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RESEARCH ARTICLE

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Impacts of African savannah elephants (*Loxodonta africana*) on tall trees and their recovery within a small, fenced reserve in South Africa

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Abstract

African savannah elephants (*Loxodonta africana*) can have detrimental impacts on trees; studies exploring elephant impact within small, fenced reserves, have lacked focus in determining elephant high-use areas, the direct effect of fence line and tree recovery. The aim of this study is to assess whether elephants cause significant impact on trees in the small-fenced Karongwe Private Game Reserve (KPGR) and to determine the levels of tree recovery. Trees ≥5 m in height were surveyed (n = 634 trees). Elephant location data were used to identify high- and low-use areas. Five species accounted for 80% of the records; these were used for further analysis. Trees in high-use areas were significantly less likely to show signs of debarking and push over. Tall trees were significantly more likely to be impacted by elephants, being associated with heightened risks of debarking and branches being broken but lower risks of being pushed over. Trees close to the fence line were not more impacted than trees near the centre of the reserve. The level of use, distance to the fence and tree height were not significant predictors of tree recovery indicators. Future mitigation efforts should focus on trees with high levels of impact and low levels of recovery.

KEYWORDS
elephant density, fence, Karongwe Game Reserve, *Loxodonta africana*, space use, tree impact, tree recovery

Résumé

Les éléphants de savane africains (*Loxodonta africana*) peuvent avoir des impacts néfastes sur les arbres; les études explorant l'impact des éléphants dans de petites réserves clôturées n'ont pas été suffisamment ciblées pour déterminer les zones d'utilisation élevée des éléphants, l'effet direct de la ligne de clôture et le rétablissement des arbres. Le but de cette étude est de déterminer si les éléphants ont un impact significatif sur les arbres dans la réserve privée de gibier de Karongwe (KPGR) protégée par une petite clôture et d'évaluer les niveaux de rétablissement des arbres. Des arbres d'une hauteur ≥5 m ont été étudiés (n = 634 arbres). Les données de localisation des éléphants ont été utilisées pour identifier les zones à forte et à faible utilisation. Cinq espèces représentaient 80 % du nombre d'individus enregistrés; elles ont été utilisées à des fins d'analyse plus approfondies. Les arbres présents dans les aires à forte utilisation étaient considérablement moins susceptibles de montrer des signes d'écorçage et de renversement. Les grands arbres étaient considérablement plus susceptibles d'être touchés par les éléphants, car ils étaient associés à des risques accrus d'écorçage et de bris de branches, mais risquaient moins d'être renversés. Les arbres situés à proximité de la ligne de clôture n'ont pas été plus touchés que les arbres situés à proximité du centre de la réserve. Le niveau d'utilisation, la distance jusqu'à la clôture et la hauteur des arbres ne constituaient pas des prédicteurs significatifs des indicateurs de rétablissement des arbres. Les efforts d'atténuation à venir devraient être axés sur les arbres exposés à un haut niveau d'impact et présentant un faible niveau de rétablissement.

1 | INTRODUCTION

African savannah elephants are keystone species for savannah ecosystems (Western, 1989), helping to promote diversity of both habitats and species (Brooks et al., 1983). Elephant populations have been declining in range and numbers for decades due to ivory poaching, habitat loss and land fragmentation (Lemieux & Clarke, 2009; Thouless et al., 2016). African savannah elephant populations have decreased by at least 60% over the last 50 years, according to the latest IUCN assessment (Gobush et al., 2021). To counteract these trends, countries such as South Africa have engaged in major elephant reintroductions to protected areas, many of which include conversion of agricultural land to fenced reserves (Lehmann et al., 2008; Lombard et al., 2001). This method has proved successful in maintaining and increasing elephant population numbers; however, suitability of land, elephant densities and vegetation sustainment is often met with concerns (Caughley, 1976; Stretch et al., 2002).

In an open ecosystem without land use restrictions, elephant herd size varies depending on resource availability (Young et al., 2009). The impact of elephants on tree species in open reserves (unfenced) has been well-studied (Holdo, 2003; Levick & Asner, 2013; Morrison et al., 2016). Tree species composition has been shown to change when elephants are present in high enough densities to modify their habitat (Cumming et al., 1997). To determine the level of elephant impact, research has focussed on high profile species such as marula trees (*Sclerocarya birrea*) (Gadd, 1997; Jacobs & Biggs, 2002; Weaver, 1995; Wiseman et al., 2004). Trees with larger stems (>10 cm in diameter) are more likely to be selected by elephants for debarking, whilst smaller trees are more likely to be toppled (Boundja & Midgley, 2010; Ihwagi et al., 2012; Ssali et al., 2013). Gaugris and Van Rooyen (2010), Kerley and Landman (2006), Landman et al. (2008) and more recently Howes et al. (2020) have documented the impact that elephants have on small reserves, where natural elephant feeding behaviours are restricted by fences.

Spatial restriction of elephants within fenced reserves can exacerbate their impacts on their habitat (Baxter & Getz, 2005; Hoare, 1999; Laws, 1970). Fencing may cause elephants to become sedentary, reduce seasonal movement, prolong, and concentrate feeding impacts (Cumming et al., 1997; Guldemond & Aarde, 2008; Lombard et al., 2001). Fencing acts as a fixed boundary, where the confinements of elephants could deprive access to seasonal habitat, in turn increasing encounters of selected tree species (O'Connor et al., 2007). The potential for elephants to utilise the same patches of vegetation increases in small, fenced reserves, because of their inability to distribute themselves effectively in response to resource availability (de Boer et al., 2015; Mackey et al., 2006; Slotow et al., 2005). Additionally, the fence line itself can prove problematic, as they may cause elephants to bunch up against the fence line (Loarie et al., 2009). Movement patterns of elephants have moreover been shown to be influenced by the proximity of the fence line, with studies suggesting increased habitat use within the centre of the reserve (see e.g. Vanak et al., 2010).

Elephant impact on vegetation is known to be affected by their feeding behaviours. Potential impacts of elephants on vegetation include broken branches, the main stem being broken, debarking, the tree being pushed over, and the elephant causing the death of the tree (Table 1). Recovery levels are generally determined as the ability of individual trees to survive after elephant browsing occurs. Bark recovery (Wigley et al., 2019), coppicing (Jacobs & Biggs, 2002) and sprouting (Bond & Midgley, 2001) are generally used as indicators of tree recovery (Table 1). Elephants consume both woody vegetation and grasses, and they characteristically select vegetation depending on seasonal availability (de Boer et al., 2000; Buss, 1961; Laws, 1970; Owen-Smith, 1988). They typically feed on tree species with high nutrients in their leaves (Holdo, 2003; Jachmann, 1989; Novellie et al., 1991; Wiseman et al., 2004) and select trees with large volumes of foliage to gain maximum energy output; the level of impact has been suggested to depend on tree characteristics, such as the tree height and canopy width (Boundja & Midgley, 2010; Howes et al., 2020; Levick & Asner, 2013; Thornley et al., 2020). Private wildlife reserves are often set on degraded livestock areas, which can force elephants to utilise woody vegetation year-round due to poor grazing conditions (O'Connor et al., 2007; Smallie & O'connor, 2000). It is, however, important to acknowledge that conflicting views remain on

TABLE 1 Scale used to record elephant browsing during field observations. Impacts and recovery levels on tree species were derived from Walker (1979)

Variable	Observation
Impact	Branches broken (A)
	Condition of the tree: Alive/Dead (B)
	Main stem broken (C)
	Main trunk debarked (D)
	Pushed over (E)
Recovery	Presence of coppicing (F)
	Bark regrowth (G)
	Presence of sprouting (H)

elephant vegetation preferences and their nutritional characteristics (Scholes and Mennell, 2008). An increase in bark and roots being consumed may indicate nutritional stress, which may in turn result in greater impact on woody vegetation (Barnes et al., 1994; Guy, 1976).

Therefore, the ability for individual trees to recover after being impacted by elephants is essential for maintaining a diversity of tree species within the reserve (Kohi et al., 2011; Moe et al., 2009; Scogings et al., 2012). The resilience of tree species depends on whether species possess recruitment and regeneration rates that match the rate of mortality over time (Cumming, 1981; Lock, 1977; O'Connor et al., 2007; Thomson, 1975). Once impact has occurred, trees can recover through coppicing, regrowth as well as seedling regeneration, but success is dependent on the level of impact. Adults of some species such as mopane (*Colophospermum mopane*) have high coppicing ability (Ben-Shahar, 1996; Lewis, 1991; Styles & Skinner, 2000), while other species have weak regrowth ability (e.g. umbrella thorn (*Vachellia tortilis*)) (MacGregor & O'Connor, 2004). Most *Vachellia* and *Senegalia* populations heavily impacted by elephants have persisted through time on account of their regeneration ability (Croze, 1974; Dublin, 1995; Leuthold, 1977; Lock, 1993; Pellew, 1983; Western & Maitumo, 2004). Studies on marula trees determined that even though impacts from elephants were high, mortality rates were low as affected trees showed signs of recovery (Gadd, 2002). Small and medium-sized herbivore species including browsers such as impala (*Aepyceros melampus*) and rodents have been suggested to prevent seedling recruitment in marula trees, due to high utilisation and seed predation (Haig, 1999; Helm & Witkowski, 2012; Helm et al., 2009). Elephants may kill regenerating stems through overbrowsing, which may be exacerbated in fenced reserves (Moseby et al., 2018). There is a need to determine the recovery levels on impacted individual trees across fenced reserves with elephant presence, so reserve managers can apply appropriate measures to maintain sustainable populations of tree species while securing elephant survival. In this study, we attempt to determine the levels of tree recovery following elephant-induced impact whilst considering habitat use and fence line effect. While doing so, we test the following hypotheses:

Hypothesis 1 *Impacts will be more prevalent on tall trees (≥5 m) within high-use areas, and low levels of recovery will be displayed on these trees.*

Hypothesis 2 *Trees close to the fence line will be less impacted by elephants than trees further from the fence line, which will display lower levels of recovery.*

2 | METHOD

2.1 | Study site

The study was carried out in Karongwe Private Game Reserve (KPGR), a 7960-ha fenced private reserve in the Limpopo province of South Africa (S24.227061, E30.603302). The reserve consists of two savannah vegetation types: Granite Lowveld and Tzaneen Sour Bushveld type (Mucina & Rutherford, 2006). Average daily temperatures range from 5–17°C in winter (June–August) and 17–28°C in summer (December–February). The altitude varies from 489 m to 520 m above sea level (Lehmann et al., 2008).

The reserve originally consisted of 10 individual private farmlands, but division fences were removed in 1998 and a Reserve was established. KPGR is bordered by public roads, which are 50 m from the fence line. The western fence line (19.1 km) runs along the paved R36 route, while gravel roads run parallel along the eastern (14.5 km) and northern (11.9 km) fence lines.

Elephants were translocated to KPGR in 1999 from Kapama Game Reserve and Maggudu, Kwaza-Zulu Natal (7 individuals). Since 2011, the elephant population has consisted of one stable family unit of adult females, both male and female subadults and juveniles. There are also two bulls present on this reserve. Owen-Smith et al. (2006) suggests that an effective elephant population density is 0.28 km², so based on this estimation, KPGR could support 22.28 elephants. KPGR currently supports 20 elephants.

Elephants within KPGR are never more than 3 km from an artificial water point: there are 70 dams across the reserve; some are pumped when the water level is low. Distance to water was therefore not considered a limiting factor of elephant distribution or an explanatory variable likely to shape tree damage; it was therefore not included within our models (Harris, Russell, van Aarde, and Pimm, 2008; Shannon et al., 2008) (Figure 1).

2.2 | Data collection

Elephant locations were determined by sightings twice a day: AM drives (05:00) and PM drives (15:00), as part of a long-term study using visual recordings (data collection began in 1999). Sightings were recorded within 5 m of the observer, where the GPS recording was taken for the elephant. After locating the focal animal, the following parameters were recorded:

- Date
- Time
- Location
- Map coordinates (derived from GarminTM GPSMap® 60CSX – GNSS)

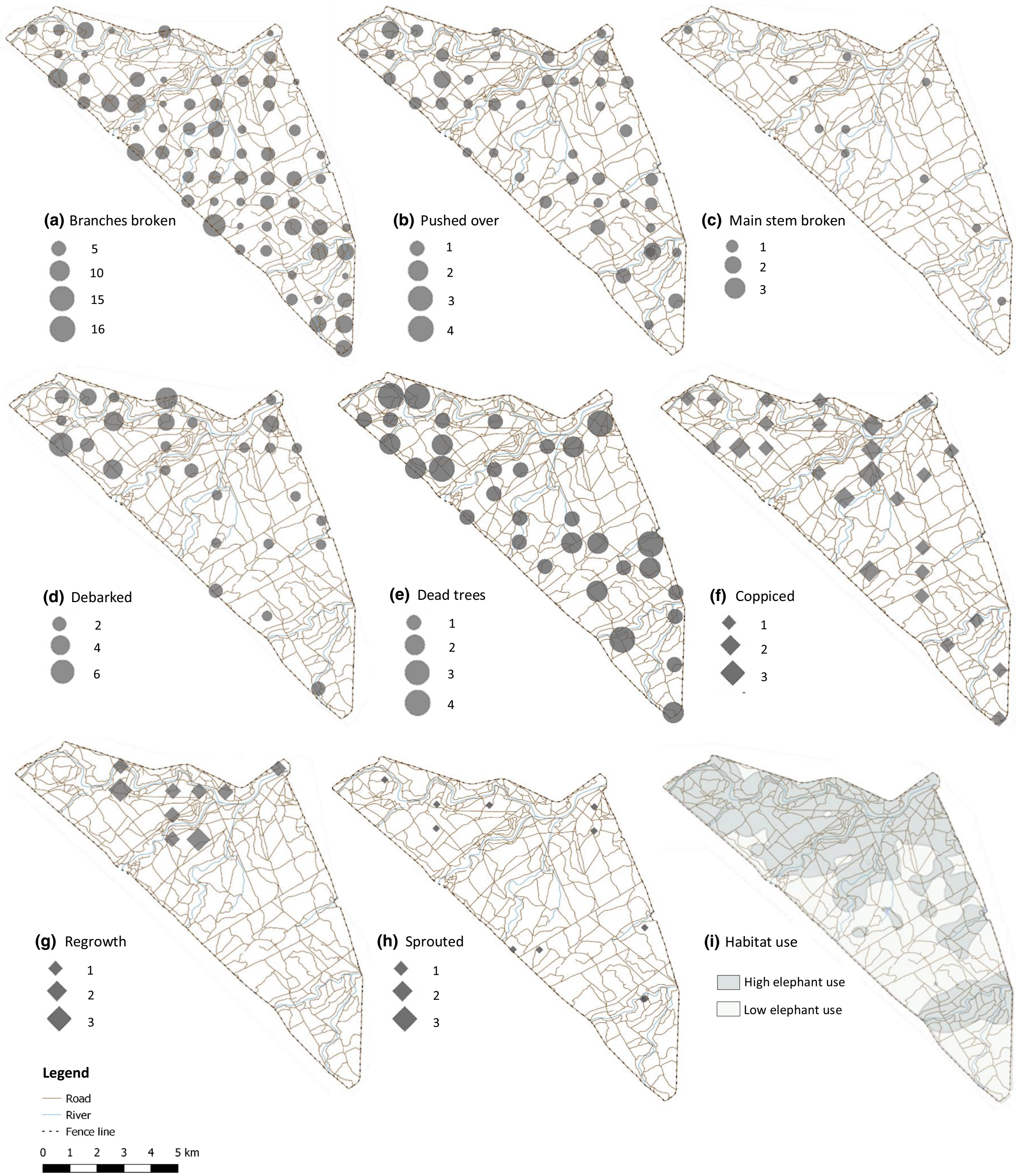

FIGURE 1 Spatial distribution of water points, vegetation transects and elephant utilisation distribution across Karongwe Private Game Reserve (KPGR)

Vegetation data were collected between July and September 2018 using 84, 10 × 100 m transects distributed across the reserve. Sampled trees near the fence line were those who were found being within 100 m of the fence line. Out of the total 84 performed, 29 transects were carried out in areas considered to be near the fence line. Navigation-grade GNSS coordinates were acquired at the start of each transect. Every tree of height ≥5 m and diameter breast height (DBH) of ≥15 cm was sampled for elephant impact and

TABLE 2 GLMM structure of dependent and independent variables (fixed effects) for all impact and recovery variables

Dependent variable	Independent variables
Branches Broken (A)	Habitat Use + Height + Habitat Use: Height + Fence line
Debarking (D)	Fence line + Habitat Use + Height + Fence line: Height + Fence line: Habitat Use
Pushed over (E)	Habitat use + Height + Fence line
Coppicing (F)	Habitat Use + Height + Habitat Use: Height + Fence line + Fence line: Height + Fence line: Habitat Use

recovery (Coetzee et al., 1979; Staub et al., 2013). Our study considered >5 m trees as research to date has focussed on elephant impact on trees within this height class as they are often targeted by elephants, but these studies lacked insights on multiple tree species within a small, fenced reserves with high elephant density (Biggs & Jacobs, 2002; Helm & Witkowski, 2012; Helm et al., 2009; Weaver, 1995). When a tree met the necessary requirements, the following parameters were recorded:

- Species, height (m), DBH (cm)
- Elephant impact type (Table 1)
- Tree recovery type (Table 1)

Tree impact types were derived from the Walker damage scale (Walker, 1979) (Table 1). Elephant impact on trees was easily distinguished from that of other browsers due to their foraging behaviours. Elephants feed on woody vegetation by breaking off branches, toppling and bark stripping using their tusks (Boundja & Midgley, 2010; Coetzee et al., 1979), whereas smaller browsers are narrowly selective for new leaves, flowers and fruits at lower heights (Owen-Smith & Chafota, 2012). We also recorded categories of recovery for each tree (Table1).

2.3 | Data analysis

'High-use' and 'low-use' areas were determined using elephant location data collected in 2018. Habitat use is described as a categorical variable, with high-use areas including areas where elephants are within their home and core range and low-use areas including areas not within their home and core range. A utilisation distribution (UD) was created to provide a measure of the probability an elephant to be found at a given location (Worton, 1989); the 'heatmap' tool in QGIS was then used to perform a quartic (biweighted) kernel density estimation (KDE) using a discrete data set to produce a continuous UD. To define the home range of the elephants, 95% of volume contours of the KDE was extracted to remove the outliners. 50% of the space use distribution, determined the elephants kernel core range (CR) and was extracted for this study.

Data exploration determined that sample sizes were too small to effectively test our two hypotheses for the following impact and recovery variables: B, C, G and H (Table 2). Therefore, Generalised linear mixed effect models (GLMMs) (binomial distribution) were used to model the likelihood of a given impact type (A, D, E, Table 2) to be found on a particular tree as a function of the height of that tree, its distance to the fence and whether or not the tree was located in an area highly used by elephants (all fixed effects). Transect identity was modelled as a random effect. We also used this approach to model the likelihood of coppicing (F; Table 2) to occur on a given tree as a function of the height of that tree, its distance to the fence and whether or not the tree was located in an area highly used by elephants (Table 2). A baseline model was constructed with all the possible interactions and main effects. Akaike information criterion (AIC) (Burnham & Anderson, 2004) and model averaging was used to select a combination of the top models. We limited the calculation of the conditional averages to models within 2 delta AIC of the best model. The conditional average for each model was used for further inference. All models were built in R using the 'lme4' package (Pinheiro & Bates, 2000; R Core Team, 2013).

Model assumptions were verified by plotting residuals for spatial dependency. We determined that the random effect approach is sufficient for spatial dependency by conducting a Moran's I test on all models (Getis, 2008). Results confirm that spatial autocorrelation is absent (p-value >0.05) in the residuals of all models.

3 | RESULTS

A total of 634 trees were considered for analysis; these data were gathered on the five most common species in the reserve: knobthorn (*Senegalia nigrescens*) 30%; marula 19%; velvet corkwood (*Commiphora mollis*) 13%; red bushwillow (*Combretum apiculatum*) 11%; leadwood (*Combretum imberbe*) 5%. 570 (90%) of these trees expressed visible signs of elephant impact. Overall, levels of impact across the reserve were thus high, but levels of recovery were low in both high-use and low-use habitats (Figure 2).

Debarking was more likely to occur on trees found in low-use areas; similarly, trees were more likely to be pushed over in low-use areas ($p = 0.04$ and $p = 0.03$, respectively). The likelihood of finding trees with branches broken was not influenced by elephant habitat use (all $p > 0.05$; Table 3).

Taller trees were significantly more likely to show signs of debarking ($p = 0.01$) and branches being broken ($p = 0.01$). However, taller trees were less likely to be pushed over ($p = 0.01$).

Trees within proximity to the fence line did not significantly experience less impacts from elephants than trees further apart from the fence line (all $p > 0.05$; Table 3). The level of use, distance to the fence and tree height were not significant predictors of tree recovery indicators.

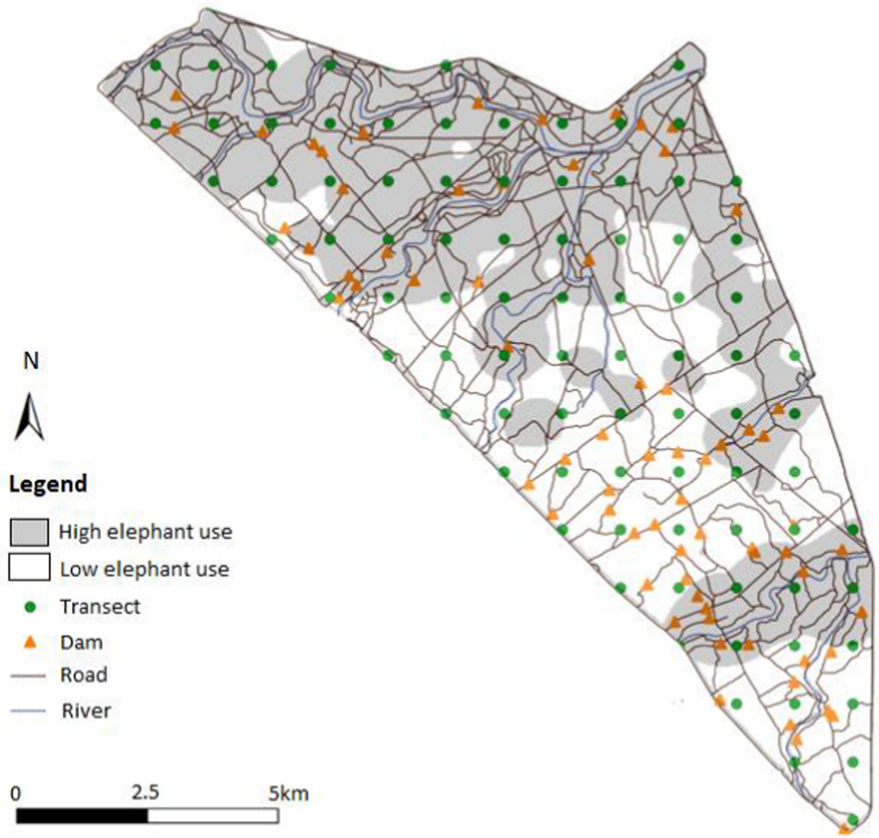

FIGURE 2 Spatial distribution of total impacts (●) and recovery (◆) on trees: (a) Branches broken; (b) Pushed Over; (c) Main Stem Broken; (d) Debarked; (e) Alive; (f) Coppiced; (g) Regrowth; (h) Sprouted and (i) Total elephant habitat use during 2018

TABLE 3 GLMM outputs of dependent and independent variables (fixed effects) for all impact and recovery variables on the best models

Variable	Fixed effects	GLMM coefficient (estimate)	SE	*Z* value	*p* value
Branches broken (A)	Habitat use	0.63	0.96	0.66	0.51
	Height	0.19	0.07	2.80	**0.01***
	Habitat use: Height	0.18	0.14	1.25	0.21
	Fence line	−0.29	0.59	0.49	0.62
Debarking (D)	Fence line	3.17	2.46	1.29	0.19
	Habitat Use	−1.07	0.52	2.07	**0.04***
	Height	0.31	0.08	3.94	**0.01***
	Fence line: Height	−0.38	0.23	1.69	0.09
	Fence line: Habitat use	−1.54	1.49	1.03	0.30
Pushed over (E)	Habitat use	−0.74	0.35	2.12	**0.03***
	Height	−0.22	0.08	2.81	**0.01***
	Fence line	0.17	0.66	0.26	0.79
Coppicing (F)	Habitat use	2.61	2.89	0.89	0.37
	Height	−0.13	0.12	1.05	0.29
	Habitat use: Height	−0.70	0.42	1.68	0.09
	Fence line	4.80	4.59	1.05	0.29
	Fence line: Height	−0.75	0.64	1.18	0.24
	Fence line: Habitat use	−2.18	2.36	0.92	0.36

*indicate significance of bold values.

4 | DISCUSSION

This study shows that African savannah elephant impact on trees does not occur randomly. Contrary to our expectations (1) the level of tree impact was not determined by the proximity to the fence line; (2) tree height and habitat use impacted differently risks of debarking, branches being broken, and tree being pushed over; (3) tree recovery could not be predicted from tree height, the level of elephant use in the area occupied by the tree, or the distance from the tree to the fence line.

The results regarding the level of impact in response to tree height were consistent with impact from elephants that have been seen on other sites across African savannahs, when considering impacts related to branches being broken and debarking. Tree height has been shown to be a significant indicator of the presence of elephant impact (Makhabu et al., 2006; Mapaure & Mhlanga, 2000; Scogings et al., 2012), which has also been the focus of many studies (Biggs & Jacobs, 2002; Cook et al., 2017; Helm & Witkowski, 2012; Helm et al., 2009). Previous studies considering elephant impacts on marula trees, have shown that tall trees between 5–11 m high showed signs of impacts, and the greatest mortality was found in trees in the 5–8 m height class (Biggs & Jacobs, 2002; Cook et al., 2017; Helm & Witkowski, 2012; Helm et al., 2009).

However, our findings do not match previous observations that the containment of elephants increases impacts on trees within the areas they mostly utilise (Cumming et al., 1997; Lombard et al., 2001). Additionally, we identified that the likelihood of a tree being debarked or being pushed over was reduced in high-use areas, which was surprising as we expected habitat use to correlate with impact. This mismatch between expectations and observations could be a result of how habitat use was determined in this study, as we only considered elephant sightings for 1 year (2018). Further work is required to establish whether the reported patterns remain consistent once more information on elephant distribution is taken into account.

We also considered the distance from the fence line in relation to tree impact, as this had not been previously studied in a small, fenced reserve. Recent research has focussed on efforts to deter elephants from fences to prevent elephants breaking out of reserves, reduce crop raiding and human–wildlife conflict (Chang'a et al., 2016; King et al., 2011; Ngama et al., 2016; Pozo et al., 2019) but little is known about the edge-effect of fences on tree damage and recovery (Vanak et al., 2010). We found no significant relationship between distance to the fence line and all impact and recovery variables. This suggests that management plans aimed at focusing efforts on reducing the impacts of elephants on trees based on their distance to the fence line may not be evidence-based.

Furthermore, we tried to identify correlates of the level of recovery on impacted trees, as this had been largely overlooked in previous studies. We considered tree recovery an important factor, as the ability for trees to recover after being impacted is essential for the long-term sustainability of tree populations (Leuthold, 1996). Our results showed a significant lack of recovery on most sampled trees, including large trees within high-use areas, which is a concern in an area of high elephant density and high levels of impact. Previous studies have emphasised that if tree species are unable to recover from impact, they will not persist through time (Kohi et al., 2011; Moe et al., 2009; O'Connor et al., 2007; Scogings et al., 2012). Even if tree species do exhibit signs of recovery, they can become sterilised if no seeds are produced (Midgley et al., 2020).

Current landscape conditions in parts of Africa, particularly in South Africa where fencing is more prevalent, limited space use increased the impacts in the savannah landscape (Loarie et al., 2009). We have attempted to address some of the explanatory factors likely to impact tree damage levels by elephants and tree recovery within this study, but there are limitations to our work. Our study is constrained both temporally and spatially, as we only collected data over one time period within one area. The study could be replicated in the wet and dry season as a comparison to determine how seasonality and water availability effects tree recovery. Tree impact could be recent, therefore, there may not have been sufficient time to display signs of recovery that could be identified during the time of study. We only looked at trees >5 m in height and focussed on the five most common species. We then had to score impact binomially, which reduced our ability to explore how responses differ according to damage level. As demonstrated in Gadd (2002), tree species may be more vulnerable to mortality if a greater percentage of bark has been stripped. Therefore, future studies should consider scoring impact levels into quantitative formats, to discern from low-level impacts and high-levels impacts (Helm et al., 2009). This would aid in identifying areas where high levels of impact occur. Future research also needs to look at information over multiple years and other factors that might impact elephant movements, such as human–wildlife conflict areas and other anthropogenic disturbances that were beyond the scope of this study.

Management of elephants on small reserves such as the study site in question is challenging as there are multiple factors to consider. This is due to the size constraints of the reserve where expansion is not a viable option, which is the case in many reserves throughout South Africa with increasing elephant population numbers. Elephant bulls have been shown to cause greater impacts to vegetation compared with cows as they are larger bodied and have more destructive tendencies, especially when in musth (Greyling, 2004). The success of contraception as a management tool to control population numbers has been shown in several small reserves in South Africa, including the Greater Makalali Private Game Reserve and Tembe Elephant Park (Bertschinger et al., 2018). Contraceptives were also administered to bull elephants in 2016 in the reserve considered in this study. Future studies should explore the impact of contraception on the behaviour of elephants in small reserves, considering any changes in behaviours and ultimately how these behaviours impact on vegetation (Stretch et al., 2002).

Our study has given an insight into impacts on trees by elephants and subsequent recovery, within a fenced reserve with high elephant density, but there is still a need to determine the long-term impacts of elephant on vegetation. We have stressed the need to focus mitigation efforts on trees with high levels of impact where recovery was not identified. A possible addition to this study would be to include trees of smaller height classes to see if impacts occur as readily to smaller trees and if so, how well they recover (<5 m in height). This has been suggested by anecdotal evidence, where elephants have shown an increase in preference for seedlings. We also suggest that future studies should consider the secondary effects of elephant impact, for example, bark stripping makes trees more susceptible to further impacts (Campbell et al., 1996; Wigley et al., 2019). As reserves vary in elephant density, methods of population control and types of vegetation vary from one reserve to another, it is imperative that we establish new holistic management methods for the sustainability of fenced reserves and to ultimately support long-term elephant conservation.

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CONFLICT OF INTEREST

The authors declare no competing interests.

AUTHOR CONTRIBUTIONS

K.T. conceived the study. K.G. performed data collection. A.P. provided some help with the analysis. N.P provided critical analysis and discussion. K.T. performed data analysis. K.T. wrote the manuscript with support of N.P., G.E. and A.F. All authors edited and approved the content.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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