Pre-translocation, generally, the Sanctuary and Pende areas (Regions 1 and 2) had consistently higher IOU than that of Pwadzi (Region 3).
- The translocation event of 2017 significantly decreased the IOU of AWPs. Although IOU changed over all months, all other changes, while significant, were much smaller in scale than the one as a result of the translocation.
- Post translocation Region 1 saw a steep decrease in IOU, most likely due to the numerous herds removed from that area, followed by a steady increase in IOU until the present.
- Region 2 remained relatively undisturbed in terms of IOU likely due to the limited number of herds removed from the region.
- Region 3 remained undisturbed during and immediately after the translocation. However, six months after the translocation event, the region's IOU began to decrease.

6.2.2.1 Diversity of Usage

- There was no significant change in the diversity of usage (DOU) i.e. number of different herds using AWPs across the three regions between 2016 2019.
- Slight seasonal fluctuations in DOU were detected as DOU increased during drier months.
- Contrary to expectations, there was no immediate change in DOU after the translocation event that occurred between June and July 2017. This suggests that remnant herds from the surrounding areas were quick to replace those removed.
- Herds traditionally sighted in Pwadzi (Region 3) were more frequently sighted in the Sanctuary (Region 1) post-translocation period. This suggests that the increases in IOU and constant DOU are largely due to herds from Region 3 beginning to utilize AWPs in the Region 1 more.

6.2.3. Chapter Five: How African elephants respond to the approach and sustained presence of a drone.

- MWR elephants did not respond differently to drone approach than elephants in LNP, despite LNP elephants being significantly more experienced with drones and helicopters.
- 54% of approach flights, regardless of protocol, were a success.

- 60% of sustained presence flights, regardless of protocol, were a success.
- Approach speed and angle affected success of approach flights; slower speeds and approach angle of 45° (vs 90°) resulted in higher percentages of successful flights. Starting altitude did not influence approach success.
- When separated into presence variables (speed and flight pattern [fixed altitude vs. varied altitude]), once again, slower speeds resulted in higher percentages of successful presence flights, as did a fixed flight pattern compared to a varied flight pattern. However, statistically, neither was found to significantly influence the likelihood of a successful sustained presence flight.
- The only significant effect on the sustained flight's success was the approach. No environmental or flight variables significantly affected the success of a sustained-presence flight.

6.3. Management Recommendations

6.3.1. Translocation impacts on source elephant population demographics and MWR elephants going forward

It is still too early to fully assess any long-term translocation effects on the demographics of MWR's elephant population, but this study provides important baseline data for future monitoring and decision making. This study was limited in the impact assessment by a lack of historic comparison data, and it is strongly recommended that in-depth counts, utilizing the mark and re-capture methods and the database created by Frances Forrer (2016) and updated by Wesley Hartmann (2019), be conducted every second year so as to accurately track the population's demographic response, as well as more detailed data collected on the elephant population, such as explicit conception, birth and death rates. More generally, for effective decision making and management, MWR urgently needs a more detailed elephant monitoring programme, which could follow a model similar to that developed by Dimond & Armstrong (2007), and would require a full-time staff member's attention. Crucial metrics such as calf survival, conception rates or population recruitment to breeding age will determine population growth rates and currently cannot be assessed for MWR. Although this may seem a significant investment, it is hard to see how management outcomes could be successful without better information on the complexities of elephant population dynamics.

This study found MWR's elephant population to have an estimated average annual growth rate (AAGR) since the translocation event of 7%. This is lower than historically calculated (11%, Forrer 2017), and could decline further due to the severe adult male bias in the sex ratio (Whyte *et al.* 1998). Even at an AAGR of 7%, MWR's elephant population numbers could surpass pre-translocation levels within five years. However, AAGR are very small snapshots on elephant dynamics, and may be too simplistic to generate good predictions about the likelihood of needing management intervention. Better demographic monitoring is urgently needed to give management authorities much stronger data on which to base decisions, all of which require significant investment of limited resources.

Before any option is considered, management must of course ask itself if there is a need for intervention. Consistent with African Park's Policy on Elephant Management (PEM) (African Parks 2017), one valid management strategy is no intervention while building knowledge through monitoring (van Aarde & Jackson 2007; African Parks 2017). The Policy quotes valuable insight from Owen-Smith *et al.* (2006), "management decisions must reconcile scientific principles with economic, political, social and aesthetic considerations in order to achieve their mandated aims". The Policy also notes that elephant impacts are site specific, and therefore there is no clear or definitive answer to the need for and timing and scale of management intervention. In this incredibly uncertain space, management is required to clearly and explicitly state well-defined goals and objectives against which site-specific impacts can be monitored and assessed (African Parks 2017). The former allows for the determination of whether elephant impact is acceptable or not.

The Policy goes on to specifically define the following by which management's decision to intervene must be evaluated:

- National policies and objectives (in this context, the National Elephant Management Plan for Malawi [NEAPW 2015])
- Area-specific management objectives
- Levels of poaching
- Localised elephant densities
- Spatiotemporal distributions and variability in resource use
- Observed impacts on vegetation, other species and people
- Habitat sensitivity and uniqueness
- Opportunities and resources available for intervention.

The above will be considered throughout the following possible strategies discussed. African Parks' PEM names five management options: 1) Self-regulation through density dependence; 2) Metapopulation management through translocation and range expansion; 3) Sterilization; 4) Immunocontraception; and 5) Culling. Options one, two, four and no intervention are discussed below. Options three and five are not discussed as they are not seen as ethical options by the primary author.

6.3.1.1. Self-Regulation Through Density Dependence – the "van Jackson" Way

Jestingly coined 'the van Jackson way', after the combination of the two authors (van Aarde & Jackson 2007), this strategy does not involve any active management intervention. Interventions without a complete understanding of how elephants are impacting their environment are reckless, which is a statement that is consistent with African Parks' policy. Rather, under this framework, management would set up clear monitoring programmes to evaluate the impacts of the remaining elephants across MWR prior to developing intervention strategies. There are several good reasons to consider this the most preferable option for MWR at this stage.

Firstly, it is currently unclear how the severe male bias in the adult sex ratio will influence the population's growth rate; increased male-male competition could have large effects on the social landscape for MWR elephants and may alter female reproductive success. Secondly, although resource competition may have decreased, there are now very few reproductively active females in this population, and few to be recruited into the reproductive cohort in the coming years. Age at first reproduction is not currently known for MWR elephants, nor is calf survivorship or inter-birth intervals. Finally, it would seem wholly unnecessary to implement a management strategy to slow population growth rates when the previous translocation seems likely to have done exactly that. In fact, it seems rather short-sighted.

This 'inaction' would be in-line with the national policies and objectives of the Department of National Parks and Wildlife of Malawi, as their National Elephant Action Plan has set a target of 22 000 elephants and largely inhibits any strategy which would seek to undermine this. By waiting to intervene again in MWR, valuable data can be collected providing management with a better understanding of how its population has responded to the translocation of 2017, as well as contributing to the National target of 22 000 elephants within Malawi by potential natural population increase. It is noted here that the 2017 translocation was also in-line with National policies as the elephants were moved within Malawi and therefore not counted as a loss of overall elephant numbers.

Levels of poaching have decreased significantly since 2003 when African Parks assumed management of MWR and as long as existing law-enforcement practices and good relationships with surrounding communities continue, it seems unlikely that illegal killing of elephants will provide justification for intervention. Localised elephant densities are commented on later whilst discussing closure of artificial waterpoints (6.3.1.4), as well as observed impacts on vegetation. The impacts of the elephants of MWR on other species and people has not been well documented and such data could be collected in newly implemented monitoring schemes. To the authors knowledge, only three studies have documented the impact of MWR's elephants (and other species) on the vegetation in MWR (Staub *et al.* 2013; Weinand 2013, Geenen, 2019) and a better understanding of habitat sensitivity and uniqueness should probably form a third component of monitoring to inform intervention policy decisions. As a non-profit organisation African Parks relies largely on donor funding (pers. comms. African Parks); although tourism revenues have been increasing in MWR, the costs of running and maintaining such a reserve still requires additional sources of funding. The minimal costs associated with the van Jackson way would entail the employment of a full-time staff member to conduct and manage the appropriate monitoring schemes, which need not be restricted to elephants or their effects.

<u>The above strategy is strongly recommended.</u>

6.3.1.2. Metapopulation Management - Translocations

Although translocation remains a management option, it is not a desirable way forward, as translocations are invasive, expensive, stressful and disruptive. Translocations have been the primary management tool for the MWR elephant population, however MWR has acted as the sink in past translocations, with the 2017 event seeing MWR as the source for the first time. African Parks' PEM mentions the major limitations to translocations as being time-consuming and expensive, as well as a lack of suitable sink destinations (van Aarde & Jackson 2007). The policy does go on to state that if opportunities do become available, translocations should be considered if the following two considerations are adhered to:

"First, it is important not to remove all the older bulls as the need for dominance in the hierarchy is most important; and second, family groups must all be removed together and should not be split up."

Reflecting on the criteria for intervention assessments, many of the same concerns regarding our predictive ability with our current knowledge for MWR elephants also apply here. According to national policies and objectives (NEAPW 2015), translocations would only be permitted within Malawi; although this may lower the costs it also limits the number of suitable sink destinations. Liwonde National Park and Nkhotakota Wildlife Reserve are both unlikely to be appropriate destinations, as LNP also seeks to lower its elephant densities and NWR was the recipient of the 2017 translocation. Should it be deemed that NWR is capable of housing more elephants, this would make it a viable destination. However, such a determination requires further monitoring, and another translocation of elephants from MWR to NWR under current knowledge levels would be careless.

Since the population growth uncertainty makes it unclear when and if further translocations might become necessary, it seems prudent to invest in strategies that minimise the likelihood of translocations becoming necessary. These strategies are discussed below.

<u>Another translocation of elephants out of MWR is strongly advised against on the grounds of the extremely disruptive nature of translocation events.</u>

6.3.1.3. Immunocontraception

Contraception management strategies, which entail the manipulation of the fertility in elephant, generally utilise two main methods: steroids and immunocontraception (Rogers & Sherwill 2008). Of the two main methods, immunocontraception has shown greater rates of success than steroids, with results in the Kruger National Park, South Africa, varying between 60 to 80% (Rogers & Sherwill 2008). Very simply, immunocontraception employs a foreign protein (porcine zona, pZP) to immunize cow elephants, by encouraging antibodies that prevent fertilization (Fayrer-Hosken *et al*. 2000; Rogers & Sherwill 2008). The pZP is preferred over steroids because it reports no deleterious health effects, is safe to use during pregnancy, is reversible and does not generate behavioural abnormalities in targeted females (Rogers & Sherwill 2008). The use of helicopters to deliver the darts raised concern, as it was suggested that these deployment periods would be too stressful for elephant social groups. However, Druce *et al*. (2012) concluded that the disruption effect of immunocontraceptive darting on family groups was minimal, but their sample size was extremely small with only five family groups in the population.

Despite the advantages of a contraception-based management strategy i.e. lengthening inter-calving intervals and increasing the age of first calving (Pimm & van Aarde 2001), there are notable disadvantages. Contraceptive methods are unable to reduce elephant populations in the short-term, and contraceptive management is correspondingly expensive and time consuming, becoming unmanageably so for large elephant populations (Whyte *et al*. 1998; Pimm & van Aarde 2001; van Aarde & Jackson 2007; Rogers & Sherwill 2008). Although MWR's elephant population is currently relatively small, the expense of this operation will add to the Park's budget, and must be repeated across years; to achieve the current goal of 0% growth. Immunocontraception models have demonstrated that contraception of 75% (36) of the breeding-age females with an annual mortality rate of 2-3% is sufficient (Delsink & Kirkpatrick 2012). However, this goal may not be reached until 11 years after the initial implementation of a contraception program (Whyte 1998). Furthermore, annual rates of increase are too simplistic a measure for the complexity of elephant population dynamics, and management needs better data tools to base good decisions on. Elephant populations are not static and rather than absolute numbers, a more realistic goal is resilience - allowing elephant populations to grow and decline in response to environmental variation. This becomes particularly important in the face of climate change and associated changing disease dynamics that pose significant threats to southern African elephant populations.

A contraceptive management strategy was recommended after MWR's previous detailed elephant demographic study (Forrer 2017), which has yet to be implemented. African Parks' PEM goes on to state that it is "firmly in favour" of the use of immunocontraceptive methods in a Malawian context as to pre-emptively manage elephant population growth rates. The author is in complete disagreement with the PEM as (1) the manipulation of AWPs would be able to achieve a similar result at a much lower cost or risk and (2) the life-history information such as the fertility rates, inter-calving intervals and mortality rates of MWR's elephant population remain unknown, and render immunocontraceptive models powerless (Whyte *et al*. 1998; van Aarde & Jackson 2007; Delsink & Kirkpatrick 2012). Collection of such data is absolutely necessary before any contraception programme could be developed for the elephants of MWR. Finally, the use of immunocontraceptives may well counteract the DNPW Malawi goal of achieving a national population of 22 000 elephants (NEAPW 2015).

<u>Much more detailed data on the MWR elephant population is needed before this option could be implemented sensibly.</u>

6.3.1.4. Self-Regulation through Density Dependence - Restriction of Artificial Water Points

Within large, water scarce reserves, the number and positioning of AWPs have the ability to manipulate local elephant densities (Chamaille-Jammes *et al.* 2007). Areas located away from any permanent water sources limit the foraging range of elephants in the dry season, potentially shielding tree species in these areas (Rogers & Sherwill 2008; Loarie *et al.* 2009). Since calf mortality is heavily mediated by access to water (Moss *et al.* 2011, CH 12) population growth rates decrease as water becomes restricted, and foraging ranges decrease in dry seasons when females are tied to restricted water sources (Duffy *et al.* 2002; Owen-Smith *et al.* 2006; Rogers & Sherwill 2008). The net effect is slower population growth and more marked seasonal movements of elephants.

Majete has 10 AWPs, with four located in the Sanctuary (Region 1), four in the Pende area (Region 2) and two in the Western region (Region 3). The Sanctuary has two perennial rivers that define the northern and eastern boundaries. There is also at least one natural spring within this region. The sanctuary is a water rich area throughout the calendar year. Previously, it had been suggested that at least one of the AWPs within the Sanctuary region be closed, as elephant habitat use within the region had become homogenized due to the overabundance of perennial water sources (Forrer 2017; Geenen, 2019). This recommendation is repeated here, with the specific mention of Nsepete Waterhole, as well as the Heritage waterhole, which are relatively close to the Mkhulumadzi River and Shire River, respectively. While the closure of these two AWPs may not be sufficient to influence population growth rates, as there are still numerous water sources within the Sanctuary, it may start to reverse the process of habitat homogeneity. Furthermore, it is recognized that MWR relies heavily on income generated via tourists visiting the park to see its wildlife, this is especially true during the dry season when game viewing is almost guaranteed at the AWPs. Game-viewing experiences could potentially be negatively impacted by restricting tourists to a small number of AWPs. However, with ample tourist roads along the rivers as well as throughout the Sanctuary in general, it is believed that closing one waterhole for an extended period of time and rotating them will have minimum impact on the tourist's experience, but may help in restoring habitat heterogeneity to the area. The vegetation around these waterholes has been drastically transformed, with piosphere effects in place (Figure 6.1).

Figure 6.1. The Pende1 waterhole in Region 2 and the transformation of vegetation around it due to almost continuous animal traffic. Photo taken by W. Hartmann at an altitude of 100 m.

The closure of AWPs in the Pende region would certainly have an effect on the foraging range, and subsequent calf mortality rates, in the dry season. Should the goal be to slow population growth rates, it is highly recommended at least one of the following waterholes be closed: Pende1, Pende2, Nthumba, and/or Kakoma. There are very few naturally occurring year-round water sources in this region and limiting the number of AWPs would essentially reinstate natural ecological limitations on calf survival and population growth by re-coupling the limitations of water and available forage, forcing elephants to travel further for water in dry seasons.

It has previously been suggested that the Sanctuary be designated as a high elephant impact zone due to the numerous sources of perennial water and tourists demands, and that other regions within MWR (i.e. Pende and the Western region) designated as low elephant impact zones (Forrer 2017). Limiting the number of water sources in these low impact zones and ensuring the functional AWPs to be no closer than 10 km apart is recommended to reduce elephant habitat use in these areas (Weinand 2013), and a management plan which strives for habitat heterogeneity with regards to elephant herbivory across MWR should be implemented. Such a plan would promote systems, such as regulating the position and number of operational AWPs, to ensure MWR is a mosaic of different elephant browsing levels in differing habitat types (Weinand 2013). Before the implementation of such a plan, further research is required, specifically investigating whether the proposed areas above are indeed suitable for the proposed impact-levels of elephant use.

Such a strategy is in-line with African Parks' PEM, where it is recognized that in order for an elephant population to self-regulate, an environment that places pressure on the population must be created by management (Joubert 2005). The historical environment seen throughout MWR, particularly in the Sanctuary, has not been one of pressure, and thus, the elephant population could not have been expected to self-regulate. However, the removal of only breeding-age females and dependent offspring has resulted in drastically fewer opportunities for males to mate with females, therefore increasing male-male competition and potentially creating an environment of escalated social pressure (see below).

<u>This option allows a low-cost low-risk exploration of restoring the connection between environmental conditions and elephant ranging and population dynamics that has the potential to be extremely beneficial to MWR.</u>

6.3.1.5. Summary

The strategies discussed above are summarized into the following recommendations below.

The Policy of Elephant Management set out by African Parks emphasizes the need to promote heterogeneity within its parks. It aims to achieve this heterogeneity by employing the direct manipulation of resource availability, through the provision of water (i.e. artificial water points), as a means to change and circumvent spatiotemporal elephant impacts. Both African Parks and the author recognize that such a strategy will require close monitoring of both the system and other species to ensure that heterogeneity and biodiversity are not unintentionally compromised. Simply put, close monitoring of the current elephant population and surrounding systems are required in combination with the closure of certain AWPs. A further possible strategy would be to translocate a number (~ 50) of males out of MWR, within a reasonable time period (~ 2-3 years), in combination with the regulation of AWPs. This would balance the adult sex ratio, whilst not influencing the growth rates since the number of breeding-age females remains constant. Lower densities might promote population growth, but that would be curbed by the pressure applied through the closure of AWPs. Finally, while contraception is technically an option, it is stated in African Parks' PEM that contraceptive measures are an option only if self-regulatory means and metapopulation management strategies are not possible. In addition, contraception would be difficult to be approved in MWR as the DNPW of Malawi would see it as a hindrance to their goal of 22 000 elephants (NEAPW 2015).

6.3.2. Translocation impacts on the utilization of artificial water points by the remnant elephant population

The Sanctuary appears to be a favoured region by the elephants of MWR. With ample forage and numerous perennial water sources this is not surprising (Weinand 2013). Although many herds were removed from the Sanctuary, it appears as if elephants from the surrounding regions are beginning to move back into this area. If left the way things currently are (i.e. all current AWPs continue to function as they have), it is likely that the elephant density and IOU will be at pre-translocation levels within the next year. This may be desirable for management, as it has previously been stated that the Sanctuary is an area within MWR with an extensive road network for game drive vehicles and tourists opting for the self-drive option.

It has previously been suggested, both in this paper and others (Weinand 2013; Forrer 2017) that the Sanctuary be designated as a high impact elephant zone. The management of MWR has to balance the ecological requirements of the reserve, as well as the needs of the tourists. Tourism plays an essential role in funding MWR, and so ensuring tourists are still able to view wildlife is a fundamental requirement of any plan that is to be enacted. However, regions which are not frequently visited by tourists, such as Pende (Region 2) and Pwadzi (Region 3), should then be designated as medium to low impact elephant zones. The goal should be to create a heterogeneous landscape with regards to elephant impact. This has been suggested above by regulating the number of AWPs in these tourist-scarce areas.

Chapter four demonstrated that large-scale anthropogenic events have significant effects on the distribution of elephants. It was found that due to the removal of many herds predominantly from Region 1, elephant herds from surrounding regions began to fill the 'vacuum' created. This is important for future translocations of elephants. If an area is of suitable habitat for elephants, and the density of elephants decreases (due to the artificial removal of elephants), it should be noted by management that elephants from the surrounding areas are likely to move into that space. If the goal is to decrease elephant densities of a certain area permanently, then the removal of individuals will not work. Other options, such as the limiting of water in the area or fencing should be considered.

Furthermore, translocations will undoubtedly disrupt well-established social hierarchies. The social dynamics of an elephant population should first be well understood before the arbitrary removal of herds and/or individuals, as the removal of key individuals or herds can be a stressor for remnant elephants (Gobush *et al.* 2008; Silk *et al.* 2010; IUCN, 2013). Individuals were predominantly removed in the 2017 translocation from the Sanctuary and Pende areas due to the road network providing ease of access for large transport trucks. No consideration was given to the social dynamics prior to the removal of certain herds. Additionally, it has been found that certain herds were split-up during the translocation event which is in direct opposition to what African Parks has stated in their Policy on Elephant Management.

Moving forward, a serious effort should be made to understand the social landscape of an elephant population, as to best ensure minimal disturbance for an already extremely disruptive event. Given the complexities of male social dynamics that centre around male-male competition, the severe male bias in the sex ratio caused by the premediated removal of only breeding herds during the 2017 translocation is of serious concern. The extreme lack of access to breeding age females now present in MWR may well intensify interactions between the breeding males of MWR. Large males (generally over the age of 35 years of age) guard females during mid-oestrus to ensure the best opportunities for reproduction, while small to medium males gain access only during early- and late-oestrus (Poole 1989). The 2017 translocation has drastically altered not only the demographics of MWR's elephant population, but the subsequent social landscape for the remaining elephants. As suggested in section 6.3.2.5, the removal of males from MWR may rebalance the social landscape, and with the regulation of AWPs allow for a 'natural' means of managing a population in 'unnatural circumstances'.

6.3.3. The use of drones to observe African elephants aerially

The use of drones, both commercially and scientifically, has increased rapidly over the last decade. This is supported by the increasing number of review papers on drones and their uses in the wildlife and ecological sciences (Jones *et al.* 2006; Allan *et al.* 2015; Ivoševi *et al.* 2015; Christie *et al.* 2016; Hodgson & Koh 2016). Drones have great potential in the field of wildlife sciences and ecology by providing advancements in aerial imagery (Shahbazi *et al.* 2014), aiding in species distribution and abundance surveys (Vermeulen *et al.* 2013), as well as assisting in general, large-scale conservation efforts (Koh & Wich 2012; Mulero-Pázmány *et al.* 2014). However, there are challenges, ranging from practical to logistical to governmental, that face the enthusiastic-drone-loving-ecologist (Linchant *et al.* 2015; Vincent *et al.* 2015). A large concern for many researchers and conservation managers alike, is that of the effect drones have on the animals themselves, and despite this appearing as the most critical issue, the studies attempting to quantify an animal's response to a drone are few and far between (Ditmer *et al.* 2015; Pomroy & Conner 2015; Vas *et al.* 2015). This study aimed to provide drone pilots with an ethical protocol to follow when observing elephants aerially by quantifying how elephants respond to various flight patterns.

If the following protocols are utilized, this study recommends the use of drones for future elephant observational studies, counts and for cinematic purposes. If a person(s) wishes to use a drone within MWR, the following should be done:

1) A request stating the explicit purpose of utilizing the drone should be submitted to MWR management, specifically the field operations manager (FOM) and reserve manager (RM).
2) Upon approval by the FOM and RM, the protocol and guidelines below should be distributed to all drone pilots and signed, regardless of the drone pilot's intention (i.e. scientific or cinematic).

3) Drone pilots should be required to log and submit all flights. Generally, drone flights are automatically logged by the drones themselves, and this flight information should be required by management in order to prevent the abuse of this aerial freedom.
4) Management should hold the right to view the data/footage captured by the drone at any time. Likewise, management should have the ability to revoke the drone pilot's ability to fly within the reserve at any point.

This study found that by utilizing certain flight protocols, it was possible to get within 30 m of both male and female African elephants. Thirty meters was chosen as the closest distance from the elephant as most modern drones are equipped with a 2x digital zoom which enables general behavioural observational studies, demographic studies and studies investigating dietary preferences (i.e. identifying the species of plant elephants are feeding on [Figure 6.2; Figure 6.3]). If a study wanted to investigate family-herd dynamics, the drone might have to first identify all individuals of interest (using unique notches on the ears of the elephants and/or tusk orientation [Figure 6.2]) and be at the 30 m threshold, before retreating to a higher altitude and observing the now identified individuals all at once (Figure 6.4).

Figure 6.2. A lone adult bull feeding in the Pende Region. Distinct ear notches and tusk orientation easily identifiable. Photograph taken by W. Hartmann at a height of 25 m and a ground distance of 15 m utilizing the 2x digital zoom on the Mavic Pro. Platinum.

Figure 6.3. Two young bulls stripping a tree of its bark in the Sanctuary Region. Photograph taken by W. Hartmann at a height of 35 m and a ground distance of 18 m. Not zoomed in.

Figure 6.4. A mixed herd feeds in the Sanctuary Region during the wet season. Photograph taken by W. Hartmann at a height of 50 m.

The aims and objectives of future studies wishing to use drones to observe elephants should be clearly stated before deciding on a specific flight protocol to use. However, the results of this study provide a broad, basic protocol which should be followed by both scientist and civilian:

6.3.3.1. Approach Protocol

When approaching an elephant or elephants, a slow speed of 2 m/s is advised. Slower speeds yielded the least amount of disturbance to the elephants droned. It is thought that the slow approach speed enables elephants to determine/decide if the approaching drone is a threat or not. Utilising an approach angle of 45° is desirable. At this angle of approach, elephants were able to lift their heads and make visual contact with the drone. Steep approach angles (i.e. 90°) prevented elephants from being able to visually identify the source of the noise and subsequently resulted in higher levels of distress. While starting altitude appeared to have no significant effect on elephant response, it is advised to start at 50 m or higher, as at this height, elephants are easy to locate as well as the drone being well clear of the canopy. Thus, the drone pilot need not worry about crashing. One hundred meters is also adequate, however, elephants in dense vegetation are challenging to locate.

Although there were no differences in successful approach rates between bulls and breeding herds, it is advised that the above protocol be strictly followed when droning herds as calves and infants were often witnessed retreating into thicker vegetation. However, this did not influence the surrounding adults' behaviour.

6.3.3.2. Presence Protocol

Although no flight or environmental variable had a significant impact on the success of a presence flight, the general trend was that slower speeds resulted in less disturbed elephants. It is therefore recommended that a flight speed of 2 m/s be used whenever and wherever possible. Generally, a flight pattern with a fixed altitude resulted in the least disturbance, and it is therefore advised that once the target altitude (no less than 30 m) has been reached, the drone is kept at that height throughout the study. However, as mentioned above, certain studies may require various altitudes throughout one flight. This is permitted, as no significant difference was found between a varied and fixed pattern, but it is advised that transition between heights be conducted smoothly (i.e. at slower speeds).

Importantly, the success of the preceding approach was found to have an extremely significant influence on the success of a sustained presence flight. It is therefore recommended that the approach protocols are always strictly adhered to, as to prevent the initial disturbance and subsequent sustained disturbance of the elephants being droned. Should the elephant(s) be disturbed upon approach, it is unlikely they will return to an undisturbed state throughout the rest of the droning session. Should this occur, it is recommended that the droning session be cancelled for that/those elephant/elephants and only attempted again the following day.

6.3.3.3. General Guidelines

Elephants, although the focus of this study, were not the only animals within MWR. A variety of responses by different species to the drone were noted. Some, such as the warthog, seemed completely unphased by it, whereas others, such as baboons, hippos and antelope, appeared incredibly sensitive to the drone's presence, even if it was merely flying over them.

It is therefore advised that the above protocols (a slow speed and height of at least 50 m) be adhered to whilst searching for elephants with a drone, as potential bystanders may be present below. Additionally, birds of prey may engage the drone. A successful defence mechanism is to simply ascend vertically at a rapid pace. However, the subsequent noise of the drone ascending rapidly could disturb the target elephants and thus, bring about a premature end to the droning session. It is therefore advised that the drone pilot has an assistant to watch the skies as the flight is conducted. This is only possible when the target elephants are within a reasonable distance of the drone pilot (~ 300 m), or if the pilot and assistant are at an elevated altitude themselves.

The alternative is to continue on as the way things currently are, which varies between reserves. Some have complete bans on any drone flying whatsoever, resulting in many opportunities lost, whereas others are more liberal, such as MWR, and allow drones to be flown for research and cinematic purposes. However, by implementing the recommendations above, the process for standardizing drone flights by appropriate personnel within protected areas has begun. Drones are here to stay and can be a great tool for the ecologist, filmmaker and reserve manager, so it is best we understand and enforce how to fly them ethically.

6.4. Conclusion

1. Majete Wildlife Reserve urgently needs monitoring resources to underpin good management decisions for its habitat heterogeneity and especially its elephant population
2. The preferred options for elephant management should be a focus on gathering data on reproductive status through regular monitoring, to allow evaluation of options for immunocontraception and translocation in the future, providing these do not counter National goals for elephant populations
3. Manipulation of AWPs offers a cheap, reversible and low-risk way to test how elephant populations may self-regulate without the need for expensive, disruptive interventions
4. The extreme male bias in the adult ratio requires special attention in monitoring to assess the potential for deleterious social effects of extreme male-male competition
5. MWR elephants tolerate drones well, despite no prior exposure. This can be a vital tool for monitoring, and all drone pilots should follow responsible protocols as set out here to ensure the tool remains available for monitoring, research or commercial (film) use.

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Appendices

Appendix 1

Appendix 1. A table representing the number of family herds and solitary bulls in MWR and their total number, sex and age structure in the years 2018 and 2019.

Code	Group Type (Family Herd, Solitary Bull)	Total Number of Individuals	Number of Large Adult Males	Number of Large Adult Females	Number of Medium Adult Males	Number of Medium Adult Females	Number of Small Adult Males	Number of Small Adult Females	Number of Male Juveniles	Number of Female Juveniles	Number of Unknown Juveniles	Number of Male Calves	Number of Female Calves	Number of Unknown Calves	Number of Male Infants	Number of Female Infants	Number of Unknown Infants
C1_C2_C10a_C11a_C13_C41	Family Herd	13	0	2	0	3	0	1	0	1	0	1	2	1	0	0	2
C19	Family Herd	3	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1
C21	Family Herd	3	0	0	0	1	0	0	1	0	0	1	0	0	0	0	0
C37_C38	Family Herd	5	0	1	0	1	0	1	0	0	0	1	1	0	0	0	0
C1a	Family Herd	3	0	0	0	1	0	0	1	0	0	0	0	1	0	0	0
C2a	Family Herd	4	0	0	0	1	0	0	1	1	0	0	0	1	0	0	0
C3a	Family Herd	2	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0
C4a	Family Herd	5	0	1	0	0	0	0	1	1	0	0	0	1	1	0	0
C6a	Family Herd	3	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0
C7a	Family Herd	2	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0
C9a	Family Herd	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
C10a	Family Herd	2	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0
C11a	Family Herd	2	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0
C12a	Family Herd	2	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0

PwC1_PwC29	Family Herd	7	0	1	0	0	0	1	0	1	0	1	1	1	0	0	1
PwC2	Family Herd	3	0	0	0	1	0	0	0	0	0	0	0	1	0	0	1
PwC3	Family Herd	6	0	0	0	1	0	1	0	0	0	1	2	0	0	1	0
PwC5_PwC6_ PwC15	Family Herd	11	0	0	0	2	0	1	2	1	0	1	2	0	0	1	1
PwC7_PwC8	Family Herd	7	0	1	0	1	0	0	0	1	0	1	1	1	0	0	1
PwC11_PwC14	Family Herd	6	0	1	0	1	0	0	0	1	1	0	1	0	0	0	1
PwC12	Family Herd	4	0	1	0	0	0	0	0	1	0	0	0	2	0	0	0
PwC18	Family Herd	3	0	0	0	1	0	0	0	1	0	0	0	1	0	0	0
PwC19_PwC22	Family Herd	5	0	0	0	1	0	1	0	0	0	1	0	1	0	0	1
PwC23	Family Herd	3	0	0	0	1	0	0	0	1	0	0	1	0	0	0	0
PwC24	Family Herd	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
PwC25_PwC26_ PwC27_PwC30	Family Herd	11	0	2	0	1	0	1	2	1	0	3	0	0	0	0	1
PwC28	Family Herd	2	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0
PeC1_PeC2	Family Herd	6	0	1	0	1	0	0	0	2	0	2	0	0	0	0	0
PeC5	Family Herd	4	0	0	0	1	0	0	2	0	0	1	0	0	0	0	0
PeC13	Family Herd	4	0	0	0	1	0	0	0	1	0	0	0	1	0	0	1
Di C1 - same as Pe C2	Family Herd	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Di C4	Family Herd	3	0	0	0	1	0	0	0	1	0	0	0	1	0	0	0
B1	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B2	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B3 = B11	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0

B4	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B5	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B6	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B7	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B9 = B49	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B10 = B36	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B11 = B3	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B12	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B13	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B14 = B21	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
B15 = B48	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B16	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B17	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B18	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B19 = PeB9	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B20	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B21	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B22	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B23	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B24 = B31	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B25	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B26	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B27	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0

B29	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B31 = B24	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B32	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B33	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B34	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B35	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B36 = B10	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B37	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B38	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B40	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B43	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B45	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B46	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B47	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B48 = B15	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B49	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B50	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B51	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B52	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B2a	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B3a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B5a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B6a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0

B7a	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B8a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B10a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B11a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B12a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B13a	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B14a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B17a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B19a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B20a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B21a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B25a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B27a	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B28a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B29a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B30a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B31a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B32a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B33a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
DiB2	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PwB2	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PwB3	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PwB4	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0

PwB7	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PwB8	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PwB9	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PwB10	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PwB11	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PwB12	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PwB13	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PwB14	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PwB15	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PwB16	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PwB17	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PwB18	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PwB19	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PwB20	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PwB21	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB1	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PeB2 = B3	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PeB3	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PeB4	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PeB5	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB6	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PeB7	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PeB8	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0

PeB9 = B19	Bull	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PeB10	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PeB14	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB15	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PeB17	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PeB1a	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
PeB2a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB3a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB5a	Bull	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PeB6a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB7a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
PeB10a	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B53	Bull	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
B54	Bull	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Total		**236**	**19**	**13**	**39**	**24**	**42**	**12**	**11**	**15**	**1**	**15**	**13**	**15**	**3**	**3**	**11**

Appendix 2

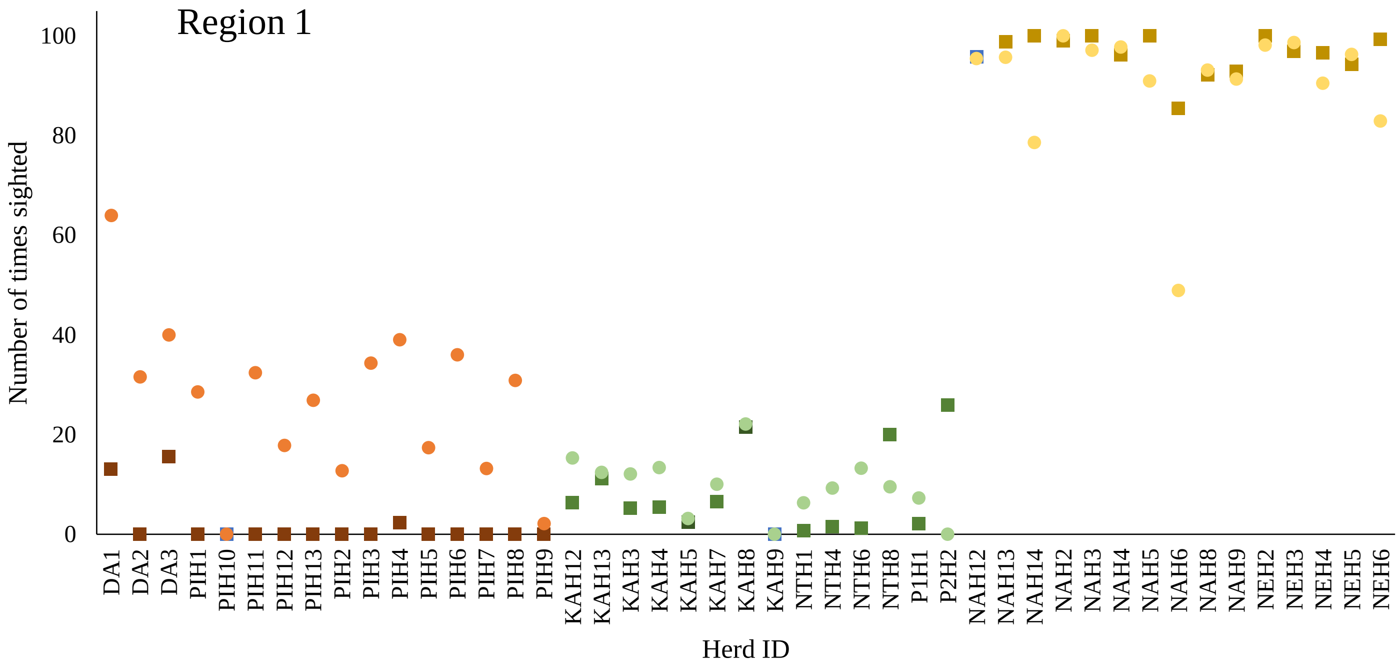
Appendix 2A. A list of all approach flight variables and their possible combinations and subsequent code.

Speed ($m.s^{-1}$)	Abbreviation	Angle of Approach (°)	Abbreviation	Initial Altitude (m)	Abbreviation	Code	Number
2	S	45	A	35	L	SAL	1
2	S	45	A	50	M	SAM	2
2	S	45	A	100	H	SAH	3
2	S	90	N	35	L	SNL	4
2	S	90	N	50	M	SNM	5
2	S	90	N	100	H	SNH	6
4	M	45	A	35	L	MAL	7
4	M	45	A	50	M	MAM	8
4	M	45	A	100	H	MAH	9
4	M	90	N	35	L	MNL	10
4	M	90	N	50	M	MNM	11
4	M	90	N	100	H	MNH	12
6	F	45	A	35	L	FAL	13
6	F	45	A	50	M	FAM	14
6	F	45	A	100	H	FAH	15
6	F	90	N	35	L	FNL	16
6	F	90	N	50	M	FNM	17
6	F	90	N	100	H	FNH	18

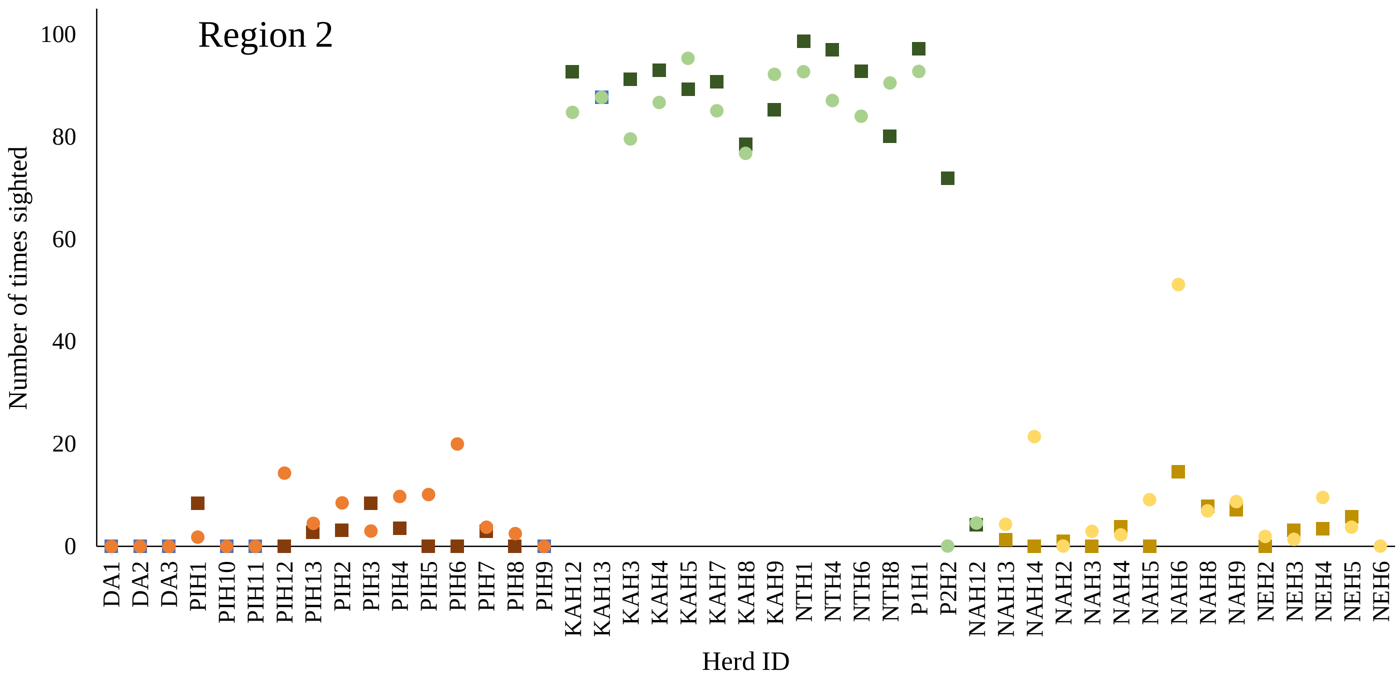
Appendix 2B. A list of all presence flight variables and their possible combinations and subsequent code.

Speed ($m.s^{-1}$)	Abbreviation	Flight Pattern	Abbreviation	Code	Number
2	S	Fixed	F	SF	1
2	S	Varied	V	SV	2
4	M	Fixed	F	MF	3
4	M	Varied	V	MV	4
6	F	Fixed	F	FF	5
6	F	Varied	V	FV	6

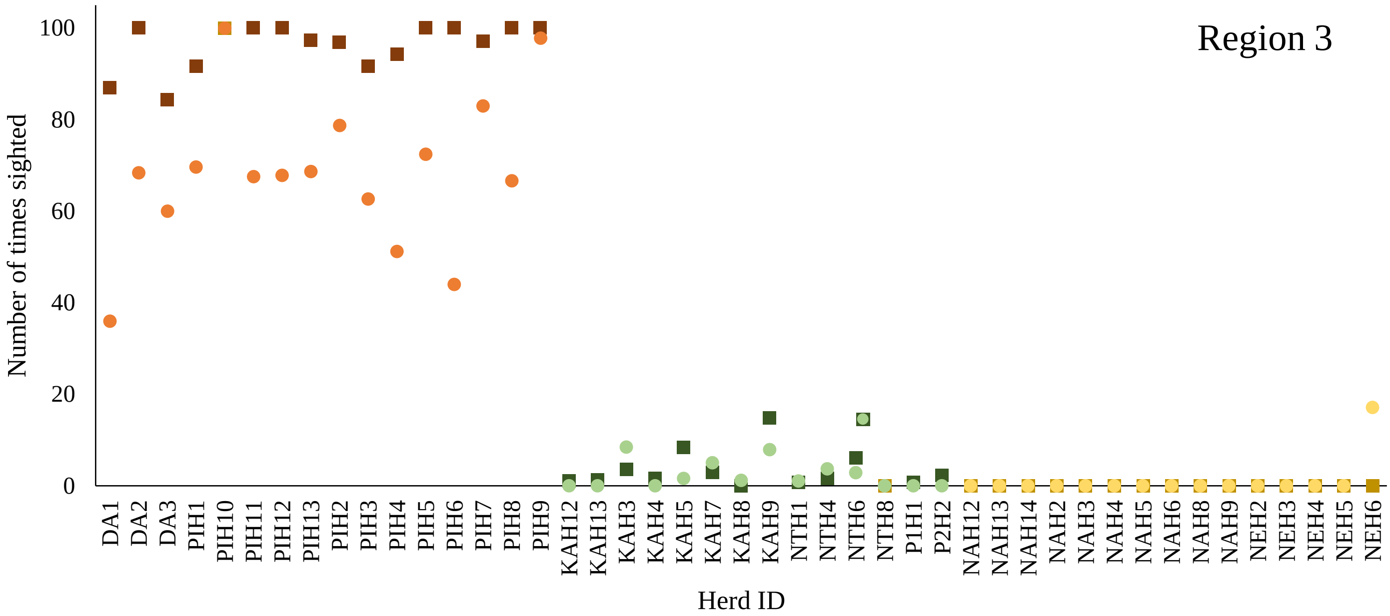
Appendix 3



Appendix 3A. The sighting frequencies of herds not removed in 2017 in Region 1. The darker square points represent the total number of times that herd was sighted pre translocation (January 2016 – June 2017). The lighter circle points represent the total number of times that herd was sighted pre translocation (July 2017 – April 2019). The red/orange points represent herd classified as Region 3 herds. The green points represent herds classified as Region 2 herds. The gold/yellow points represent herds classified as Region 1 herds.



Appendix 3B. The sighting frequencies of herds not removed in 2017 in Region 2. The darker square points represent the total number of times that herd was sighted pre translocation (January 2016 – June 2017). The lighter circle points represent the total number of times that herd was sighted pre translocation (July 2017 – April 2019). The red/orange points represent herd classified as Region 3 herds. The green points represent herds classified as Region 2 herds. The gold/yellow points represent herds classified as Region 1 herds.



Appendix 3C. The sighting frequencies of herds not removed in 2017 in Region 3. The darker square points represent the total number of times that herd was sighted pre translocation (January 2016 – June 2017). The lighter circle points represent the total number of times that herd was sighted pre translocation (July 2017 – April 2019). The red/orange points represent herd classified as Region 3 herds. The green points represent herds classified as Region 2 herds. The gold/yellow points represent herds classified as Region 1 herds.