
Effect of selective logging on tree and understory regeneration in miombo woodland in western Tanzania

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Abstract

Tree regeneration and understory response to selective cutting of *Pterocarpus angolensis* DC, and *Sterculia quinqueloba* Sim, was measured outside Katavi National Park, Tanzania. Contrary to expectations, a selective harvest had no effect on tree recruitment for either species in this miombo woodland. In unlogged plots along transects taken at increasing distance from a main road, the stand densities of young trees declined away from the road. Since anthropogenic activity along the road results in a high fire frequency but low native ungulate densities, the increased rate of tree regeneration near the road may result from either reduced grazing and browsing pressure by native ungulates, or reduced fire intensity and severity along roads. These effects appear to mask any possible effect of selective harvest on tree recruitment. Currently, there is no evidence of compensatory recruitment of trees of any species into the canopy to replace logged trees and this will lead to a gradual thinning in overstory stand density.

Résumé

On a mesuré la régénération des arbres et la réponse de l'étage végétal inférieur à la coupe sélective de *Pterocarpus angolensis* DC et de *Sterculia quinqueloba* Sim en dehors du Parc National de Katavi, en Tanzanie. Contrairement à ce qu'on attendait, une récolte sélective n'a aucun effet sur la repousse des arbres d'aucune espèce dans cette forêt à miombo. Dans les parcelles, où il n'y a pas eu de coupe, situées le long de transects choisis à une distance de plus en plus grande de la route principale, la densité des groupes de jeunes arbres diminuait en s'écartant de la route. Étant donné que les activités humaines le long de la route aboutissent à une haute fréquence des feux mais à une faible densité d'ongulés locaux, le taux accru

de régénération des arbres près de la route peut être le résultat soit d'une diminution de la pression du broutage par les ongulés locaux, soit d'une diminution de l'intensité et de la gravité des feux le long des routes. Ces effets semblent masquer tout effet éventuel de la récolte sélective sur la repousse des arbres. Actuellement, il n'y a aucune preuve d'une repousse compensatoire d'arbres de quelque espèce que ce soit dans la canopée pour remplacer les arbres coupés, et ceci va entraîner un éclaircissement progressif de la densité de l'étage supérieur.

Introduction

Numerous factors affect the structure and composition of dry woodland, or miombo, in south-central Africa. For convenience, these can be categorized into abiotic and biotic factors. Abiotic factors include edaphic components such as nitrogen or phosphorus concentrations (Stromgaard, 1992; Chidumayo, 1994) and disturbances such as fire (Kikula, 1986; Chidumayo, 1988a). Biotic factors can be divided into naturally occurring forces such as damage by herbivores (Campbell *et al.*, 1996), especially by elephants (*Loxodonta africana* Blumenbach) (Cumming *et al.*, 1997); and anthropogenic factors, including commercial charcoal production (Monela, O'Kting'Ati & Kiwele, 1993), collection of fuel wood (Abbot & Homewood, 1999) and cutting of building poles (Luoga, Witkowski & Balkwill, 2000). Of these, anthropogenic factors give cause for the greatest concern with respect to maintenance of forest diversity. Although elephants can have a dramatic adverse effect on the regeneration of some species (Guy, 1989; Western & Gichohi, 1993) their populations are relatively stable or declining in most areas. In contrast, human populations are growing rapidly in southern African countries and charcoal is still the cheapest form of fuel in most areas, including the area studied here (e.g. Vermeulen, 1996).

Many studies conducted in miombo woodlands have attempted to document various types of exploitation by

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local people (McGregor, 1994) and to estimate the amount of woody vegetation remaining (Chidumayo, 1987, 1988b, 1993; Grundy, 1995). Within this context, we can divide anthropogenic woodland use into categories that range in intensity: light fuel wood coppicing, selective harvest of canopy trees for boards, multi-species harvest for charcoal production, and land clearing. Surprisingly, however, the response of the miombo woodland to most of these forms of disturbance has received little attention. This is unfortunate, because regeneration within exploited woodlands will determine the extent to which they can sustain exploitation. If tree regeneration increases in response to selective logging, then sustained use is at least possible. If, however, the removal of woody vegetation results in little tree regeneration, or a change to a different woodland structure, then heavy exploitation of timber resources could have adverse effects on species composition and possibly on the animal species living there (see Campbell *et al.*, 1996).

To investigate the effect of a common form of anthropogenic disturbance in miombo, we examined the effects of selective logging on tree and woodland regeneration in western Tanzania. In this region, the commercial logging of two species is increasing as timber resources are depleted. *Pterocarpus angolensis* is felled, cut into boards and transported by road to a railhead in Mpanda, Rukwa Region, and then sent to the capital, Dar es Salaam, for furniture construction (Schwartz *et al.*, 2002). *Sterculia quinqueloba* is felled and cut, and then used locally in the region as a building material, principally for rafters (Caro, 1999a). Because this form of exploitation involves cutting individual trees but little other disturbance, we used pair-wise comparisons to test the effects of a selective removal of trees on sapling regeneration and understory cover. We also surveyed woodland composition in 0.1 ha undisturbed plots across a wider area, in order to determine the extent to which non-anthropogenic, biotic factors, such as fire and herbivory, affect tree regeneration in this little studied region of the country.

Methods

The study area

The study site was in, and immediately south of, Katavi National Park (KNP), latitude 6°45'–7°05'S, longitude 30°45'–31°25'E at the north end of Rukwa Valley in

Rukwa Region, western Tanzania. The site consisted of three legally designated areas (Fig. 1). Katavi National Park consists largely of miombo woodland, characterized by a variety of species in the following genera: *Acacia*, *Combretum*, *Grewia*, *Kigelia*, *Pterocarpus* and *Terminalia* (Lawton, 1978; Rodgers, 1996) but also encompasses two seasonally inundated flood plains, Lakes Katavi and Chada (Caro, 1999a,b). No temporary or permanent settlements are allowed, except for Park headquarters and two outlying ranger posts; no livestock, beekeeping, hunting, fishing or timber extraction are tolerated. Tanzania National Park rangers conducting vehicle and foot patrols enforce these laws.

The second area is Msaginia Forest Reserve (FR), immediately adjacent to and north-east of the Park. Settlements, cattle grazing and tourist hunting are forbidden but hardwood extraction is allowed under license. By far the most commonly cut hardwood species is *P. angolensis*, but *Swartzia madagascariensis* Desv. is taken if encountered. The third, Usevya Open Area (OA), immediately to the south of the Park boundary, sanctions permanent settlement, most notably in three small villages (Mirumba, Manga and Kibaoni), and allows for selective tree cutting under license. Although people cultivate, graze cattle, collect grass and firewood, cut hardwood, keep beehives, and hunt illegally in OA, the woodland remains largely intact within 5 km of the Park boundary. North of the villages, where we were working, *S. quinqueloba* is the principal hardwood species harvested, as *P. angolensis* does not grow in the area.

In both the FR and OA, tree cutters walk in from the main Sitalike to Kibaoni road (Fig. 1), cut trees, roll them to sawpits, and cut them into boards. Minor logging tracks connect the sawpits to the main road and are used by lorries to haul the finished boards. In August and September 2000 during the middle of the dry season, we sampled vegetation close to these tracks and at increasing distances from the main road.

Regeneration under trees

To assess local effects of selective logging, we sampled 10 pairs of trees of each focal species, where one tree had been felled while the other had not. We estimated that the trees had been felled between 2 and 5 years prior to our study. These estimates were based on personal communication with local townspeople as well as the condition of the stumps. Both *S. quinqueloba* and *P. angolensis*

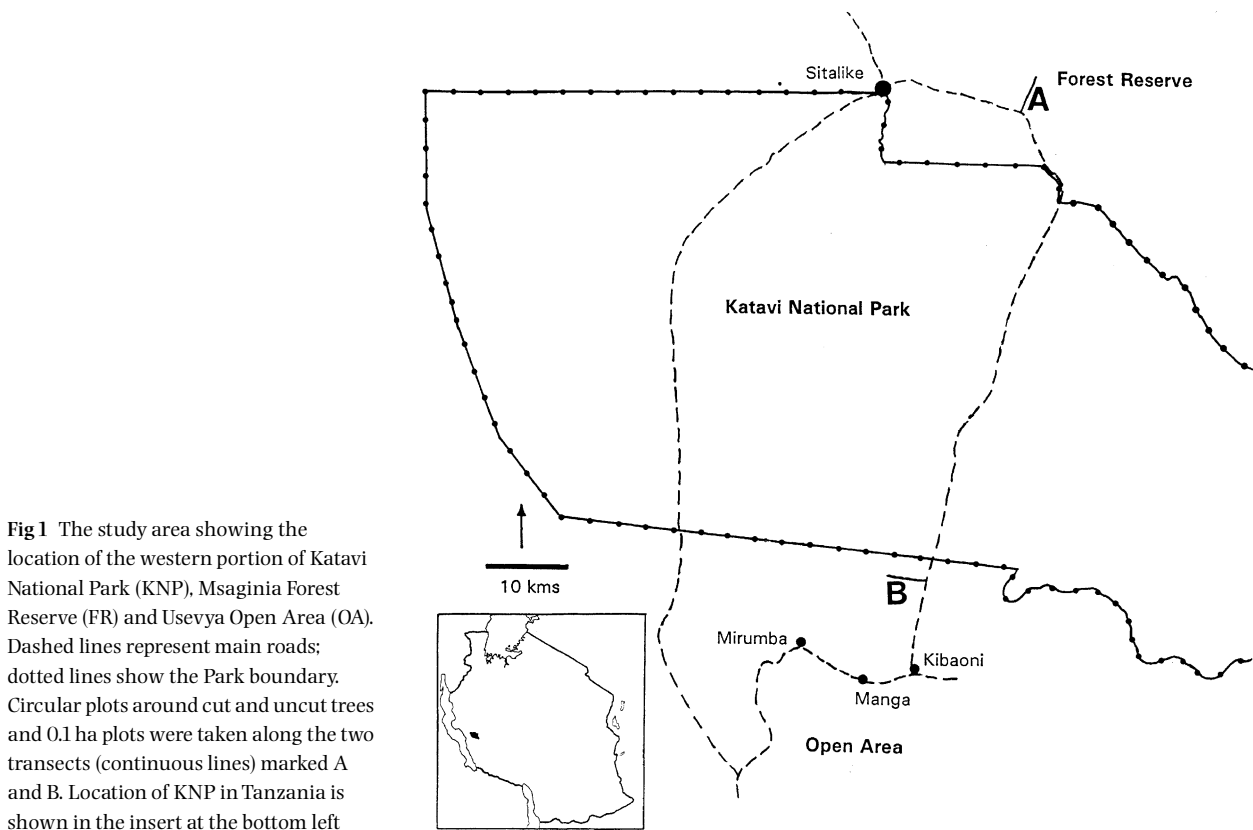


Fig 1 The study area showing the location of the western portion of Katavi National Park (KNP), Msaginia Forest Reserve (FR) and Usevy Open Area (OA). Dashed lines represent main roads; dotted lines show the Park boundary. Circular plots around cut and uncut trees and 0.1 ha plots were taken along the two transects (continuous lines) marked A and B. Location of KNP in Tanzania is shown in the insert at the bottom left

are subject to hollowing of live trunks such that not all individuals are suitable for harvest. Trees with hollow or twisted trunks are frequently left standing. As a result, we could find paired standing trees for comparison to logged stumps. At 0.5 km intervals along transects (A and B, Fig. 1) away from the main road we sampled 10 pairs of cut and uncut trees of each species. Paired plots were sampled across distance away from the main road because personal observations and discussion with Park personnel suggested that proximity to the main road is a proxy for logging age and intensity. Stumps closer to the main road were, in general, more degraded.

Paired cut and uncut trees were within close proximity (<50 m) and were of roughly comparable size. For *S. quinqueloba* we made an effort to select uncut trees that were larger than the basal diameter of cut trees, because the loss of bark and rotting of the stems results in cut tree sizes being an under-estimate of true tree size. For *S. quinqueloba*, size differences between cut and uncut stems were significant (cut trees = 73.3 cm d.b.h. vs. uncut trees = 90.1 cm d.b.h., $n = 10$, paired t -test, $t = 4.4$, $P < 0.05$). For

P. angolensis, where the stumps are more robust and less visibly decayed than those of *S. quinqueloba*, we made no effort to select larger trees and there was no difference in tree sizes among matched pairs (cut trees = 44.8 cm d.b.h. vs. uncut trees = 41.2 cm d.b.h., $n = 10$, paired t -test, $t = 0.83$, $P = 0.43$).

Circular plots were established, centred on the bole of the target tree or stump. The plot size for each species was chosen to reflect a radius representative of the average canopy breadth. Our aim was to compare the vegetation structure under the canopy of the uncut tree to the comparable gap left by the harvest of a cut tree. Plot radii were 5 m for *P. angolensis* and 10 m for *S. quinqueloba*. Within plots we measured and identified all trees greater than 2 cm diameter at breast height (d.b.h.); counted all small (<2 cm d.b.h) woody stems into one of three size categories (<50 cm tall, 51–100 cm tall, >100 cm tall); and estimated understory vegetation density using a vegetation density board.

Our vegetation density board consisted of a 7×4 grid of 100 cm^2 squares, alternating black and white for

greater visibility. We used the cardinal directions N-S and E-W to collect two sets of measurements of the degree to which the grid cells were visually obscured across the length of the plot diameter. At each measurement location, we counted the number of obscured grid cells. A grid cell was considered obscured if more than 25% of that cell was covered by vegetation when viewed from the far side of the circular plot. Thus, the viewing distance was 10 m for *P. angolensis* and 20 m for *S. quinqueloba*. For each cardinal direction, obscured cells were counted with the observation board placed at ground level and 1 m above ground, resulting in a total of four observations of 28 grid cells in each plot. Vegetation density was measured as a proportion of the 112 total grid cells obscured.

We analysed vegetation response measures (plot basal area of trees ≥ 2 cm d.b.h., density of small woody vegetation, and vegetation density) using the paired *t*-tests function of JMP 4.0 (SAS Institute). We hypothesized that harvested trees would result in a vegetation response with increased vegetation density. Furthermore, we hypothesized that this effect would increase with the age of the cut, such that the differences between cut and uncut plots would increase towards the main road.

Vegetation plots

To assess trends in vegetation that could be associated with legal boundaries and gradients in human use, we also surveyed a series of vegetation plots along transects

that varied in distance from the main road, which has existed for well over 50 years. We compared these data to similar data from 15 transects inside KNP. We surveyed vegetation in five plots at 1 km intervals along a minor logging track (A) in the FR where *P. angolensis* is being extracted and along a logging track (B) in the OA where active harvest of *S. quinqueloba* has been ongoing for at least 5 years (Fig. 1).

Vegetation plots were 20×50 m. We measured all woody stems ≥ 2 cm d.b.h and identified them to species. For the purpose of this paper we report only stem density values. No plot in either transect contained any individuals of either target species. We hypothesize that logging, being greater near the main road, would result in the lowest stem density near the road. We also compared tree regeneration values from these two sets of five vegetation plots to the same data collected from an additional 15 plots within KNP.

Results

None of our paired comparisons of vegetation response (tree density > 2 cm d.b.h.; seedling/sapling density; or vegetation density) for either *S. quinqueloba* or *P. angolensis* yielded a significant difference between cut and uncut tree locations. With a sample size of 10 for each species, one may question our power to detect differences, but only 55% of the 18 responses tested trended in the predicted direction, and none resulted in a $P < 0.15$ (Table 1). Similarly, we found no differences in trends in vegetation

Table 1 Paired comparisons of understory vegetation growth between plots centred on cut and uncut trees of *Pterocarpus angolensis* and *Sterculia quinqueloba*. For each species, 10 sets of paired plots were compared

Measure	<i>P. angolensis</i>				<i>S. quinqueloba</i>			
	Cut	Uncut	<i>t</i> -test	<i>P</i>	Cut	Uncut	<i>t</i> -test	<i>P</i>
Density of trees								
Small (d.b.h. 2–5 cm)	2.1	1.2	0.87	0.41	6.0	5.2	0.17	0.87
Medium (d.b.h. 5–10 cm)	2.1	2.7	–0.53	0.61	5.2	7.5	–0.95	0.38
Large (d.b.h. 10–20 cm)	1.0	1.6	–1.5	0.17	2.7	4.7	–1.10	0.30
Total basal area	289.7	413.6	–1.5	0.17	764.6	1214.8	1.33	0.21
Understory shrubs								
< 50 cm tall	16.6	15.3	0.22	0.85	15.2	11.7	0.69	0.51
51–100 cm tall	2.2	0.6	1.12	0.29	1.2	1.7	0.89	0.40
> 100 cm tall	2.0	1.6	1.58	0.15	8.8	8.6	–0.13	0.90
All	20.8	16.9	0.46	0.65	25.2	22.0	–0.53	0.61
Understory vegetation	29.6	22.7	0.94	0.37	51.6	54.7	–0.33	0.75

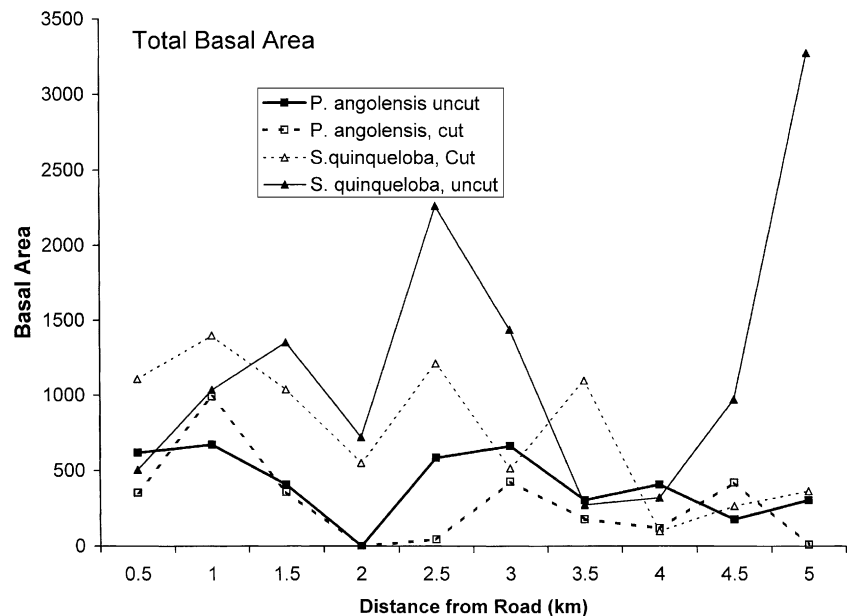


Fig 2 Total basal area sampled within circular plots surrounding cut and uncut stems of mature *Pterocarpus angolensis* and *Sterculia quinqueloba* plotted against distance from main road. Sampled plots ($n = 10$) were collected at 0.5 km intervals along a 5 km transect away from the main road. For both species, total basal area within plots does not vary between paired cut and uncut plots, but increases slightly toward the main road

responses along the gradient in distance from the main road. We present data for total woody basal area within plots (Fig. 2), but the results were consistent for each response variable.

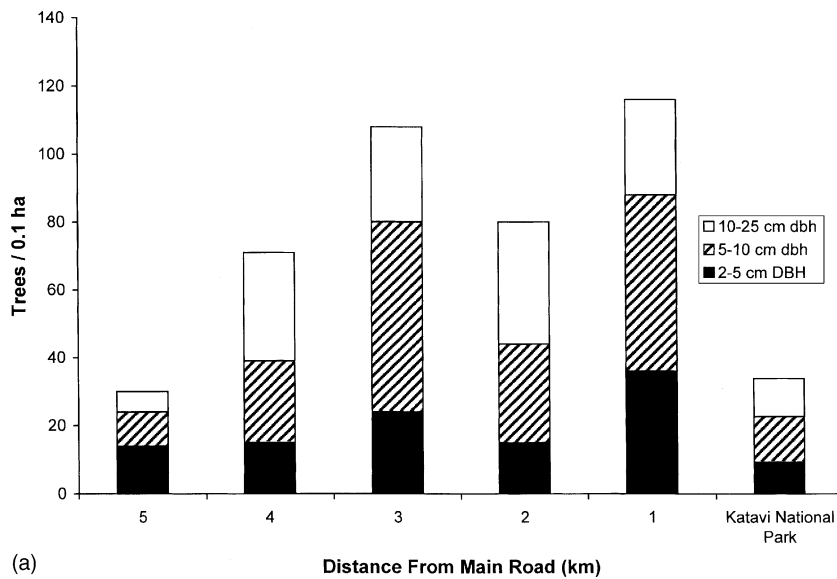
In contrast, vegetation plots demonstrated strong trends in total stem density and basal area of small trees in the transects leading away from the main road (Fig. 3). In both the FR and OA, total stem density increased rapidly towards the main road. Total stem densities at distances furthest from the main road were very similar to plots sampled within KNP. The KNP plots would have had a lower mean stand density were it not for a single outlier plot with a very high stem density. Excluding this outlier, the four plots in the 4 km closest to the road along each transect contained significantly higher stem densities (2–25 cm d.b.h) than the 15 plots sampled within KNP (all eight t -tests, $P < 0.01$). In contrast, the two plots furthest from the road (5 km plots), although having stem densities above KNP plot means, were not significantly different than the KNP plots ($P > 0.2$). Finally, a linear regression of total stem density by distance from the main road for the combined transects was highly significant ($n = 10$, $r_{\text{adj}}^2 = 0.58$, $P = 0.006$).

Discussion

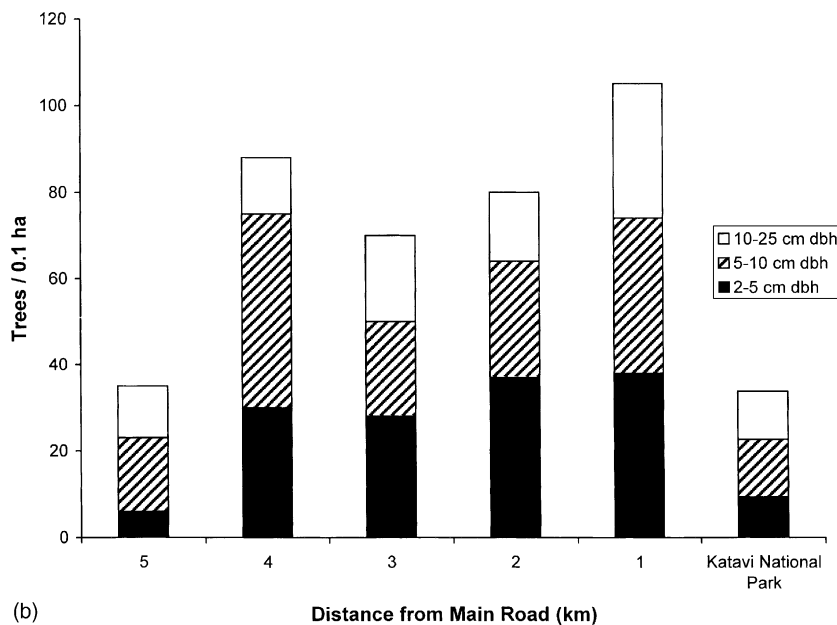
Contrary to our *a priori* expectations, we found no apparent response of understory vegetation to logging for

either of the two selectively harvested species. We saw no recruitment increase of trees under selectively harvested trees. Furthermore, we observed no recruits of either target species in any of our cut or uncut circular canopy plots. In fact, in sampling ten 0.1 ha quadrats we found no small (<15 cm) individuals of either target species. Thus, we found no evidence of compensatory recruitment of trees into the canopy to replace logged trees. The consequence of these observations is that both targeted species are predicted to decline toward economic extinction within harvested stands. Moreover, the lack of compensatory tree recruitment of any species suggests a gradual thinning of stand densities of canopy-sized trees as logging proceeds.

Despite our observed lack of any tree recruitment within canopy gaps, there was a clear trend in the recruitment of young trees in these western Tanzanian miombo woodlands. Stand densities of young trees (<25 cm d.b.h) increased towards the main road irrespective of selective logging. Both selectively harvested species grow at low densities in the sites that we surveyed, and our transects typically included no cut stumps from harvested trees. These results were unexpected because we had predicted a decrease in woody recruitment due to the increasing intensity of human use on the landscape close to roads. For example, fires are routinely set in the OA in order to open up habitat for cattle grazing and in the FR by poachers who are better able to spot and track



(a)



(b)

Fig 3 Small tree density (a) in the Msaginia Forest Reserve and (b) in Usevya Open Area measured as trees/0.1 ha. Densities were sampled in 0.1 ha rectangular plots and categorized into three size classes (2–5 cm, 5–10 cm, 10–25 cm d.b.h). Plots were set up along transects at 1 km intervals away from the main road. The mean density of trees within Katavi National Park, as sampled from 15 × 0.1 ha plots, is shown for comparison for each transect

native herbivores after a fire. For convenience, people walk along the main road with a torch in both areas, even though starting fires is illegal. Because fires are typically set along roads and allowed to run into woodlands, frequency of burning should be greatest near the main road. We expected fire to limit recruitment of woody trees and result in a decreased density of trees near roads.

Nonetheless, such a fire regime could also increase woody tree recruitment. Increased fire frequency may

decrease fire intensity near roads by keeping fuel loads low and low fire intensity could allow stem density to increase near thoroughfares. In addition, fires set inside the park by park rangers tend to be concentrated early in the dry season in the hope of reducing fire intensity (A. Kyambile, pers. comm.) but trees may not be entirely dormant and fire may be more damaging than believed. In contrast, fires outside the park tend to be set later in the dry season (T.M.C. pers. obs.) when trees are dormant.

Studies of woodland response to fire have varied. Where forest and savanna habitats meet, exclusion of fire generally increases tree recruitment (e.g. Swaine, Hawthorne & Orgle, 1992). In contrast, in drier areas fire exclusion tends to reduce tree recruitment rates (e.g. Brookman-Amissah *et al.*, 1980; Chidumayo, 1988a). In our study region, frequent low intensity fire may stimulate the recruitment of trees. Similarly, fires set late in the season may be relatively benign to fire-adapted species of woodland trees and stimulate increased recruitment (e.g. Lock, 1998). We cannot yet adequately predict how fire season, frequency and intensity interact to affect tree recruitment in miombo vegetation.

In addition, native ungulate densities are very low near the road, as a consequence of anthropogenic disturbance. Poachers move out from the main road into the FR and OA and the density of large mammals is very depressed near the road as it travels along the side of the FR and through the OA (T.M.C. pers. obs.). If species such as buffalo (*Synercus caffer* Sparrman), giraffe (*Giraffa camelopardalis* Linnaeus) and zebra (*Equus burchelli* Gray) which are the principal species targeted by poachers (Caro, 1999a) and hartebeest (*Alcelaphus lichtensteinii* Peters), topi (*Damaliscus korrigum* Ogilby) or impala (*Aepyceros melampus* Lichtenstein) were the critical limiting factor for tree recruitment, then we would expect recruitment to be higher near the road than far (5 km) from it. Furthermore, if native ungulates were a driving force in woodland structure we would expect tree recruitment to be low within KNP, where ungulates are highly concentrated (Caro, 1999a,b). Observing increased recruitment near roads and low recruitment within the National Park supports the hypothesis that native ungulates may be critical in determining tree recruitment rates.

Our working hypothesis is that woodland tree recruitment is driven by a complex interaction among native herbivore densities, varying fire regimes and human habitat uses. The result is low tree recruitment rates in protected areas relative to unprotected areas. Independent of selective harvest, tree densities are increasing in areas near human activity, as a result of ungulate avoidance, native grazers and/or fire management. The consequence is that miombo woodlands along highways and near villages show a natural tendency to increase in standing biomass. These increases in standing biomass are most likely offset in many locations by firewood collection and clearing for grazing that might reduce in the eventual recruitment of mature reproductive trees.

We do not yet understand the conservation implications of selective logging for woodland structure, but we imagine that there will be a gradual loss of timber species and an increase in other less preferred tree species in selective logged woodlands. In addition, across a broader scale and contrary to initial predictions, we predict a general increased regeneration of trees species associated with road networks in miombo habitats in south-central Africa. The mechanism driving this increased small tree density in association with roads is not clear, but could be a result of changes in at least two interacting factors: reduced ungulate density near roads reducing the browsing of tree seedlings, and/or increased fire frequency and thus reduced intensity. Predictions regarding the generality of this observation need to be verified by studies across a broader range of sites. The mechanisms driving these patterns also require testing to ascertain potentially important interactions.

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