Woody vegetation change and elephant water point use in Majete Wildlife Reserve: implications for water management strategies

By

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Declaration

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Abstract

The confinement of many elephant (*Loxodonta africana*) populations in southern Africa to fenced reserves has made the management of such reserves difficult as elephants are a keystone species. Elephants are also water-dependent; therefore the availability of water affects the location, extent and intensity of elephant impacts on vegetation. Majete Wildlife Reserve (Malawi) has undergone reform, during which it was fenced, artificial waterholes (AWPs) were created and wildlife reintroduced, including 220 elephants. Concerns have arisen as to the impact elephants may be having on the vegetation. In this thesis, two studies were conducted, along with a review of literature on elephant interactions with surface water.

Woody vegetation changes in Majete were assessed by comparing woody vegetation cover datasets (based on remote-sensed vegetation classifications) of the reserve for 1985, 1990, 2000 and 2010. Woody cover loss was high between 2000 and 2010, therefore points of woody cover loss were further analysed in a spatial analysis. Using spatial and non-spatial environmental data, the effects of rainfall, fires, terrain (altitude, aspect, slope, hill and valley characteristics) and proximity to perennial water on woody vegetation cover were tested. Data analyses indicated that woody cover loss may have been caused by differing combinations of drought and herbivory or fire in different areas of the reserve. Where woody cover loss was attributed to herbivory, points of loss were not associated with proximity to perennial water. It was suggested that this is due to high perennial water availability in Majete, which would not limit herbivore foraging ranges in the dry season. Woody cover loss could not, however, be attributed to elephants in this study and further information on their use of and impacts around perennial water points was required.

In a further study, the hypothesis tested was that different water point types (rivers, AWPs and springs) would be used at different intensities by elephants, and that perennial rivers would experience the most use. Elephant usage (including visits to water points, browsing levels and path use around water points) of selected perennial water points in Majete was monitored in the wet and dry season. The effects of season, water point characteristics (type, size and water quality) and habitat context (surrounding vegetation type, elevation and proximity to other water points) on elephant water point use were then tested. Elephant water point use was affected by season, as well as water point altitude and surrounding vegetation type. In areas of high perennial water availability, elephant browsing around water points did

not decrease with increasing distance. It was suggested that this too could be because elephant browsing activity is not limited by water availability in Majete.

Based on the findings of both studies, recommendations for water, elephant and fire management in Majete were proposed and discussed.

Opsomming

Die beperking van baie olifantpopulasies tot omheinde reservate in Suider-Afrika bemoeilik die bestuur van hierdie parke omdat olifante (*Loxodonta africana*) as 'n hoeksteenspesie beskou word. Olifante is ook water-afhanklik en daarom word die plek, omvang en intensiteit van olifante se impak op plantegroei deur die beskikbaarheid van water beïnvloed. Die Majete Wildlife Reserve (Malawi) het hervorming ondergaan waartydens dit omhein is, kunsmatige waterpunte (AWP's) geskep is en wilde diere, insluitend 220 olifante, hervestig is. Kommer het ontstaan rondom die moontlike impak wat olifante op die plantegroei mag hê. In hierdie tesis word twee studies sowel as 'n literatuurstudie aangaande olifantinteraksies met oppervlakwater uitgevoer.

Houtagtige plantegroei veranderinge in Majete is geassesseer deur data van plantegroei-bedekking (op afstandswaarnemings van plantegroeiklassifikasies gebaseer) vanuit 1985, 1990, 2000 en 2010 met mekaar te vergelyk. Die verlies van houtagtige-bedekking tussen 2000 en 2010 was hoog en gevolglik is hierdie areas verder deur 'n ruimtelike analise ontleed. Deur die gebruik van ruimtelike en nie-ruimtelike omgewingsdata, kon die effek van reënval, brande, terrein (hoogte, aspek, helling, heuwel- en vallei-eienskappe) en nabyheid aan standhoudende water op houtagtige plantegroei getoets word. Data-analise het aangedui dat verlies van houtagtige-bedekking deur verskillende kombinasies van droogte, herbivooraktiwiteit of brande in die verskillende dele van die reservaat veroorsaak word. Waar verlies van houtagtige-bedekking toegeskryf kon word aan herbivore, was die punte van verlies nie geassosieer met nabyheid aan standhoudende water nie. Daar is voorgestel dat hierdie waarneming verband hou met die hoë beskikbaarheid van water in Majete, wat gevolglik nie herbivoorbeweiding in die droë seisoen beperk nie. Hierdie studie kon dus nie die verlies van houtagtige-bedekking aan olifantteenwoordigheid toeskryf nie en verdere inligting rondom die verbruik en impak van olifante op standhoudende waterpunte word benodig.

In 'n verdere studie is die hipotese dat verskillende tipes waterpunte (riviere, AWP's en fonteine) teen verskillende intensiteitsvlakke deur olifante benut word, en dat standhoudende riviere die meeste verbruik sou ervaar, getoets. Verbruik van geselekteerde standhoudende waterpunte deur olifante (insluitend besoeke aan waterpunte, beweiding en paadjiegebruik rondom waterpunte) in Majete is gedurende die nat- sowel as droë seisoene gemonitor. Die

effek van seisoen, waterpunt-eienskappe (tipe, grootte en watergehalte) en habitatkonteks (omliggende plantegroei, hoogte bo seespieël en nabyheid aan ander waterpunte) is op die waterpuntverbruik van olifante getoets. Die waterpuntverbruik van olifante word deur seisoen, hoogte bo seespieël en omliggende plantegroei beïnvloed. In gebiede met hoë beskikbaarheid van standhoudende waterpunte, het olifantbeweiding rondom waterpunte nie met toenemende afstande afgeneem nie. Daar is voorgestel dat hierdie waarneming ook toegeskryf kan word aan die feit dat olifantbeweiding nie deur die beskikbaarheid van water in Majete beperk word nie.

Gebaseer op die bevindinge van beide studies, word daar aanbevelings vir water-, olifant- en brandbestuur in Majete voorgestel en bespreek.

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Chapter One:

Thesis introduction

1.1. Background

Protected areas and wilderness are under increasing pressure in southern Africa due to high human population growth in rural areas, which have resulted in the expansion of settlements and subsistence agriculture. As a result, reserves are increasingly being fenced in order to protect both people and wildlife (Hayward & Kerley, 2008). Fences have multiple effects on ecological processes within reserves, including severing of wildlife dispersal routes, and limiting the amount of food, water and space resources available to wildlife (Newmark, 2006). Fencing essentially creates artificially closed systems or ecological islands (Boone & Thompson Hobbs, 2004; Whyte & Joubert, 2010). Elephants are recognized as a keystone species in savanna systems as they can cause large-scale changes in woody vegetation structure, diversity and cover through their destructive feeding habits and large forage requirements (Laws, 1970; Cumming et al., 1997; Hayward & Zawadzka, 2010). When elephant populations are enclosed in fenced reserves the management of such a system becomes increasingly complex. In fenced reserves elephants are unable to shift ranges seasonally as they do in natural systems, and vegetation is therefore exposed to increased foraging pressure, a situation which can lead to severe loss of woody vegetation cover (Duffy et al., 2002; van Aarde & Jackson, 2007; Loarie et al., 2009a). As mixed-feeders, elephants mostly graze in wet seasons, but browse in the dry season when herbaceous forage becomes moribund (Bax & Sheldrick, 1963; Osborn, 2004). Elephants are also a water-dependent species and therefore their impacts on woody vegetation are highest closer to water and at dry times of year when surface water availability is limited (Ben-shahar, 1996; Chamaillé-Jammes et al., 2009; Gaugris & van Rooyen, 2009). As water availability decreases in dry seasons, elephant ranges become restricted to areas holding perennial water (Laws, 1970; Chamaillé-Jammes et al., 2007b; Smit & Ferreira, 2010). The concentration of elephants (and other herbivores) around perennial water in the dry season results in the development of vegetation use patterns around water points or piosphere effects (Lange, 1969). Piospheres consist of what has been termed as a 'sacrifice area' devoid of vegetation immediately surrounding a waterhole, followed by increasing vegetation quality with increasing distance from water, a function of herbivore water and forage needs (Lange, 1969; Thrash & Derry, 1999). Water point placement within fenced reserves can therefore strongly influence the distribution of elephant impacts on woody vegetation, and as a result surface water management constitutes an important activity of effective reserve management.

Majete Wildlife Reserve (Majete) in southern Malawi, is a small (700km^2) fenced reserve surrounded by high density rural human settlements and agriculture, which prevents the reserve's expansion. By the mid-1990s local extinctions of most large mammal species had been caused by high levels of poaching and a lack of law enforcement. In 2003 African Parks Majete (Pty) Ltd. entered into a public-private partnership (PPP) with the Malawian government, more specifically the Department of National Parks and Wildlife, through which they undertook responsibility for the rehabilitation of the reserve. Rehabilitation activities included fencing the reserve and large-scale wildlife re-introductions, both to restore ecological processes and increase Majete's tourism prospects. Pre-1990 the number of elephants inhabiting Majete and the surrounding regions was approximately 300 individuals (Sherry, 1989). Between 2006 and 2009, 215 elephants were reintroduced into the reserve and the population currently stands approximately at 270 individuals (reserve manager, pers. com.).

Majete's expected annual precipitation falls between 680-1000mm in a distinct wet season commencing in November and ending in early April. No other significant rainfall is received for the rest of the year. Few perennial water points exist in the reserve other than two large perennial rivers in the north-east and 11 springs (perennial and seasonal springs) scattered throughout the reserve (Figure 1.1.). The perennial rivers run through the 'Sanctuary' area, a 14 000ha site where most of the re-introductions took place. Water has been supplemented in Majete through the addition of seven borehole-fed artificial water points (AWPs), four of which have been placed in the Sanctuary area. AWPs are used, as in other reserves, to stabilise surface water availability (Chamaillé-Jammes et al., 2007a), to increase access to dry season forage areas for wildlife (Redfern et al., 2005; Loarie et al., 2009a), to facilitate species population growth, and to provide game-viewing opportunities for tourists (Shannon et al., 2009). Additionally, tourism is an important source of revenue for Majete and the AWPs are key attractions for tourists due to the good game viewing opportunities they provide during the dry season when wildlife concentrates at remaining perennial water.

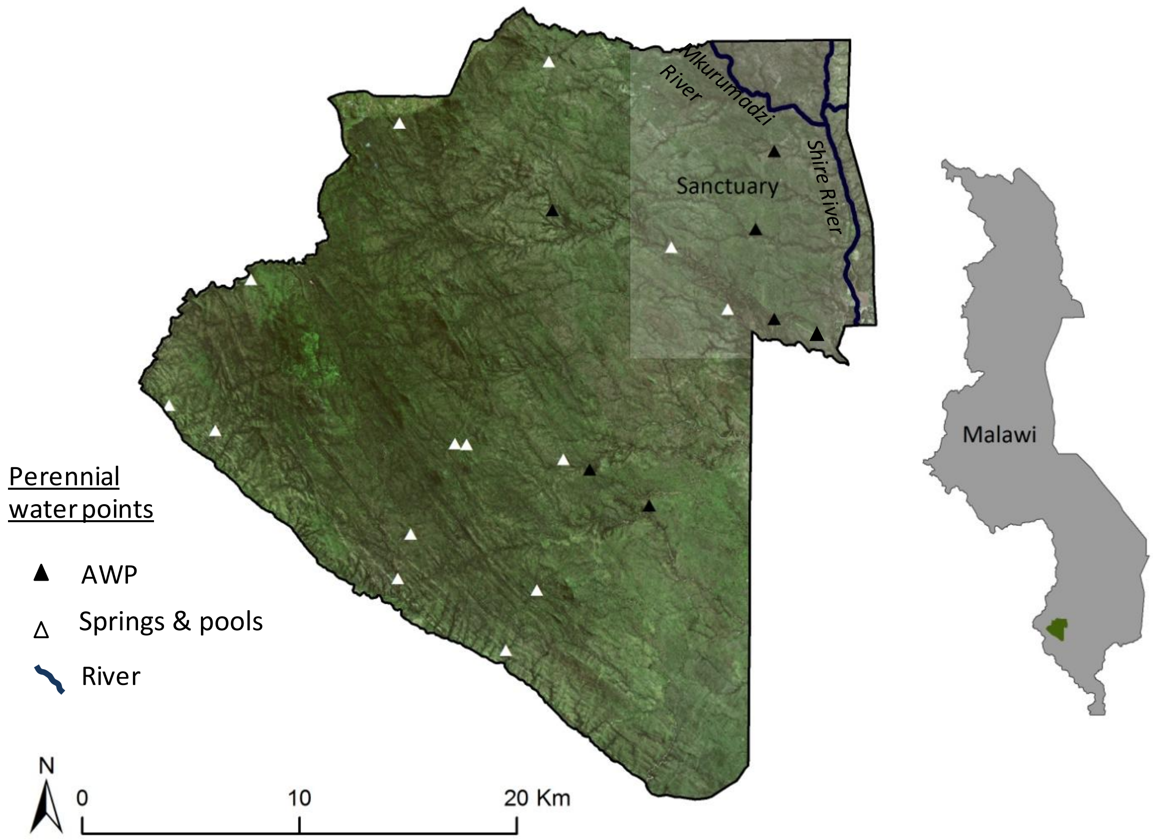

Figure 1.1. Perennial water points, including rivers, AWPs (artificial water points) and springs in the Majete Wildlife Reserve, and location of the reserve in Malawi

The impacts of altering the natural state of water availability are complex and potentially detrimental for vegetation in fenced areas (Owen-Smith, 1996; Gaylard et al., 2003; Owen-Smith et al., 2006). AWPs create foraging opportunities across a wider area during the dry season than would naturally be available to elephants (Leggett, 2006; Chamaillé-Jammes et al., 2007b; Shannon et al., 2009). By effectively buffering against the dry season water shortages that naturally occur in savanna systems (Chamaillé-Jammes et al. 2007a), water supplementation allows year-round use of foraging areas by elephants (Loarie et al., 2009a; Young et al., 2009). If water availability becomes too high in an area, elephants may not need to shift home ranges seasonally to access new foraging areas (Shannon et al., 2006), as extensive foraging is possible due to the wide availability of surface water (Grainger et al., 2005; Harris et al., 2008; Loarie et al., 2009b). This creates a more persistent elephant presence in their preferred areas, which can inhibit plant recovery from browsing and lead to a loss in woody vegetation (O'Connor et al., 2007). Wider dry season elephant ranging is enabled by higher water availability and this drives the development of homogenous browsing levels across a landscape (Redfern et al., 2005), thus compromising ecosystem spatial heterogeneity (Gaylard et al., 2003).

In water-scarce landscapes, elephant impacts are concentrated around water points (Ben-Shahar, 1993; Franz et al., 2010) but vegetation further from water remains undisturbed in plant refuges – a situation which contributes to ecosystem heterogeneity (Chamaillé-Jammes et al., 2009). Ecosystem heterogeneity affects most ecological processes and has significant consequences for biodiversity and ecosystem resilience (Rogers, 2003; Farmer, 2010). If water is supplemented in a reserve, then the position of AWPs has consequences for vegetation and ecosystem spatial heterogeneity. Over-utilization of vegetation can result if AWPs are positioned too close to one another or to existing natural water, as this causes piospheres around individual water points to merge and a homogenization of browsing levels across a landscape (Gaylard et al., 2003; Boundja & Midgley, 2009; Landman et al., 2012). These effects on vegetation of changing surface water distribution are more pronounced in smaller fenced reserves (Duffy et al., 2002; Loarie et al., 2009a).

Majete's vegetation needs to be carefully monitored to prevent large-scale woody cover loss, particularly in areas such as the Sanctuary where there is an abundance of perennial water points and the terrain is flatter. Past studies have found that elephants use hilly areas less than areas of flatter terrain (Nellemann et al., 2002) and are likely to be present at greater densities in areas with high water availability, provided that adequate forage material is available (de Beer & van Aarde, 2008; de Knegt et al., 2011). Additionally, Staub (2009) found that elephant browsing levels were higher in vegetation types which are dominant within the Sanctuary. Therefore, reserve managers are particularly concerned over elephant impact on woody vegetation in the Sanctuary. Riverine vegetation is a preferred vegetation type of elephants in Majete (Sherry, 1989; Staub et al., 2013) and occurs in mature stands along the perennial rivers situated in the Sanctuary. These habitats are also important for other wildlife species, so vegetation degradation needs to be prevented. Water management strategies may provide a suitable method for reducing elephant impacts within Majete. Baseline data collection and frequent monitoring needs to be undertaken in the reserve in order to document elephant-induced vegetation changes, particularly in vulnerable areas such as near perennial water points. Water management policies should be based on sound ecological principals, so that potentially deleterious effects of water supplementation in Majete can be avoided.

This study presents findings on the historic vegetation changes which have occurred in Majete, and on elephant interactions with water points in the reserve and the subsequent impacts on vegetation. An assessment was made of woody vegetation cover changes in Majete between 1985 and 2010. Whether points of woody cover loss were associated with particular environmental variables, including proximity to perennial water points, was also assessed. The perennial water points in the reserve were then monitored to determine which were visited most by elephants, and thus affected by higher levels of elephant browsing and pathway use. Factors affecting overall elephant use of water points were examined. Data gathered in this study will aid Majete in developing sound water management strategies so that elephant distribution and their impacts on vegetation can be appropriately managed.

1.2. Research goal and objectives

1.2.1. Goal

To provide Majete Wildlife Reserve with guidelines for water point placement and management based on scientific, ecologically sound research. The guidelines will aim to address future water point placement to manage elephant vegetation impacts, preserve spatial heterogeneity and prevent biodiversity loss.

1.2.2. Objectives

1. To determine if woody vegetation cover has changed in Majete since 1985 and identify whether woody cover changes, if any, are associated to particular environmental variables.

2. To determine whether elephant perennial water point use (elephant visits to water points, browsing and path use levels around water points) in Majete is random, or whether water points are used differentially. If differential use is observed, identify factors affecting water point usage.

1.3. Research questions

a) Has woody cover in Majete decreased since the elephant reintroductions?

b) In which areas has loss of woody cover been highest?

c) Are points of reduced woody cover found closer to perennial water points?

d) Which other factors (fire and other environmental variables) are associated with points of reduced woody cover?

e) Do elephants use water points in Majete randomly?

f) If elephant water point use is not random, which factors affect use?

g) Should any artificial water points in the reserve be closed or constructed?

1.4. Thesis structure

This thesis is composed of five chapters, three of which (Chapters Two, Three and Four) have been compiled as stand-alone manuscripts to facilitate publication in peer-reviewed journals. As such, there is some repetition within the thesis. Some cross-referencing of results between Chapters Three and Four is included in the thesis, to aid the reader in understanding the links between chapters. Chapter Five has been prepared to serve as both a discussion chapter for this thesis and to provide a management recommendations document for African Parks Majete (Pty) Ltd. Therefore, the literature review results are not discussed in Chapter Five.

Chapter Two presents a literature review on the nature of surface water impacts on African savanna elephant (*Loxodonta africana*) distribution and their associated impacts on vegetation. The results presented show when, where and how past research on elephant-water interactions has been undertaken. Findings of past research were then synthesized to provide guidelines to conservation practitioners as to how water supplementation can be used to manipulate elephant distribution and their effects on vegetation.

Chapter Three describes the historical woody vegetation cover changes in Majete through the spatial analysis of remotely-sensed images from 1985, 1990, 2000 and 2010. The extent and distribution of point of woody cover loss were identified and effects of a range of environmental variables (proximity to perennial water points, fire history and other environmental variables) were assessed. Woody plant conservation in Majete was then discussed in light of the study findings.

In Chapter Four, perennial water points used most by elephants in Majete were identified and factors affecting this use were analysed. The effects of water point characteristics (type, size and water quality) and habitat context (surrounding vegetation type, elevation and proximity to other water points) were tested. The implications for elephant and water point management were then discussed.

Chapter Five summarizes the main research findings of the thesis, and proposes recommendations for water point management in Majete.

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Chapter Two:

The influence of natural surface water and artificial water points on African elephant (*Loxodonta africana*) distribution and vegetation impacts: A review

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2.1. Abstract

African elephant (*Loxodonta africana*) populations are being increasingly isolated in remaining wilderness areas and fenced reserves, particularly in southern Africa. The restriction of elephant populations to limited areas requires that their impacts (as a keystone species) on vegetation are carefully managed. Elephants are water-dependent and their foraging activities are constrained by surface water availability; therefore the position and availability of surface water in reserves affects the distribution and intensity of elephant impacts on vegetation. In this review, research findings of peer-reviewed literature studying the impacts of surface water on elephant distribution and their impacts on vegetation were synthesized. Using an analysis of sampled peer-reviewed articles, insight is provided as to when, where and how information on this subject was gathered. Whether researchers synthesized their findings into practical management recommendations for conservation practitioners was also assessed. The potential to use surface water manipulation as a tool to manage elephant distribution and their vegetation impacts is also discussed, and factors to be taken into consideration before making changes to natural surface water availability are outlined.

2.2. Introduction

In the last century, African elephant (*Loxodonta africana*) populations have been restricted to smaller ranges than were previously occupied, in response to a range of anthropogenic pressures (de Boer et al., 2013). Elephant populations were drastically reduced in the last century by safari hunting and ivory-poaching (Skarpe et al., 2004). The rapid increase in sub-Saharan human populations has driven an increase in rural settlements, subsistence agriculture, deforestation and unsustainable hunting of wildlife (Newmark, 2006), and as a result remaining elephant populations are being increasingly isolated. There has also been a rise in the number of reserves being fenced, particularly in southern Africa, in response to human-elephant conflict (van Aarde & Jackson, 2007; Hayward & Kerley, 2008). Despite the continent-wide decrease of elephant populations, the species' population growth in well-protected conservation areas is high, requiring reserve managers to manage high population densities (Blanc et al., 2007; Loarie et al., 2009a). This poses many challenges for ecosystem conservation, especially in fenced areas, as elephants are a driver of change in woody vegetation (Baxter & Getz, 2005; Mapaure & Moe, 2009; Vanak et al., 2012). Elephants are also water-dependent and their distribution is constrained by the presence of surface water (Redfern et al., 2003; Chamaillé-Jammes et al., 2007b; Martin et al., 2010), consequently position and availability of water points affects elephant impacts on vegetation. Networks of artificial water points (AWPs) have been developed in many reserves to stabilise dry season water availability, provide viewing opportunities for tourists and increase the extent of dry season forage areas for wildlife (Redfern et al., 2005; Chamaillé-Jammes et al., 2007a; Loarie et al., 2009a). The effects of increasing surface water availability in fenced areas supporting elephants are complex and can lead to degradation of vegetation (Gaylard et al., 2003; Owen-Smith et al., 2006; van Aarde & Jackson, 2007; Loarie et al., 2009a). There is a profusion of research effort devoted to understanding how surface water affects elephant distribution and elephant impacts on vegetation - this highlights the importance of water management in African conservation areas.

This review synthesizes research findings of peer-reviewed literature on (i) how surface water affects elephant distribution and their impacts on vegetation, (ii) how surface water manipulation can be used to manage these and (iii) what factors should be taken into consideration before changing natural surface water availability. An analysis of peer-reviewed research articles was used as an indication of when, where and how information on this subject was gathered. Whether researchers synthesized their results into practical management recommendations was also assessed. This review does not include studies of how surface water provision affects elephant population growth rates.

2.3. Surface water influence on elephant distribution and impact on vegetation

Elephant dependence on surface water means that their distribution in a landscape is strongly shaped and determined by surface water availability and distribution (Verlinden & Gavor, 1998; Redfern et al., 2005; Chamaillé-Jammes et al., 2008; Smit & Ferreira, 2010). In regions with distinctly seasonal rainfall, surface water availability and location dictate regional elephant distribution during dry seasons (Chamaillé-Jammes et al., 2007b; de Beer & van Aarde, 2008; de Knegt et al., 2011). In the wet season water is highly available, through ephemeral pools or rivers, which facilitates a wider dispersal of elephants in a landscape (Stokke & du Toit, 2002; Smit et al., 2007b; Ngene et al., 2009). They are able to utilize both ephemeral and perennial water points (Redfern et al., 2005) and so shift ranges between seasons according to water availability. In the dry season elephants are concentrated around remaining perennial water points, so at this time of year woody vegetation around water points is vulnerable to overuse (Laws, 1970; Ben-shahar, 1996; Gaugris & van Rooyen, 2009). As mixed-feeders, elephants switch their dietary preferences between seasons depending on the available forage quality (Shrader et al., 2012). In the wet season they preferentially graze, as green grass provides high quality forage, then change to browsing in the dry season as woody plant material offers a more nutritious fodder choice than moribund grasses (Bax & Sheldrick, 1963; Osborn, 2004). The concentration of elephants (and other herbivores) around perennial water in the dry season results in the development of vegetation use patterns around water points, known as the piosphere effect (Lange, 1969). Piospheres consist of a ‘sacrifice area’ devoid of vegetation immediately adjacent to a waterhole, followed by increasing vegetation quality with increasing distance from water, which is a function of herbivore water and forage needs (Lange, 1969; Thrash & Derry, 1999). Vegetation further away from water points is protected from elephant browsing in a range of partial to absolute spatial refuges (Ben-Shahar, 1993; O’Connor et al., 2007). Piospheres do not develop symmetrically around water points (Farmer, 2010; Fullman & Child, 2012; Landman et al., 2012) as herbivores are distributed unevenly in landscapes in response to a range of factors (Chamaillé-Jammes et al., 2007b; Harris et al., 2008; de Knegt et al., 2011). African savanna systems support herbivores from a variety of feeding guilds, so piosphere effects are expressed in both herbaceous and woody vegetation, the latter of which is particularly affected by elephants (Thrash & Derry, 1999).

Seasonal changes in elephant range location occur in response to changes in rainfall (Young et al., 2009; Martin et al., 2010; Birkett et al., 2012). Natural surface water availability and vegetation productivity are primarily driven by rainfall, and both resources determine local elephant densities (Loarie et al., 2009b; Martin et al., 2010; de Boer et al., 2013). Young et al. (2009) measured whether spatial use by elephants differed between wet savannas (disturbance-driven) and dry savannas (climatically-driven). They found that vegetation productivity influenced elephant spatial use according to seasonal resource limitations. In water-limited environments, greener than average vegetation is consistently sought after, yet elephants are restricted to those areas holding water (Harris et al., 2008; Loarie et al., 2009b; Ngene et al., 2009). High rainfall (or an increase in water availability) reduces water restrictions and this allows elephants to access a wider foraging area (Martin et al., 2010). Hence, where surface water is not limiting, vegetation preferences have a greater influence on elephant movements than water (Grainger et al., 2005; Shannon et al., 2006; Loarie et al., 2009b; Matawa et al., 2012).

The shift in elephant range in response to seasonal water availability has important implications for woody vegetation. In the wet season, elephant dispersal and the dietary switch to include more grass provides respite to woody plants in dry season foraging areas, allowing recovery during high rainfall periods (O'Connor et al., 2007). Franz et al. (2010) described how rainfall affects elephant impact on woody vegetation around waterholes in Etosha National Park, Namibia. With a decrease in rainfall, vegetation use gradients shifted away from water (i.e. enlargement of piosphere effects) as elephant browsing pressure increased around the remaining waterholes due to the lack of ephemeral surface water. Higher rainfall led to a decrease in the use gradient (lessening of piosphere effects), but the recovery time for woody vegetation around waterholes was lengthy and the legacy effects of heavy browsing from dry years remained evident in woody vegetation for years.

Surface water availability also has varying impacts on elephant home range size and location depending on the characteristics of the environment. Elephant home ranges are functions of key resource abundance and distribution, with home range size balanced between the need to drink daily and the need to access enough food (de Beer & van Aarde, 2008; Wittemyer et al., 2008). Generally, when water and good-quality forage are easily accessible locally, home ranges are smaller (Grainger et al., 2005; Young et al., 2009) and more intensively used as travel distance to key resources is short (Grainger et al., 2005; de Beer & van Aarde, 2008; Thomas et al., 2011). Although home ranges are small when resources are abundant, elephant landscape-scale distribution is wider as a greater range of habitats can be utilized (Thouless, 1995; Dolmia et al., 2007). This creates a spreading of elephant effects on vegetation across a landscape (Loarie et al., 2009a) with patches of high impacts on vegetation within small home ranges (Shannon et al., 2006; de Beer & van Aarde, 2008; Birkett et al., 2012). In resource-scarce landscapes elephants are concentrated in regions where surface water and forage are available (Chamaillé-Jammes et al., 2008; Shannon et al., 2010) and home ranges in arid areas are usually larger than for elephants in wetter, resource-rich areas (Young et al., 2009). Limited water availability causes a concentration of elephants around water points, but the poor quality forage present in arid environments forces elephants to keep large home ranges so that sufficient foraging areas are incorporated (Lindeque & Lindeque, 1991; Verlinden & Gavor, 1998; Harris et al., 2008).

The effect of surface water distribution on elephant home range size has also been found to differ between the sexes, due to sexual differences in physiology and behaviour. Elephants have been found to be sexually segregated in habitat selection and feeding preferences (Stokke & du Toit, 2002; Smit et al., 2007b; Shannon et al., 2010). Bulls have respectively lower metabolic demands to meet than individuals in breeding herds (consisting of pregnant or lactating cows, and juveniles) (du Toit & Owen-Smith, 1989) and are therefore less constrained by water (Stokke & du Toit, 2002; Smit et al., 2007b). In Chobe National Park, Botswana, Stokke and du Toit (2002) found that elephant bulls roamed further from water and used a greater variety of habitat types in the dry season than breeding herds. They proposed that bulls can utilize habitats further from water points than breeding herds, and that these habitats usually offer high quality forage material as they are less impacted by water-restricted breeding herds. This was confirmed by Smit et al. (2007c) in the Kruger National Park, South Africa, who found that elephant bulls used a much wider variety of habitats and were generally found further from water than breeding herds. They reasoned that breeding herds maintained home ranges closer to rivers, which provide reliable sources of water and forage, allowing individuals to meet higher metabolic demands. This sexual difference in ranging behaviour in response to water and forage availability therefore has a critical influence on how elephants affect vegetation in a landscape. Shannon et al. (2011) found that bulls were more likely to exhibit destructive feeding behaviour, browse longer and feed less selectively than females. Therefore, elephant bulls individually affect vegetation more than the average elephant in a breeding herd, with bull-impacts on vegetation located at greater distances from water.

Ecological systems are naturally complex. Elephant distribution in a landscape in response to water determines the distribution of elephant impact on vegetation, which in turn has an impact on the ecological heterogeneity of an ecosystem. Natural variation in surface water availability causes spatial heterogeneity in elephant usage of vegetation in a landscape. In the past, conservation management policies were aimed at stabilizing ecosystems and maintaining an even-usage of vegetation by herbivores across areas (Rogers, 2003; Farmer, 2010). It is now widely accepted that savanna systems are inherently unstable (Owen-Smith et al., 2006; Munyati & Sinthumule, 2013), and that heterogeneity in herbivore distribution and foraging is integral to maintaining overall ecosystem heterogeneity and resilience (Rogers, 2003). The drivers of vegetation change in savannas - rainfall, herbivory and fire - have manifest differently in vegetation depending on landscape characteristics, such as terrain and soil depth (Baxter & Getz, 2005; Vanak et al., 2012). As a result, ecosystems naturally consist of differing levels of vegetation structure, condition and cover. This natural variation contributes positively to ecosystem heterogeneity, and therefore to overall ecosystem resilience (Gaylard et al., 2003; Rogers, 2003; Farmer, 2010). As a key resource, water point spatial location and availability will create regions of differential elephant use of vegetation which will affect vegetation heterogeneity (Rogers, 2003; Chamaillé-Jammes et al., 2009). Piospheres are created by limited water availability which causes elephants to concentrate at remaining perennial waterholes (Thrash & Derry, 1999; Franz et al., 2010). In Hwange National Park (Zimbabwe) vegetation heterogeneity was found to persist in piospheres, and piospheres actually contributed positively to landscape vegetation heterogeneity (Chamaillé-Jammes et al., 2009). Where water points are closely spaced, elephants move easily between them and this may cause merging of piospheres, as well as the development of an even level of browsing across a landscape (Owen-Smith, 1996; Landman et al., 2012). If elephants are able to forage in all parts of a reserve, the regional differences in browsing intensity are lost which results in the homogenization of elephant vegetation impacts (Gaylard et al., 2003; Boundja & Midgley, 2009). Vegetation degradation and plant species loss can occur in these situations, especially in fenced reserves where elephant dispersal is limited (Duffy et al., 2002; O'Connor et al., 2007; van Aarde & Jackson, 2007).

2.4. Trends in elephant-water relations literature

2.4.1. Review methods

To provide insight as to where and how information on elephant distribution and their impacts on vegetation in relation to water were obtained, available online peer-reviewed literature was selected and examined. The objective of this review was to select scientific papers to represent the body of literature referring to how surface water affects elephant distribution and impacts on vegetation. Please note that this review did not follow a systematic review process and the group of articles selected for this review was not a comprehensive collection of all the information available on this topic. Article selection was restricted to include only published articles in peer-reviewed journals. Grey-literature was excluded from the selection as the credibility of research methods and results could not be verified in such a range of papers of varying levels of quality. Research on forest elephants (*Loxodonta africana cyclotis*) was also excluded from the review because surface water is not limited seasonally in tropical forests as it is in savanna systems and does not restrict forest elephant movements (Young et al., 2009). Only the online search engines Google Scholar and Web of Science were used in the search process, and keywords were selected through an iterative process. Keywords which produced the most relevant results included "African elephants"; "*Loxodonta africana*"; water; vegetation; distribution; landscape. Searches returned 87 articles on Web of Science and around 1500 on Google Scholar. Only the first 20 pages on Google Scholar were used, as a sharp decrease in relevance of articles occurred after this point. Papers were first filtered for relevance by title, and then by abstract, after which articles were subjected to a full-text review to assess final relevance. Final relevance was based on whether the article specifically aimed to test (or to discuss in a review article) the effect of surface water on elephant distribution or their impacts on vegetation. If a study aimed to achieve a more general objective (such as determining elephant habitat use or ranging behaviour in a particular area), but specifically tested for the effect of surface water on elephant distribution or vegetation impacts, it was included in this review. Through this process, 64 articles were selected, five of which were review papers (Appendix I).

2.4.2. Results and discussion

Of the 64 articles selected 89% were published between 2000 and 2013. Glover (1963) and Laws (1970) were two of the earliest articles reviewed, and both simply acknowledged that surface water distribution affects elephant dry season distribution. After exploring the literature, it appears that the intricacies of how surface water affects elephant movements, and the myriad effects of this on vegetation, were primarily studied in the last decade. Opinions on the use of surface water manipulation to manage elephant distribution and vegetation impacts also seemed to diversify most after 2000, during which time conflicting ideas developed as to the usefulness of this strategy (see the debate between Chamaillé-Jammes et al., 2007c and Smit et al., 2007d).

Nine countries were represented in the reviewed literature (excluding review papers), with six outlying articles that studied elephants in multiple countries in Sub-Saharan Africa. There were 18 studies (31%) in the reviewed literature which were based in South Africa, and eight from both Botswana and Zimbabwe – literature from these three countries accounted for over half of the articles selected. Of the studies conducted in South Africa, 11 (61%) were based in the Kruger National Park. Studies in Botswana were mostly based in Chobe National Park and Northern regions (7 articles, 88%), while those in Zimbabwe were mainly located in Hwange National Park (5 articles, 63%). The prominent locations of research - Kruger, Chobe and Hwange National Parks - also currently support some of the highest density elephant populations in Africa (IUCN Elephant Database, 2013 – see www.elephantdatabase.org). This research effort is evidence of existing concerns over how to manage such large populations and their impacts on vegetation in these parks (see Owen-Smith et al., 2006; van Aarde & Jackson, 2007).

Of the 52 studies conducted in single countries, 30 (59%) were undertaken in unfenced reserves, 13 (26%) in partially fenced reserves (or some fencing present) and only 9

(18%) were based in fenced reserves. Most of the fenced and partially fenced study areas within the selected literature were located in South Africa. Vegetation in fenced reserves is usually impacted differently by elephants than in open systems, as it is under a greater degree of continuous browsing pressure as seasonal shifts in elephant ranges are limited or prevented (Duffy et al., 2002; Loarie et al., 2009a). With the large number of fenced reserves in southern Africa it is critical that sufficient data is generated to inform the management of elephant impacts in fenced areas (Wiseman et al., 2004). In large enough fenced reserves, surface water management can be used to influence the intensity and distribution of elephant impacts on vegetation (Smit et al., 2007c), and it appears from the reviewed literature that more research is needed on this management option specifically in fenced reserves.

To determine whether studies in the sampled literature were conducted at small or large scale (Table 2.1.), the study site size distribution was explored. For this, study site size was specified as only continuous areas.

Table 2.1. Study site sizes in sampled peer-reviewed literature studying elephant interactions with surface water

Study site size (km^2)	Number of sampled articles	Percentage
Total	**59**	**100**
82-1000	12	20.3
1000 - 4000	5	8.5
4000 - 10000	3	5.1
10000 - 15000	7	11.9
15000 - 20000	18	30.5
20000 - 80000+	6	10.2
Multiple study sites	8	13.6

Research on elephant-water relationships in multiple countries and multiple study sites was categorized as ‘Multiple’, and reviews were excluded. The smallest study site was Pongola Game Reserve in South Africa at 82 km^2 and the largest study site was around 80 000 km^2, covering unprotected open areas and National Parks across northern

Botswana. In the sampled literature, there was a large range in study area size, although most studies (18 of 59) were undertaken in larger reserves between 15000 and 20000 km^2. This is due to the abundance of studies conducted in the Kruger National Park, which is over 19 000 km^2 in size (this only includes studies which specify that their study area was within the Kruger and not in conservation areas surrounding the Kruger, such as private land or conservation land over the Mozambique border). The distribution of studies across such a range of study site sizes is encouraging. Elephant impacts on vegetation and their distribution across landscapes vary in response to water at different spatial scales (Chamaillé-Jammes et al., 2007b; de Knegt et al., 2011). Elephant impacts on vegetation differ with reserve size, notably if the reserve is fenced or movement is prevented by unfavourable surrounding habitats (Duffy et al., 2002; van Aarde & Jackson, 2007; Loarie et al., 2009a). Gathering information on elephant interactions with surface water at different spatial scales is, therefore, important to furthering our understanding of how populations can best be managed.

A variety of methods have been used in gathering data on elephant distribution and vegetation impacts. Of the 59 sampled articles (excluding review articles), 46 studies (78%) explored elephant distribution in relation to water, 16 (27%) studied elephant vegetation impacts in relation to water and 4 (7%) studied the both elephant distribution and their impacts on vegetation in relation to surface water. Elephant distribution data in the sampled articles were mainly obtained using telemetry sampling methods (via satellite- or GPS-collared elephants) (Table 2.2.). Other well utilised methods included the use of aerial survey methods and waterhole monitoring, among others.

Data on elephant vegetation impacts in the reviewed literature was most commonly collected using direct vegetation sampling methods (Table 2.2.). Yet, few studies used remote-sensing of vegetation or long-term fixed-point photography to document elephant impact on vegetation. The usefulness of satellite imagery to monitor elephant vegetation impacts is only just beginning to be explored and it is a tool that has great potential (Chamaillé-Jammes et al., 2009; Simms, 2009; Munyati & Sinthumule, 2013). Remote-sensing of vegetation is advantageous as it is more time-efficient than direct sampling of vegetation, it is often a cheaper alternative (freely available imagery from Landsat and MODIS satellites) and it is possible to monitor long-term changes in vegetation condition as archived images are available dating back to 1972 (see www.landsat.usgs.gov). However, vegetation changes detected through remote sensing are very difficult to specifically attribute to a particular herbivore species, even when using high resolution imagery. Therefore, further research in the use of remote-sensing to detect elephant impacts in vegetation is required.

Table 2.2.: The methods of data collection used in sampled literature which studied either elephant distribution or elephant impacts on vegetation

Elephant distribution studies		**Elephant vegetation impact studies**	
Data collection method	No. of articles	Data collection method	No. of articles
Telemetry	25	Crop-raiding data	1
Physical tracking	4	Vegetation sampling	11
Vehicle sighting	2	Remote sensing	2
Presence/absence transect	3	Fixed-point photos	2
Pathway data	1		
Aerial survey	12		
Waterhole monitoring	6		
Total:	**46**	Total:	**16**

Overall, of the 64 reviewed articles, only 52% synthesized their results into practical recommendations as to how to manage elephant distribution and impacts on vegetation through surface water manipulation practices. Many of the articles which did not make recommendations for surface water management were those with more general objectives, for example, those assessing elephant distribution outside of reserves to inform conservation corridor design. Of those articles using methods that only assessed elephant distribution (46), 21% proposed recommendations on how elephant impacts on vegetation could be managed through surface water manipulation. Although inferences can be made as to elephant vegetation impacts based on distribution data, the variation and details of elephant impacts on vegetation cannot be fully understood without obtaining data on vegetation condition. Understanding particular impacts of elephants on vegetation is important in determining how they affect ecosystem spatial heterogeneity and diversity, which need to be accounted for when designing water management policies in reserves (Gaylard et al., 2003; Farmer, 2010). Of the articles that synthesized results into management recommendations, only 49% acknowledged the importance of spatial heterogeneity when recommending water management actions. Farmer (2010) showed that only recently, in the late 2000s, have the effects of water management on ecosystem spatial heterogeneity been recognized or acknowledged as an important system characteristic to manage and preserve. Water point management in reserves needs to acknowledge and encourage spatial heterogeneity in elephant browsing (Chamaillé-Jammes et al., 2007b; Farmer, 2010).

2.5. Water supplementation

Many articles sampled in the review process discussed whether water should be supplemented in conservation areas and some gave practical advice as to how this management intervention strategy should be undertaken. A synthesis is provided of findings in the literature on the impacts that surface water supplementation can have on elephant distribution and their impacts on vegetation.

2.5.1. Effects of water supplementation on elephant distribution and vegetation impacts

Natural water sources in reserves have often been supplemented with artificial water points (AWPs) to stabilize natural water fluctuations, so that wildlife densities can be maintained in arid areas throughout the dry season (Chamaillé-Jammes et al., 2007a). This may be to provide good wildlife viewing opportunities for tourists (Omphile & Powell, 2002; Shannon et al., 2010), to increase the extent of herbivore foraging ranges, or to facilitate faster population growth of water-dependent species (Redfern et al., 2005; van Aarde & Jackson, 2007; Loarie et al., 2009a). Artificial water provision can buffer against the natural seasonal fluctuations in surface water availability by ensuring that water is available even in arid areas during the dry season (Grainger et al., 2005; Chamaillé-Jammes et al., 2007a; Smit & Ferreira, 2010), thus overriding normal biological processes (Smit et al., 2007a; de Knegt et al., 2011). Higher water availability in previously dry areas can support elephant habitation in areas that naturally would have been avoided during dry seasons (Chamaillé-Jammes et al., 2007a; Shannon et al., 2010). With reduced water limitations elephant presence can be sustained in arid areas throughout the year (Leggett, 2006), but seasonal variation in local elephant densities is minimized (Shannon et al., 2006) which can lead to vegetation degradation through persistent elephant browsing (Loarie et al., 2009a; Landman et al., 2012). In natural systems, seasonal water shortages protect vegetation in dry areas from elephant browsing through the creation of spatial refuges (O'Connor et al., 2007; Gaugris & van Rooyen, 2009).

Examples of the impacts that surface water supplementation have on elephant ranging behaviour, and the consequences for woody vegetation, are plentiful. In Tembe Elephant Park, South Africa, there is a concern over elephant impacts on rare sand-forest vegetation. Shannon et al. (2009) showed, by mapping and modelling elephant pathways in relation to water, that elephants would not be present in many sand-forest areas if water had not been provided. Leggett (2006) found in his observations of elephant ranging behaviour in the Hoanib River catchment (Namibia) that in the dry season breeding herds were only observed to range 2% of the observation time in areas further than 10km away from natural perennial water points. After the addition of AWPs, 98% of the time they were found over 10km away from natural water points. The seasonal difference in their ranging patterns was reduced and elephants, therefore, concentrated in particular areas year-round. Leggett (2006) speculated that once good quality foraging areas had been reduced through perennial elephant browsing, elephants would once again resume seasonal range changes. This, however, is unfortunately only likely to occur after the degradation of foraging areas.

AWPs were also placed in Chobe National Park (Botswana) in an attempt to attract elephants away from the Chobe River to reduce elephant browsing pressure in the dry season on riverine vegetation (Omphile & Powell, 2002; Kalwij et al., 2010). Kalwij et al (2010) found that vegetation structure around the AWPs changed, as elephant impacts on vegetation became similar around AWPs as they were around natural pans. Essentially, adding AWPs changed the distribution of elephant impacts on vegetation in Chobe. Fullman and Child (2012) showed that elephant impacts on vegetation away from the Chobe River now extend much further than previously thought, and that their impacts have been extended into previously arid areas due to the presence of AWPs.

A consideration, brought to light by Fullman and Child's study (2012), is that if impacts on vegetation around water points are so extensive, then the danger of piosphere-merging increases if AWPs are not positioned far enough from other water sources (Owen-Smith, 1996; Landman et al., 2012). In small fenced or isolated reserves where elephant presence is perennial, the impact of artificial water provision on vegetation can be accentuated. Fenced reserves create artificially closed systems by creating resource limitations, preventing elephant dispersal and limiting their access to water (Boone & Thompson Hobbs, 2004; van Aarde et al., 2006; van Aarde & Jackson, 2007). Even without water supplementation distances to water points may be minimal and easily accessible to elephants when they are confined to a fenced reserve. The Hluhluwe-Imfolozi Park (South Africa) is not excessively small at 900km^2, but distances to water points do not exceed 8km and elephants are able to browse anywhere, so the spatial relation between elephant vegetation impacts and proximity to water is weak (Boundja & Midgley, 2009). Vegetation degradation can arise through the loss of variation in seasonal elephant ranging patterns caused by fencing and AWP provision (Loarie et al., 2009a). Homogenisation of elephant browsing levels on vegetation can then occur should water points be positioned too close to one another.

Perhaps the best-known example of the effects of regional-scale water provision in a fenced reserve comes from the Kruger National Park, South Africa. The water provision policy implemented in the Kruger (1930s to 1990s) saw the creation of over 300 AWPs in the reserve which was entirely fenced at the time (Gaylard et al., 2003). It is thought that by stabilizing the natural fluctuations in seasonal water availability (Redfern et al., 2005) this not only facilitated high rates of elephant population growth (Gaylard et al., 2003; Chamaillé-Jammes et al., 2008; Shrader et al., 2010), it also allowed wider-ranging of elephants across the reserve (Gaylard et al., 2003; Owen-Smith et al., 2006; van Aarde et al., 2006). Landmark research by Redfern et al. (2005) showed that in the Kruger the majority of the park areas were within 5km of perennial water. Borehole removal would increase the area further than 5km from perennial water by 19% in the northern regions, and 8% in the southern regions. Construction of AWPs created a super-abundance of surface water in the Kruger, although the natural availability of surface water still ensures that the park is not a particularly water-limited landscape for elephants in normal rainfall years (Redfern et al., 2005; Smit et al., 2007c). Owen-Smith (2006) suggested that reducing the number of AWPs in the park would contribute to restricting the dry season distribution of elephants. After Smit et al. (2007b) found that breeding herds were usually found closer to rivers (hotspots of high forage quality and water availability) and bulls closer to AWPs, they proposed that AWPs were more important as water sources for bulls. They speculated that closure of AWPs would primarily affect bull distribution (Smit et al., 2007b, 2007c), but this could help mitigate elephant impacts on vegetation as bulls feed more destructively than females or juveniles (Shannon et al., 2011). Elephants are able to forage in most areas in the Kruger throughout the year, which inhibits vegetation recovery from browsing (Owen-Smith et al., 2006; O'Connor et al., 2007). There has been a decline in woody plant components in the Kruger, thought to partly be due to the wide, unrestricted use of vegetation in the park by elephants (Shannon et al., 2008; Vanak et al., 2012; Munyati & Sinthumule, 2013); although authors caution that this could also be attributed to other drivers. Levick et al. (2009) assessed the relative impacts of fire and herbivory on savanna vegetation structure in the Makhohlola exclusion plots (southern Kruger) and suggested that mega-herbivores, notably elephants, have likely had a significant impact on the vegetation in the Kruger.

Although use of AWPs in conservation areas has caused changes to the nature of elephant interactions with vegetation, artificial water supplementation is considered one of the few tools available for reserve managers to use in managing elephant impacts on vegetation (Chamaillé-Jammes et al., 2007b; Smit et al., 2007c).

2.5.2. Surface water manipulation to manage elephants

Manipulation of surface water through the addition or removal of AWPs has been frequently cited as an option for controlling local elephant densities and impacts on vegetation (Glover, 1963; Owen-Smith, 1996; Chamaillé-Jammes et al., 2007b; Young & van Aarde, 2010). However, many factors need to be taken into consideration when changing the natural surface water availability. From the literature, guidelines can be drawn to advise reserve managers as to whether or not to use water manipulation to manage elephants, and if used what should be taken into consideration.

In reserves where natural surface water is sufficient, managers should be cautious of adding AWPs (such as for tourism purposes) because this is likely to spread elephant effects across the landscape (Loarie et al., 2009a), encourage locally high, inter-seasonally persistent elephant densities (Shannon et al., 2006; van Aarde & Jackson, 2007), and intensive use of home ranges (de Beer & van Aarde, 2008; Birkett et al., 2012). These changes in elephant foraging activity can cause homogenisation of browsing levels or overuse of vegetation (Grainger et al., 2005; Owen-Smith et al., 2006; de Beer & van Aarde, 2008). In fenced reserves particularly, vegetation may become degraded if water is supplemented. If water is abundant within a fenced reserve seasonal variation in browsing pressure is reduced as elephants do not need to shift their ranges to access water in dry seasons, which places plants under year-round browsing pressure (Owen-Smith et al., 2006; van Aarde & Jackson, 2007).

In water-limited environments where natural water availability is scarce, it is possible with the addition or removal of AWPs to influence elephant distribution and impacts should the need arise (Leggett, 2006; Chamaillé-Jammes et al., 2008; Loarie et al., 2009a). The effect which water manipulation can have on elephants will, however, be influenced by rainfall levels and the resulting development of ephemeral water points. In the Kruger, seasonal water contributes significantly to the overall availability of surface water and elephants (due to their high mobility) are able to exploit these seasonal water points and are, therefore, less limited by perennial water points than other herbivores in the Kruger (Redfern et al., 2005; Smit et al., 2007c). Hwange National Park is naturally water-limited and elephants mainly rely on AWPs in the dry season. Large-scale water supplementation has allowed the persistence of one of the highest elephant population densities in Africa through the dry season, and those areas without AWPs contain low numbers of elephants (Chamaillé-Jammes et al., 2008).

Elephant population density affects how the population reacts to water manipulation (Smit et al., 2007c; Martin et al., 2010). In Hwange, Chamaillé-Jammes et al. (2007b) showed that reducing numbers of AWPs would create a more even distribution of elephants among water points through crowding at individual waterholes. At a low population density the consequences of this may not be severe on woody vegetation around water points, although the impact may be deleterious at a high population density. Franz et al. (2010) showed that removing AWPs may prevent wide-ranging impacts on vegetation in water-limited environments (i.e. Etosha National Park in Namibia), but side-effects such as crowding around remaining water points could cause an increase in piosphere effects and loss of woody vegetation around perennial water.

AWPs should be carefully positioned, as placement too close to one another can cause overuse of woody vegetation in high-water density areas (Owen-Smith, 1996; Shannon et al., 2008; Landman et al., 2012). Elephants are capable of travelling large distances daily, if required, so water point additions may still result in utilization of preferred vegetation further from water than expected (Fullman & Child, 2012). The goal of any water supplementation programme should be to create a range of elephant vegetation use levels across a reserve, which will promote ecosystem heterogeneity (Gaylard et al., 2003).

Ecosystem heterogeneity will differ at a range of scales, as will elephant distribution in relation to surface water (Chamaillé-Jammes et al., 2007b). If water points are spaced widely then piosphere impacts will be intensive around water points (Harris et al., 2008; Franz et al., 2010). However, this can contribute to ecosystem heterogeneity through the concentration of herbivores at fewer perennial water points (Chamaillé-Jammes et al., 2009) and offer woody plants respite from persistent browsing in a range of partial and absolute spatial refuges from water (O'Connor et al., 2007).

An understanding of elephant vegetation preferences at a local-scale is also needed to anticipate which areas may be impacted by elephants through changing surface water availability (Smit & Ferreira, 2010; Thomas et al., 2011). Surface water placement should be avoided in or near sensitive vegetation, as providing water may create greater foraging opportunities for elephants in these areas (see Shannon et al., 2009; Landman et al., 2012). Vegetation heavily utilized by elephants because of its proximity to water should also be protected from intensive fires as the combination of these drivers may lead to woody plant loss and degradation of foraging areas (Levick et al., 2009; Vanak et al., 2012). Perennial rivers are particularly vulnerable to changes in elephant browsing caused by water supplementation as they are a preferred breeding herd habitat type (Smit et al., 2007b; Smit & Ferreira, 2010). Rivers function as resource hotspots as they provide reliable dry season sources of water, and riverine vegetation provides shade and high quality forage (Leggett, 2006; Smit & Ferreira, 2010). Water point positioning should also take proximity to seasonal rivers into account as these rivers act as spatial refuges for riparian vegetation (see Smit & Ferreira, 2010). O'Connor et al. (2007) recommends that if water supplementation is required, then AWPs position should be limited to those areas in which water was historically available in past dry seasons.

Scientists have reached a high degree of consensus on managing elephants (Owen-Smith et al., 2006) and with regards to the use of surface water manipulation there are clearly some circumstances in which supplementing of water can be helpful in controlling elephant impacts on vegetation, but others in which it would be detrimental to increase water availability. The characteristics of the reserve which vary spatially and temporally should be taken into account (Thomas et al., 2011). These include how water-limited the area is, the availability of other key resources (e.g. forage quality) and the elephant population density and sex ratio (Smit et al., 2007c). Each individual reserve differs in these characteristics and Smit et al. (2007c) even state that, "the influence of water provision will be area and population specific and it is cautioned to transfer conclusions between populations and areas".

Aside from fencing and controlled burning, surface water manipulation is one of the few intervention strategies available to manage elephant distribution and impacts in reserves (Smit et al., 2007a). Most researchers agree that altering natural water availability should be approached with extreme caution because the impacts of this practice are difficult to predict, complex and potentially detrimental (Owen-Smith et al., 2006; van Aarde et al., 2006; van Aarde & Jackson, 2007). In some circumstances, such as in fenced reserves with limited access to reliable water sources, water supplementation is required. Additionally, managers cannot only manage water availability to control elephant impacts as they must provide water to a variety of other less mobile water-dependent species. However, any alterations to natural surface water availability should be approached with an understanding of the potential consequences of elephant impacts on vegetation (Smit et al., 2007c). Long-term monitoring programmes of elephant impacts on vegetation should be implemented if natural surface water availability is changed, so that management has appropriate information upon which to base future actions (see Kalwij et al., 2010; Landman et al., 2012). There is currently greater awareness of the need to manage elephant population effects, particularly on ecological heterogeneity, as well as population growth or size (Smit & Ferreira, 2010). Surface water manipulation, in appropriate contexts, can offer managers one potential intervention strategy to control elephant distribution and impacts on vegetation.

2.6. Conclusions

Surface water availability affects elephant distribution, home range size and shape, and elephant impacts on vegetation. The influence that surface water can have on elephants, and subsequently on vegetation, is affected by a number of factors. These include how naturally water-limited the landscape is, the availability and quality of forage, and the density and sex ratio of the elephant population.

Findings from this review indicated that there are a number of gaps in research on elephant-water interactions. Most research on this topic has been conducted in only a few African countries: South Africa, Botswana, Zimbabwe and Namibia. Large elephant populations in other African countries, such as Zambia and Mozambique, remain largely unstudied. Additionally, more research should be undertaken in fenced reserves, or areas in which elephants are isolated to provide greater insight into how the elephant-water relationship can be used to manage confined elephant populations and their impacts on vegetation. There are also some methods, such as remote-sensing of vegetation, which remain largely unexplored in their application to studying the impact of elephants on vegetation. Lastly, it was found that there were few studies in which results were synthesized into practical management recommendations for conservation practitioners. This should be addressed if we are to enable the effective management of increasing elephant populations, particularly in fenced or isolated reserves.

The complexity of the impacts that surface water manipulation can have on an ecosystem warrants careful consideration of water point placement or closure. Understanding of the ecological effects of surface water availability on elephant distribution and impacts on vegetation is essential when making changes to water availability in an area.

2.7. Acknowledgements

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Chapter Three

Woody vegetation change in Majete Wildlife Reserve, Malawi: response to wildlife re-introduction, fire and water management

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3.1. Abstract

Savanna systems are inherently dynamic and vegetation fluctuates between woody and grass-dominated states. Changes in vegetation structure and composition are primarily driven by rainfall, fire and herbivory. Majete Wildlife Reserve (Malawi) has recently undergone reform. During this process wildlife was reintroduced, artificial water points were built, wildfire management strategies implemented and the reserve was fenced. Concerns have arisen as to the impact of reintroduced herbivores and fire regime changes on the woody vegetation in the reserve. Within fenced reserves, herbivores are confined to a limited area and forage availability is therefore limited; a situation which can produce changes in vegetation. Alterations to surface water availability may also cause changes in foraging pressure on vegetation. Using available remote-sensed vegetation classifications of Majete for 1985, 1990, 2000 and 2010 historic woody vegetation changes were identified. Woody vegetation cover loss was high between datasets for 2000 and 2010. Therefore, the 2010 dataset was further analysed to identify environmental features of those areas that experienced woody cover loss. The effect of fires, proximity to perennial surface water, altitude, slope, aspect and ridge or valley characteristics on woody cover loss were explored using descriptive statistics and Principal Components Analyses. Rainfall data indicated the occurrence of drought

periods between 2000 and 2010 which likely affected woody cover in Majete. Points of woody cover loss were found to be distributed differently in different regions of the reserve and woody cover loss appears to have been caused by the synergistic effects of drought and different disturbances (fire and herbivory). Water, wildlife and fire management policies are discussed in the light of the findings of this study.

3.2. Introduction

Savanna systems are composed of a fluctuating mixture of woody and grass-dominated states, and changes between these states are driven by rainfall and disturbance mechanisms - notably fire and herbivory (Baxter & Getz, 2005; Sankaran et al., 2008). Variations in fire seasonality, frequency and intensity create landscape mosaics of differing vegetation structure (Trapnell, 1959; van Wilgen, 2009). High rainfall can promote woody plant growth, whilst low rainfall or drought leads to woody plant loss (de Beer et al., 2006), through synergisms with disturbances, such as fire and herbivory (Sankaran et al., 2005, 2008; Mapaure & Moe, 2009). Savanna ecosystems are inherently dynamic and heterogeneous, making them difficult to manage as isolated systems within fenced reserves (Gaylard et al., 2003; Rogers, 2003; van Aarde & Jackson, 2007). The complexity of interactions between herbivory and fires, rather than their impacts as single drivers of vegetation change, creates differing vegetation mosaics across landscapes, contributing to spatial heterogeneity in a landscape (Midgley et al., 2010; Vanak et al., 2012). Vegetation management and monitoring are, therefore, critical facets of reserve management because managers must provide sufficient forage for wildlife, preserve plant diversity and maintain ecosystem health and resilience (O’Connor et al., 2007; Farmer, 2010).

Monitoring of vegetation changes produces data upon which reserve management decisions can be based. Remote sensing of satellite imagery is a useful method for detecting and monitoring vegetation changes (Palmer & van Rooyen, 1998; Xie et al., 2008; Farmer et al., 2010). Analysing satellite imagery is often a more time- and cost-effective way to assess vegetation changes (particularly structural changes) than through physical vegetation sampling (Turner et al., 2003). Remote sensing can allow detection of changes over a much wider landscape than normally possible through physical sampling, and images from different seasons can be easily obtained. In addition , archived images are available dating back to the 1970s, providing invaluable historical data on vegetation which would otherwise not be available for many reserves (see www.landsat.usgs.gov).

Majete Wildlife Reserve (Majete) is a small (700km^2) fenced national park in southern Malawi which underwent rehabilitation, beginning in 2003. Prior to this, indigenous large mammal species had been eradicated in the area due to excessive poaching and human settlements had encroached upon the reserve (reserve manager, pers. com.). Since 2003, Majete has been fenced, wildlife has been reintroduced, artificial water points (AWPs) were constructed and fire control policies were implemented. Over 200 elephants were re-introduced to Majete, along with other herbivorous species. The indigenous wildlife populations which previously inhabited the area were able to migrate seasonally to and from the reserve (Sherry, 1989) as Majete was unfenced.

With the reintroduction of wildlife (notably elephants), fencing and the addition of AWPs to Majete between 2003 and 2012, concerns have arisen as to how woody vegetation in the reserve may have been affected. Fences affect ecological processes within reserves in many ways, such as by severing wildlife dispersal routes and creating a limit to the amount of food, water and space available to wildlife (Boone & Thompson Hobbs, 2004; Newmark, 2006; Whyte & Joubert, 2010). The confinement of wildlife to a fenced reserve, such as Majete, also requires that conservation managers monitor the effects of wildlife on vegetation. Monitoring vegetation in Majete is particularly necessary as past research has found that when a continuous elephant presence exists in a fenced reserve, it can lead to woody vegetation loss (van Aarde et al., 2006; van Aarde & Jackson, 2007; Loarie et al., 2009). Elephants play a key role in shaping savanna woody vegetation (Baxter & Getz, 2005; Vanak et al., 2012), as their feeding habits and large dietary requirements can produce dramatic changes in woody cover (Mapaure & Moe, 2009), structure and diversity (Laws, 1970; Cumming et al., 1997; Hayward & Zawadzka, 2010). Artificial water supplementation can also change the intensity and distribution of herbivore impacts on woody vegetation. In many reserves surface water is supplemented by AWPs to provide tourists with good wildlife viewing opportunities (Omphile & Powell, 2002; Shannon et al., 2010), to increase the extent of herbivore foraging ranges, or to facilitate faster population growth of water-dependent species (Redfern et al., 2005; van Aarde & Jackson, 2007; Loarie et al., 2009). Furthermore, because artificial water provision buffers against seasonal fluctuations in surface water availability, (Grainger et al., 2005; Chamaillé-Jammes et al., 2007a; Smit & Ferreira, 2010) this allows herbivores to remain in previously drier or water-devoid areas throughout the year (Leggett, 2006; Chamaillé-Jammes et al., 2008). This causes a decrease in seasonal variation of herbivore movements, which can result in woody plant loss caused by year-round browsing pressure (Shannon et al., 2006; Landman et al., 2012).

The objectives of this study were to quantify woody vegetation changes which occurred in Majete between 1985 and 2010, and to gain insight into why changes might have occurred by identifying environmental features of areas that underwent change. The underlying hypothesis for the study was that woody cover loss would be greater closer to perennial water points at lower elevations. Woody vegetation cover maps, classified from Landsat TM satellite imagery by GeoTerraImages (Pty) Ltd. were provided by the reserve for use in this study. These maps were used to quantify the extent to which woody cover had changed in Majete between 1985, 1990, 2000 and 2010. A decrease in woody cover was detected between 2000 and 2010. This dataset was further analysed to identify environmental features which could explain the decrease in woody cover. Variables (including fire return rate, proximity to perennial surface water, altitude, slope, aspect and ridge or valley characteristics) were compared for areas where change had occurred and those which remained woody. Correlations between variables were also tested. Finally, the impacts that rainfall, fire, wildlife and artificial water provision may have had on woody vegetation in Majete were discussed based on the study findings.

3.3. Methods

3.3.1. Study area

Majete covers 700km^2 of the lower Shire valley region in southern Malawi and the reserve's eastern boundary is situated along a section of the Shire River. Terrain is highest in the western region, consisting of steeply undulating hills dissected by river valleys. Eastwards, closer to the Shire River, the terrain flattens with the decrease in altitude. The Shire River and Mkurumadzi River are the only perennial rivers transecting the reserve. Majete contains approximately five perennial springs, although in the wet season ephemeral pools contribute greatly to surface water availability. The range of expected annual precipitation is between 680-800mm in the eastern low-lands and 700-1000mm in the western highlands, falling in a distinct wet season commencing in November and ending in early April (Malawian Department of Climate Change and

Meteorological Services).

Vegetation classes in Majete are often merged and difficult to definitively classify, but their distribution is strongly associated with soil type and depth (Sherry, 1989). Majete is primarily covered in woodland. Vegetation types present in the reserve fall within the Miombo woodland ecoregion, although tree density and canopy height vary between types. Miombo woody plant density and structure is driven by rainfall, herbivory (notably by elephants) and fire (Trapnell, 1959; Lawton, 1978; Malmer, 2007; Sankaran et al., 2008). Vegetation types include riverine associations (along the larger river systems, characterized by *Kigelia africana, Philenoptera violacea* and *Euphorbia ingens*); low altitude mixed deciduous woodland (Eastern areas of Majete at altitudes between 205-280m, *Acacia spp.*, *Sclerocarya birrea* and *Sterculia spp.*); ridge-top mixed woodland (on flatter ridge-tops and higher ground, 220-300m, *Terminalia sericea*, *Diospyros kirkii* and *Diplorhynchus condactylcarpon*); medium altitude mixed deciduous woodland (eco-tonal vegetation between eastern low lands and western hilly area, 230-410m, *Brachystegia boehmii*, *Pterocarpus rotundifolius*, *Diospyrus kirkii* and *Combretum spp.*); and high altitude miombo woodland (western hilly areas, 410-770m, *Brachystegia boehmii, Julbernardia globiflora*, *Burkea africana, Diplorhynchus condylocarpon* and *Pterocarpus angolensis*) (Sherry, 1989).

Wildlife reintroductions in Majete were undertaken in stages. Initially, only a 140km² 'Sanctuary area' of the North-eastern section of the reserve was fenced in 2003 for wildlife re-introduction purposes. Fencing of the remainder of the reserve was only completed later, by 2008, after which wildlife was reintroduced to this area. The Sanctuary fence was removed between May and September 2011. Over 2550 individual animals of 14 species have been reintroduced to Majete; species reintroductions to the Sanctuary and the rest of the reserve are detailed in the Appendix (Appendix Two).

Seven solar-powered AWPs were created in Majete, four of which were placed in the Sanctuary and three in the rest of the reserve. No fire management took place prior to 2003 in Majete, after which efforts have been made to reduce wildfires and to practice prescribed controlled burning treatments, in which the Sanctuary has been protected from fire. Fires are, therefore, an annual occurrence in the dry season. Conversely, wildfires occur randomly in the dry season and are often started in rural settlements outside the reserve.

3.3.2. Woody cover data

This study used a time series dataset of woody cover classifications created by GeoTerraImage (Pty) Ltd. for African Parks Majete (Pty) Ltd. The datasets represented woody vegetation cover in Majete for four periods, 1985, 1990, 2000 and 2010. Maps for each of the periods were derived independently through unsupervised classification of multiple Landsat TM images, following protocols used in both the South African Standard Classification for Land-Cover and FAO Land-Cover Classification System (LCCS) standards. For each of the time frames, multi-seasonal images were used in the classification (Table 3.1.) to avoid seasonal bias in vegetation classification.

Table 3.1. Details of the individual Landsat images used to produce the woody cover classifications, subsequently used in this study

Year	Landsat 168-071	Landsat 167-071
1985	19841030	
	19850118	
	19861223	
1990	19900711	19890701
	19910511	19910723
2000	20000425	20000824
	20000714	20001230
	20000831	20011014
	20001002	
	20010701	
2010	20090426	20110220
	20100328	20111018
	20110416	
	20110822	
	20111110	

As the resultant datasets were comprised of multiple images, they were considered as representative of the state of vegetation in Majete around the specified date of the woody datasets. The classified data showed two classes, namely woody and non-woody (open) vegetation. Due to the coarse spatial resolution of the raw satellite images (30m), the classification was based on the structural characteristics of the vegetation rather than floristic composition. Pixels (30x30m) were classified as either ‘woody’ if they contained vegetation communities with over 15% tree or shrub cover (as discernible on the Landsat imagery through desk-top mapping procedures), or ‘non-woody’ if tree or shrub cover was below 15%. The low threshold of woody cover used to classify non-woody pixels facilitated the detection of transformed pixels that had shifted from a woody to an open, grassy state. However, data on the magnitude of the change in vegetation types was undocumented. For example, dense woody vegetation transformed to a more open state with less than 15% woody cover would have been classified as a non-woody point, as well as vegetation which had shifted to a non-woody point from a naturally lower level of woody cover. The nature of the datasets also means that woody cover loss in more dense vegetation types (such as riverine associations) may have been underrepresented.

Datasets were checked to confirm the logical sequence of woody and non-woody cover between years. Accuracy assessment statistics were not available from GeoTerraImage (Pty) Ltd. for use in this study. Hence, there are no available confidence levels of the accuracy of the datasets. However the datasets were developed by a singular person, using consistent methodology, and therefore woody vegetation cover changes between years are well-represented and the datasets are comparable.

3.3.3. Environmental variables

Altitude, slope, aspect and TPI (Topographical Position Index) of pixels

A 30m resolution Digital Elevation Model (DEM) (Aster, 2012) (raster dataset) was obtained for Majete and used in the study analyses. Pixel altitude, slope, aspect and Topographical Position Index (TPI) values were derived from the DEM using ArcGIS version 10.1 (ESRI, 2012). Slope was presented as angular degrees (between 0-90˚) where 90˚ indicated a vertical slope and 0˚ a flat area. Aspect, was presented as bearing (-1 to 360˚) and provided a directional measure. An aspect value of -1˚ indicated a flat area with no directional facing, while values between 0˚ and 360˚ corresponded to circular compass readings (North at 0˚ or 360˚). TPI values were produced using TPI version 1.3a (Jenness, 2001) in ArcGIS with a neighbourhood radius of ten pixels (300m). TPI classifies landscape by relative elevation (low areas surrounded by high areas and vice versa) and, thus, highlights ridge tops, valley bottoms and mid-slope areas. Positive index values indicated that a pixel was higher than its surrounds, and negative values that it was lower.

Table 3.2. A summary of information about the environmental spatial data used in this study

Environmental spatial variables	**Source**	**Resolution**	**Uses**	**Range**
DEM	Aster	30m	Shows elevation above sea level	100m to 1200m
Slope	derived from DEM	30m	Shows the steepness of slopes	0˚ to 90˚ (90˚ a vertical slope, 0˚ flat areas)
Aspect	derived from DEM	30m	Indicates the directional facing of slopes	-1 to 360˚ (-1 no slope, North at 0˚ and 360˚)
TPI	derived from DEM	30m	Shows valleys and ridges	-1 to 1 (ridges positive values, valleys negative values)
Fire	MODIS	90m	Display of historical fire record over a landscape	Fire frequency between 2000 and 2011
Water points	Majete Wildlife Reserve	~10m accuracy (GPS)	Location of perennial water points in the reserve	-

Fire

MODIS standard fire products (MODIS, 2012) (part of NASA Land Use and Land Cover Program and the International Global Observation of Forest Cover Project) were used to supply 10 vector datasets of the fire history in Majete for each year between 2000 and 2011. Each yearly dataset was converted to raster (90m spatial resolution) and fire return rate was calculated by summing these datasets to show areas that had burnt several times during the decade. The MODIS fire products provided information as to when and where fires occurred in Majete.

Water points

Locations of perennial water points in Majete were obtained from the reserve's GIS records. Permanent water points present in 2010 included six AWPs, 13 springs and pools, and two perennial rivers. A distance raster was generated in ArcGIS and showed proximity to perennial water points, and to perennial rivers.

Rainfall

Non-spatial data of the annual precipitation received between 1984 and 2011 were obtained for the nearest weather station to the reserve in Chikwawa Boma, approximately 10km from the southern boundary of Majete. These were provided by the Malawian Department of Climate Change and Meteorological Services. Rainfall data were available as a cumulative amount per wet season. The wet season occurs in the summer months, usually beginning in November and ending in April of the following year. Therefore, a rainfall data point is presented for the wet season, and is written, for example, as the wet season of 2010/2011 (2010/'11). The Chikwawa Boma weather station lies within the same altitudinal range as the Eastern part of Majete and experiences a similar range of average annual rainfall (680-800mm). Thus, historic rainfall data were not representative of the Western highlands of Majete, but it was assumed that the rainfall pattern recorded for the Eastern lowlands region would have persisted in the Western highlands. I.e. if rainfall received in a year was average for the Eastern part of the reserve, it was assumed that the rainfall level would also be average for the Western part of Majete.

For each woody cover dataset, the satellite imagery used was sourced from different years (Table 3.1.). Rainfall received in the period between datasets was taken as what fell between the latest dates of images used in developing the datasets. For example, images from 1991 were used in developing the 1990 dataset, and images from 2001 were used for the 2000 dataset. Therefore, rainfall received by Majete between 1990 and 2000 will be taken from the end of 1991 (i.e. the wet season of 1991/1992; rain falls from November to April) to the beginning of 2001 (i.e. the wet season of 2000/2001).

3.3.4. <u>Data analysis</u>

Data processing

A mask was created to exclude artificially bare pixels from the analysis. This was achieved by buffering roads, fences, buildings, riverbeds and waterholes, effectively excluding these areas from the analyses. A 150m buffer was used to exclude cover along fences and a 60m buffer used along roads. A wide buffer was used along the park border to exclude management dirt roads that ran along the fence, as well as illegal logging activities that may have occurred on Majete's boundary pre-2003. A section of the reserve (approximately 40km^2) to the north-west of the reserve was also excluded from the analysis due to encroachment by subsistence farmers which occurred pre-2003. Pixels occurring in 150m of perennial river courses and waterholes were also excluded to avoid analysis of non-woody pixels created along river courses by flood events.

Woody cover change analysis

For the time series comparison, values were extracted from the masked woody cover datasets using ArcGIS. The numbers of woody and non-woody pixels in Majete were then compared across the time series. To determine the differences in woody cover loss between different areas in Majete the reserve was divided into three regions, namely the Western highlands, the Eastern lowlands and the Sanctuary area (Figure 3.1.). The division between the Western highland and the Eastern lowland regions was based on altitude, and also accounted for vegetation type and rainfall region. The Sanctuary was split from these regions due to its different management history. The Sanctuary area had contained a higher density of wildlife for a longer period of time than the rest of Majete and efforts were made to prevent the occurrence of fires in the Sanctuary. The Western highland included areas above 300m in elevation and was characterized by hilly and steeply-sloped topography at high altitudes, *Brachystegia* and *Julbernadia*-dominated vegetation types and relatively higher average annual rainfall (700–1000mm). The Eastern lowland included areas below 300m in altitude and was characterized by gentle topography at low altitudes, the predominance of vegetation types associated with lower altitudes (low altitude mixed deciduous woodland, ridge-top vegetation, riverine associations) and a relatively lower average annual rainfall (680–800mm). The Sanctuary area was delineated from the rest of the reserve along the Sanctuary fence line, but fell into the same rainfall region as the Eastern lowlands. Woody cover changes were compared between years for different areas to determine whether woody cover was moving towards a more open or closed state, and to identify when significant changes occurred.

To determine whether the historic rainfall patterns may have affected woody vegetation cover in Majete, rainfall received in the time periods between woody cover datasets was compared. Annual rainfall levels differ widely between years; therefore, rainfall data were temporally smoothed using a three year running mean. This provided an historic pattern of rainfall received between 1985 and 2010 (see Mills et al., 1995). Based on the running mean values of image dates used in developing the woody cover datasets, a weighted mean was also calculated for the dates of the woody cover datasets (for a detailed description see Appendix Two) and compared between datasets. The weighted mean of rainfall provides a indication of the rainfall which fell in the years of images used in creating the woody cover datasets, i.e. rainfall which would have directly affected vegetation in the image years.

To examine which environmental variables were associated with recent vegetation transformation, only pixels that had changed from woody to non-woody between 2000 and 2010 were used in the analysis (i.e. only those ‘new’ non-woody pixels) (Figure 3.1.). Per pixel point data (representing woody cover) were extracted from the 2010 woody cover dataset. Environmental values (fire return rate, altitude, slope, aspect, TPI, distance to permanent water) were extracted for each point and subsequently grouped by the three regions of Majete for use in statistical analyses. Descriptive statistics were then used to explore similarities between environmental variables of ‘new’ non-woody points in each of the areas. The data was spatially autocorrelated (an inherent property of spatial data); therefore, only descriptive statistics could be used to examine the data. Spatially autocorrelated data violates the requirement of statistical analyses (in testing for significant differences) for variables to be independent.

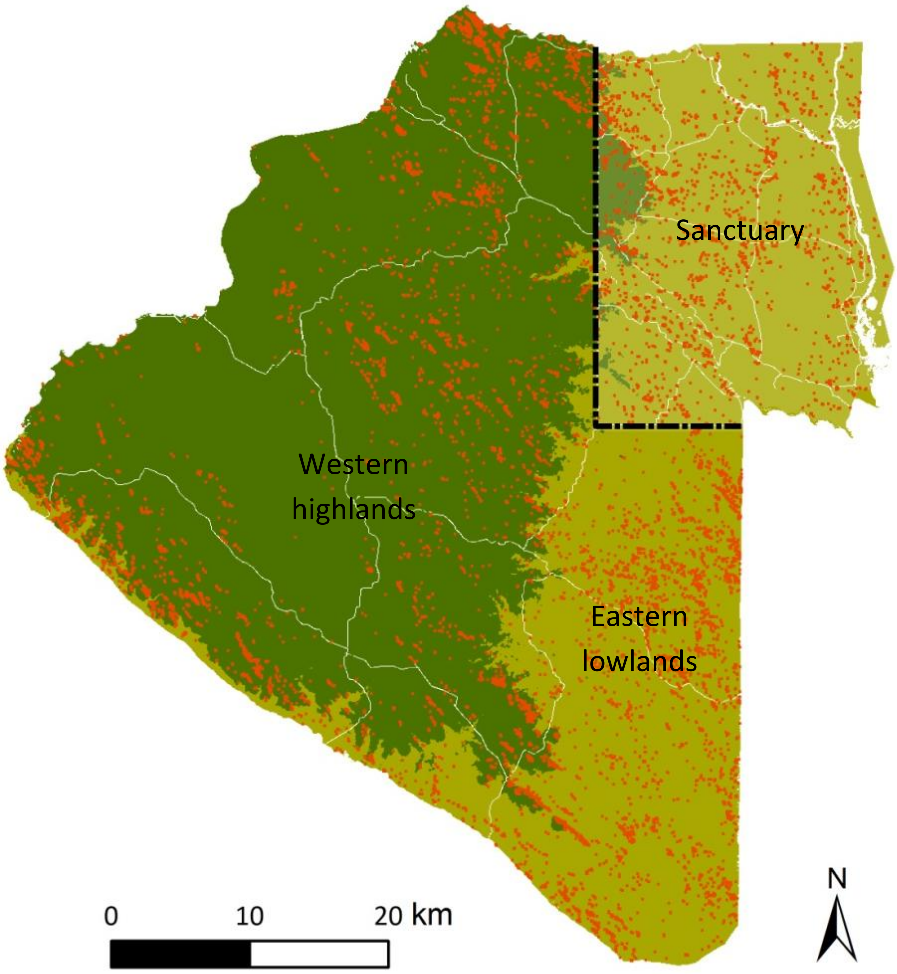

Figure 3.1.: Majete Wildlife Reserve was split into three different regions, the Sanctuary, Eastern lowlands and Western highlands. The divisions were based on elevation, rainfall region, vegetation type and topography. Regions outside of the Sanctuary were split along the 300m elevation contour. The Eastern lowlands are shown in light green and the Western highlands are shown in dark green. New non-woody points to be analysed are shown in red points on the map. The white points are areas which were buffered from the analyses (e.g. roads, fences).

A point alignment analysis was undertaken on PAST v. 1 to assess the alignment of new non-woody point clusters. The point alignment algorithm produced lines which intersected clustered new non-woody points. The geographic orientation of lines was further analysed to test whether alignment occurred more prominently in any direction and to gauge whether new non-woody point clusters were aligned differently in the three regions. The geographic orientation of lines lies in either the North/South, North-East/South-West, East/West or South-East/North-West direction. The line angle is calculated as degrees between North and South, 0˚ to 180˚ (Figure 3.2.).

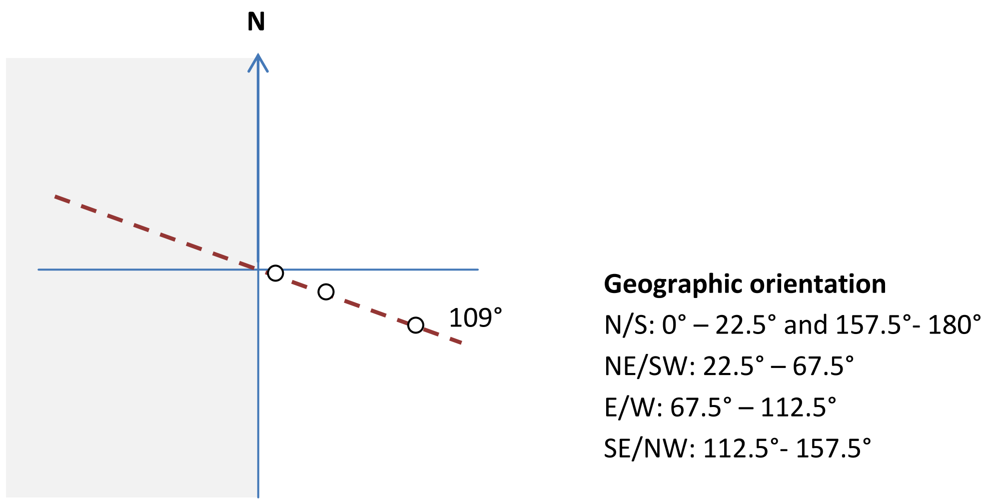

Figure 3.2.: Diagram of how the geographic orientation of lines between clustered points is calculated. The range degrees used to classify the orientation of a line are listed in the diagram

To test and interpret possible correlations between the topography-related variables of new non-woody points, Principal Components Analyses (PCAs) were conducted of slope, elevation (altitude) and TPI (ridge or valley characteristics) values of new non-woody points for each region of Majete. Although the data were spatially autocorrelated, PCAs were simply used to gauge the relationship between point variables. PCAs of new non-woody points were compared to PCAs generated for all points for each region. PCAs consisted of a correlation matrix of attribute data (of new non-woody points and 'all points' of a region) and were performed using Statistica for Windows version 11 (Statsoft Inc., 2013). Slope, elevation and TPI data were all included as active variables in the PCAs. Aspect values of points were not included in the PCA because the values are an index in which North is indicated both by values closer to 0˚ and closer to 360˚. Factors generated from the correlation matrix were then selected for the PCAs based on their Eigen values (generated from the correlation matrix) using the Kaiser Criterion and scree plot analysis for guidance, and only two factors had Eigen values greater than 1.

3.4. Results

3.4.1. Woody cover changes 1985-2010

In 1985 only 1.1% of Majete's total land cover was classified as non-woody cover. Many non-woody points were found along rivers in the form of sandbanks or open river beds (naturally devoid of vegetation as a consequence of seasonal river flow). Some non-woody points were also sites of known exposed rock outcrops (at high altitudes). There was little change in non-woody point cover between 1985 and 2000; however, non-woody cover increased between 2000 and 2010 (Figure 3.3.). Between 1985 and 1990, non-woody point cover increased to 1.2% (Table 3.3.), and between 1990 and 2000 non-woody pixel cover had increased to cover 1.4% of Majete's total land cover. Then between 2000 and 2010, non-woody point cover increased to 3.09% of Majete's total land cover.

In the three delineated regions of Majete (Sanctuary, Eastern lowlands and Western highlands), there were differences in the increase of non-woody cover (Table 3.3.). Between 2000 and 2010, non-woody cover increased from 0.5% to 1.4% in the Sanctuary, 0.4% to 1.8% in the Eastern lowlands, and 0.1% to 1.1% in the Western highlands.

Table 3.3. Area specific percentage change of non-woody cover in Majete detected between 2000 and 2010.

	Percentage of non-woody cover in:			
Year	**Majete**	**Sanctuary**	**Eastern lowlands**	**Western highlands**
2000	1.4%	0.5%	0.4%	0.1%
2010	3.09%	1.4%	1.8%	1.1%
Change	+1.7%	+0.9%	+1.4%	+1%

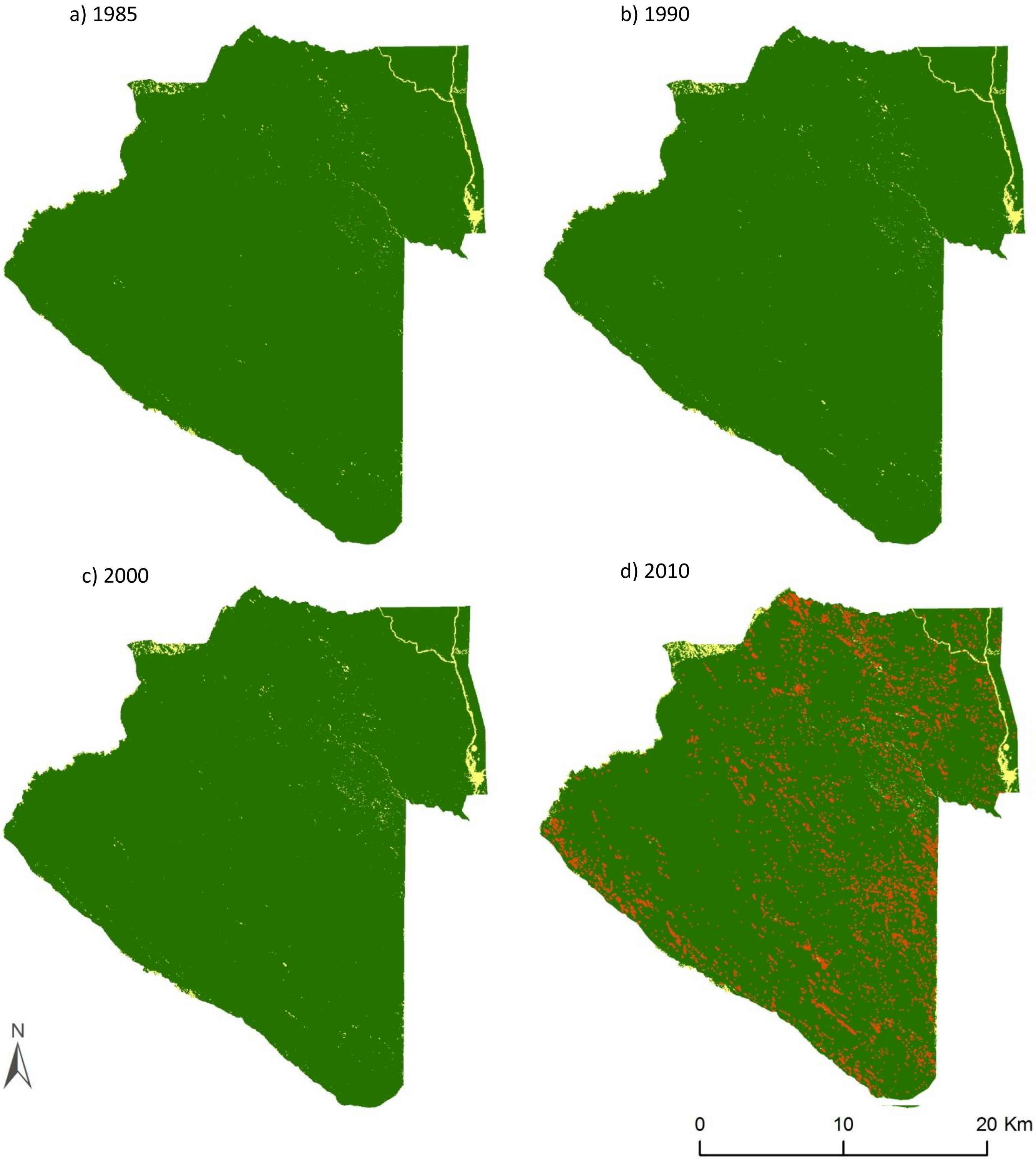

Figure 3.3.a, b, c and d - Maps showing the change in woody cover in Majete Wildlife Reserve from 1985 to 2010. Few points of 'non-woody' cover (representing a 30 x 30m plot with <15% woody cover) existed up until 2000, but an increase in non-woody points was noticed between 2000 and 2010. The 'new' non-woody points (those which appeared between 2000 and 2010) are shown in red dots in the 2010 image.

3.4.2. Woody cover change analysis

Rainfall

The rainfall pattern between 1985 and 2010 was erratic (Figure 3.4.). The smoothing action of the three year running mean of rainfall facilitated detection of dry periods (below the expected annual rainfall range for the Eastern region of Majete, 680-800mm) in the rainfall pattern. A dry period between the early and mid-1990s was followed by a long spell of above expected rainfall from the late 1990s through to the early 2000s. Two dry periods were experienced between 2000 and 2010, but these were divided by a three year wet period.

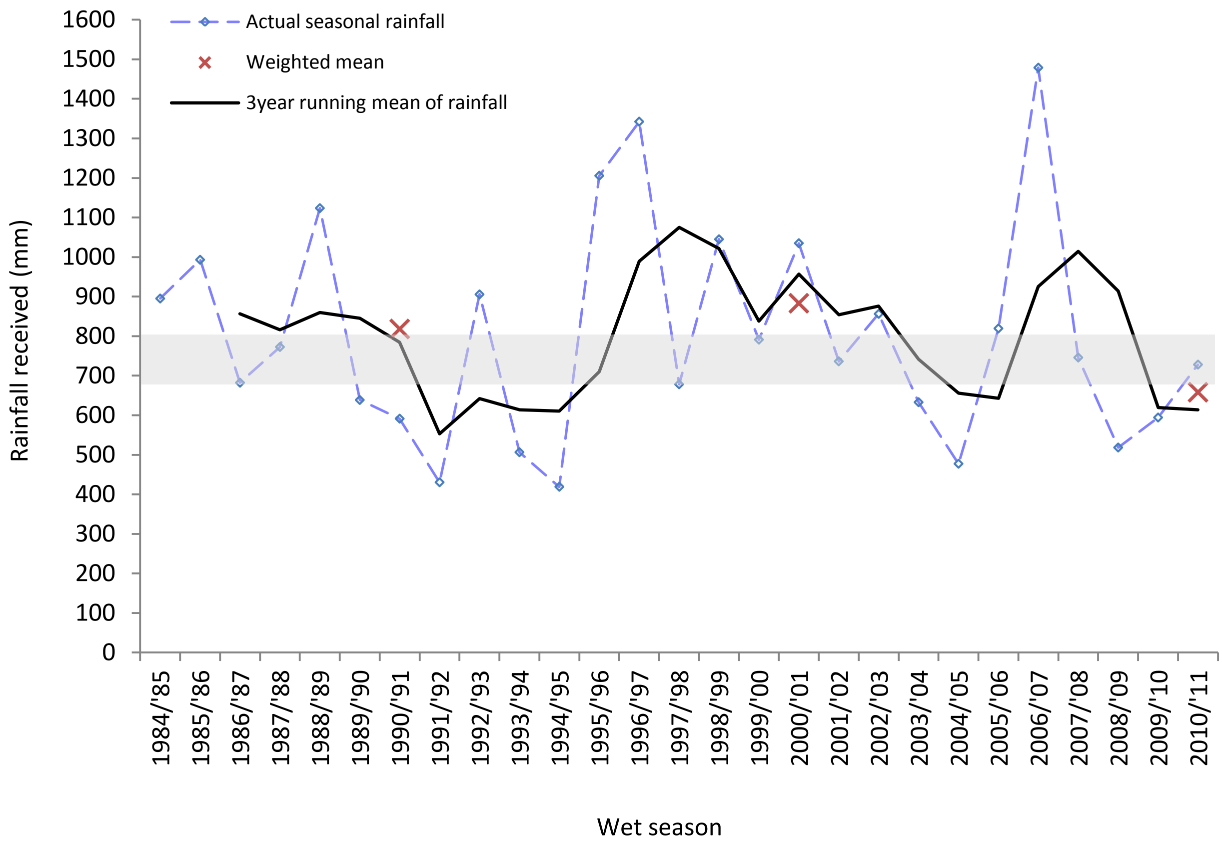

Figure 3.4. Seasonal rainfall recorded at Chikwawa Boma weather station (Malawi) between 1985 and 2011. The wet season begins in November and ends in April the following year and is written as, for example, 2011/'12 (rain which fell in the wet season of Nov.2011 to April 2012). The blue dotted line indicates the actual rainfall received in wet seasons. The black line shows the three year running mean of rainfall, calculated (by taking the mean rainfall for a year, and for the two years preceding it) to smooth the rainfall data and show the historic pattern of rainfall between 1985 and 2010. The red crosses indicate the weighted mean rainfall for each woody cover dataset (woody cover image of Majete Wildlife Reserve, generated through remote sensing of satellite imagery). The grey area shows the expected range of annual rainfall for the area (680-800mm).

Rainfall which fell in the imagery dates used in developing the woody cover datasets is shown by the weighted mean rainfall values (calculated for each woody cover dataset). The three year running mean pattern and the weighted mean values show that rainfall received in the area prior to the 1990 and 2000 datasets was above average, but immediately before the 2010 dataset, a dry spell occurred.

Proximity to perennial water

Much of Majete is in close proximity to perennial water (Figure 3.5., Table 3.4.), but 'new' non-woody points were not associated with proximity to perennial water points.

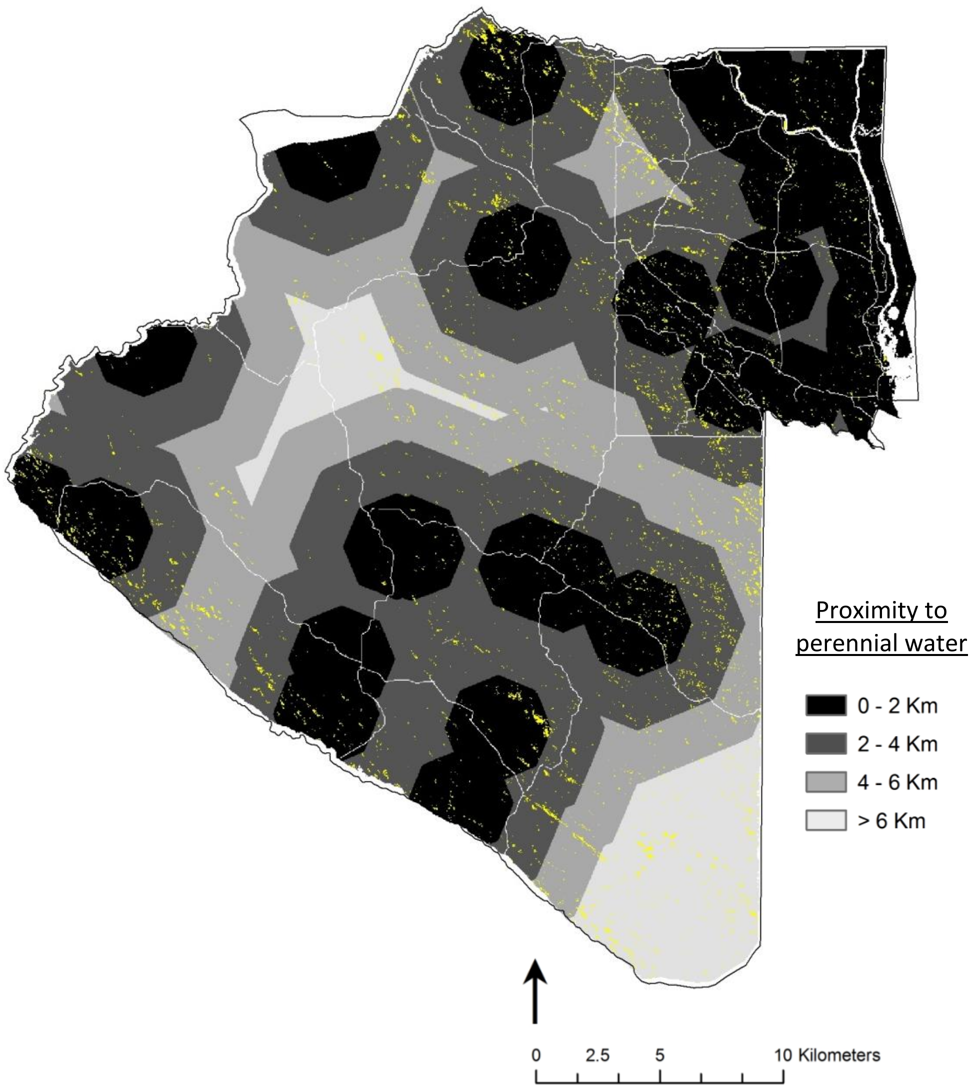

Figure 3.5. The map illustrates the distribution of new non-woody points (points of recent woody cover loss) across different distances to water points in Majete Wildlife Reserve. New non-woody points are shown in yellow. White areas are features which have been masked from the analysis (e.g. fences, rivers and roads).

Within the Sanctuary and the Western highlands, 8km was the furthest distance to perennial water and in the Eastern lowlands 10km was the furthest distance to perennial water. In the Sanctuary, 61% of the total area was within 2km of perennial water and 87% of the area was within 4km. In the Eastern lowlands, 27% of the area was within 2km of perennial water and 57% was within 4km. In the Western highlands, 25% of the area was within 2km of perennial water and 71% was within 4km of perennial water.

New non-woody point cover was calculated for each distance to perennial water class. There appeared to be no association between proximity to perennial water and new non-woody point distribution (Table 3.4.). The percentage cover of new non-woody points was calculated as a percentage of the available area in each distance to perennial water class.

Table 3.4.: Proximity of different areas of Majete Wildlife Reserve to perennial water and the percentage of new non woody points in the region found in different distance to water categories

	Proximity to perennial water	**% of Region**	**% cover of new non-woody points**
Sanctuary	0-2km	61	1.2
	2-4km	25	1.5
	4-6km	11	1.9
	6-8km	2	1.5
Eastern lowlands	0-2km	27	1.7
	2-4km	30	1.8
	4-6km	25	1.9
	6-8km	10	2.3
	8-10km	8	1.7
Western highlands	0-2km	25	1.5
	2-4km	46	1.0
	4-6km	25	0.8
	6-8km	4	1.0

Fire

The occurrence of fire differed across the different regions of Majete (Figure 3.6.). Very few fires occurred in the Sanctuary between 2000 and 2010. The new non-woody points in the Sanctuary had, on average, been burnt only once during this period. In the Eastern lowlands and Western highlands, new non-woody points experienced, on average, three and four fires respectively between 2000 and 2010.

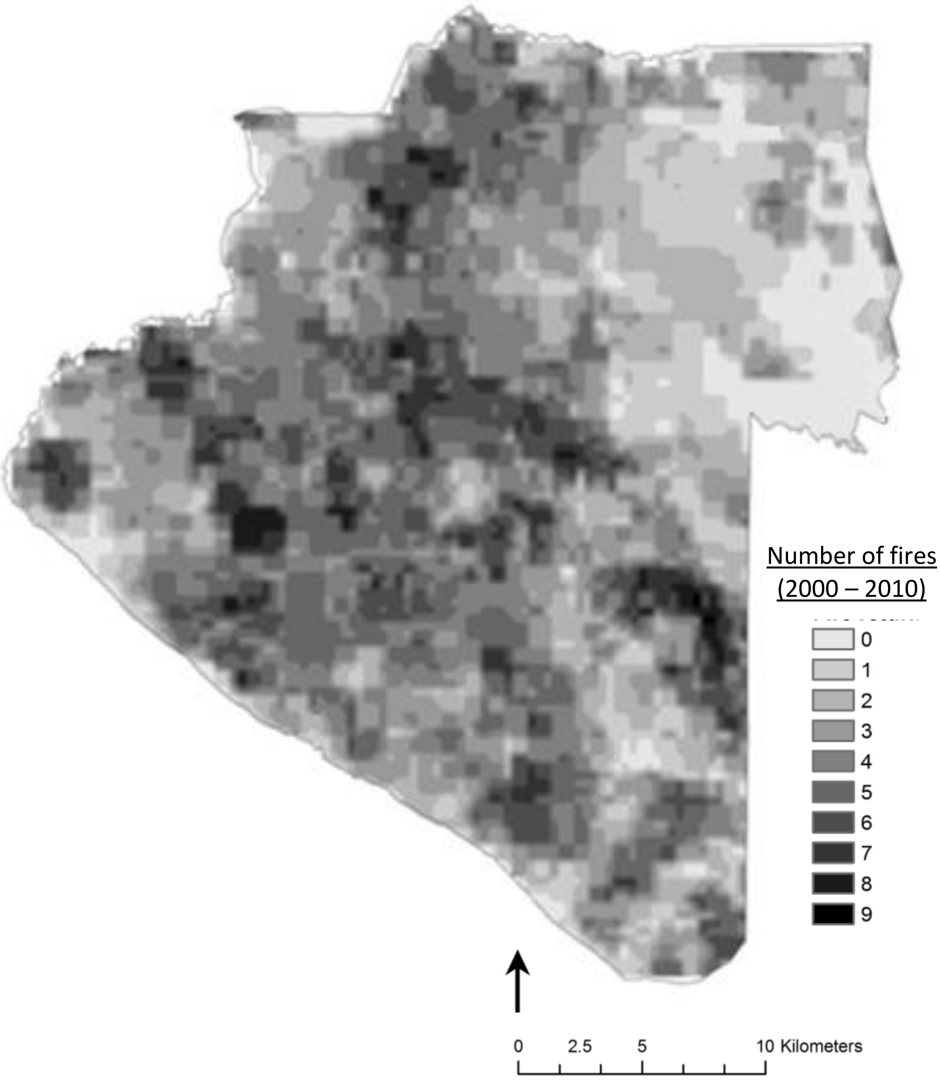

Figure 3.6. The map shows the number of fires in Majete Wildlife Reserve between 2000 and 2010. Darker areas indicate a higher occurrence of fire

Topography

Some of the topographically descriptive variables, namely slope, elevation and TPI (ridge or valley indicator) values of new non-woody points and of ‘all points’ in the different regions of Majete were assessed for correlation using PCAs. Aspect was not included as the values are directionally indexed based on compass degrees (North-facing slopes are indicated by values closer to 0˚ and closer to 360˚).

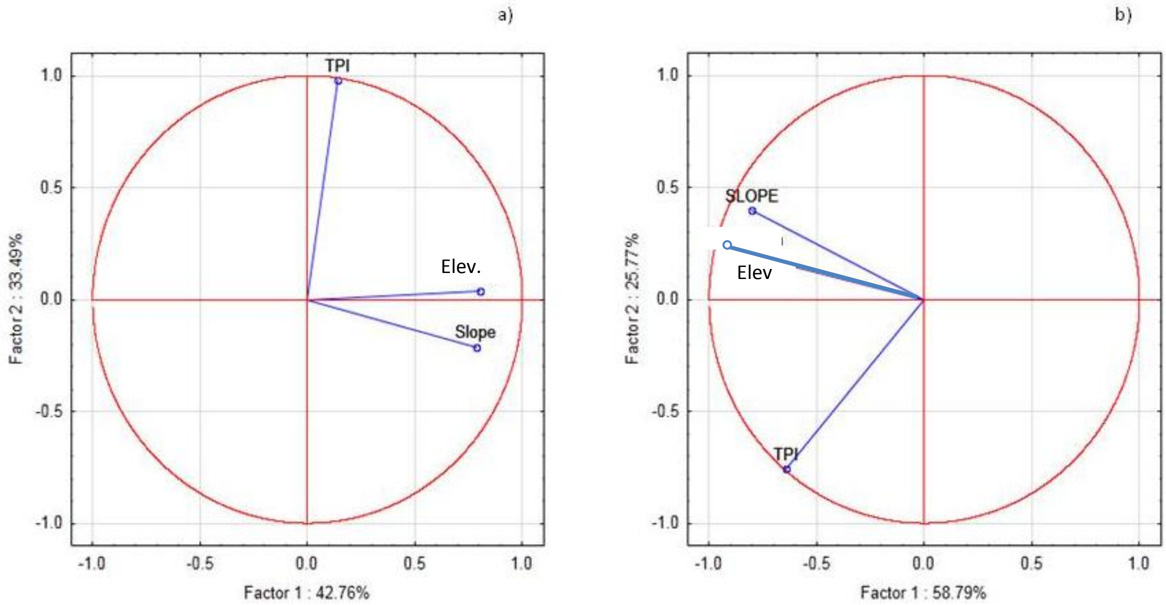

Figure 3.7. a) & b): PCA biplot of the correlations between TPI, slope and elevation values of a) all points and b) new non-woody points in the Sanctuary area of Majete. Point data were extracted from a 2010 woody cover dataset of Majete Wildlife Reserve.

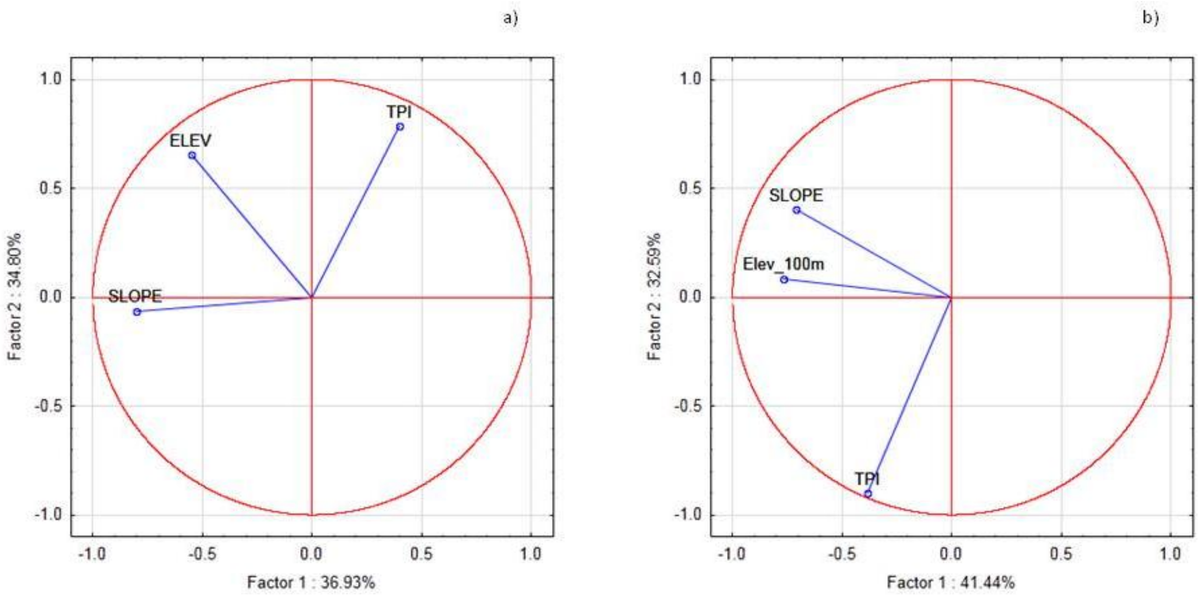

Figure 3.8. a) & b): PCA biplot of the correlations between TPI, slope and elevation values of a) all points and b) new non-woody points in the Eastern lowland area of Majete. Point data were extracted from a 2010 woody cover dataset of Majete Wildlife Reserve.

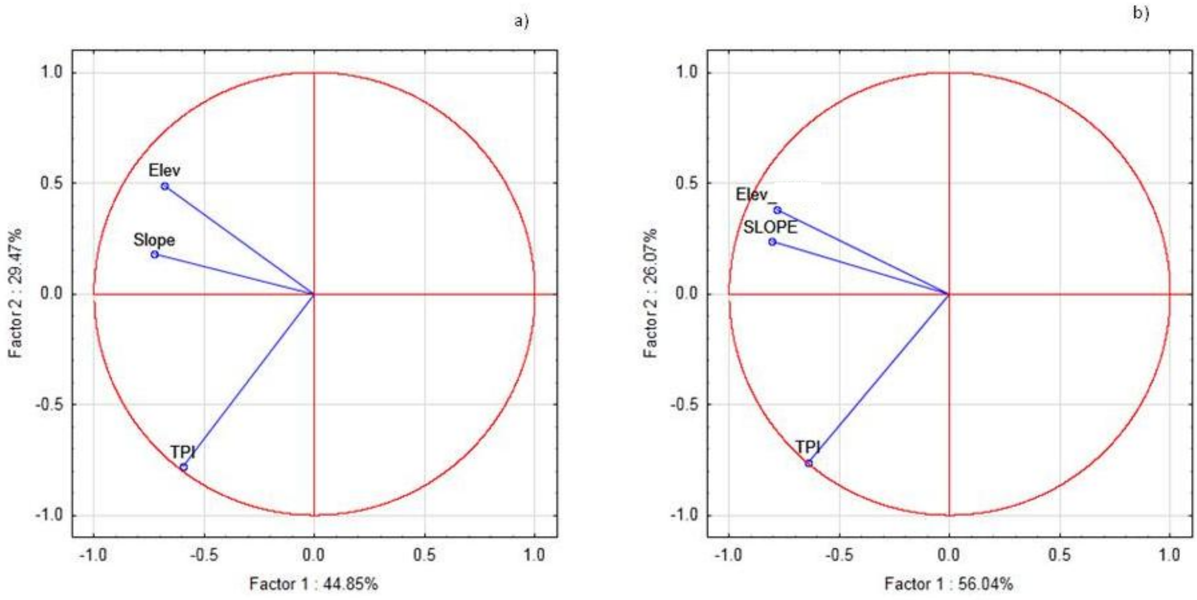

Figure 3.9. a) & b): PCA biplot of the correlations between TPI, slope and elevation values of a) all points and b) new non-woody points in the Western highland area of Majete. Point data were extracted from a 2010 woody cover dataset of Majete Wildlife Reserve.

On the PCA biplots the unit circle provides a visual representation of how well each variable is represented by the current set of Factors (1 and 2), i.e. the closer the variable to the circle margin, the better it is represented by the factor coordinate system which was used. The Factors used in creating the biplots are also listed on the x and y-axes of the graphs.

For the Sanctuary, the Factor 1 and 2 represented 76% of the variance in the data (42.8% and 33.5% respectively) in the PCA for all points in the Sanctuary (Figure 3.7. a). Factor 1 and 2 of the PCA plot for new non-woody points in the Sanctuary represented 84.6% (58.8% and 25.8% respectively) of the variance in the data (Figure 3.7. b). New non-woody point variables showed the same correlations as for all points in the Sanctuary. Elevation and slope were positively correlated.

For the Eastern lowlands, Factors 1 and 2 in the PCA represented 71.7% of the variance in the data (36.9% and 34.8% respectively) for all points (Figure 3.8. a), and represented 74% of the variance in the data (41.4% and 32.6% respectively) for new non-woody points (Figure 3.8. b). In the Eastern lowlands, slope and elevation were more strongly positively related for new non-woody points than for points in general in this region. This indicated that at higher altitudes, new non-woody points were more likely to be found on steeper slopes than what was normal for the area.

For the Western highlands, Factors 1 and 2 in the PCA represented 74.3% of the variance in the data (44.9% and 29.5% respectively) for all points (Figure 3.9. a), and represented 72% of the variance in the data (56% and 26% respectively) for new non-woody points (Figure 3.9. b). In this region, new non-woody point variables were associated in a similar manner to the variables of all points in this region. The association between elevation and slope was, however, stronger for new non-woody points. This also indicated (to a lesser extent than for the Eastern lowlands) that at higher altitudes, new non-woody points were more likely to be found on steeper slopes than what was normal for the area. In all the PCAs for all regions, TPI was not associated with elevation or slope. Therefore, a point could have been on either a concave or convex slope at any different altitude.

New non-woody points were distributed at a range of altitudes in Majete (Figure 3.10.). In the Sanctuary, 82% of the total area was below 300m in altitude, 83% of the Eastern lowland region was between 200-300m, while 78% of the Western highland region fell between 300 and 500m in altitude.

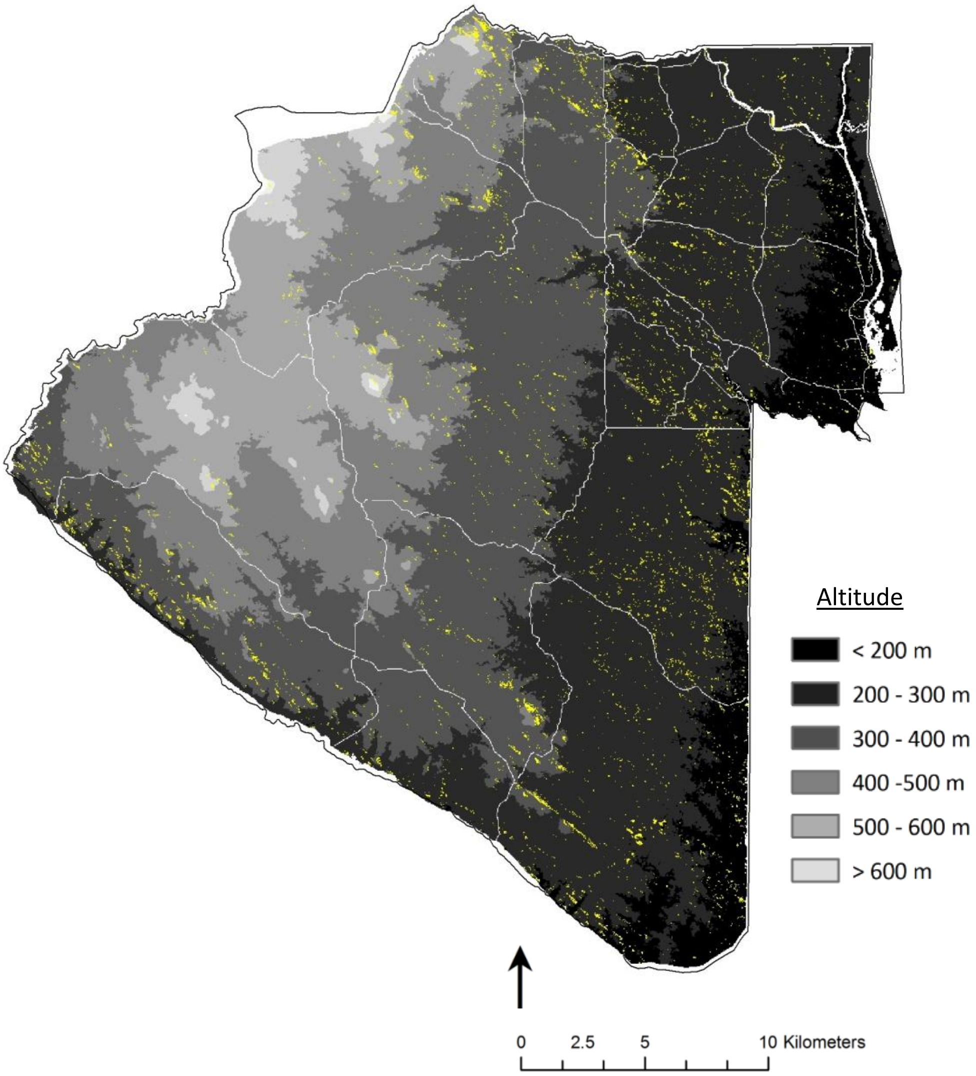

Figure 3.10. The map illustrates the distribution of new non-woody points (points of woody cover loss) across different altitudes in Majete Wildlife Reserve. New non-woody points are shown in yellow. White areas are features which have been masked from the analysis (e.g. fences, rivers and roads).

In the Sanctuary, new non-woody point cover did not differ much between the different altitudinal classes (Table 3.5.). In the Sanctuary, new non-woody cover was highest in the lower altitude classes (under 500m). In the Eastern lowlands, new non-woody point cover was similar between the altitude classes. For the Western highlands, a relatively larger proportion of new non-woody points were found in between 300-400m and also between 700-800m.

Table 3.5.: The altitudinal distribution of different regions in Majete Wildlife Reserve and the percentage of new non woody points in the region found in different altitude classes

	Altitude	% of Region	% cover of new non-woody points
Sanctuary	<200	16	1.0
	200-300	66	1.4
	300-400	10	2.0
	400-500	3	1.6
	500-600	5	0.4
	600-700	0.2	0.9
Eastern lowlands	<200	17	2.0
	200-300	83	1.8
Western highlands	300-400	47	1.4
	400-500	31	0.9
	500-600	19	0.8
	600-700	3	0.7
	700-800	0.1	2.4

Analyses also showed that on steeper slopes the percentage of new non-woody point cover (calculated as a percentage of the number of points within each class) was generally higher for all regions (Table 3.6.). In the Western Highlands particularly, new non-woody point cover was much higher on steeper slopes (between 30˚ and 50˚).

Table 3.6.: The distribution of different regions in Majete Wildlife Reserve on differently angled slopes, and the percentage of new non woody points in the region found on different slopes. A higher slope angle indicates a steeper slope.

	Slope	% of All points	% Cover of New non-woody points
Sanctuary	0-10	93.9	1.3
	10-20˚	5.8	1.7
	20-30˚	0.3	7.2
	30-40˚	0.02	5.7
Eastern lowlands	0-10	89	1.7
	10-20˚	10.5	2.5
	20-30˚	0.5	4.3
	30-40˚	0.02	0.00
Western highlands	0-10˚	75.54	0.7
	10-20˚	21.57	1.8
	20-30˚	2.59	5.8
	30-40˚	0.29	11.2
	40-50˚	0.01	13.3

New non-woody points in the different regions of Majete were also distributed differently in relation to aspect (North-facing) (Figure 3.11). The percentage of new non woody points in each aspect class was calculated as a percentage of the total new non-woody points (in each region), rather than a percentage of new non-woody cover in each aspect class. This allowed for a visual comparison against the percentage of the region which fell into the relative aspect class. New non-woody points were mostly found on North and North-East facing slopes, particularly in the Western highlands. Of all points (cover) in the Western highlands region 11% of slopes were North-facing, 13% were North-East facing and 14% were East-facing. However, 16% of new non-woody points were North-facing, 40% of new non-woody points were North-East facing and 28% were East-facing. This trend was similar for new non-woody points in the other regions of Majete, although new non-woody point distribution between aspect classes was more even in the

Sanctuary and Eastern lowlands.

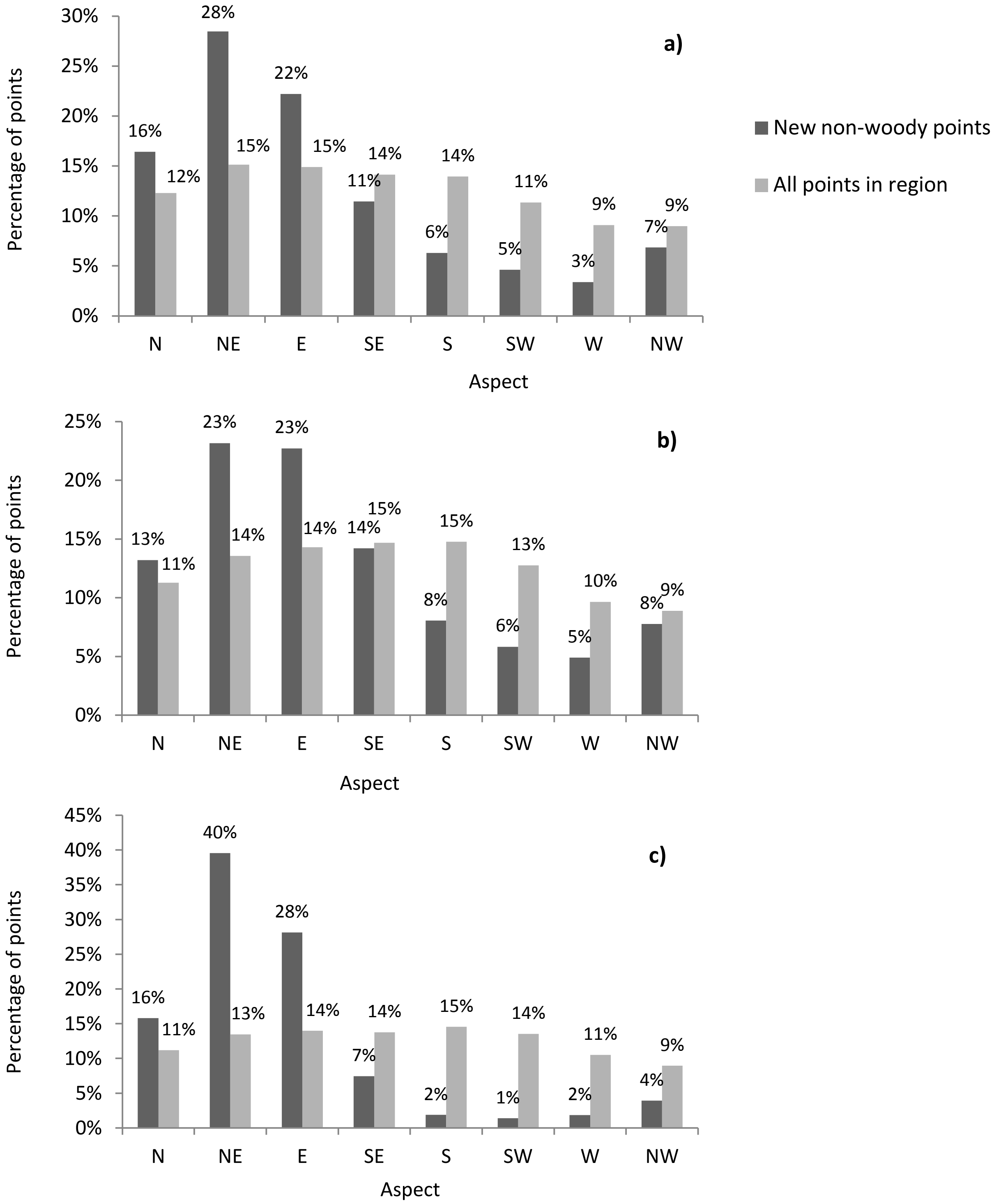

Figure 3.11.: The percentage of all points (light grey) and new non-woody points (dark grey) on slopes of different aspects (North-facing) in the a) Sanctuary, b) Eastern lowlands and c) Western highland region of Majete Wildlife Reserve.

Results of the point alignment analysis showed that new non-woody point clusters were aligned differently in regions of Majete. The percentage of lines (intersecting new non-woody point clusters) in each geographic orientation class is presented for each region. In the Sanctuary, new non-woody points were not linearly distributed, i.e. lines drawn between new non-woody points were distributed evenly between geographic orientation classes (Table 3.6). Conversely, in both the Eastern lowlands and Western highlands, a high percentage (45% in both) of lines was South-east/North-west oriented (Table 3.6.).

Table 3.6.: Results of the point alignment analysis used in this study to determine whether points of woody cover loss (new non-woody points) were linearly distributed, and the geographic orientation of their distribution

	Percentage of lines between clustered new non-woody points		
Geographic orientation of lines	**Sanctuary**	**Eastern lowland**	**Western highland**
North /South	24%	14%	17%
North-east /South-west	21%	13%	18%
East/West	24%	27%	21%
South-east/North-west	31%	45%	45%

3.5. Discussion

3.5.1. Factors affecting woody cover change

In Majete, it is apparent that a loss of woody cover occurred between 2000 and 2010, differing considerably from the consistency in woody cover detected between 1985 and 2000. Although the overall increase in non-woody point cover from 1.4% to 3.1% as a proportion of the entire reserve does not seem high, it must be considered that this increase means that non-woody point cover has doubled between 2000 and 2010. The main factors which influence woody vegetation change in Miombo woodland vegetation types are fires, herbivory and rainfall (Lawton, 1978; Ribeiro et al., 2008; Mapaure & Moe, 2009). The increase in open areas in Majete indicates that pressure from one or a combination of these drivers has increased. Interestingly, the highest increase in new non-woody points occurred in the Eastern lowlands, followed by the Western highlands and least change occurred in the Sanctuary (however, even in this area non-woody point cover doubled).

The 1990 woody cover dataset was developed using imagery from years at the start of a long dry period in the rainfall history of the area, which occurred from the early to mid 1990s. It is likely that changes in woody cover may not have been detected so early on in this dry period due to the lag response of woody vegetation to the lack of rainfall (Desanker & Prentice, 1994). In addition to this, the 2000 dataset was developed after a long period of above average rainfall and woody vegetation cover may have been high at the time (Higgins et al., 2000; Sankaran et al., 2005). However, the two dry spells between 2000 and 2010 (especially the one which fell immediately before the 2010 dataset) may have had some influence on woody vegetation cover in the 2010 dataset, and perhaps explain some of the increase in non-woody point cover in the 2010 dataset. Rather than directly affecting woody vegetation density, Desanker and Prentice (1994) found that severe soil moisture stress inhibits seedling growth in Miombo and restricts adult plant growth. Sankaran et al. (2005) modelled the effects of rainfall, nutrients and disturbance (fire and herbivory) on African savannas and found that where average annual precipitation exceeded 630mm, savannas were “disturbance-driven” rather than “climatically-driven”. Vegetation in Majete falls into the disturbance-driven category (as the expected rainfall range is above 630mm), in which woody cover density is mainly affected by fire and herbivory. Therefore, the impact of the two dry periods between 2000 and 2010 may not have directly led to much loss in woody cover, but would have compounded the effects of fire and herbivory on woody vegetation. Vegetation recovery in the wet season, after experiencing fire and herbivory during the dry season, would then have been limited by successive below average rainfall wet seasons (see Desanker & Prentice, 1994; Higgins et al., 2000; Sankaran et al., 2005, 2008).

Fire data were unavailable for Majete pre-2000 and there is no way of knowing whether fire frequency may now be higher in Majete than in the past. There was, however, a 22% increase in the human population surrounding the reserve between 1998 and 2008 (NSOMalawi, 2008). Rural communities around Majete traditionally burn indigenous vegetation annually to prepare fields for planting, improve visibility for walking or to produce a green grass flush for grazing cattle on (Mzumara, 2009). Despite current efforts to control wildfires in Majete, fires started outside the reserve often burn across Majete’s boundary fence. In addition to this, poachers hunting inside the reserve tend to use fire to flush out game (reserve manager, pers. com.). The increase in the surrounding human population may have caused an increase in fire occurrence in the reserve.

3.5.2. Area specific changes

Sanctuary

In the Sanctuary, points of new non-woody cover had experienced very few fires and it is clear from Figure 3.6. that the Sanctuary was seldom burnt between 2000 and 2010 in comparison to the rest of Majete. As wildlife was reintroduced to the Sanctuary first, the area has been actively protected from fires since 2003 to protect foraging areas for reintroduced wildlife (reserve manager, pers. com.). Woody cover loss in the Sanctuary is probably due, therefore, to the interacting effects of low rainfall and herbivory (Sankaran et al., 2005, Sankaran et al., 2008). Wildlife was reintroduced to the Sanctuary beginning in 2003 and only reintroduced to the outer areas of the reserve beginning in 2008 (Appendix Two). Wildlife has, therefore, been present in the Sanctuary for longer than in other areas and wildlife was contained to this region by the Sanctuary fence. The elephant population density in the Sanctuary was higher for longer than in the rest of Majete (approximately 0.5 elephants per km^2, reintroduced in 2006 and 0.26 elephants per km^2, reintroduced in 2008 respectively) (reserve manager, pers. com.). Although elephants are a keystone species and generally have a greater impact on woody vegetation than other herbivorous species (Laws, 1970; Cumming et al., 1997; Hayward & Zawadzka, 2010), the results of this study in no way allow woody cover loss to be attributed to elephants or to any other herbivorous species without field validation. Impala, for example, can inhibit sapling recruitment and trampling effects by herds of buffalo can result in loss of woody plants (Mosugelo et al., 2002; Augustine & McNaughton, 2010; Midgley et al., 2010). However, it is very difficult to attribute vegetation changes to particular herbivores, even if high resolution imagery is used.

New non-woody points in the Sanctuary were not distributed at any particular distance to perennial water either. 87% of the Sanctuary is found within 4km of perennial water and it is probable that water availability is not a limiting factor to herbivore foraging (Redfern et al., 2005). The high availability of water in this area would allow a wide distribution of herbivores even at the height of the dry season. Therefore, if new non-woody points have been influenced by herbivory, they would not necessarily be distributed close to perennial water (Gaylard et al., 2003; Chamaillé-Jammes et al., 2009; Landman et al., 2012).

In the Sanctuary, new non-woody points were distributed at different altitudes but in proportion to the portion of the Sanctuary which fell into the different classes. For example, although 82% of new non-woody points in the Sanctuary were under 300m in elevation, 87% of the available area in the Sanctuary was under 300m. Slope and altitude were positively correlated for this region (for all points and for new non-woody points); therefore, new non-woody points in the lower altitude areas of the Sanctuary would not have been positioned on very steep slopes. Interestingly though, non-woody points in the Sanctuary did have a tendency to be situated on North and North-East facing slopes. These slopes are characteristically drier than slopes facing other directions, due to the longer exposure to sunlight (Huntly, 1982). Coupled with drought conditions between 2000 and 2010, woody vegetation on dry Northward-facing slopes could have been more vulnerable to the impacts of herbivory than Southward-facing slopes, for example.

Eastern lowlands and Western highlands

There were many similarities between the distribution of new non-woody points in the Eastern lowlands and Western highlands. In both regions, new non-woody points had been burnt more often than in the Sanctuary and it is evident that overall, fire frequency in the regions outside of the Sanctuary was higher (Figure 3.6.). Additionally, herbivore reintroductions to the areas outside of the Sanctuary only began in 2008 - much later than to the Sanctuary itself (see Appendix Two). Images for the 2010 dataset were taken from 2009, 2010 and 2011 and it is unlikely that much impact from herbivory would have been detected in the 2010 dataset, as such a short time span had elapsed since wildlife reintroduction in this area. New non-woody points were also distributed evenly in the different proximity to perennial water classes in both regions.

New non-woody points were also distributed at altitudes in proportion to the available land at these altitudes in the both regions. For both regions, slope and elevation were more strongly positively correlated for new non-woody points that what was normal for the respective regions. This indicated that if new non-woody points were at higher altitudes, they were also more likely to be found on steeper slopes. Aspect again appeared to play a significant role in determining new non-woody point location. New non-woody points in the Western highland were distributed disproportionately more on Northward-facing slopes in comparison to what was normal for this area. In the Western region, slope data also suggested that new non-woody cover was much higher on steeper angled slopes than it was on more gentle slopes. New non-woody points in the Eastern lowlands also tended to be distributed on Northward-facing slopes and new non-woody point cover was slightly higher on steeper slopes; but neither of these trends was as strong as in the Western highland. In addition, in both regions, a high percentage of the new non-woody point clusters were distributed linearly.

Woody cover loss in these regions is likely due to the impacts of a combination of frequent fires and drought (Midgley et al., 2010). This was indicated by the linear distribution of new non-woody points on mainly Northward-facing slopes, and the high fire frequency experienced by these points. Such sites would be characterized by thin, well-drained soils and would be exposed to sunlight for longer than South-facing slopes - consequently vegetation is sparser on slopes such as these in Majete (Huntly, 1982; Sherry, 1989). Sparse vegetation communities on this type of slope would predispose these areas to woody cover loss by fire, particularly as detected in the woody cover datasets. Many of the dominant plant species in Majete's high altitude areas have also been shown to be sensitive to fires. Results from long-term burning experiments in Zambia showed that frequent fires in *Brachystegia*- and *Julbernadia*-dominated vegetation types can result in the transformation of vegetation towards a more open, shorter, fire-hardy, shade-intolerant community of woody plants (Trapnell, 1959; Lawton, 1978; Mapaure & Moe, 2009). Low soil moisture availability during drought periods would have limited woody vegetation recovery after fire (Sankaran et al., 2005) on the slopes where new non-woody points developed.

3.5.3. Management implications

Considering the change which occurred in woody vegetation cover between 2000 and 2010 it is vital that vegetation condition is monitored regularly in Majete (in exclusion plots within specific vegetation communities, for example) so that rapidly changing areas can be detected. The results of this study indicate that fire and herbivory have had varying levels of impact on woody vegetation in different areas of Majete. Specific management interventions therefore, may be needed for each area to protect woody vegetation from degradation.

In the Sanctuary, less woody cover loss occurred than in the other regions of Majete, however, data analyses showed that woody cover loss was likely as a result of the combined effects of herbivory and below average rainfall between 2000 and 2010. Wildlife reintroduction to Majete requires that their impacts on woody vegetation are monitored as vegetation in fenced reserves is vulnerable to overuse (Ben-Shahar, 1993; Duffy et al., 2002; Shrader et al., 2010). During drought periods in particular, the impact of herbivores on vegetation needs to be monitored, as recovery from dry season herbivory may be hampered if soil moisture remains low in below average rainfall wet seasons. With the removal of the Sanctuary fence, many of the herbivores contained in the area may have dispersed into other regions of Majete. Threshold levels of acceptable herbivore impacts on woody vegetation will need to be defined for the reserve so that woody vegetation changes can be more easily monitored and degradation prevented.

New non-woody points were not distributed closer to perennial water points in the Sanctuary and most of the Sanctuary is in close proximity to perennial water. When surface water becomes scarce in a landscape, i.e. in the dry season, herbivore foraging is concentrated around remaining perennial water points (Redfern et al., 2003). If surface water is supplemented (using AWPs) then herbivore foraging activities are less restricted by water availability (Smit et al., 2007; Smit & Grant, 2009; Young et al., 2009). If herbivory was a main driver of the increase in new non-woody points in this area, then herbivory is not restricted by water availability in the dry season in this region. Browsing levels may be homogenized across the Sanctuary which could result in widespread woody cover changes and loss of vegetation heterogeneity in the Sanctuary (Rogers, 2003; Chamaillé-Jammes et al., 2009; Farmer, 2010). Closure of at least one of the AWPs in the Sanctuary may alleviate extensive browsing pressure by herbivores. However, the AWPs are an important feature in the Sanctuary because they provide good game-viewing opportunities for tourists. Before such changes are considered, further data on the condition of the woody vegetation in the Sanctuary in relation to distance to perennial water is needed.

Although the Sanctuary has experienced a widespread decrease in woody vegetation cover, mega-herbivores have been absent from the reserve (or present at low densities) for over 20 years. Vegetation may simply be returning to a state similar to what might have existed in the area when elephants and other browsing species were present at higher densities in Majete (see Cormack, 1992; Mosugelo et al., 2002; Skarpe et al., 2004). However, as Majete has been fenced and the resident mega-herbivore populations are increasing (reserve manager, pers. com.), it remains important for regular monitoring to be undertaken of the vegetation dynamics in different areas of Majete. This will allow the early detection of woody vegetation changes so that appropriate management action can be taken, particularly in drought situations.

The impact of fire and periods of below average rainfall on woody vegetation cover in the two outer regions was apparent from the analyses. Northward-facing slopes in particular appear to be vulnerable to woody vegetation loss. The Eastern lowlands and Western highlands had both been exposed to frequent fires between 2000 and 2010, despite reserve management efforts to control wildfires. Control of wildfires is exceptionally difficult in the hilly regions of Majete, because fires burn more hotly up slopes and the terrain makes mobility of fire-fighting teams difficult. Additional fire breaks should be created in some of these areas to reduce the extent of wildfires and to allow specific areas to be targeted for controlled burns. New firebreaks may make fire management strategies easier to implement in this difficult terrain. The impacts of herbivory are likely to be minimal in the high altitude areas of Majete as the terrain is steeply hilly and rocky (see Nellemann et al., 2002; de Knegt et al., 2011) and the *Brachystegia-Julbernadia* dominated vegetation communities of this area are of limited foraging quality (Sherry, 1989).

3.6. Conclusions

The comparison of woody cover datasets from 1985, 1990, 2000 and 2010 for Majete showed that woody vegetation cover remained similar between 1985 and 2000, but that the reserve experienced higher woody vegetation cover loss between the 2000 and 2010 datasets. Two below average rainfall periods occurred between 2000 and 2010, one of which occurred immediately prior to the 2010 dataset.

In the Sanctuary, woody cover loss was not associated with a high fire frequency as the area had been protected from fires since 2003. Woody cover loss appeared to have been driven by the synergistic effects of drought and herbivory in the Sanctuary. Most of the Sanctuary, and correspondingly most new non-woody points, were situated at lower altitudes. New non-woody point elevation and slope were positively correlated, and new non-woody points were mainly found on Northward-facing slopes. Points of woody cover loss were not associated with proximity to perennial water and were not clustered. If herbivory was a key driver of woody cover change in this area, herbivore foraging is, therefore, not limited by water availability in the area. Reducing water availability in this region by removing at least one of the AWPs may help to limit the extent of herbivory; however, additional data on vegetation condition is required before such changes can be considered. With the reintroduction of herbivores to Majete, it may also be possible that vegetation is simply returning to a state similar to what it was before the indigenous herbivore populations were exterminated.

In the Eastern lowlands and Western highlands, new non-woody points experienced a higher fire frequency than those in the Sanctuary. They were also not associated with proximity to perennial water. New non-woody points were distributed at altitudes in proportion to the amount of area situated at different altitudes, and their slope and elevation attributes showed a stronger positive association than what was normal for the area. New non-woody points were clustered and linearly distributed along the South-East line, and were mainly situated on Northward-facing slopes. From the data analyses, it appears that the synergistic effects of fire and continuous periods of below average rainfall have driven the development of most new non-woody points. Woody vegetation may continue to be degraded in these areas if exposed to such regular fires. Additional fire breaks should be created in the hilly, high altitude regions of Majete to facilitate early burning strategies and wildfire control.

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Chapter Four

Elephant water point usage in a fenced reserve: a case study from Majete Wildlife Reserve, Malawi

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4.1. Abstract

Elephants, as a keystone species, have a significant impact on woody vegetation in savanna systems, particularly when confined to fenced reserves. However, elephant distribution and their impacts on vegetation are restricted by surface water availability. The construction of artificial water points (AWPs) in reserves can alter the distribution of elephant impact on woody vegetation. Majete Wildlife Reserve (Malawi) is currently being developed, and as part of this process elephants were reintroduced, the reserve was fenced and AWPs were constructed. Understanding elephant usage of and impacts around perennial water points in Majete will provide critical information upon which future management decisions on AWP positioning or closure can be based. A greater understanding of factors affecting elephant water point usage will also aid managers of other reserves in designing more effective water management strategies for controlling elephant impacts on vegetation. The objectives of this study were to determine whether elephants used water points differentially in the reserve, and which factors affect elephant water point selection. Elephant usage (including visits to water points, browsing levels and path use around water points) of selected perennial water points in Majete was monitored in the wet and dry season. The effects of season, water point characteristics (type, size and water quality) and habitat context (surrounding vegetation type, elevation and proximity to other water points) on elephant water point use were then tested. Elephant water point use was affected by season, as well as water point altitude and

surrounding vegetation type. The implications of the study findings for elephant and water point management were then discussed.

4.2. Introduction

Fences affect ecological processes within reserves in numerous ways, such as by severing wildlife dispersal routes and limiting food, water and space availability, which creates artificially closed systems or ecological islands (Boone & Thompson Hobbs, 2004; van Aarde & Jackson, 2007; Whyte & Joubert, 2010). As the largest terrestrial herbivore, the African savanna elephant (*Loxodonta africana*) plays a key role in savanna systems by altering woody vegetation cover through its feeding habits and large forage requirements (Laws, 1970; Cumming et al., 1997; Hayward & Zawadzka, 2010). In fenced reserves elephants are unable to shift ranges seasonally as they do in natural systems, so vegetation is exposed to year-round foraging pressure (Duffy et al., 2002; van Aarde & Jackson, 2007; Loarie et al., 2009a). Elephant distribution and their impacts on vegetation within a landscape, however, are not random (Harris et al., 2008; Loarie et al., 2009b; Young et al., 2009). They are primarily affected at coarse-scales by rainfall and vegetation productivity (de Knegt et al. 2011), and at finer-scales by surface water availability, terrain and landscape heterogeneity (Grainger et al. 2005; Chamaillé-Jammes et al. 2008; de Beer & van Aarde 2008). Elephants are water-dependent, therefore their distribution is strongly constrained by surface water availability in the dry season (Redfern et al. 2003; Smit et al. 2007a) and their ranging patterns are limited to areas within a daily commutable distance to water (Stokke & du Toit, 2002; Leggett, 2006). Elephants predominantly browse in the dry season as grasses and forbs are moribund (Bax & Sheldrick, 1963; Osborn, 2004), therefore browsing pressure by elephants is greatest on woody species closer to water during this time of year (Ben-Shahar, 1993; Chamaillé-Jammes et al., 2009; Gaugris & van Rooyen, 2009). The concentration of elephants around water in the dry season thus contributes to the development of piospheres.

Piospheres consist of a ‘sacrifice area’ devoid of vegetation immediately adjacent to a waterhole, followed by increasing vegetation quality with increasing distance from water. Piospheres are a function of herbivore water and forage needs (Lange, 1969; Thrash & Derry, 1999). Vegetation further away from water points is protected from elephant browsing in a range of partial to absolute spatial refuges (Ben-Shahar, 1993; O'Connor et al., 2007). African savanna systems contain herbivores from a variety of feeding guilds, so piosphere effects are expressed in both herbaceous and woody vegetation, the latter of which is particularly affected by elephants (Thrash & Derry, 1999). Piosphere size and shape is affected by the distance to which elephants will forage away from water in the dry season (dependent on individual metabolic requirements), the quality of forage surrounding a water point and levels of intraspecific competition (Stokke & du Toit, 2002; Leggett, 2006; Young et al., 2009). With the onset of the wet season elephants are again able to move away from their water-restricted dry season ranges (Lindeque & Lindeque, 1991; Verlinden & Gavor, 1998; Ngene et al., 2009), allowing woody vegetation around perennial water to recover. Water point placement within fenced reserves can therefore determine elephant distribution, local density and impacts on woody vegetation.

Artificial water points (AWPs) have frequently been used to stabilise fluctuations in natural water availability in reserves (such as Hwange National Park in Zimbabwe, and Kruger National Park in South Africa) so that wildlife densities can be maintained in arid areas throughout the dry season (Chamaillé-Jammes et al., 2007a). This may be to provide tourists with good wildlife viewing opportunities (Omphile & Powell, 2002; Shannon et al., 2010), to increase herbivore foraging range size, or to facilitate faster population growth of water-dependent species (Redfern et al., 2005; van Aarde & Jackson, 2007; Loarie et al., 2009a). However, the impacts of increasing surface water availability on woody vegetation are complex and potentially detrimental in fenced areas supporting high numbers of elephants (Owen-Smith, 1996; Gaylard et al., 2003; Owen-Smith et al., 2006). Where surface water has been supplemented, elephant presence can be sustained in previously drier areas throughout the year (Leggett, 2006), which can lead to vegetation degradation through persistent browsing (Loarie et al., 2009a; Landman et al., 2012). When water is abundant, elephant home ranges tend to be smaller (Grainger et al., 2005; Young et al., 2009) and more intensively used because travel distances between key resources are minimised (Grainger et al., 2005; de Beer & van Aarde, 2008; Thomas et al., 2011). Intensive use of home ranges creates areas of high impact on vegetation (Shannon et al., 2006; de Beer & van Aarde, 2008; Birkett et al., 2012) which can further cause woody vegetation loss (Chamaillé-Jammes et al., 2008; Hayward & Zawadzka, 2010). Although home ranges might be smaller, elephant landscape-scale distribution becomes wider as foraging opportunities increase with increased water availability (Thouless, 1995; Dolmia et al., 2007). This spreads elephant effects on vegetation across a landscape (Loarie et al., 2009a).

In natural systems, seasonal water shortages protect vegetation from elephant browsing through the subsequent development of plant spatial refuges in areas far from water (O'Connor et al., 2007; Gaugris & van Rooyen, 2009). Where water is supplemented, the spread of elephant browsing through increased water availability can produce a more homogenous level of elephant impact across a landscape, compromising ecosystem spatial heterogeneity (Gaylard et al., 2003; Redfern et al., 2005; Chamaillé-Jammes et al., 2009). Ecosystem heterogeneity affects most ecological processes and has significant consequences for biodiversity and ecosystem resilience (Rogers, 2003; Farmer, 2010). Heterogeneity in the distribution and intensity of elephant browsing can, therefore, contribute positively to ecosystem heterogeneity (Rogers, 2003; Whyte et al., 2003). Perennial water point positioning and availability in fenced reserves affects the extent and level of elephant impacts on vegetation, as well as ecosystem heterogeneity. Elephant dispersal and migration is prevented in fenced reserves, and so browsing is more persistent (Owen-Smith et al., 2006; van Aarde et al., 2006; van Aarde & Jackson, 2007). Hence, the impacts of AWPs on elephant use of vegetation are more pronounced in smaller fenced reserves supporting elephants, where vegetation and space resources are limited (Duffy et al., 2002).

Majete Wildlife Reserve (Majete), in southern Malawi, is a small fenced reserve without room for expansion and with limited access to surface water for wildlife. Majete is undergoing transformation, which has included the re-introduction of over 200 elephants and the construction of AWPs. In a previous study (Chapter Three), it was found that woody vegetation cover in Majete had decreased between 2000 and 2010. Data analyses suggested that woody cover loss could be attributed to herbivory and drought (due to the low incidence of fire) in the previously separately fenced 'Sanctuary area' in the North-East of Majete. Most of this area (87%) was found to be within 4km of perennial water. Points of woody cover loss were not associated with distance to perennial water but were widespread throughout the area. These findings suggested that browsing by herbivores may not be limited by water availability in this region. It was then suggested that some of the AWPs in the area may need to be closed in order to reduce water availability in the area and, by doing so, limit the extent of herbivory. As elephants are a keystone species and are a major driver of change in woody vegetation (Sankaran et al., 2008; Hayward & Zawadzka, 2010), their reintroduction to Majete may have contributed to woody cover loss in the Sanctuary area.

The distribution and intensity of elephant impacts on vegetation are affected by perennial water distribution and availability. Understanding elephant usage of and impacts around perennial water points in Majete will provide critical information upon which future management decisions on AWP positioning or closure can be based. A greater understanding of factors affecting elephant water point usage will also aid managers of other reserves in designing more effective water management strategies for controlling elephant impacts on vegetation. The objectives of this study were to determine which perennial water points in Majete were being used most by elephants and to determine which factors may have affected this. Data was collected and analysed for three different indicators of elephant water point use: the number of visits to water points, as well as browsing levels and intensity of path use around water points. It was hypothesized that different water point types (rivers, AWPs and springs) would be used at different intensities by elephants, and that perennial rivers would experience most use. Through this research greater insight is provided into elephant water point usage in Majete to inform future water placement or closure, which will also aid in preserving ecosystem spatial heterogeneity in the reserve.

4.3. Methods

4.3.1. Study area

Majete, situated in the lower Shire valley in southern Malawi, lies along the southern reaches of the Great Rift Valley. In the western section the terrain consists of steeply undulating hills dissected by river valleys. Towards the east, closer to the Shire River, slopes become more gentle and flat. Two perennial rivers transect the reserve, the Shire River and Mkurumadzi River, although many seasonal streams exist throughout the reserve. There are numerous perennial springs, a hot spring and seasonal pools which contribute to the available surface water in the wet season. The average annual precipitation is between 680-800mm in the eastern low-lands, and 700-1000mm in the western uplands, falling in a distinct wet season commencing in November and ending in early April (Sherry, 1989; Malawian Department of Climate Change and Meteorological Services). No other significant rainfall is received for the rest of the year. Summers are hot with an average daily temperature of 28.4°C and winters slightly cooler with an average daily temperature of 23.3°C.

Vegetation classes in Majete are often merged and difficult to definitively classify, but their distribution is strongly associated with soil type and depth. These include riverine associations (found along river systems, characterized by *Kigelia africana, Philenoptera violacea* and *Euphorbia ingens*); low altitude mixed deciduous woodland (between 205-280m, *Acacia spp., Sclerocarya birrea* and *Sterculia spp.*); ridge-top mixed woodland (on flatter ridge tops and higher ground, 220-300m, *Terminalia sericea, Diospyros kirkii* and *Diplorhynchus condactylcarpon*); medium altitude mixed deciduous woodland (eco-tonal vegetation between low lands and hilly area, 230-410m, *Brachystegia boehmii*, *Pterocarpus rotundifolius, Diospyrus kirkii* and *Combretum spp.*); and high altitude miombo woodland (hillier areas, 410-770m, *Brachystegia boehmii, Julbernardia globiflora*, *Burkea africana, Diplorhynchus condylocarpon* and *Pterocarpus angolensis*) (Sherry, 1989).

No indigenous large-mammal populations remained in the reserve by the mid-1990s and in 2003, African Parks Majete (Pty.) Ltd undertook the rehabilitation and development of Majete. The entire reserve was fenced and the smaller 140km² Sanctuary area (Figure 4.1.) was separately fenced for wildlife re-introduction purposes. Over 2550 individual animals, of 14 species have been reintroduced and seven solar-powered AWPs added to Majete, although at the time of study only six of these were functioning. Wildlife reintroductions included 217 elephants, 70 of which were released in 2006 in the Sanctuary and 147 in the outer section of the reserve between 2008 and 2009. In 2011 the Sanctuary fence was removed allowing elephant movement throughout the reserve. The latest estimate of the elephant population size in Majete is around 270 individuals (reserve manager, pers. com.).

4.3.2. <u>Water point data collection</u>

Water point selection

Few roads exist in Majete and much of the terrain in the reserve is rugged, steep and hilly, particularly in the interior of the reserve. Past research has shown that elephant are generally found in areas with more gentle terrain and that they avoid steep or very hilly areas (Nelleman et al. 2002; de Knegt et al. 2011). Due to these factors, and to the considerable difficulties in obtaining fuel in Malawi during the fieldwork period, water point selection was restricted to three main regions of the reserve; the Sanctuary, the Pende area, and the Diwa hilly interior area (Figure 4.1.). The Sanctuary area towards the north-east has a less hilly terrain and contains both permanent rivers and numerous water points, mainly located in low altitude woodland and riverine vegetation. Elephant density was historically higher in the Sanctuary than the rest of Majete (respectively 0.5 elephants per km^2 and 0.26 elephants per km^2) (reserve manager, pers. com.) before the Sanctuary fence was removed in 2011. The Pende area in the south-east is flat and only contains one major water point (the closest perennial water point is in a hilly area, 4km away) surrounded by low altitude woodland vegetation. The hilly interior of Majete is steeply hilly and rocky, and contains some water points (although difficult to access) in *Brachystegia*-dominated vegetation types (mid altitude and high altitude woodland).

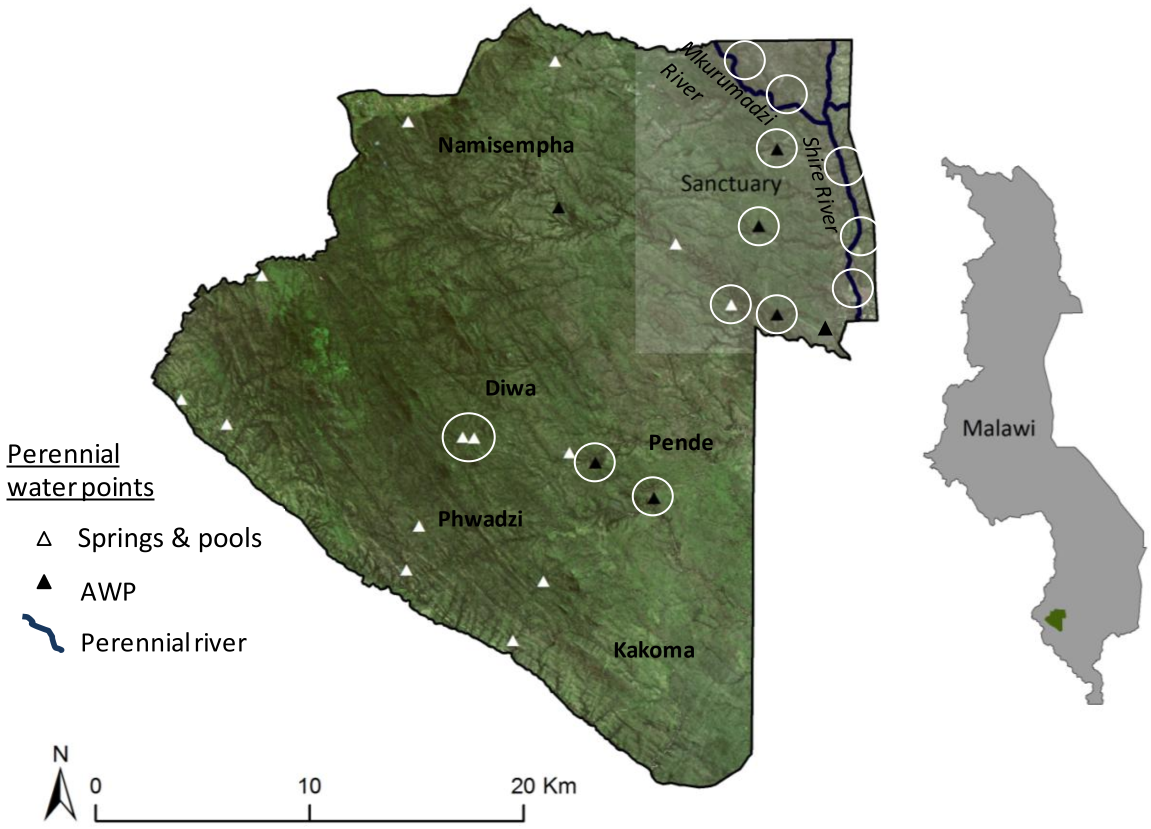

Figure 4.1. A map of the perennial water points in Majete Wildlife Reserve. Majete's position in Malawi is also shown. Water points selected for sampling in this study are encircled in white. These included five artificial water points (AWPs), three springs and five river sites.

Water points could not be systematically selected for this study because water points are not distributed evenly across Majete, and each water point is unique in its position and characteristics. To allow testing of the study hypothesis, that water point type would affect elephant water point usage, a similar number of AWPs, springs and river sites were selected to allow comparisons between types. Only perennial water points were selected. Three springs were accessible for monitoring - one in the Sanctuary and the other two in the southern hilly interior. Five accessible AWPs were selected - one in the Pende area, one in the hilly interior and three in the Sanctuary. Three sites along the Shire River and two on the Mkurumadzi River were chosen after consultation with Majete scouts and managers as to site accessibility and sites at which elephants were regularly observed.

Data collection at selected water points took place during two sampling periods in the wet (January-March) and dry season (July-August) of 2012. Three indicators of elephant activity around water points were measured for each season to provide evidence for usage of water points. Indicators included: i) the average number of elephant visits to water points in a 24 hour period, ii) levels of browsing (or piosphere development) around water points, and iii) intensity of elephant path use around water points. Time constraints prevented long-term data collection, but the sampling of browsing levels and path use around water points provide longer-term evidence of water point use.

Monitoring of visits to water points

Elephant visits to water points were monitored for two sample periods of 10 days in each season. Water points were not monitored consecutively for the sample period, but a minimum of 3 consecutive days of monitoring was stipulated. This monitoring period was shorter than in other studies due to time constraints, but visit data is used as a complimentary data set to browsing and path data. Monitoring of water points was achieved using a sweep and spoor-count method similar to that of Leggett, (2006), Epaphras *et al.* (2008) and Pastorini *et al.* (2010). Radiuses of 40m around water points were swept clean of elephant tracks and dung prior to the start of monitoring. They were then checked every 24 hours for spoor, dung or other signs of elephant presence (i.e. digging, damage to vegetation or path use) and approximate counts of elephants that had visited the sites were taken with the help of trained scouts. On occasions when large numbers of elephants (or other game animals) had visited the site making spoor difficult to read, the area surrounding the 40m radius was also checked for tracks to provide additional evidence of visits, especially along paths frequently used by elephants. After counts had been made the area was swept clean of tracks and dung to ensure that evidence of elephant visits was not counted again the next day. Daily variables recorded at each water point are detailed in Table 4.1. For rivers it was not possible to monitor all the variables. Random observations at water points (lasting between five to six hours) were conducted throughout the monitoring season during peak drinking hours and used to confirm the accuracy of counts. Data on the fixed characteristics of each water point were also recorded. This included surrounding vegetation type, surrounding topography, altitude, distance to nearest perennial water or river; and distance to nearest road or fence.

Table 4.1. Variables recorded daily at perennial water points when monitoring water points for elephant usage at Majete Wildlife Reserve

Variable	Measurements
Approximate number of elephants	Adults and juveniles counted
Water point dimensions (springs and AWPs only)	Max length, width or overflow from main pool (m)
Water movement	Qualitative index: 1-3; 1 flowing strongly, 3 still
Water turbidity	Qualitative index: 1-3; 1 clear, 3 very muddy
Water algae level	Qualitative index: 1-3; 1 least green, 3 very green
Water odour	Qualitative index: 1-3; 1 no smell, 3 strong unpleasant smell
Daily weather conditions (from Majete records)	Min. and max. temperatures and precipitation

Browsing levels and path sampling

Vegetation transects were utilised to determine the intensity of elephant browsing around water points. Transects are commonly used to determine herbivore impacts on vegetation around water points (Brits et al. 2000; Gaugris & van Rooyen 2010) and have been used specifically to quantify elephant impacts on woody vegetation around water (Mukwashi, 2006; Fullman, 2009; Landman et al., 2012). Sampling took place in the wet and dry seasons so that seasonal differences in browsing levels around water points could be detected. Replicate transects were sampled around water points so as to obtain an average measure of browsing levels. Three replicate transects radiating outwards in different directions from every spring or AWP were sampled. For the two perennial rivers, five replicate transects perpendicular to each perennial river were used. Transects were not continuous, but rather consisted of 20m x 20m plots spaced at 500m, 1000m and 1500m intervals from the water point. Elephants mostly graze in the wet season, but prefer to browse during the dry season as woody plant forage is of higher quality (Bax & Sheldrick, 1963; Osborn, 2004). Therefore, an additional plot at 2000m on all transects was sampled in the dry season to give a wider perspective of browsing around water points. In summary, for every discrete water point (AWP and springs) three 20x20m at each specified distance (500m, 1000m, 1500m and 2000m) were sampled around water points. For each river, five plots at each specified distance were sampled.

Plots were placed along elephant paths leading away from water points (see Steyn & Stalmans 2001). Elephants make use of paths between key resource areas; these reflect the population’s long-term feeding and movement patterns (Shannon et al., 2009; von Gerhardt-Weber, 2011). Elephant paths were identified by presence of spoor and dung along the path. The presence of fresh dung piles and path width (Table 4.2.) were used as indicators of path use intensity (see Shannon et al., 2009). Paths leading away from the water point were walked to each 500m interval distance, at which point the plot was positioned at least 30m from the path (side of the path was selected randomly) to avoid sampling trees damaged by animal movement. Paths were graded from 1 to 3; 1 being most used and 3 least used (rest areas were not taken into account). Paths varied in sinuosity so the linear distance of plots to water points was used. Plot location was recorded using a hand-help GPS unit (Garmin GPSMap 60CSx ®) and records were kept of plot vegetation type, terrain and percentage shade. Recently burnt areas were avoided.

Within plots, only woody plants above 1.0m in height were measured as it was not feasible to measure all woody plants due to the density of woody vegetation in most areas. Therefore, in this study when elephant impact on woody vegetation around water points in Majete is discussed, it refers only that portion of woody vegetation above one metre.

Table 4.2. Details of the elephant-path classification system (adapted from Shannon et al., 2009) which was used in this study to rate the level of intensity to which a path, leading away from monitored water points, was used by elephants in Majete Wildlife Reserve

Path grade	Path width	Evidence of elephant use	Path character
1 – well used	0.6-1.0m	Dung present on path every 10-50m	Path surface has been heavily eroded and no vegetation is found on or around the path
2 – some use	0.3-0.7m	Dung only found every 50m	Some erosion of path down below the surrounding surface. Vegetation growth on the path is absent, but some plants hang over the path.
3 – seldom used	0.15m	Fresh dung seldom sighted on path	Path visible but very narrow and encroached on by surrounding vegetation.

Plants were identified and evaluated visually for browse markings or canopy removal (measured in increments of 5%) by herbivores (see Ben-Shahar, 1993; Gaugris and van Rooyen, 2009). The type of markings indicative of browsing by any herbivore (bark stripping, pushed or bent over, trunk snapped, roots eaten, and branches broken) were recorded, as well as whether the plant had coppiced. Plants browsed by smaller herbivores were grouped into one category, whilst elephant browsing was aged as either NE (new elephant use) or OE (old elephant use). NE was defined as a browsing event from within the season of sampling and OE as browsing events from prior to the sampling season. Additional measurements of plant dimensions were taken to be used in future analysis. These included plant height, trunk diameter at breast height and canopy width, length and height.

4.3.3. Data analysis

Elephant visit data

Data analyses were conducted using Statistica, version 11 (©Statsoft Inc., 2012). The effects of water point type (springs vs. rivers vs. AWPs) and season on elephant visits to water points were tested using Fixed Effect Tests and Least Squares Designs (LSDs). LSDs were also used to determine whether water points in different regions of Majete

(Sanctuary vs. hilly interior vs. Pende) were preferred differently by elephants. Spearman's correlation tests were used to determine whether a relationship existed between visits to a water point and its qualitative water measurements, size and habitat context (altitude and proximity to roads, forest patches, perennial water points and rivers). River sites were not included in the correlation analyses, as measuring the proximity of an artificially discreet river site to other features was felt to be theoretically unsound as rivers are continuous bodies. When testing the effect of proximity to permanent water points on visits, the number of perennial water bodies in a 5km radius around the water point were counted (rivers were counted as one water body) and this cumulative value used. The same analysis was attempted for a 10km radius; however, this overly reduced the sample sizes due to the close proximity of all water points in Majete.

Elephant browsing data

The 'average percentage canopy removal' in plots was selected as the browsing level indicator for all the statistical analyses (Gaugris & van Rooyen 2009). Old and new canopy removal by elephants (OE and NE respectively) was averaged for all woody plants sampled in a plot. This provided three indicators of elephant browsing in each plot: OE average percentage canopy removal, NE average percentage canopy removal and OE+NE average percentage canopy removal (an overall browsing indicator).

ANOVA and LSDs were used to test the main effects of season, 'distance from water point' (500m, 1000m, 1500m) and 'water point type' (AWP, spring, river) on canopy removal in plots. For comparisons between seasons the 2000m dry season plots were left out to ensure comparability with wet season data. The main effects were tested for OE, NE and OE+NE average percentage canopy removal in plots. OE and NE canopy removal were tested separately to allow comparison of how elephant browsing levels changed between the wet and dry seasons.

Only the browsing indicator 'OE+NE average percentage canopy removal' was used for the remainder of the analyses and simply referred to as 'elephant browsing level'. Spearman's correlations were used to test the relationship between plot elephant browsing level and the plot variables: 'proximity to perennial water points' (any water type, including rivers), 'proximity to perennial rivers', altitude and percentage shade. When calculating the variable 'proximity to perennial water points' the water point from which the transect started at was included. The effects of vegetation type and region (Sanctuary vs. Pende vs. hilly interior) on elephant browsing level were tested using ANOVA and LSDs.

The relationship between indicators of elephant use of water point ('visits to water point', 'surrounding browsing levels' and 'path use') for the wet and dry season was then analysed using Spearman's correlation tests. For this test the number of elephant visits to a water point was averaged over monitoring periods. To obtain an overall level of elephant browsing for each water point, OE+NE average percentage canopy removal was averaged across i) plots in a transect, and then across ii) replicate transects for each water point. Path use frequencies were averaged similarly. Landman (2012) and Farmer (2010) warn against averaging browsing levels across 'distance to water' categories because levels of browsing by elephants around water points are not symmetrical (Farmer, 2010; Landman et al., 2012). However, for this study an indication of elephant use of vegetation around water points was required rather than a specific measure of piosphere size.

4.4. Results

4.4.1. Elephant water point use

Contrary to the hypothesis, elephant water point preference was not affected by water point types in either season (LSD $p = 0.5$). The region in which water points were situated did, however, significantly affected elephant water point preference in both seasons. Elephants visited the Pende water point more than water points in the Sanctuary (wet and dry season LSD $p = 0.00$) or those in the hilly interior (wet and dry season $p = 0.00$) (Table 4.3.). Only in the dry season were Sanctuary water points used more than those in the hilly interior ($p = 0.02$). Standard deviation in the average daily number of elephant visits to water points, however, was high (Table 4.3.).

Season had some impact, although not significant (LSD $p = 0.059$), on elephant water point preference (i.e. the average number of elephants visiting a water point over a 24 hour period). In the dry season water points were visited, on average, by four more elephants than they were in the wet season. Evidence of visits was seldom found at any water point during the wet season. Qualitative water measurements (water movement, water turbidity, water algae levels and water odour), water point size (not including rivers) and substrate did not influence elephant water point preference.

Table 4.3.: Results of water point monitoring for elephant visits in the wet and dry seasons in Majete Wildlife Reserve. The water point located in the Pende experienced significantly more visits than those in the Sanctuary (wet and dry season $p = <0.00$) or in the Hilly area (wet and dry season $p = <0.00$).

Season	Water point location	Daily average # of elephant visits to water points recorded	Std. Dev.
Wet	Sanctuary	1	3
Wet	Pende	5	7
Wet	Hilly area	0	1
Dry	Sanctuary	4	6
Dry	Pende	25	10
Dry	Hilly area	1	4

Water point proximity to other perennial water (in a 5km radius) did not significantly affect elephant preference of water points in either season (Dry season $p = 0.37$, $R = -0.1$). However, testing this was problematic as most water points in Majete are in close proximity to other perennial water. There was an average of three perennial water points in a 5km radius around discrete water points (AWPs and springs), an average of two water points around water points in the hilly interior and only one water point near the AWP in Pende. In the dry season, water points closer to the perennial rivers were visited significantly more by elephants than water points further away (Spearman's correlation: $p = 0.01$, $R = -0.03$); this relationship was not significant in the wet season. Whether this relationship held at a smaller-scale, i.e. for only those water points found in the Sanctuary, or if it was significant at the reserve-scale only (both perennial rivers are located within the Sanctuary) was then assessed. Sanctuary water points were a maximum of 5km away from perennial rivers, whilst those in Pende and the hilly interior were over 19km away from rivers. For Sanctuary water points alone, proximity to rivers at this small-scale did not have a significant effect on elephant preference during either season (wet season $p = 0.43$, $R = -0.13$, dry season $p = 0.09$, $R = 0.27$).

Vegetation types surrounding water points included low altitude woodland (n = 4), riparian associations (n = 6) and *Brachystegia*-dominated vegetation types (medium altitude woodland and high altitude Miombo woodland) (n = 3). Vegetation type only significantly affected elephant water point preference in the dry season (ANOVA, p = <0.00). Water points in low altitude woodland were visited, on average, nine times a day by elephants, four times if in riparian vegetation, and once if situated in *Brachystegia*-dominated vegetation types .

Water points were located at a range of altitudes. Water points at lower altitudes received more visits than those at higher altitudes and this was significant in the dry season (Spearman's correlation: p = 0.02, R = -0.21) (Figure 4.2.).

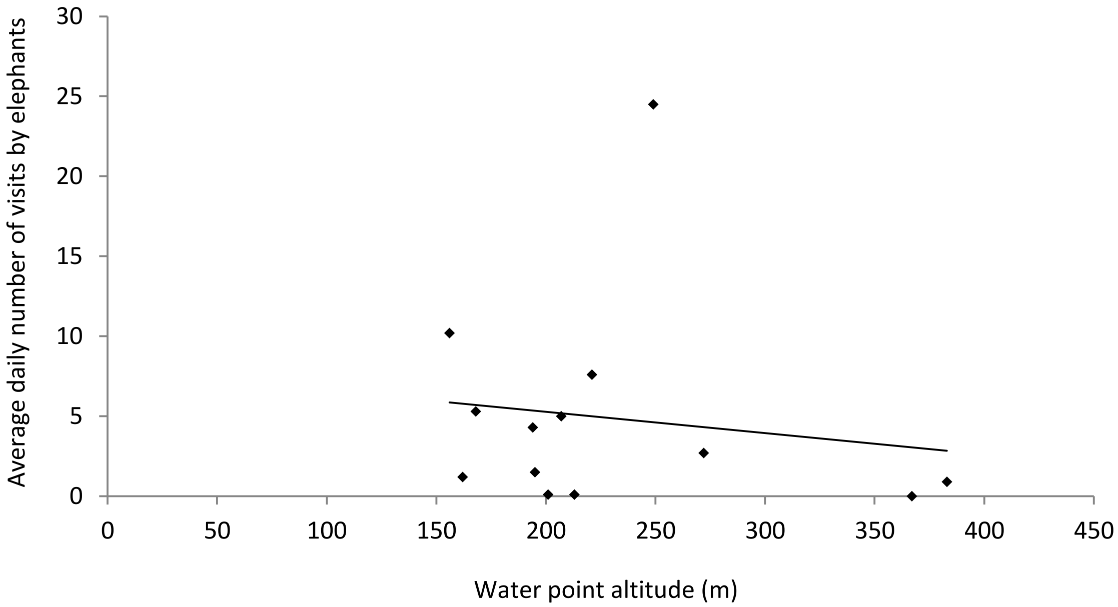

Figure 4.2.: The average daily number of times that a perennial water point was visited by elephants in the dry season is plotted against the water point's altitude. Water points at a lower altitude were visited significantly more than water points at a higher altitude (p = 0.02, R = -0.21), demonstrated with the linear trendline.

4.4.2. Elephant browsing levels around water points

Water point type was again not found to have any significant effect on levels of elephant browsing (average percentage OE+NE canopy removal) around water points (LSD p = 0.2). Season appeared to significantly influence browsing levels in plots close to water points (LSD p = 0.04). Browsing in plots at 500m was higher in the dry season than the wet season (11% versus 8% average canopy removal in plots). When browsing levels were split into OE and NE canopy removal, levels of OE canopy removal around water points did not differ between seasons, but levels of NE canopy removal were significantly higher (LSD p = 0.01) in the dry season. In the dry season, plots showed on average, 2.4% canopy removal in the NE category, compared to a 0.9% canopy removal in the wet season.

Elephant browsing levels were significantly different between regions in the wet (ANOVA p = 0.05) and dry season (p = 0.00) (Figure 4.3.). Browsing levels were higher (wet season p= 0.01, dry season p = 0.00) around water points in the Sanctuary (n = 6) than in the hilly interior area (n = 3). Browsing levels around the water point in Pende did not differ significantly to the Sanctuary in either season and only differed significantly from browsing levels around water points in the hilly interior in the dry season (p = 0.01).

The effect of plot distance from water point (500m, 1000m, 1500m and 2000m) on browsing levels also differed between regions in the dry season (Figure 4.3.).

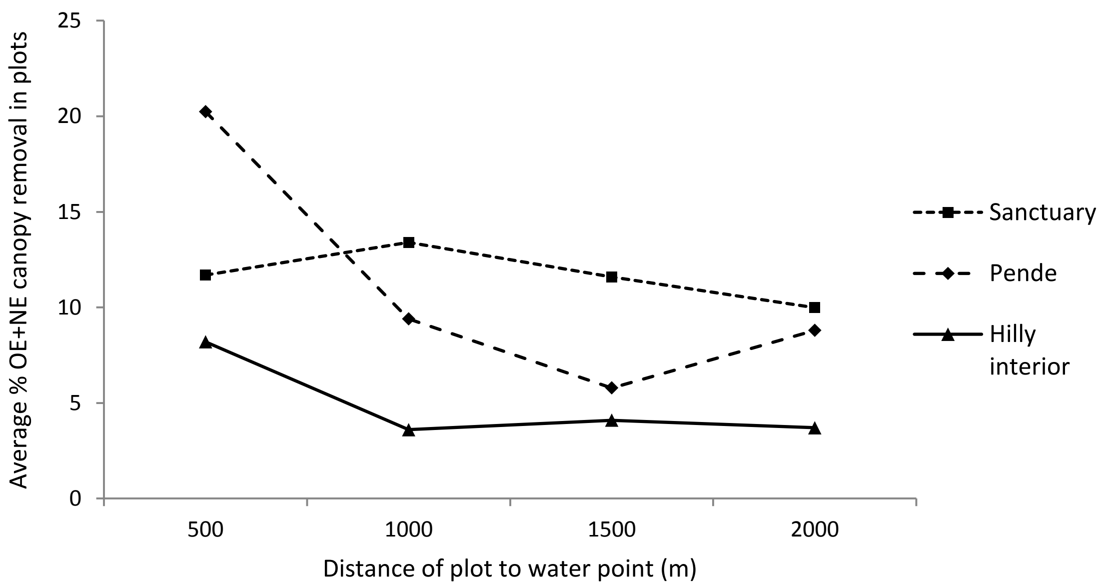

Figure 4.3. In plots sampled around perennial water points, dry season elephant browsing levels (average percentage OE+NE canopy removal) differed significantly (p = <0.00) with increasing distance from water points in different sampling areas of Majete Wildlife Reserve (the Sanctuary, Pende and the hilly interior region).

At the Pende water point browsing levels decreased sharply from 500m to 1000m (p = <0.03) and remained similar between subsequent plots. In the hilly interior there was a trend for browsing levels to be highest at 500m, but browsing levels remained consistently low across the rest of the distances on the transect. Around water points in the Sanctuary where browsing levels were greatest, distance to water point had no effect on canopy removal.

There was no significant relationship between elephant browsing levels and plot proximity to a perennial water point (Dry season p = 0.33, R = -0.08). Correlation analyses also showed that the level of elephant browsing around a water point (averaged across transect distances, then averaged between replicate transects) was not affected by water point proximity to other perennial water points. However, due to the close proximity of perennial water points in Majete the maximum distance of any plot to perennial water was only 5.5km.

Browsing levels were significantly higher in plots closer to perennial rivers during the wet (Spearman's correlation: p = <0.01, R = -0.28) and dry season (p = <0.00, R = -0.46) (Figure 4.4.). When this relationship was tested at a smaller spatial scale - i.e. only for Sanctuary water points - it was not significant. The maximum distance of any plot in the Sanctuary from perennial rivers was 7.6km.

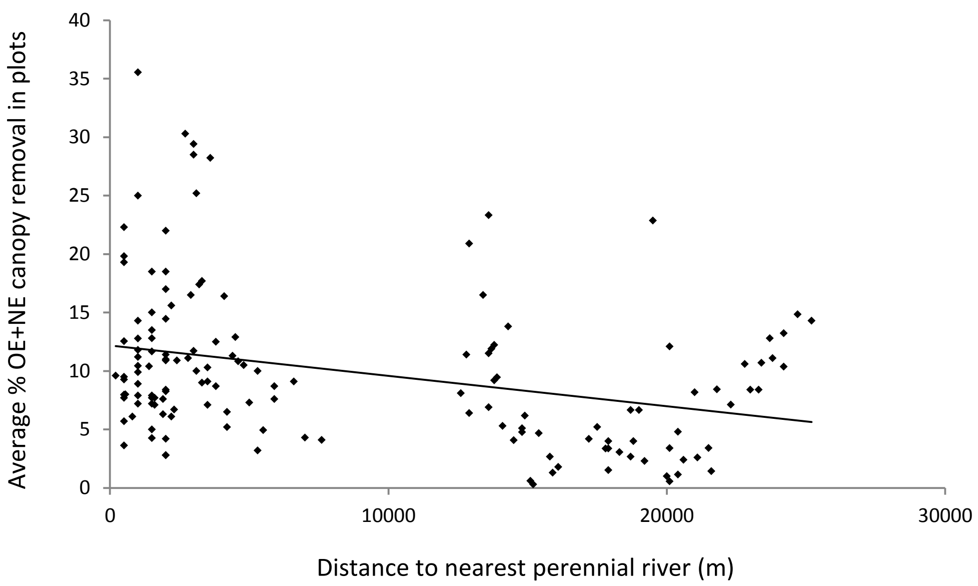

Figure 4.4. In plots sampled around perennial water points, dry season elephant browsing levels **(**average percentage OE+NE canopy removal**)** were significantly correlated with distance to the nearest perennial river (p = <0.00, R = -0.46), in Majete Wildlife Reserve

A significant negative relationship between plot browsing level and altitude was detected. Browsing levels were significantly lower for plots situated at higher altitudes in both the wet (Spearman's correlation: p = <0.00, R = -0.395) and dry season (p = <0.00, R = -0.56) (Figure 4.5.).

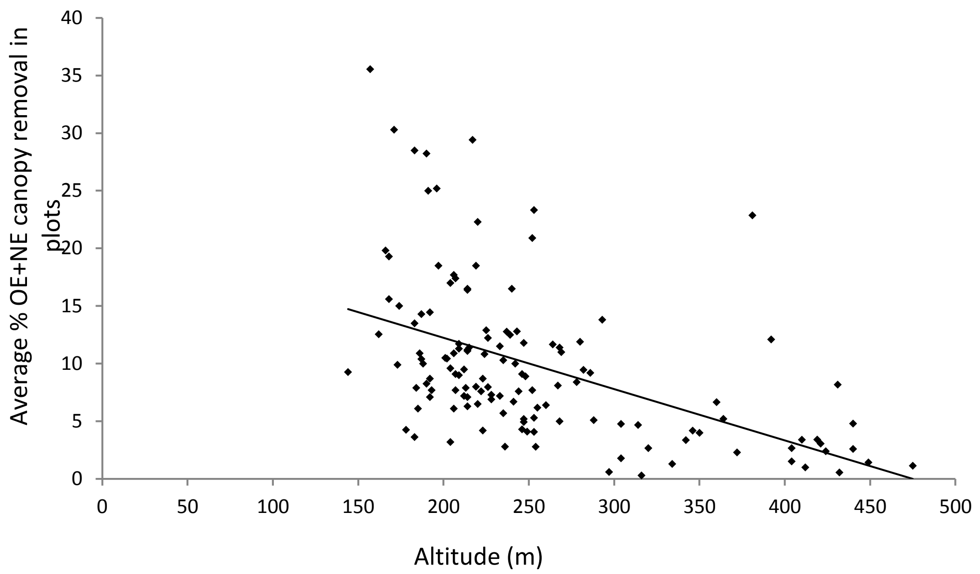

Figure 4.5. In plots sampled around perennial water points, there was a significant correlation between dry season elephant browsing levels **(**average percentage OE+NE canopy removal**)** and altitude (p = <0.00, R = -0.56), Majete Wildlife Reserve

Plot vegetation type had a significant impact on elephant browsing levels (ANOVA p = 0.03) during the dry season but not during the wet season. In the dry season, browsing levels were highest in plots situated in ridge-top and low altitude woodland, followed by riparian vegetation, then mid and high altitude woodland types (Figure 4.6.). Browsing levels were similar for plots in low altitude woodland, ridge-top woodland and riparian vegetation, and similar between mid and high altitude woodland plots. Browsing levels only differed significantly between low altitude woodland plots and mid-altitude (p = 0.03) and high-altitude woodlands (p = 0.01), then also between ridge-top woodland and mid-altitude (p = 0.03) and high-altitude woodland (p = 0.01).

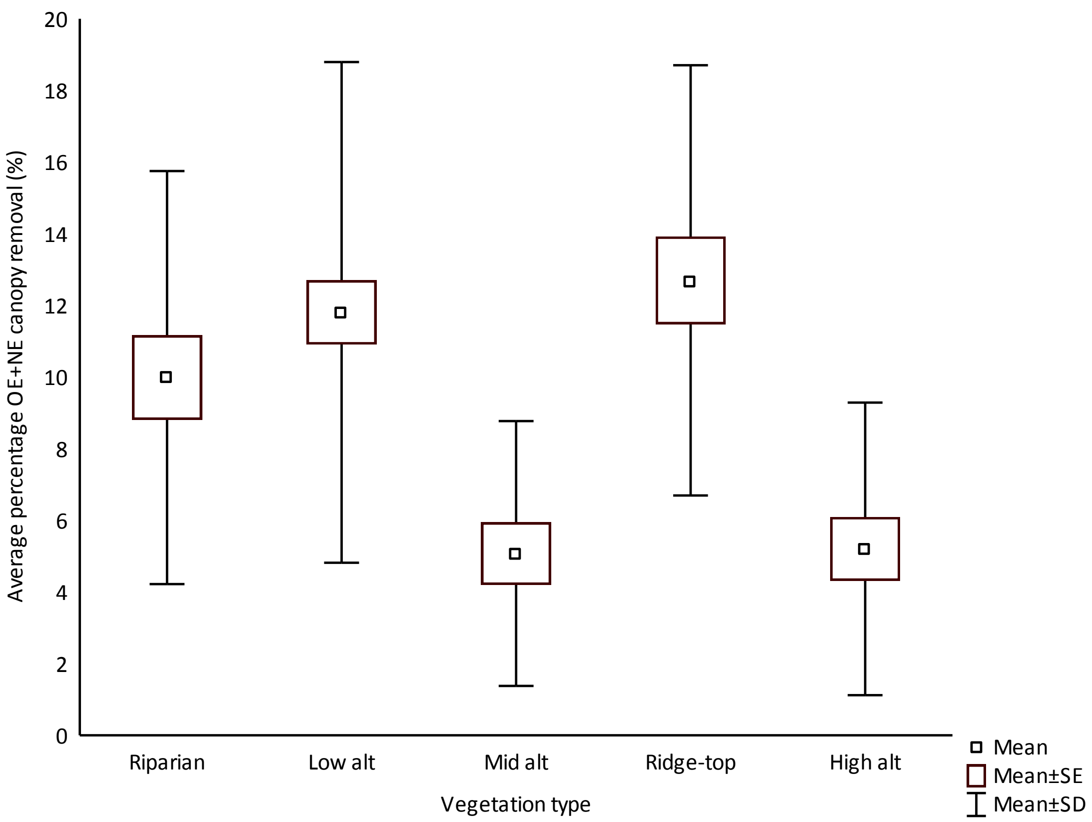

Figure 4.6.: The average percentage of canopy removal (browsing) by elephants in plots sampled around perennial water points. Vegetation type was found to differ significantly in plots of different vegetation types (p = 0.03). Canopy removal was highest in ridge-top vegetation, low altitude vegetation and riparian vegetation, but low in *Brachystegia*-dominated vegetation types, namely high and mid altitude miombo woodland.

4.4.3. Correlation between visits, canopy removal and path use

There was a significantly positive correlation (Spearman's correlation: p = 0.03, R = 0.60) between water point browsing levels and average visits to water points for the dry season and a strong trend towards a significant positive correlation for the wet season (Spearman's correlation: p = 0.06, R = 0.53). Intensity of pathway use was negatively correlated (path use index: 1 = intensely used, 3 = less used, effectively a positive correlation) with browsing levels in the wet season (Spearman's correlation: p = 0.02, R = -0.22) and in the dry season (p = <0.00, R = -0.37), i.e. path use was greater at plots that also exhibited greater canopy removal. Therefore water points which were visited frequently, had correspondingly surrounding high levels of canopy removal and path use by elephants.

4.5. Discussion

Elephant movements are not random (Stokke & du Toit, 2002; Grainger et al., 2005; Harris et al., 2008), and this is supported by the findings of this study. In the dry season, particularly, certain water points were used more by elephants. These exhibited distinctly higher numbers of visits, greater piosphere development and surrounding path use. Previous studies indicate that a number of factors influence elephant use of water points (see Chamaillé-Jammes et al. 2007; Harris et al. 2008; de Knegt et al. 2011). The results of this study showed that season and the nature of the surrounding habitat (i.e. vegetation type, terrain or altitude) affected elephant water point use. It is also likely that proximity to other perennial water points affected elephant water point usage. Contrary to the hypothesis, whether a water point was a spring, river or AWP appeared to have no impact on elephant water point usage (as for Epaphras et al. 2008; Kalwij 2010).

4.5.1. Season

There was some indication from the analyses that water points are used differentially by elephants between seasons. Surface water is readily available in the wet season, which allows elephants to range away from permanent water points (de Beer et al. 2006; Shannon et al. 2010; Smit & Grant 2009; Thomas et al. 2011) and forage in preferred vegetation types (Smit et al. 2007; de Beer & van Aarde 2008), whereas in the dry season, they are restricted to areas with available water (Stokke & du Toit, 2002; Redfern et al., 2005). This was confirmed in Majete, as in the wet season water points were visited minimally by elephants and in the dry season all monitored water points were used more frequently. The increase in elephant browsing levels close to water points provides evidence of an increase in elephant activity closer to perennial water in Majete.

4.5.2. Terrain and vegetation type

The least-used water points were all situated at higher altitudes. Altitude can be used as a proxy for terrain in Majete as increased elevation corresponds with increased steepness of slopes and rockiness (Sherry, 1989). Past studies on elephant distribution have shown that bulls avoid terrain extremes (very steep and very flat areas) and mixed herds avoid undulating areas but favour flatter habitats (Nellemann et al., 2002; de Knegt et al., 2011). Vegetation type also affected water point usage. Water points used most were situated in low altitude woodland and least-used water points were those situated in

Brachystegia-dominated vegetation types. This confirmed the findings of Sherry (1989) and Staub (2009) who documented higher elephants browsing levels in vegetation communities at lower altitudes in Majete, due to the presence of preferred plant species such as *Sclerocarya birrea, Acacia* species, *Adansonia digitata*, *Dichrostachys cinera* and *Combretum* shrub species.

4.5.3. Proximity to perennial water

Elephant water point use did not appear to be correlated with water points' proximity to other perennial water. This is contrary to findings of other studies which indicate that elephant browsing levels and path use intensity usually increase closer to water (Ben-shahar 1993; Shannon et al. 2009; Gaugris and van Rooyen 2010). This leads to the conclusion that the close proximity of perennial water points in Majete masked the effect of proximity to other water points on elephant water point usage in the analyses (see Boundja & Midgley, 2009; Matawa et al., 2012).

4.5.4. Regional water point use

Water points used most by elephants were located in the Sanctuary. The findings of this research and evidence of previous studies show that there are a number of reasons for this. Most of the Sanctuary lies in a lower altitude region, characterized by gentle slopes favoured by elephants (Nellemann et al., 2002; de Knegt et al., 2011). It is also mainly covered by vegetation types preferred by elephants (low altitude mixed deciduous woodland, riverine associations and ridge-top mixed woodland) (Sherry, 1989; Staub, 2009). There is also a higher density of perennial water points, including the two perennial rivers, in this region than in other areas of Majete. Although the rivers in the Sanctuary were not used more than other water points, elephant browsing in riparian vegetation was higher than for most other vegetation types. Previous studies have documented elephant preference for riverine habitats as they provide elephants with a source of water, shade and high quality forage (Laws, 1970; Smit et al., 2007b; Smit & Ferreira, 2010). The presence of perennial rivers in the Sanctuary could be attracting more elephants to this area, causing a related increase in elephant use of other Sanctuary water points.

Although in the analyses proximity of a water point to other perennial water also did not appear to affect elephant use of water points, the high availability of perennial water in the Sanctuary has probably contributed to attracting a greater number of elephants to this region. Personal observation, and the intensity and evenness of elephant water point use in the Sanctuary compared with water points in other regions indicate that the elephant population density is highest in the Sanctuary. Chamaillé-Jammes et al. (2007b) found that at a higher population density, elephants used water points less differentially than at lower population densities due to a 'saturation process' occurring at water points. Increased intra-specific competition forces elephants to use different water points more evenly in areas of greater elephant population density.

The spatial relationship between elephant browsing levels (and path use) around water points and distance to perennial water was also weak in the Sanctuary. This indicates that piospheres around water points in this area are likely merged due to the closeness of perennial water points in the Sanctuary (all within 5km of perennial water) (Gaylard et al., 2003; Landman et al., 2012). Water therefore does not necessarily limit the extent of elephant browsing in the Sanctuary (Grainger et al., 2005; Smit et al., 2007a; Loarie et al., 2009a). As a result woody vegetation may be used more intensively and homogenously by elephants in this region than in other areas of Majete (Gaylard et al., 2003; van Aarde & Jackson, 2007; de Beer & van Aarde, 2008). In addition to this, elephants were present in the Sanctuary at a higher density than in the rest of Majete until 2011 when the Sanctuary fence was deconstructed. It is possible that the elephants which were confined to the Sanctuary previously may have not shifted their home range location much and, therefore, higher elephant density may still persist in this area. This will only be determined, however, through future game counts.

The AWP in Pende experienced very high frequencies of visits by elephants. Browsing levels by elephants were high close to this AWP, but decreased significantly with increasing distance from water as described by traditional piosphere theory (Lange, 1969; Thrash et al., 1991). The isolation of this point from other perennial water demonstrates the impact of AWPs in extending elephant habitat use to areas previously lacking in perennial surface water (Leggett, 2006; Owen-Smith et al., 2006; Shannon et al., 2009), and how limited access to water concentrates elephant activity in an area (Chamaillé-Jammes et al., 2008). When little perennial water is available to elephants, their impacts on vegetation surrounding the remaining perennial water points may be intense (Franz et al., 2010). However, vegetation further from remaining perennial water will be protected from elephant browsing in spatial refuges (Ben-Shahar, 1993; O'Connor et al., 2007;

Gaugris & van Rooyen, 2009). This type of situation increases landscape heterogeneity and contributes positively to ecosystem heterogeneity (Chamaillé-Jammes et al., 2009).

Elephants used water points found in the hilly interior of Majete least. Evidence from this study and previous research indicates that elephants likely avoid water points in this region due to the surrounding hilly, rocky terrain of the land, and the lack of preferred vegetation types (Sherry, 1989; Staub, 2009). Observations from the field lead to the conclusion that elephants only use the water points in this region when moving between other regions. These water points play a facilitative role in providing water for elephant passage through unfavourable and arid habitats in the reserve. These water points are likely important in aiding elephant dispersal between different areas.

4.6. Conclusions

In conclusion, water points were used at different intensities by elephants in Majete and that this was affected by season and nature of the surrounding habitat, but not water point type. Water points used least were those at high altitudes, in steep, rocky hilly areas and surrounded by vegetation types preferred least by elephants. Water points situated in favoured vegetation types, at low altitudes where the terrain was flatter were used most. Proximity to other perennial water points did not appear to affect elephant water point usage in Majete, likely due to the close proximity of perennial water points in the reserve.

Water points in the Sanctuary were used most by elephants. This could be because this area consists of flat, low altitude terrain, favoured vegetation types are prevalent, there is high availability of perennial water and perennial rivers are accessible. There are four AWPs in the Sanctuary which create ideal tourist viewing opportunities, but they are providing water in an already water-abundant area. An overabundance of water points in this area could result in decreased heterogeneity of elephant impacts (Gaylard et al., 2003; Redfern et al., 2005; Farmer, 2010) and potential loss of woody diversity and cover (Owen-Smith et al., 2006; O'Connor et al., 2007). The suggestion that browsing may not be restricted by perennial water availability in the Sanctuary is supported by findings of a previous study (Chapter Three). Woody cover loss in the Sanctuary was shown to be widespread and not associated with distance to perennial water.

The number of water points should be reduced in the Sanctuary, or a rotation system could be established where AWPs are alternately switched off every few years. Changes to surface water availability should, however, be carefully considered as elephants would likely make greater use of the perennial rivers, placing further pressure on surrounding riverine vegetation (Smit & Ferreira, 2010). Vegetation sampling demonstrated that elephant browsing levels were also high in riparian vegetation. Regular monitoring should be undertaken along the perennial rivers in the Sanctuary so that vegetation changes can be detected. Should the number of AWPs be reduced in this area, then others should be provided in suitable habitats further away from the Sanctuary to encourage elephant dispersal to other areas.

Additionally, the concept of creating spatial heterogeneity in elephant browsing levels across Majete needs to be incorporated into future water management plans (Gaylard et al., 2003; Farmer, 2010). If water is supplemented then AWPs should be positioned far away enough from other perennial water so that spatial heterogeneity in elephant browsing in encouraged (Gaylard et al., 2003; Farmer, 2010). The distinct piosphere (in elephant browsing levels) around the isolated Pende water point, for example, contributes to landscape heterogeneity in elephant browsing because arid areas are protected from browsing in spatial refuges (Chamaillé-Jammes et al., 2009). Management should not aim to create similar levels of elephant impact on vegetation throughout the reserve, but rather mosaics of differing intensities of browsing. This could be achieved in Majete using strategic positioning of waterholes and fencing to exclude elephants from particular areas (Owen-Smith et al., 2006; Chamaillé-Jammes et al., 2007b; Loarie et al., 2009a). Long-term monitoring programmes should be implemented so that the impacts on woody vegetation due to changes in surface water availability can be detected. Monitoring of elephant vegetation impacts is vital in a fenced reserve such as Majete, in which a large population of elephants is already supported. Surface water manipulation has been cited as a tool for managing elephant distribution and vegetation impacts (Gaylard et al., 2003; Chamaillé-Jammes et al., 2007b; Martin et al., 2010) in suitably large and water-limited reserves (Smit et al., 2007c). Majete is sufficiently large and water-limited for water point manipulation to be effective, and possibly be used as a tool to manage elephant impacts on vegetation.

4.7. Acknowledgements

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Chapter Five

Research findings, conclusions and management recommendations – for submission to African Parks Majete management staff

5.1. Overview

The artificial supplementation of surface water in Majete has implications for the conservation of woody vegetation in the reserve. This study essentially addressed the landscape- and local-scale impacts of increasing water availability on woody vegetation. At a landscape-scale, this study provided baseline information on changes in woody vegetation cover in Majete since the elephant reintroductions by means of a GIS analysis of remotely-sensed images of woody vegetation cover. The relationship between woody cover change and proximity to perennial water points was explored. At a local-scale, elephant seasonal-use of perennial water points in the reserve was analysed with the aim of determining which water points were used most by elephants, and which factors affected this. The findings of this research, coupled with supporting information from a literature review, will aid Majete in developing a water management strategy to manage elephant impacts on vegetation in the reserve. In this report, the key study findings are summarized and the management implications of these findings are discussed in detail.

5.2. Study findings

5.2.1. <u>Chapter Three: Woody vegetation cover changes in Majete</u>

- Change in woody cover in the reserve was most pronounced between 2000 and 2010, during which time ‘non-woody’ vegetation cover doubled in Majete. Woody cover loss was highest in the Eastern lowlands region of the reserve, then in the Western highlands and then in the Sanctuary.

- Analysis of historic rainfall data showed that the reserve experienced consecutive years of below expected annual rainfall prior to 2010. Drought could possibly explain some of the woody cover loss seen in the 2010 map.

- The Sanctuary experienced few fires between 2000 and 2010; therefore woody cover loss in this region was likely due to a combination of herbivory and drought. Woody cover loss was widespread throughout this region and was not associated with proximity to perennial water. This indicated that herbivory in the Sanctuary may not be limited by water availability; this is highly likely as most of the region (87%) is within 4km of perennial water. Woody cover loss could not be attributed specifically to a particular herbivore species in this study. It was suggested that water availability may need to be reduced in the Sanctuary to limit the extent of herbivore foraging pressure on vegetation.

- In the Eastern lowlands and Western highlands, new non-woody points experienced a higher fire frequency than those in the Sanctuary. Their location was also not associated with proximity to perennial water or altitude. New non-woody points were clustered and linearly distributed along the South-East line, and were mainly situated on Northward-facing slopes. New non-woody point cover was also higher on steeper slopes. This suggested that the synergistic effects of fire and periods of drought may have driven woody cover loss in this region. Woody vegetation may continue to be degraded in these areas if exposed to such regular fires. Additional fire breaks should be created in the hilly, high altitude regions of Majete to facilitate early burning strategies and wildfire control.

5.2.2. <u>Chapter Four: Elephant usage of perennial water points in Majete</u>

- Water points were used more in the dry season and evidence was found of increased elephant browsing levels close to water points.

- Water points in Majete were not used equally by elephants in the dry season. Specific temporal and site characteristics influenced elephant use of water points.

- Elephant water point usage was not correlated with proximity to other perennial surface water. This was likely because perennial water points in Majete are abundant and in close proximity to one another. Water points closer to the perennial rivers were used more at the reserve-scale, but this relationship did not hold at a local-scale (i.e. for waterholes in the Sanctuary, closer to the rivers).

- Elephants seldom-used water points situated at higher altitudes, where the terrain is steep, hilly and rocky. These water points were mainly surrounded by *Brachystegia* and *Julbernadia*-dominated vegetation types, and very little elephant browsing took place around them. Water points which were used most by elephants were found at lower altitudes, particularly if they were situated in ridge-top mixed woodland, low altitude mixed deciduous woodland or riverine vegetation.

- In general, the most well-utilised water points were found in the Sanctuary. Elephant browsing levels around water points in the Sanctuary were high and similar around individual water points. The weak spatial relationship between elephant browsing levels and distance to water in this area indicated that elephant browsing may not be restricted by water availability (supported by findings of Chapter Three) – perennial water is abundant in the Sanctuary. It was also suggested that perennial water availability may need to be reduced in this area.

5.3. Specific conclusions and management recommendations

5.3.1. Sanctuary

Despite seldom being burnt between 2000 and 2010 woody cover loss (as detected in Chapter Three) was similarly high in the Sanctuary as in areas which had been frequently burnt during this time. Although woody cover loss in this area was attributed to herbivory in Chapter Three, the data did not provide evidence for the loss to be attributed to any one herbivore species. Results in Chapter Four indicated that there is a greater use of the Sanctuary area by elephants than for other areas of the reserve. In the Sanctuary, most of the area (87%) is within 4km of perennial water. In Chapter Four it was also found that elephant browsing levels did not decrease with increasing distance from water and that proximity of plots to perennial water points or rivers had no effect on elephant browsing levels. It is, therefore, likely that elephant browsing is not limited in this area by surface water availability, even in the dry season. The fact that points of woody cover loss were not associated with proximity to perennial water supports this notion and shows that herbivory may be widespread in the Sanctuary. Providing water artificially for wildlife buffers against dry season water shortages and creates wider foraging opportunities for wildlife (Chamaillé-Jammes et al. 2007a). This can reduce seasonal variation in elephant use of foraging areas (Shannon et al., 2006; Loarie et al., 2009a; Young et al., 2009) and allow extensive elephant foraging in previously arid areas (Grainger et al., 2005; Harris et al., 2008; Loarie et al., 2009b). Past research has shown that were perennial water is abundant in an area, piospheres around individual water points are not discrete but merge with piospheres of other water points (Landman et al., 2012), which results in the homogenisation of elephant browsing levels (Gaylard et al., 2003; Chamaillé-Jammes et al., 2009) – a situation which may exist in the Sanctuary. If no water restrictions to elephant foraging exist, elephants are then able to forage in preferred vegetation types, rather than only in vegetation closer to perennial water (Grainger et al., 2005; Shannon et al., 2006; Loarie et al., 2009b), which could make preferred vegetation types vulnerable to high elephant impacts. Results from Chapter Four, as well as findings from Staub et al. (2009) and Sherry (1989), show that elephant browsing is highest in vegetation types found at lower altitudes (i.e. riverine associations, low altitude mixed deciduous woodland and ridge-top mixed woodland), which are also the dominant vegetation types of the Sanctuary.

From the findings of this research it is apparent that there are too many perennial water points in the Sanctuary area. With the addition of the AWP at the Day Centre (between Thawale AWP and the Shire River), four AWPs supplement water already available at two springs in the Mwembezi River and in the perennial rivers. Closure of one of the AWPs in the centre of the Sanctuary, Nakamba or Nsepete, may be the simplest option for reducing water availability in the region. Closure of one of these waterholes may, however, lead to increased foraging by herbivores around the remaining waterhole and along the perennial rivers (Franz et al., 2010; Smit & Ferreira, 2010). Alternatively, instead of permanently closing one of these AWPs, they could be switched on and off on a rotational basis every three or four years. The effects of this on woody vegetation should be monitored regularly and managed adaptively according to management objectives. However, even if one or both of these AWPs were closed, water availability may remain high in the Sanctuary due to the presence of perennial rivers, springs (in Mwembezi River) and the other two AWPs at Thawale and the Day Centre.

Majete also relies heavily on revenue generated from tourism and the AWPs in the Sanctuary are a key tourist attraction as they provide good wildlife viewing opportunities in the dry season. Management of water points in the Sanctuary will need to balance tourism interests and the need for water by smaller less-mobile herbivores, with the management of elephant impacts in this area. Although water is abundant in the Sanctuary, future water management strategies cannot be aimed only at controlling the distribution of elephants, as water needs to be available for smaller game species as well (Owen-Smith, 1996; Smit et al., 2007b). In Majete, thresholds of acceptable levels of elephant impacts on vegetation need to be defined for particular areas. For tourists, it would be desirable for elephants to be encountered frequently. However, loss of woody cover caused by locally high elephant densities in the Sanctuary would be not only be aesthetically unappealing, but also detrimental to other herbivorous species and vegetation diversity. It may be that vegetation in particular areas in the Sanctuary will need to be sacrificed in order to reach tourism objectives and provide water for smaller herbivores. For example, if it is decided that frequent encounters with elephants between Thawale and the Day Centre are critical to attracting tourists, then degradation of woody vegetation in this area should be expected. It is recommended that if certain areas are designated as elephant high-impact areas, then key areas of woody plant diversity should be identified in the Sanctuary for protection from elephant browsing, either by reducing water availability or by fencing exclusion plots.

The results from Chapter Four indicated that there is a higher population density of elephants in the Sanctuary, compared with other areas; however, further information is needed on the size of the elephant population in Majete and on its distribution. Now that the Sanctuary fence has been removed, the high density of perennial water points and availability of high quality forage in the Sanctuary may attract elephants to this region in the dry season. If the elephant density remains high in this region, then we may see a further reduction in woody vegetation cover and the structure changed to a more open state.

Local elephant densities can be manipulated by water positioning in large-enough, water-limited reserves (Chamaillé-Jammes et al., 2007b; Smit et al., 2007b; Martin et al., 2010). AWPs could be positioned away from the Sanctuary area, preferably at least 10km away, and in this way attract elephants to other suitable areas of the reserve. Previous plans to position another AWP in the Phiringombe area, between the Sanctuary and Pende could provide a suitable initial waterhole to facilitate the attraction of elephants to other parts of the reserve. However, this location is less than 10 km away from Mwembezi Spring and Thawale waterhole. Another point of consideration is that new AWPs should not be placed directly in elephant-preferred vegetation types, such as in dry streambeds bordered by riverine vegetation, as this will result in intensive elephant use of vegetation around the new AWP which could facilitate erosion (Smit et al., 2010).

5.3.2. Eastern and Western regions

Results from Chapter Four indicated that there was a classic piosphere around Dam 1 in the Pende area, Eastern lowlands reigon (Figure 5.1.). Elephant browsing levels were high close to Dam 1, but decreased clearly with increasing distance from water. Although Dam 2 is only within 5km of Dam 1, during fieldwork sessions few elephants were found to be using this water point - although it is clearly an important water source for smaller herbivores (evident from the proliferation of tracks).

Figure 5.1. Undergrowth and mid-canopy cover closer to the water is absent at Dam 1, but these increase with increasing distance from the waterhole. Piosphere effects may be intensive around isolated water points (those far from other perennial water), but woody plants further away from water are protected in spatial refuges.

If elephants preferentially use Dam 1, then elephant impacts on vegetation will be concentrated around this waterhole but plants further away will be protected from browsing in spatial refuges (O'Connor et al., 2007; Franz et al., 2010). This process contributes positively to spatial heterogeneity (Chamaillé-Jammes et al., 2009).

In Chapter Three, woody vegetation loss in this Eastern lowland region was attributed mainly to fire and drought. Browsing impacts from elephants and other wildlife, however, may not have been evident in this study as the map used in the analyses was from 2010 – only a few years after wildlife was reintroduced to this area. The high level of browsing by elephants around Dam 1 (Chapter Four) requires that this area be protected from frequent fires if woody vegetation degradation is to be avoided (particularly in drought situations). Woody vegetation cover could be further reduced in this area if exposed to high levels of both herbivory and fire (Sankaran et al., 2008; Vanak et al., 2012).

During the dry season fieldwork for Chapter Four, I included Phwadzi Spring in the perennial water point sample set and some sampling was undertaken, however, due to time restrictions and the distance to Phwadzi, I was not able to complete sampling at this water point. Phwadzi Spring is large and is very well visited by elephants throughout the dry season (Figure 5.2. a and b). During fieldwork over 30 elephants were observed at Phwadzi Spring every day and scouts report that it is frequented by elephants even in the wet season. Intensive elephant browsing is visible mostly in the riverine vegetation along the Phwadzi River, although some browsing is evident in *Brachystegia*-dominated vegetation between Phwadzi and Diwa Springs.

In Chapter Three, fire was found to have occurred in this region frequently between 2000 and 2010. The Phwadzi drainage system is another area in which fire control is critically important. The intensive browsing of vegetation around this drainage system by elephants and other game species, combined with the effects of regular fires may lead to the degradation of woody cover along the Phwadzi River. Burning of fire breaks around the main Phwadzi drainage area should be implemented so that riverine vegetation is protected from wildfires. Combined effects of fire and elephant browsing can exacerbate change and degradation of woody vegetation (Levick et al., 2009; Midgley et al., 2010; Vanak et al., 2012) and in areas where browsing levels are high, vegetation should be protected from fire.

a)

b)

Figure 5.2.a and b: Phwadzi Spring is extensive and is regularly frequented by large numbers of elephants. Monitoring of elephant impacts on riverine vegetation in this area, and protection of the Phwadzi drainage area from fires is essential.

The high incidence of fires in the Western highlands region of Majete between 2000 and 2010, coupled with drought likely caused the woody cover loss detected in this region in Chapter Three. The sensitivity to fires of some of the plant communities (such as *Brachystegia*/*Julbernadia*-dominated vegetation types) in this area (Trapnell, 1959; Lawton, 1978; Cauldwell & Zieger, 2000) requires that additional fire breaks are developed in this region to enable more focused early burning and control of wildfires.

The water points monitored in the hilly interior of Majete - namely Diwa and Madzinchulu Springs - were seldom used by elephants. However, it seems that they function importantly as drinking points that facilitate elephant movement through this hilly area between the Phwadzi and Pende regions. Another nearby spring in the Masakala River was selected for monitoring initially; however, it dried up by June and therefore did not contribute to dry season surface water availability.

5.4. Reserve-scale management recommendations

Manipulating surface water availability has complex impacts on ecosystems (Gaylard et al., 2003; van Aarde et al., 2006). In a fenced reserve, such as Majete, it will become increasingly necessary to manage AWPs to control the impacts on vegetation of a growing elephant population (Chamaillé-Jammes et al., 2007b; Loarie et al., 2009a). As a fenced reserve, Majete could experience future woody plant loss or structural changes as a consequence of high water availability, particularly in the Sanctuary (Duffy et al., 2002; Owen-Smith et al., 2006; van Aarde & Jackson, 2007).

The concept of creating spatial heterogeneity in elephant browsing across Majete should be incorporated into water management plans (Gaylard et al., 2003; Farmer, 2010). Management should not aim to create similar levels of elephant impact on vegetation throughout the reserve, but rather mosaics of different levels of browsing in individual vegetation types. For example, if areas of low altitude mixed deciduous woodland in the Sanctuary succumb to heavy usage by elephants, then other areas of low altitude woodland should be protected from elephant use. Management needs to determine which regions of the reserve will be designated as high elephant impact areas, and which need to be protected from elephants. Decisions should take into account vegetation type, plant diversity, terrain and proximity to water (Rogers, 2003; O'Connor et al., 2007). Differing levels of elephant browsing across Majete could be achieved using the strategic positioning of waterholes and the use of fencing to exclude elephants from particular areas (Owen-Smith et al., 2006; Chamaillé-Jammes et al., 2007b; Loarie et al., 2009a). New AWPs should preferably be placed in areas where water was historically available, but not near riverine vegetation of seasonal rivers. The aridity of seasonal rivers protects riverine vegetation from elephant use in spatial refuges and can function to preserve areas of riverine habitat in Majete (Smit & Ferreira, 2010).

Long-term monitoring needs to be conducted regularly in Majete so that the distribution and intensity of elephant impacts on woody vegetation can be detected. A long-term programme monitoring elephant impact on different vegetation types in Majete was started by Caroline Staub in 2009. Regular re-sampling should be undertaken at the permanently marked study sites approximately every five years, so that the level and nature of elephant impacts on vegetation can be monitored. This programme consists of sufficient sample plots in many of the areas vulnerable to high elephant browsing levels.

However, plots distribution in the programme did not take proximity to perennial water points into account. Additional plots should be established in Majete which specifically account for proximity to perennial water, so that long-term impacts of elephants around water points can be monitored. This is of great importance along the perennial rivers. Exclusion-plots (to prevent elephant browsing) in different vegetation types should also be used, and would provide long-term data against which the level of elephant impacts in the rest of Majete could be compared (see Young et al., 1998; Levick et al., 2009). Fixed-point photography could also be used at water points and at proposed waterhole sites to monitor the changes in vegetation taking place around perennial water (Tafangenyasha, 1997; de Beer et al., 2006). Chapter Three provides baseline data on the changes in woody vegetation cover in Majete and the relationship of change to proximity to perennial water. Based on the level of change observed in a ten year period, it would be advisable for follow-up studies be undertaken approximately every five years. Although woody cover loss has occurred in Majete, it should be remembered that elephants and other mega-herbivores were absent, or only present in low densities in the reserve for over 20 years. Vegetation may simply be reverting to an historic state, similar to when wildlife inhabited the region at higher population densities (see Cormack, 1992; Mosugelo et al., 2002; Skarpe et al., 2004). However, the prevention of elephant dispersal, in particular, from Majete by fencing may result in greater changes to woody vegetation by elephants than in the past when Majete was unfenced (van Aarde et al., 2006; van Aarde & Jackson, 2007). This highlights the need for a monitoring programme of vegetation dynamics to be implemented in the reserve.

Water point management in a fenced reserve, such as Majete, is not a straight-forward task when one considers that surface water management needs to address tourism and biodiversity issues. There is scope to reduce the number of waterholes in the Sanctuary and to provide alternative waterholes elsewhere in the reserve to attract elephants to different areas. Any changes to the current water management situation need to be carefully considered, and the impacts of altering water availability on woody vegetation and elephant distribution should be monitored consistently over a long time period. A priority for Majete at this stage is the defining of ecological objectives for each area of the reserve. Thresholds of acceptable levels of elephant impacts need to be identified for specific areas and surface water managed accordingly (see Kruger National Park's elephant management policy). Where certain areas are designated as high elephant impact zones, then areas of similar habitat should be protected from elephants. Crucially, an objective of any water management policy should be to create spatial heterogeneity of elephant impacts (Gaylard et al., 2003; Rogers, 2003; Farmer, 2010).

5.5. References

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Appendix One

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Appendix Two

Supporting information for Chapter Three

Running mean of rainfall

Annual rainfall levels vary substantially between years. Therefore, in studies where the long-term pattern of rainfall may affect vegetation it is of more use to study the rainfall trend rather than the actual level of rainfall received annually (see Mills et al., 1995). In Chapter Three a three year running mean was calculated for each year in the time period between 1985 and 2010. The running mean rainfall value for a given year was calculated by averaging the actual rainfall amount received in that year, and in the two years before it (Table A.1.); this effectively smoothes the rainfall time series. A five year running mean was also calculated but this overly smoothed the data and reduced most visible rainfall trends between the datasets (the time spans between the woody cover datasets were between five and ten years).

Table A.1.: Calculations of 3 year running mean rainfall for Majete Wildlife Reserve, and weighted means of rainfall for each woody cover dataset year

Dataset	Image year used in dataset	No. of images used	Weight	3 year running mean rainfall value (mm)	Weighted mean of rainfall for dataset (mm)
1990	1989	1	0.25	859	
	1990	1	0.25	845	
	1991	2	0.50	784	
					818
2000	2000	5	0.63	838	
	2001	3	0.38	957	
					883
2010	2009	1	0.14	914	
	2010	1	0.14	619	
	2011	5	0.71	614	
					656

Weighted mean of rainfall

As the woody cover datasets used in this study were developed from multiple images, from different years (e.g. the 2010 dataset was developed using 2009, 2010 and 2011 images), an indication of the amount of rainfall which could have affected woody vegetation cover in the images was also needed. Due to the composite nature of the datasets, a weighted average of rainfall for each dataset was required. For example, for the 2010 dataset, five of the seven images used in developing it were from 2011, one was from 2010 and one from 2009. The weighted mean was calculated for the 2010 dataset using the running mean rainfall values for 2009, 2010 and 2011. The running mean rainfall average for 2009 was 914mm, for 2010 it was 619mm and for 2011 it was 614mm. The calculations of weighted mean rainfall for each dataset is detailed in Table A.1.

Reintroduction history of wildlife reintroduction to Majete

Wildlife was reintroduced to Majete in stages (Table A.2.). All reintroductions were made to the Sanctuary before 2007 as only the Sanctuary fence was in place. After fencing was completed in 2008, wildlife was also reintroduced to the rest of Majete. The Sanctuary fence was only deconstructed in 2011.

Table A.2.: Details of wildlife reintroductions made to different regions of Majete Wildlife Reserve between 2003 and 2010.

	Sanctuary								Areas outside Sanctuary			
Species	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**Total**	**2008**	**2009**	**2010**	**Total**
Black Rhino	2				6			**8**				
Elephant				70				**70**	64	83		**147**
Buffalo	120	100						**220**	86			**86**
Sable	100							**100**	153		99	**252**
Hartebeest		4		10		15	30	**59**				**0**
Waterbuck	98							**98**	198		106	**304**
Zebra		37	50	9		38		**134**		40		**40**
Eland		20				32		**52**		25		**25**
Impala	216							**263**	210		311	**474**
Nyala	6	15						**25**	38			**34**
Warthog	60							**60**	98			**98**