IMPACT OF ELEPHANTS (*LOXONDONTA AFRICANA L.*) ON BAOBAB TREES (*ADANSONIA DIGITATA L.*) IN MANA POOLS NATIONAL PARK OF THE MID-ZAMBEZI VALLEY REGION

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A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Tropical Resource Ecology (2013).

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ABSTRACT

An assessment was made on elephant impact on a baobab population in Mana Pools National Park. Study sites were selected on the basis of their distance from the Zambezi River, a major source of perennial water in the park. The study also compared current levels (2005) of baobab density, damage and mortality to those obtained during 1980s in the northern part of Mana Pools National Park.

Results showed that baobab density was significantly lower in 2005 than that obtained in the 1980s for northern Mana Pools National Park (Mann-Whitney, U=256, p=0.037). However there was no significant difference in basal area per km^2 for the two study periods (Mann-Whitney, U=330, p=0.417995). The girth at breast height (gbh) size class distribution showed a shift in pattern from a J-inverse shape with more saplings in 1984 to a bell-shaped pattern with fewer plants in the smallest gbh size class in 1988 and 2005. Baobab mortality for the intervening period (1984-2005) was found to be lower (4.42%) than that experienced in the 1980s (7.3%). Results from this study showed that baobab density is significantly different between southern, central and northern Mana Pools National Park (Kruskal-Wallis ANOVA by ranks, H=7.61, df =2, p=0.022). The density increased with increased distance from the Zambezi River. Mean basal area per km^2 and mean height was also significantly different between study sites. Chi-square test of independence showed a significant difference between damage categories for the three study sites and it was concluded that there is a relationship between damage and study sites. Elephant induced damage on baobab trees decreased with increasing distance from the Zambezi River.

The study concluded that elephants are shaping the baobab structure in Mana Pools National Park. If baobab mortality in northern Mana Pools National Park continues at the present rate then only few old baobab trees will remain.

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1.1 Background

1.1.1 General introduction

Ecological effects of elephant (*Loxodonta africana* Blumenbach) has emerged a contentious issue for instance the concept of the 'elephant problem and the concerns that elephants may irrevocably alter the remaining areas that are available to them' (Caughley, 1976a). There is a serious debate on whether elephants should be culled or not. Some scholars support the notion that elephants should be culled (Cumming *et al*., 1997; Van Aarde *et al*., 1999; Whyte *et al*., 1999) while some authors have discounted this (Owen-Smith *et al*., 2006).

Elephants exert a direct impact on woody vegetation by feeding on plant parts, breaking or trampling of trees, stem push over, uprooting, snapping or debarking (Barnes, 1983a; 1983b; Boundja *et al*., 2010; Gandiwa *et al*., 2011; Kohi *et al*., 2011; O'Connor *et al*., 2007; Shannon *et al*., 2008; Swanepoel and Swanepoel, 1986). When such impacts are sustained and selective, they affect the relative growth, survival and reproductive output of these plants, with consequences on vegetation structure, community composition, cover and ecosystem processes (Jachmann and Bell, 1985; Smallie and O'connor, 2000; Kohi *et al*., 2011).

The baobab (*Adansonia digitata* Linnaeus) is a tree species widely susceptible to elephant damage, due to its soft, pithy wood (Owen-Smith, 1988). Elephants strip off bark, and then gouge deepening holes into the trunk, until the tree collapses (Barnes, 1980; Weyerhaeuser, 1985; Swanepoel, 1993b). Several studies have shown that baobab populations have decreased where elephants have reached high local densities especially during drought years (e.g., Robertson-Bullock, 1960; Napier Bax and Sheldrick, 1963; Laws, 1969; Caughley, 1976b;

Barnes, 1980; Geerling, 1983; Weyerhaeuser, 1985; Swanepoel, 1993b, Edkins *et al*., 2007; Mpofu *et al.*, 2012).

The baobab is a very long-lived tree species characterized by a low regeneration rate (Baum, 1995a). Trees can reach an age of over 1000 years (Baum, 1995a). The exceptional longevity of the baobab means any short-term changes in mortality as a result of elephant damage may have long-term consequences (Weyerhaeuser, 1985). Using carbon-dating, Swart (1963) estimated that a large baobab in Lake Kariba, Zimbabwe, could be some 1010 ± 100 years old. Using the same method, Patrut *et al*. (2007) proposed the age of another tree in north-eastern Namibia at about 1275 ± 50 years 2007).

Elephants are the only herbivores that can kill adult baobabs, and are linked to reduction in baobab densities in most protected areas in Africa (Swanepoel and Swanepoel, 1986; Swanepoel, 1993b; Mpofu *et al*., 2012; Kupika *et al*., in press; Barnes *et al*., 1994; Weyerhaeuser, 1985; Whyte *et al*., 1996; Kelly, 2000; Edkins *et al*., 2007). In a 10-year study in Tanzania, Barnes *et al*. (1994) found that baobab populations declined as elephant numbers increased and that the baobab recovered when elephant populations declined due to poaching.

Similar with other species, the impact of elephants on baobabs is confounded by interactions with drought (Whyte *et al.*, 1999; 2001a; 2001b), other herbivores (Edkins *et al*., 2007) and fire (Venter and Witkowski, 2013). Furthermore, the pattern of elephant effects on baobabs is inconsistent across size-classes, either showing selection against small trees (Weyerhaeuser, 1985; Barnes *et al*., 1994) or no size class selection (Swanepoel, 1993b).

There has been considerable debate over the lack of long-term ecological studies to guide policy and decisions related to elephant management in protected areas (Owen-Smith *et al*., 2006). Bakker *et al.* (1996) and Tilman (1989) suggest that long-term studies are vital for the

understanding of global change. They argue that very long-term studies, probably beyond the lifetime of the researchers, will continue to show important effects. This study revisit baobab studies in Mana Pools National Park by Swanepoel and Swanepoel (1986) and Swanepoel (1993b). This was made possible because in these studies baobab trees were numbered making it possible to relocate them.

1.1.2 Historical background of elephants in Mana Pools National Park since 1900

Historical evidence suggests that elephant populations in the mid-Zambezi Valley were severely reduced by 1900 and that the present population has increased since that time and has also been compressed into a more restricted range(Kerr and Fraser, 1975).The first attempts at aerial census of wildlife in Zimbabwe took place during the 1960s (GOZ, 1996).

In October 1966, the Department of National Parks and Wildlife Management conducted an aerial survey covering the Nyakasanga Controlled Hunting Area (CHA), Mana Pools Game Reserve and Sapi CHA (Kerr and Fraser, 1975). The survey revealed only 324 elephants. In light of a hunting quota of 215 and a kill of 158 elephants in the three CHA areas during the 1966 season, the apparent decline in elephant numbers caused concern. The outcome was the suspension of elephant hunting in all CHA's during 1967.Contrary to this, it has been reported that over 330 elephants were culled in Mana Pools National Park from 1969 to 1976 (Dunham, 1994).

A survey was subsequently conducted in February 1967. The aim was to investigate the seasonal distribution and movement patterns of elephants in the area as a basis for future management. The study revealed that there was no evidence of seasonal movements to or from the study area and differences in seasonal totals were attributed to differences in visibility in the

generally markedly deciduous vegetation types. The February-March surveys suggested that the populations were generally widely dispersed during the rainy season. Some preferences are shown for the major tributaries of the Zambezi, in particular the Rukomechi and Nyakasanga Rivers. By mid dry season no major changes in the distribution patterns were observed.

The dry season is well advanced by September and by then the major changes in the distribution patterns are apparent in the west of the Chewore River. The dispersed population observed in June has now concentrated in certain areas. The major annual concentration areas were along the Mana Pools and Sapi mouth alluvia, and along the lower Rukomechi River near its confluence with the Zambezi River. Fairly large aggregations were also seen just east of Chirundu village on the Mwangu River in 1968 and adjacent to the Zambezi River in the Urungwe CHA in 1969.However subsequent studies have shown that elephant cows which use the Zambezi riverine woodland in the dry season usually stayed within 20 km of the Zambezi River during the rains (Dunham, 1986).As elephants become concentrated close to sources of water they begin to cause serious damage to vegetation. Baobab trees become easy targets for elephants during the dry season.

Mana Pools National Park has been surveyed (aerial) on eight occasions since 1980 (Table 1.1) and numerous ground counts especially for northern Mana Pools (Jarman, 1968; 1972; Dunham, 1994, Dunham and du Toit, 2012). The density for the mana alluvium has shown an increase for this period (1980-2003) from 1.7 to 3.1 elephant km^{-2}. Only 2 strata were sampled during the period 1980 to 1989 and were increased to 3 during the 2001 and 2003 aerial surveys (Table 1.1). Mana Pools I stratum is in the northern part of Mana Pools National Park where baobabs for the Sapi study site were sampled (Table 3.1). Mana Pools II is in the central

part of the park and this is where baobabs for Fourways study site where sampled and Mana Pools III is in the southern part of the park where the Rukomechi study site is located.

However large mammals on the floodplain during April and October 1965 and 1966 were counted by Jarman (1968; 1972). Dunham (1994) repeated large mammal counts in the Zambezi alluvial woodlands and on the adjacent ecotone in the north of Mana Pools National Park from April 1989 until October 1989 using similar methods used by Jarman (1968; 1972). He found that elephant density declined after a management cull. Elephants were culled on the alluvium during 1985 and in the south of the park in 1988 (Dunham, 1994).

Dunham (1994) has reported that density on the floodplain in October decreased from 1981 to 1982, but increased during the 1984 drought. Density declined to $3 \cdot 8$ km^{-2} in 1985-87, after a cull of 500 elephants (mainly cow herds) near the study area early in the 1985 dry season. Despite the cull, it was noted that density on the alluvium during October 1988 was high. On the ecotone, density during October increased from $0 \cdot 6$ km^{-2} in 1981-82 to $1 \cdot 0$ km^{-2} in the 1983-84 drought, before dropping after the cull to $0 \cdot 4$ (0.08) km^{-2} in 1985-89. Dunham (1994) reported that the density of elephant cows and young on the floodplain during October declined from $3 \cdot 7$ km^{-2} in 1981-84 to $1 \cdot 6$ km^{-2} in 1985-87. Thus, the cull removed about 57% of the cows and young which regularly used the alluvium at the end of the dry season.

Table 1.1: Elephant density estimates for Mana Pools National Park, for the period 1980-2011.

Year	Mana Pools I	Mana Pools II	Mana Pools III
1980	1.71[*]	0.55*	
1982	1.44[*]	0.63[*]	
1983	2.04[*]	0.62[*]	
1984	1.59[*]	0.16[*]	
1986	1.70[*]	0.49[*]	
1989	2.30[*]	0.77[*]	
2001	3.08[**]	1.61[**]	1.02[**]
2003	2.74[***]	1.76[***]	1.95[***]
2011	8.11*		

Sources:

[*]Elephant Census in Zimbabwe 1980-1995 an analysis and review (Price Waterhouse).

[**]Zambezi Valley Survey Report 2001, (Mackie, 2002).

[***]Zambezi Heartland Report (2003) AWF, (Dunham, 2004).

*Dunham and du Toit (2012).

1.1.3 Elephants and baobabs in Zimbabwe

In Zimbabwe elephants and baobabs have been studied in Gonarezhou National Park (GNP) and Mana Pools National Park. Swanepoel (1993b) noted that there was an absence of written records on elephant baobab damage in Mana Pools National Parks records dating back to 1960. In Mana Pools National Park it was observed that elephant foraging behavior is a distinctly seasonal phenomena related to the position of trees relative to perennial water (Swanepoel, 1993b). A significant mortality of baobab trees was caused by elephant damage between 1984 and 1988. Swanepoel (1993b) concluded that by the year 2000 the baobab population studied will be halved if annual mortality continues at 7.3%. He recommended that reducing elephant population was the only way to ensure the long term survival of baobab trees, especially those growing to areas close to perennial water. The purpose of the present study, therefore, is to repeat the survey conducted in 1984 (Swanepoel and Swanepoel, 1986; Swanepoel, 1993b), either to confirm them or to revise them.

In GNP, Mpofu *et al*. (2012) recently determined the abundance, distribution and status of baobabs in three land categories namely, (i) plains, (ii) riverine and rocky outcrops, and (iii) development areas. It was concluded that elephants and possibly fire among other factors may be influencing baobab structure, abundance and distribution in southern GNP.The impact of African elephant on baobab population structure in northern GNP, southeast Zimbabwe, was assessed between 2008 and 2012 (Kupika *et al*., in press). It was found that baobab density and size class distribution did not significantly differ across study sites. However, significant difference in baobab damage was recorded between study sites.

1.1.4 Study species

1.1.4.1 Taxonomy, botanical and morphological description

Several authors have described the African baobab tree (Wild, 1961; Robyns, 1963; Villiers, 1975; Coates-Palgrave, 1983; Wickens, 1982; Baum, 1995a; 1995b). The African baobab belongs to Equisetopsida class, subclass Magnoliidae, superorder Rosanae and of Malvales order. The African baobab and its related species belong to the subfamily Bombacoideae of the family *Malvaceae*, a family which includes about 200 genera and 2300 species (Baum, 1995b). *Adansonia digitata* L. is one of eight species of *Adansonia*.

A. digitata was first described by Adanson in 1771. The African baobab is mostly known for its exceptional height and girth. The trunk tends to be swollen and stout and can reach a diameter of up 10m and usually bottle shaped (Sidibe and Williams 2002). Baobabs are characterized by palmately compound leaves. The trunks consist of soft, fibrous wood that can store water. The leaves of juvenile trees are simple and gradually change to 5-7 foliate compound leaves as the tree gets older. Flowers are borne in the axils of leaves and comprise a single, large, odoriferous white flower made up of both male and female reproductive parts. The fruits are large, ovoid, and covered in a yellow/green velvety indumentum. The pericarp is woody and indehiscent. Seeds are reniform, embedded in a soft dry matrix (Baum, 1995b).

Baobab flowers are white and large, 12 cm across, have 5 petals that are hairy inside and are generally leathery; the sepals are cup-shaped and 5-cleft; the stamens divide into multiple anthers and the styles are long and 7-10 rayed. The flowers rarely have a life span of more than 24 hours and are pollinated by bats, insects, and wind (Gebauer *et al*., 2002, Sidibe and Williams 2002). The fruits are variable in how they look, but tend to be ovoid and covered with velvety hairs; the pulp is dry and mealy, contained within a woody pericarp (Sidibe and Williams 2002).

The root system of baobabs, while shallow, spreads further than the height of the tree, contributing to its ability to survive in dry climates. The range of the shallow root system allows the trees to collect and store massive amounts of water during the heavy, but infrequent rainfalls, which they then use to photosynthesize in the trunk during the 8 months in which they are leafless. This species is found to be among the most effective trees at preventing water loss. The tree tends to grow in sandy-textured soils, but can be found on rocky hillsides or in places where there is runoff. Every part of the African baobab tree has been used by humans for multiple purposes, including medicinally and nutritionally, however it is not widely cultivated (Ebert *et al*. 2002).

1.1.4.2 Distribution of species *Adansonia digitata*

In Africa baobabs are found mostly in the drier plant communities of the Sudano-Zambesian lowlands where annual rainfall is 200-800 mm annually (Wickens, 1982). In southern Africa, baobabs are associated with *Colophospermum mopane*, *Cordyla* and *Kigelia* woodlands in low lying, dry, hot and frost free areas. They occur on a variety of soil types, but seem to do best on deep, well-drained soils (Wickens, 1982).

Adansonia digitata occurs in semi-arid areas, Africa south of the Sahara (Figure 1.1), where annual rainfall is in the range of 150 to 1500 mm (Fenner, 1980). It is also found from sea level up to areas 1250 in altitude (Fenner, 1980). Its range extends across the African continent south of the Sahel region, also into East, Central and Southern Africa (Fenner, 1980).

Figure 1.1: Distribution map of *A. digitata* (source: Wickens & Lowe, 2008).

1.1.4.3 Phenology of *Adansonia digitata*

In Mana Pools National Park, baobab phenology was studied by Swanepoel (1993a). He found that baobabs are generally deciduous, with the appearance of leaves being correlated with season. He noted that the African baobab revealed that leaf buds began to activate before the first rains, and generally took approximately a month to fully develop (Swanepoel 1993a). However, the rate of development slowed if rains were poor, suggesting a dependence of leaf development on rainfall.

The flowers of the baobabs in Kenya open shortly after dusk, and last only 12 hours (Wickens, 1982). The flowers are large, white and pendulous, and open very rapidly (Fenner, 1980). The stamens are numerous, being born on a stout column, and dehisce from one-celled anthers shortly after the flower opens (Fenner, 1980). Fruit bats (*Rousettus aegyptiacus*) were observed feeding on the open flowers, and are considered a significant pollination agent of the baobab on the Kenya coast (Fenner, 1980). Bush babies (*Galago crassicaudatus*) were observed feeding on newly open flowers near Mombasa, Kenya, and systematically moved from flower to

flower burying their faces within the flowers and around the sepals (Fenner, 1980). They can be considered as pollinators.

According to Palmer and Pitman (1961), seeds apparently keep their viability for years. They are reported to germinate well under nursery conditions on moist conditions. In their natural condition, baobabs are believed to germinate in exceptionally good rainy seasons. Young trees soon develop a swollen underground organ for storing water, from which the taproot and side roots develop (Palmer and Pitman, 1961). Attempts to reproduce baobabs vegetatively from truncheons have failed and it appears that planting by seed is possibly the only way of propagating the baobab.

1.1.4.4 Ecological significance of *A. digitata*

The baobab acts as an important source of food, water (during drought) and shelter, for a wide range of animals (Fenner, 1980). Baobabs are known for their associated fauna, of special insects but also bats and birds use baobabs. Hollows in baobab trunks provide shelter to leopards, genets, porcupines, west African lesser bush baby (*Galago senegalensis*) and the greater bush baby (*Otolemur crassicaudatus*) (Cashel 1995). The stripped hyena (*Hyaena hyaena*) also use baobab hollows to rear its young (Sweeney, 1973).

Wickens (1982) notes that baboons eat baobab fruits, either eating the young soft fruit or breaking them open when they are ripe. The tree also provides edible fruits and leaves and is also a source of fibre (Fenner, 1980). Due to their great size and bizarre shapes, baobabs are often the most prominent tree species where they occur and form the subject of many legends (Fenner, 1980).

1.1.4.5 **Socioeconomic importance of** *A. digitata*

Adansonia digitata has many uses medicinally and non-medicinally and every part of the plant is reported to be useful (Gebauer *et al*., 2002). The leaves, for instance, are used in the preparation of soup. Seeds are used as a thickening agent in soups, but they can be fermented and used as a flavouring agent, or roasted and eaten as snacks (personal observation). The pulp is either sucked or made into a drink while the bark is used in making ropes. The different parts of the plant provide food, shelter, clothing and medicine as well as material for hunting and fishing (Venter and Venter, 1996; Sidibe and Williams, 2002). Baobab tree provides income and employment to rural and urban households. The bark is formed into strong fiber that is used for all sorts of textiles, while the wood is used as fuel (Gebauer*et al*. 2002). In Zimbabwe the baobab craft industry is an important source of rural employment for instance in the Save-Odzi valley, especially among women and young people (Romero, 2001).

1.2 Research Problem

Elephants can have major impacts on vegetation in the habitats in which they occur. Baobabs are often targets of elephant foraging. Elephants can damage and even kill these trees. The damage elephants can inflict on baobabs is well known from sites throughout the range of this species in Africa (see Robertson-Bullock, 1960; Napier Bax and Sheldrick, 1963; Laws, 1969; Caughley, 1976; Barnes, 1980; Geerling, 1983; Swanepoel and Swanepoel, 1986; Swanepoel, 1993b, Edkins*, et al*., 2007; Mpofu *et al*., 2012).

A cause of concern in Mana Pools National Park, Zimbabwe, has long been the lack of regeneration of some tree species, notably *Faidherbia albida* (Del) (Dunham, 1989a; 1989b) and *Adansonia digitata* (Swanepoel and Swanepoel, 1986; Swanepoel, 1993b). Swanepoel (1993b)

predicted that by the year 2000, the baobab population studied in Mana Pools National Park would be halved if annual morality continues at 7.3%. Thus, he recommended that reducing the elephant population may be the only way to ensure the long term survival of the baobab trees, particularly those growing in areas close to perennial water.

The present study seeks to understand the effect of elephant browsing and herbivory on *A. digitata* in the Mana Pools National Park.

1.3 Objectives of the study

1.3.1 The primary objective of this study was to estimate the level of elephant induced baobab mortality in Mana Pools National Park and to compare current levels of elephant-induced mortality with those found in 1984 by Swanepoel.

1.3.2 The specific objectives were to determine elephant induced baobab damage in relation to the distance from the Zambezi River a major source of perennial water.

1.4 Research questions

The study attempted to answer the following questions:

a. Does the current level (2005) of baobab mortality and density in northern Mana Pools National Park differ from that experienced during 1984?

b. Does the level of baobab damage due to elephants differ between the three study sites of different distances from Zambezi River?

c. Does the size structure of three baobab population differ in relation to distance from the Zambezi River?

d.   Does the density of baobab differ in the three study sites in Mana Pool National Park?

1.5 Justification of the study

Managing the alleged impact of elephants is riddled with ethical, political, economic, and ecological implications and is often motivated by information from studies carried out elsewhere (Guldemond and van Aarde, 2008). Inferences made from elephant impact studies could be complicated and confounded by poor replication, lack of suitable controls, confounding environmental factors, and response variables that are incorrectly assigned to elephants (Guldemond and van Aarde, 2008). Thorough scientific information on which to evaluate elephant impact and to motivate their management is lacking (Owen-Smith *et al.*, 2006; van Aarde and Jackson 2007). This paucity of information has made it difficult for park managers to come up with informed management decisions.

The purpose of the present study, therefore, was to verify the validity of results of surveys conducted in 1984 by Swanepoel (1993b). Although elephant impacts on baobabs have been studied before, there is need to establish how these impacts are distributed over space and under local conditions. The increasing elephant population (Table 1.1) in northern Mana Pools National Park has been a cause for concern. The Park was designated a world heritage site (UNESCO) in 1984. Loss of biodiversity is likely to reduce the wilderness value of the park and its economic value as a tourist destination. Loss of baobab trees in Mana Pools National Park is likely to result in overall reduction in species richness.

1.6 Study Area

1.6.1 Location

The Mana Pools National Park is part of the mid-Zambezi Valley region, which is located in the north of Zimbabwe, between latitudes $15^0\,40'$ to $16^0\,20'$S and longitudes $29^0\,08'$ to $29^0\,45'$ E. The middle or mid-Zambezi valley normally represents that stretch of the Zambezi between the Victoria Falls and Cabora Bassa Gorge. Mana Pools and Chewore Game Reserves were proclaimed in 1963. Final Boundaries were gazetted in 1968 (Kerr and Fraser, 1975).

Mana Pools National Park covers 2196 km^2, and is bounded to the west by Hurungwe Safari area-Nyakasanga section (2878 km^2) and to the east by Sapi (1180km^2) and Chewore safari areas (3390km^2). Mana Pools National Park is, therefore, part of more than 10000 km^{2}of wildlife land under the jurisdiction of Zimbabwe Parks and Wildlife Management Authority (Figure 1.2). To the north, the park is bordered by the Zambezi River which forms the international boundary with Zambia. Opposite Nyamepi camp (Figure 1.2) at the Chongwe River in Zambia, starts the Zambia Game Reserve. Mana Pools National Park is bordered by Mukwichi Communal Land to the south of the escarpment.
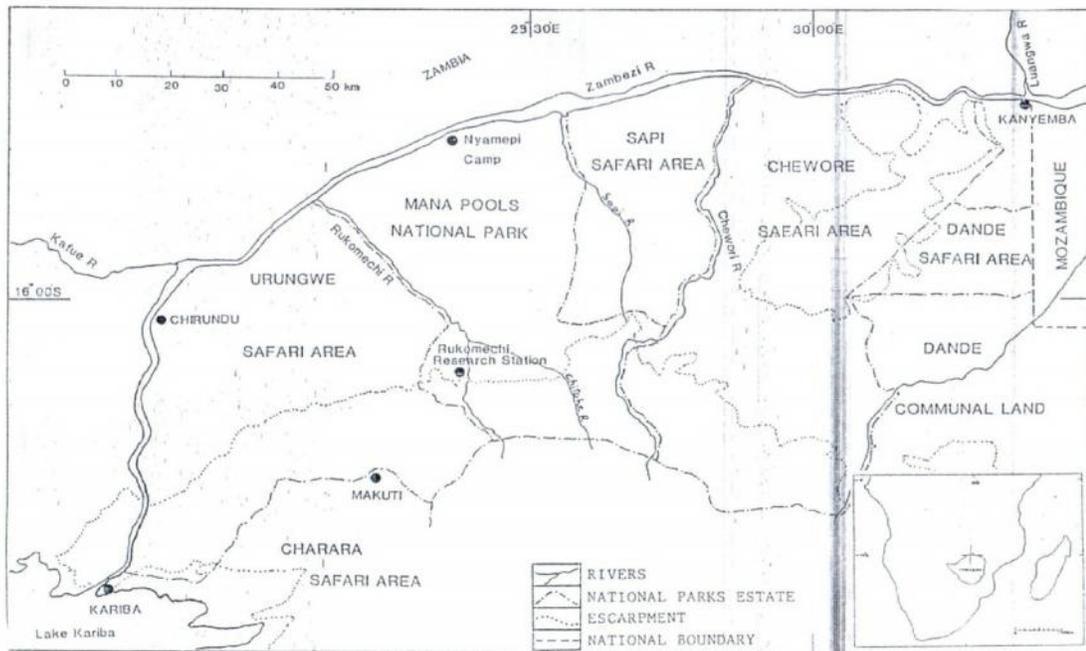
Figure 1.2: Geographic location of Mana Pools National Park within the lower mid-Zambezi valley in northern Zimbabwe (Swanepoel, PhD thesis).

1.6.2 Geology and Geomorphology

The Zambezi Valley is a down-faulted valley with main fault-line running along the length of the escarpment (Guy, 1970). Bennett, Anderson and Brinn (1985) noted that the valley floor within the Mana Pools National Park is almost underlain by Karoo rocks, products of fluviate and Aeolin depositions between the Permian and the Jurrassic (about 280-135 million years B.P.). The sediments are faulted and tilted to give a gently undulating topography.

1.6.3 Soils

Mandinyenya (2004) reported on the relationship between soil type and vegetation type. The soils of the Zambezi valley floor are described by Bennett, Anderson and Brinn (1985). Soils covering much of the southern section of the valley floor (i.e., covered by Mopane vegetation) are derived from karoo sandstones. The surface horizons are mainly fine or medium grained sandy loams, and are often sodic. They cover compact, largely impervious, strongly alkaline horizons. In the northern half of Mana Pools National Park are deep fine to medium-grained sands, often reddish, derived from aeolian sandstones. These soils ("Jesse-soils') are acidic, with a low available water capacity.

Colluvial deposits (fairly deep loamy sands overlying sandy clay or clay loams), occur between the Chitake River and the escarpment. Alluvial deposits are restricted to the larger rivers (Rukomechi and Sapi) and are especially pronounced and older along the Zambezi, forming the mana pools "floodplain", covered by what is referred to as Zambezi riverine vegetation. The gneissic derived soils on and above the escarpment are generally shallow, medium-grained lithosolic sands (Thompson and Purves, 1978).
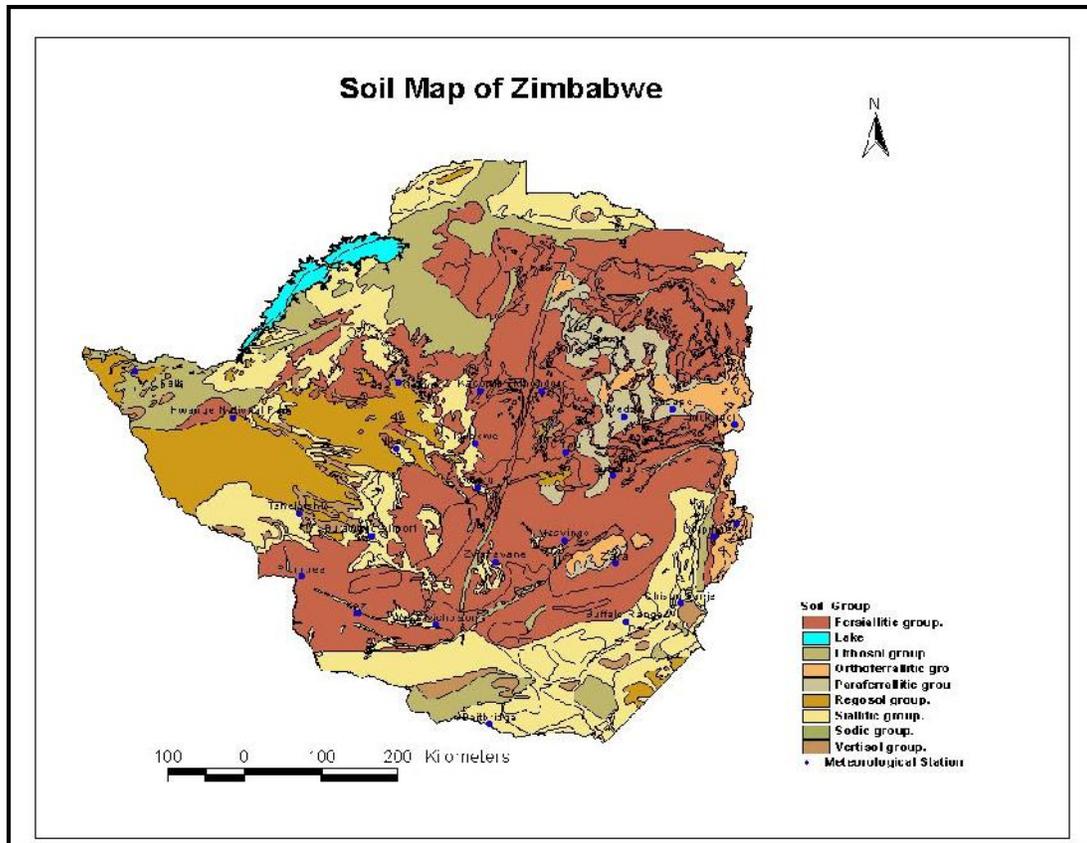
Figure 1.4: Distribution of major soil groups in Zimbabwe (Source: Department of Surveyor General 1979)

1.6.4 Hydrology

Of the Zambezi tributaries flowing through the Mana Pools National Park, only Rukomechi and its tributary, the Chitake, rise in the escarpment. Perennial water, apart from in the Zambezi, is available only from springs and seeps associated with the streams and rivers in the escarpment. Surface water on the Valley floor occurs only in pans, filled by rainfall runoff and these pans normally dry by mid-dry season.

The Zambezi River streamflow was disrupted by the building of Kariba Dam in late 1958. Details of the past and recent hydrology of the Zambezi are given by Nugent (1988). Before the dam was built, the flow of the Zambezi River at Mana Pools gradually increased through the wet season, to peak in about May with the arrival of floodwaters from the upper

catchment in Barotseland. Superimposed on this gradual rise were series of brief but intense flood peaks, representing flash floods in the lower catchment. Neither these peaks nor the maximum flow were enough to regularly inundate the alluvium terraces. After the Kariba dam was built, streamflow was regulated by baseflow through the turbines and by artificial flooding created by the opening of floodgates. The baseflow has increased as more turbines were commissioned. The effect on the Zambezi has not been to degrade its bed to any great extent (due to controlling bars of hard rock crossing the channel at various points), but rather to cause major widening (increasing by approximately 200 m), resulting mainly from the constant baseflow, broken only by sudden flooding when the floodgates are opened (see also Guy, 1980, 1981). During the period from June 1981 to June 1988, no floodgates were opened and the channel changed little over this period. This artificial stability has resulted in a situation whereby sandbanks normally seasonally inundated by floodgate waters have now been stabilized by vegetation; similarly, shallower river channels have begun to close due to vegetation build-up and siltation (*Swanepoel C.M., personal communication*).

1.6.5 Climate

Within the mid-Zambezi Valley region climatological observations by the Department of Meteorological Services have only been made at Chirundu (close to the Zambezi river), and Rukomechi Research Station (near the base of the escarpment). Rainfall records have been kept at Mana Pools National Park (Nyamepi Camp) since 1967, and screen equivalent maximum and minimum temperatures since 1981. These data show that rainfall is usually confined to the period from November to March (the hot wet season). Figure 1.4 shows annual rainfall recorded at Rukomechi Research Station between 1983 and 2002.
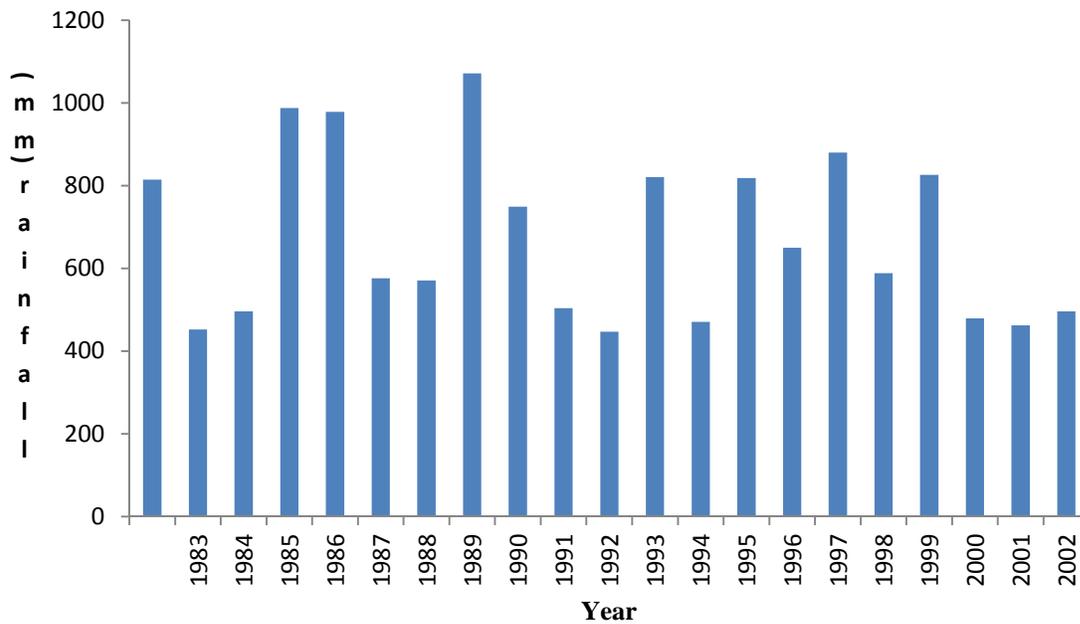
Figure 1.4: Annual rainfall (mm) recorded at Rukomechi Research Station, Mana Pools National Park (Department of Meteorological Services).

A cool dry season lasts from May to August, followed by increasingly hot, dry weather until the rains break. The coolest month is July (Figure 1.5), but depending on the rainfall distribution, mean monthly maximum temperatures above 40°C have been recorded in November, December and January (Department Meteorological Services).
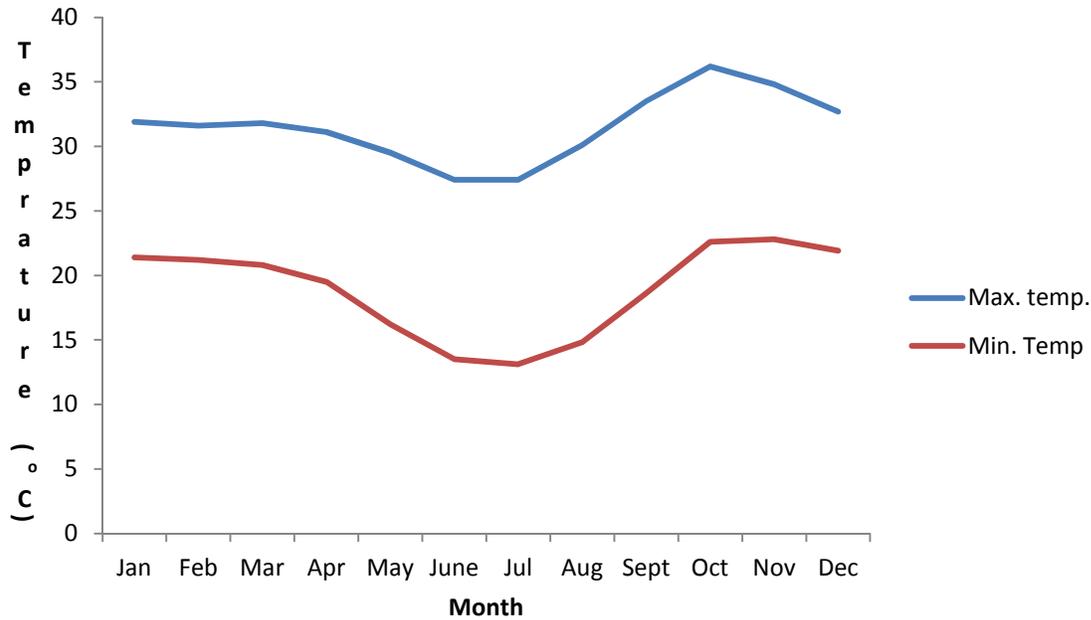
Figure 1.5: Mean monthly temperatures (^{0}C) at Rukomech Research Station, Mana Pools National Park (Department of Meteorological Services).

1.6.6 Vegetation

The first vegetation classification and descriptions of the study area were done by Wild (1961). The second was done by Guy (1977) who recognized seventeen vegetation types. Muller and Pope (1982), however, found these to be inadequate, and in their study, they recognised twelve vegetation communities as follows;

   i.   Riparian vegetation

Riparian vegetation exists on the alluvial areas alongside the larger rivers which support *Acacia albida* woodlands.

   ii.   Mixed species layered dry forest (Jesse-Bush)

This is found on the jesse-soils which are deep sands rich in both tree and shrub species. Large areas of dry deciduous *Commiphora-combretum* thickets (jesse-bush) exist.

    iii.   *Colophospermum mopane* woodland

Well grown *Colophospermum mopane* woodland is the most extensive vegetation type, with the majority of the valley floor covered by medium or tall *Colophospermummopane* savanna woodland. *Adansonia digitata* is common in *Colophospermum mopane* woodland and mixed species woodland.

    iv.   *Colophospermum mopane* woodland with understory

This is common in areas adjacent to jesse-bush or to riparian communities.

    v.   Scrub*Colophospermum mopane*woodland

This occurs essentially on the edges of the Zambezi alluvial system between densely wooded ridges in the Chewore area. Trees are only up to 3 m in height.

    vi.   *Combretum/Terminalia/Diospyros kirkii* wooded grassland

    vii.   *Colophospermum/Kirkia/Terminalia prunioides* woodland on ridges

    viii.   Mixed species woodland on larger hills

    ix.   *Acacia robusta/Terminalia prunioides* woodland

    x.   Mixed *acacia/Lonchocarpus capassa* woodland

    xi.   Vegetation associated with alkaline pans

    xii.   Grasslands

The vegetation of the Zambezi valley displays very close relationships between vegetation types and soils (Guy, 1977). Because of this close relationship it is not difficult to see why the structure within each vegetation type changes over short distances where pockets of soils unlike those surrounding them occur. Differences in soils no doubt account for the very

heterogeneous structure of the *Colophospermum mopane* woodland with its many different associated species resulting in considerable variations in physiognomy over relatively short distances. Because of this marked variation in the physiognomy of the dominant species of the largest vegetation type (mopane) within comparatively small areas, it is difficult to apply any one classification system satisfactorily.

2. LITERATURE REVIEW

2.1 Elephant-vegetation interactions in savannah ecosystems

The African elephant is the largest herbivore species in the savannah ecosystem and therefore have a significant impact on vegetation (Guldemond and van Aarde, 2008). As African savanna elephants become increasingly confined to smaller fragmented landscapes coupled with their increasing densities, concern over their potential detrimental impacts on vegetation and biodiversity has increased (Loarie *et al*., 2009). Understanding elephant vegetation preferences across relevant spatial and temporal scales is a critical step towards managing protected areas for the persistence of both elephants and biodiversity (Loarie *et al*., 2009).

The impacts of elephants, in combination with fire, have been suggested as being important in converting the vegetation structure of the many protected areas in east and southern Africa (Laws, 1970; Dublin, Sinclair and McGlade, 1990; Ben-Shahar, 1993; Tafangenyasha, 1997; Eckardt, van Wilgen *et al*., 2000; Mapaure and Campbell, 2002; Augustine and McNaughton, 2004). In particular, where elephant populations are high, tree-dominated savannas are converted to a grass-dominated state. The African elephant is one of the principle drivers regulating shifts between savannah states. Elephants can independently impede woodland formation (Western and Maitumo, 2004; Holdo, 2007), raising concerns that increasing elephant populations may lead to a loss of biodiversity in protected areas (Cumming *et al*., 1997; Lombard *et al*., 2001; Botes *et al*., 2006; Kerley and Landman, 2006; Van Aarde *et al*., 2006; Hayward and Zawadzka, 2010).

Elephants supplement grass and browse intake with bark in the dry season (O'Connor, Goodman and Clegg, 2007; Ihwagi *et al.*, 2010). Ringbarking, or girdling, kills trees, which can no longer transport sugars (Noel, 1970).African elephants preferentially browse on trees such as

baobabs and Acacia trees (*Acacia spp*) by stripping the bark, removing foliage, uprooting the tree and breaking branches. This foraging behaviour has been linked to the decline of woody vegetation in African game reserves (Edkins *et al.*, 2007; O'Connor *et al.*, 2007; Gandiwa *et al.*, 2011).

Elephant vegetation interactions are complicated by human modifications to the landscape. Networks of water holes now expand the areas where elephants are found which will affect well-wooded areas in the dry season (Chamaille-Jammes *et al.*, 2007; Grainger *et al.*, 2005; Smit *et al.*, 2007; Van Aarde *et al.*, 2008). These waterholes may enable elephants to reach browsing habitats that were out of reach in the dry season. Moreover, away from water, these habitats would be passed over in favour of grasslands and shrublands in the wet season. Fences may interfere with elephant vegetation preferences to concentrate elephant feeding, which likely increases their impact on the vegetation (Grant *et al.*, 2011). Potentially more importantly, they may act as barriers to these seasonal movements and their associated vegetation shifts, forcing elephants to utilize the same landscape year-round. This could be especially important in relatively small protected areas and ones designed to represent a few special habitat types.

Landscape heterogeneity is extremely important for maintaining biodiversity as it provides variation in shelter and food resources (Loarie *et al.*, 2009). Recent evidence suggest that when elephants are at densities within the carrying capacity of a given habitat, they actually increase browse heterogeneity (White *et al.*, 2001a; 2001b; Kohi *et al.*, 2011), suggesting that the key to successful elephant management is to ensure that population density remains at a level where elephant impact on vegetation remains a natural part of the ecosystem rather than causing widespread vegetation extinction.

In a meta-analysis study, Guldemond and Van Aarde (2008) reported that most elephant vegetation selection evidence comes from localized studies from single sites. If elephant vegetation preferences vary across precipitation gradients, we would expect different localized studies to yield different results. Likewise, other factors that trade-off with herbivore foraging choices such as predator avoidance (Kie, 1999) and water availability (Redfern *et al*., 2003) vary geographically. They noted that separating the influence of these factors from the influence of vegetation greenness requires a large study area with diverse combinations of landscapes.

2.2 Elephant distribution in relation to surface water

Elephant distribution is regulated by water availability (Redfern *et al*., 2003; Chamaille´-Jammes, Valeix & Fritz, 2007), with herds staying <10 km from permanent water on average during the dry season (Stokke and du Toit, 2002; Loarie *et al*., 2009). Increased utilization of vegetation around water sources by elephants has been seen in multiple protected areas across Africa (Afolayan, 1975; Brits, Van Rooyen and Van Rooyen, 2002; De Beer *et al*., 2006; Gaugris & Van Rooyen, 2010), but the causes underlying heterogeneity in impacts remain poorly understood (Chamaille´-Jammes, Fritz and Madzikanda, 2009). A better understanding of the spatial heterogeneity of elephant impacts on vegetation will improve management of plant biodiversity as well as other large herbivore species, which often cluster around water in semi-arid savannahs and may be greatly affected by piospheres (Chamaille´-Jammes, Fritz and Madzikanda, 2009).

Other studies have also shown the tendency of elephants to concentrate close to water source for example in Rwenzori National Park and Chambura Game Reserve, Uganda (Eltringham, 1977) elephant distribution tended to be more aggregated in the dry than wet seasons. In the Ruaha-Rungwa area of southern Tanzania elephant densities were higher close to

water (Barnes and Douglas-Hamilton, 1982). In Luangwa valley in Zambia elephant cows were observed to have a distribution pattern probably influenced by food abundance and quality and distribution of surface water in the dry season (Caughley, 1976a). In Kidepo Valley National Park (Uganda) it was observed that at Pirre, a dry area, elephant occurred in largest numbers when the Park was relatively wet and elephants were not seen to use this area during the dry seasons, probably due to lack of surface water.

In Mana Pools National Park, the densities of large herbivores were reported to be high, based on citizens counts recorded during the late dry season in an area of the park where animals are known to congregate and as such the densities are not typical of the remainder of the park (Dunham and du Toit, 2012). The densities of most species were similar to the maximum densities recorded in the same area during the 1980s (Dunham, 1994), but the 2011 densities for impala, eland and kudu were greater, and more similar to those recorded during the 1960s (Jarman, 1972). These high densities means vegetation damage is high in northern Mana Pools National Park due to herbivory pressure. Elephant density was reported to be 8.11 km^{-2} and Impala density was 203.8 km^{-2}.

2.3 Impact of elephants on baobabs

2.3.1 Elephant and baobabs in protected areas

Robertson-Bullock (1960) was the first to report elephant eating and destroying baobab trees in the Luangwa valley of Northern Rhodesia. During this period out of thirty-eighty baobab trees that were examined, only two had not been badly scarred by elephant. In Tsavo Royal National Park (East) of Kenya, Napier Bax and Sheldrick (1963) found that in the dry season bark and leaves of bushes and trees became important food source for elephant. In this study it was found

that grasses, creepers and herbs, when available, form the bulk of elephants' diet even in a normal dry season. Damage to woody vegetation reached a peak in periods of drought, when the herbs and grasses disappear or have a very low nutritional value, and trees and shrubs maintain a relatively high food value.

Large-scale destruction of trees by elephants and the resultant conversion of woodland into open or wooded grassland have been observed in Tsavo National Park (Glover, 1963; Napier Bax and Sheldrick, 1963; Laws, 1969). Laws (1969) estimated a baobab mortality of 12 % over 2 or 3 years in Tsavo East National Park. In 1977, Leuthold reported that baobab trees had been nearly exterminated by elephants and few baobab individuals which were found, presented too small a sample for any analysis.

In Ruaha National Park (Tanzania) tree surveys made in 1976 and 1977 were repeated in 1982. The three species measured were *Commiphora ugogensis*, *Acacia albida*, and *Adansonia digitata*. All three showed significant decreases due to elephant browsing. The baobab was found to have declined at a higher rate than predicted by the 1976 survey.

Barnes (1980) in his study in Ruaha National park showed that with the present rate of destruction, assuming no further regeneration and a constant elephant density the baobab population in the park may be eliminated in 30-170 years. Dry season elephant density was estimated to be 4.6 per km^{-2} when Barnes (1980; 1983a; 1983b) investigated the effects of elephants on three tree species between 1976 and 1982. Baobabs were particularly plentiful at the beginning of the study but the density of baobabs had decreased by 45% by 1982, with no young trees.

In his study Barnes found only ten individuals in the 0-24 year age class. Conversely in a healthy population the regeneration class would be expected to be the largest class. There was a

decline in recruitment probably due to a decrease in fruiting or regeneration rates or to an increase in the mortality rate of this class. Whilst the factors, which influence seeding, fruiting, or germination in baobabs, were not known, there was some evidence concerning mortality in this class. It was evident that elephant density had increased since the park was gazetted. Elephants appeared to kill a higher proportion of younger trees and the gap in the youngest class was therefore probably due to the increasing elephant density.

The change in size distribution reported by Barnes (1980) confirmed Caughley's (1976b) observation that elephants select young baobab trees. Guy (1971) suggested the decline in the number of baobab trees might be attributable to desiccation and a decrease in rainfall but there was no evidence at Ruaha to support this hypothesis. There were no baobabs observed to show signs of desiccation, and rainfall records in Ruaha National Park did not show long-term decrease in annual rainfall.

Based on these studies Barnes (1980, 1983a; 1983b) predicted the elimination of baobabs in a section of Ruaha National Park. However in 1989 the baobab population had started to recover in spite of relatively high elephant densities, probably because virtually all adult males had been killed by poachers (Barnes *et al.*, 1994), supporting the observation that males are primarily responsible for killing baobabs.

In the Serengeti National Park elephants were first recorded in 1955 and were probably absent for at least the previous 40 years (Lamprey *et al*., 1967). In this study it is reported that increases through immigration caused vegetation changes and destruction of the larger trees at the rate of at least 6 % per year. In the past elephants had vast areas of country to occupy, but under present conditions, their range has been severely restricted so that they tend to be confined to the limited areas of National Parks and Reserves.

Caughley (1976b) reported that elephants find young baobabs irresistible, but do not restrict themselves to eating young baobabs, but also fell mature trees with a frequency that decreases progressively with the size of the tree. Caughley (1976b) using an age estimate proposed that a stable cycle of approximately 200 years exists between elephants and forests in eastern and southern Africa. However, Guy (1971) recommended that baobabs not be used in the determination of links in time between baobabs and elephants due to their pronounced fluctuations in growth rate. He suggested that some other tree species or method be used unless it can be shown that decreases in girth have not occurred.

In Tanzania's Lake Manyara National Park, during the dry season, elephants were observed to eat the bark and soft pulpy wood of baobabs. By gouging out the wood with their tusks, they can excavate large cavities in the trunk. If the cavity becomes large enough, it can result in the collapse and death of the tree (Weyerhauser, 1985). At Lake Manyara in Tanzania, damage to baobabs was compared for surveys conducted in 1969 and 1981. Damage and mortality increased over the 11-year interval (Weyerhauser, 1985). It was found that while bark damage increases in large size classes, the opposite was true with elephant induced mortality. All the trees killed by elephants in the Lake Manyara survey were in the smaller size classes.

Swanepoel (1993b) found that elephant foraging on baobabs is distinctly seasonal phenomena related to the position of the trees relative to perennial water. Although elephants concentrated their feeding on individuals trees, in Mana Pools National Park no significant prevalence of damage in any particular size of baobab was found. Studies in other National parks have however suggested that elephants preferred to feed on smaller baobab trees (e.g. Caughley, 1976b; Barnes, 1980; Weyerhauser, 1985).

The impact of elephants on baobabs was also studied in Kruger National Park by Whyte (2001) in two areas with different histories of elephant occupancy. Where elephants had been present for longest, a significantly higher proportion of trees showed severe damage, smaller size classes were poorly represented in comparison to an area that had been occupied more recently. At a control site with no elephant, small size classes formed a higher proportion of the population than at either site with elephants. Actual mortality was highest during a period of low rainfall between 1981 and 1984, suggesting that damaged trees were more liable to die from other stress factors. These impacts occurred even at low elephant densities.

The abundance, distribution and status of baobabs in three land categories namely, (i) plains, (ii) riverine and rocky outcrops, and (iii) development areas, in southern Gonarezhou National Park (GNP), southeast Zimbabwe, were determined. Baobabs were sampled between April and August 2010 using transects along existing roads and the Mwenezi River. Height, basal circumference and elephant damage for each baobab tree was measured. A total of 117 baobabs were sampled using 17 transects with a combined length of 238 km. Mean baobab density was significantly higher in the development areas as compared to the plains, riverine and rocky outcrops. However, there were no significant differences in mean diameter at breast height and height for baobab trees across the three land categories. Elephants and possibly fire among other factors may be influencing baobab structure, abundance and distribution in southern GNP. Baobab densities in southern GNP do not seem to indicate that baobabs are in danger of extirpation.

2.3.2 Baobab density

Baobab densities are very variable in the landscape in which they occur. They are affected by a number of factors, such as soil requirements, competition for water (related to baobab's extensive root system), baboon-mediated seed dispersal, elephant populations and human settlements (Sidibe and Williams 2002, Duvall, 2007, Edkins *et al*., 2007, Wickens and Lowe, 2008).

Diverse population densities have been reported in different countries and in a number of land-use types (Table 2.1). One reason might be the different methodologies used to estimate tree density (Wickens and Lowe 2008). In some studies outside protected areas it was found out that generally baobab density is higher in cropland than in fallows or grazing land as seedlings are protected from fire and grazing (Johansson, 1999; Dhillion and Gustad, 2004; Venter and Witkowski, 2010). Baobab density might also be higher in cropland as farmers are interested in maintaining this tree species. Boffa (1999) reported that farmers tend to reduce tree density and the number of species and favour preferred species in the agroforestry systems of West Africa.

Table 2.1: Baobab densities in some parts of Africa sub-Sahara

Country	Site	Land-use type	Estimated density km^{-2}	Source
Benin	Northern part	Traditional agroforestry system	1-5	Assogbadjo *et al.,* 2005b
Burkina Faso	Sudano-Sahelian zone	Traditional agroforestry system	6	Kyndt *et al.,* 2009
Ghana	Sudano-Sahelian zone	Traditional agroforestry system	4	Kyndt *et al.*, 2009
Kenya	Kibwezi district of Kenya	Farmlands and natural woodlands	0-6000 (woodlands) 0-200 (farmlands) a	Mohamed 2005
Malawi	Southern Lake Malawi area	Cropland, fallow and villages	1000[a]	Chirwa *et al.*, 2006
Mali	–	–	10.7	Wilson 1988
	South-western part	Cultivated and non-cultivated land	6.69	Duvall 2007
	Cinzana, central	Cropland, fallow	40-200 a	Dhillion and Gustad

	part (central Nigerian Delta)	and villages		2004
Senegal	Sudano-Sahelian zone	Traditional agroforestry system	7	Kyndt *et al*., 2009
South Africa	Kruger National Park	Protected Area	0.3-32 (different areas of KNP)	Kelly 2000
	KNP	Protected area		Edkins *et al*., 2007
	(northern Venda northern part)	Plains, rocky areas, fields and villages	103 a	Venter and Witkowski 2010
Tanzania	Ruaha National Park	Protected Area	27.6 - 51 (year 1982 and 1976 respectively) b	Barnes 1980, 1994)
	Lake Manyara National Park	Protected Area	49.2 - 72.8 (different areas within protected area)	Weyerhaeuser 1985
	Kondoa Irangi Hills	Cutlivated and grazing land	l7 .5- 50 (different areas)	Johansson 1999
Zimbabwe	Mana Pools National Park	Protected Area	l3.1 - 18.4 (year 1988 and 1984 respectively) b	Swanepoel1993
	Save-Odzi Valley, eastern part	–	840 a	Romero *et al*., 2001
	GNP	Protected area	8.3-21.6 in (different areas in the park)	Mpofu *et al*., 2012

2.3.3 Baobab recruitment and regeneration

Regeneration is a key issue when elephant damage to vegetation is to be assessed and is best measured over a long period of time (Boot and Gullison, 1995). However, an analysis of age or size structure of a population at one point in time can also provide valuable information, as an identification of poorly represented stages of life history could suggest a negative impact of elephants on regeneration. This may be particularly revealing for a slow growing and long-lived species.

Different size class distributions for the baobab tree have been reported by several authors. Bell-shaped size class distributions, with few individuals in small and large size classes,

have been reported from Benin (Assogbadjo *et al*., 2005b), Mali, Kenya and Sudan (Wilson, 1988), Zambia (Caughley, 1976b), Tanzania (Barnes, 1980), South Africa (Edkins *et al*., 2007; Venter and Witkowski, 2010) and Malawi (Chirwa *et al*., 2006). Several authors (Caughley, 1976b; Weyerhaeuser 1985; Swanepoel, 1993b; Barnes 1994; Edkins *et al*., 2007) have shown that in National Parks, the lower number of individuals in the smaller size classes is mainly due to overpopulation of elephants who have the habit of destroying younger baobabs in their search for water during the dry season. Outside National Parks, the lower number of individuals in the smaller size classes is generally attributed to fires, droughts and increased grazing pressure from domestic livestock (Wilson, 1988; Assogbadjo *et al*., 2005b; Wickens and Lowe, 2008).

Reverse J-shaped size class distributions, with more individuals in smaller than in larger size classes, have also been reported for the baobab tree: in south-western Mali (Duvall, 2007), in Mozambique (in Limpopo National Park, where there are few elephants, Edkins *et al*., 2007), in W National Park in Burkina Faso (Schumann *et al*., 2010) and in wooded plains in northern South Africa (Venter and Witkowski, 2010).

However, baobab natural regeneration has been reported to be rather poor (Wickens, 1982). Several authors have noted a lack of recruitment in baobab populations and raised concern about the survival of baobab populations (Romero *et al*., 2001; Swanepoel, 1993b; Assogbadjo *et al*., 2005b; Edkins *et al*., 2007). However, a recent study carried out in South Africa by Venter and Witkowski (2010) pointed out that for long-lived species, such as the baobab tree, recruitment and mortality might be episodic events, as the baobab population in that area was stable and had healthy numbers of mature baobab trees.

Several factors affect baobab recruitment, rainfall and drought being two major factors. Baobab seedlings, which lack the extensive shallow rooting system and the accumulation of

water in the trunk, as such are thought to be more sensitive to droughts than adults (Wickens and Lowe 2008). It is believed that recruitment of baobab seedlings is linked to a series of particularly wet years (O'Connor *et al*., 2007, Wickens and Lowe 2008). Fire is another factor playing an important role in baobab regeneration (Gebauer *et al*., 2002, Edkins *et al*., 2007). Although mature baobab trees have a thick corky bark which enables them to tolerate burning (Greenway and Vesey-Fitzgerald 1969), seedlings and young trees are not tolerant to fire. As mentioned earlier, elephants and other grazing animals such as cattle have an impact on baobab recruitment as they eat and kill baobab seedlings and young baobabs (Wilson 1988; Gebauer *et al*., 2002; Edkins *et al*., 2007; Wickens and Lowe, 2008). Another factor affecting baobab recruitment, as baobab commonly occurs in cultivated land, is whether the tree is desired or not by the farmer (Boffa, 1999).

Because of these factors, it seems that baobab regeneration mainly takes place in association with settlements and cultivated land, and in steep slopes and rocky areas, probably because they have less fire frequency and lower level of herbivore disturbance, especially by elephants (Duvall, 2007; Edkins *et al*., 2007).

Low natural regeneration has also been observed for other parkland tree species, especially in West Africa. It seems that for a number of parkland species, regeneration is linked to the parkland system of cultivated and fallow land, regeneration mainly taking place in the later. Boffa (1999) suggested that *V. paradoxa* and *P. biglobosa* parklands are threatened by shortening or eliminating fallows due to increasing population levels. Probably, baobab tree regeneration is also affected by these. Increased human-induced fire frequency has also been linked to declining population sizes of some parkland species in Senegal (Lykke, 1998).

However, other animals such as baboons *Papio ursinus* and monkeys *Chlorocebuspygerythrus* are known to pull baobab seedlings and eat the tuber that constitutes the roots (Wickens 1982). Similarly where baboons are prevalent, they are known to destroy the majority of baobab fruit and hence greatly reduce seed production (Venter and Witkowski, 2011).

Recently, Venter and Witkowski (2013) confirmed that seedling establishment is severely hampered by poor and inconsistent rainfall. Poor rainfall is given as the reason for poor baobab seedling survival in many other regions (Chirwa *et al*., 2006; De Villiers, 1951; Dhillion and Gustad, 2004), but it is more likely that seedling mortality is a result of inconsistent intra-seasonal rainfall rather than low rainfall *per se.* It has been shown with other semi-arid African savanna species, that seedlings need frequent, and not necessarily high rainfall to survive initially (Wilson and Witkowski 1998). The results of this study show that seedling emergence can occur in years even when early season rainfall is as low as 287 mm and that in the year that received 589 mm, seedlings emerged but were unable to survive due to erratic post -emergence rainfall.

Venter and Witkowski (2013) also found that saplings died more quickly after goat browsing than cattle trampling, but that after repeated damage, of either browsing or trampling, their reserves were too depleted to survive. In conservation areas, goats and cattle are replaced by a variety of wild herbivores such as impala, kudu, eland, zebra and buffalo. Baobab recruitment in conservation areas, may be hampered by the presence of ungulate browsers and grazers, for similar reasons that marula declined in the Kruger National Park (Helm *et al*., 2009; Helm and Witkowski, 2012).

Baobab populations could possibly be maintained with low episodic recruitment (Venter and Witkowski, 2010), but if conditions for recruitment do not re-occur, then there may be cause

for concern. Climate change models predict that many areas where baobabs occur will get dryer and hotter (Cuni Sanchez *et al*., 2011a; 2011 b). This, in addition to rapidly increasing elephant, human and livestock populations, implies that future recruitment events may be even scarcer.

3. MATERIALS AND METHODS

3.1 Research Approach

The study aimed at determining baobab mortality in Mana Pools National Park by comparing data from the 1984 surveys (Swanepoel and Swanepoel 1986; Swanepoel, 1993b) to the data collected during the course of this study. To facilitate direct comparison with these previous surveys, data for 1984, were extracted from original data sheets provided by Swanepoel in 2005 and reanalyzed. Baobabs measured in 1984, were therefore identified using distance along the road and distance from the road as recorded by Swanepoel. Baobab trees were measured and assessed for both damage and growth. The study also aimed at investigating the relationship between baobab density and elephant damage to distance from the Zambezi River which is a major source of perennial water in Mana Pools National Park.

3.2 Research design and data collection

Stratified sampling was used where three strata were surveyed for the baobab population in the Mana Pools National Park study area in April 2005. The three strata were delineated according to distance from the Zambezi River (Figure 3.1). All transects were parallel with a general west-east compass alignment. Non-random purposively sampling of baobab trees within a belt 300 m wide on either side of existing roads were evaluated. Edkins *et al*. (2007) noted that baobabs are difficult to sample, which in turn invalidates random sampling methods because they tend to under-sample juveniles as mentioned by Mpofu *et al*., (2012).
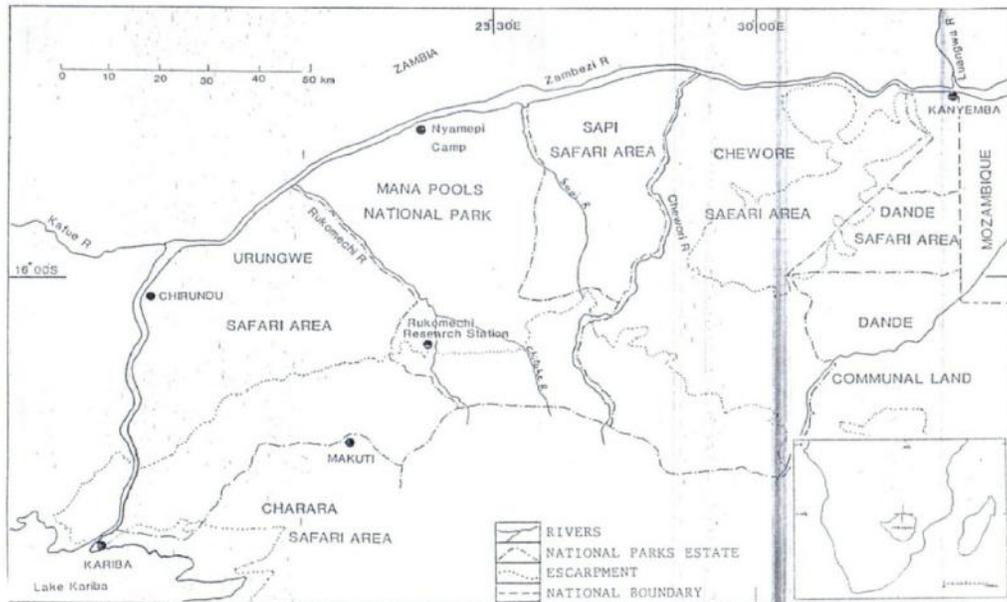
Figure 3.1 Location of the study sites in Mana Pools National Park

Three sites, based on distance from the Zambezi River, were sampled from the three strata described above namely: (i) Sapi River road (in the Zambezi flood plain, of the northern Mana Pools National Park), (ii) Fourways road (in the middle of Mana Pools National Park) and (iii) Rukomechi road which is at the base of the escarpment (southern Mana Pools). Table 3.1 provides a brief description of these study sites.

Table 3.1: Study sites in Mana Pools National Park

Study site	Distance from the Zambezi River (average)	Area covered in km^2	Number of belt transects
Sapi River road	4	16.8	28
Four Ways	26	15	25
Rukomechi road	50	6.6	12

A transect is a path along which one counts and records occurrences of the phenomena of study (e.g. plants). Belt transects were used to sample baobabs in all the three study sites using existing roads. Twenty eight transects were evaluated every 1 km from Sapi River road, twenty six transects from the Fourways road, while 12 transects were evaluated from the Rukomechi road (Table 3.1). The method used to measure density, damage and mortality was similar to the one used by Swanepoel in the 1984 and 1988 surveys. A vehicle was used to traverse the selected roads, and baobabs within 300m from either side of the road were sighted, counted and assessed for elephant damage. The same method was also used in a similar study in Gonarezhou National Park (Mpofu *et al*., 2012). Distance along the road was measured using the vehicle odometer, while distance from the road to the baobab tree was measured using a rangefinder. Baobab tree position was logged into a Garmin Geographical Positioning System (GPS) unit.

Sighting distances of baobabs on either side of the road were truncated at 300m in order to increase the detectability of baobabs in the sampling widths considering limitations of

adequately sampling baobab saplings using driven transects as used in the present study. Some baobab trees had leaves during the study period, and this made identification easy. When small, and without leaves, trees were identifiable from terminal buds, which have characteristic red leaflets (Kupika *et al*., 2014). An additional identification character of the baobab was the twig which can twist without breaking (Edkins *et al.,* 2007).

3.2.1 Population structure of *A*. *digitata* in in the study area

The size of each baobab tree was determined by its girth and height. These two variables have previously been used to measure the size of baobab trees in Mana Pools National Park and other protected areas (Swanepoel and Swanepoel, 1986; Swanepoel, 1993; Barnes *et al*., 1994; Mpofu *et al.*, 2012, Wilson, 1988; Weyerhauser, 1985). Girth at breast height (1.3 m above the ground level) of each tree was measured using a 50 m tape measure. Where girth at breast height (gbh) could not be measured due to elephant damage or because trees have not reached breast height, girth measurements were taken at ground level and converted to gbh (Weyerhaeuser 1985).

Height for each tree was estimated to the closest 1 m using a 12 m graduated pole and baobabs whose heights were more than 12 m, their heights were visually estimated. This measurement was chosen because it was used in the previous studies by Swanepoel and Swanepoel (1986); Swanepoel, (1993b) and, therefore, allowed comparison between these studies. The structure of baobab populations was determined by placing trees into various size classes with girth intervals of 2.5 m giving a total of 6 size classes. However, to allow for a comparison between 1984 and 2005 data, similar size classes to those used by Swanepoel (1993b) were used. Size classes with girth intervals of 1.5m were used, giving a total of 9 size classes.

3.2.2 Utilisation of *Adansonia digitata* by elephant in Mana Pools National Park

Elephant induced damage on 284 baobab trees, observed in the three study sites was assessed. Categorical data on the extent of elephant damage to sampled baobabs was collected and subjectively classified into five scales (Swanepoel and Swanepoel 1986). Table 3.2 shows the ranked data collection scales used in the present study.

Table 3.2: Damage categories used in this study

Scale	Damage Category	Description
0	No damage	No damage
1	Slight	Few scars on trunk
2	Moderate	Scars deeper or more numerous, trunk not ring barked
3	Severe	Trunk completely ring barked and/or scarred deeply
4	Dead	Tree collapsed

The frequency of baobabs within each damage class per study site was calculated and presented graphically. These data were used to test for variation in elephant- induced bark damage among the study sites.

3.2.3 Mortality of baobabs in Mana Pools National Park

Mortality can be measured directly or indirectly (Krebs 1994). Direct measurement is achieved by marking individuals and observing how many survive through time. Indirect measurement can be achieved by determining the abundance of successive age groups in the population (Krebs 1994). In the present study, direct measurement of mortality of baobabs along the Sapi River road transect was used based on the individual trees that were previously marked and numbered (Swanepoel and Swanepoel 1986). In the 1984 survey, 124 baobabs were numbered, and these were observed over time. Numbered baobabs were searched using previously recorded distances along the road and distance from the road since GPS coordinates were not recorded in earlier studies. The baobabs which were not found were assumed to be dead. Mortality rate was then calculated for the intervening period (1984-2005).

3.2.4 Baobab density and distribution in Mana Pools National Park

To estimate density, it was assumed that all baobabs within 300m of either side of the road were observed and enumerated. Baobab density for each transect was then calculated and used in data analysis.

3.3 Data Analysis

3.3.1 Data exploration

Baobab data on all measured variables were tested for normality using Kolgomorov-Sminorv test in STATISTICA version 7 for Windows and was not normally distributed (Appendix 2).

3.3.2 Descriptive statistics

Descriptive statistics were used to explore the following: mean baobab density, basal area and height in STATISTICA version 7 for Windows.

3.3.3 Non-parametric analysis

Due to lack of normality of all variables, Kruskal-Wallis tests (ANOVA by Ranks) in STATISTICA version 7 for Windows were used to determine significant differences between study sites for:

   i.   Mean baobab density
   ii.   Basal area
   iii. Height

Comparisons between 1984 and 2005 data were performed using Mann-Whitney tests in STATISTICA version 7 for Windows.

3.3.4 Principal Components Analysis

Principal Components Analysis (PCA) was used to compare baobab density, basal area and height data between study sites.

3.3.5 Chi-square test

A chi-square test was used to compare elephant induced baobab damage among study sites.

3.3.6 Size Class distribution

Graphs were plotted on baobab frequency against size classes and compared. Original graphs by Swanepoel (1993b) were redrawn and compared to current data (2005).

4. RESULTS

4.1 Comparison of 1984 and 2005 baobab data for Sapi study site

In 1984, Swanepoel sampled 124 baobab trees along the Nyamepi-Sapi River road in 27 transects. A repeat survey in the current study only identified 48 of the previously sampled baobab trees, although a total of 76 baobab trees were counted during this period. Results of a comparison of the 1984 and 2005 samples are presented below.

4.1.1 Comparison of Baobab density for 1984 and 2005

Comparison of baobab tree density for 1984 and current study indicated a decline from 7.65 in 1984 to 4.7 baobab trees km^{-2} (Table 4.1).

Table 4.1: Mean baobab density by study period for Sapi study site in Mana Pools National Park

Period of survey	No. of belt transects	No. of trees observed	Area inkm^2	Density (trees km^{-2})
1984	27	124	16.2	7.65
Current	28	76	16.8	4.70

The 1984 baobab density data had a higher median and interquartile range compared to the current data (Figure 4.1).Overall, there was a statistically significant difference between baobab density between 1984 and current (Mann-Whitney, U=256, p=0.037).
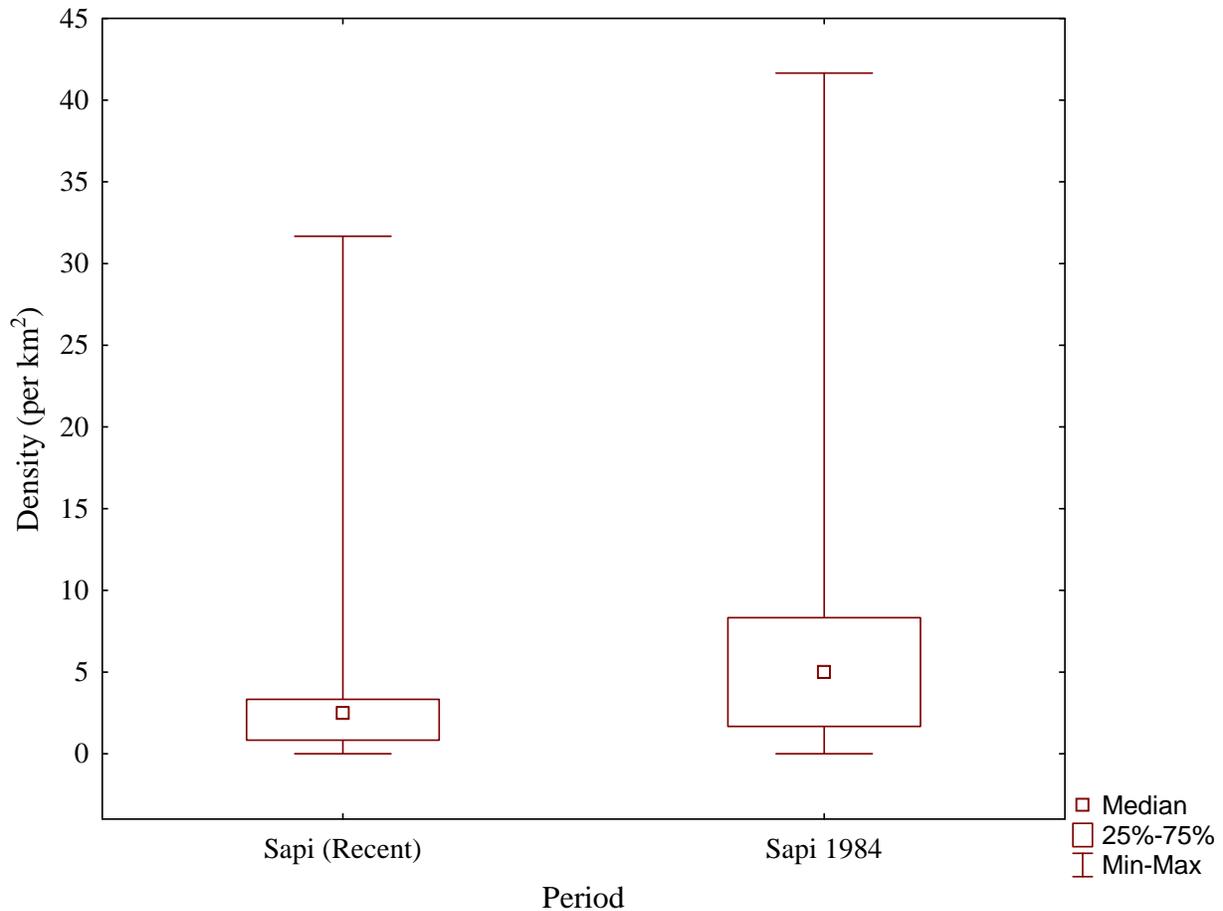
Figure 4.1: Comparison of baobab density between 1984 and 2005 in Mana Pools National Park, Sapi study site

4.1.2 Comparison of basal area between 1984 and 2005

Median basal area and the associated inter-quartile ranges for 1984 and 2005 data are shown in Figure 4.2. Basal area km^{-2} for the 1984 survey was slightly higher than that for 2005. However a comparison between 1984 and 2005 mean basal area km^{-2} indicates that there is no significant difference in basal area km^{-2} for the two study periods (Mann-Whitney, U=330, p=0.418).
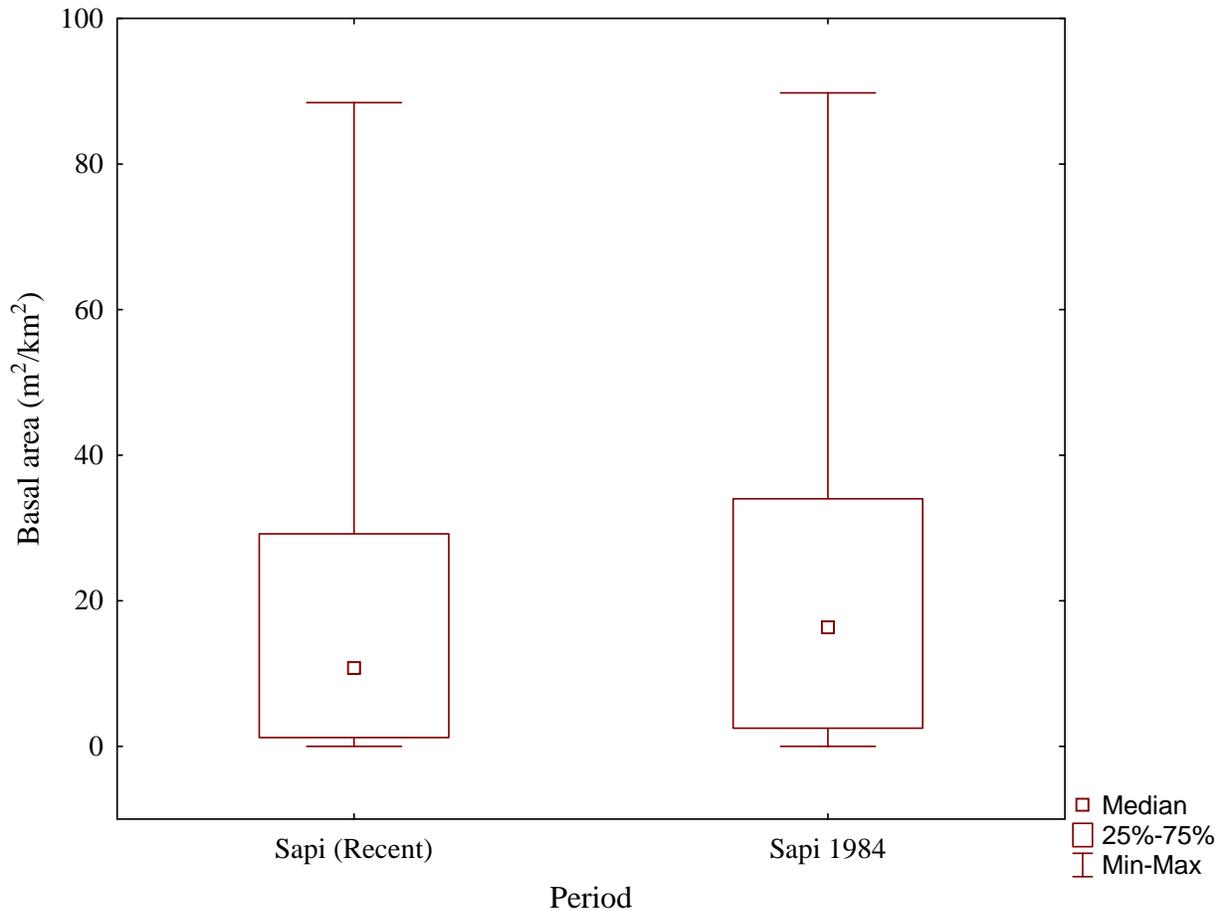
Figure 4.2: Comparison of the baobab basal area between 1984 and 2005

4.1.3 Baobab size class distribution

The highest frequency of baobabs sampled in 1984 was in the least size class (0-2.5m gbh category) and the lowest frequency was in the largest size class (15.01-17.5 m gbh category (Figure 4.3). However from the 2005 survey data the highest frequency of baobabs was in the middle class (5.01-7.5m gbh category) and the least frequency was in the largest size class category.

The size-class distribution for the population of baobab trees along the Sapi River road study site showed an inverse J-shaped pattern in 1984 with the majority of trees falling in the four smallest dbh size classes (Fig. 4.3). In 2005 the number of baobabs in the first two gbh size classes continued to decrease.

N=124   for   1984   and   N=76   for   2005

Figure 4.3: Comparison of gbh size class distribution for the baobab tree population of Sapi study site for 1984 and 2005

Figure 4.3 shows a significant difference in the first size class category where there were more baobab trees in this size class in 1984 and very few in 2005. The last three size classes had similar frequencies of baobab trees for 1984 and 2005.

4.1.4 Baobab mortality and damage

4.1.4.1 Baobab mortality

In 1984, Swanepoel and Swanepoel (1986) recorded 124 baobab trees and marked their positions using distances along the road and distance from the road. Only 48 of the numbered trees were identified in the 2005 survey. The rate of change (slope) was calculated, and was found to be 4.42%. Thus, for the intervening period (1984-2005), baobab trees along the Sapi River road died at an annual rate of 4.42%.

4.1.4.2 Baobab damage

Table 4.2: Elephant induced baobab damage categories by size class in 2005 along the Sapi River Road transect

Size class	Nil	Slight	Moderate	Severe	Dead	Total
0-2.5.0	0	2	0	0	0	2
2.51-5.0	0	8	11	4	0	23
5.01-7.5	0	5	12	9	0	26
7.51-10.0	0	0	6	8	0	14
10.01-12.5	0	0	3	5	0	8
12.51-15.0	0	0	1	1	0	2
15.01-17.50	0	0	0	1	0	1
Total	0	15	33	28	0	76

There were no undamaged baobabs in the Sapi study site during 2005. However, no dead baobabs were recorded at Sapi during this survey (Table 4.2). Moderate and severe damage categories had high frequencies in the second, third and fourth size classes (Figure 4.4).
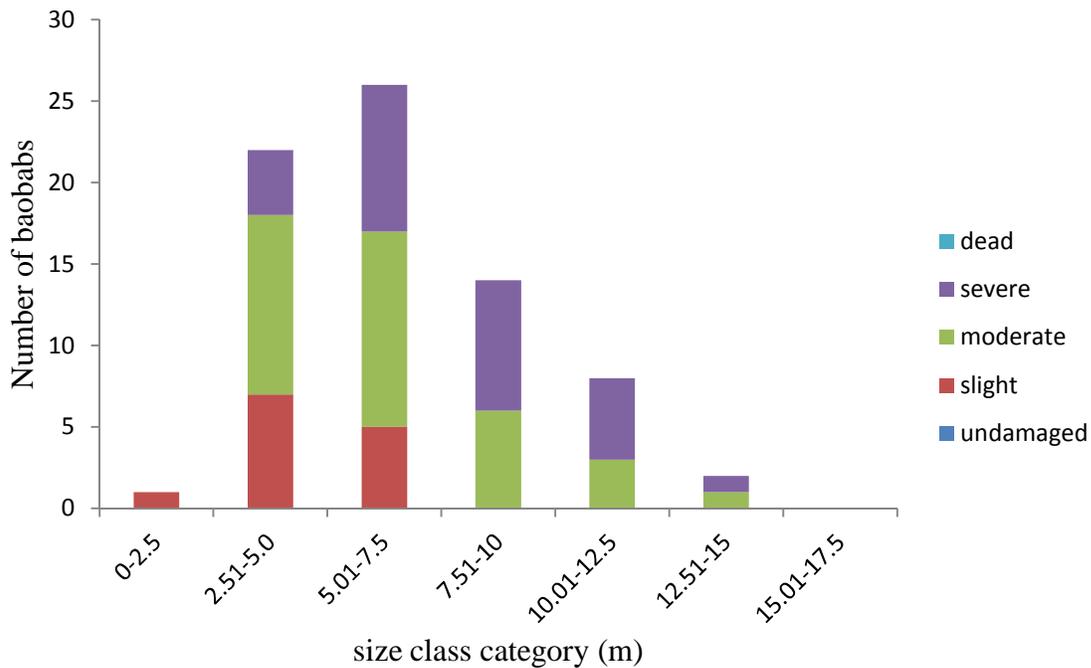

Figure 4.4: Distribution of damage by elephants relative to the size class of baobab trees, for 1984 for Sapi River Road transect.

4.2 Baobab status in Mana Pools National Park during 2005

A total of 284 baobab trees were sampled in 2005 using 65 transects in all study sites. In general, more baobabs were recorded in areas further away from the Zambezi River, the Rukomechi area at the base of the escarpment. Among these sampled baobabs, two were dead or decomposing and were recorded in the Rukomechi study site.

4.2.1 Baobab density by study site in 2005

Baobab density was higher for the Rukomechi study site, while Sapi and Fourways sites had nearly identical densities (Table 4.3).

Table 4.3: Baobab density according to study site in 2005

Study site	No. of trees	Area (km^2)	Density (tree/km^2)
Sapi	76	16.8	4.70
Four-ways	60	15	4.0
Rukomechi	148	6.6	22.43

There was a statistically significant difference on baobab tree density between the different study sites (Kruskal-Wallis ANOVA by ranks, H = 7.61, df = 2, p= 0.022), with a mean rank of 35.33 for Rukomechi, 34.83 for Sapi and 21.35 for Fourways. Rukomechi had the highest median value while Sapi and Fourways had nearly identical median values (Figure 4.5). Rukomechi had the highest interquartile range (IQR) with an extreme value further from the median value.
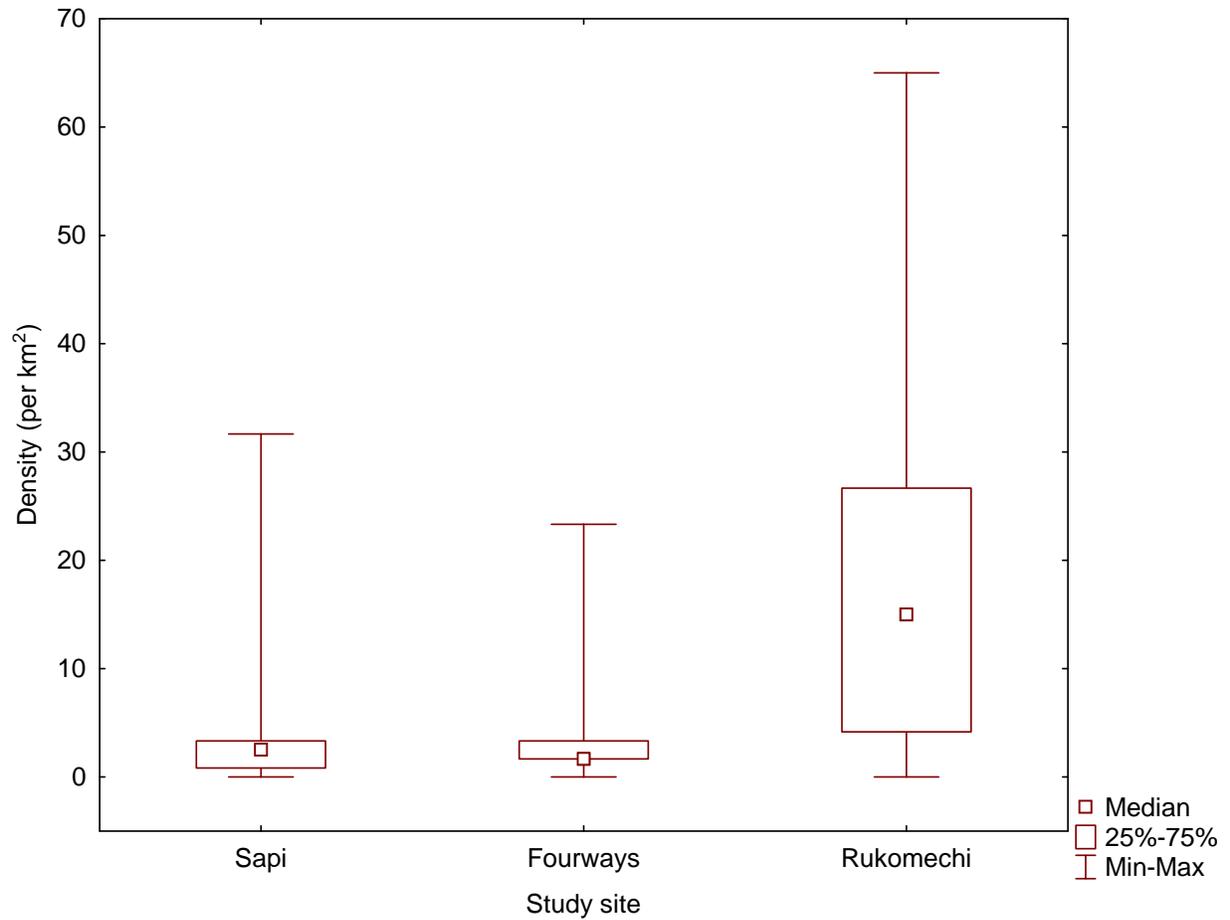
Figure 4.5: Comparison of mean baobab density by study site in 2005

4.2.2 Basal area

Rukomechi had the highest basal area, while Fourways had the least. Figure 4.5 below illustrates a box and whisker plot of the basal area by study site.
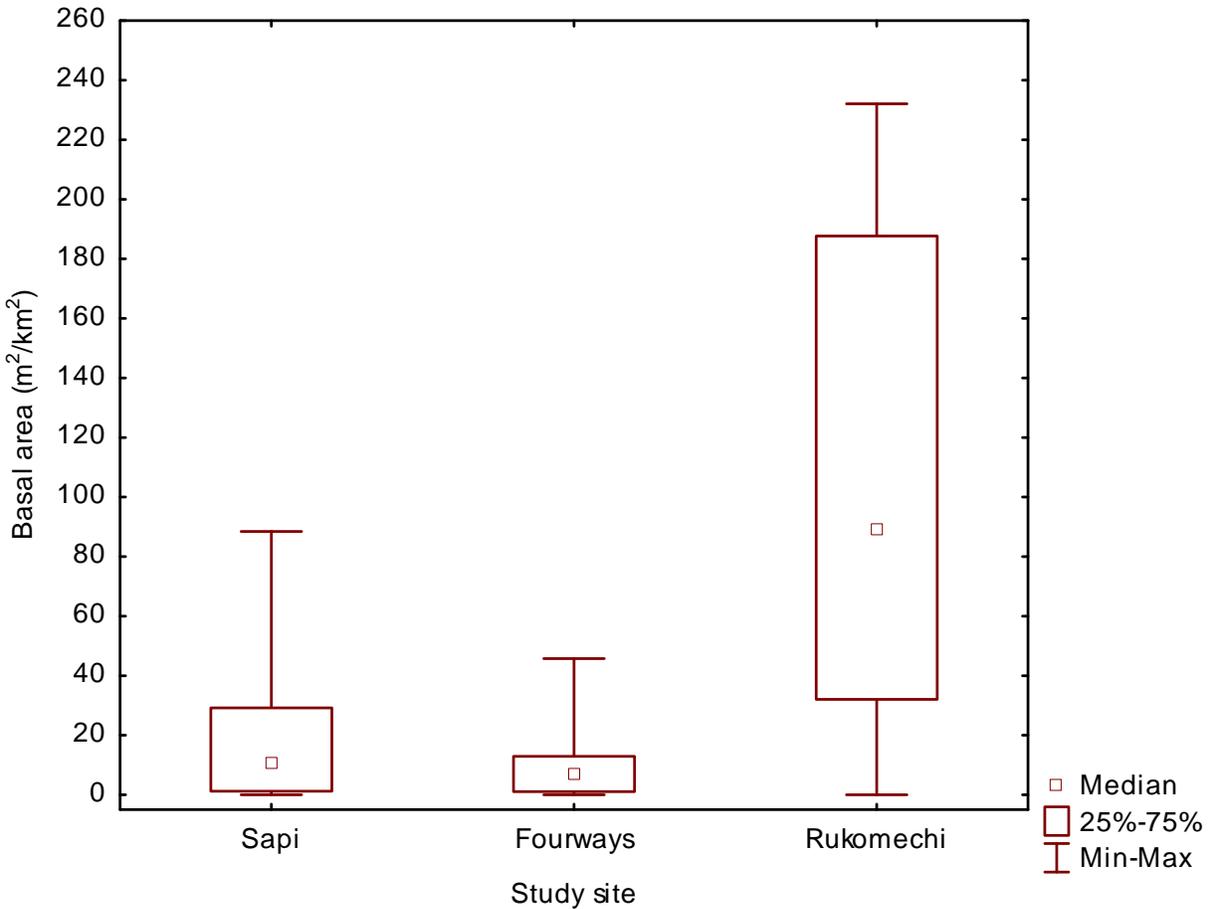
Figure 4.6 Basal area by study site in 2005

Kruskal-Wallis (ANOVA by Ranks) indicates a significant difference in basal area for the three study sites (H=10.11, df= 2, p =0.006)

4.2.3 Baobab mean height

Upper quartile range for mean baobab height decreased with increasing distance from the Zambezi River and was, therefore, lowest for Rukomechi (Figure 4.6). However, Rukomechi had

a higher median than Fourways. Basal area was significantly different between the three study sites (Kruskal-Wallis ANOVA by ranks, H = 3.69, df = 2, *p*= 0.158).
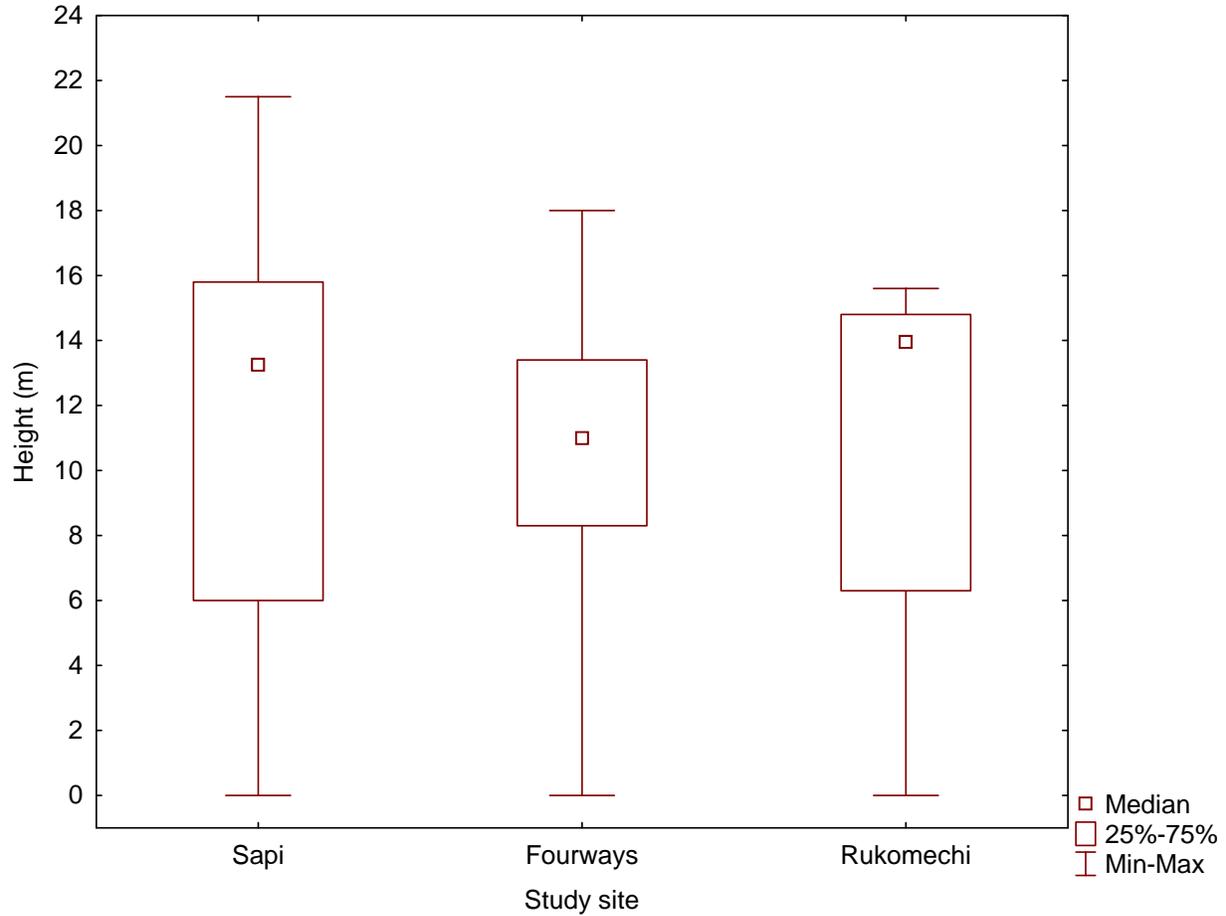
Figure 4.7: Mean Baobab height by study site in 2005

4.2.4 Size class distributions of baobab trees within study sites

The three study sites had very few baobabs in the first gbh size class (0-2.5m), and showed a bell-shaped pattern. Fourways had no baobabs recorded in the last three size class (Figure 4.8).
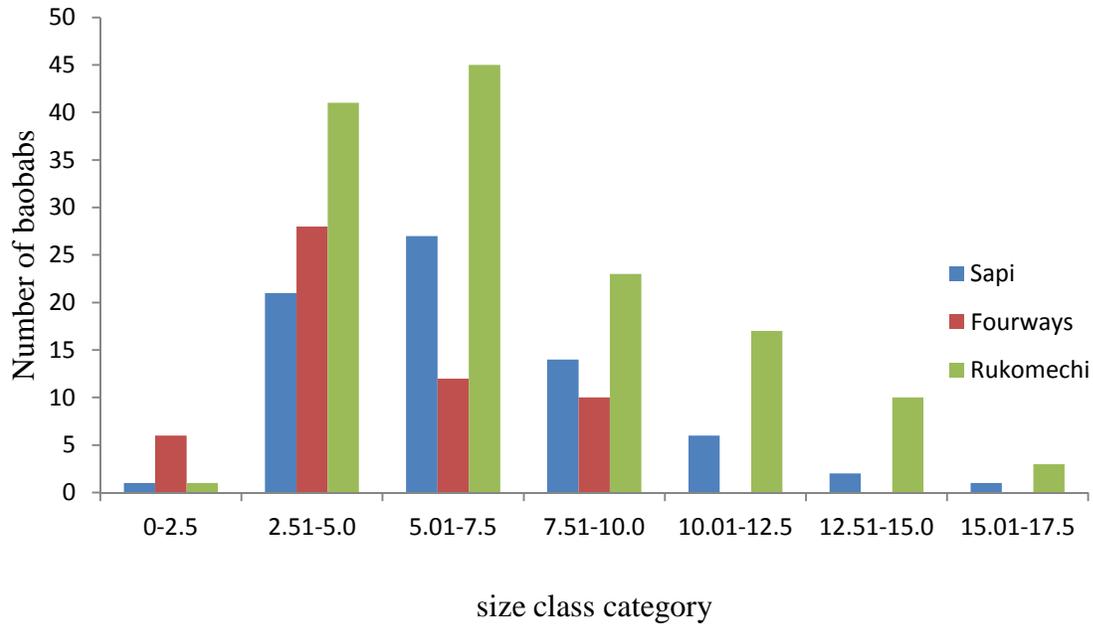
Figure 4.8: Frequency distribution of baobab trees in different size class categories for the study sites

4.2.5 Comparison of baobab damage between study sites

From a sample of 284 baobab trees, 0.7% were found dead or decomposing, 2.82% were not damaged and 96.48% were damaged. The highest frequency of baobab trees (51.76%) was slightly damaged, while 22.89% were moderately damaged, and 21.83% were severely damaged. The highest frequency of baobab damage was in Sapi (100%), and the lowest was in Rukomechi (93.92%).
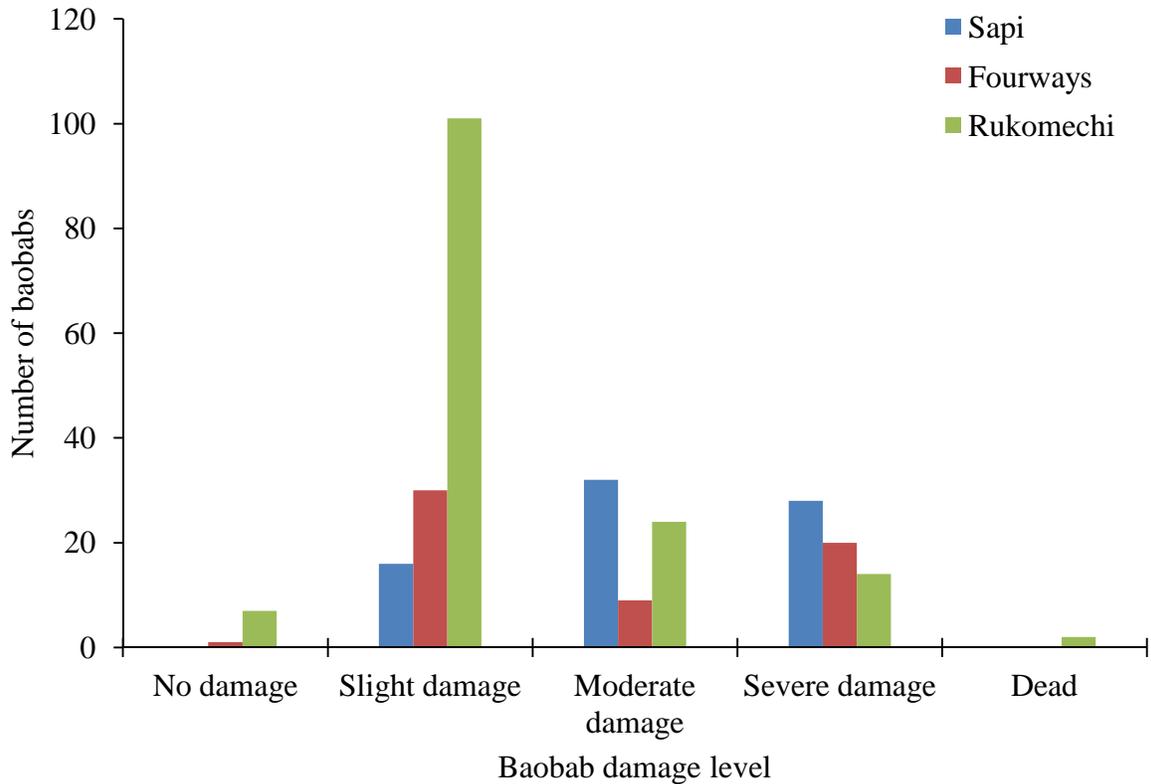
Figure 4.9: Number of baobabs found in different damage categories across study sites

Chi-square tests of independence showed that there is a significant difference between damage categories for the three study sites (2 = 60.73, df = 4, P < 0.0001).Thus, it can be concluded that there is a relationship between damage and study sites.

4.3 Principal Component Analysis

Principal Components Analysis (PCA) results showed that three Principal Components (PC) explained the variance (Table 4.7). The first two principal components explained 93.89% of the variation. Principal Component 1 accounted for 67.78% of variance, while Principal Component 2 accounted for 26.11 % of the variance. Density and basal area were strongly positively correlated with PC 1. However, baobab height was highly positively correlated to PC 2 (Figure

4.10). Transects sampled from Rukomechi study site showed a strong correlation with PC 1 (figure 4.11). For instance, Transects 3 to 7 from the Rukomech study site had high mean densities and basal area. As an example, Transect 7 had a mean baobab density of 63 baobab trees per km^{-2} and mean basal area of 193.93per km^{-2}.

Table 4.4: Eigenvalues and corresponding variance accounted by Principal Components (PC).

PC	Eigenvalue	% Total variance	Cumulative Eigenvalue	Cumulative %
1	2.03	67.78	2.03	67.7791
2	0.78	26.11	2.82	93.8906
3	0.183281	6.11	3.00	100.00
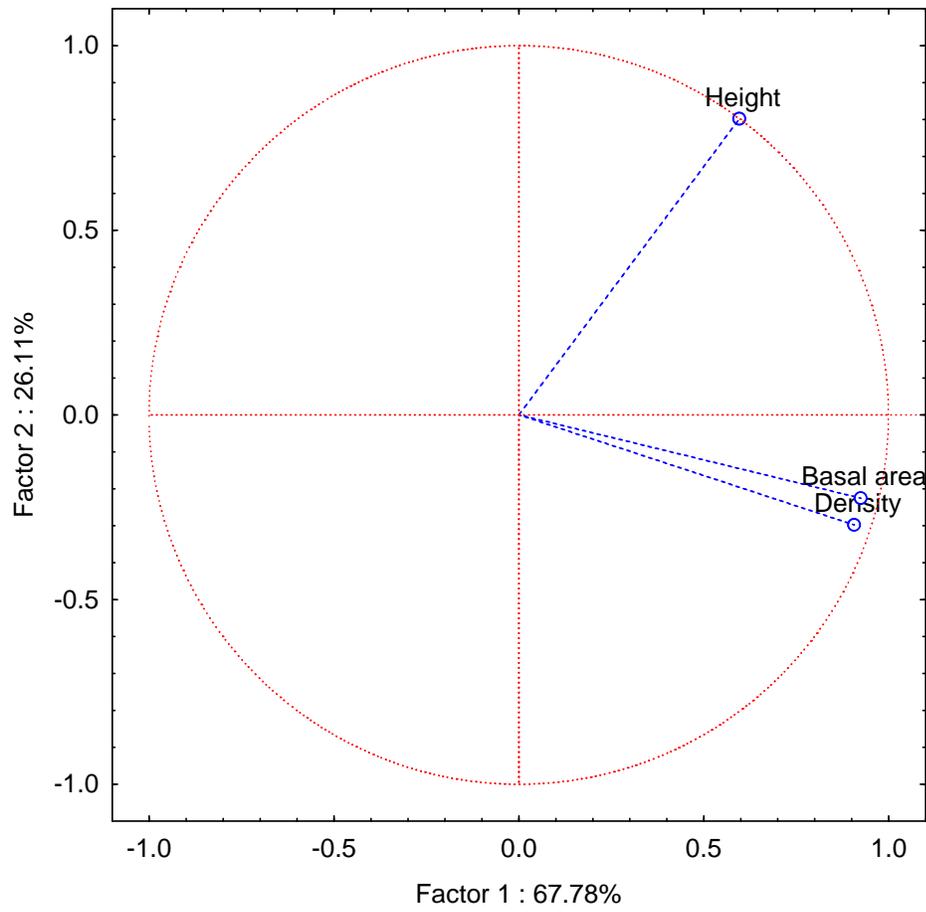
Figure 4.10: Plot of PCA variables
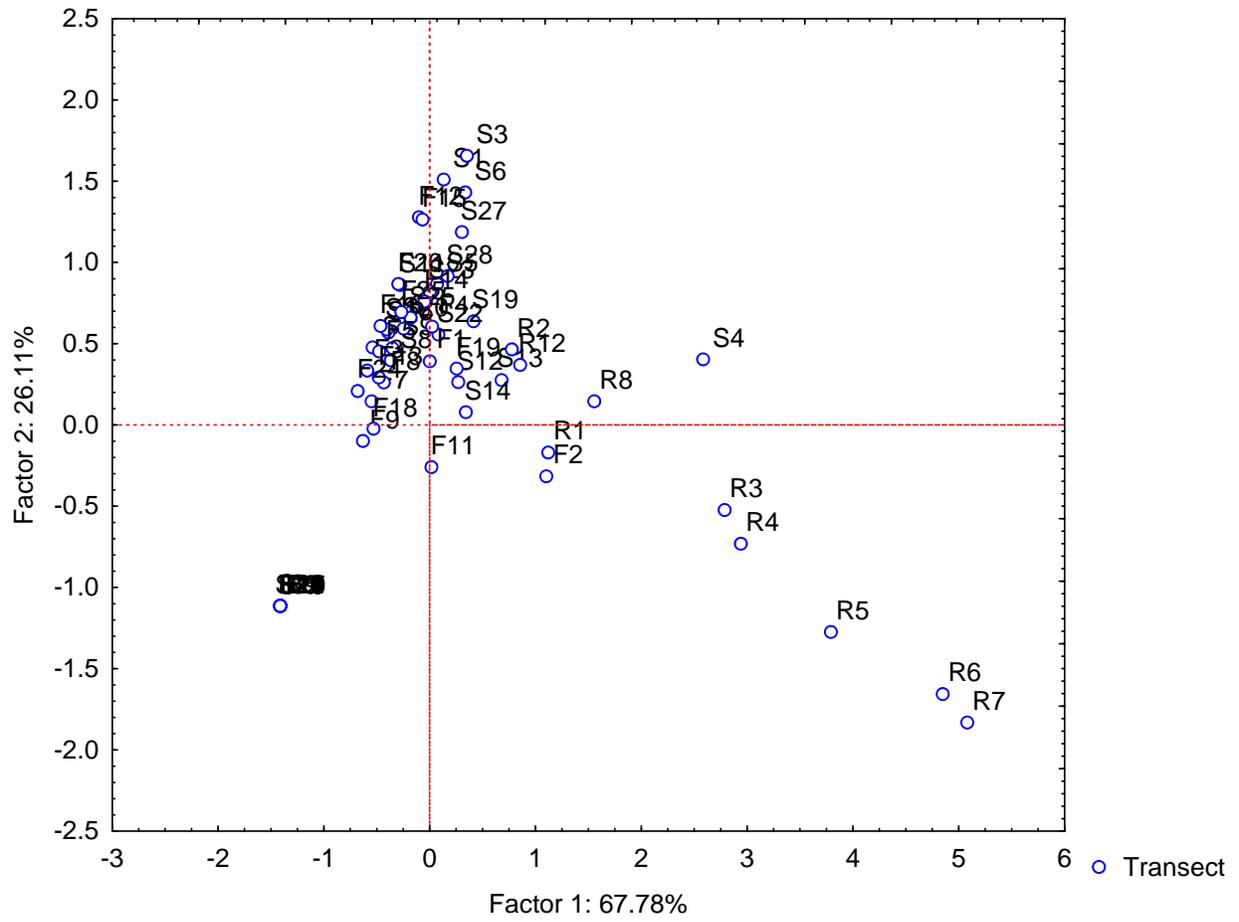
Figure 4.11: PCA scatter plot of sample transects from the study sites

5. DISCUSSION

5.1 Trends in the baobab population of Mana Pools National Park

5.1.1 Baobab density

The study revealed that the 2005 baobab density of northern Mana Pools National Park was significantly lower than that of in 1984 (Table 4.1). This decline is attributed to elephant browsing pressure, especially during the dry season and also during drought periods. Owen-Smith (1988) suggested that declines in baobab populations have occurred widely where elephants reached high densities that result in shortage of food during the dry season. Previously, elephants have been reported to be killing adult baobabs, and have been linked to the reduction in baobab densities in Mana Pools National Park (Swanepoel, 1993b).

The present study revealed that there is a significant difference in the density of baobabs between three sites studied in Mana Pools National Park. Mean baobab density was high further away from the Zambezi River in the southern part of the park, at the Rukomechi study site, and low in both central (Fourways) and northern (Sapi) parts of the park (Table 4.5). However, the average baobab density for the park was found to be 10.23 trees km^{-2} which is lower when compared to other protected areas and land types (Table 2.1). In addition, Principal Components Analysis results showed that baobab density is strongly correlated to Principal Component 1, which accounted for 67.78% of variance (Table 4.4). Baobab densities are very variable in the landscape in which they occur. They are affected by such factors as soil, competition for water (related to baobab's extensive root system), baboon-mediated seed dispersal, elephant populations and human impacts (Sidibe and Williams, 2002; Duvall, 2007; Edkins *et al*., 2007;Wickens and Lowe, 2008). In the present study, high elephant densities along the Zambezi

River during the dry season are considered to be responsible for the observed baobab density gradient.

In a similar study where the long-term impact of elephant browsing on baobab trees at Msembe (Ruaha National Park, Tanzania) was monitored over a period of thirteen years, Barnes *et al*. (1994) reported a baobab populations decline as elephant numbers increased, and the species recovered when elephant populations declined due to poaching. They reported a significant drop in baobab density between 1976 and 1982, but the population started recovering between 1982 and 1989. Baobab density decreased from 51 trees km^{-2} in1976 to 27.6 km^{-2}in 1982. However, the density increased from 27.6 trees km^{-2}in 1982 to 34.6 km^{-2}in 1989.

Moreover, Barnes *et al*. (1994) noted that if one needed to manage elephants as a way of protecting baobabs, the sex-difference in elephant feeding behaviour showed that only bulls needed be shot. The success of such a strategy was observed in the Msembe baobab population where bulls were culled by poachers and consequently the baobab mortality rate dropped. However, this strategy has a weakness in that the resulting imbalance in the sex ratio may have undesired consequences for elephant social organization (Poole, 1989). In any case, the selective culling of bulls would not necessarily prevent over-browsing of other woody species. Although there are similar sex-differences in elephant browsing on other trees and shrub species (Barnes and Douglas-Hamilton, 1982), they are not as marked as the sex-difference when feeding on baobabs.

The range of baobab density in Mana Pools National Park (4-22 trees km^{-2}) appear to be lower than previously recorded in other protected areas ,except for Sudan where baobab densities varied between 0.25 and 20.0 km^{-2}. In other protected areas in Africa, baobab densities higher

than this have been recorded (Table 2.1). In Gonarezhou National Park (GNP), southeast Zimbabwe, baobab densities ranged between 8.3 to 21.6 km^{-2} (Mpofu *et al*., 2012) and 34.3km^{-2} to 69.8km^{-2} (Kupika *et al*., 2014). At Lake Manyara, baobab density ranged from 49.2 to 72.8 trees km^{-2}in different areas of the park. In Kruger National Park, baobab density ranged from 0.3-32 baobabs km^{-2} (Kelly, 2000) and 103 baobabs km^{-2} in fields and villages (Venter and Witkowski, 2010).

Outside protected areas, diverse population densities were reported in different countries and in a number of land-use types (Table 2.1). High baobab densities in cropland than in fallows or grazing land were reported as seedlings are protected from fire and grazing (Johansson, 1999; Dhillion and Gustad, 2004; Venter and Witkowski, 2010). It was suggested that baobab density might also be higher in cropland as farmers were interested in maintaining the tree species.

5.1.2 Baobab mortality

One of the objectives of the present study was to compare mortality rates between 1984 and 2005. Monitoring of mortality rate would be a better indicator of the potential of Mana Pools baobab populations to survive than monitoring utilization alone. In 1984, the mortality rate for the baobab population in northern Mana Pools was found to be 7.3% (Swanepoel, 1993b). Swanepoel reported that if elephant densities in Mana Pools National Park were not reduced or if the baobab mortality rate persisted at 7.3%, then the baobab population was going to be reduced by 50% by the year 2000. The present study revealed that by the year 2005, this population had declined by only 38.71%. Mortality rate for the intervening period (1984-2005) was found to be 4.42%. Out of the 124 original baobab trees numbered by Swanepoel in 1984, only 48 were identified in 2005. Baobab annual mortality rates from populations across Africa were reported

to range between 0.69 and 3.7% (Kelly, 2000; Wilson, 1988). This is lower than what was recorded for Mana Pools National Park.

The 50% decline in northern Mana Pools National Park baobabs predicted by Swanepoel (1993b) was slightly higher probably due to the following reasons. Firstly, the 1992 drought could have resulted in elephant mortality in the study area. The culling that was conducted in the 1980s in Mana Pools National Park could have lowered the impact of elephants on baobabs. Secondly, trophy hunting of bull elephants in the neighboring Safari areas could have influenced the utilization of baobab trees in the study area. These factors, however, remain speculative as this aspect was not determined during the current study.

Declines in baobab populations have occurred widely where elephant densities have resulted in food shortages during the dry season (Owen-Smith, 1988). In Mana Pools National Park, elephants tend to concentrate and forage along the Zambezi River during the dry season (Swanepoel, 1993b; Dunham). The late dry season elephant distribution patterns were found to clearly reflect the distribution of water, with the exception of lone bulls or small bull herds, elephant were not observed more than 13 km from the river (Kerr and Fraser, 1975). Thus, water proximity to water becomes a determining factor during the dry season or periods of drought. Elephant distribution is regulated by water availability (Redfern *et al*., 2003; Chamaille´-Jammes, Valeix & Fritz, 2007), with herds staying <10 km from permanent water on average during the dry season (Stokke and du Toit, 2002; Loarie *et al*., 2009). In Mana Pools National Park, elephant densities continue to rise along the Zambezi River in the dry season, and recently Dunham and du Doit (2012) reported an elephant density of 8.11 km^{-2}.

5.1.3 Damage of *A. digitata* by Elephants in Mana Pools National Park

Assessments on utilisation of 284 baobab trees which occur in the study area showed that elephant damage on baobabs is greater in the northern section than in the south (Table 4.3). Elephant induced damage decreased with increasing distance from the Zambezi River. Most trees in the northern section were either severely or moderately damaged, with few trees at least slightly damaged, while trees in the south were generally only slightly damaged (Figure 4.9). The highest frequency of baobab damage was in Sapi, and the lowest in Rukomechi. Chi-square tests of independence showed significant differences among damage categories across the three study sites and it was concluded that there is a relationship between damage and study sites. These findings support the observations of Edkins *et al*. (2007) who suggested that damage incurred on any tree depends on its position relative to water, elephant population density, and timing of the initial damage (early or late in the dry season).

Results from the present study suggest that elephants target large baobabs (girth> 5m) (Weyerhaeuser, 1985; Swanepoel, 1993). The prevalence of moderate to severe elephant damage in larger classes (gbh >5m) supports the view that elephants prefer larger baobabs than smaller ones. This contradicts suggestions that elephants prefer to feed on small baobab trees (Caughley, 1976; Barnes, 1980; Weyerhaeuser, 1985). Large trees provide opportunities for more than one elephant to forage on at any given time (Owen-Smith, 1988), hence the severe damage recorded on such trees.

Elsewhere, Kupika *et al*. (2014) found that elephant damage to baobab trees was most severe in areas located closest to water sources in Gonarezhou National Park. The present study noted that elephants encountered baobabs closer to permanent water sources more regularly before or after drinking water during the dry season. This increases the chances of bark

stripping since forage is scarce during the dry season. The present results are consistent with finding from previous studies (Conybeare, 2004; Swanepoel, 1993, van Wyk & Fairall, 1969) which revealed that elephant impact on vegetation is usually higher near water sources. Tafengenyasha (1997) also observed that tree loss measured from aerial photographs was highest close to water in GNP.

These findings support the overwhelming evidence that of the latter part of the last century that indicate that elephants are major agents of habitat change. The impact of elephants on habitats increases in area where restrictions are placed on their movements and the control of elephant populations become necessary. Proper management of elephant populations, however, requires information on the trends of the populations, especially their pattern of resource utilisation (Ben-Shahar, 1993). The effect of elephant utilisation on a tree is dependent on the type of damage, the ability of the tree to recover, the role played by the tree in the plant community and the relationship between the plant community and other ecosystem components (Engel Brecht, 1979).

5.1.5 Size class distribution

A comparison of 1984, and 2005 Size Class Distribution (SCD) showed a decrease of baobabs in smaller gbh size classes baobabs (gbh <1.5 m and 1.51-3.0m.), indicating a recruitment bottleneck (Figure 4.3). However, the largest gbh size class had the same number of baobabs in 1984, and 2005. The 1984 SCD showed a reverse J-shaped pattern, with more individuals in smaller than in larger gbh size classes. This is normal for a healthy population. Reverse J-shaped SCDs were reported elsewhere (Duvall 2007; Edkins *et al*., 2007; Schumann *et al*., 2010; Venter and Witkowski 2010).

The 1984 pattern shifted to bell-shaped size class distributions, with few individuals in small and large size classes (Figure 4.3). Elsewhere, bell-shaped SCDs were reported (Assogbadjo *et al.*, 2005b; Wilson, 1988; Caughley, 1976b; Barnes 1980; Edkins, *et al.*, 2007; Venter and Witkowski, 2010; Chirwa, *et al.*, 2006). In most of these studies, the lower number of individuals in the smaller size classes was attributed to overpopulation of elephants which have the habit of destroying younger baobabs in their search for water during the dry season. This trend is believed to be due to the adverse impact of elephants on baobabs. Similarly, a comparison of SCDs for the three study sites revealed that Sapi and Rukomech had bell and positively skewed patterns, while Fourways had a bell shaped pattern (Figure 4.8). Generally, the three study sites had very few baobabs in smaller gbh size classes.

While results from the present study support the suggestion that elephants impact on baobab demography by removing individuals from the smaller gbh size-classes (Caughley, 1976; Barnes, 1980; Weyerhaeuser, 1985; Wilson, 1988; Swanepoel, 1993b; Barnes *et al.*, 1994; Edkins,*et al.*, 2007), a knowledge gap still exists on recruitment of baobabs in protected areas. This is so as elephants target larger baobabs (Weyerhaeuser, 1985; Swanepoel, 1993b). Thus, mortality only seems to be recorded in the lowest size classes. Edkins *et al.* (2007) reported that if small baobabs die and rot away, and recorded elephant utilization is based on trees that are still standing, the apparent selective utilization of larger trees would not be so apparent. Edkins *et al.* (2007) concluded that the baobab population of Kruger National Park was experiencing a drastic reduction in regeneration. . Recruitment was observed to be only taking place in the so called 'baobab refugia'. Similarly, this pattern was observed in GNP, where it was concluded that it was unlikely that baobabs would ever become locally extinct as a result of elephants and fire as

there are a number of trees growing in rocky outcrops and development areas that may act as refugia (Mpofu *et al.*, 2012).

Recent findings by Venter and Witkowski (2013) that saplings died more quickly after goat browsing than cattle trampling is important in trying to understand factors affecting baobab recruitment in protected areas. They also reported that after repeated damage due to either browsing or trampling, the baobab reserves were too depleted to survive. In conservation areas, goats and cattle are replaced by a variety of wild herbivores such as impala, kudu, eland, zebra and buffalo. Baobab recruitment in protected areas may be hampered by the presence of ungulate browsers and grazers, for similar reasons that marula declined in the Kruger National Park (Helm *et al.*, 2009, Helm and Witkowski, 2012) and *Acacia alibida* in Mana Pools National Park (Dunham, 1989a; 1989b; 1990). While these findings support earlier suggestions that impact of elephants on baobabs is confounded by interactions with drought (Whyte *et al.*, 2001) and other herbivores (Edkins *et al.*, 2007) and fire, further research in this area is required. Tafangenyasha (1992) suggested that herbivores (e.g. tree squirrels), drought and increased density of associated species could have brought about deaths of baobabs in Gonarezhou National Park. In Mana Pools National Park, high densities (Dunham and du Doit, 2012) of herbivores were recorded along the Zambezi River, and are a cause for concern in view of these findings.

Venter and Witkowski (2013) also confirmed that seedling establishment is severely hampered by poor and inconsistent rainfall. Poor rainfall is given as the reason for poor baobab seedling survival in many other regions (Chirwa *et al.*, 2006; De Villiers, 1951; Dhillion and Gustad, 2004), but it is more likely that seedling mortality is a result of inconsistent intra-seasonal rainfall rather than low rainfall as such. Rainfall in Man is very seasonal, with first rains coming in November and lasting till April. Baobabs have a limited growing season, with leaf

flush adhering closely to these months (Swanepoel, 1993a). Thus, it may be important that germination occurs relatively early in the season so that the young plants have enough time to acquire resources to survive the winter.

5.1.6 Basal Area and mean Baobab height

Basal area km^{-2} differed significantly between study sites, but there was no significant difference between 1984 and 2005. The 1984 and 2005 data did not differ because in 1984 the population had a lot of juveniles with very small circumference. Although few baobabs were recorded in 2005, they were in large size classes, with bigger circumferences. The mean baobab height was significantly different between study sites, and was positively correlated to PC 2 (Figure 4.11). Environmental factors such as soil could be explaining for this difference. Mandinyenya (2004) found that vegetation physiognomic types and individual plant species are influenced by a complex combination of environmental factors, including soil and drainage.

5.2 Conclusion

It can be concluded that baobab density in northern Mana Pools National Park significantly declined between 1984 and 2005. There were few baobabs in the small size classes in 2005 than in 1984. The level of baobab damage was significantly different between northern, central and southern Mana Pools National Park and increased with increased distance from water. Baobab density was significantly different between the three study sites, and was highest in southern Mana Pools National Park which is far away from the Zambezi River. Elephants are shaping the baobab structure in Mana Pools National Park. If baobab mortality in northern Mana Pools National Park goes unchecked, only few old baobab trees will remain.

5.3 Recommendations

Future studies should concentrate on the effects of fruit predation on seed availability, natural dispersal and scarification by wild animals and the fate of seed in soil seed banks. Management should come up with elephant management policies that will halt loss of baobabs in northern Mana Pools National Park. Continued monitoring of the marked baobab population is recommended. However, the results clearly indicate a declining trend. Only management can decide whether the observed baobab tree loss will conflict with the mandate for the park, being to conserve biodiversity in all its natural facets and fluxes.

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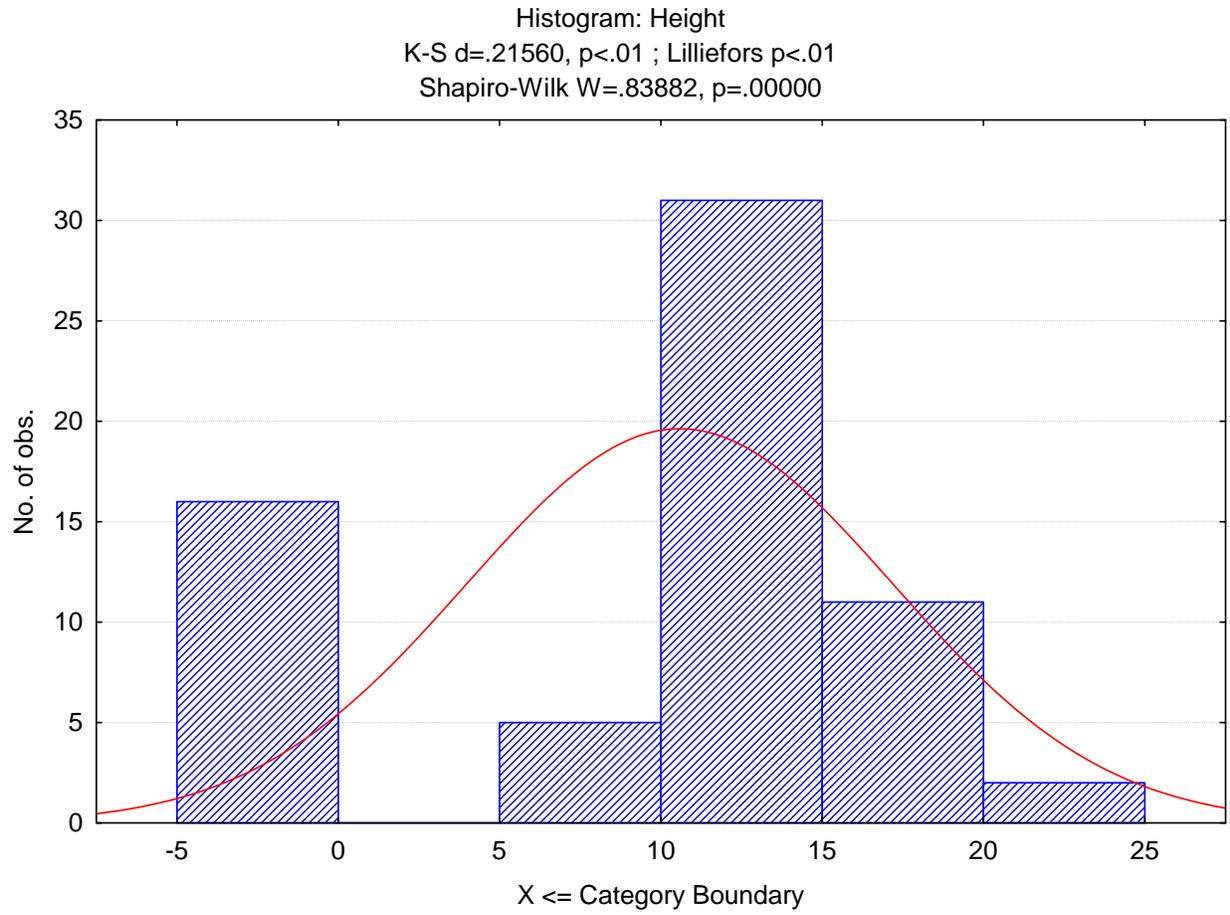Wild, H. 1961. Bombacaceae.Flora Zambesiaca, pp.511-517. London.

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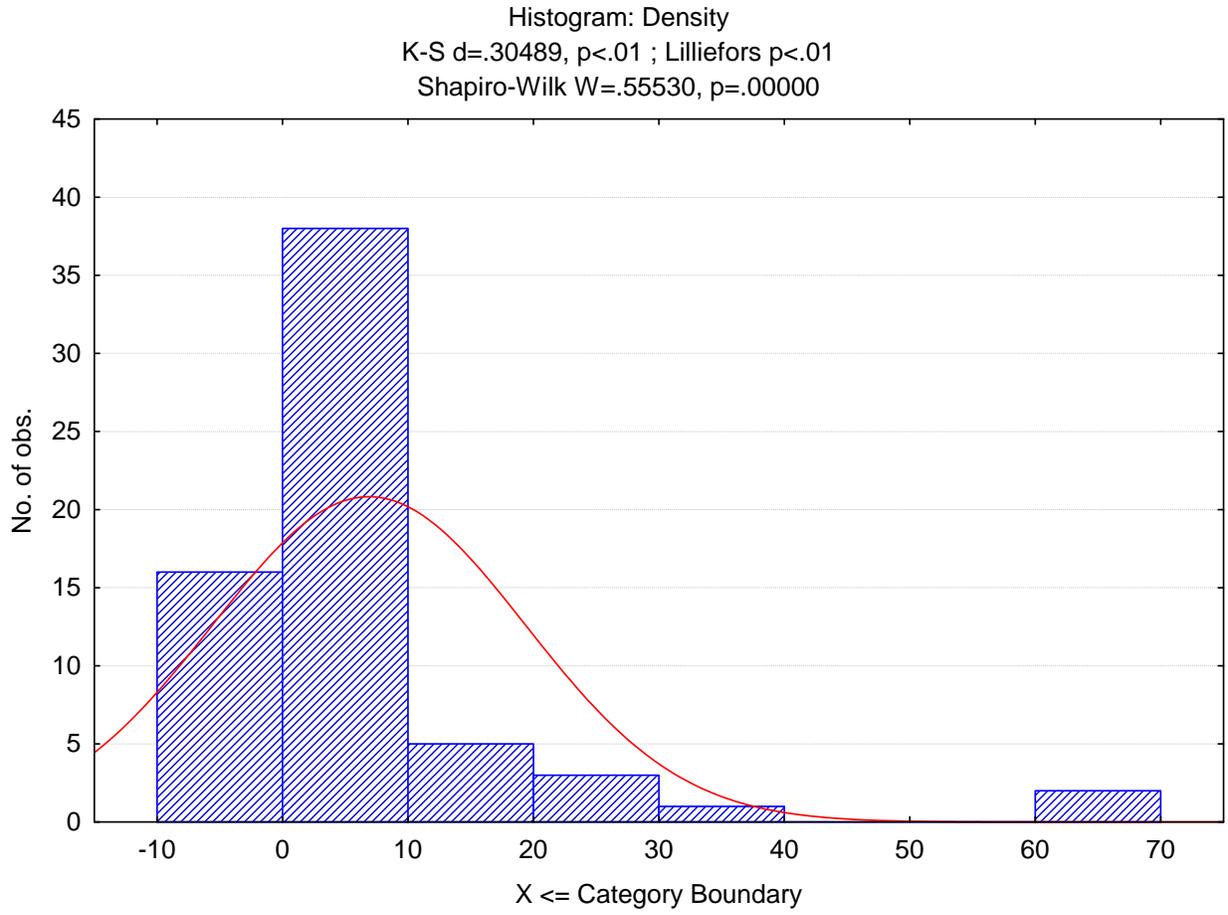
Appendix 1: Data collection tool

Sighting number	Distance along road (km)	Angle of baobab tree(degrees)	Distance from road	Girth(m) At breast height	Height	Degree of damage	GPS coordinates

Appendix 2: Data Normality tests for the three study sites (2005)

    i.   Baobab height data

Histogram: Height
K-S d=.21560, p<.01 ; Lilliefors p<.01
Shapiro-Wilk W=.83882, p=.00000

ii. Baobab density data

Histogram: Density
K-S d=.30489, p<.01 ; Lilliefors p<.01
Shapiro-Wilk W=.55530, p=.00000

iii.   Baobab basal area

Histogram: Basal area
K-S d=.29568, p<.01 ; Lilliefors p<.01
Shapiro-Wilk W=.62278, p=.00000