

Ecological Impact of Large Herbivores at Artificial Waterpoints in Majete Wildlife Reserve, Malawi

By

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Declaration

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December 2019

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Abstract

The management of water requires an important understanding of the impacts of supplementation on the ecosystem. In semi-arid environments worldwide, access to water is essential for the survival of wildlife and may affect the distribution of species within a reserve. Water supplementation, via artificial waterpoints (AWPs), has become common practice in maintaining wildlife densities when other water sources are limited in the dry season. Herbivores, especially those which are water-dependent, tend to congregate near water points and this congregation results in increased trampling and intense herbivore utilisation of the surrounding vegetation, eventually forming gradients of utilisation with increased utilisation pressure close to the AWP. Majete Wildlife Reserve (MWR) in southern Malawi has undergone a rehabilitation phase, which included a large wildlife reintroduction programme and as a necessity the construction of 10 AWPs. Concerns have now arisen as to the impact herbivores may be having on the vegetation surrounding these water points. In this study, vegetation change and structure surrounding AWPs and wildlife AWP usage was investigated.

AWP utilisation by 12 species was studied with the use of camera traps from June 2017 to May 2018. African elephant (*Loxodonta africana*) and Cape buffalo (*Syncerus caffer*) were the most frequently observed species. AWPs were predominantly visited from 10:00 - 11:00, apart from black rhinoceros (*Diceros bicornis minor*) and Cape buffalo which preferred to utilise AWPs at dawn and dusk. Seventy-nine percent of AWP use observations were in the dry season, compared to 21% in the wet season. AWP utilisation was significantly higher in the south-eastern region of the reserve due to a lack of natural perennial water sources. Vegetation was sampled along three 2000m long transects in the N, SE and SW directions from six AWPs between July and November 2017 for woody vegetation, and between February and May 2018 for herbaceous vegetation. Both herbaceous vegetation condition and woody vegetation cover increased significantly with distance from water. Fixed-point photography was used to determine the change in vegetation cover between 2013 and 2018. Eight of the ten AWPs had an average decrease in vegetation cover. Woody vegetation cover surrounding AWPs decreased over the five-year period primarily due to elephant activity, and there was a coincided decrease in herbaceous vegetation cover. Soil erosion surrounding AWPs increased, due to an increase in water run-off and a decrease in soil stability when herbaceous vegetation cover decreased. AWPs are an important source of water for herbivores in the dry season in MWR, however the planning of AWP placement, maintenance and management options require careful consideration.

Opsomming

Die bestuur van water vereis 'n goeie begrip van die gevolge van sodoende aanvulling op die ekosisteem. In semi-droë omgewings wêreldwyd is toegang tot water noodsaaklik vir die oorlewing van wilde diere en dit kan die verspreiding van spesies binne 'n reservaat beïnvloed. Wateraanvulling, deur kunsmatige waterpunte (AWPs), het gereelde praktyk geword in die handhawing van digtheid van wild in die natuur as ander waterbronne in die droë seisoen beperk is. Herbivore, veral waterafhanklike spesies, is geneig om naby waterpunte te vergader en hierdie gewoonte lei tot verhoogde vertrapping en oorbeweiding van die omliggende plantegroei, wat uiteindelik gradiënte van benutting vorm met 'n verhoogde gebruiksdruk naby die AWP. Majete Wildlife Reserve (MWR) in die suide van Malawi het 'n rehabilitasiefase ondergaan, wat 'n groot hervestigingsprogram vir wild insluit en word gesien as 'n noodsaaklikheid vir die oprigting van 10 AWP's. Daar is nou kommer oor die impak wat herbivore op die plantegroei rondom hierdie waterpunte kan hê. In hierdie studie is plantegroei-verandering en -struktuur rondom AWP's en AWP-gebruik deur diere ondersoek.

Die gebruik van die AWP deur twaalf spesies is bestudeer met behulp van 'n kameratrap van Junie 2017 tot Mei 2018. Afrika-olifant (*Loxodonta africana*) en Kaapse buffels (*Syncerus caffer*) was die soorte wat die meeste voorkom. Afgesien van swartrenosters (*Diceros bicornis minor*) en Kaapse buffels, wat verkies om AWP's teen dagbreek en skemer te gebruik, is AWP's oorwegend van 10:00 - 11:00 besoek. Nege-en-sewentig persent van die AWP-waarnemings was in die droë seisoen, in vergelyking met 21% in die nat seisoen. AWP-benutting was aansienlik hoër in die suidoostelike gebied van die reservaat weens 'n gebrek aan natuurlike permanente waterbronne. Plantegroei opnames is onderneem op drie transekte van 2000 meter in die N-, SO- en SW-rigting vanaf ses AWP's tussen Julie en November 2017 vir houtagtige plantegroei, en tussen Februarie en Mei 2018 vir kruidagtige plantegroei. Bedekking van beide kruidagtige plantegroei en houtagtige plantegroei neem aansienlik toe met die afstand van water.. Vastepuntfotografie was gebruik om die verandering in plantegroei tussen 2013 en 2018 te bepaal. Agt van die tien AWP's het 'n gemiddelde afname in plantbedekking gehad. Houtagtige plantegroei rondom AWP's het gedurende die vyfjaarperiode afgeneem, hoofsaaklik as gevolg van olifantaktiwiteit, en daar was 'n afname in die plantbedekking van plante. Wanneer die kruidagtige plantegroei bedekking laag was, het gronderosie rondom AWP's toegeneem, as gevolg van 'n toename in die afloop van die water en 'n afname in grondstabiliteit. AWP's is 'n belangrike bron van water vir herbivore in die droë seisoen in MWR, maar die beplanning van AWP se plasing, instandhouding en bestuur moet noukeurig oorweeg word.

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Chapter One

General introduction and thesis outline

1.1 Introduction

Large herbivores have a substantial effect on vegetation and ecological processes, which includes altering the composition and structure of their surrounding landscapes (De Garine-Wichatitsky, Fritz, Gordon, & Illius, 2004). Herbivores cause disturbance within ecosystems as they utilise the landscape and are therefore, a core focus in conservation (Fuhlendorf, Engle, Kerby, & Hamilton, 2008; Maron & Crone, 2006; Riginos & Grace, 2008; Schippers et al., 2014). Large herbivores such as elephant (*Loxodonta africana*), buffalo (*Syncerus caffer*), giraffe (*Giraffa camelopardalis*) and zebra (*Equus quagga*) continue to contribute to structural ecological changes present in a variety of African landscapes (Birkett & Stevens-wood, 2005; De Garine-Wichatitsky et al., 2004; Hamandawana, 2012; Morrison, Holdo, & Anderson, 2016; Mosugelo, Moe, Ringrose, & Nellemann, 2002; Riginos & Grace, 2008; Skarpe, 1990). Therefore, understanding the fundamental factors affecting herbivore abundance and distribution is an important aspect for optimal management (De Garine-Wichatitsky et al., 2004). Important resources including food and water determine the distribution of herbivores and influence their seasonal movements (Seydack et al., 2012).

As a vital resource, water is able to influence wildlife behaviour, distribution and ecology within various ecosystems (Hayward & Hayward, 2012). Historically, herbivores relying on natural perennial water sources, migrated between forage resource areas according to water availability during the critical dry months in order to maintain stable populations (Epaphras et al., 2007; Grant, Peel, & Bezuidenhout, 2011; Walker, 1979). Therefore, many conservancies have resorted to supplementing water in areas with little perennial surface water (Owen-Smith, 1996; Smit, Grant, & Devereux, 2007; Tefempa, Ngassam, Nkongmeneck, Mapongmetsem, & Gubbuk, 2008). This resource is often supplemented for two primary reasons in protected areas, namely i) to boost water-dependent herbivore population sizes, (ii) to manipulate herbivore distribution patterns and, iii) to enhance biodiversity as a result of increased habitat heterogeneity. However, waterpoints often result in the opposite effect by homogenizing vegetation over a large landscape (Harrington et al., 1999). Additionally, the placement of artificial waterpoints (AWPs) to improve wildlife viewing for ecotourism is undertaken (Farmer, 2010).

1.1.1 *The effect of water supplementation on vegetation*

Water supplementation has a noticeable effect on rangeland utilisation, which largely effects soil and vegetation over time (Graetz & Ludwig, 1978). Large migratory herbivores become sedentary due to the fencing in of reserves and the provision of water supplementation. This results in constant intense herbivore utilisation and browsing leading to increased utilisation pressure and vegetation change (Walker, 1979). Vegetation surrounding AWP's requires relief from grazing pressure during the wet season in order to recover from intensive dry season trampling and grazing (Cronje, Cronje, & Botha, 2005; Owen-Smith, 1996). Water dependent herbivores congregate in areas surrounding perennial surface water during the dry season and disperse into forage areas away from these sources during the wet season, resulting in seasonal variation (Thrash, 2000b; Thrash & Derry, 1999; Young, Ferreira, & Van Aarde, 2009). The placement of AWP's in regions with few natural perennial water sources allows for further wildlife dispersal. This dispersal distance increases in the dry season when water is not readily available, resulting in wildlife travelling vast distances in search of forage areas. Another result may be wildlife concentrating in areas close to AWP's, increasing herbivore utilisation in these areas (Cronje et al., 2005; Thrash, 2000a). This requires an area with suitable and sufficient forage, and perennial water, to encourage herbivore population dispersal. Certain regions in protected areas may experience vegetation homogenisation as a result of limited or no vegetation recovery periods, which in turn results in a decrease in wildlife abundance during extreme conditions such as droughts (Matchett, 2010; Owen-Smith, 1996). Therefore, wildlife populations may fail to recover from such extreme conditions once the circumstances have improved (Owen-Smith, 1996).

An thorough understanding of how water supplementation impacts an ecosystem is required for best water management practices. Herbivores, particularly those which are water-dependent, tend to congregate near water. This increases in the dry season as natural surface water gradually becomes depleted (Owen-Smith, 1996; Smit et al., 2007). Such congregation results in gradients of utilisation with increased utilisation pressure closer to the AWP, as intense grazing and trampling of the surrounding vegetation increases (Graetz & Ludwig, 1978; Perkins & Thomas, 1993), resulting in a piosphere effect. Lange (1969) defined a piosphere as "an ecological system of interactions between a permanent watering point, its surrounding vegetation and the grazing animal."

The piosphere effect is caused by a trade-off between the forage and water requirements of wildlife, forming noticeable degradation surrounding a water source. The extent of this degradation is limited by distance travelled between drinking events which is related to the quantity and quality of the forage adjacent to water sources (Farmer, 2010; Smit et al., 2007). The impact of AWP's is visualised as circular piospheres surrounding a single water source, with a high level of impact adjacent to the AWP. A

habitat surrounding an AWP shows its true ecological potential the further you move from surface water, moving through a zone of change where herbivore impact tails off (Graetz & Ludwig, 1978). A region with a high density of perennial water has an increased probability of merging biospheres, resulting in habitat homogeneity (Graetz & Ludwig, 1978). This depicts a negative relationship between herbivore utilisation and distance from water for herbivores (Child, Parris, & Le Riche, 1971; Senzota & Mtahko, 1990; Thrash, Theron, & Bothma, 1993; Weir, 1971).

The impact seen in the immediate vicinity of the AWP is caused more by trampling than intense herbivore utilisation (Senzota & Mtahko, 1990). This trampling is caused by herds of herbivores waiting around the AWP for their turn to drink when competition is high (Young, 1972). Herbaceous species diversity tends to decrease near waterpoints while an increase is seen in weed cover (Tolsma, Ernst, & Verwey, 1987). Additionally, a decrease in herbaceous vegetation results in an increased chance of water run-off on soil, causing severe erosion (Young, 1972). Veld condition, herbaceous biomass and cover increase with distance from water. Change in woody vegetation structure occurs at waterpoints and the majority of the impact is inflicted by elephants. Over time a decrease in woody vegetation density is seen as distance to water decreases (Thrash, Nel, Theron, & Bothma, 1991).

1.1.2 *The effect of water supplementation on large herbivores*

Numerous protected areas located in water stressed countries throughout Africa, make use of AWPs (Child et al., 1971; Epaphras et al., 2007; Makhabu, Balisana, & Perkins, 2002; Parker & Witkowski, 1999; Redfern, Grant, Biggs, & Getz, 2003; Shannon, Matthews, Page, Parker, & Smith, 2009; Tefempa et al., 2008; Valeix, 2011). These AWPs provide a vital source of water for mammals in the dry season, sustaining them until the rains arrive in the wet season, replenishing natural waterpoints. This resource may therefore act as an important management tool, controlling mammal movements and population sizes (Owen-Smith, 1996; Smit et al., 2007; Tefempa et al., 2008).

Essential resources such as water determine the carrying capacity of mammal populations within fully fenced protected areas (Chamaillé-Jammes, Valeix, & Fritz, 2007; Smit & Grant, 2009; Valeix et al., 2009). Water supplementation alters wildlife distribution as some herbivores no longer need to migrate between areas, and instead become localised (Walker, 1979). This localisation may cause a rapid population increase of water-dependent species such as zebra and wildebeest (*Connochaetes taurinus*), increasing competition with the more vulnerable, low-density species, for example roan (*Hippotragus equines*) or sable antelope (*Hippotragus niger*) (Harrington et al., 1999; Joubert, 1976). Thus, competition usually results in the decline of vulnerable low-density species' population sizes (Grant et al., 2011; Harrington et al., 1999; Nicholls, Viljoen, Knight, & van Jaarsveld, 1996; Seydack et al., 2012). Large herbivore species such as common reedbuck (*Redunca arundinum*), roan and sable

antelope rely on tall grass to hide their altricial young (Estes, 2012). An increase in perennial water sources may decrease herbaceous vegetation cover, including tall grass species, resulting in increased effort for certain antelope when trying to hide their young (Collinson, 1983; Owen-Smith, Le Roux, & Macandza, 2013).

The dispersal ranges of herbivores can be extended by placing AWP in regions lacking natural perennial surface water, allowing for herbivores to occupy these now resource rich regions (Owen-Smith, 1996). In order to attract herbivores, these regions require enough forage situated at a minimum of 10km from a water source. Without sufficient distance between surface water, herbivore impact on vegetation adjacent to the surface water will be exaggerated and result in vegetation homogenisation. This will decrease habitat diversity and resilience, as such regions become vulnerable to extreme conditions which are difficult to recover from (Owen-Smith, 1996).

Waterpoints are dangerous sites for herbivores as they are often ambushed by predators as they are vulnerable while drinking (Périquet et al., 2010; Valeix et al., 2009). Some herbivores require access to water on a daily basis, depending on rainfall seasonality, allowing predators to predict patterns of AWP use by some herbivores (Crosmar, Valeix, Fritz, Madzikanda, & Côté, 2012; Périquet et al., 2010; Valeix et al., 2009). Predation may strongly influence AWP utilisation by herbivores, even resulting in some species changing peak AWP utilisation times in order to avoid predators. Most herbivores drinking times, with the exception of elephants, peak diurnally (Crosmar et al., 2012; Valeix et al., 2009). This may be due to avoidance of nocturnal predators, however, it may also be due to competition with elephants as they are more dominant at waterpoints (Hayward & Hayward, 2012; Valeix et al., 2009). Thermoregulation may influence the time spent accessing water as herbivores avoid open areas during hotter periods of the day when solar radiation is maximised (Matsika, Matsvimbo, Valeix, Madzikanda, & Fritz, 2008). This is noticeable in larger species such as buffalo, zebra and giraffe as they heat and cool less rapidly than smaller species (Matsika, Matsvimbo, Valeix, Madzikanda, & Fritz, 2008).

Many studies focussing on the effect of water supplementation within ecosystems have been conducted in eastern (Ayeni, 1975; Castelda, Napora, Nasser, Vyas, & Schulte, 2011; Egeru, Bernard, Henry, & Paul, 2015; Epaphras et al., 2007; Mukinya, 1977) and southern Africa (Brits, van Rooyen, & van Rooyen, 2000; Hayward & Hayward, 2012; Kasiringua, Kopij, & Şerban, 2017; Parker & Witkowski, 1999; Redfern et al., 2003; Sutherland, Ndlovu, & Pérez-Rodríguez, 2018; Thrash et al., 1993; Valeix, 2011), however, there is a gap in the literature for the regions in between, countries such as Zambia, Malawi and Mozambique, for example.

1.2 Study area

Majete Wildlife Reserve (MWR, hereafter) is a 700km² fenced area located in the lower Shire valley of southern Malawi (Figure 1.1). MWR was established as a non-hunting region surrounding Majete Hill in 1951 (Spies, 2015). The intention was to create a refuge area for larger animals experiencing increased pressure due to an increasing human population in the region during the early 20th century. This was followed by the declaration of a 500km² area of land as Majete Game Reserve in 1955, with the main intention of restricting the elephant population to this area (Morris, 2006). In 1969 MWR was extended to include the Mkulumadzi River in the north and a small area east of the Shire River (Morris, 2006; Sherry, 1989). By the early 2000's most mammal species had been depleted or completely eradicated from the reserve due to a lack of finances, an ill-equipped anti-poaching unit and poor management.

In 2003 African Parks (Pty) Ltd., and the Malawian government entered a 25-year public-private partnership (PPP), in which African Parks took on the responsibility to develop, manage and rehabilitate MWR (Spies, 2015). This was followed by the fencing of a sanctuary region (14 000ha north-eastern area) and the initial reintroduction of more than 2550 individuals from 14 species over several years (Wienand, 2013; Appendix 1). The boundary fence of the reserve was completed in 2008, and in 2011 the sanctuary fence was removed, allowing the animals to repopulate the entire reserve. An increase in funding and improved management has resulted in the population of 2550 reintroduced individuals growing to well over 13 000 animals according to the most recent aerial census conducted in 2015 (C. Hay, personal comms.). MWR has a "no hunting" or "culling" policy, with intentions of using surplus animals to repopulate other reserves within Malawi.

Two distinct seasons characterise MWR, the dry season from June to November and the wet season from December to May. Estimated mean minimum and maximum temperatures are 12.5°C and 26.3°C for the coolest month (June) and 20.9°C and 34.8°C for the hottest month (November). The lowest recorded temperature in MWR was 11°C and the highest 45°C (Martin, 2005; Wienand, 2013). MWR has a clear gradient of precipitation ranging from 700-1000mm in the western highlands and 680-800mm in the eastern lowlands (Wienand, 2013). Evaporation rates in the Lower Shire valley are high due to the high temperatures experienced. These rates range from 107mm in June (winter) to 274mm in October (summer), with a total annual evaporation rate of ~2000mm. MWR may experience lower rates of evaporation due to its elevation (Martin, 2005).

The topography of the reserve slopes from over 900m in the western highlands to 150m in the south-eastern lowlands (Sherry, 1989). This includes numerous rocky outcrops and hills with moderately gentle undulations. The geology of the reserve includes rock formations of Precambrian Basement

Complex gneisses and schists with overlying pockets of recent alluvial deposits (Sherry, 1989). MWR's soil composition includes limited deposits of fertile alluvial soils along small areas of some rivers. The majority of the reserve is composed of shallow, stony ferruginous soils, and lithosoils, or poorly fertile lithosoils, with loamy or sandy soils (Sherry, 1989).

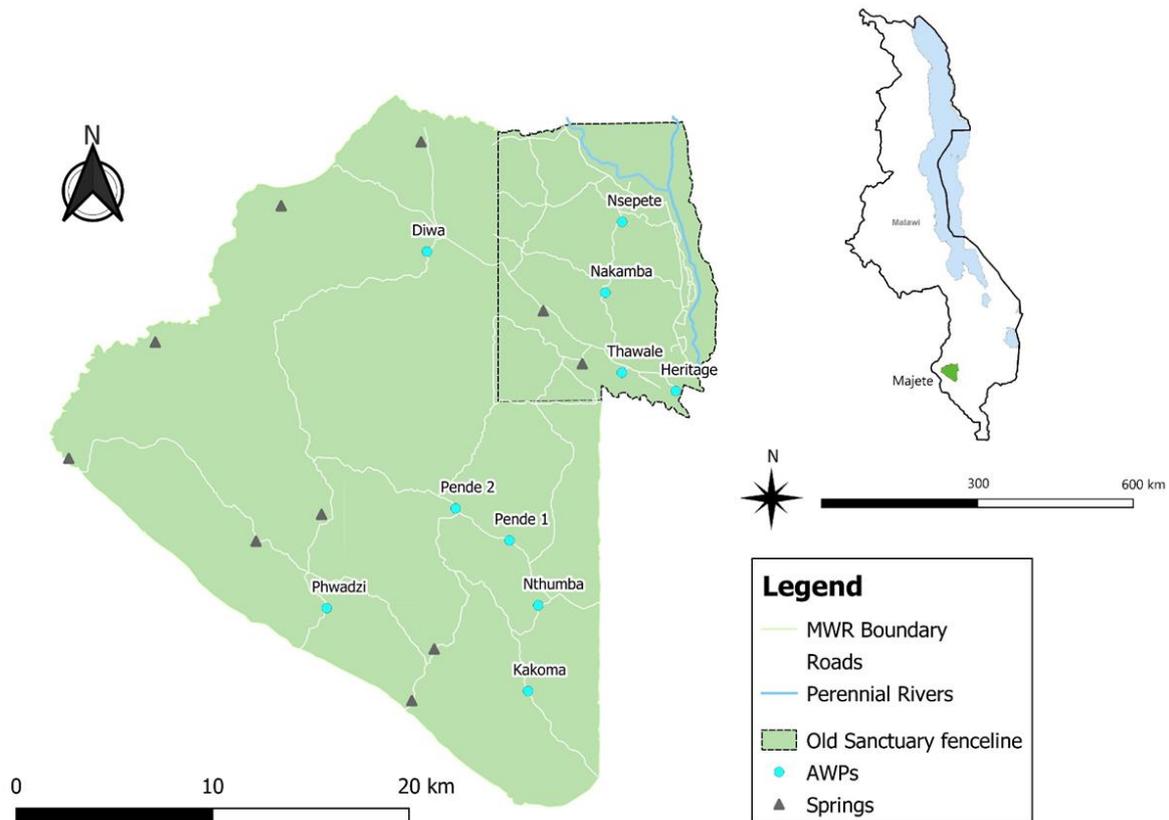


Figure 1.1 The location of Majete Wildlife (MWR) Reserve in Malawi, and the position of the ten artificial waterpoints and two perennial rivers in the reserve (Shapefiles provided by African Parks (Pty) Ltd.)

The reserve has two perennial rivers, the Mkhulumadzi and Shire, situated in the north and east respectively. Water availability in the reserve is dependent on season, and is additionally provided from six non-perennial springs, five perennial springs and ten AWP (Figure 1.1).

Vegetation in MWR is classified as primarily miombo savanna woodland (de Vos, 2017). This classification can be divided into four vegetation types: savanna characterised by *Panicum*, *Vachellia*, *Senegalia* and *Combretum* species; low altitude mixed woodland (<250m) characterised by *Sterculia*, *Vachellia* and *Senegalia* species; medium altitude mixed woodland (250 – 400m) characterised by

Combretum species, *Diospyros kirkii* and *Brachystegia boehmii*; and high altitude miombo woodland (>400m) characterised by *Pterocarpus*, *Burkea africana* and *Brachystegia boehmii* (Figure 1.2).

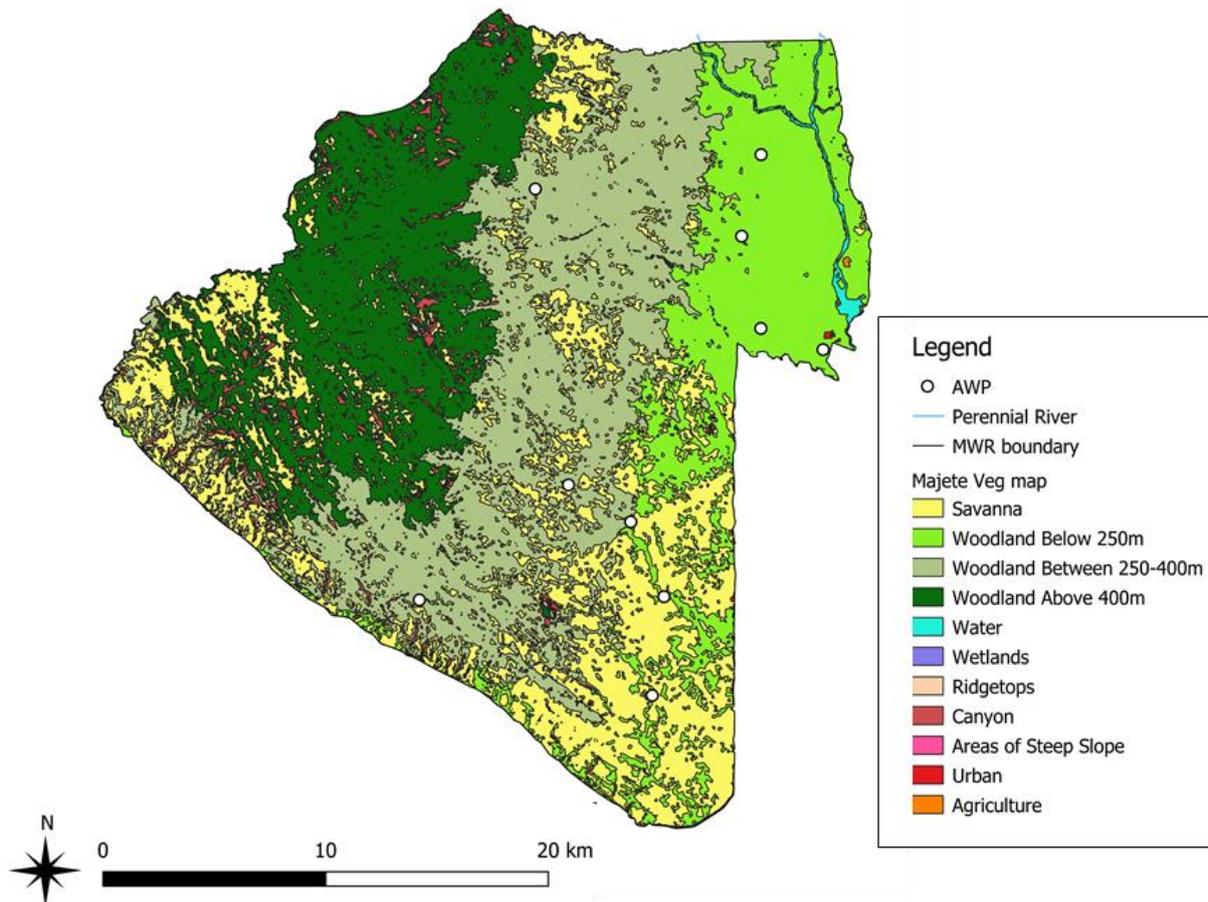


Figure 1.2 A vegetation map of MWR and the location of ten artificial waterpoints (Shapefiles provided by African Parks (Pty) Ltd.)

Miombo woodland is the most extensive dry deciduous woodland in southern and central Africa, covering over 2.7 million km² (Frost, 1996). Trees in the *Brachystegia*, *Julbernardia* and *Isoberlinia* genera are characteristic of this woodland. The recent vegetation mapping study is in the process of being ground-truthed in order to ensure an accurate classification of vegetation types (C. Hay, personal communication, February 13, 2018).

1.2.1 Artificial waterpoints of MWR

The ten AWP's located in MWR are borehole-fed and have been placed throughout the reserve, with four located in the sanctuary area, four in the south-east (Pende) area, one in the north-western

(Diwa) area and one in the south-western (Pwadzi) region (Figure 1.1). Herbivore carrying capacity was largely limited by water availability in MWR before the development of AWP (Bell, 1984). These AWP were established in order to stabilise surface water availability, by supplementing the available natural water thereby contributing to increased population growth and stability of the various species (Chamaillé-Jammes, Fritz, & Murindagomo, 2007; Shannon et al., 2009). The four AWP in the sanctuary were placed to enhance wildlife viewing for tourism, as two are placed in front of human habituated areas (lodge/restaurant), and the remaining two have viewing hides located at the AWP. The six remaining AWP were established in order to encourage wildlife movement from the sanctuary to the southern and western regions of the reserve, therefore removing grazing and browsing pressure from the sanctuary region. This will allow for the spread of herbivore utilisation evenly throughout the reserve, decreasing the probability of habitat homogenisation. Habitat homogenisation results in decreased ecosystem resilience and should therefore be avoided when managing water supplementation in reserves (Gaylard, Owen-Smith, & Redfern, 2003).

The topic for this thesis was decided upon to gain a better understanding of the utilisation and impact of herbivores on AWP and the surrounding vegetation in a fenced reserve with limited natural perennial water. Understanding the utilisation of AWP will help with possible future placement or removal of AWP for management purposes, whether to assist with dispersal of herbivore populations or vegetation recovery, for example, and contribute to an understanding of the impacts that herbivore populations have on vegetation surrounding AWP.

1.3 Research goal and objectives

1.2.2 Goal

To provide a guideline for MWR with regards to waterpoint management and placement. These guidelines will aim to integrate vegetation condition, water point usage and distance from other perennial sources when addressing future water point placement and the conservation of habitat heterogeneity, managing herbivore vegetation impacts and preventing biodiversity loss.

1.2.3 Objectives

1. To determine the impact of water supplementation and herbivore water utilisation on vegetation cover in MWR.

- a) What is the relationship between distance from water and vegetation cover (e.g. woody and herbaceous vegetation)?
- b) What is the status of the Ecological Index of herbaceous vegetation surrounding the AWP?

- c) Do areas with a higher density of AWP experience decreased vegetation cover and condition?
2. To determine the utilisation of AWP by selected herbivores in MWR and to quantify factors associated with AWP use patterns. Specifically, to determine:
- a) At what frequency and time of day herbivores use the AWP?
 - b) If there is a seasonal variation in AWP utilisation by herbivores?
 - c) If there are differences between specific species and feeding strategies (grazers, browsers and mixed feeders) in AWP use?
 - d) If AWP in close proximity to other perennial water sources experience less herbivore utilisation?
3. To determine if vegetation cover and erosion surrounding AWP has changed in MWR over a five-year period (2013-2018).
- a) What is the extent of vegetation change surrounding waterpoints using fixed-point photography over a five-year period?
 - b) Do older AWP show a decrease in vegetation cover?
 - c) Does erosion surrounding AWP increase with time?

1.4 Thesis structure

This master's thesis comprises of five chapters. Chapter Two, Three and Four have been written in the format of stand-alone manuscripts. This format was chosen to aid with publication in peer-reviewed journals. This has resulted in some repetition and cross-referencing between chapters.

Chapter one includes a general introduction to AWP and their use in conservation, details regarding the study site, research aims and objectives.

Chapter Two investigates AWP utilisation by 12 herbivore species in Majete Wildlife Reserve. Temporal and seasonal variation in artificial water point usage for all species is quantified and described. This is followed by a discussion regarding the various study species and artificial waterpoint management.

Chapter Three investigates vegetation cover and condition surrounding artificial waterpoints and how this changes with distance from AWP.

In Chapter four, the temporal change in vegetation cover and erosion at AWP is described by comparing fixed-point photographs over a five-year period.

Chapter Five collates the results and conclusions from all the previous chapters and discusses the implications for AWP management in MWR.

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Chapter Two

Artificial Water Point Utilisation by Selected Herbivores in Majete Wildlife Reserve, Malawi

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2.1 Abstract

Access to water is essential for the survival and reproduction of wildlife and may affect the distribution of species within a reserve. Water supplementation, via artificial waterpoints (AWPs), has become common practice in maintaining wildlife densities as natural water becomes scarce during the dry season. Herbivore AWP utilisation was studied at ten AWPs in Majete Wildlife Reserve (MWR), Malawi, using camera traps from June 2017 to May 2018. Chi-Square Goodness of Fit Tests were conducted to determine if the frequency of observations were significantly different per season, AWP and time of day. A total of 219 105 observations were made of 12 herbivore species. African elephant (*Loxodonta africana*) and black rhino (*Diceros bicornis minor*) were the most frequently observed species and impala the least. The results show a significant difference in waterpoint utilisation between seasons, AWPs and time of day ($p < 0.001$). The majority of observations occurred in the dry season (79%), with emphasis on the late dry season (56%), compared to the wet season (21%). Overall peak utilisation times were from 10:00 - 11:00, apart from black rhinoceros (*Diceros bicornis minor*) and Cape buffalo (*Syncerus caffer*) which preferred to utilise AWPs at dawn and dusk. AWP utilisation was significantly higher ($p < 0.001$) in the south-east of the reserve, possibly due to a lack of natural perennial water sources in this region. This study provides insight into herbivore AWP utilisation in MWR and will aid in future management with regards to the seasonal opening and closing of AWPs. This study shows that AWPs are an important source of water for herbivores in areas devoid of perennial water within MWR, with an emphasis on the dry season.

2.2 Introduction

Access to water is essential for the survival and reproduction of wildlife and may affect the distribution of species within a reserve (Epaphras et al., 2007; Hayward & Hayward, 2012; Kasiringua et al., 2017; Redfern et al., 2003; Western, 1975). Water controls the daily activities and land use patterns of most species throughout the globe, with emphasis on herbivores (Halidu, Ayodele, Lameed, & Oyleye, 2013; Smit et al., 2007; Sutherland, Ndlovu, & Perez-Rodriguez, 2018). With the increase in habitat fragmentation and the change in climatic conditions, water has become a limited resource (Ayeni, 1975; Fullman, Bunting, Kiker, & Southworth, 2017; Simpson, Stewart, & Bleich, 2011; Zvidzai, Murwira, Caron, & Wichatitsky, 2013). Historically, herbivores relied on natural perennial water sources, seasonally migrating out of areas lacking this resource during the critical dry months in order to maintain stable populations (Epaphras et al., 2007; Grant et al., 2011). However, many reserves in southern Africa are fenced in order to protect wildlife from the surrounding increasing human population. This fragmentation of protected areas has resulted in limiting or interruption of historical migratory routes to perennial water sources which, in turn, can result in a dramatic decrease in many species' population numbers (Tefempa et al., 2008).

The distribution, ecology and conservation of herbivores will be negatively affected by a lack of vital resources such as forage and water that are essential requirements for survival. Water supplementation via artificial waterpoints (AWPs) has become common practice where natural water sources become scarce during the dry season, thereby maintaining wildlife density during these harsh conditions (Sutherland, Ndlovu, & Perez-Rodriguez, 2018). AWPs allow for the expansion of herbivore forage ranges as well as aiding in maintaining water-dependent species population numbers. Water provision is therefore ranked as one of the core management interventions throughout African wildlife reserves (Owen-Smith, 1996; Rosenstock et al., 1999; Sirot, Renaud, & Pays, 2016; Smit et al., 2007).

Despite the various positives, some studies have found that AWPs may be disadvantageous for some species. AWPs may cause increased herbivory on the surrounding vegetation as herbivores forage in close proximity of waterpoints, causing subsequent vegetation degradation known as the piosphere effect (Egeru et al., 2015; Makhabu et al., 2002; Sutherland, Ndlovu, & Perez-Rodriguez, 2018; Van Rooyen, Bredenkamp, Theron, Bothma, & Le Riche, 1994). In the Kruger National Park, South Africa, researchers found that during severe times of drought, a higher frequency of more common grazers were present in areas they were ordinarily absent from due to the presence of AWPs. The rare antelope species in these areas were out competed by the common grazers, which in turn attracted more predators into these areas, resulting in an increased exposure of the rare antelope to these predators (Harrington et al., 1999; Rosenstock et al., 1999).

Understanding the utilisation of AWP by wildlife is vital for determining the impact of water supplementation on habitat integrity. Factors affecting herbivore AWP utilisation may include competition (Sirota et al., 2016; Valeix, Chamaillé-Jammes, & Fritz, 2007), predation risk (Crosmary et al., 2012; Valeix et al., 2009), seasonality (Halidu et al., 2013; Hayward & Hayward, 2012; Sutherland, Ndlovu, & Perez-Rodriguez, 2018), water availability and species activity patterns (Hayward & Hayward, 2012; Mukinya, 1977). Seasonality occurs as herbivores depend highly on AWP when water availability decreases during the dry season (Hayward & Hayward, 2012; Smit et al., 2007; Sutherland, Ndlovu, & Pérez-Rodríguez, 2018). As AWP influence herbivore movement patterns, habitat integrity on the landscape scale may be significantly influenced (Smit et al., 2007). The region surrounding the AWP may experience a gradient of this influence up to 5kms from water due to increased wildlife traffic, causing increased foraging pressure and trampling (Chamaillé-Jammes et al., 2009; Smit et al., 2007).

For example, the Kruger National Park in South Africa historically had more than 300 borehole-fed AWP and 50 reservoirs (Gaylard, Owen-Smith, & Redfern, 2003). These provided sufficient water for wildlife, allowing species to be at a minimum distance of 5km from a water supply, which instead resulted in various negative effects on the ecosystem (Smit et al., 2007). These negative effects included a decrease in rare antelope population sizes and distribution, an increase in predator numbers near AWP (Harrington et al., 1999; Owen-Smith, 1996) and vegetation over-utilisation in the form of the piosphere effect (Brits et al., 2002; Parker & Witkowski, 1999). In 1997, reserve management in the park decided to implement a new water management strategy which aimed at closing many AWP in order to create large water remote areas, aiding in the decrease in landscape homogenisation (Smit 2013). This resulted in a change in wildlife abundance and distribution within the park, with browsers being least affected due to browse having a higher water content than grass (Smit & Ferreira, 2010; Smit et al., 2007). Therefore, it is integral that reserve management is aware of both the negative and positive aspects of AWP and how AWP are utilised in order to predict and assess the impact the distribution of herbivores have on the surrounding vegetation (Owen-Smith, 1996; Smit et al., 2007).

The aim of this study was to determine how selected herbivores utilised artificial waterpoints located in MWR and if there was any seasonal and/or temporal variation in waterpoint use. The hypothesis was that herbivores would prefer to utilise AWP during daylight hours when water was pumped via solar power (sometimes water was depleted by the late afternoon as large buffalo herds emptied the concrete troughs). Additionally, it was hypothesised that the utilisation of AWP by selected herbivores would vary between the different seasons, with more frequent visits occurring in the dry season when compared to the wet season. This study will provide insight into herbivore water

utilisation in terms of time preference and seasonal activity, in order to assist management with future decision-making regarding AWP positioning or closure within MWR.

2.3 Methods

2.3.1 Study site

Majete Wildlife Reserve (MWR), covering 700km², is located in the Lower Shire valley of southern Malawi (Figure. 2.1). The topography of the eastern region of the reserve is relatively flat in comparison to the steep undulating hills and valleys of the western region. Annual precipitation is highest in the western region (700-1000mm) and lowest in the eastern region (680-800mm), with a wet season from December until May (Wienand, 2013). Vegetation types in MWR fall within the Miombo woodland ecoregion as it is primarily woodland dominated. There are two perennial rivers bordering the north-eastern boundary of the reserve, the Shire River and Mkulumadzi River (Figure 2.1). A total of 12 species were reintroduced into the reserve between 2004 and 2009, expanding their distribution according to the available resources within the reserve. In order to provide reliable water sources for wildlife during the dry season, 10 solar powered AWPs were installed in MWR between 2007 and 2014 (Wienand, 2013).

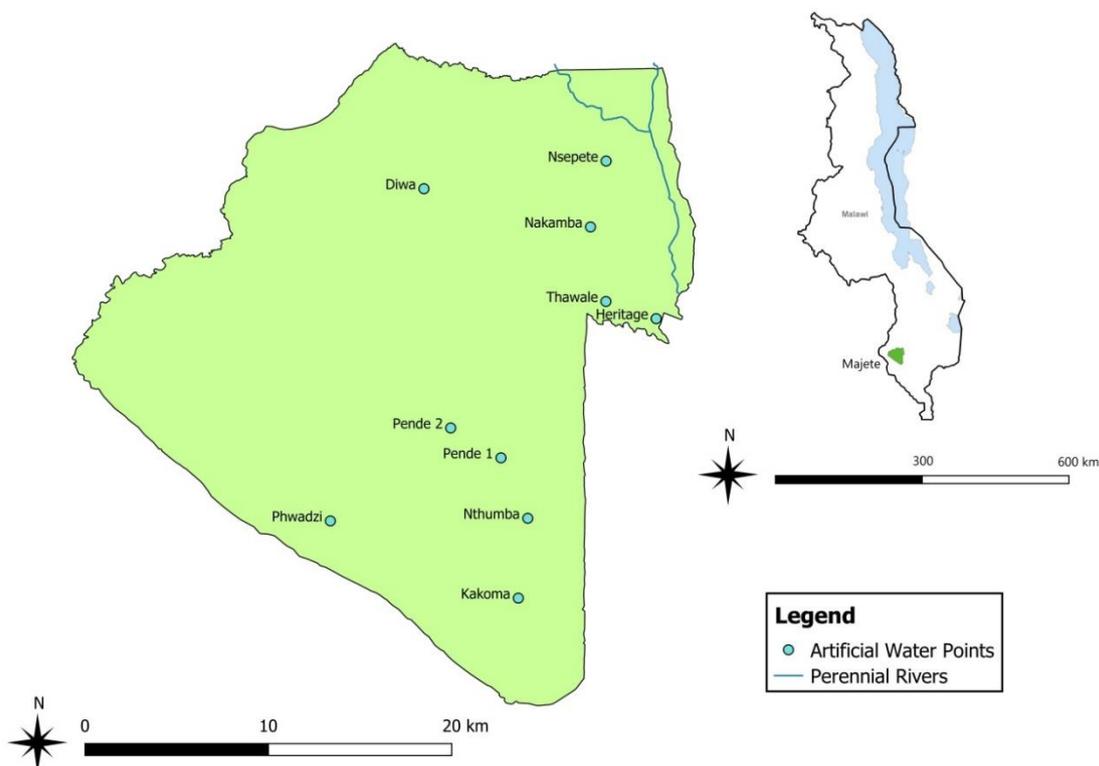


Figure 2.1 Location of the ten artificial waterpoints in Majete Wildlife Reserve (MWR) situated in the southern region of Malawi (Shapefiles provided by African Parks (Pty) Ltd.).

The AWP's are situated throughout the reserve, and are all serviced via boreholes and solar pumps. These AWP's included Diwa in the north-western region; Phwadzi in the south-western region; Pende 1, Pende 2, Nthumba and Kakoma situated in the south-eastern region; and Thawale, Nakamba, Nsepete and Heritage in the North-eastern region (Figure 2.1).

2.3.2 Camera trap surveying

The non-invasive research technique of camera trapping has been used worldwide to determine species population sizes as well as to monitor behaviour (Ancrenaz, Hearn, Ross, Sollman, & Wilting, 2012). In this study, camera traps were a useful method of sampling, as they increased the sampling area and sampling period while providing continued monitoring of the study sites. The drinking behaviour of 12 ungulate species was studied, including: three browsers, namely black rhinoceros (*Diceros bicornis minor*), greater kudu (*Tragelaphus strepsiceros*) and bushbuck (*Tragelaphus scriptus*); five grazers, namely African savanna buffalo (*Syncerus caffer*), plain zebra (*Equus quagga*), Waterbuck (*Kobus ellipsiprymnu*), Lichtenstein's hartebeest (*Alcelaphus lichtensteinii*), sable antelope (*Hippotragus niger*); and four mixed feeders, namely African elephant (*Loxodonta africana*), common eland (*Tragelaphus oryx*), Nyala (*Tragelaphus angasii*), and impala (*Aepyceros melampus*).

Cuddeback™ Ambush© and Attack© camera traps were positioned on large trees between 1m and 1.5m off the ground, at all ten of the AWP's from 1 June 2017 to 15 May 2018. Cameras were positioned so that the entire concrete water basin of the AWP was in view. Once a photo was taken, the cameras were set to wait for a period of one minute before another photo was taken if triggered.

Due to technical issues with some cameras at some of the waterpoints, camera trap active days varied between 204 and 327 days. Generally, each camera trap was serviced after a 14 to 30-day period which involved changing batteries and SD cards. Once downloaded onto the computer, the photographs were sorted into files according to species, date and location. Animals in the photographs taken were not necessarily drinking which is why the term waterpoint "utilisation" was used as opposed to actual "drinking". That said, water sources are areas of high risk for herbivores as they are exposed to ambush predation, therefore, the close proximity of herbivores to water suggests they were or had been seeking water (Valeix et al., 2009). Every photo was counted as a visitation, regardless if the same individual or herd was present in a string of photos. This was done as we were interested in observing how the entire area surrounding the AWP was utilised and not only the AWP itself. This would give us an indication of the pressure herbivores exert on the vegetation and the soil between the different seasons. Therefore, utilisation was used as this refers to not only water but also general utilisation of the entire area surrounding the AWP, including animals that loiter around the

AWP. This may however bias the data towards species that are known to loiter around AWP's such as elephant, buffalo, waterbuck, etc.

2.3.3 Statistical analysis

All statistical analyses were conducted in R version 3.5.2 (The R Foundation for Statistical Computing, 2018). Chi-Square Goodness of Fit Tests were run to determine if the frequency of visits by species to AWP's was significantly different according to AWP, season and time of day. The frequency of visits was calculated as the number of photographs containing a section or an entire study species, divided by the number of camera trapping days. This was then compared to an extrinsic theoretical expectation derived from dividing the total photo count by the number of seasons, hours or AWP.

A correspondence analysis was conducted in order to determine if all 12 species had a preferred time of day for AWP utilisation, corresponding to certain seasons within MWR. A Chi-square Test for Independence was then conducted in order to determine if the pattern derived from the correspondence analysis was significant.

2.4 Results

A total of 219 105 observations (a photograph that contained a portion of or an entire study species), from 12 species were recorded using camera traps at the ten AWP's monitored within MWR during the study period (Figure 2.2). All ten AWP's were not visited equally (Chi-square Goodness of Fit Test: $\chi^2_{(df = 9)} = 30690$, $p < 0.001$). Likewise, the number of visitations to AWP's by season was significantly different (Chi-square Goodness of Fit Test: $\chi^2_{(df = 3)} = 53395$, $p < 0.001$). Of the 219 105 sightings recorded, 28 160 (12.85%) were recorded in the late wet season (March - May), 48 725 (22.24%) in

the early dry season (June - August), 123 672 (56.44%) in the late dry season (September – November) and 18 548 (8.47%) in the early wet season (December - February).

Pende 1 was the most frequently visited AWP (12.94%) and Phwadzi was the least visited AWP (0.86%). The observations of species at the AWPs were standardised by population size (Figure 2.3). Overall, elephants (26.52%) were the most abundant species recorded, followed by black rhino (16.15%) while impala were the least (3.83%) recorded according to the 2015 estimates from the 2015

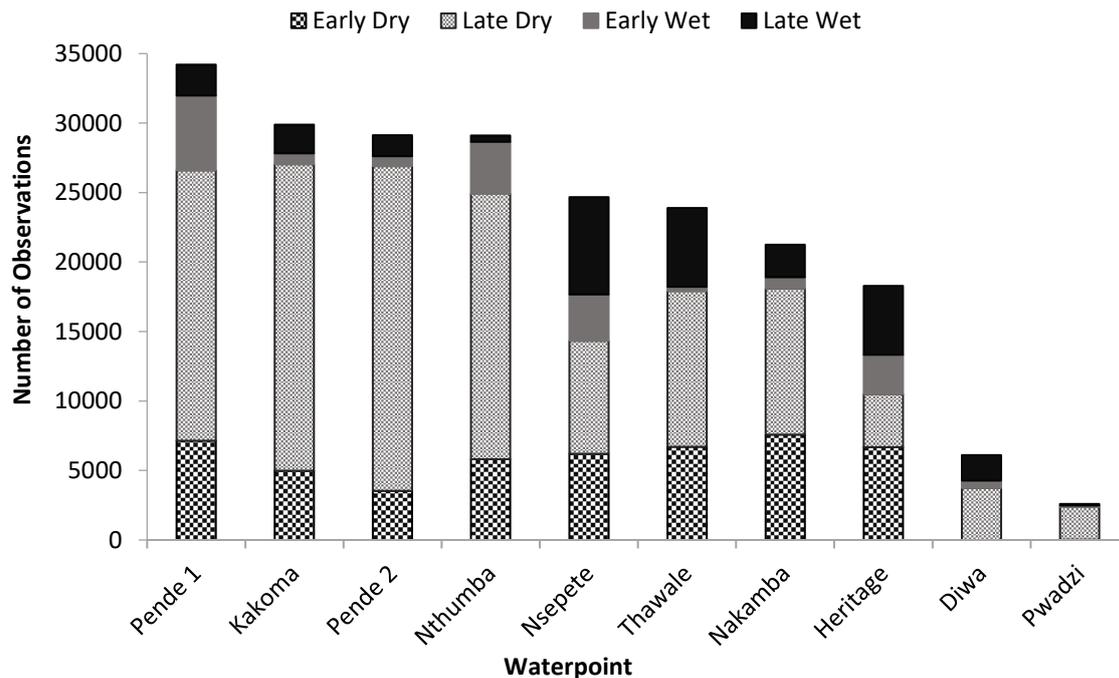


Figure 2.2 The total number of observations of the twelve study species at 10 artificial waterpoints over the four seasons within MWR, Malawi.

aerial census (Appendix 1) in MWR (Figure 2.3). Black Rhino were not observed visiting the Heritage, Phwadzi and Thawale AWPs. Hartebeest did not visit the Heritage and Phwadzi AWPs, and Nyala were not observed at both Diwa and Phwadzi AWPs. The remaining study species were observed at all ten AWPs (Figure 2.3).

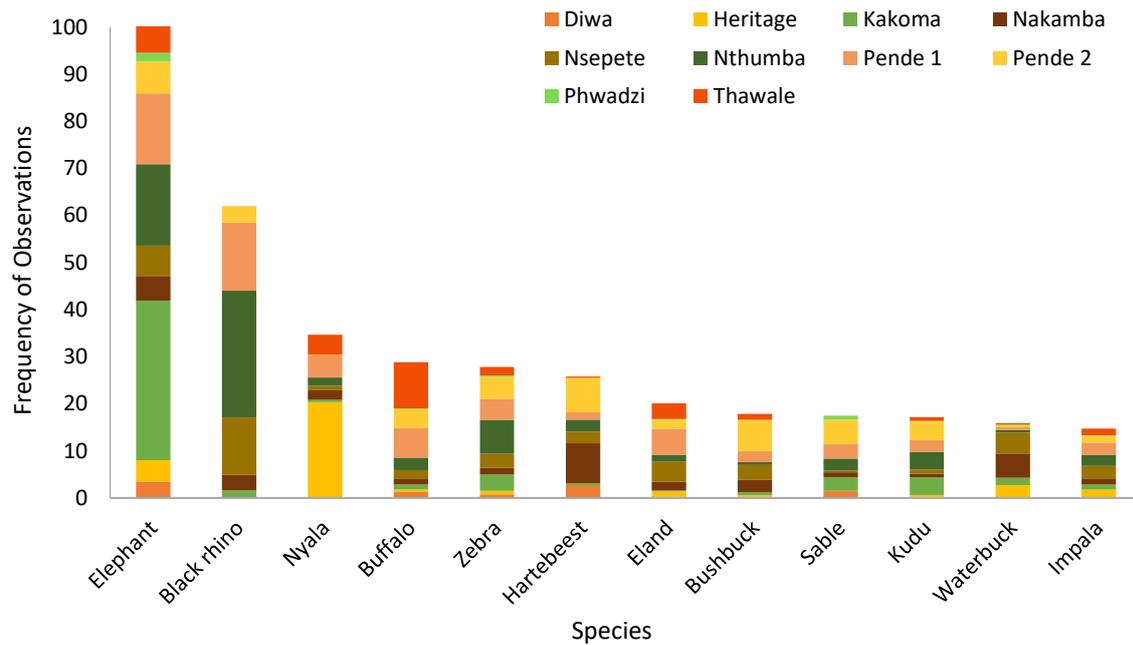


Figure 2.3 Frequency of observations of the twelve study species at each of the ten artificial waterpoints within MWR, Malawi.

Raw data was divided by camera trap effort for each AWP in order to obtain observation frequencies. Overall, the total number of observations for all species at the ten AWP was significantly different for time of day (Chi-square Goodness of Fit Test: $\chi^2(df = 23) = 248.08$, $p < 0.001$). All species combined AWP use peaked at 10:00 and 11:00 and use was lowest at 03:00 and 04:00 (Figure 2.4).

With the exception of black rhino, all species were recorded visiting the various AWP during all 24-hourly periods for the entire study period (Figure. 2.5). Black rhino, however, did not visit any of the AWP from 09:00 and 11:00 (Figure. 2.5). Black rhino, bushbuck, eland, hartebeest, impala, kudu, nyala and sable were not observed visiting the various AWP during all 24-hourly periods across the four seasons.

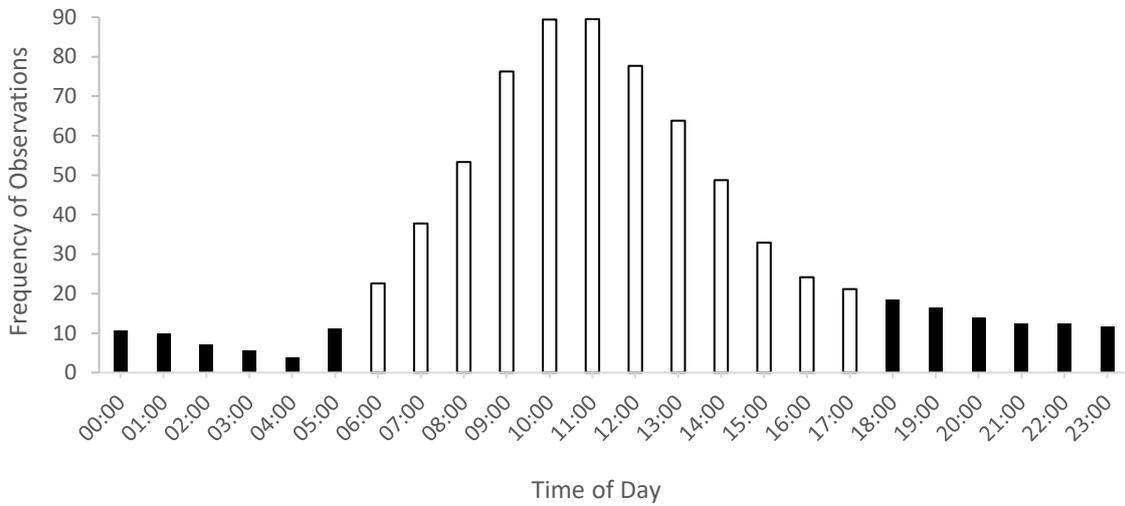


Figure 2.4 The total number of observations recorded each hour over a 24-hour period for all species in MWR, Malawi. Raw data was divided by camera trapping effort to obtain observation frequencies. Dark bars depict non daylight hours.

All species AWP utilisation peaked between 09:00 and 12:00 (Figure 2.5 to 2.7), except black rhino (Figure 2.7) and buffalo that peaked at dawn and dusk (Figure 2.6.).

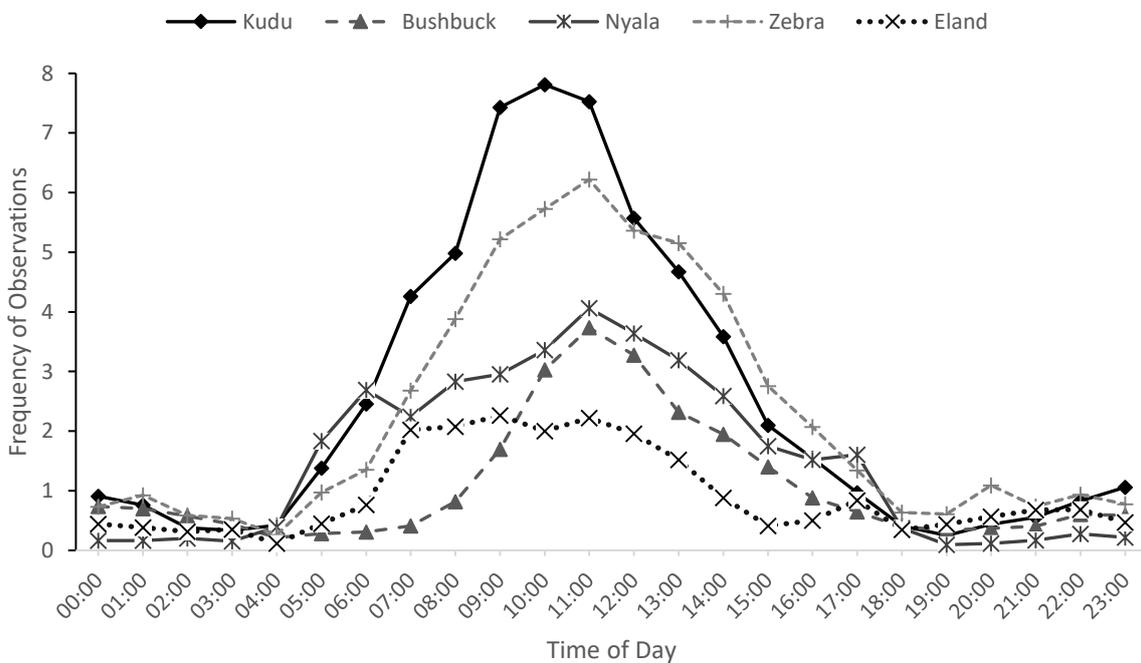


Figure 2.5 AWP utilisation observed for kudu, bushbuck, nyala, zebra and eland at the ten AWP. Raw data was divided by camera trapping effort to obtain observation frequencies.

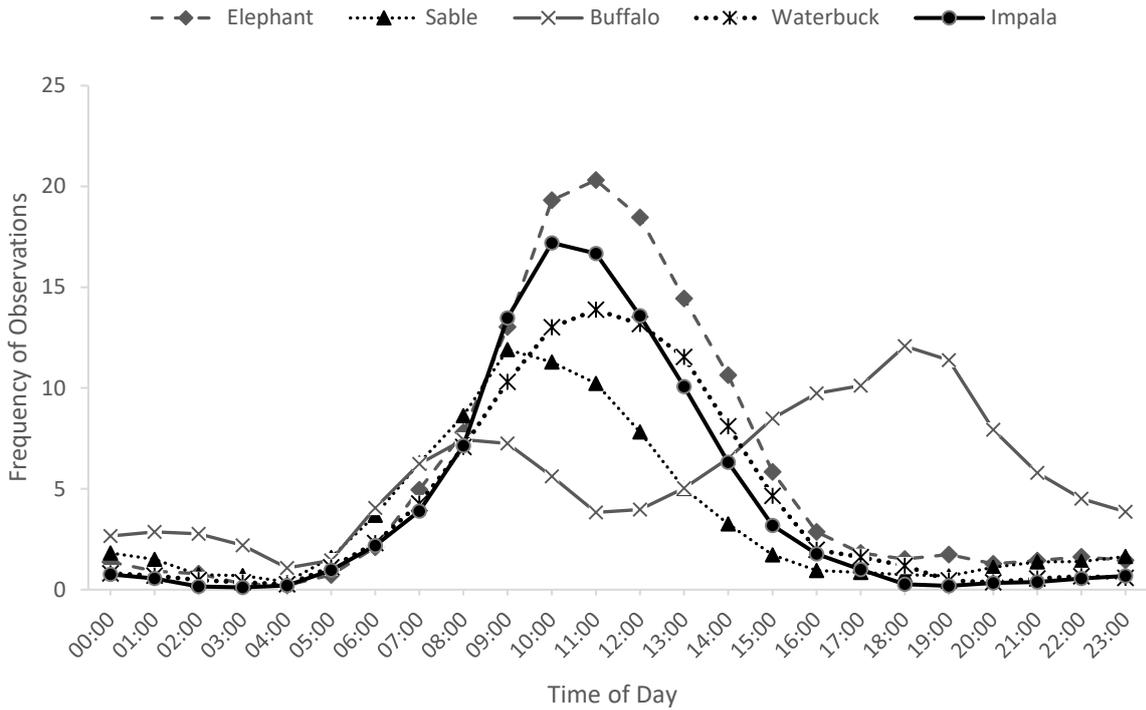


Figure 2.6 AWP utilisation observed for elephant, sable, buffalo, waterbuck and impala at the ten AWP. Raw data was divided by camera trapping effort to obtain observation frequencies.

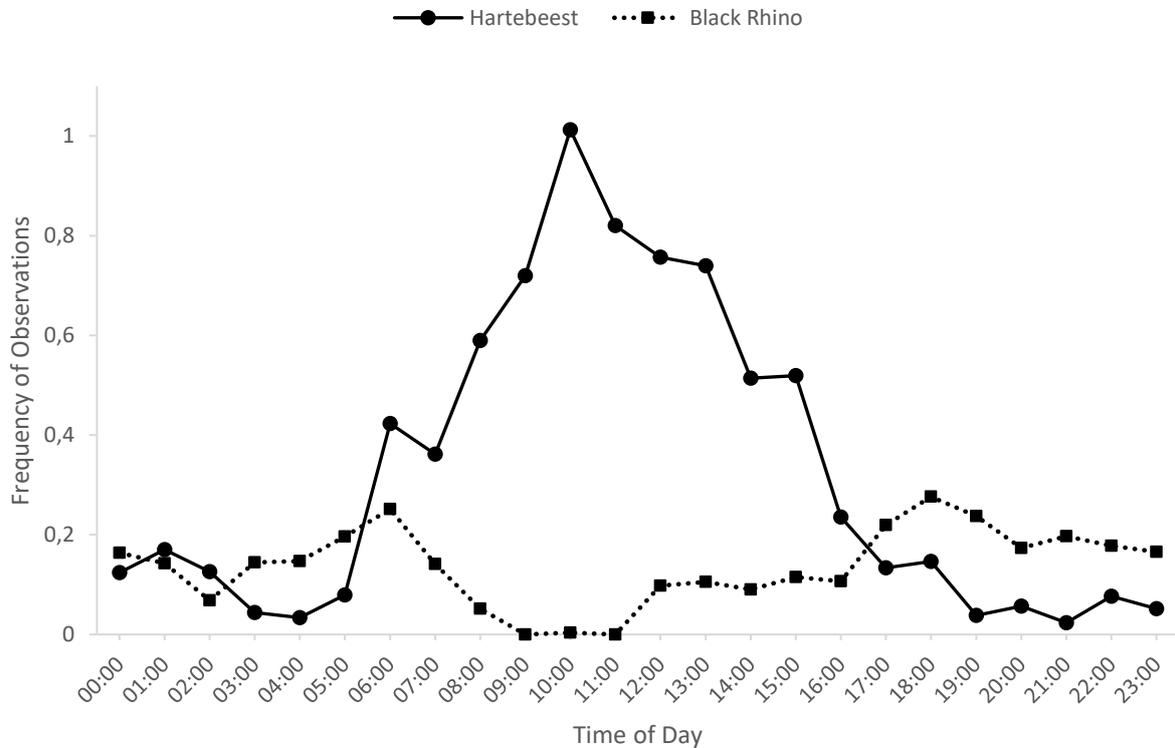


Figure 2.7 AWP utilisation observed for hartebeest and black rhino at the ten AWP. Raw data was divided by camera trapping effort to obtain observation frequencies.

The time of day buffalo preferred to visit AWP's differed significantly amongst the four seasons (Chi-square Test of Independence: $X^2_{(df = 69)} = 7018.4$, $p\text{-value} < 0.001$). The early dry season showed the strongest correspondence to 17:00, late wet to 21:00, early wet to 22:00 and late dry to 08:00. Elephants preferred to visit AWP's at certain times of day, differing significantly amongst the four seasons (Chi-square Test of Independence: $X^2_{(df = 69)} = 4022.2$, $p\text{-value} < 0.001$), with early dry season showing the strongest correspondence to 21:00, late wet to 17:00, early wet to 09:00 and late dry to 11:00.

The time of day waterbuck preferred to visit AWP's also differed significantly (Chi-square Test of Independence: $X^2_{(df = 69)} = 3346.4$, $p\text{-value} < 0.001$). The early dry season showed the strongest correspondence to 10:00, late wet to 23:00, early wet to 14:00 and late dry to 08:00. Lastly, zebra timing of waterpoint use also differed significantly (Chi-square Test of Independence: $X^2_{(df = 69)} = 20.798$, $p\text{-value} < 0.001$), with early dry season showing the strongest correspondence to 22:00, late wet to 14:00 – 14:59, early wet to 19:00 and late dry to 09:00.

2.5 Discussion

Water utilisation by wildlife differs between habitat, season and species (Cain, Owen-Smith, & Macandza, 2012; Crosmar et al., 2012; Hayward & Hayward, 2012). Regions experiencing harsh dry seasonal conditions or those more susceptible to drought often experience a higher water demand (Kasiringua et al., 2017). Of the ten AWP's within the reserve, six were visited by all 12 species. Heritage and Thawale are situated 100m from areas of human habitation (lodges and restaurants), and this may very well deter rare or shy species such as Lichstenstein's hartebeest and black rhino (Odendaal-holmes, Marshal, & Parrini, 2014). Phwadzi and Diwa AWP's are situated in the far south-western and north-western regions of the reserve. These regions consist of deep valleys and undulating hills with many small natural springs providing alternate areas for water utilisation throughout the year. The four AWP's in the southern region of MWR (Pende 1, Kakoma, Pende 2 and Nthumba) were visited more than the other six remaining waterpoints. This is most likely due to fewer available perennial water sources in the southern region of the reserve when compared to the two perennial rivers that are present in the northern region.

Overall AWP utilisation peaked between 10:00 to 11:00 which differs from previous studies conducted in the Waterberg National Park, Namibia, and Hwange National Park, Zimbabwe, reporting peak utilisation times between 15:00 and 22:00 (Kasiringua et al., 2017; Weir & Davison, 1965). Du Preez and Grobler (1977) studied waterpoint utilisation in Etosha National Park, Namibia and found a general peak in waterhole utilisation between 12:00 and 13:00, differing slightly from the current study results. Black rhino AWP utilisation in MWR peaked at dawn (06:00 – 06:59) and dusk (18:00 –

18:59). The latter corresponds to previous studies conducted by Ayeni (1975), Du Preez and Grobler (1977), Mukinya (1977) and Valeix et al. (2007).

According to the literature, elephants have been observed utilising waterpoints at various times of day. Many studies suggest that the peak waterpoint utilisation time for elephants is at dusk or during the cooler hours of the evening (Ayeni, 1975; Chamaillé-Jammes, Valeix, et al., 2007; Sungirai & Ngwenya, 2016; Valeix et al., 2007). Elephant AWP utilisation in MWR was similar to the results obtained from a study in the southern region of Kruger National Park (Sutherland, Ndlovu, & Perez-Rodriguez, 2018).

The results obtained for nyala and impala in this study were similar to those in a study conducted at five different sites in southern Africa, including one site in Botswana and four in South Africa (Hayward & Hayward, 2012). Zebra and kudu AWP utilisation patterns in MWR were similar to those in Etosha and Hwange (Du Preez & Grobler, 1977; Valeix et al., 2007). Buffalo AWP utilisation peaked at 08:00 and 18:00, proving similar to that of Buffalo in Hwange (Valeix et al., 2007). Hartebeest AWP utilisation within MWR was similar to waterpoint utilisation in Etosha (Du Preez & Grobler, 1977). Waterbuck, eland and sable AWP utilisation patterns observed in MWR differed from previous studies (Ayeni, 1975; Du Preez & Grobler, 1977; Hayward & Hayward, 2012; Kasiringua et al., 2017; Valeix et al., 2007; Weir & Davison, 1965). Very little research has been conducted on bushbuck AWP utilisation.

AWP utilisation differed between seasons. During the late dry season when natural water sources were diminished, AWP utilisation was at its highest for all species. This was the opposite for the early wet season when natural water sources were replenished, corresponding with previous studies (Ayeni, 1975; Cain et al., 2012; Valeix et al., 2007). Some of the AWPs contained large amounts of water in the erosion run-off. Therefore, animal visitations to this part of the AWP may be overlooked any could affect the overall results.

The preferential diurnal use of AWPs by herbivores in MWR may be due to the AWPs running on solar power, which results in optimum pumping of water when UV radiation is high, between 10:00 and 14:00. It may be that some herbivores have adjusted their peak utilisation times in order to visit AWPs when there was plentiful, free-flowing available water. However, this may need further and more detailed investigation to form any tangible conclusions. Occasional visits by herbivores after dark may be due to heat stress avoidance (Hayward & Hayward, 2012) as well as MWRs low predator density (Briers-Louw, 2017). Some AWPs in MWR often dry out during the evening as herds of species, such as buffalo, drink and bathe in the remaining water. This water is not replenished until the next morning when the solar pump is triggered by the sun. This provides a serious water limitation as there is a lack of water pumping and no batteries to store charge for pumping during the night.

Utilisation hotspots were not monitored along the two perennial rivers in the reserve and it therefore cannot be determined if these areas would be preferred over the AWP. Rivers provide an important source of forage and water, therefore, water-dependent herbivores such as buffalo, waterbuck and elephant may prefer utilising these river lines (Smit et al., 2007). Rivers are often preferred by browsers and mixed-feeders as there is good quality browse available as well as more cover for protection from predation (Smit et al., 2007). However, it is challenging to monitor extensive areas along river lines as it is not always easy to determine where along the river herbivores might prefer to drink.

In conclusion, this study demonstrates that herbivores utilise AWPs at a higher rate in the dry season, with emphasis in the south-eastern (Pende) region where perennial water is scarce. This research provides insight into the importance and benefits of AWPs in MWR. Future studies should focus on longer study periods to account for potential drought years and variations in rainfall which may affect AWP utilisation, water quality, differences between AWPs and natural waterpoints, and waterpoint types.

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Chapter Three

Understanding the impact of large herbivores on vegetation surrounding artificial waterpoints in Majete Wildlife Reserve, Malawi

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3.1 Abstract

Water availability is an important resource for herbivore populations in semi-arid ecosystems as these regions are more likely to experience water scarcity in the dry season. The construction of artificial waterpoints (AWPs) as water supplementation is a common population management tool for indigenous herbivores in these ecosystems. Intense vegetation utilisation is often visualised as gradients of disturbance surrounding the waterpoints, with a decrease of disturbance as distance from water increases and is often termed a piosphere. Understanding how the supplementation of artificial waterpoints shapes the landscape and influences biodiversity is important for reserve management. The aim of this study was to investigate to what extent large herbivores impact vegetation surrounding selected AWP in Majete Wildlife Reserve (MWR), Malawi. Vegetation was sampled along three 2000m long transects in N, SE and SW directions from each of the six AWP. Data were collected between July and November 2017, for woody vegetation, and between February and May 2018, for herbaceous vegetation. The step-point method was used to analyse herbaceous vegetation composition and cover in 10m² plots placed at 0m, 25m, 50m, 75m, 100m, 150m, 200m, 300m, 500m and 1000m from the AWP. For the woody vegetation analysis, eight 20m² plots were placed along the transects at 0m, 50m, 150m, 300m, 500m, 1000m, 1500m and 2000m from the waterpoint. Within these plots, vegetation cover was recorded in height categories <2m, 2-3m, 3-5m and >5m. The resultant Shannon-Weiner diversity index (H')($F_{(9, 169)}=6.22, p<0.001$), vegetation condition score (VCS)($F_{(9, 45)}=9.27, p<0.01$) and percentage woody vegetation cover ($F_{(7,35)}=7.04, p<0.01$) were significantly different with distance from water. There was a significant positive correlation ($R=0.423$,

$p < 0.01$) between plot distance from AWP, herbaceous vegetation cover and VCS ($R = 0.366$, $p < 0.01$). Bare ground was negatively correlated with plot distance from AWP ($R = -0.423$, $p < 0.01$). The results suggest that permanent AWP in MWR have led to the structural changes of woody and herbaceous vegetation adjacent to surrounding waterpoints, with vegetation cover and condition increasing with an increase in distance from water.

3.2 Introduction

Fenced reserves play an important role in conservation as they help reduce human-wildlife conflict, poaching and the spread of diseases from livestock to wildlife and visa-versa (Boone & Hobbs, 2004; Durant et al., 2015; Pirie et al., 2017). Fences may, however, negatively affect wildlife by preventing migration to areas abundant in essential resources, especially in the dry season when adequate forage and water availability is low (Hopcraft, Olf, & Sinclair, 2010; Oates & Rees, 2013; Skarpe, 1992; Tefempa et al., 2008). Resources such as forage and water availability therefore determine large herbivore behaviour and distribution within ecosystems (Ogutu et al., 2014; Redfern et al., 2003).

Food availability is one of the limiting resource for herbivore population growth in semi-arid ecosystems (Redfern et al., 2003; Western, 1975). Starvation induced mortality increases in protected areas with a high waterhole density. This is due to increased herbivore utilisation of regions at short distances between waterholes than those with increased distance from waterholes (Walker et al 1987). The construction of artificial waterpoints (AWP) for water supplementation is a common population management tool for indigenous herbivores in these ecosystems (Mwakiwa et al., 2012; Owen-Smith, 1996; Redfern et al., 2003; Thrash, 2000a). There are both natural ecosystem benefits and ecotourism benefits of AWP supplementation in situ. Networks of AWP are established within a reserve to stabilise dry season water availability, increase the range of dry season forage areas and provide wildlife viewing opportunities for ecotourism (Redfern, Grant, Gaylard, & Getz, 2005).

Additionally, ecotourism may benefit from increased herbivore populations due to water supplementation, however an overabundance of AWP can be detrimental to the health of an ecosystem as they may affect biodiversity (Harrington et al., 1999). The distribution patterns of herbivores change due to increased surface water availability which in turn influences vegetation composition and structure (Rosenstock, Ballard, & Devos, 1999). Herbivores in semi-arid savannas are attracted to this resource due to its limited availability, often resulting in over-utilisation via browsing, grazing and trampling in the vicinity of the waterpoint. The result of this increased utilisation will be a homogenisation of vegetation and a decrease in species diversity (Owen-Smith, 1996; Thrash, 1998), with serious consequences for biodiversity with emphasis on herbivores, ecosystem processes and resilience (Davidson & Funston, 2002; Owen-Smith, 1996; Walker, Emslie, Owen-Smith, & Scholes,

1987). An increase in water availability in certain areas results in an increase in water-dependent herbivores, as they are forced to reside within a 10km radius from a waterpoint in the dry season, proving costly to water-independent species (Redfern et al., 2005, 2003). Some rare antelope species, such as sable (*Hippotragus niger*), are sensitive to habitat changes which may result in a decrease in population numbers, as other less sensitive species gather around waterpoints, altering vegetation structure (Davidson & Funston, 2002; Grant & van der Walt, 2000; Harrington et al., 1999; Owen-Smith, Le Roux, & Macandza, 2013).

Vegetation over-utilisation is often visualised as gradients of disturbance surrounding waterpoints (Cronje et al., 2005; Landman, Schoeman, Hall-martin, & Kerley, 2012; Thrash & Derry, 1999; Thrash et al., 1993). This biological indicator is described as a localised zone that is visualised as gradients of disturbance surrounding waterpoints, expressing traits of decreasing disturbance as distance from water increases (Cronje et al., 2005; Thrash & Derry, 1999; Thrash et al., 1993). This gradient of disturbance is often termed a piosphere and is an ecological interaction between herbivores, a waterpoint and its surrounding vegetation. Variations in herbaceous species composition, range condition, biomass and basal cover have previously been found directly proportional with distance from water (Thrash & Derry, 1999). Soils with a high clay and silt content surrounding waterpoints are more susceptible to soil compaction, erosion and capping (Thrash & Derry, 1999). The effect of this disturbance is heightened if waterpoints are closely spaced, reducing biodiversity further with a consequence of reduced ecosystem resilience to drought and disturbance. For woody and herbaceous vegetation communities, changes seen with distance from water vary with proximity to other perennial water sources, herbivore numbers and rainfall (Parker & Witkowski, 1999; Thrash, 2000a).

Understanding how the addition of AWP shapes the landscape and influences biodiversity is important for reserve management. The aim of this study was to investigate to what extent large herbivores impact the vegetation surrounding selected AWP in Majete Wildlife Reserve, Malawi. Results from this study will aid in future artificial water management in the reserve, in particular when considering closing or providing additional AWP.

3.3 Methods

3.3.1 Study site

Majete Wildlife Reserve (MWR) is a 700km² protected area located in the Lower Shire Valley of southern Malawi (Figure 3.1). The Shire and the Mkulumadzi Rivers are the only perennial rivers flowing through the reserve and are both located in the north-eastern sector of the reserve. Majete Wildlife Reserve has an increasing altitudinal gradient from 100m a.s.l. in the east to over 900m a.s.l.

in the west. The western region is characterised by large rocky outcrops and steep hills with deep valleys, while the eastern region gently slopes down from the base of the steep western hills to the Shire River. The reserve experiences a gradient in rainfall from $\pm 700\text{mm}$ in the eastern lowlands to $\pm 1000\text{mm}$ in the western highlands. Average daily temperatures range from 23.3°C in winter (June to August) and 28.4°C in summer (December to February) (Spies, 2015; Wienand, 2013). The geology of the reserve includes rock formations of Precambrian Basement Complex gneisses and schists with overlying pockets of recent alluvial deposits (Sherry, 1989). Vegetation can be classified into four broad types including savanna, woodland $<250\text{m}$, woodland from $250 - 400\text{m}$, woodland $>400\text{m}$, with miombo savanna being the dominant vegetation type (Forrer, 2017).

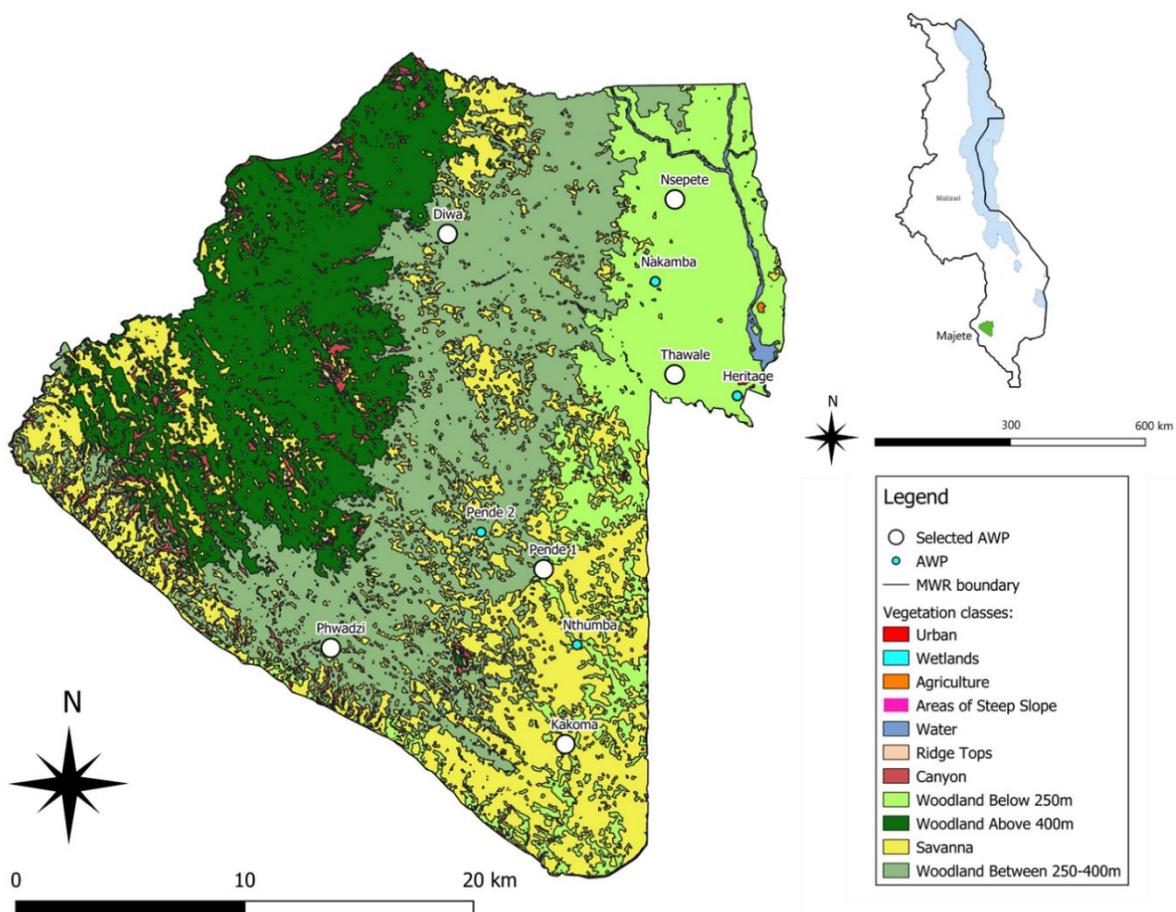


Figure 3.1 Location of the six selected artificial waterpoints (large white circles) in the different vegetation types of Majete Wildlife Reserve (MWR) situated in the southern region of Malawi (Shapefiles provided by African Parks (Pty) Ltd.).

3.3.2 Herbaceous vegetation cover

Herbaceous vegetation was sampled along three transects radiating from six artificial waterpoints between February and May 2018, during the grass flowering season for optimal species identification.

These transects were 1000m in length and radiated in the north (N), south east (SE) and south west (SW) directions from the six AWP. The well-known step-point method was selected (Mentis, 1981) as it provides an accurate determination of botanical composition and herbaceous vegetation cover.

A 10m² plot was surveyed via 100 points separated by 1m (each step) in a grid like pattern. These plots were placed at 0m, 25m, 50m, 75m, 100m, 150m, 200m, 300m, 500m and 1000m from the waterpoint so the linear distance of plots to waterpoints was used. A hand-held GPS (Garmin GPSMap 60CSx[®]) was used to record the plot locations and plot vegetation type and terrain was recorded. At each point the sampler lowered a sampling pin to the ground, guided by a definite notch in a boot. The sampler placed their boot at a 30° angle to the ground to avoid disturbing the plants. By lowering the pin perpendicularly to the boot, it would either come into contact with a herbaceous plant or the ground. Grasses were identified to species level while other herbaceous species were grouped as ‘herbs’ and ‘sedges’. The herbaceous species were identified and recorded, while samples of unknown species were collected for later identification (Fish, Mashau, Moeaha, & Nembudani, 2015; van Oudtshoorn, 1999). This method provided both species composition and estimated total ground cover of the herbaceous layer, whilst moving outward from an artificial water source, therefore allowing one to investigate the possible piosphere effect.

The Ecological Index Method was used to determine veld condition scores (VCS) as it has shown to be valuable for determining condition of the herbaceous vegetation layer and formulating best veld management practices (Barnes, Rethman, Beukes, & Kotze, 1984; Foran, Tainton, & Booysen, 1978; Teague, Trollope, & Aucamp, 1981; Trollope, 1986; Vorster, 1982). This involves grouping grass species into ecological categories based on their reaction to grazing pressure, biomass production and palatability (Appendix 2.1). The percentage composition of each ecological status category was calculated and multiplied with the specific ecological value allocated to that category. The sum of those values represents an ecological condition index with a theoretical maximum value of 1000. Grass cover was calculated from the percentage of herbaceous vegetation strikes and a relative index of density was provided by the percentage of bare ground.

The Shannon-Wiener index (H') was used to estimate herbaceous vegetation diversity per vegetation plot and was calculated as follows:

$$H' = -\sum p_i \ln p_i$$

Where p_i is the relative abundance of each species in the sample (Reyes, Kneeshaw, De grandpre, & Leduc, 2010). This index is sensitive to both species’ richness and abundance per species (Keller, 2002). A high H' value suggests high species diversity, a greater number of species and a higher proportion

of each species (evenness) within plots (Begon, Townsend, & Harper, 2006; Keller, 2002; Smith & Smith, 2001). The Shannon-Wiener index provides a rough measure of diversity or heterogeneity, providing a less biased approach by sample size than species richness.

3.3.3 *Woody vegetation cover*

Vegetation was sampled along three transects radiating from six AWP's between July and November 2017. Two AWP's were selected in each of the three vegetation types due to ease of accessibility. Nsepete and Thawale were located in woodland <250m, Diwa and Phwadzi in woodland >400m woodland, and Pende 1 and Kakoma in savanna vegetation. These transects were 2000m in length and radiated in the N, SE and SW directions from each waterpoint. Eight 20m² plots were placed along the transects at 0m, 50m, 150m, 300m, 500m, 1000m, 1500m and 2000m from the waterpoint. A hand-held GPS (Garmin GPSMap 60CSx[®]) was used to record the plot locations and plot vegetation type, dominant species and terrain was recorded. Within these plots, vegetation cover was visually estimated within 5% increments. Woody vegetation cover was estimated within each plot in height categories <2m, 2-3m, 3-5m and >5m. All plots were a minimum of 10m from a road, drainage line or river to account for any influence these variables may have had on the vegetation in that area.

3.3.4 *Statistical analysis*

All statistical analyses were performed using the lmer package of Rstudio version 3.5.2 (The R Foundation for Statistical Computing, 2018) as well as Statistica version 13.2 (Dell Software, 2018) to obtain mixed-effects repeated measure model estimates and to calculate descriptive statistics. A simple logistic regression was calculated to predict percentage cover of the various ecological statuses and bare ground based on distance from AWP.

Mixed Model ANOVAs were used to test the main effects of distance from water point (0m, 25m, 50m, 75m, 100m, 150m, 200m, 300m, 500m, 1000m) and transect direction (N, SW, SE) on the vegetation condition scores in plots. Mixed Model ANOVAs were used to test the main effects of distance from water point (0m, 50m, 150m, 300m, 500m, 1000m, 1500m, 2000m) and transect direction (N, SW, SE) on woody vegetation cover (<2m, 2-3m, 3-5m and >5m). All significant ANOVAs (assumptions of normality and homoscedasticity being met) were followed-up with Fisher's LSD post-hoc tests. For all the above-mentioned statistical tests the significance level was set at $p = 0.05$, unless otherwise stated.

Spearman's correlations were used to test the relationship between plot variables (distance from AWP and proximity to perennial water) and vegetation cover (<2m, 2-3m, 3-5m, >5m, bare ground,

herbaceous vegetation cover and VCS). The variable “proximity to other perennial water” excluded the AWP from which the transect started.

3.4 Results

3.4.1 Herbaceous vegetation

A total of 38 herbaceous species were recorded from 180 vegetation plots (Appendix 2.2). Logistic regression showed that distance from AWP impacted some of the ecological categories. Bare ground decreases dramatically with increasing distance from AWP ($p < 0.001$). The percentage cover of herb species decreases with increasing distance from AWP ($p < 0.001$). The percentage cover of Increaser I and Increaser IIa species increases with increased distance from water ($p < 0.001$).

The Shannon-Weiner diversity index (H') was significantly different with distance from water ($F_{9, 169} = 6.22$, $p < 0.001$) (Table 3.1, Figure 3.2). Post Hoc comparisons indicated that the mean H' for 0m from the AWP ($M = 0.76$, $SD = 0.38$) was significantly different to 100m ($M = 1.18$, $SD = 0.25$), 200m ($M = 1.27$, $SD = 0.36$), 300m ($M = 1.41$, $SD = 0.29$) and 1000m ($M = 1.44$, $SD = 0.39$) from the AWP. The mean Shannon-Weiner diversity index for 1000m ($M = 1.44$, $SD = 0.39$) from the AWP was significantly different to 25m ($M = 0.91$, $SD = 0.38$) and 50m ($M = 1.04$, $SD = 0.39$) from the AWP and the mean Shannon-Weiner diversity index for 300m ($M = 1.41$, $SD = 0.29$) from the AWP was significantly different to 25m ($M = 0.91$, $SD = 0.38$).

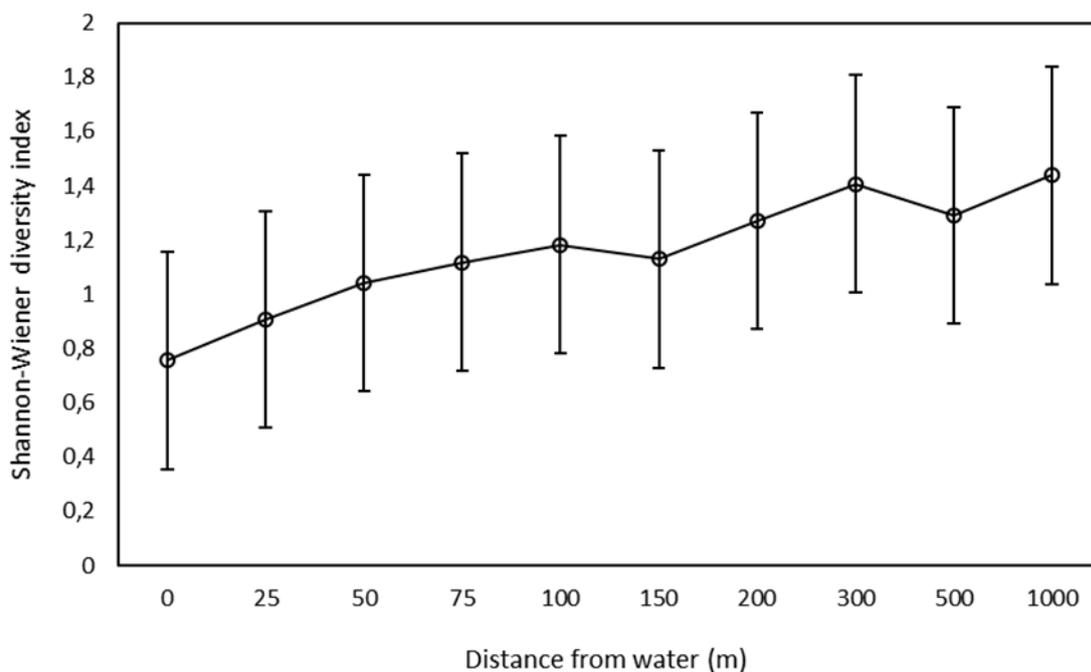


Figure 3.2 Shannon-Wiener diversity index as a function of distance from water (m) for plots surrounding AWP in Majete Wildlife Reserve, Malawi ($F_{(9, 169)} = 6.22$, $p < 0.001$). Vertical bars denote 95% confidence intervals.

VCS were significantly different amongst distances from water ($F_{9,45}=9.27$, $p<0.01$) (Table 3.1, Figure 3.3), with VCS at 0m from the waterpoint significantly different to all other distances (LSD $p<0.01$). VCS were significantly lower at 25m from the AWP compared to VCS at 100m, 300m, 500m and 1000m from the AWP. (LSD $p<0.05$). The VCS at 50m ($M=419.72$) and 150m ($M=419.22$) were relatively similar. Additionally, there was no significant effect of transect direction ($F_{2,10}=1.11$, $p=0.37$) on VCS. VCS at 500m from water were significantly higher than all other plot VCS (LSD $p<0.05$).

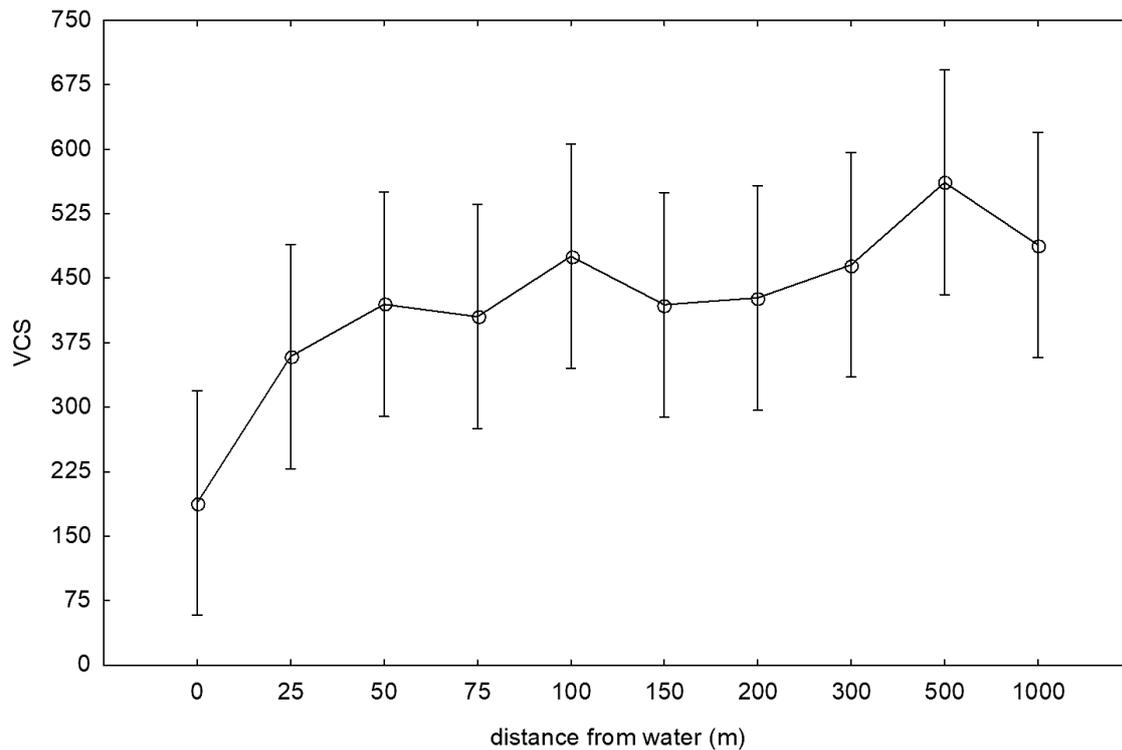


Figure 3.3 Vegetation condition score (VCS) as a function of distance from water (m) for plots surrounding AWP in Majete Wildlife Reserve, Malawi ($F_{(9,45)}=9.27$, $p<0.01$). Vertical bars denote 95% confidence intervals.

There was no significant interaction between distance from water and transect direction ($F_{18,90}=0.74$, $p=0.76$). Descriptive statistics showed that vegetation 500m from the AWP had the highest VCS ($M=561.5$, $SD=195.28$), compared to vegetation directly adjacent to the AWP (0m) [mean=188.61, $SD=126.43$] which had the lowest VCS. A veld condition score lower than 450 may reflect low grass cover, biomass production, unpalatable grasses and annual grass species. Subsequently, these results indicated poor veld condition for grazers.

Table 3.1 Attributes of herbaceous vegetation composition for plots (mean \pm standard error) with distance from artificial waterpoint (AWP) in Majete Wildlife Reserve, Malawi. Significance from one-way ANOVA tests displayed.

Variables	Category	Distance from AWP (m)										One-way ANOVA
		0	25	50	75	100	150	200	300	500	1000	<i>p</i>
Ecological status	Decreaser	9.11 \pm 3.14	16.11 \pm 3.62	22.44 \pm 5.11	11.67 \pm 3.16	23 \pm 5.59	20.5 \pm 3.0	16.06 \pm 4.12	22.72 \pm 3.45	23.22 \pm 1.78	19.39 \pm 4.4	0.178
	Increaser I	0.17 \pm 0.17	5.11 \pm 2.16	8.78 \pm 3.43	19.93 \pm 5.09	11.39 \pm 3.4	13.44 \pm 4.61	13.17 \pm 4.01	15.06 \pm 3.91	23.06 \pm 5.03	21.17 \pm 5.47	0.189
	Increaser II	7.33 \pm 2.2	10.11 \pm 4.3	6.94 \pm 2.03	7.6 \pm 2.73	6.89 \pm 3.22	2.39 \pm 0.91	3.61 \pm 1.57	5.72 \pm 1.70	3.61 \pm 1.51	3.33 \pm 1.07	0.086
	Increaser IIa	7.22 \pm 2.78	21.94 \pm 6.16	19.28 \pm 5.54	29 \pm 5.28	21.78 \pm 4.97	25.28 \pm 4.56	26.33 \pm 5.94	31.33 \pm 4.41	28 \pm 3.55	30.67 \pm 5.68	0.386
	Increaser IIc	26.89 \pm 5.66	31.89 \pm 8.38	30.89 \pm 7.4	28.93 \pm 8.56	26.44 \pm 5.88	32.72 \pm 7.34	31.5 \pm 7.05	22.11 \pm 5.17	19.56 \pm 5.36	22.83 \pm 6.58	0.936
Shannon-Wiener diversity index (H)		0.76 \pm 0.09	0.91 \pm 0.10	1.04 \pm 0.09	1.12 \pm 0.11	1.18 \pm 0.06	1.13 \pm 0.09	1.27 \pm 0.08	1.41 \pm 0.09	1.29 \pm 0.08	1.44 \pm 0.09	< 0.001
Vegetation condition score (VCS)		188.61 \pm 29.8	359.06 \pm 35.44	416.72 \pm 47.93	405.28 \pm 52.61	475.56 \pm 44.34	419.22 \pm 43.59	427.28 \pm 50.2	465.67 \pm 44.45	561.5 \pm 46.03	488.78 \pm 37.73	< 0.001
Percentage bare ground (%)		38 \pm 5.35	6.78 \pm 2.14	6.72 \pm 2.51	2 \pm 0.84	5.17 \pm 3.52	1.78 \pm 0.63	3.56 \pm 1.26	2.06 \pm 0.66	1.22 \pm 0.55	1.06 \pm 0.36	< 0.001

3.4.2 Woody vegetation

The percentage woody vegetation cover was significantly different amongst plot distance from water ($F_{7,35}=7.04$, $p<0.01$) (Figure 3.4), with percentage cover at 0m and 50m from the waterpoint significantly different to 300m, 500m, 1000m, 1500m and 2000m (LSD $p<0.05$) from the AWP. Percentage cover was significantly lower at 0m from the AWP compared to 150m (LSD $p=0$), and cover at 150m was significantly lower than that at 1500m from the AWP (LSD $p=0.02$). Once again, there was no significant main effect of transect direction ($F_{2,10}=0.02$, $p=0.98$) and vegetation height category ($F_{3,15}=1.40$, $p=0.28$) with woody vegetation cover.

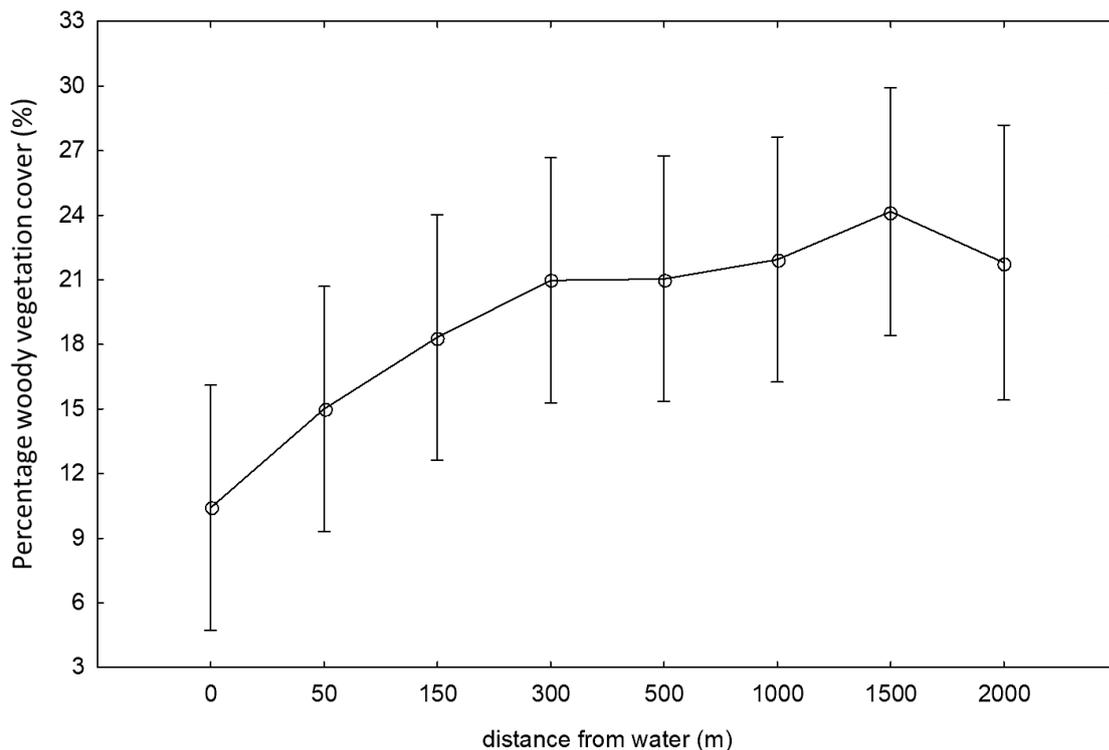


Figure 3.4 Estimated percentage woody vegetation cover as a function of distance from water (m) for plots surrounding AWP in Majete Wildlife Reserve, Malawi ($F_{(7,35)}=7.04$, $p<0.01$). Vertical bars denote 95% confidence intervals.

There was a significant interaction between distance from water and the vegetation height category ($F_{21,400}=1.76$, $p<0.05$) (Figure 3.5). Woody vegetation cover increased significantly between 0m and 2000m from an AWP for <2m (LSD $p=0.01$), 2-3m (LSD $p=0$) and 3-5m (LSD $p=0$) height categories. Vegetation cover >5m increased significantly between 0m and 1500m from water (LSD $p=0.05$).

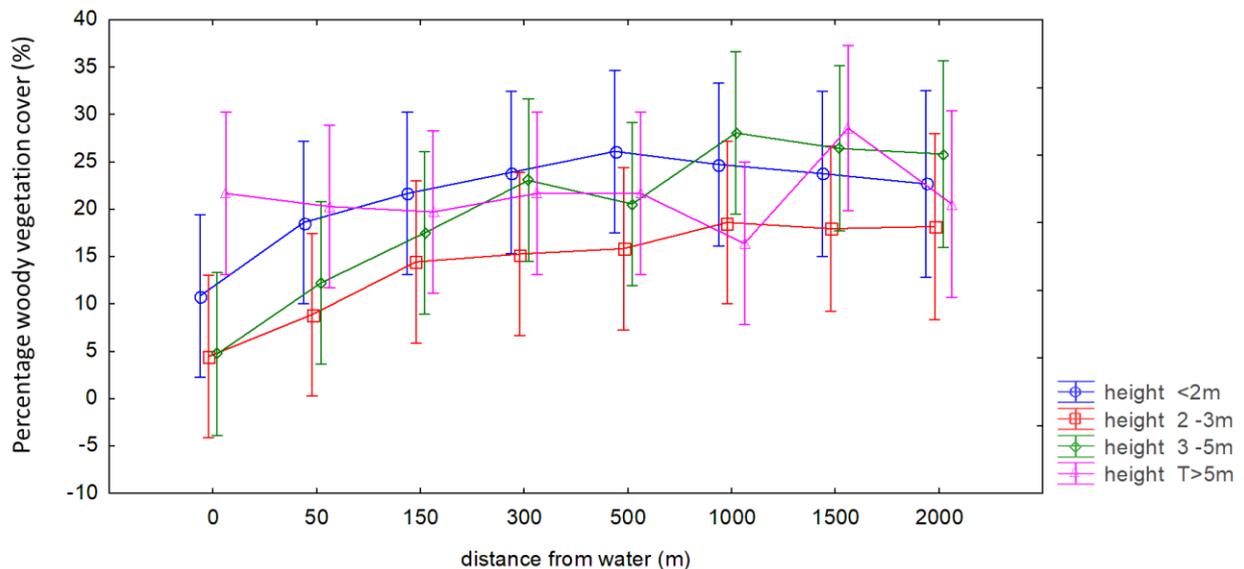


Figure 3.5 Estimated percentage woody vegetation cover for height categories as a function of distance from water (m) for plots surrounding AWP in Majete Wildlife Reserve, Malawi ($F_{(21,400)}=1.76$, $p=0.02$). Vertical bars denote 95% confidence intervals.

Descriptive statistics showed that 1500m from the AWP had the highest percentage vegetation cover ($M=23.53$, $SD=13.52$) compared to 0m ($M=10.42$, $SD=14.24$) which had the lowest percentage vegetation cover. Percentage vegetation cover was lowest for the 2-3m height category ($M=14.04$, $SD=12.03$) and highest for the <2m height category ($M=21.54$, $SD=12.05$).

3.5 Discussion

Surface water availability influences herbivore movements on a spatial and temporal scale (Shannon et al., 2009). This could affect the level of herbivore distribution and foraging, altering the vegetation on a landscape and habitat scale. Some researchers suggest that vegetation change present in protected areas is due to herbivore browsing and grazing pressure, with noticeable contributions by high impact species such as elephants (Sankaran, Ratnam, & Hanan, 2008; Smit et al., 2007; Thrash, 1998; Thrash, Theron, & Bothma, 1995). Elephants are mixed feeders, utilising the entire woody plant as a source of food, with increased utilisation usually evident along rivers and other permanent water sources (Laws, 1970). The aggregation of herbivores, including elephants, around AWP in the dry season results in increased trampling, browsing and grazing pressure in the immediate areas. This will in turn result in the suppression of seedling regeneration and plant growth (Senzota & Mtahko, 1990; Thrash & Derry, 1999).

The results from this study suggest that permanent AWP in MWR have led to structural changes of woody and herbaceous vegetation adjacent to the waterpoints, with vegetation cover and condition increasing with an increase in distance from water. These findings are consistent with the general assumption that herbivores change the structure of vegetation in the form of piospheres through

trampling and increased herbivory due to artificial water provision (Owen-Smith, 1996; Thrash et al., 1993).

3.5.1 *Herbaceous vegetation*

Herbaceous vegetation condition responded significantly to distance from water. The ecological index method using the veld condition score (VCS) was used to evaluate herbaceous vegetation condition. VCS were highest at 500m and 1000m from the AWP, while regions adjacent to the AWP (0m) were the lowest. Variations in VCS with distance from AWP were due to herbivores congregating in high numbers in the vicinity of water, therefore causing disturbance by means of trampling and foraging, decreasing VCS near AWP (Thrash, 2000a). High VCS are a result of a lack of disturbance and little utilisation by herbivores as intense grazing results in decreased grazing quality and biodiversity (Arsenault & Owen-Smith, 2002; Olff & Ritchie, 1998). This essentially shows that there is a sacrifice area in close proximity to the AWP consisting of bare ground caused by increased trampling (Thrash & Derry, 1999). There have been recordings of similar negative effects on grazing quality and biodiversity caused by a complete lack of grazing (Arsenault & Owen-Smith, 2002). One study found wildebeest grazing in the Serengeti increased grass density and stimulated the regrowth of leafy grass species (McNaughton, 1976).

A veld dominated by decreaser grass species produces a high VCS while those which are dominated by increaser I and II species result in a low VCS (Dyksterhuis, 1949; Foran et al., 1978). Increaser grass species are usually pioneer and sub-climax grasses which thrive and increase in overgrazed veld, and decreaser grass species are palatable, climax species which are sensitive to overgrazed or undergrazed veld (Dyksterhuis, 1949; van Oudtshoorn, 2002). Decreaser species are characteristic of good veld management while increaser species indicate poor veld management (Dyksterhuis, 1949; Foran et al., 1978). Sites closer to water were dominated by increaser II grasses, suggesting that these regions have experienced heavy grazing as these grass species are more tolerant to intense grazing (Dyksterhuis, 1949; van Oudtshoorn, 2002). Increaser I species increased significantly with distance from water. This was not expected as these species thrive in areas which are overgrazed such as those near water (Dyksterhuis, 1949; van Oudtshoorn, 2002). Previous studies have found that AWP with high animal activity and intense herbivore utilisation exhibit low percentage cover of increaser I and decreasers in general, with a high cover of increaser II species (Parker & Witkowski, 1999; Seithamo, 2011).

Having said this, VCS have many limitations when applied to ecological systems consisting of herbivores with differing forage requirements. The VCS system assigns herb and forb species low condition scores. However, these species increase ecosystem resilience and biodiversity in extreme

conditions such as drought and provide an important food source for browsers and mixed feeders (Grant et al., 2011, Siebert and Dreber, 2019). Therefore, assigning such robust scores to certain plant groups and species may not be entirely accurate as not all herbivores have the same forage requirements.

Grazing gradients may increase when the veld experiences continuous grazing pressure for extended periods of time, after the establishment of water sources (Tarhouni, Belgacem, Neffat, & Henchi, 2007). The lowest species diversity was recorded for 0m from water and increased with distance from water. Intense herbivore utilisation of the veld results in an increase in increaser I and II grass species. This in turn suggests decreased species diversity and increased homogenisation of the veld (Collinson, 1983; Noy-Meir, Gutman, & Kaplan, 1989; Olsvig-Whittaker, Hosten, Marcus, & Shochat, 1993). An unnaturally high density of AWP inhibits the rotation of utilisation pressure on grazing areas between seasons and therefore reduces habitat diversity (Collinson, 1983). In eastern Botswana, species diversity was found to decline closer to waterholes (Tolsma, Ernst, & Verwey, 1987), while in southern Australia species diversity did not display a distinct pattern and instead varied with distance from water (Landsberg et al., 1997).

The relationships between distance of plots from AWP to VCS were significant. This could be related to the low density of AWP in some areas, decreasing the chances of a potential overlap of transects from adjacent AWP. This will result in the formation of more distinct piospheres, with increased degradation close to the AWP (Brits et al., 2000; Owen-Smith, 1996; Thrash & Derry, 1999). Clegg (1999) found that species diversity was lower closer to water when compared to vegetation >200m from water, indicating a point of equilibrium which was reached. Similar results were obtained in this study as vegetation cover and condition increased slowly 100m from the AWP. This suggests that the piosphere is saturated to the region adjacent to the AWP.

3.5.2 *Woody vegetation*

Woody species cover tends to increase with increasing distance from water due to increased foraging and trampling on seedling recruitment triggered by increased herbivore concentrations with proximity to water (Brits et al., 2000; Chamailé-Jammes, Fritz, & Madzikanda, 2009; Roques, O'Connor, & Watkinson, 2001). An increase in trampling and grazing pressure may result in the increased establishment of young trees and shrubs, as competition with the herbaceous layer decreases (Roques et al., 2001; Smet & Ward, 2005; Smit et al., 2007). A study conducted in the Kruger National Park, South Africa, found a weak, although significant direct relationship between woody plant density and distance from water (Brits, van Rooyen, & van Rooyen, 2002). In Hwange National Park, Zimbabwe, Chamailé-James et al. (2009) found woody cover to be greatly affected within 2km of the piosphere

surrounding a water point, with woody cover decreasing closer to waterholes. A decreasing gradient of vegetation utilisation with distance from AWP was recorded in Tembe Elephant Park, South Africa (Augris & van Rooyen, 2010).

Browsers appear to have a marginal effect on the woody vegetation as they are less water-dependent than grazers, due to their forage consisting of a higher water content (Clegg, 1999; Western, 1975; Young, 1970). Less-mobile water dependent herbivores do not occupy areas out of their forage range during the dry season as they are not able to travel vast distances in search of water. In contrast, for large herbivores with greater mobility, foraging may occur up to 10km from water (Collinson, 1983).

In the current study extensive impact did not exceed 150m from the AWP, with the highest impact seen adjacent to the AWP (0m from AWP). This is most likely due to trampling and overcrowding of AWP, decreasing vegetation cover, increasing disturbance and percentage of bare ground. However, woody vegetation cover did increase with distance from water for all height categories. This general trend indicates less vegetation utilisation further from a water source. Some studies have suggested that high-intensity fires and increased levels of elephant impact may result in the homogenization of the woody vegetation community at a landscape scale (Eckhardt, Wilgen, & Biggs, 2000; Scholes, Bond, & Eckhardt, 2003). Establishing AWP eventually results in changes in the heterogeneity of habitats for herbivores, effecting species diversity and community composition (Knight, 1995; Owen-Smith, 1996).

This study provides substantial evidence that vegetation condition and cover varied considerably with distance from the AWP. The intensification of vegetation utilisation near AWP by large herbivores coincided with a negative trend in vegetation cover and condition. The impact of AWP on vegetation is most likely due to the placing of water in areas void of natural supplies or where water is merely seasonal (Joubert, 1976; Owen-Smith, 1996; Smuts, 1972; Weir, 1971). This impact is due to large water-dependent herbivores utilising areas in the dry season that were strictly natural foraging areas in the wet season.

3.6 Conclusion

Future studies should include other factors that may affect vegetation structure and condition such as: fire, droughts, frost, edaphic factors, topography, disease and previous human activity (Chafota & Owen-Smith, 2009; Guldemond & van Aarde, 2008). These factors may be interconnected, exerting a stronger effect when acting simultaneously. Previous studies have found that herbivory and fire have a synonymous effect on vegetation structure and can alter habitats by causing a decline in woody canopy and herbaceous plant cover (Gandiwa & Kativu, 2009; Tafangenyasha, 1997, 2001). High

browsing pressure surrounding AWP's reduces fuel load, resulting in decreased fire frequencies. There is scope for comparing vegetation structure and condition at AWP's to natural waterpoints within the reserve. Large herbivores tend to prefer natural waterpoints over AWP's, which may result in the vegetation surrounding natural waterpoints being subjected to high utilisation for many years prior to the establishment of the AWP's (Thrash, 1998; Young, 1970).

Therefore, it is recommended that continued monitoring of vegetation structure and condition within the plots surrounding the AWP takes place, in order to investigate the long-term impact of herbivores around AWP's in MWR. Future studies should include plots situated further than 2000m from water to establish the impact of AWP's at greater distances from water. In conclusion, vegetation experienced a greater impact by large herbivores with distance from water, with vegetation cover and condition decreasing close to AWP's.

3.7 Acknowledgements

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Chapter Four

A basic assessment of artificial waterpoint condition in Majete Wildlife Reserve, Malawi.

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4.1 Abstract

Large herbivores can change the structure and composition of their surrounding landscapes through vegetation utilisation on a spatial-temporal scale. Fixed-point photography is a valuable tool for evaluating short- and long-term vegetation and landscape change. The objective of this study was to use fixed-point photography to determine vegetation change surrounding artificial waterpoints (AWPs) between 2013 and 2018. A total of 166 photographs (84 in the wet season, 82 in the dry season) containing the AWPs and their immediate surrounds, were taken during 2013 in Majete Wildlife Reserve (MWR), Malawi. All sites were revisited in 2018, and the photograph pairs were compared and analysed for changes via a visual assessment. Photographs were rated according to the severity vegetation change with scores between -1 and -3 showing a reduction in vegetation, 1 to 3 showing an increase in vegetation and 0 for no change in vegetation. Vegetation reduction was highest in the south-eastern region and lowest in the sanctuary region. Erosion at each AWP was measured using tape measures and compared between 2014 and 2018. The severity of erosion increased at nine of the ten AWPs, six of which showed an increase in erosion in the form of gully erosion. The remainder of the AWPs had extensive rill erosion. In order to minimize erosion and over utilisation of vegetation, it is recommended that AWPs are fitted with a ball valve system and rotated annually within the network of AWPs. This study has shown that fixed-point photography can be an effective monitoring tool within MWR as it is relatively easy and inexpensive for reserve management. This method was able to determine and document short-term changes in vegetation cover surrounding AWPs within MWR and provided essential information needed for making informed management decisions.

4.2 Introduction

Large herbivores have the ability to physically change the structure of their surrounding landscapes through vegetation utilisation and their effect on ecological processes (De Garine-Wichatitsky et al., 2004). Herbivores are drivers of disturbance in ecosystems as they utilise the landscape and are thus a core focus in conservation (Fuhlendorf et al., 2008; Maron & Crone, 2006; Riginos & Grace, 2008; Schippers et al., 2014). Large herbivores such as elephant, zebra, buffalo and giraffe have contributed to such structural changes evident in the various African landscapes (Birkett & Stevens-wood, 2005; De Garine-Wichatitsky et al., 2004; Hamandawana, 2012; Morrison et al., 2016; Mosugelo et al., 2002; Riginos & Grace, 2008; Skarpe, 1990). Understanding factors affecting herbivore abundance and distribution is an important aspect of reserve management (De Garine-Wichatitsky et al., 2004). Such factors include vital resources such as food and water, which can determine the distribution and seasonal movements of herbivores (Seydack et al., 2012).

Water accessibility is a limiting resource for herbivore population growth in semi-arid ecosystems (Redfern et al., 2003; Western, 1975). Artificial waterpoint (AWP) construction for water supplementation is a common population management practice for herbivores in these ecosystems (Mwakiwa et al., 2012; Owen-Smith, 1996; Redfern et al., 2003; Thrash, 2000). These AWP are established within reserves to increase water availability during the dry season, increase dry season forage range and improve wildlife viewing opportunities for tourists (Redfern et al., 2005).

A high density of AWP can be detrimental to ecosystem health within reserves, thus effecting biodiversity (Harrington et al., 1999). Increased surface water availability influences the distribution patterns of herbivores, thus effecting vegetation composition and structure (Rosenstock et al., 1999). Herbivores in semi-arid savannas gather around AWP, resulting in increased over-utilisation via browsing, grazing and trampling. This results in the homogenisation of vegetation and a decrease in species diversity (Owen-Smith, 1996; Thrash, 1998). This has serious consequences for biodiversity, ecosystem processes and resilience (Davidson & Funston, 2002; Owen-Smith, 1996; Walker et al., 1987).

Reducing biodiversity loss through the preservation and conservation of natural areas are amongst the core principals of reserve management programmes (Bruner, Gullison, Rice, & Da Fonseca, 2001; Ehrlich & Pringle, 2008; Rodrigues et al., 2004). Protected area managers worldwide have to make important decisions with regards to anticipating and assessing the effect of land use impacts on biodiversity (Maiorano, Falcucci, & Boitani, 2008). Such management decisions are improved when using ecological data from reliable monitoring programmes which document details of environmental

change and their drivers over time (Gitzen, Millsbaugh, Cooper, & Licht, 2012; Scholes, Mace, Turner, Geller, Jurgens, & Larigauderie, 2008).

Ground-based fixed-point photography provides a platform for the comparison and analysis of changes between past and existing vegetation patterns. By comparing photographs, one can depict how vegetation has changed over time and develop a historical profile of the landscape which aids in the prediction of responses to future changes in similar areas (Butler & DeChano, 2001; Moseley, 2006; Pickard, 2002; Vale, 1987). This method of monitoring has been used to quantify vegetation change due to climate change as well as depicting trends in landscape vegetation change including land conversion, habitat fragmentation and tree encroachment (Veblen & Lorenz, 1988).

Semi-arid regions are highly vulnerable to water erosion due to sufficient rainfall to cause erosion, but insufficient to ensure adequate plant cover throughout the year (Elkins, Sabol, Ward, & Whitford, 1986; Pressland, 1973; Scotney & McPhee, 1991). Grass is essential for soil stabilisation and decreased erosion as root networks below the soil surface provide better stabilisation than the taproots of shrubs. The possibility of soil compaction and erosion is increased for bare ground as trampling by herbivores results in the recruitment of unpalatable annual grasses and forbs (Batey, 2009; Fatunbi & Dube, 2008; Hayes & Holl, 2003).

Erosion usually occurs in areas of high herbivore density, where increased trampling and grazing reduces the protective soil crust layers (Belnap & Gillette, 1998; Eldridge & Leys, 2003; James, Landsberg, & Morton, 1999). A loss of vegetation cover may be caused by erosion as the climatic variations are not absorbed when soil health is poor (MacGregor & O'Connor, 2002). Soil nutrients decrease in areas with increased trampling (Smet & Ward, 2005). These effects cause a reduction in plant biomass near the AWP (Ludwig, Wilcox, Breshears, Tongway, & Imeson, 2005).

Fixed-point photography provides a platform for spatial and temporal analysis of landscape and successional change in vegetation which is important for reserve managers (Moseley, 2006; Nichols, Ruyle, & Nourbakhsh, 2009). This method has aided in forest policy change due to the information it has provided, for example, in China (Moseley, 2006). A long-term repeat fixed-point photography monitoring programme was established in Majete Wildlife Reserve (MWR), Malawi, in 2013 but the effectiveness and value of the results have not yet been analysed. The main objective of the current study was to quantify and identify changes in vegetation cover between 2013 and 2018 from fixed-point photographs taken at AWPs. This study was important as it will determine if fixed-point photography should be recommended to reserve management as an appropriate monitoring tool for vegetation change over short and long-time frames.

4.3 Methods

4.3.1 Study site

Majete Wildlife Reserve (MWR) is a 700km² fully fenced, protected area located in the Lower Shire Valley of southern Malawi (Figure 4.1). The altitudinal gradient of the reserve increases from 100m above sea level (a.s.l) in the east along the rivers to over 900m a.s.l in the west. The western region consists of large rocky outcrops and steep hills with deep valleys. The eastern region gently slopes downwards from the base of these steep western hills to the Shire River. Annual rainfall in the reserve ranges between 680 and 1000mm, with the highest rainfall occurring in the western highlands. Average daily temperatures range from 23.3°C in winter (June to August) to 28.4°C in summer (December to February) (Spies, 2015; Wienand, 2013). The geology of the reserve consists of rock formations including Precambrian Basement Complex gneisses and schists with overlying pockets of recent alluvial deposits (Sherry, 1989).

Vegetation is dominated by miombo woodland and can be classified into four broad types including woodland <250m, woodland between 250 – 400m, >400m and savanna (Forrer, 2017). Two perennial rivers are present in the reserve, the Mkhulumadzi and Shire, bordering the north eastern boundary of the reserve. Water availability is dependent on seasonality and is sourced from six non-perennial springs, five perennial springs and ten artificial waterpoints (AWPs). The solar-powered AWPs are borehole-fed and have been placed throughout MWR with four located in the sanctuary, four in the south-east, one in the north-west and one in the south-west region (Figure 4.1).

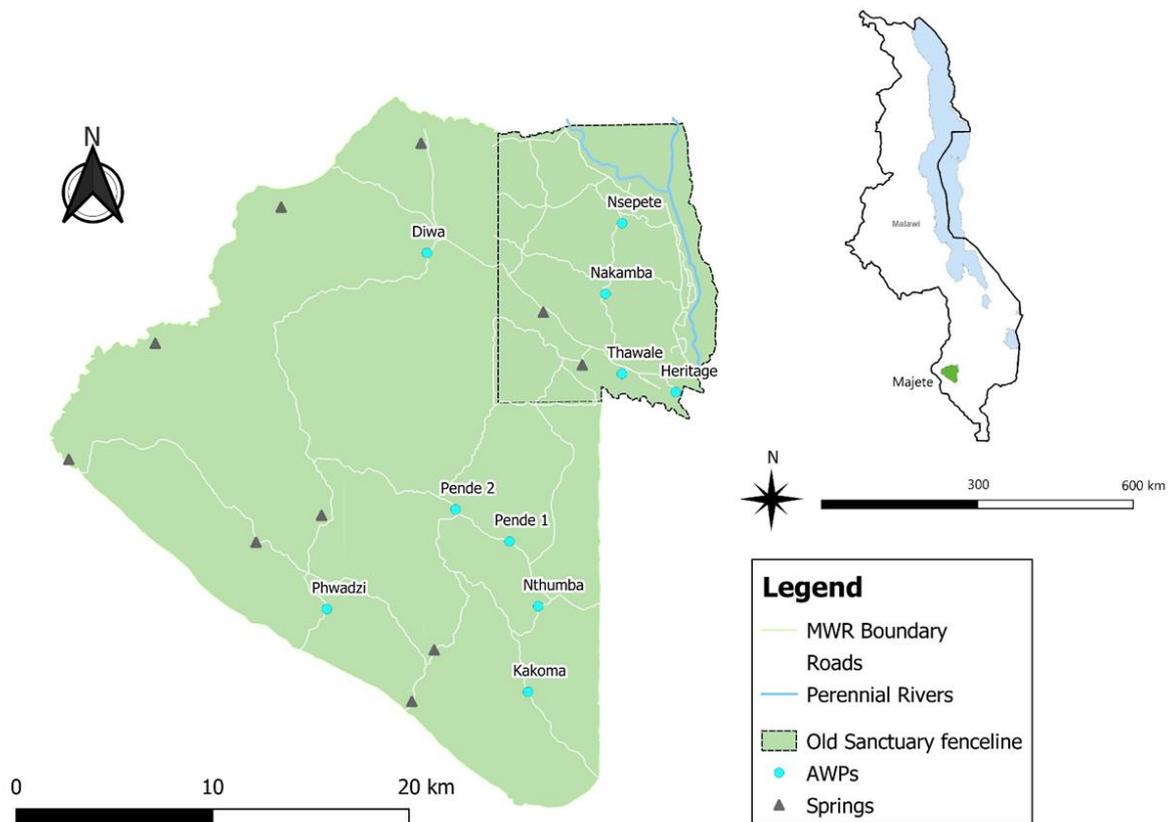


Figure 4.1 Location of the ten artificial waterpoints in Majete Wildlife Reserve (MWR) situated in the southern region of Malawi (Shapefiles provided by African Parks (Pty) Ltd.).

4.3.2 Fixed-Point photography

Fixed-point photography is a useful and reliable method for the long-term monitoring of vegetation (Hall, 2002). This involves taking photographs of the landscape or vegetation from the same standpoint (height, direction, zoom) at the same time of year at regular intervals. This provides a visual record which may be subjected to an objective analysis at a later stage. Photographs were taken at all ten AWP using a 700D canon camera fitted with a 15 - 55 mm canon lens during March (wet season) and November (dry season) of 2013 and 2018 (Appendix 3.1). The number of photographs taken at each AWP varied, however, this number included four photographs taken facing north and moving clockwise to capture photos in the north, east, south and west directions

4.3.3 Soil erosion

The length, width and depth of erosion present at AWP was measured using a tape measure during the wet season of 2014 and 2018. Erosion was also described and classified as sheet, rill or gully erosion (Appendix 3.2).

Sheet erosion is formed by removing the top layer of soil over a large area and might not be immediately noticed (Bergsma, Charman, Hurni, Moldenhauer, & Panichapong, 2000). Rill erosion forms as runoff water creates small channels as it flows downward, concentrating at the bottom of a slope. Such small channels reach up to 0.3m in depth and form gulley erosion if they exceed 0.3m (Bergsma et al., 2000). Gulley erosion develops in watercourses or other places with concentrated runoff. The depth of gulley erosion is usually restricted by the underlying rock material, which usually limits them to less than 2.0m deep (Bergsma et al., 2000).

4.3.4 Data analysis

Photos were analysed via visual assessment, with one observer comparing photos taken in 2013 and 2018 for each waterhole (except Heritage). These photos were allocated a score according to the change in vegetation cover observed (Table 4.1). A mean score was then calculated for each waterhole to determine overall change in vegetation between 2013 and 2018.

Table 4.1 Vegetation cover change scores and descriptions allocated to each of the four fixed-point photos at the AWP in MWR.

Score	Description
3	large increase in cover
2	moderate increase in cover
1	slight increase in cover
0	no change
-1	small reduction in cover
-2	moderate reduction in cover
-3	large reduction in cover

Fixed point photography within a long-term monitoring programme may be negatively affected by observer bias, which may influence the estimation of vegetation cover from photographs (Symstad, Wienk, & Thorstenson, 2008). To avoid this, vegetation cover analysis was conducted by a sole observer. However, this could not be avoided during the field work as multiple trained observers took photographs between 2013 and 2018.

4.4 Results

Change in vegetation cover was highest at Pende 1 and Nthumba, both situated in the south-eastern region of the reserve with a moderate reduction in cover observed (Table 4.2). No change in vegetation cover between 2013 and 2018 was observed at Thawale and Nsepete, both situated in the Sanctuary (north-eastern region) (Table 4.2). A slight reduction in cover was observed at five AWP. A large reduction in vegetation cover and any increase in vegetation was not observed at any of the AWP.

Table 4.2 Mean scores for change in vegetation cover between 2013 and 2018 for nine AWP in Majete Wildlife Reserve, Malawi.

AWP	Mean score
Thawale	0
Nsepete	0
Nakamba	-1
Diwa	-1
Phwadzi	-1
Kakoma	-1
Pende 2	-1
Pende 1	-2
Nthumba	-2

Soil erosion at eight of the ten AWP sites deteriorated over time (Table 4.3). Little erosion was evident at Nthumba AWP in 2014. However, by 2018, extensive gulley and rill erosion was recorded extending up to 110m from the waterpoint and ended in a large 40m wide mud pool. In 2014, Pende 1 had the most extensive erosion of all the ten studied sites. The erosion gulley was over 100m long, up to 5m wide and 1.5m deep. The concrete water-basin edge had collapsed as a result of the erosion. In 2018, extensive erosion was still present at Pende 1. Efforts attempting to rehabilitate erosion by filling and levelling out the gulley erosion via mechanical methods failed in 2017, due to a continued water runoff problem. The next attempt at rehabilitating erosion should be conducted after the water overflow issue has been resolved to avoid the same result as in 2017. The gulley erosion measured in 2018 extended 240m from the AWP reaching a depth of 0.65m in some areas.

Table 4.3 A basic description of erosion for 2014 and 2018 at the ten AWP sites within Majete Wildlife Reserve, Malawi.

AWP	2014	2018
Heritage	Severe gulley erosion & chance of water-basin-edge loss	Severe gulley erosion due to drainage line and slight water-basin-edge loss
Pende 1	100m long deep gulley erosion, Basin edge overflow	240m long gulley erosion, 7m wide, 0.65m deep
Nsepete	45m deep gulley erosion	59.3m long rill erosion into drainage line, 10m wide, 0.3m deep
Nakamba	25m rill erosion	36m long rill erosion into drainage line, 3m wide, 0.35m deep
Diwa	90m of rill erosion, water basin overflow	66m long rill erosion, 4m wide, 0.3m deep
Nthumba	No significant erosion issues observed	110m long gulley erosion, 40m wide, 1m deep
Thawale	Mild erosion around basin edging	173.3m long gulley and rill erosion, 13m wide, 1.2m deep
Phwadzi	130m shallow gulley erosion above (east of) road, 40m shallow gulley erosion below (west of) road	147.4m long gulley erosion ends at a lower pool, 14.8m wide, 0.45m deep
Pende 2	No significant erosion issues observed	28m long rill erosion into the river, 3.5m wide, 1.2m deep
Kakoma	No significant erosion issues observed	37m long rill erosion, 3.1m wide, 1.2m deep

Little erosion was observed at Pende 2 and Thawale during 2014, with most extensive erosion observed in the form of sheet erosion. By 2018, Pende 2 had a 28m long rill erosion area flowing to a river which was most likely due to excessive water runoff. This reached a considerable depth (1.2m) in some areas. Thawale showed an extensive increase in erosion by 2018. Gulley and rill erosion were

present, extending 173.3m from the AWP in the form of runoff and large mud wallows. These wallows were 13m wide and 1.2m deep in some areas.

In 2014, the concrete water-basin at Nakamba showed some erosion and undercutting due to water overflows, resulting in a considerable amount of soil loss. By 2018, Nakamba had 36m long rill erosion areas ending in a natural drainage line. Nsepete waterpoint had a long section of gulley erosion (45m), with multiple terraces in 2014. This erosion ended in a 1.5m deep gulley opening into an erosion fan formed as a result of the excessive water overflow. There was an attempt to halt edge erosion in 2017 that failed, primarily due to the constant volume of water overflowing the concrete basin. By 2018, the gulley erosion was 59.3m long with a maximum depth of 0.3m. One area had a maximum width of 10m in a form of a mud wallow. The Heritage AWP experienced some undercutting of the concrete basin in 2014. There was a considerable amount of erosion downslope from the visitor centre. By 2018, the erosion at Heritage was very similar to that of 2014. However, the concrete water basin edge had deteriorated, resulting in undercutting erosion surrounding the basin. Continued sheet and rill erosion were present on the side of the stream bank due to runoff from the slope up to the visitor's area.

In 2014, Diwa AWP had approximately 90m of rill erosion downstream from the concrete water-basin overflow, with 3 to 4 erosion terraces. By 2018, a 66m long rill erosion area was recorded with a maximum depth of 0.3m, which was due to a data error from the 2013 survey. In 2014, the Phwadzi AWP consisted of a non-concrete natural area filled by a solar-powered borehole pump and there was approximately 130m of rill erosion downstream from the waterpoint. By 2018, extensive gulley erosion was present and extended up to 147.4m from the AWP. These included several mud pools with runoff, some reaching depths of 0.45m and widths of 14.8m.

4.5 Discussion

Fixed-point photography enables the analysis of changes between historical and existing vegetation patterns (Butler & DeChano, 2001; Moseley, 2006; Pickard, 2002; Vale, 1987). Vegetation cover is one of the most frequently used indicators of change in ecosystems (Godínez-Alvarez, Herrick, Mattocks, Toledo, & Van Zee, 2009). Herbivore density remains moderately low in the western region of MWR, whereas higher densities were documented along the perennial rivers and near perennial waterpoints. Due to African Parks' management interventions wildlife densities within the reserve increased substantially between 2003 and 2018 (Appendix 1). Ecosystem management requires an extensive understanding of landscape use by herbivores (Bailey et al., 1996). Herbivores disturb ecosystem structure and processes through manipulating vegetation cover and productivity

(Tanentzap & Coomes, 2012). Understanding the spatial-temporal dynamics of herbivory is therefore important on the landscape scale within MWR.

In this study, there was considerable variation in total vegetation cover between AWP sites. Vegetation score changes at some AWPs within MWR were evident for over a short five-year period. A moderate reduction in vegetation cover was recorded for Pende 1 and Nthumba. These AWPs are situated in the south-eastern region of the reserve which is water scarce. Therefore, wildlife in this region rely on these two AWPs immensely during the dry season. This increased pressure by herbivores results in a reduction in vegetation cover surrounding the AWPs.

For Diwa AWP, a small reduction in vegetation cover was recorded in 2018. This AWP is situated at a relatively high altitude in the north western region of the reserve, where perennial water is relatively scarce. As herbivore population sizes have increased since 2013, as mentioned above, wildlife has spread throughout the reserve to access more suitable forage areas. This may be why Diwa has experienced a small decrease in vegetation cover in the last five years. This may be the same for the Phwadzi AWP, however, this AWP is exposed to more extensive fires than other AWP areas during the dry season due to its close proximity to the fence line and surrounding villages. This requires further investigation in future studies. Kakoma is a relatively new AWP (established in 2014) and therefore changes in vegetation may not be as robust and evident as those established before 2010 (Nakamba, Nsepete, Thawale, Pende 1, Pende 2).

There was no reduction in vegetation cover at Nsepete and Thawale, situated in the sanctuary region. This region has experienced extensive pressure from herbivores compared to the remaining regions as it consists of a high density of AWPs. Long term pressure from herbivores homogenises vegetation and vegetation change by large herbivores may not be as noticeable over a short time period (Graetz & Ludwig, 1978).

Grazing around AWPs results in a sacrifice zone within 500m of the water's edge. This region usually consists of broken soil crusts, increased erosion and unpalatable herbaceous vegetation (James et al., 1999; Thrash et al., 1995). Erosion was present at all the monitored waterpoints, some of which was more extensive than others. Erosion seen forming at AWPs is the result of excessive water flow from the solar powered boreholes and resultant overflow of the AWP pools. Water overflow from the AWP concrete basins has led to undercutting and cracking or breaking of the basins edge, as there is insufficient consolidation of soil material surrounding the AWPs. Downslope from the basins overflow point there is often a substantial length of rill and gully erosion. An increase in water runoff from

AWPs results in changing vegetation patterns as grass seedling growth is limited (Chartier, Rostagno, & Pazos, 2011).

AWP age and limited large game numbers in the area, may explain the decreased/reduced erosion damage that is evident at Kakoma and Diwa AWP sites when compared to the other AWP sites in this study. The topography is relatively flat in these areas, thus reducing flowrates and resultant erosion. In the case of the Heritage AWP, a considerable amount of gulley erosion on the side of the stream bank was experienced due to runoff from the slope up to visitor's center. The vegetation on this slope has been removed to open the view to the AWP. Natural measures to slow or divert rainfall runoff from continuing straight down slope have therefore been reduced, increasing the impact on the AWP.

The erosion surrounding and caused by AWP sites can be relieved by gabion constructions, stone packing and the use of fallen logs and tree trunks from the immediate surrounds. In many instances the reduction of water flow in the wet season would significantly help to reduce the rates of erosion. As the AWP sites are solar powered and water run-off therefore continuous on sunny days, a long-term solution would be the installation of ball-valves that prevent pumping when the required levels are reached. This would greatly reduce the development of the erosion problems that were observed, and reduce wastage of groundwater resources, especially in the wet season. The erosion problems will increase over time and corrective actions will cost considerably more in the future if no remedial work is undertaken in the near future.

In future, areas proposed as potential AWP sites can be monitored via fixed-point photography to determine the changes in vegetation taking place around AWP sites before and after placement (de Beer, Kilian, Versfeld, & van Aarde, 2006; Tafangenyasha, 1997). In order to minimize erosion and increased herbivore pressure on vegetation, AWP sites must be annually rotated within the network of AWP sites (Ayeni, 1975; Scholes et al., 2003; Thrash, 1998; Todd, 2006). This will involve the rotational closing and opening of various AWP sites to relieve pressure from certain areas.

A set of fixed-point photographs should always be accompanied by a more detailed data collection and description of the site itself. This was lacking for this study's data set (except for the Kakoma AWP) and should be incorporated in future surveys. Including vegetation survey data at each site adds depth to the analysis to monitor changes in range condition, species diversity and ecosystem functions (Lucey & Barraclough, 2001). Metadata such as rainfall, stocking rates and management interventions are essential for photo-monitoring programmes as they may aid in the clarification of landscape changes. Fixed-point photography studies with shorter observation periods of 5 years are preferred

as they may capture the dynamic nature of ecosystems in semi-arid environments (Masubelele, Hoffman, Bond, & Burdett, 2013).

Biodiversity monitoring frameworks are needed within protected areas in southern Africa (Reyers & McGeoch, 2007; Scholes et al., 2008). Reserve managers are using photographs more often to advise and implement policy change (Hall, 2000; Hendrick & Copenheaver, 2009; Nichols et al., 2009). Long-term monitoring will help document overall landscape change compared to short-term variability in ecosystems. The drivers of this change may be determined by the addition of wildlife census records and weather data. This information is essential for the daily and long-term management decisions and plans made by reserve managers (Hendrick & Copenheaver, 2009; Munson, Webb, & Hubbard, 2011; Peters, Yao, Sala, & Anderson, 2011; Webb, Belnap, Scott, & Esque, 2011).

This study has shown that fixed-point photography is a relatively easy and inexpensive monitoring tool which is valuable when regularly updated and archived. This basic monitoring method conducted over a five-year period at ten AWP's in the MWR was able to determine and document short-term changes in vegetation cover surrounding AWP's. This shows potential for an extensive long-term monitoring program with additional sites spread throughout the reserve.

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Chapter Five

Research findings, conclusions and management recommendations

5.1 Overview

The supplementation of water in semi-arid regions is essential for wildlife conservation and ecosystem management. This has implications for the conservation of vegetation and herbivores within Majete Wildlife Reserve (MWR). This study essentially investigated the impacts of large herbivores on artificial waterpoints (AWPs) and the surrounding vegetation within MWR. Herbaceous and woody vegetation structure and condition was explored for selected AWPs; along with vegetation change over a five-year period using fixed-point photography. Herbivore AWP utilisation was explored in detail. The findings from this study have provided a basis for understanding the ecological impacts of large herbivores at AWPs within MWR and will assist with the development of a viable and effective Conservation Management Plan in order to better manage the AWPs. In this report, the major findings of this study are summarised and their implications for water management within MWR are further discussed in detail.

5.2 Research findings

5.2.1 Chapter Two: Artificial waterpoint utilisation by selected herbivores

- AWPs were utilised more in the dry season, specifically the late dry season (September - November) which is as a result of a depletion of surface water availability in the reserve.
- Pende 1 was the most frequently visited AWP (12.94%) and Phwadzi was the least visited AWP (0.86%). In general, the most utilised AWPs were found in the Pende region. Perennial water sources are limited in the Pende region and therefore herbivores heavily rely on the AWPs as a source of water.
- Black Rhino (*Diceros bicornis minor*) were not observed visiting the Heritage, Phwadzi and Thawale AWPs. Lichtenstein's hartebeest (*Alcelaphus lichtensteinii*) did not visit Heritage and Phwadzi AWPs, and Nyala (*Tragelaphus angasii*) were not observed at Diwa and Phwadzi AWPs. The remaining study species were observed at all ten AWPs. This could be due to shy species avoiding human habituated areas such as lodges and restaurants situated close to the Heritage and Thawale AWPs.

- All species AWP use peaked between 10:00 – 11:59 and was lowest between 03:00 – 04:59. This peak in drinking time may be a result of running on solar power, with an optimum pumping of water occurring when UV radiation is high between 10:00 and 14:00. Herbivores have adjusted their utilisation times to ensure visits occur when water is plentiful, free-flowing and available.
- Heat stress avoidance may be a reason for the occasional visit by herbivores after dark.
- A serious water limitation issue is present in the reserve as AWP's dry out during the evenings, when water stops pumping and large animals such as black rhino and buffalo wallow in the remaining water. This water is not replenished until the following morning when the solar pump is triggered by the rising sun.
- This study demonstrates that AWP's provide an important source of water for herbivores in the dry season.

5.2.2 Chapter Three: Herbivore impact on vegetation surrounding artificial waterpoints

- Herbaceous vegetation cover, Veld Condition Score (VCS) and woody vegetation cover increased with distance from AWP, while Bare ground decreased with distance from AWP. This demonstrates the sacrifice zone theory, as AWP's experiencing high utilisation by herbivores results in increased trampling and grazing. This creates an area devoid of grass surrounding the AWP.
- Percentage woody vegetation cover was lowest for the 2-3m height category and highest for the <2m height category. This suggests that medium size trees are not as common possibly due to utilisation by large herbivores such as elephants.
- The impact on vegetation cover is more prominent in the Pende region, at Pende 1 AWP specifically. This is due to Pende 1 experiencing the most utilisation by herbivores (see Chapter two) in all seasons.
- Increased browsing and grazing pressure surrounding AWP's reduces fuel load, resulting in decreased fire frequencies. Going forward, it would be beneficial to determine how this effects the fire regime and management of the reserve.
- Large herbivores tend to prefer natural waterpoints over AWP's, which may result in the vegetation surrounding natural waterpoints enduring high utilisation for many years prior to the establishment of the AWP's.

5.2.3 Chapter Four: Using fixed-point photography to determine artificial waterpoint condition

- Soil erosion at eight of the ten AWP sites has deteriorated substantially, by increasing in length and depth, forming extensive rill erosion and gulleys.
- Pende 1 and Nthumba AWPs have high levels of herbivores utilisation (Chapter two) which have resulted in a moderate reduction in vegetation cover and an increase in erosion over time.
- The water overflow issue and erosion surrounding the AWPs needs to be urgently addressed. A long-term solution would be the installation of a ball-valves that prevent pumping when the required levels are reached. This would greatly reduce the development of the erosion problems that were observed, and reduce wastage of groundwater resources, especially in the wet season. Once the overflow issue is resolved, the erosion can be relieved by gabion constructions, stone packing and the use of fallen logs and tree trunks from the immediate surrounds.
- In order to minimize erosion and increased herbivore pressure on vegetation, AWPs must be annually rotated within the network of AWPs. This will involve the rotational closing and opening of various AWPs to relieve pressure from certain areas.
- Long-term monitoring will help document vegetation and overall landscape. The drivers of this change may be determined by the addition of wildlife census records and weather data. This information will be useful for the daily and long-term management decisions and plans made by reserve managers.
- Fixed-point photography is a relatively easy and inexpensive monitoring tool which may be valuable when trying to understand vegetation change over time. This method shows potential for an extensive long-term monitoring program with additional sites spread throughout MWR.

5.3 Management recommendations

The incorporation of knowledge on resource utilisation and impact of herbivores is integral for future wildlife conservation and the management of protected areas (Grant et al., 2011; Parker & Witkowski, 1999; Wienand, 2013). Water supplementation in the form of AWPs is regularly used as a management technique by conservation agencies and reserve management in semi-arid regions (Smit, Grant, & Devereux, 2007). Fenced reserves with AWPs require management in order to maintain habitat heterogeneity and control the impacts of herbivores on vegetation (Chamaillé-Jammes, Valeix, et al., 2007). This manipulation of water supplementation has multifaceted impacts on the ecosystem,

which is mostly evident in the vegetation surrounding AWP (Gaylard et al., 2003; van Aarde, Jackson, & Ferreira, 2006). The supplementation of AWP in reserves has been reported as detrimental for the surrounding vegetation (Smit et al., 2007; Thrash, 2000; Thrash, Theron, & Bothma, 1995). Impact on vegetation due to water supplementation arises from placing AWP in areas void of natural water or where these natural sources are simply seasonal (Joubert, 1976; Owen-Smith, 1996; Smuts, 1972; Thrash, 1998; Weir, 1971). This opens up areas to large water-dependent herbivores for foraging in the dry season that were naturally only available in the wet season (Thrash, 1998).

The four AWP present in the sanctuary region of MWR were established to offer ideal animal viewing opportunities for ecotourism; however, this region has a high density of perennial water sources and therefore such AWP are not needed. The remaining six AWP in the reserve were established to encourage wildlife dispersal from the sanctuary, into the entirety of the reserve. A fenced reserve such as MWR could undergo increased vegetation cover loss and structural changes due to surplus water, such as that in the sanctuary region, if not managed correctly (Owen-Smith, Kerley, Page, Slowtow, & van Aarde, 2006). This overabundance of AWP could result in increased homogeneity of herbivore impacts, loss of vegetation cover and diversity (Farmer, 2010; Gaylard et al., 2003; O'Connor, Goodman, & Clegg, 2007; Owen-Smith et al., 2006; Redfern, Grant, Gaylard, & Getz, 2005). The number of AWP in this region should be reduced, or a rotation system should be implemented, where AWP are alternately closed and opened periodically (Wienand, 2013). Any changes in water supplementation should be cautiously considered as increased pressure and utilisation will be diverted to perennial water sources, increasing the pressure on riverine vegetation (Smit & Ferreira, 2010). If AWP are reduced in this region, regular vegetation monitoring along the perennial rivers should be undertaken to detect changes. AWP in other areas with suitable habitats should be provided further from the sanctuary to encourage herbivore dispersal away from the sanctuary as it is no longer fenced.

The analyses indicated that AWP are seasonally utilised by herbivores. Water sources are plentiful in the wet season, allowing herbivores to move away from AWP and forage in various regions (de Beer & van Aarde, 2008; Smit & Grant, 2009; Smit et al., 2007). In the dry season, this foraging is restricted to available perennial water sources (Redfern et al., 2005). In MWR, AWP were visited predominantly in the late dry season with little visitation in the early and late wet months. The decrease in vegetation cover close to AWP provides evidence of an increase in herbivore activity with a decrease in distance to perennial water in MWR. This indicates that MWR is in need of a water management plan addressing the positioning of AWP, the management or prevention of erosion and the conservation of vegetation heterogeneity.

Several management actions may assist to address the potential vegetation and erosion issues identified by this study. Some management options for the AWP are as follows:

5.3.1 *Water overflow control*

AWPs which have a constant overflow and are utilised by high numbers of wildlife are ideal for the transmission of parasites such as roundworms and liver flukes (Coetzee, 2016). MWR faces a unique issue with water supply to the AWP. There is extensive overflow of water during the day, particularly in the wet season. This is as a result of the constant uncontrolled pumping of water via solar power into the concrete water basins, increasing the loss of water via soil infiltration and erosion severity. This occurs during the dry season as some of the concrete basins have been damaged, leaving openings for water to either overflow or leak through. However, the borehole pumps stop pumping in the evening which often results in a lack of water as large herds of buffalo (peak drinking at 18:00), deplete the water supply in the early evening. By sunrise there is little to no water left in the concrete basins and the mud baths or gullies. The latter has an overabundance of water in the wet season and provides an important source for wallowing and mud bathing in the dry season.

The best approach to such a problem is to treat the cause, followed by the symptoms. Firstly, the water runoff issue should be resolved, then the extensive erosion at the AWP can be rehabilitated (Coetzee, Bothma, van Rooyen, & Breedlove, 2016). A quick fix to this solution would be to repair the erosion, with water runoff still occurring (has been attempted in the past in MWR). The best chance of success involves a long term and continuous commitment to rehabilitation and management (Coetzee et al., 2016). A solution to this problem may be to implement a ball valve system along with water storage tanks or reservoirs (Coetzee, 2016). Ball valves allow for the continuous and measured flow of water into a concrete basin, eliminating excessive and unnecessary water wastage. Water can be stored in the tanks and held back by the ball valve, allowing just enough water through to ensure the AWP is filled but not overflowing. A ball valve system consists of a float on the water in a protected area of the AWP. When wildlife drink from the AWP and lower the water level, the float drops and opens up the valve to allow water to flow into the basin (Appendix 4). These tanks will have to be protected from animals such as elephants and baboons, for example.

5.3.2 *Erosion control*

Erosion is formed due to soil particle removal and deposition via wind and water. Soil erosion becomes more severe when the protective vegetation layer is removed, exposing bare soil to the elements of nature (Roodt, 2011; van Oudtshoorn, 2002). Water runoff dislodges soil particles and transports the sediment downhill. Over time, this concentrates in small ditches known as rill erosion. Gully erosion

is formed as the ditches increase in size due to sediments removing soil from the edge of the ditches (Coetzee et al., 2016; van Oudtshoorn, 2002).

In order to decrease gulley erosion, an attempt to decrease the flow speed of runoff water needs to be made. This can be achieved in various ways such as:

- Creating walls out of stones in wire baskets (gabions) in the gulley to catch sediment, allowing water to slowly pass through the gabions.
- Branches may be used instead of stones; however, this seems less successful.
- Common reed grass can be planted across the gulley, creating a natural wall over time which will collect sediments (MWR currently has a nursery where these stabilising grasses are grown).

According to Coetzee et al. (2016) the successful control of soil erosion is achieved by abiding to the following general principles:

- Treating the cause before the symptoms
- Realistic long-term goals
- Continued commitment and maintenance
- Using the most cost-effective method to achieve an objective

Habitat rehabilitation should focus on stabilising areas which have been eroded and encouraging increased vegetation establishment (Coetzee et al., 2016). The treatment of serious erosion such as larger gulleys should focus on the shallow starting point of the erosion instead of the deeper areas that may be later destroyed by upstream runoff. This will prevent the shallow points from expanding further (Coetzee et al., 2016). Gabions can be used to control this advanced gulley erosion by building walls or contoured weirs along the head of the erosion. This involves the construction of stones packed in mesh wire and placed at the base or sides of the gulley for stabilisation and to improve silt accumulation (Coetzee et al., 2016). It is important to combine this with the planting of grass seeds as increased grass growth in these areas will increase the silt accumulated (Coetzee et al., 2016; Roodt, 2011). However, the planting of grass seeds will not be effective when the soil is experiencing constant trampling. Therefore, AWP's should be closed for a few months during the wet season if this approach is taken. The closing of the AWP will remove pressure from the surrounding areas, allowing for grass and other plant species to recover and grow. The aim of habitat rehabilitation should be to improve soil water retention and infiltration.

Several methods may assist to resolve erosion issues identified by this study at the various AWP, however, these will not be effective until the water overflow problem is addressed. These suggested methods for each AWP are as follows:

- *Nthumba*

The topography at this AWP is flat, thus reducing flow-rates and resulting is the development of natural pools along shallow gulley erosion. Some breakage of the concrete water-basin should be repaired immediately. Mechanical equipment should be used to level out the runoff wallows. These can then be packed with reeds to help protect the soil and grass seed bank from wind erosion and further trampling. The rill erosion can be rock packed to help rehabilitate the soil.

- *Diwa*

The concrete water-basin was broken, and rock packing is required on either side of the overflow of the concrete water-basin to prevent undercutting of the basin edge due to the excavation of this area by wallowing wildlife. The use of rock packing is needed immediately downstream of the overflow and along the rill terraces to help prevent these becoming serious erosion problems.

- *Phwadzi*

Brush and rock packing along the length of the erosion area would aid in the rehabilitation of the area, allowing the growth of grasses, reeds and other vegetation. This will in turn assist with the reduction of this erosion problem. If not contained this erosion may eventually create a gulley across the road. The larger gulley wallows can be levelled out by mechanical intervention. Since the conclusion of this study, the waterpoint has been cemented, however the overflow is still ongoing and the gulleys broadening and getting deeper.

- *Nakamba*

Rock-packing the area surrounding the concrete water-basin would help to contain further damage to the basin itself and the immediate surrounding land. Further rock packing can be used to resolve the rill erosion leading down into the dry riverbed. This will aid in soil and grass rehabilitation.

- *Nsepete*

Rock packing at the concrete-basin edge and the construction of gabion structures at numerous terraces along the gulley course will be required to slow the progress of erosion. Using mechanical

means to fill the erosion followed by brush packing the site would assist with the growth of perennial grasses. Brush-packing the gulleys will assist with the trapping of seed to facilitate vegetation regrowth and thus assist with slowing future erosion. It is worth noting that there was already an attempt to halt edge erosion at this AWP that failed, possibly due to the continued overflowing over water from this basin.

- *Pende 1*

The deep gully erosion problem can be controlled with the construction of multiple gabions and rock packing around the water-basin edge. Mechanical intervention is needed to fill the deep gully erosion which can then be brush packed to help promote plant growth and rehabilitation. Reduction of water overflow is highly recommended at this site to prevent further degradation of this landscape. Pende 1 may prove the costliest site to repair as it has the most extensive erosion of all observed AWP's.

- *Pende 2*

Rock packing at the site of water overflow from the concrete water-basin would assist with reduction of future erosion, with undercutting and resultant damage of the basin edging. The concrete water-basin should be repaired as there are some cracks present.

- *Thawale*

Mechanical breaking up of the soil and levelling of the gully erosion needs to be implemented to aid soil recovery. This should be followed by reed packing the graded area to ensure protection from trampling and wind erosion. Rock packing surrounding the water basin will protect the concrete from breaking apart and decrease erosion surrounding the basin.

- *Heritage*

There is some undercutting of the concrete water-basin as well as a considerable amount of gully erosion on the side of the stream bank which, will require rock packing to prevent eventual loss of this side of the water-basin as the stream bank erodes away. Brush packing along the slope from the visitors centre to the riverbed will be required to slow erosion. Rock packing of the gully's is a short-term solution, but a gabion structure in the stream edge would be an improved long term solution for this AWP.

- *Kakoma*

Kakoma has not undergone any extreme erosion as yet, as it is a relatively new AWP. The edge of the concrete water basin has eroded due to water overflow, forming some rill erosion. The concrete water basin needs to be repaired and the rill erosion can be filled via rock packing. This will allow for sediment to collect and vegetation to grow.

The construction of earth dams (large wallows) in the runoff below the concrete basins of the AWPs may be recommended, given the high frequency of utilisation including wallowing. Stocking rates in reserves may determine the impact of herbivores on the vegetation surrounding AWPs (Owen-Smith, 1996). Therefore, the monitoring of vegetation adjacent to AWPs should be continued proactively along with the management of wildlife stocking rates as increased herbivore populations lead to advanced ecological impacts.

5.3.3 *Considerations for future AWPs*

The conservation and increased population densities of wildlife can be achieved through creating AWP networks within these semi-arid regions (Ayeni, 1975; Chamaillé-Jammes, Fritz, et al., 2007; Owen-Smith, 1996; Redfern et al., 2005). However, if not correctly managed, these AWP networks negatively impact the ecosystem by resulting in increased structural changes of vegetation, alter prey-predator relationships and eventually landscape homogenization (Cain et al., 2012; Chamaillé-Jammes, Charbonnel, Dray, Madzikanda, & Fritz, 2016; Sirot et al., 2016). Hence the need of a defined goal and vision for water supplementation policies, preferably with a defined sustainable action plan (Smit & Grant, 2009)

Surface water manipulation has previously been used to manage herbivore distribution and vegetation impacts within large, water-limited reserves (Chamaillé-Jammes, Fritz, & Murindagomo, 2007; Chamaillé-Jammes, Valeix, & Fritz, 2007; Gaylard et al., 2003; Smit, Grant, & Devereux, 2007). Habitat diversity can be maximised through carefully designing a pattern for the placement of AWPs (Thrash, 2000). Encouraging an even distribution of wildlife within a reserve via uniformly distributed AWPs does not adhere to the maintenance of a healthy ecosystem, but rather for the management of properties breeding animals with the aim of maximum production (Thrash, 1998). A standard distance between AWPs should therefore be avoided and instead a scattered pattern of AWP placement should be implemented within the reserve.

The distance wildlife travels on a daily basis needs to be considered when proposing sites for new AWPs with the aim of maintaining habitat heterogeneity with intermittent stable forage areas during drought conditions. AWPs should be spaced three times the distance wildlife travels for water, whether it be natural or artificial water sources (Smit et al., 2007). Some species, such as the plains

zebra, cover up to 5km a day in search of water which would result in the recommendation of spacing water sources at least 15km apart (Owen-Smith, 1996; Smit et al., 2007). The correct management and implication of water policies within reserves could increase habitat heterogeneity, in turn decreasing management intensity (Owen-Smith, 1996).

Future water management plans in MWR need to incorporate the idea of creating spatial heterogeneity in herbivore foraging levels (Farmer, 2010; Gaylard et al., 2003). When supplementing water, AWP's should be situated at a distance from other natural perennial water to encourage spatial heterogeneity and avoid homogeneity through increased grazing and browsing (Farmer, 2010; Gaylard et al., 2003). Reserve management should aim to encourage the creation of mosaics of differing foraging intensities by strategically positioning AWP's to exclude high densities of large herbivores in certain areas (Chamaillé-Jammes, Fritz, et al., 2007; Loarie, Aarde, & Pimm, 2009; Owen-Smith et al., 2006). Instead of spreading AWP's and herbivores evenly across the landscape, fewer AWP's with intense utilisation should be implemented. This will minimise the chances of vegetation homogenisation and reducing ecosystem resilience during drought periods when areas far from AWP's are important buffer forage areas.

The addition of AWP's in MWR is not needed and should be avoided going forward. Landscapes consist of complex ecological systems. A surplus of AWP's could result in ecological homogeneity as the distribution of AWP's in a landscape determines the distribution of herbivores and herbivore impact on vegetation. A natural variation in surface water results in spatial heterogeneity of herbivore impact on vegetation. AWP location and water availability (Chamaillé-Jammes et al., 2007). Piospheres are created when water availability is limited, causing herbivores to congregate around perennial water sources (Thrash & Derry, 1999). Closely spaced AWP's results in herbivores moving easily between them, creating an even level of browsing and grazing across the landscape, merging piospheres (Owen-Smith, 1996). Regional differences in vegetation utilisation are lost, resulting in vegetation impact homogenisation, when herbivores are able to graze and browse throughout the reserve (Gaylard et al., 2003). In fenced reserves, where herbivore movements are limited, vegetation impact and species loss may occur (O'Connor et al., 2007; van Aarde & Jackson, 2006).

The following should be considered if the construction of future AWP's is intended (Coetzee, 2016; Owen-Smith, 1996):

- Avoid slopes exceeding 5° as this will increase trampling and water spillage which accelerates erosion.
- Avoid dense woodland areas as herbivores prefer areas with less chance of ambush predation.

- Avoid areas with extensive bedrock as ungulate species have difficulty moving over these surfaces.
- Avoid placing AWP in or close to areas with sensitive vegetation communities. These include wetlands and areas with high endemic plant diversity.
- Ensure AWP are placed at the bottom of valleys or along drainage lines as these areas are well adapted to trampling and frequent utilisation.
- Select areas with alluvial soils or well-drained coarse sand instead of clay soils.
- Ensure AWP are not placed in areas already overgrazed, desiccated and degraded which as this will increase pressure on vegetation and soil.
- Ensure AWP borehole pumps can be controlled to be opened or shut, encouraging animal movements.
- The AWP must be constructed to inflict minimum wildlife disturbance while allowing for enhanced wildlife viewing.
- Ensure AWP are not positioned within 5km of perennial water sources to reduce the possibility of merging biospheres.
- The construction of hides at AWP should include screened walkways from a parking area to the hide to ensure minimum wildlife disturbance.
- Areas should be selected which would not be sensitive to the overexploitation by wildlife.

In order to maintain habitat diversity and ecosystem resilience, an alternating system of open and closed AWP has been suggested (Smit & Grant, 2009). According to Chamaillé-Jammes et al. (2016) this alternating system can be effective if: 1) the distribution of herbivores can be accurately predicted under different management scenarios; 2) the water supplementation policy can be altered according to changes in herbivore distribution and environmental changes and 3) an ensured adaptive management policy is applied. Without the abovementioned points, deciding which AWP to open and close annually can be challenging.

5.4 Conclusion

Results from this study and supporting literature suggest a rotational system of open and closed AWP in MWR needs to be implemented, specifically in the sanctuary region. This, along with a combination of erosion and overflow management techniques, will ensure the successful management of water within MWR, as it is a small fenced reserve (Owen-Smith, 1996; Thrash & Derry, 1999). To guarantee that the most suitable management strategies are identified, selected and implemented, a Conservation Management Plan for water sources must first be compiled, with a clearly defined vision and goal which is realistically achievable (Grant et al., 2011).

Water management policies should include the creation of spatial heterogeneity in large herbivore vegetation utilisation throughout MWR through the creation of vegetation mosaics (Gaylard et al., 2003; Redfern et al., 2003). Strategically positioning AWP's could achieve these mosaics on a small scale varying in levels of herbivore utilisation by encouraging herbivores to move into different regions of the reserve (Owen-Smith, 1996; Thrash, 1998; Thrash, Nel, Theron, & Bothma, 1991). However, on a larger scale, homogenisation may occur as animal distributions are spread over wider areas. Therefore, the closing of some AWP's in MWR will discourage the spread of wildlife throughout the reserve and localise the impact on the landscape.

The information gained from this study has improved our current understanding of large herbivore impact on vegetation at and surrounding AWP's and is critical for advancing the conservation of herbivores, vegetation and water supplementation (Grant et al., 2011; Rosenstock et al., 1999). Long-term monitoring programmes such as fixed-point photography and vegetation impacts should be continued to detect the impacts on vegetation owing to changes in available surface water (Grant et al., 2011; Simpson et al., 2011). Monitoring of herbivore impacts on vegetation is important in a fenced reserve such as MWR, in which a large population of herbivores is already sustained. The results from this study should be encompassed into conservation efforts in order to utilise this information significantly to ensure optimum management.

5.5 References

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Appendices

Appendix 1

Table A.1 Details of Majete Wildlife Reserve wildlife introductions between 2003 – 2018, along with aerial survey estimates from September 2015.

Species	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total	Aerial Count (2015)
Black Rhino	2				6									1			9	16
Buffalo	120	100				86											306	1319
Bushbuck																	-	400
Eland		20				32	25										77	320
Elephant				70		64	83										217	389
Hartebeest		4		10		15	30										59	80
Impala	216					210		311						400			1137	2000
Kudu																	-	1022
Leopard									2	4							6	-
Lion										3						5	8	-
Nyala	6	15				38											59	300
Sable	100					153		99									59352	1337
Warthog	60					98											158	1500
Waterbuck	98					198		106									402	1782
Zebra		37	50	9		38	40										174	571

Appendix 2.1

Grasses are good indicators of veld condition as they are well adapted to specific growth conditions and may be sensitive to change. The ecological status of grasses refers to the ecological grouping of grasses according to their reaction to different levels of grazing, forage and fuel production.

Ecological status categories include (van Oudtshoorn, 2002):

Category	Description	Value
Decreaser:	Grasses abundant in good veld, but decrease when the veld is under or overgrazed	10
Increaser I:	Grasses which are abundant in underutilised veld	7
Increaser IIa:	Grasses abundant in veld with light overgrazing	4
Increaser IIb:	Grasses abundant in veld with moderate overgrazing	4
Increaser IIc:	Grasses abundant in veld with severe overgrazing	1
Herb:	Unpalatable annual grass and forb species with a low productivity and low grazing value	1
Bare:	Bare ground encountered on a miss	0

The ecological status of grasses is used to determine veld condition. This is done by means of calculating the species composition of a plot via a veld survey. The veld condition score determined from the ecological values of species within the plot provides an ecological index. This index classified the veld as poor, moderate or good. A veld in good condition would consist mostly of Decreasers or Increaser I's, while poor veld would consist mostly of Increaser II's and herbs.

Veld condition scores:

Ecological index	Veld condition
0 – 399	Poor Veld
400 – 600	Moderate Veld
601 – 1000	Good veld

Appendix 2.2

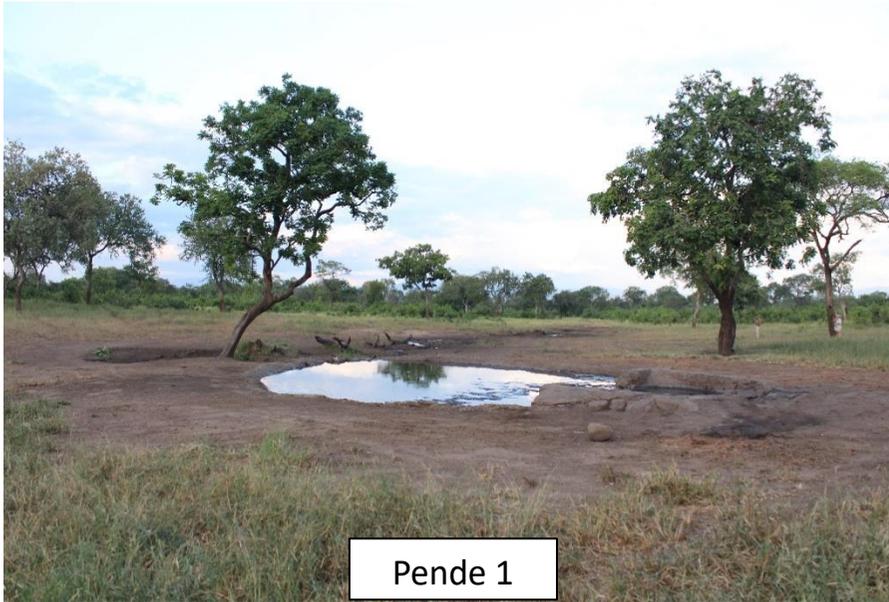
Grass List

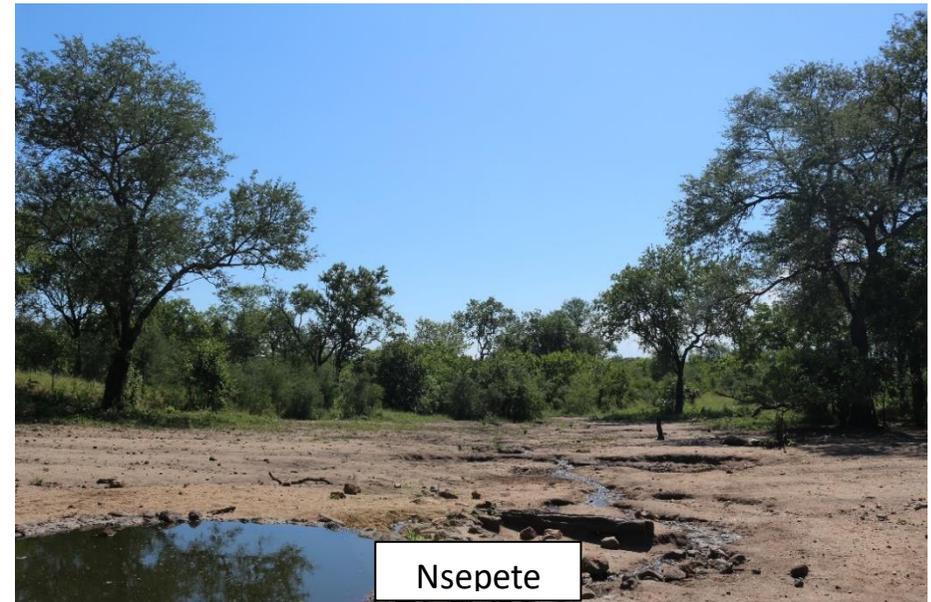
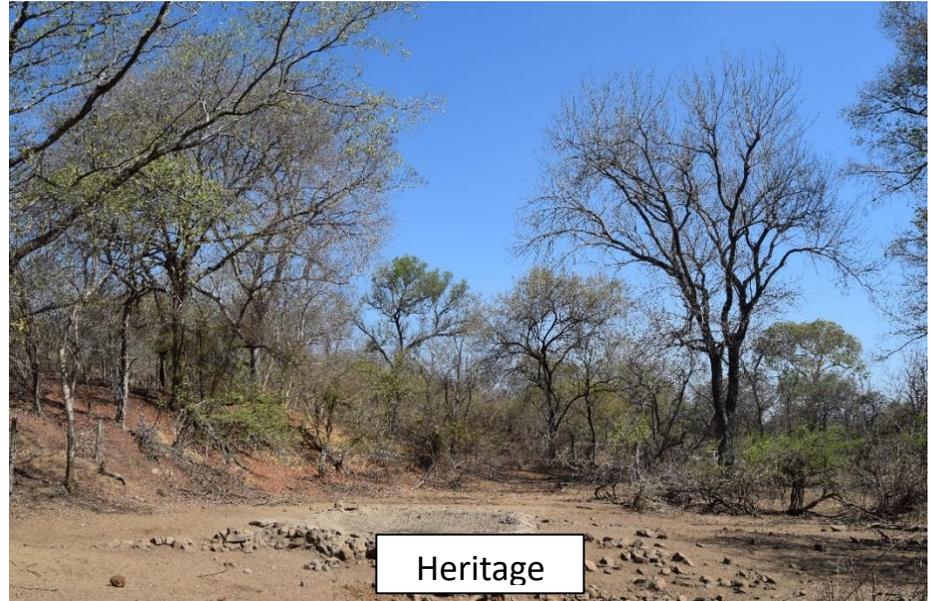
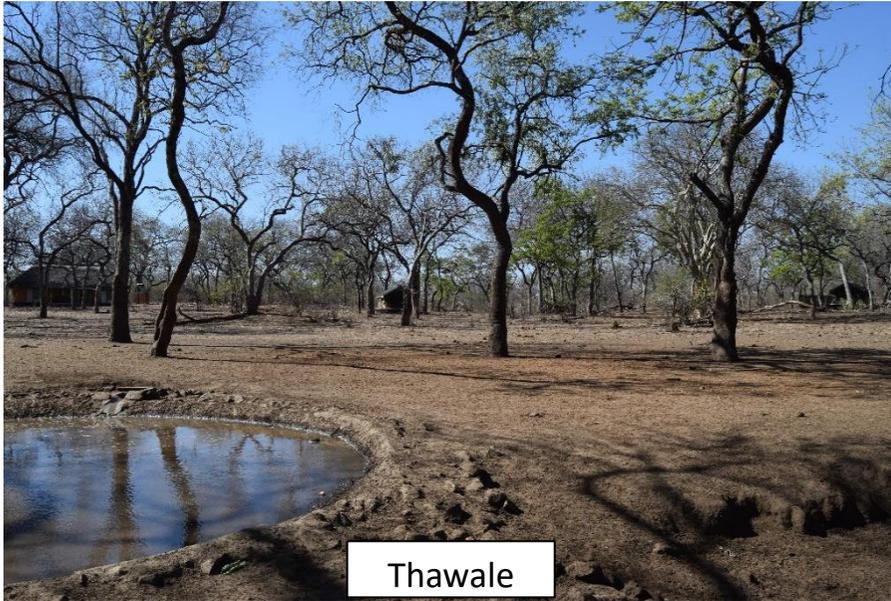
Scientific name	Common name	Ecological status
<i>Andropogon gayanus</i>	Blue grass	Increaser I
<i>Aristida adscensionis</i>	Annual three-awn	Increaser IIc
<i>Bewsia biflora</i>	False love grass	Increaser IIa
<i>Bothriochloa bladhii</i>	Purple plume grass	Increaser I
<i>Chloris pycnothrix</i>	Spiderweb grass	Increaser IIc
<i>Chloris virgata</i>	Feather-top chloris	Increaser IIa
<i>Cymbogon plurinodis</i>	Narrow-leaved turpentine	Increaser I
<i>Cymbopogon caesius</i>	Broad -leaved turpentine grass	Increaser I
<i>Dactyloctenium giganteum</i>	Giant crowfoot	Increaser IIc
<i>Dactyloctenium aegyptium</i>	Common crowfoot	Increaser IIc
<i>Digitaria eriantha</i>	Common finger grass	Decreaser
<i>Diheteropogon amplexans</i>	Broad-leaved Bluestem	Decreaser
<i>Eragrostis aspera</i>	Rough Love Grass	Increaser IIc
<i>Eragrostis chloromelas</i>	Curly leaf (Broad)	Increaser II
<i>Eragrostis cilianensis</i>	Stink love grass	Increaser IIc
<i>Eragrostis ciliaris</i>	Woolly Love grass	Increaser II
<i>Eragrostis superba</i>	Saw-tooth love grass	Increaser IIa
<i>Heteropogon contortus</i>	Spear grass	Increaser IIa
<i>Hyparrhenia anamesa</i>		Increaser I
<i>Hyparrhenia dissoluta</i>	Yellow thatching grass	Increaser I
<i>Hyparrhenia filipendula</i>	Fine Thatching grass	Increaser I
<i>Hyparrhenia hirta</i>	Common thatching grass	Increaser I
<i>Leptochloa chinensis</i>	Red Strangletop	Increaser IIa
<i>Panicum maximum</i>	Guinea grass	Decreaser
<i>Phragmites australis</i>	Common Reed	Decreaser
<i>Pogonarthria squarrosa</i>	Herringbone grass	Increaser IIc
<i>Rottboellia cochinchinensis</i>	Itch grass	
<i>Schimdtia pappophoroides</i>	Sand quick	Decreaser
<i>Schizachyrium sanguineum</i>	Red Autumn grass	Increaser I
<i>Setaria sagittifolia</i>	Arrow grass	Increaser IIa

<i>setaria sphacelata var. sericea</i>	Golden Bristle grass	Decreaser
<i>Sorghum versicolor</i>	Black-seed Sorghum	Increaser II
<i>Themeda triandra</i>	Red grass	Decreaser
<i>Trachypogon spicatus</i>	Giant Spear grass	Increaser I
<i>Tragus berteronianus</i>	Carrot seed grass	Increaser IIc
<i>Urochloa mosambicensis</i>	Bushveld Signal grass	Increaser IIc
<i>Brachiaria deflexa</i>	False Signal Grass	Increaser II

Appendix 3.1

An example of fixed-point photographs for each of the ten AWP in Majete Wildlife Reserve. These photographs were taken during wet season 2018 (Diwa, Pende 1, Pende 2, Nthumba, Phwadzi, Nsepete and Nakamba) and dry season 2018 (Heritage, Kakoma and Thawale).

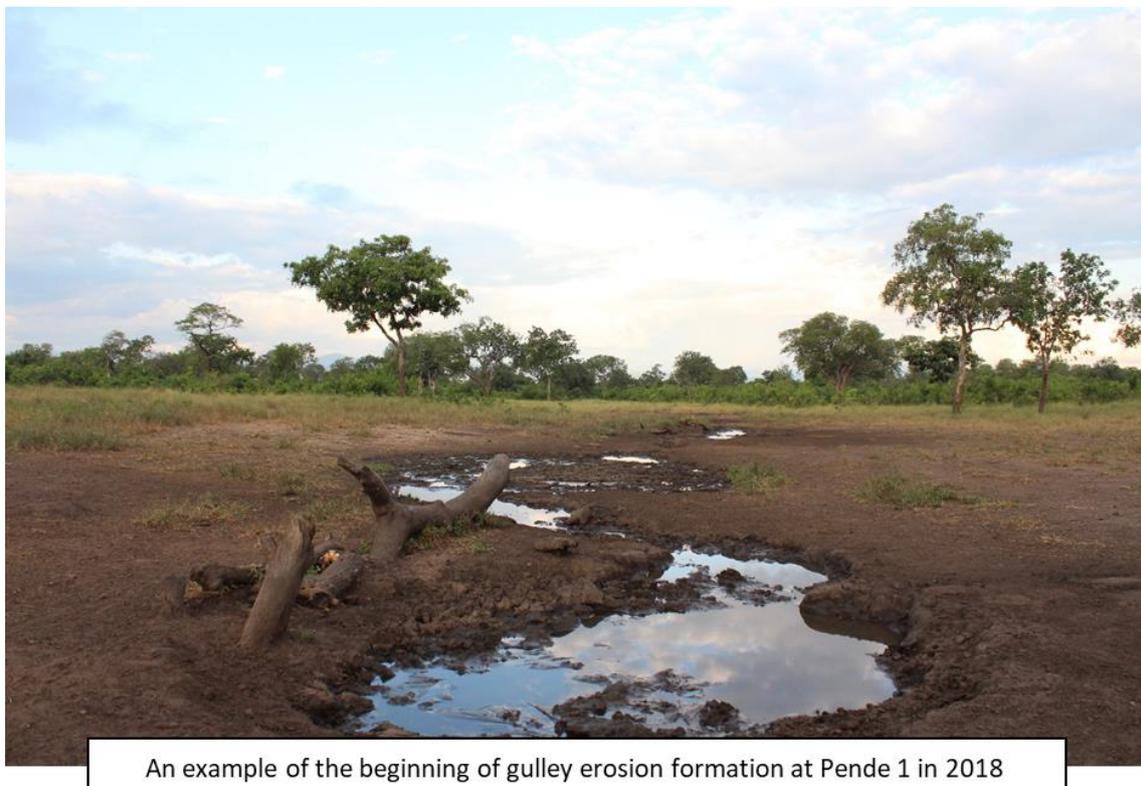






Appendix 3.2

Examples of rill and gulley erosion measured in 2018 at the AWP in MWR.





Appendix 4

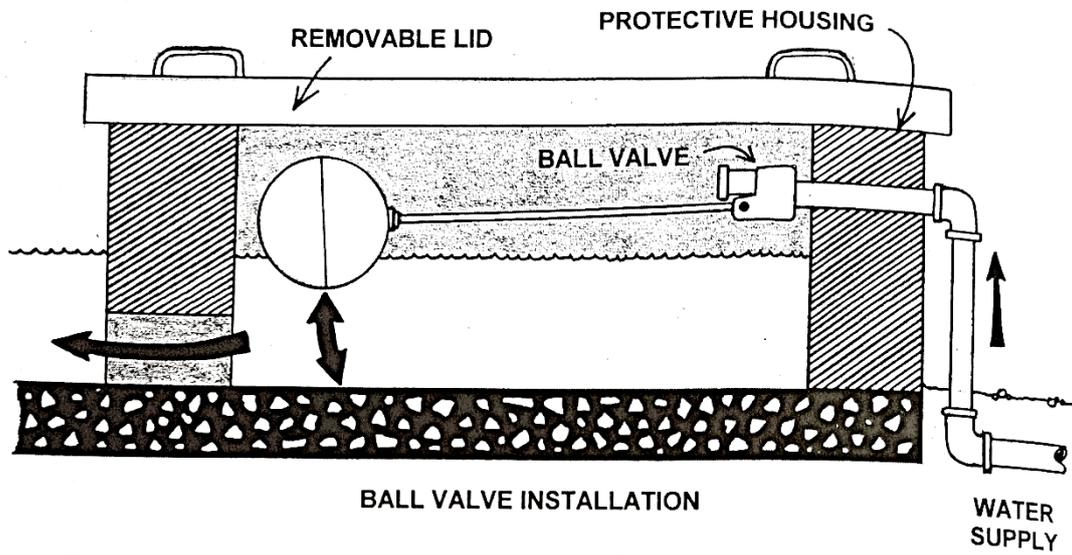


Figure A.1 An example of a ball-valve system which could be implemented at the AWP's to control water levels (Coetzee, 2016).