
Variability in survival and mortality of *Acacia drepanolobium* Sjøstedt following prescribed burning at Olpejeta Conservancy, Kenya

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Abstract

Mortalities to *Acacia drepanolobium*, a main item in the diet of the eastern black rhino (*Diceros bicornis* Michaeli) in Olpejeta conservancy, Kenya, are caused by three main factors: drought, browsers and fire. The effect of controlled fires on *A. drepanolobium* was examined by monitoring survival and growth in experimental plots before and after fire episodes between 2003 and 2007. Density, survival and growth in *A. drepanolobium* trees were compared eighteen months before and after burning. Tall trees were more likely to die from fire compared to short trees ($R^2 = 0.575$; $df = 6$; $P = 0.048$), while seedling densities increased after fire ($\chi^2 = 36.57$; $df = 1$; $P = 0.001$). Even with increased seedling densities, burned areas attracted large numbers of seedling predators, lowering the possibility of seedling recruitment into adult, as mean seedling heights reduced significantly (ANOVA, $F = 204.42$; $df = 1$; $P = 0.036$). Fires also significantly lowered flowering ($F = 346$; $df = 1$; $P < 0.05$) in *A. drepanolobium*, thereby affecting fruit production. Although fires caused mortalities to adult *A. drepanolobium*, the most significant effect was tree reversals into seedling height class as trees resprouted. Although fire may increase browse biomass of *A. drepanolobium* available for black rhino, it is not an appropriate black rhino habitat management tool because burnt areas attract many seedling predators that lower seedling recruitment into adult trees.

Key words: *Acacia drepanolobium*, fire, mortality, Olpejeta, recruitment, survival

Résumé

La mortalité d'*Acacia drepanolobium* dans l'aire de conservation d'Olpejeta, au Kenya, est causée par trois facteurs

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principaux, la sécheresse, les herbivores et les feux. On a examiné l'effet des feux contrôlés sur *A. drepanolobium* en suivant la survie et la croissance dans des parcelles expérimentales 18 mois avant et après les feux, entre 2003 et 2007. On a comparé la densité, la survie et la croissance d'*A. drepanolobium* 18 mois avant et après les feux. Les grands arbres étaient plus susceptibles de mourir à cause des feux que les petits ($R^2 = 0,575$; $ddl = 6$; $P = 0,048$) alors que la densité des jeunes plants augmentait après les feux ($\chi^2 = 36,57$; $ddl = 1$; $P = 0,001$). Même si la densité de jeunes plants était accrue, les zones brûlées attiraient de grands nombres de prédateurs de ces plants, diminuant ainsi la possibilité de recrutement vers l'âge adulte, parce que la hauteur moyenne des jeunes plants était significativement réduite (Anova, $F = 204,42$; $ddl = 1$; $P = 0,036$). Les feux réduisaient aussi significativement la floraison ($F = 346$; $ddl = 1$; $P < 0,05$) chez *A. drepanolobium*, affectant dès lors la production de fruits. Même si les feux causaient une certaine mortalité chez les *A. drepanolobium* adultes, l'effet le plus significatif était le retour des arbres dans des classes de hauteur des jeunes plants lorsque les arbres rejetaient. Même si les feux peuvent augmenter la biomasse de brouillis d'*A. drepanolobium* disponible pour les rhinos noirs, ils ne sont pas un outil de gestion approprié pour l'habitat des rhinos parce que les zones brûlées peuvent attirer des nombreux prédateurs des jeunes plants qui réduisent le recrutement de ceux-ci vers l'âge adulte.

Introduction

Fire is an ecological process that profoundly affects plant community composition, structure and function (Briggs & Knapp, 2001). Fire influences changes in vegetation on its

own and in conjunction with other factors like herbivory, climate, and soils (Whelan, 1995). Many studies have demonstrated that fire may cause short-term changes in forage quality and availability that attracts ungulates to recently burned areas (Wilsey, 1996). Managers therefore often manipulate natural fires or apply prescribed burning to areas where expected increase in productivity may benefit wild or domestic herbivores, particularly grazing ungulates (Van Dyke & Darragh, 2006). Once attracted to a burned site, grazers facilitate changes that alter community structure and function, including increased rates of nutrient cycling (Risser & Parton, 1982), reduced litter and fuel accumulation (Hobbs *et al.*, 1991; Frank & Groffman, 1998), and increased species richness (Collins, 1987; Puerto *et al.*, 1990).

Although it is believed that *Acacia drepanolobium* has increased in density and expanded to areas where it had previously been absent (see Pratt & Gwynne, 1977) many ranches in Laikipia are harvesting it for charcoal or for bush control (Okello, O'Connor & Young, 2001). Pratt & Gwynne (1977) and Dall, Maas & Isselstein (2006) proposed that fire was an appropriate management tool for encroaching *A. drepanolobium*. *Acacia drepanolobium* is important for the conservation of the endangered black rhino at Olpejeta as it contributes up to 75% of the rhinos' diet at the site (Birkett, 2002). In Olpejeta however, *A. drepanolobium* is facing threats from large concentrations of megaherbivores that are fenced in and using it as the main food item (see Birkett, 2002). At the same time, there has been need to use controlled fires to improve habitat for grazers that constitute the highest densities of wild ungulates in the conservancy. Controlled fires are used to remove moribund grass and allow it to rejuvenate as well as control bush encroachment especially of *Euclea divinorum*. However, although fires and *Acacias* are vital components of the African savanna dynamics, little is known about the impact of fire on *Acacia* life cycles (Midgeley & Bond, 2001). Midgeley & Bond (2001) express the need for more research relating fire-sensitivity to size and age of *Acacia* species. Studies on the relationship between shrub size and resprouting ability for example have not generated consistent results the relationship is negative for woody species in wet prairies; (Pendergrass, Miller & Kauffman, 1998), not significantly different from zero for a fynbos conifer (Keeley, Keeley & Bond, 1999), and positive for several shrubs including *Acacia* species (Hodgkinson, 1998).

Acacia drepanolobium can either be a tree or shrub that grows to 7.5 m in height and has swollen thorns that are

inhabited by ants (*Crematogaster mimosae*, *C. nigriceps*, *C. sjoestedtii* and *Tetraponera penzigi*) (Coe, 1993). *Acacia drepanolobium* forms nearly mono-dominant stands in many arid uplands with impeded drainage in East Africa (Okello & Young, 1999). In the Laikipia plateau of north-central Kenya, *A. drepanolobium* forms a virtual monoculture, often comprising more than 90% of the total tree and shrub density in the black cotton soils (Young, Stubblefield & Isbell, 1997).

In this study, we report the effect of controlled burns between 2004 and 2007 on the survival and recruitment of *A. drepanolobium*, which constitutes the main diet of the black rhino in Olpejeta conservancy.

Materials and methods

Olpejeta Conservancy (formerly Sweetwaters Game Reserve) is located in central Kenya, 230 km north of Nairobi, near Nanyuki, on the Laikipia plateau between Mt. Kenya and the Aberdare Mountains. It lies at an altitude of 1800 m on the equator 368560E. The vegetation is a mosaic of grassland, *Acacia* woodland, *Euclea* scrub woodland and riverine woodland (Birkett, 2002). The Conservancy is on the southern and wettest area of the greater Samburu-Laikipia ecosystem with a mean annual rainfall of 900 mm and a bimodal rainfall pattern. The conservancy is currently the largest Black rhino Sanctuary in East Africa with a population of 80 black rhinos (*Diceros bicornis*). Other large mammal herbivores at the study site include elephants (*Loxodonta africana*), giraffes (*Giraffa camelopardalis*), cape buffalos (*Syncerus caffer*), elands (*Taurotragus oryx*), Grevy's zebras (*Equus grevyi*), Burchell's zebras (*Equus burchelli*), Beisa oryx (*Oryx beisa*), Jackson's hartebeests (*Alcelaphus buselaphus jacksoni*), Waterbuck (*Kobus defassa*), Grant's gazelles (*Gazella granti*), steinbucks (*Raphicerus campestris*) and domestic cattle. Predators include Lion (*Panthera leo*), leopard (*Panthera pardus*), cheetah (*Acinonyx jubatus*), wild dog (*Lycan pictus*), Silver-backed jackal (*Canis mesomelas*) and Spotted hyena (*Crocuta crocuta*). Primates include Olive baboon (*Papio anubis*), Patas monkey (*Erythrocebus patas*), Vervet monkey (*Cercopithecus aethiops*) and Lesser bushbaby (*Galago senegalensis*).

Two species, the whistling thorn tree *A. drepanolobium* and the multi-stemmed shrub *Euclea divinorum* dominate the conservancy (Birkett, 2002). Currently the whistling thorn is one of the principal foods of the black rhinoceros, comprising more than 75% of its diet in this conservancy

(Birkett, 2002). *Acacia drepanolobium* has undergone a lot of browse pressure from elephants, rhinos, giraffes, drought (Birkett, 2002) as well as controlled fires since 2005.

A long-term study set up to monitor changes to the *A. drepanolobium* woodlands (Birkett, 2002; Birkett & Stevens-Wood, 2005) has now run for ten years. Results of the impacts of the three megaherbivores and drought on the *A. drepanolobium* before fire was introduced in 2005 are contained in Birkett (2002) and Birkett & Stevens-Wood (2005). This study examines the effect of fire on *A. drepanolobium* by comparing their performance before and after 2005.

Tree and seedling measurements

Acacia drepanolobium trees and seedlings from eleven of the 30 plots set up in 1998 (See Birkett, 2002; Birkett & Stevens-Wood, 2005) were subjected to fire treatment between 2005 and 2006. Each tree was identified with a numbered aluminium tag fixed to trees above 1 m by a nail and to seedlings below 1 m by a plastic covered wire. The plot locations were recorded by GPS, accurate to 100 m and the individual tree locations by bearing and distance (in metres) from the first tree (see Birkett, 2002). Each plot initially contained between 40 and 60 tagged trees selected as a stratified sample of seven height classes. Data from 528 trees and seedlings that had been monitored since at least the year 2000 were available before and after fire. We compared tree and seedling performance data eighteen months before and after fire. The height and damage status (damaged by fire but alive or dead) of each tree and seedling were compared before and after fire. Tree height was measured to an accuracy of 2 cm using a Dynamis Telescopic Measuring Rod (Stanton Hope, 1998) that could be extended to 7 m. Measurements were recorded in the field using a US Robotics 3ComPalmPilot Professional loaded with software supplied by the University of Kent (Pascoe, Morse & Ryan, 1998). Data were then downloaded from the Palm Pilot into Microsoft Excel 97 spreadsheets and later analysed using SPSS (SPSS Inc., Chicago, IL, USA).

Results

In all the eleven burnt plots, mean tree heights significantly reduced after burning from a mean of 111.38 ± 49.45 to 27.33 ± 15.12 cm (ANOVA;

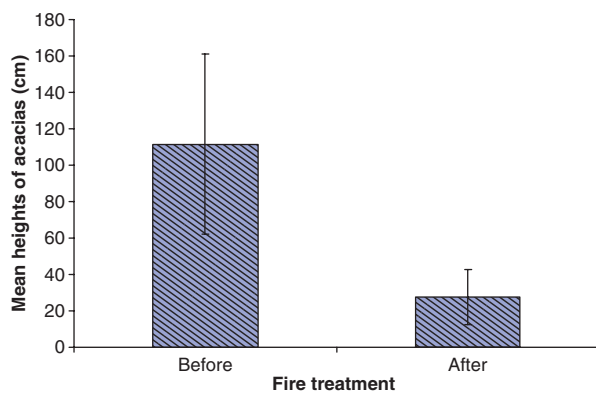


Fig 1 Mean heights of *Acacia drepanolobium* measured eighteen months before and after fire treatment

$F = 366.73$; $df = 1$; $P = 0.009$) (Fig. 1). Most tall trees, especially those above 3 m tend to have desiccated bark and colonies of lichen that make them very susceptible to fire than shorter trees. These trees were seen to dry from top and drop off chunks of their canopy after the first year of burn. The burnt trees were also more susceptible to breakage by animals rubbing and scratching against them. All these factors lead to rapid reductions in heights.

Measurements taken of trees within the first year following fire indicated no significant change in height. During this interval, no mortalities were recorded but growth was arrested following death of growth meristems. After the first year however, calculated mean growth of trees started to decrease rapidly following rapid decreases in heights.

We compared mean growth rates eighteen months before and after fire using a one-way ANOVA. We define growth here as change in height between any two time periods so that dead trees or burnt trees that broke off recorded negative growth. We recorded significant differences between mean growth rates before and after fire ($F = 245.24$; $df = 1$; $P = 0.0001$). Mean growth rates before fire were significantly higher (2.86 ± 7.44 cm) compared to after fire (mean = -5.23 ± 10.22 cm) (Fig. 2).

We categorized individual trees into each of seven height classes before the fire episodes in each plot. Height classes were 1 (≤ 1 m), 2 ($>1 \leq 2$ m), 3 ($>2 \leq 3$ m), 4 ($>3 \leq 4$ m), 5 ($>4 \leq 5$ m), 6 ($>6 \leq 7$ m) and 7 (>7 m). For all height classes, mean growth significantly reduced following burning but some height classes were more affected than others (Fig. 3). When the effect of fire on growth across

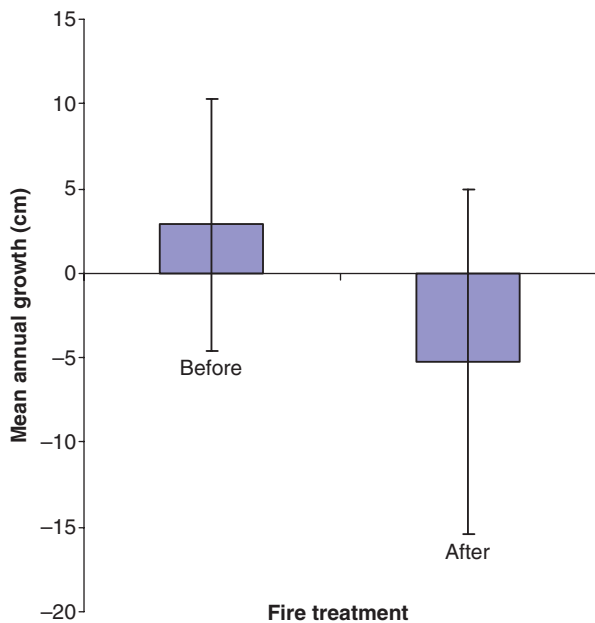


Fig 2 Mean annual growth of *Acacia drepanolobium* measured eighteen months before and after fire treatment

height classes was analysed using linear regression, there was a significant increase in mortality as height increased; difference in the mean growth across height classes increased with height ($F = 6.77$; $df = 6$; $P < 0.05$). However, the most affected height class was of trees above 7 m that also suffered significant additive mortality from drought.

We defined seedlings as plants below 1 m in height before burning. For *A. drepanolobium* at Olpejeta, trees above this height are commonly reversed into the height class

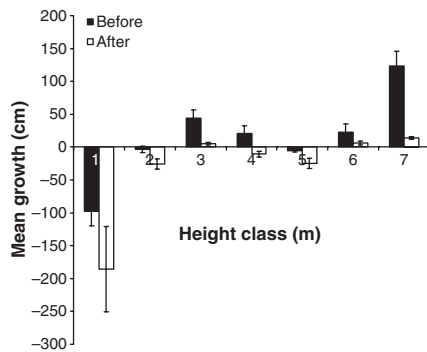


Fig 3 The effect of fire on *A. drepanolobium* showing differences in heights before and after fire on each height class

especially by browsing rhinos or from breakage and browsing by elephant. We considered all plants, even old reversed trees below half a metre as seedlings. Our results show that burning seems to favour seedling establishment by increasing after-burn seedling density (Fig. 4).

Seedling density may have increased as a result of sprouting, germination and tree reversals. In burnt areas however, seedlings may take long to recruit into trees as they are more exposed and suffer heavy seedling predation and reversals, with resultant decreases in heights. We observed that mean seedling heights significantly reduced (ANOVA, $F = 204.42$; $df = 1$; $P = 0.036$) by about half from 32.77 ± 13.22 to 17.96 ± 9.12 cm after burning (Fig. 5).

We examined the effect of fire on flowering and fruiting in mature *A. drepanolobium*. Significantly more mature trees (twenty times more) flowered in the unburnt plots

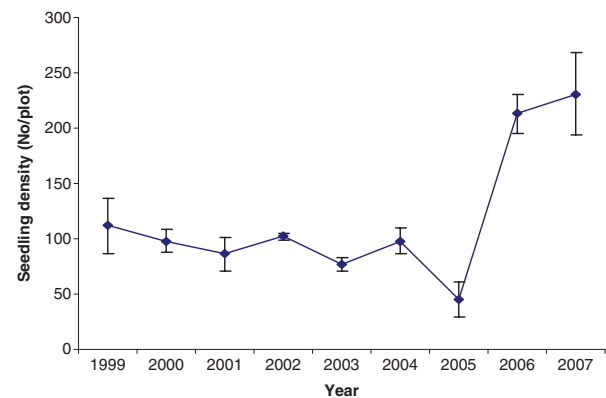


Fig 4 The effect of fire on *A. drepanolobium* showing the increase in mean seedling density following fire for eight plots burned in 2005

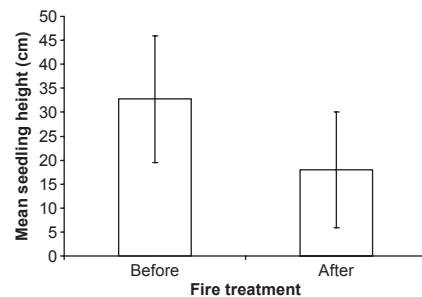


Fig 5 The effect of fire on *A. drepanolobium* showing the decrease in mean seedling height following fire for eight plots burned in 2005

than in burned plots ($F = 346$; $df = 1$; $P < 0.05$) in 2007 (Fig. 6). We note here that flowering and fruiting at Olpejeta are affected by both fire and giraffe browsing. We did not control for giraffe browsing but took the assumption that giraffe would in any case feed uniformly across both treatments or tend to affect more of the unburnt trees.

We observed that although trees were affected by fires, 45% of the trees that appeared to have been killed by fire sprouted at the bottom within the first year of burn, even though highly significant reduction in height continued to be recorded from the crown. Eventually, most of these trees were reversed into seedling class.

Discussion

Increase in woody cover of *Acacia* woodlands often leads to declines in the productivity of understory grass or herb layer species (Sabiiti & Endroma, 1991). In Olpejeta, the use of prescribed fires in range forage management is aimed at improving forage for grazers. Elsewhere in Laikipia plateau, fires are currently infrequently used by pastoralist to control bush encroachment in *Acacia* dominated woodlands to enhance fresh growth of forage for grazers (Okello *et al.*, 2007). In Olpejeta conservancy, *A. drepanolobium* is one of the key forage species for the endangered eastern black rhino (*Diceros bicornis* Michaeli). The density and distribution of this species is therefore important and will influence distribution and habitat use of the black rhino. Therefore, the ability of *A. drepanolobium*

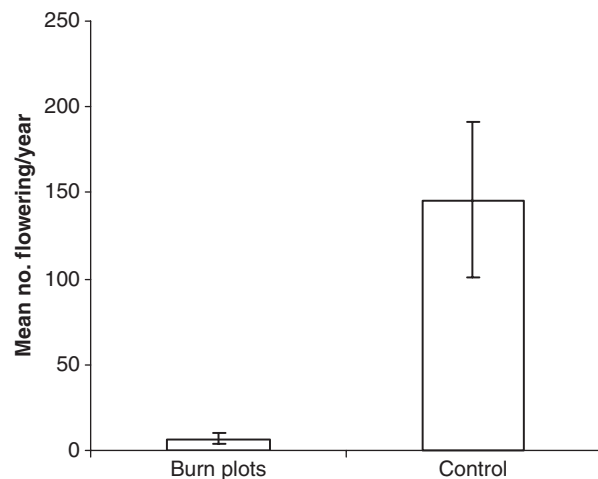


Fig 6 The effect of fire on *A. drepanolobium* showing the difference in the number of trees in flower between burnt and unburnt plots

to flower and regenerate is of major importance to the conservation of the black rhino. Germination and early growth of *Acacias* have been reported to be the major phases in the population dynamics of *Acacia* species (see Sringer Fenner, 1980), as they determine their density and distribution (Oba, 1990), and because these phases are of importance because of the high mortalities that characterize them (Coe & Coe, 1987).

Burning attracts a wide range of grazing and browsing herbivores to the affected site. Frank, McNaughton & Tracy (1998) report that there is a close association between grazing ungulates and the spatio-temporal arrangement of such forage, as ungulates apparently track changes in forage production and nutritional levels to increase their grazing efficiency and diet quality. We observed that ungulates concentrated on the burned sites for several months, and transformed the entire landscape into maintained grazing lawns. To explain the lag in adult tree mortality following fire, Okello *et al.* (2007) hypothesized that large mammalian herbivores prefer to visit burnt areas or to feed upon individual trees that have experienced a burn, perhaps either because of higher quality grass forage or because of increased visibility. Our own observations are not consistent with their hypothesis; although we observed heavy use of burned areas by herbivores, most animals using these areas were grazers and browsers on seedlings while large browser damage was negligible. It appeared as if elephants, for example avoided these burned areas in the first two years after burn.

The effect of fire on at Olpejeta is more pronounced because of its reversal effect rather than in mortality. It has already been shown that topkill is much more frequent than complete mortality after fire in *Acacias* (Hoffmann & Solbrig, 2003), because fire stimulates below-ground tissues (Pendergrass *et al.*, 1998). This study reported that trees of all height classes were reduced in height with most being reversed to seedling class rather than being killed. Okello *et al.* (2007) in their study on the effect of fire on *A. drepanolobium* at Mpala near Olpejeta observed that virtually all trees had survived the fire nine months after the burn. Similarly, Meyer *et al.* (2005) observed that 91% of *Acacia mellifera* shrubs survived two years after a fire in a semi-arid South African savanna. In our case, nearly half of the trees coppiced from the bottom after the first year of burn but continued to reduce in height as dead stems broke or were pushed over by animals. Such low mortalities and high resprouting abilities of savanna woody

species are in accordance with earlier findings (Keeley *et al.*, 1999; Owens, Mackley & Carroll, 2002).

Contrary to our own observations however, that tall trees suffered significantly more mortality than smaller trees in the first year, Okello *et al.* (2007) reported more mortality on smaller trees. Nevertheless, Okello *et al.* (2007) clarify that this observation where trees below 2 m suffered more damage was because the average fire height was 1.5 m and fire temperature decreased with height.

However, the two studies show consistent results with regard to long-term mortality, where smaller trees survive better than large trees. This is because tall trees continued to be broken or pushed over by animals such as elephants, buffalo and zebra after rubbing or scratching against them (see also Okello *et al.*, 2007). An alternative explanation for this observation is that tall and older trees suffer from an additional senescent effect caused by an accumulation of physiological dysfunction (Meyer *et al.*, 2005). Our findings also support those of Meyer *et al.* (2005) for *A. mellifera* where mortality after fire was higher in taller shrubs than in shorter ones. Elsewhere, mortalities in savanna woody species reported decreasing postfire mortalities with increased size (Hodgkinson, 1998; Hoffmann & Solbrig, 2003).

Fires may not be the best tool in enhancing germination in *A. drepanolobium*. Despite its reported effect on seed germination, fire is not a prerequisite for field germination (Cavanagh, 1980). Although burning leads to increased soil temperatures that may enhance earlier germination in some species (e.g. Antos, McCune & Bara, 1983), Okello & Young (1999) observed that fire killed most of *A. drepanolobium* seeds in an experiment in a nearby ranch to Olpejeta. They observed that fire is unlikely to play a significant role in the germination of *A. drepanolobium*, but rather may be an effective tool in the control of *A. drepanolobium* regeneration from seed.

Tree recruitment is likely to be limited by survival of tree seedlings, since plants are usually most vulnerable to herbivores at this stage. By killing seeds and thereby preventing germination, fire will then have a significant role in suppressing new seedling recruitment with a subsequent implication for the population dynamics of *A. drepanolobium* at Olpejeta (see also Harper, 1982).

Okello *et al.* (2007) observed that in addition to adult mortality to *A. drepanolobium*, few surface seeds of *A. drepanolobium* germinated suggesting that fire may reduce recruitment. In this study, we observed that fire inhibited flowering in *A. drepanolobium* with 20 times more trees

flowering in unburnt plots than in burnt plots. Reduction in flowering and consequently in fruiting would have a significant consequence for the self propagation of the species.

Burning may be a useful tool in habitat management for savannahs but may be suitable for certain habitat types and not others. In Olpejeta conservancy, our results show that whereas burning is a useful tool in turning moribund grass vegetation into nutritious and reproductive forage, it is detrimental to *A. drepanolobium* woodlands because it reverses growth in trees through topkill, and reduces flower and fruit set in mature trees. Burning also results in reduced seedling recruitment as burnt areas attract seedling predators.

We therefore recommend that to maintain a stable and productive habitat for black rhino, controlled burning should be practiced away from *A. drepanolobium* dominated or mixed woodlands.

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