

The impact of elephant browsing on vulture nest sites in the Klaserie Private Nature Reserve, Limpopo Province, South Africa

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Elephants are recognised keystone species of the African savannas, as they have the ability to restructure plant communities through their feeding habits. Since the banning of the ivory trade, elephant population numbers have risen dramatically over much of southern Africa. In certain well protected areas, such as the Kruger National Park and the surrounding Private Nature Reserves in South Africa, elephant numbers are expanding since culling operation ceased in the mid 1990s. The expanding elephant population has raised concern amongst managers of protected areas and land owners within privately owned reserves as to the influence of elephant feeding habits on the tall tree component within these protected areas. I investigated the effects of elephant browsing on trees utilized by vultures as nest sites within the Associated Private Nature Reserves (APNR) in the Limpopo Province of South Africa. The data collected for 2012 was compared with data collected from 2008 to 2011. In addition, 200 control trees (without vulture nests) in the immediate vicinity of trees that were used as nesting sites were also sampled. The morphological attributes of nest and control trees were not significantly different ($p > 0.05$). However, significant differences ($p < 0.05$) were found between nest and control tree diameters and mean elephant impacts. The trees that had active nests had thicker diameters and less trunk damage compared to the control trees. Preliminary results suggest that trees with active nests are less frequently subjected to bark stripping and of lower overall impact classes than control trees. These results may be due to unequal sample sizes and require further investigation. The impacts induced to the trees with active nests from 2008-2012 were significantly different between years. However, current levels of elephant browsing within the APNR are not having an effect on the vulture nest trees. Future research should focus on a combination of factors, such as fire, importance of artificial watering points, growing elephant populations and senescence as driving influences facilitating vulture nest abandonment within the protected areas of southern Africa.

Keywords: vultures; nesting; elephants; browsing impacts, *Acacia nigrescens*

INTRODUCTION

The increased pressure from human expansion to accommodate people's demands for resources heightened human-wildlife conflict (Verlinden 1996). Man's ability to make inhospitable habitats more suitable has reduced and 'pushed back' the numbers of existing faunal species (Van Sickle 1990, Preston 1996). The resulting competition for grazing and water has often seen wildlife being persecuted in order for livestock to survive (Verlinden 1996). This has implications on the survival strategies utilized by many animals and herbivores in particular are strongly inhibited from migrating to regions that present better quality food patches (Verlinden 1996). The ability to move over vast areas in search of better grazing has been hampered by fencing brought in by the growth in the human population and this is followed by an increase in settlements and associated developments (Verlinden 1996). Human demands have restricted the range and habitat of many wildlife species and to date, many of these species only exist in protected areas throughout southern Africa.

The confinement of wildlife within game reserve borders often induces internal conflicts between many species. One of the many conflicts of interest is the impact of elephants (*Loxodonta africana*, Blumebach 1797) and their feeding habits on the surrounding vegetation. Teren and Owen-Smith (2010) stated that having a high elephant population not only affected the structure and composition of the vegetation but generated concern for the overall biodiversity of the region of interest. Herremans (1995) said that the African elephant has been reported to modify its' environment, and when they occur at particularly high densities, the long term degradation inflicted to the environment, as well as the loss of woodland, has been documented many instances. The alteration of woodland structures by a composition of large herbivores has been linked to the losses of habitat dependent species (Parker 2008).

Elephants are described as ecosystem engineers due to their ability to restructure and modify their surrounding environment (Owen-Smith 1992). Due to the relatively large physical presence an elephant imposes on their environment, their impacts are noticeable (Henley and Henley 2007, Haynes 2011). As elephants have the ability to modify their environment they have been termed as keystone species (Paine 1969, Owen-Smith 1992). Elephants are mixed feeders and are relatively unselective feeders, feeding on a wide range of species (Fritz *et al.* 2002, Roux 2006, Parker 2008). Having a wide tolerance for many plant species often results in the removal of large quantities of vegetation and this can have negative effects on plant dynamics and on species dependent on certain types of vegetation (Fritz *et al.* 2002).

Henley and Henley (2007) stated that the direct impacts of elephant browsing behaviours were evident in the form of knocked over trees, broken branches and bark stripped from the trees trunk. Anderson (2000) and Mundy *et al.* (2002) proposed that the losses of large trees from the environment could be one of the many factors contributing to the global decline of vultures. Vulture (genera *Aegyptius* and *Gyps*) populations have shown a 10 % decline in recent years (Monadjem and Garcelon 2005). The main factors affecting the vulture populations in southern Africa are the reduction of prey availability through the reduction and confinement of wild game because of grazing pressures, habitat loss and fragmentation (Monadjem and Garcelon 2005).

Elephant feeding behaviour

Gandiwa *et al.* (2011) found the top-down regulation of ecosystems by large herbivores to be an active debate between scientists and a clear example was the interactions between elephants and trees in the African savannas. Despite the alterations to many environments being a result of fire and herbivore interactions, none were more radical than the elephant's ability to change an ecosystem through its feeding behaviour (Gandiwa *et al.* 2011). The

changes in seasons resulted in the elephants changing/shifting their diets from grass to browse in response to food availability (Roux 2006, Gandiwa *et al.* 2011).

A study conducted by Greyling (2004), found that the different elephant sexes have a varied feeding style. Breeding herds were found to be more selective with regards to the vegetation they consumed. Males had a less diverse diet and consumed more in terms of quantity and quality (Greyling 2004, Henley and Henley 2007). The type of feeding varied between sexes, as cows preferentially stripped the leaves, while bulls were more prone to tree felling (Greyling 2004, Parker 2008).

Valiex *et al.* (2011) found elephant impacts on woody vegetation were prolonged during the dry season and this correlated to the decrease in large trees. The loss of large, mature trees resulted in the change of canopy structures and this effect could influence the species that are dependent on large trees as a part of their life cycle, as in the case of vultures.

Vulture impacts

There have been many proposals put forward as to why vulture populations are in global decline (Komen 1991, Mundy *et al.* 1992, Anderson 2000, Virani *et al.* 2011). Many authors are in agreement that poisoning, persecution for the traditional market, food shortages, habitat loss and collisions with power lines are the main driving factors contributing to vulture declines (Virani *et al.* 2011).

The distribution of vultures, like all other species subjected to the direct or indirect influences of burgeoning human populations persecution, are confined to the protected areas of South Africa such as the Kruger National Park and surrounding conservation areas (Virani *et al.* 2011). Anderson (2000) said that the threats to vultures outside of protected areas have been well documented but the threats to vultures within protected reserves are not well understood. Virani *et al.* (2011) stated the main focus on vultures has been directed at their importance as scavengers and minimal attention had been directed to the requirements relating to successful reproduction.

A study by Mundy (1997) found that vultures have a preference for large, mature *Acacia* species as nesting and roosting trees. Dean *et al.* (1999) concluded that large and small trees do not play identical roles in the maintenance of avian biodiversity in arid savannas. Virani *et al.* (2011) observed the loss of nest sites to be a contributing factor to further vulture declines. There could be a variety of influences affecting vultures within reserves and Rode (2010) stated that a continued growth in elephant populations could result in increased interactions between trees used for nesting and elephant browsing preferences.

Anderson (2000) stated that the wide spectrum/conflicting processes such as habitat loss was affecting vultures within areas that are meant to be safe havens reflects how little we really understand vulture requirements.

AIMS

The main aim of this study was to assess the impact of elephant feeding habits on selected trees used by large avian species as nest sites. In the process of addressing the main question, further questions will be addressed:

- a) Does elephant induced damage to nest trees change over time?
- b) Do trees with active nests have similar physical characteristics/features to inactive nest trees?
- c) Is elephant damage species specific, with regard to the tree with an active nest?

METHODS

Study site

The study was conducted within the Associated Private Nature Reserves (APNR). These reserves collectively make up 180 000 hectares (ha) of bushveld (see appendix 1) and are composed of the Timbavati, Klaserie, Balule and Umbabat Private Nature Reserves (figures 1 and 2). The APNR is situated within the Limpopo Province of South Africa on the

western side of the Kruger National Park. The nests belonging to the raptors and vultures were situated within the Klaserie Private Nature Reserve. The initial area of interest, where the data was to be collected was situated in the Timbavati, Klaserie and Umbabat Reserves. Due to the extensive flooding that occurred during February of 2012, many of the proposed sites within these particular reserves could not be accessed and this included all the Southern Ground Hornbill (*Burcovus leadbeateri*) nest sites. Consequently, the Klaserie Nature Reserve (figure 2) was used for the vulture nest surveys, as this section of the APNR has the highest concentration of these birds' nests (figure 3). The vegetation composition of the area where both active and control nests were found can be seen in appendix 1.

Fauna and Flora within the APNR

The different species of nesting vultures found within the APNR were White-backed (*Gyps africanus*), White headed (*Trigonoceps occipitalis*) and Hooded (*Necrosyrtes monachus*) Vultures. During the data collecting period, it was speculated that the majority of the nests being recorded belong to the White-backed Vultures, as these species were the only ones recorded at or near the trees that had active nests (figure 3).

Study Sites within the APNR

Greyling (2004) conducted a questionnaire in the APNR and the results showed that many of the landowners expressed their concern about the impact of elephants on certain tree species and particularly on Marula (*Sclerocarya birrea*) and Knob thorn (*Acacia nigrescens*) trees.

An ongoing study began in 2003 which focused on the impact of elephant browsing on certain tree species. Trees have been defined as single-stemmed woody plants; more than 2 metres tall (van Wyk and van Wyk 1997). There are five study sites within the APNR and 164 nest trees that are being monitored for elephant damage. For the purpose of this study and because of the effects from the February flooding, 50 trees that had known nests and 200 control trees were surveyed for elephant damage.

The data collection was based upon the methods established by Greyling (2004) and Henley and Henley (2007). The tree species being monitored were the Marula, Knob Thorn and False Marula (*Lannea schweinfurthii*) trees. The majority of the vulture nests were situated in Knob Thorn trees and these trees have been studied since 2008. Four control trees (with no vulture or raptor nests) were monitored for every tree used as a nesting tree during the 2012 survey in order to establish whether nesting trees were specifically being selected by vultures based on certain unique attributes. The control trees were determined by the same tree species having the same height and situated in each of the four cardinal directions radiating from the central nesting tree which was first established in 2008. The tree location was recorded by recording the GPS (Garmin GPS 72) co-ordinates after waiting until the accuracy of the reading was within 5m and standing as close as possible to the tree of interest. Thereafter each tree was labelled with a metal washer and code. The stem diameter of each tree was recorded at breast height in centimetres (cm), as well as the height of the canopy in meters (<1m, 1-2m, 2-3m, 3-5m or >5m). The height of each measured tree was determined by one person holding a four meter pole parallel to the trunk and a photograph was taken (figure 4). The photographs taken of the nest and control trees were uploaded onto Photoshop (Adobe Photoshop CS, Version 8.0, 2003) and height and width of the tree canopy was determined relative to the given length of the pole.

At each tree, the type of elephant impact was recorded and these were categorised as:

- a) BS- Barking Stripping
- b) BBA- Primary branch breaking to access smaller plant parts
- c) MS- Main stem breakage (snapped off)
- d) UR- Uprooting (main stem having been pushed over)

The impact for each of the different types was placed into a specific class in accordance to the extent of the overall damage and this was done for both the trees with nests and control trees (figure 8):

Class 1: none, Class 2: <1%, Class 3: 1-5%, Class 4: 5-10%, Class 5: 10-25%

Class 6: 25-50%, Class 7: 50-75%, Class 8: 75-90%, Class 9: 90-99%, Class 10: 100%.

The extent of barking stripping was estimated using the above classes. Bark stripping was determined by looking at the proportion of the circumference stripped of bark and not the length of bark that had been stripped (Henley 2004). Primary branch breaking is a feeding habit used by elephants to access the smaller, more palatable parts of the tree and the damage is estimated in accordance to proportion broken off. For example, if a tree only has two primary branches and one is broken off, this would be categorised as class 6 or 7. If the tree has been uprooted or the main stem has been broken but the tree is still alive it would be placed in class 9 but if it had died then it would be class 10.

Not all trees die after being impacted by elephants and if they persisted to the next period of study, they were recorded as follows:

- a) Die back – bark stripping events classed as 8 or more and have persisted.
- b) Bark stripping but the tree was still alive and no die back was evident on the branches.
- c) Hedging was recorded as persistence after primary branch breakage and classed as 8 or more.
- d) Pollard was recorded when the main stem had been snapped and the tree had regrown or found to be still alive.
- e) Uprooting was recorded when the tree had regrown or the branches were still alive.
- f) Death was recorded by determining the type of damage such as bark stripping or snapping of the main stem. There was also natural death, where trees have died due to other factors other than elephant impacts.

STATISTICAL ANALYSIS

The data collected for 2012 was added to and compared with the data collected from 2008 to 2011. The data analysed for this study spanned five years and was sufficient to determine any short term trends with regard to the extent of elephant damage to the nest trees.

The mean tree heights (m), crown width (m) and diameter at breast height (DBH in cm) of trees with active nest as opposed to the control trees were compared using a t-test for dependent samples after testing if the data was normally distributed using Shapiro-Wilks test (Statistica Statsoft version 10.0). Where data did not pass the test for normality, a non-parametric Mann Whitney U test was used on data (Statistica Statsoft version 10.0). Due to the unequal sample size of control and trees that were used by raptors and vultures and which proved to be in active use, 30 control trees located on the eastern side of the trees with active nests were selected and compared to the 30 trees with active nests. The 30 control trees were also tested for normality by using a Shapiro-Wilks test (Statistica Statsoft version 10.0). The data was normally distributed and a dependent sampled t-test was used.

A Chi-Squared test was used to determine any differences in elephant browsing damage to the respective trees but the various impact values were grouped with 1-2: no damage, 3-6: moderate damage and 7-10: heavily damaged in order for the Chi-Squared test to be statistically valid (Statistica, Statsoft, version 10.0). Although elephant browsing behaviour was dominated by bark stripping a Chi-Squared test was again used to determine the significance in the proportional occurrence of different impact types for the nest and control trees and the all the categories except bark stripping were added together for both the active and control trees. Bark stripping was kept separate from impact types which involved the possible structural modification of a tree. These included uprooting, main stem breakage and branch breakages to access smaller plant parts. As the latter impact types occurred

infrequently, they were grouped together to ensure large enough samples each of the respective categories for both the active and control trees.

The annual impact to active nest trees in 2012 was compared to the same active nest trees from the previous four year period. After testing for normality, all the results from the different years were not normally distributed and another Chi-Squared test was used to determine the differences of elephant impact classes over the five year period (Statistica, Statsoft, version 10.0). However, the data was grouped using the same ranges, as used with browsing impacts in order for the test to be statistically valid (Statistica, Statsoft, version 10.0).

RESULTS

When comparing the overall morphological differences between all the trees with active nests as opposed to the control trees, the heights of the trees that had active nests and the control trees showed that they were not significantly different ($p > 0.05$, $t = 1.801$, $df = 229$) (figure 5). The mean diameter at breast height showed a significant difference in diameter ($U = 1736.00$, $p < 0.05$). The trees that had the active nests had thicker diameters as opposed to the control trees (mean: 66.7 cm vs 57.76 cm). The mean crown widths were not significantly different ($p > 0.05$, $t = 1.023$, $df = 229$) (figure 5) between the trees that had active nests and control trees. The data was revised for all morphological features and when compared to the eastern control trees there was no significant difference between the tree heights ($p > 0.05$, $t = 1.469$, $df = 28$). There was a significant difference between the diameter at breast height ($p = 0.05$, $t = 2.028$, $df = 28$). The crown widths were not significantly different ($p > 0.05$, $t = -0.249$, $df = 28$) between the trees that had an active nests and the control trees.

Elephants display an extensive range of browsing behaviours but bark stripping was found to be the most prominent form of impact on the nest and control trees during the 2012 study period (figure 6). There was a significant difference ($\chi^2 = 86.93$, $df = 1$, $p < 0.05$) between the frequency of specific impact types when comparing trees that had an active nest as opposed to control trees. The remaining forms of elephant impacts were recorded in low numbers and often in combination with other impact forms. Observed trends revealed that bark stripping on trees with nests were less frequent than bark stripping on control trees. However, 23 (76%) of the 30 trees that had an active nests had their barked stripped by elephant feeding behaviours. 118 (64%) of the 185 control trees had their barked stripped off by elephants feeding and thus the statistical significance of these differences may be due to the unequal number of trees with nests that were surveyed as compared to control trees that were sampled. The lower sample size of the control trees was because not all the trees had damage inflicted to them.

The extent of impacts inflicted by elephants in 2012 to the trees that had an active nest and the control trees had a significant difference ($\chi^2 = 146.52$, $df = 2$, $p < 0.05$) (figure 7). The trees that had an active nest had a lower, overall mean impact class as opposed to the control nest trees. The average impact score was found to be category 3 (see methods) for the trees that had active nests. The control trees had a mean impact score of category 4. The vultures appeared to be nesting in trees with significantly less damaged and the trees that had either no nest or nests that were inactive were significantly more damaged (figure 7). However, these statistical differences could be ascribed to the large sample size of the control trees. When 30 samples located on the eastern side of the active tree were chosen from the 201 control trees and compared with the 30 active trees, there was no significant difference ($\chi^2 = 5.07$, $df = 2$, $p > 0.05$) in impact levels between trees with nests or those without.

Another Chi-Squared test was conducted to determine the difference among the mean impact classes (irrespective of impact type) over the five years (figure 8). The trees monitored in 2008 through to 2011 were added to the trees monitored in 2012 but only the same trees over the five years were compared. The results showed there to be a significant difference among the impact classes over the five year study period ($\chi^2 = 108.99$, $df = 14$, $p < 0.05$). However, the mean impacts classes were relatively consistent but 2012 impact types were significantly lower, with 53% of the trees with active nests having minimal browsing impacts (category 1-2) (Table 1). The moderate impact category (3-6) noted a marked decrease from 2011 (73%) to 2012 (33%). Finally, the heavily impacted trees have remained consistent (13%) over the five year study period.

DISCUSSION

The browsing effect induced by elephants has the ability to and does alter the structure and composition of many woody species within reserves (Roux 2006, van Aarde *et al.* 2006, Parker 2008, Rode 2010). However, elephant sexes had different feeding abilities and structures and the resulting impacts were varied in accordance to the social dynamics of the elephant herd (De Villiers 1994, Greyling 2004, Roux 2006, Druce *et al.* 2008, Parker 2008, de Knecht *et al.* 2011). Greyling (2004) found that both family and bull groups had equal preferences for certain species such as the Knob Thorn and Marula trees within the APNR.

Shannon *et al.* (2008) found that many of the largest trees occurring in the southern section of the Kruger National Park were found close to drainage lines and major river courses. The large size of the trees was attributed to the soil moisture and nutrient levels being the highest in these particular areas (Shannon *et al.* 2008). It has been well established that browse becomes an important part of an elephant's diet during the dry season or during periods of low rainfall (Parker 2008, Loarie *et al.* 2009).

Elephant movements become localised during the drier periods and their foraging movements never stray far from permanent water bodies (Roux 2006). Elephant dependence on water and vulture dependence on large trees, (that are usually located along drainage lines), in which to nest enhances the interaction rates between the species. Vultures showed a preference to nest in large Knob Thorn trees (appendix 1) and de Villiers (1994) found elephants to preferentially browse upon the same species of trees. Rode (2010) found 98% of the vultures to nest within Knob Thorn trees. Thus, the browsing habitats of elephants could be detrimental to the nesting sites of vultures.

There were no significant differences between the trees that had an active nest and control tree heights. The crown widths of the trees with an active nest and control trees were not significantly different either. The only observed significant difference occurred with the basal diameters between the active nest and control trees. It appeared that the vultures were actively nesting in trees that had thicker trunk diameters. A study by Teren and Owen-Smith (2010) on the vegetation along the Linyati River in the northern part of the Chobe River found a progressive reduction in the number of *Acacia* trees from the riparian woodland. However, a number of large, mature *Acacia* trees still remained standing; the most prominent species remaining was the Knob Thorn Tree. Teren and Owen-Smith (2010) observed these trees to be too large to be felled by elephants. This could provide an accurate proposal as to why the vultures are selecting thicker trunked trees in which to nest. However, some of the largest trees with active nests were located along and often closer to drainage lines (figure 3) than many of the control trees. Shannon *et al.* (2008) observed the largest trees within a particular region were located along or near drainage lines. It is possible that vultures were selecting certain trees because on their position within the landscape rather than the thickness of their diameters (M. Henley *per. comms.*).

A short term analysis (2008-2012) of the overall impacts of elephant browsing to vulture nest trees showed there to be a significant difference between the years and extent of damage. However, the impacts showed a remarkable consistency with the highest levels of impact been recorded in 2011 and the lowest in 2012 (Table 1). This suggests that the current extent of elephant browsing behaviours is not localised within a confined region and is allowing the trees to recover from previous browsing events. The observation of the browsing impacts being relatively consistent over time was supported by Rode (2010) who found the survival rates of the vulture nest trees to be 96.30% during her three year study. Teren and Owen-Smith (2010) found the most prominent damage inflicted to trees while browsing was bark stripping (figure 6), similar observations were recorded by Owen-Smith *et al.* (2006), Roux (2006), Henley and Henley (2007), Moncrieff *et al.* 2008, Parker (2008), Rode (2010) and Gandiwa *et al.* (2011). Teren and Owen-Smith (2010) stated, trees that had previously been damaged by bark stripping had their bark regrow and made those particular individuals even more resistant to the effects of elephant browsing. Rode (2010) found the vultures to nest in trees that were persisting and not at risk of mortality; the same trees were still present during the 2012 study. The overall results from this study suggested that the current levels of browsing by the elephants are having no significant impact on the vulture nest trees.

The reduction of suitable land has restricted vultures to protected areas and this induces its own problems such as intra-specific competition. The vultures not only compete for food resources but are also limited by the availability of suitable nest sites and nest locations. Monadjem and Garcelon (2005) found vultures to preferentially nest along drainage lines. Hockey *et al.* (2005) found White-backed Vultures to be tolerant of conspecifics while nesting and stated that loose colonies of up to 10 pairs have been recorded. Nests have been found at distances varying from 50 to 200 m apart from each other and this spatial pattern has been observed since the start of this study in 2008 (Hockey *et al.* 2005,

Rode 2010). The increased frequency of suitable trees occurring along drainage lines could be a plausible explanation as to why the vulture nest trees were clumped together (see figure 3). Herbolt and Anderson (2006) observed White-backed Vultures nesting in small colonies for a year or two and then suddenly vacating those nest locations. The phenomenon of vultures abruptly deserting particular nest locations was an interesting find but by no means unique (Herbolt and Anderson 2006). Mundy *et al.* (1992) had found similar behaviours in Botswana and in Hwange National Park in Zimbabwe. The proposed theories as to why vultures show such behaviour is thought to be related to the temporal availability of food or to disturbance to the nest sites (Herbolt and Anderson 2006). This observed phenomenon could help explain why only 30 of the 50 nest sites surveyed were still active.

Rode (2010) found that the most important feature for nesting vultures was their accessibility to watering points and more specifically artificial water points. While raptor nests were associated with their close proximity to roads (Rode 2010). The findings by Kemp and Kemp (1975) were also found by Rode (2010) who observed vulture nests to be clumped within the vicinity of watering points. The need to regularly bathe is to ensure that the vulture's carrion feeding behaviours do not result in the spread of diseases that can be acquired from feeding on various carcasses (Rode 2010). The establishment of artificial watering points has been introduced to many reserves by owners and managers in an attempt to boost game viewing for tourists (Chamailé-Jammes *et al.* 2007, Parker 2008). However, the presence of year round water reduces the need for elephants to migrate between suitable habitats in search of browsing (Loarie *et al.* 2009). Loarie *et al.* (2009) argued that elephant-vegetation interactions have been extensively complicated by human modifications. The network of water holes within many reserves has allowed elephants to expand into regions that were previously inaccessible during the dry season (Loarie *et al.* 2009).

Cumming *et al.* (1997) found that in regions where elephants have existed in high densities (>0.5 individuals per km^2) for extended periods of time, the structure and biodiversity of woodlands has been reduced. It has been established that localised woodland damage and the loss of certain tree species will occur even when elephant densities are as low as <0.2 elephants per km^2 (Cumming *et al.* 1997). Herremans (1995) found specialist canopy bird species to be extensively reduced in woodlands cleared by elephants. Rode (2010) found vultures to preferentially build their nests at the very top of a suitable tree and by association could be defined as a canopy specialist. Herremans (1995) observed that despite the loss of woodland, there was no loss of species but rather a replacement of species and change of composition of the community.

If densities persist at >0.5 per km^2 , elephants have the potential to reduce several woodland areas to scrub and result in a loss in spatial heterogeneity (Cumming *et al.* 1997, Teren and Owen-Smith 2010). Owen-Smith *et al.* (2006) makes mention of the importance of the loss of large trees from sections in the Kruger National Park and how these observations influence management decisions concerning elephants. There have been observations from certain regions that once had large trees, that this habitat has been replaced by a dense shrub layer (Owen-Smith *et al.* 2006). The changes in vegetation structures referred to by Owen-Smith *et al.* (2006) occurred during a period when the estimated elephant population was around 7000 individuals and if elephants were solely responsible, a far lower population of elephants would be required to halt the loss of large trees. The implementation of artificial watering points could create unnaturally high concentrations of elephants within specific regions during specific seasons (Chammailé-Jammes *et al.* 2007). Despite the extensive impacts induced by elephant browsing on woodland habitats, Herremans (1995) stated that experiences from other regions in Africa have demonstrated a remarkable ability of impacted woodland to regenerate after the controlling factors have been reversed.

Parker (2008) found that ecosystems were in a constant state of flux, not equilibrium and that certain levels of disturbance were vital for the promotion and maintenance of biodiversity in certain ecosystems. Mapaure (2001) observed African savannas to be highly dynamic systems and alternated between different vegetations types and states throughout the year. The frequency of disturbance events can cause plant communities to respond differently, and in regions where movements are concentrated, the disturbance is likely to have a cumulative effect to the overall ecosystem (Parker 2008).

Since the population reduction programmes ended by the mid-1990s, elephant numbers have begun increasing within southern Africa (Hall-Martin 1992). Druce *et al.* (2007) stated that over the last decade many national parks and private reserves in South Africa have experienced exceptional growth rates in their elephant populations. Van Aarde and Jackson (2007) re-enforced the need for wildlife corridors to facilitate the movements of large herbivores between certain habitats and potentially decrease localized feeding impacts. Elephants were found to disperse when given the opportunity or if the circumstances allowed for movement (van Aarde and Jackson 2007). Roux (2006) observed how large bodied species had seasonal changes in the use of their habitats. By being able to cover large distances, megaherbivores can access and utilize resources in different regions that are often only temporary available (Roux 2006). The ability of large species, such as elephants, to move between feeding areas often determines how they utilize those particular resources (Chammailé-Jammes *et al.* 2007, van Aarde and Jackson 2007).

Teren and Owen-Smith (2010) said that large herbivores introduced into environments in which they have previously been absent usually changed/alterd the vegetation in some way. There have been mixed reports with regard to the overall impacts elephant browsing has had on certain bird groups inhabiting woodland habitats (Herremans 1995, Cumming *et al.* 1997). The studies addressing the elephant impacts found that birds

with specific habitat requirements were negatively impacted by the effects of browsing, and similar theories could be proposed with regard to the vulture nesting requirements (Herremans 1995, Cumming *et al.* 1997, Monadjem and Garcelon 2005). Continued impacts on the preferred nest trees could force the vultures to abandon their nests and possibly resort to nesting outside protected areas (Herbolt and Anderson 2006). However, the results from my study found that the elephants' feeding behaviours did not have an effect on the trees used by vultures in which to nest. Rode (2010) found that the elephant impacts did not lead to the desertion of nests sites by vultures in the short term. The results suggest that other factors must be contributing to the decline in vulture nest tree activity within the Klaserie Private Nature Reserve.

The majority of the observed nests were located in the largest trees within that particular region and Shannon *et al.* (2008) found large trees to experience fluctuating levels of mortality due to a combination of driving effects (e.g. herbivory, fire, droughts and diseases). Shannon *et al.* (2008) proposed that the high numbers of large trees within the Kruger National Park are unlikely to be a common event within a dynamic savanna ecosystem. Skarpe *et al.* (2004) proposed the growth of large trees to be a result of episodic events when elephant and other herbivores numbers were low enough to allow for extensive recruitments of woody species. If the largest trees are also the oldest, the role of senescence could be another factor worth addressing. Shannon *et al.* (2008) said that the role of senescence in the mortality of large trees is poorly understood. The mass mortality of large trees in the future, purely due to aging, could be a new factor that holds greater significance than the impacts induced by elephant browsing.

CONCLUSIONS

The overall impacts induced by elephant browsing behaviours did not have a significant impact on the trees that had active nests over the five year study period (2008-2012). There were no significant differences between tree heights of trees with nests and the control trees. The same result was found between the crown widths of the trees that had an active nest and the control trees. However, there was a significant difference between the basal diameters of the trees with an active nest and control trees and between the extent of damage between nest and control trees in 2012. It appeared that vultures were selecting trees that would persist irrespective of accumulated damage. Rode (2010) found elephant damage not to have caused desertion of the nest sites over the short term, but longer studies would be required in order to fully understand the overall influences that elephants have on nesting vultures.

Rode (2010) further found vulture nest trees to be located near artificial-water points and this had significant behavioural importance and leads to the conclusion that the decommissioning of water points could have serious consequences for vulture populations but simultaneously could lower the impact to these trees as elephants move to other areas in search of water. My results are in agreement with previous studies that found elephants did not have a significant impact on the trees that had active nests within the APNR over the five year study period. Despite the *Acacia* species being preferentially browsed upon by elephants during the drier periods, increasing these tree species interactions rates, their survival was very high and persistence continued to the next year, despite the elephant damage. My proposed hypothesis was not supported by my results and elephant browsing does not have a significant impact on avian nest trees.

Parker (2008) stated that due to their large size, energy requirements, generalist feeding habits and ability to restructure the environments they inhabit, elephants are keystone herbivores within the African savanna ecosystem. Roux (2006) said that when reserves were <300 km², the movements by elephants became restricted and this aggregation of elephants could then drive habitat change. An increase in elephant populations will further push the demand for more space in which to accommodate these expanding populations. Many ecologists are in agreement that ecosystems require certain levels of disturbance to maintain their functions and diversity (Parker 2008, Dupont *et al.* 2011). However, heavily impacted localized disturbances in areas used by vultures and raptors as nesting sites could have adverse consequences to the overall biodiversity in the long term.

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LEGENDS TO FIGURES AND TABLE

Table 1: The categorised percentage of the impacts inflicted to same trees with active nests over the five year period (2008-2012) within the APNR.

Figure 1: The Associated Private Nature Reserves situated on the Western Boundary of the Kruger National Park and collectively constitutes 180 000 ha of bushveld (Motswari 2010).

Figure 2: The 180 000ha APNR, the Klaserie Private Reserve was the main focal point of the study. The star denotes the position of where the majority of the vulture nests were located (Taken from Rode, 2010).

Figure 3: The distribution of the vulture nest and control trees within the Klaserie Private Nature Reserve. Note how the nest trees show an aggregation along drainage lines.

Figure 4: An example of how tree heights were measured, using a 4 m high pole for scale. The arrow represents the length of the pole, field assistant is Kate Meares.

Figure 5: The tree morphology of active and control nest tree included; (A) tree heights (m), (B) the diameter at breast height (cm), (C) the crown width (m) (n = 30 active nest trees, n = 201 control nest trees (+ SD)).

Figure 6: The impacts as a result of elephant browsing behaviours on both trees that had an active nest and control trees (n = 185) (bb – branch breakage, bba – primary branch breakage, bs – bark stripping).

Figure 7: The mean (+SD) comparison of the elephant feeding impacts to trees that had an active nest and control trees in 2012 (n = 30 for active trees, n = 201 for control trees).

Figure 8: The mean (+SD) impact classes induced by elephant browsing behaviour to trees that had active nest over the 5 year study period (n = 30).

Table 1: The categorised percentage of the impacts inflicted to same trees with active nests over the five year period (2008-2012) within the APNR.

Categories of Impact	2008	2009	2010	2011	2012
1-2	27%	27%	27%	13%	53%
3-6	60%	60%	60%	73%	33%
7-10	13%	13%	13%	13%	13%



Figure 1: The Associated Private Nature Reserves situated on the Western Boundary of the Kruger National Park and collectively constitutes 180 000 ha of bushveld (Motswari 2010).

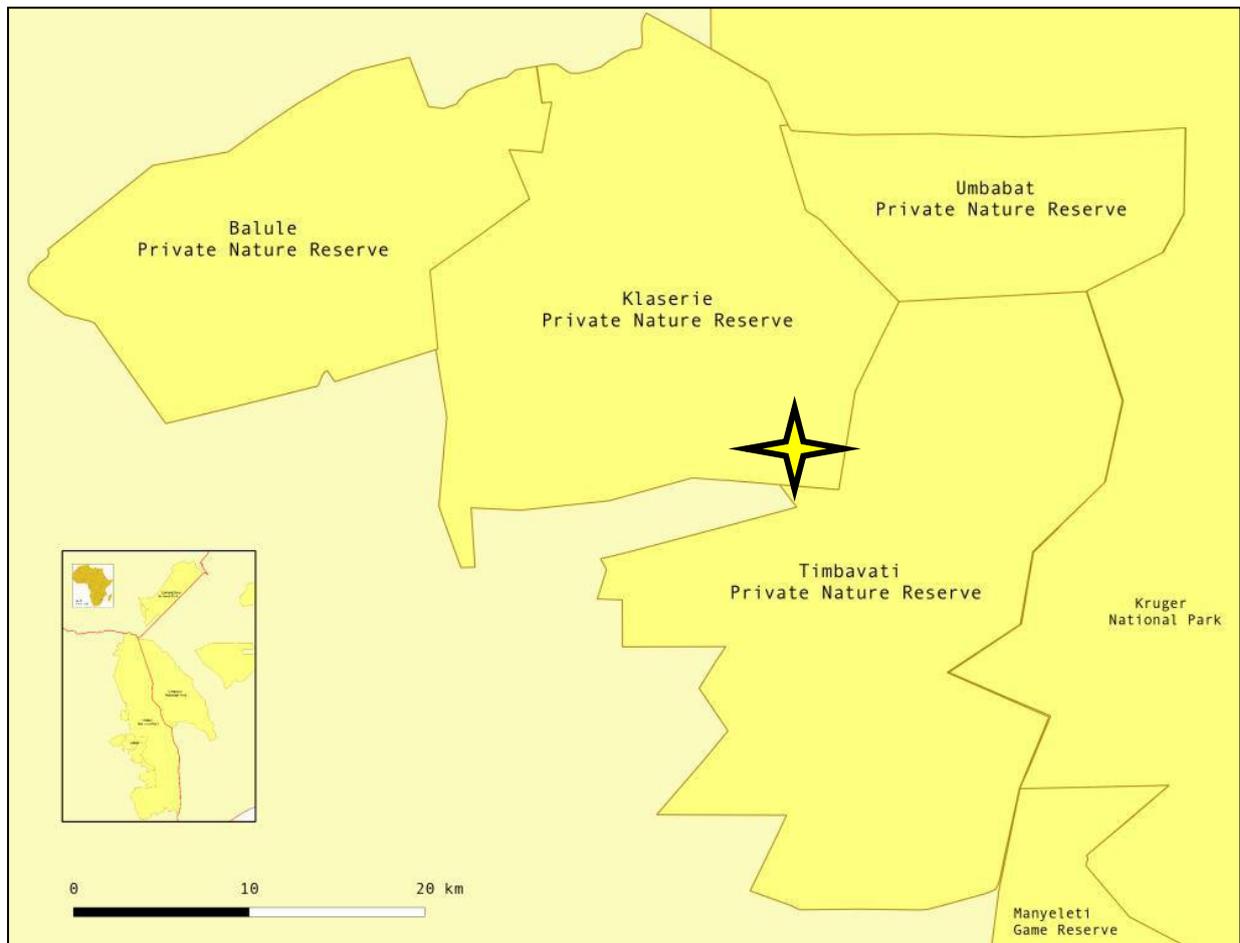


Figure 2: The 180 000ha APNR, the Klaserie Private Reserve was the main focal point of the study. The star denotes the position of where the majority of the vulture nests were located (Taken from Rode, 2010).

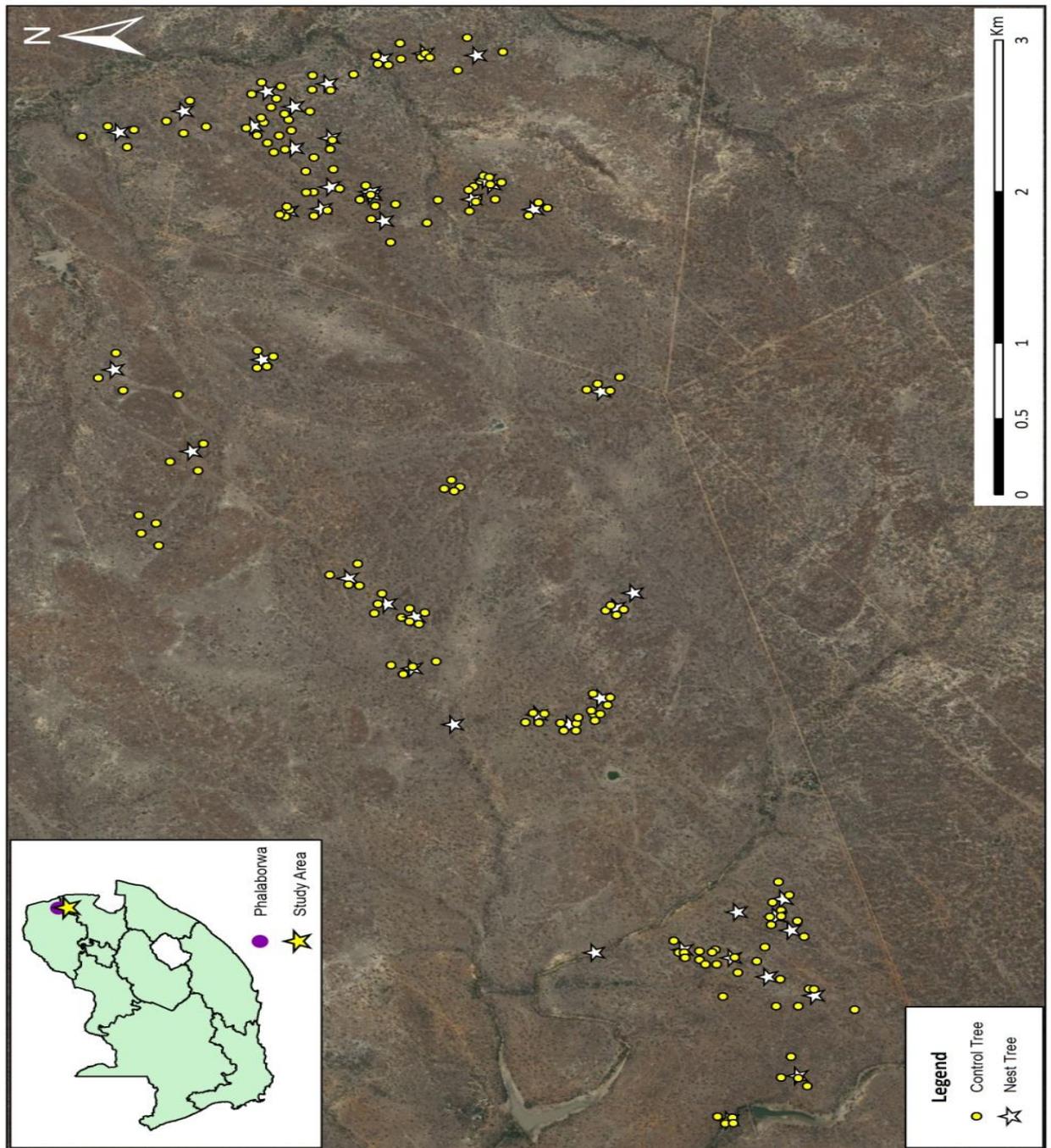


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Figure 4: An example of how tree heights were measured, using a 4 m high pole for scale.

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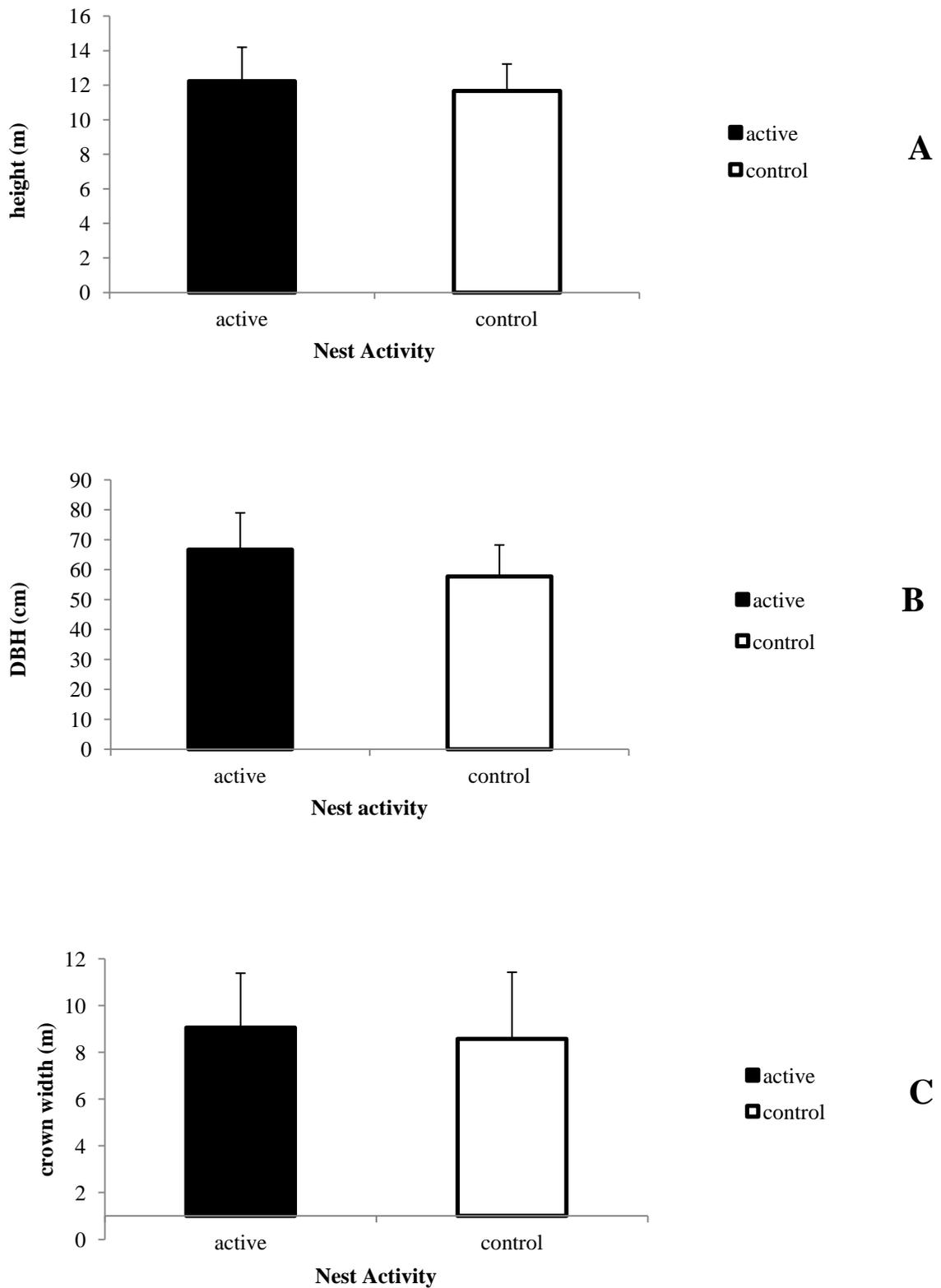


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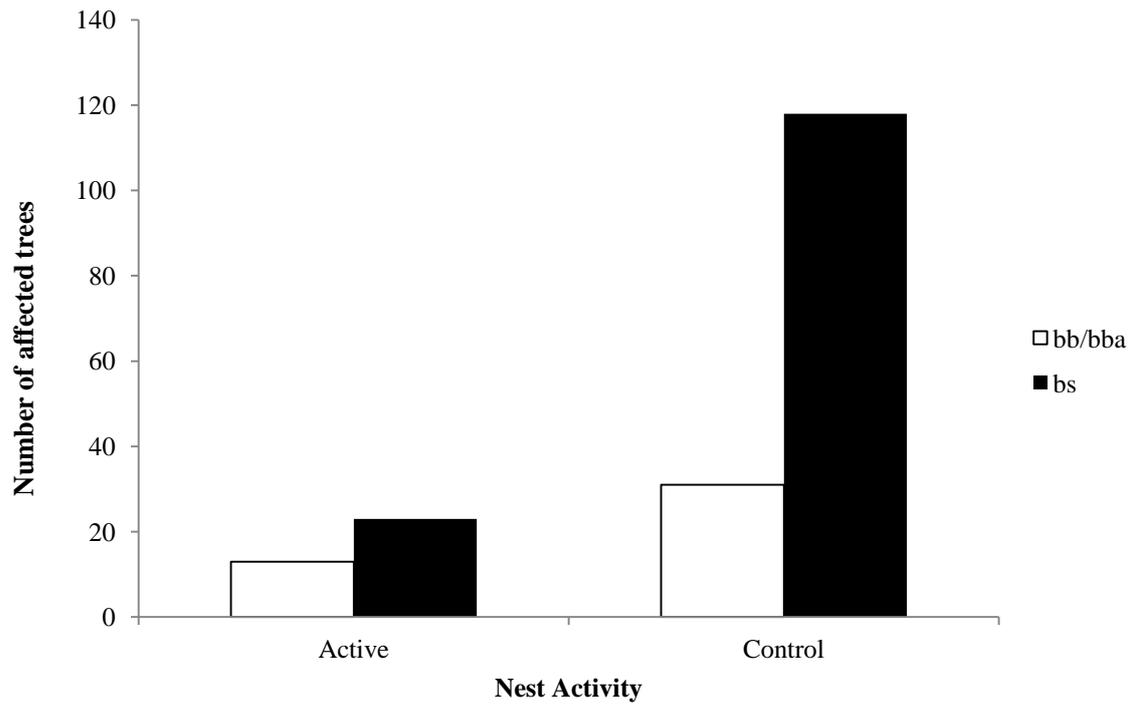


Figure 6: The impacts as a result of elephant browsing behaviours on both trees with an active nest and control trees (n = 185) (bb – branch breakage, bba – primary branch breakage, bs – bark stripping).

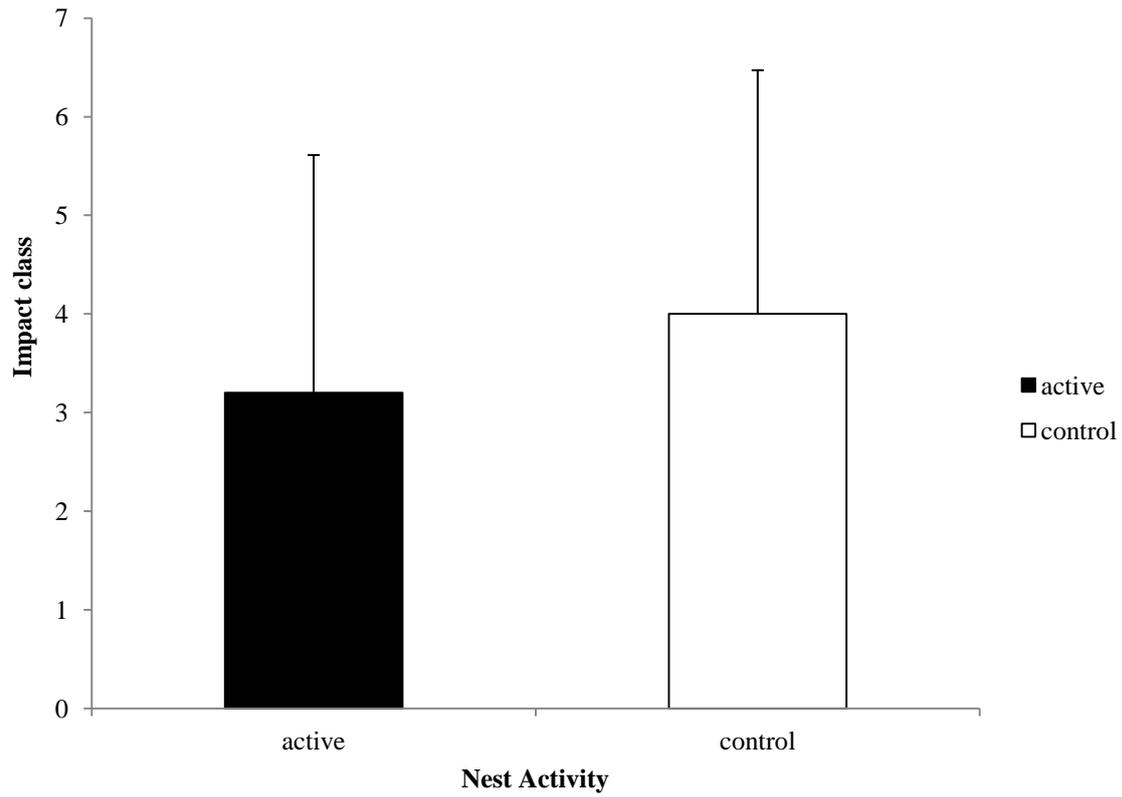


Figure 7: The mean (+SD) comparison of the elephant feeding impacts to trees that had active nests and control trees in 2012 (n = 30 for active nest trees, n = 201 for control trees).

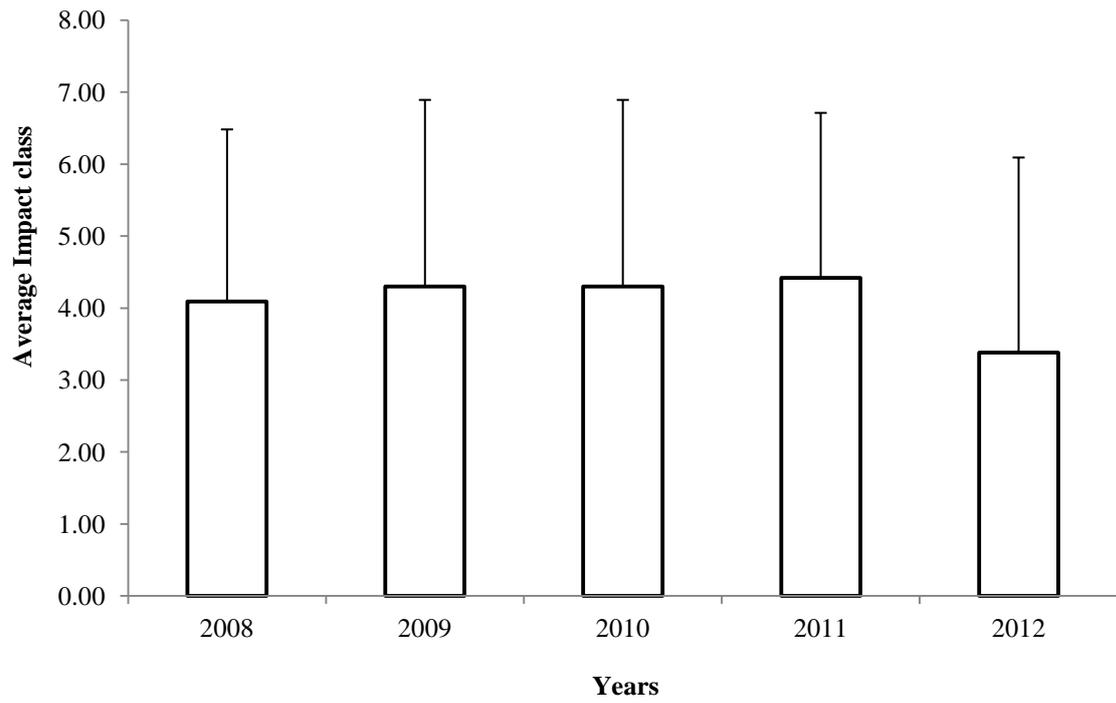


Figure 8: The mean (+SD) impact classes induced by elephant browsing behaviour to trees that had active nests over the 5 year study period (n = 30).

Appendix 1: The composition of the active, control nest and control trees with the surrounding vegetations' species compositions in the APNR for 2012. (n = 30 for active nest trees, n = 201 for control and control trees).

Nest Activity	n	Species	Surrounding species Composition
active	6	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> and <i>Acacia nigrescens</i>
active	5	<i>Acacia nigrescens</i>	<i>Combretum apiculatum</i> and <i>Acacia nigrescens</i>
active	8	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> , <i>Combretum apiculatum</i> and <i>Acacia nigrescens</i>
active	2	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> , <i>Combretum apiculatum</i> and <i>Terminalia prunioides</i>
active	4	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> and <i>flavescens</i>
active	1	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> and <i>Dichrostachys cinerea</i>
active	1	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> and <i>Lannea stuhlmannii</i>
active	2	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> , <i>Grewia Flavescens</i> and <i>Combretum apiculatum</i>
active	1	<i>Berchemia discolor</i>	<i>Grewia bicolor</i> , <i>Combretum apiculatum</i> and <i>Euclea divinorum</i>
non-active	8	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> , <i>Combretum apiculatum</i> and <i>Acacia nigrescens</i>
non-active	57	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> and <i>Acacia nigrescens</i>
non-active	46	<i>Acacia nigrescens</i>	<i>Combretum apiculatum</i> and <i>Grewia bicolor</i>
non-active	29	<i>Acacia nigrescens</i>	<i>Grewia bicolor</i> , <i>flavescens</i> and <i>heximita</i>
non-active	23	<i>Acacia nigrescens</i>	<i>Euclea divinorum</i> , <i>Dicrastachus cinerea</i> , <i>Combretum apiculatum</i> and <i>Acacia nigrescens</i>
non-active	4	<i>Acacia nigrescens</i>	<i>Lannea stuhlmanii</i>
non-active	25	<i>Acacia nigrescens</i>	<i>Terminalia prunioides</i> , <i>Combretum apiculatum</i> , <i>Grewia bicolor</i> , <i>Acacia nigrescens</i>
non-active	16	<i>Acacia nigrescens</i>	<i>Combretum hereoense</i> , <i>Acacia exuvailis</i> , <i>Dicrastachus cinerea</i> and <i>Euclea divinorum</i>
non-active	1	<i>Acacia nigrescens</i>	On a river island, surrounding by debris and sand