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Impacts of land use, anthropogenic disturbance, and harvesting on an African medicinal liana

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ABSTRACT

African medicinal plant species are increasingly threatened by overexploitation and habitat loss, but little is known about the conservation status and ecology of many medicinal species. *Mondia whitei* (Apocynaceae, formerly Asclepiadaceae), a medicinal liana found in Sub-Saharan Africa, has been subject to intensive harvesting and habitat loss. We surveyed *M. whitei* in Kakamega Forest, the largest of three remnant Kenyan forests known to contain the species. In 174 100 m² plots, we quantified the status of *M. whitei* and investigated its relationships with land use, disturbance and harvesting. With average adult densities of 101 plants/ha, *M. whitei* is not locally rare in Kakamega. However, the absence of flowers and fruits, together with a spatial disconnect between adults and juveniles, suggests that sexual regeneration is patchy or infrequent. Comparing among habitat types, we found that plants were most abundant in regenerating indigenous forest managed by the Forest Department, which permits some extractive uses. Conversely, plants were largest in indigenous forest managed by the Kenya Wildlife Service, which prohibits extractive uses. Most anthropogenic disturbances were not associated with *M. whitei*, but plant occurrence and density were higher along paths used by livestock than along other types of paths. Larger individuals appeared to be preferentially harvested, but adult plants were more likely to occur in harvested plots than un-harvested plots. This work emphasizes that restrictions on disturbance and extractive use do not automatically promote medicinal plant conservation. Moreover, harvesting may have important genetic and demographic consequences that are overlooked by studies focused on numerical losses.

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1. Introduction

The conservation of African medicinal plant species is critical for local health as well as for international drug development. Africans depend on medicinal plants for as much as 95% of their drug needs (Anyinam, 1995), and as many as 5000 plant species in Africa are used medicinally (Taylor et al., 2001). Although there have been intensive efforts to identify medicinal plants and explore their biochemistry (Fabricant and

Farnsworth, 2001; Taylor et al., 2001; Fennell et al., 2004), very few studies have investigated the ecology or conservation biology of medicinal species (Pandit and Babu, 1998; Shanley and Luz, 2003; Ticktin, 2004). A lack of ecological knowledge can seriously hinder the conservation and sustainable use of medicinal plant species, especially in the face of anthropogenic threats such as overexploitation and land use change.

Overexploitation is a growing problem for many medicinal species in Africa, where human population growth, lack of

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access to western medicine, poverty, and growing markets fuel unsustainable harvesting practices (Cunningham, 1993; Anyinam, 1995; Ticktin, 2004). For example, *Prunus africana*, a globally recognized treatment for prostate cancer, is now critically endangered due to unsustainable harvesting for international sale (Cunningham, 1997; Stewart, 2003). For many species, the ecological impacts of harvesting are unknown, and this lack of knowledge hinders the identification of sustainable harvesting levels or methods (Grace et al., 2002; Ticktin, 2004; Ghimire et al., 2005). Medicinal lianas have received exceptionally little attention, despite their abundance, diversity and pharmacological importance (On et al., 2001; Schnitzer and Bongers, 2002). Ticktin (2004) could find only two vines or lianas for which the impacts of harvesting had been quantitatively assessed. More recently, Ndangalasi et al. (2007) reported that harvesting was associated with denser, smaller stems of a liana collected for basket-weaving in Uganda's Bwindi Impenetrable National Park.

Along with overexploitation, land use change threatens many medicinal plant species in Africa (Anyinam, 1995; Giday et al., 2003; Alves and Rosa, 2007). Research has shown that medicinal plants with ruderal life history characteristics tend to be more tolerant of habitat disturbance and degradation (Giday et al., 2003; Shanley and Luz, 2003; Voeks, 2004). Yet for many species, including most medicinal lianas, basic gaps in ecological knowledge make it hard to predict how populations will respond to land use change. The few studies that have attempted to quantify a relationship between disturbance and medicinal liana abundance (e.g., On et al., 2001; Ndangalasi et al., 2007) have reached contradictory conclusions, paralleling the findings of more general research on disturbance and liana abundance (Putz, 1984; Gerwing, 2004; Mascaro et al., 2004; Rice et al., 2004; Schnitzer et al., 2004). On the other hand, at least two studies agree that lianas tend to be larger in older or less-disturbed forests (Clark, 1996; Ndangalasi et al., 2007).

The paucity of ecological knowledge about medicinal plants is a serious problem for resource managers. The creation of protected areas may facilitate the conservation of medicinal plant species by reducing habitat loss and, via restrictions on access and extractive use, reducing disturbance and overexploitation (On et al., 2001; Ndangalasi et al., 2007). Conversely, restrictions on extractive use can have negative impacts on health and income generation in park-adjacent communities (Botha, 1998; Hamilton et al., 2000; Maikhuri et al., 2001). Today, managers striving to balance conservation and community development goals need ecological information on medicinal species in order to identify sustainable levels of disturbance and harvesting.

In this study, we examined the status of and threats facing *Mondia whitei* (Hook. f.) Skeels (Apocynaceae, formerly Asclepiadaceae) in a Kenyan forest reserve. *M. whitei* is a woody climber that grows on trees and shrubs in tropical moist forests. Mature adults are generally three to six meters in length (Beentje, 1994), but can occasionally exceed ten meters in length (personal observations). Under good conditions, a single mature individual can produce more than fifty large (~8 × 3 cm), fleshy follicles annually; each pod dehisces to release numerous hair-tufted, wind-dispersed seeds (personal observations). *M. whitei*'s strongly aromatic root bark is eaten

by itself or used as a food additive to increase appetite, treat gonorrhoea and schistosomiasis, increase milk production in mothers and cattle, improve sexual performance, boost energy, and treat colds, coughs and stomach disorders (Kokwaro, 1976; Maundu et al., 1999; Venter and Verhoeven, 2001; Mukonyi et al., 2002; Matu and van Staden, 2003). Roots grow laterally just under the ground surface, making them easy to harvest via excavation.

Although ethnological, medicinal and biochemical properties of *M. whitei* have been explored in previous studies (see Maundu et al., 1999; Koorbanally et al., 2000; McGaw et al., 2000; Mukonyi et al., 2002; Matu and van Staden, 2003; Patnam et al., 2005; Pedersen et al., 2006; Watcho et al., 2006; Ndawonde et al., 2007; Watcho et al., 2007; and references therein), very little information is available on the species' physiology, ecology, or conservation. *M. whitei* is reported to be a primary colonizer, found commonly in forest clearings, along forest edges, and in savanna thickets (Hemp, 2001; Mukonyi et al., 2002). Like many lianas, *M. whitei* appears to regenerate well from root cuttings and stems, and seeds have a high germination rate (McCartan and Crouch, 1998; Kavaka Mukonyi, personal communication).

M. whitei occurs in moist to wet forests across Sub-Saharan Africa; it is recorded from Guinea, Nigeria, Cameroon, the Sudan, Uganda, Kenya, Tanzania, Zimbabwe, Malawi, Mozambique, South Africa, Swaziland, and Angola (Hall and Okali, 1979; Cunningham, 1993; Watcho et al., 2001; Mukonyi et al., 2002). Due to its medicinal and monetary value, however, the species has been subject to overexploitation and extirpation in various parts of its range (e.g., South Africa, Cunningham, 1993; McCartan and Crouch, 1998; Koorbanally et al., 2000). In Kenya, *M. whitei* is known from only three remnant forest fragments: Kakamega, Mukua, and Kariti (Beentje, 1994). Most of the roots sold in Kenyan markets are collected in or around these fragments, although a few farmers cultivate *M. whitei* from harvested seeds or seedlings (Mukonyi et al., 2002; personal observations). Harvesting is fueled by impressive demand; in 2000, 1 kg of fresh roots collected at minimal cost could be sold for 500–800 Kenya Shillings (~7–12 US\$) (Mukonyi et al., 2002).

Kakamega Forest is the largest and probably the best protected of the three Kenyan forest fragments where *M. whitei* is known to occur. Mukonyi et al. (2002) estimated that harvesters extract approximately one ton of fresh *M. whitei* roots from Kakamega Forest every month, and suggested that overexploitation and destructive harvesting techniques threaten the survival of *M. whitei* in Kakamega. Despite these assertions, no studies to date have quantitatively evaluated the conservation status or vulnerability of the plant. Lack of information about *M. whitei* has precluded the creation of a unified regulatory strategy for its conservation or sustainable use in Kakamega Forest and surrounding areas.

Non-governmental organizations working in the Kakamega area are also in need of ecological information about *M. whitei*. In 1999, the Kenya Forestry Research Institute (KEFRI) and the International Center of Insect Physiology and Ecology (ICIPE) initiated an on-farm cultivation project focused on *M. whitei*. The project is intended to (1) improve local livelihoods by generating income and improving access to a valuable medicinal plant and (2) promote forest conservation by reducing financial

and health-related dependence on extractive activities. Cultivation efforts will benefit from knowledge about *M. whitei*'s resilience to land use change, harvesting and disturbance. Perhaps more importantly, it will be impossible to evaluate the cultivation project's success, on the conservation front at least, without understanding the current status of wild *M. whitei* in Kakamega Forest.

To improve the state of ecological knowledge about *M. whitei*, and by extension, other medicinal lianas, this study addressed the following questions:

- (1) Is *M. whitei* locally rare or declining in Kakamega Forest?
- (2) What is the distribution and size structure of *M. whitei* in Kakamega Forest?
- (3) How does *M. whitei* fare in different land use types and forest age classes?
- (4) How do habitat disturbance and harvesting affect *M. whitei*?

To answer these questions, we conducted an intensive survey of wild *M. whitei* in Kakamega Forest and outlying forest

fragments. Data from the survey provided a detailed snapshot image of *M. whitei* in Kakamega. We also used informal interviews and participant observation to gain preliminary insights into temporal trends.

2. Methods

2.1. Study site

Kakamega Forest, the largest remnant rainforest in Kenya, is situated northeast of Lake Victoria at approximately 0.27°N and 34.8°E. The forest receives about 2 m of precipitation annually, has high levels of biodiversity and endemism, and serves as an important watershed for Lake Victoria (Lung and Schaab, 2006). Kakamega Forest covers approximately 241 km² (Fig. 1), including about 135 km² of indigenous forest and 57 km² of plantations (Kenya Indigenous Forest Conservation Programme, KIFCON, 1993). Eighty-one percent of Kakamega Forest (196 km²) is managed as a Forest Reserve by Kenya's Forest Department (Kenya Indigenous Forest Conservation Programme, KIFCON, 1993). Forest Department

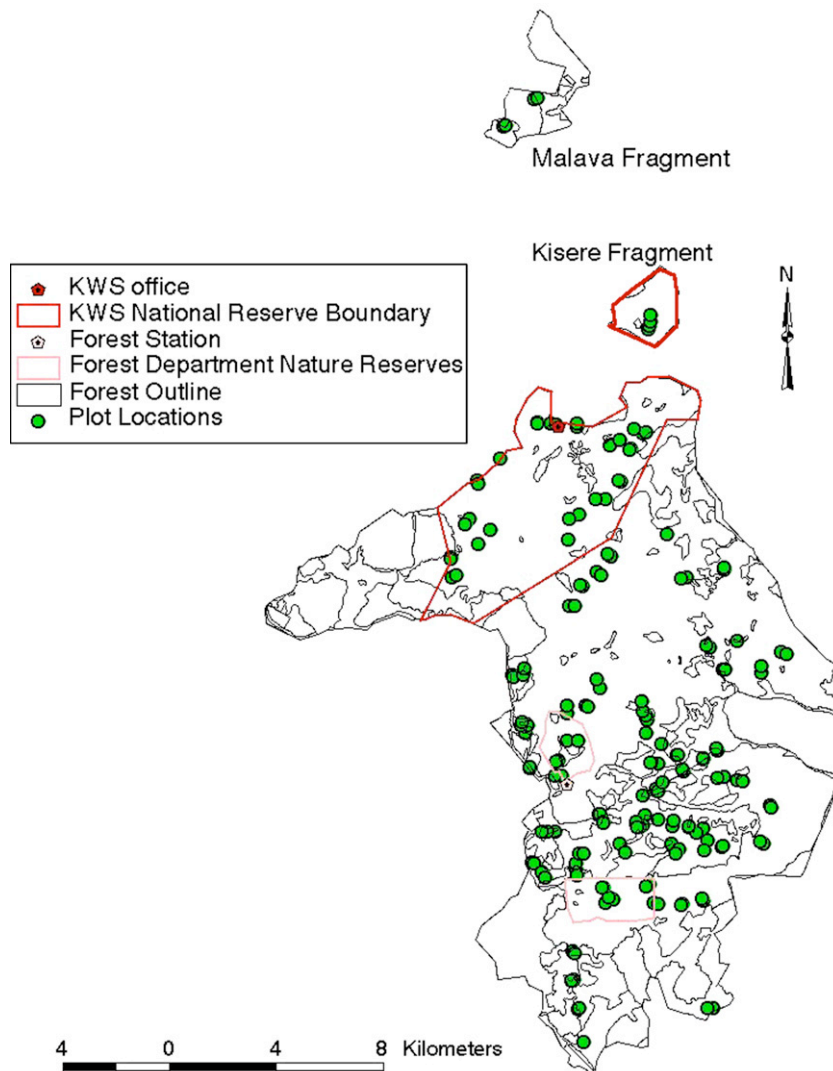


Fig. 1 – Locations of 174 10 × 10 m field plots within Kakamega Forest and outlying fragments. KWS = Kenya Wildlife Service.

regulations allow for limited extractive activities, such as grazing, fuelwood collection, and harvesting of mushrooms, fruits and medicinal plants (Guthiga et al., 2008). Extraction limits are vague and loosely enforced, and levels of illegal human disturbance are high (Bleher et al., 2006). The remaining 19% of the forest (45 km²) is managed as a National Reserve by the Kenya Wildlife Service (Kenya Indigenous Forest Conservation Programme, KIFCON, 1993). Kenya Wildlife Service regulations prohibit extraction or utilization of forest resources; restrictions on access and extraction are strictly enforced, and levels of disturbance are relatively low (Bleher et al., 2006; Guthiga et al., 2008).

Kakamega Forest is located in a low-income area of high population density. Average population density in Kakamega District is 461 people/km² (Guthiga et al., 2008), and there are about 600 people/km² in communities adjacent to the forest (Lung and Schaab, 2006). The majority of Kakamega District's inhabitants are engaged in small-scale agriculture and make less than a dollar/day (Guthiga et al., 2008). Average farm sizes range from 0.5 to 2.0 ha (Tittonell et al., 2007). Because of poverty and land shortages, local communities rely on forest products for food, fuel, building materials, and medicine (Guthiga et al., 2008). In 1993, 99% of forest-adjacent households used forest products to fulfill household needs; for 78% of the same households, forest products provided at least one source of income (Kenya Indigenous Forest Conservation Programme, KIFCON, 1993). Most parts of Kakamega Forest are accessible for extractive use via an extensive network of trails (Bleher et al., 2006).

2.2. Data collection

Between June and August 2003, we collected data in 174 10 × 10 m plots distributed across Kakamega Forest and the Kisere and Malava forest fragments (Fig. 1). Plots were stratified across five land use classes and two forest age classes. The five land use classes included three plantation classes and two indigenous forest classes differentiated by management (Table 1). Using 1986 Landsat imagery, we were able to distinguish between plots that had been subject to the same land use since 1986 ('old' plots) and plots that had been converted to the current land use after 1986 ('young' plots, see also Glenday, 2006). The number of plots placed in each land use/age class was roughly proportional to the amount of for-

est area covered by that land use/age class in 2003 (Table 1). Within each land use and age class, we scattered plots spatially to minimize geographic bias. This effort was limited by the fact that there were few plantations in the northern part of the forest and no Kenya Wildlife Service land in the southern part of the forest. Both age classes were common throughout the forest.

The study area is crisscrossed by numerous roads and footpaths. After deciding how many plots to sample in each land use/age category, we placed plots at random distances along paths in each category. Half of the plots were immediately adjacent to paths (i.e. with one edge of the plot running along the edge of the path). The other half were located at a random distance (between 0 and 500 m) from a path in a direction perpendicular to the path. Plots were not placed at distances greater than 500 m because we almost always encountered another path within 500 m of the original path. For path-adjacent plots, we recorded all observed path uses (in addition to foot travel), including bicycle travel, motor vehicle travel, livestock travel, fuelwood collection, grass collection, and logging. These potential path uses were not mutually exclusive. We also recorded any signs of anthropogenic disturbance in or near each 100 m² plot. Specifically, we noted the presence or absence of evidence of logging, fuelwood gathering, *M. whitei* harvesting, charcoal burning, and hunting (i.e. traps) in the plot, as well as grazing, grass collection, or cropland within approximately 200 m of the plot. *M. whitei* harvesting was identified by the presence of holes or trenches clearly dug for root removal and often accompanied by dead or dying *M. whitei* stems that were severed above the ground surface. We were unable to determine how long these signs of harvesting persist; litter accumulation and decomposition may obliterate evidence of harvesting fairly quickly in the moist forest understory.

Within each 100 m² plot, we counted the number of large (>1 m stem length, hereafter called adult) *M. whitei* individuals and measured the stem length of each individual. For each adult plant, we noted the presence or absence of harvesting (i.e. severed stem and root removal). Within five 1 m² quadrats placed in the four corners and the center of each 100 m² plot, we recorded the number and stem length of all small (<1 m stem length, hereafter called juvenile) *M. whitei* individuals. One meter stem length was chosen as the cutoff between adults and juveniles because on farms around Kakamega Forest, flowers and fruits were observed on indi-

Table 1 – Distribution of plots among land use classes and forest age classes

Land use type	Age ^a	Number of plots
Hardwood Timber Plantation (HP)	Old	10
	Young	9
Indigenous Forest managed by Forest Department (IFFD)	Old	62
	Young	20
Indigenous Forest managed by Kenya Wildlife Service (IFKWS)	Old	26
	Young	11
Mixed Indigenous Plantation (MI)	Old	13
	Young	6
Softwood Timber Plantation (SP)	Old	10
	Young	7

a Age estimated based on 1986 Landsat imagery: old = more than 17 years old in 2003; young = less than 17 years old in 2003.

viduals greater than 1 m tall, but never observed on individuals shorter than 1 m. Although wild *M. whitei* may not begin flowering at such a small size, we chose this conservative size cutoff to ensure that we included all potentially reproductive plants in our exhaustive 'adult' survey of each 100 m² plot. Extensive underground connections made distinguishing between genets and ramets inexact (see Schnitzer et al., 2004). Unless stems were visibly connected or very obviously root-sprouts, we treated them as separate individuals.

We supplemented our quantitative survey by talking with harvesters and vendors of *M. whitei*. In July and August 2003, we conducted 10 semi-structured intercept interviews with vendors selling fresh *M. whitei* roots in Kakamega town. Vendor interviews took approximately 15 min, were always anonymous, and were always conducted with an interpreter. Interview questions focused on the uses of *M. whitei*, reasons why one would choose to sell it, how and where vendors obtain *M. whitei* roots, the monetary value of roots, and changes in scarcity and monetary value over time (see Supplementary materials). In January 2004, two *M. whitei* harvesters allowed us to spend two days talking with them and observing *M. whitei* harvesting. Harvesters shared knowledge about *M. whitei*'s abundance, growth rates, root-sprouting capabilities, and reproductive characteristics. They also provided insight into the details of the harvesting process. We used information from harvesters and vendor interviews to assess local perceptions of scarcity or population decline over time, given the lack of historical data on wild *M. whitei* in Kakamega.

2.3. Data analysis

2.3.1. Distribution and size structure

A likelihood ratio test was used to assess the forest-wide association between adults and juveniles, and a Wilcoxon paired-sample test was used to compare mean juvenile and adult densities across all 174 plots. In plots where *M. whitei* was present, a Wilcoxon paired-sample test was used to compare mean juvenile and adult densities, and a Wilcoxon/Kruskal–Wallis two-sample test was used to evaluate the relationship between adult presence and juvenile density.

2.3.2. Habitat specificity

For each size class, we used multiple logistic regression to examine associations between habitat variables (land use, plot age, and the land use \times plot age interaction term) and plant presence. For plots where a given size class was present, we used ANOVA to examine relationships between habitat variables and the density of that size class. In this and all subsequent analyses, adult and juvenile densities were log-transformed to meet the assumptions of ANOVA. Because plants were not observed in all combinations of land use and plot age, we were forced to exclude the MI land use type from the adult density analysis and the SP land use type from the juvenile density analysis. For adult density, the model's residuals did not formally pass a Shapiro–Wilk test ($P = 0.03$), but the test's correlation value was very high ($W = 0.95$) and non-parametric tests produced qualitatively identical results.

In plots where adults were present, we tested for associations between habitat variables and adult plant size. To obtain

a plot-scale estimate of average adult plant size, we averaged the stem lengths of all adults in each plot. We then used ANOVA with land use type and plot age as predictors (the interaction term was non-significant and was pooled with the error) and average adult plant size as the response variable. In this and all subsequent analyses, average adult stem length values were log-transformed to meet the assumptions of ANOVA.

2.3.3. Anthropogenic disturbance and harvesting

For each size class, we used multiple logistic regression to examine associations between the disturbance variables (e.g., logging, charcoal burning, harvesting, path presence) and plant presence. For plots adjacent to paths, a separate multiple logistic regression model was used to explore relationships between the six path use variables and adult or juvenile plant presence. When predictor variables displayed collinearity, a backwards stepwise procedure were used to eliminate non-significant ($P > 0.10$) predictors from the model.

In plots where a given size class was present, ANOVA models were used to determine whether disturbance variables or path use variables were associated with the density of that size class. If variances were unequal across the different classes of one of the predictor variables, data were inverse variance weighted so that observations from the more variable class received less weight than observations from the less variable class. In the model for path use and adult density, car use and grass use were strongly correlated with bicycle use, and were therefore removed from the model. Similarly, grass use was removed from the model for path use and juvenile density, since it was strongly correlated with other variables.

To explore the relationships between habitat disturbance and adult size, we used ANOVA models with disturbance variables or path use variables as predictors and log-transformed average adult stem length as the response variable. As above, car and grass use were removed from the model for path use. To further examine the relationship between harvesting and adult plant size, we compared the stem lengths of harvested and un-harvested individuals. Using the data from plots where we observed both harvested and un-harvested plants, we conducted a two-way ANOVA with plot (a random effect) and harvesting status as predictors and log-transformed stem length as the response variable.

3. Results

3.1. Distribution and size structure

In total, 178 adult (>1 m stem length) and 70 juvenile (<1 m stem length) individuals were surveyed in our 174 100 m² plots. The average stem length of adult plants was 350 ± 18 cm, and the average stem length of juveniles was 30 ± 2.5 cm. Distributions of adults and juveniles did not overlap strongly. In 70.4% of the plots where we observed adults, no juveniles were present in subplots (Table 2). Although juveniles were more likely to occur in plots where adults were present (likelihood ratio $\chi^2 = 15.2$, $df = 1$, $n = 174$, $P < 0.0001$), adults were absent from 33.3% of the plots where we observed

juveniles (Table 2). When we only considered the adults found in subplots (to control for differences in sampling area), 41.7% of plots with adults lacked juveniles, while 70.8% of plots with juveniles lacked adults (Table 2).

Across all plots, juvenile plants occurred at densities eight times higher than adult plants (8.0 ± 2.8 vs. 1.0 ± 0.17 individuals/100 m²; Wilcoxon paired-sample test $df = 173$, $n = 174$, $P = 0.10$). In plots where *M. whitei* was observed, mean juvenile density was nearly ten times higher than mean adult density (23 ± 7.2 vs. 2.8 ± 0.37 individuals/100 m²; Wilcoxon paired-sample test $df = 61$, $n = 62$, $P = 0.009$). Spatially, juvenile density did not mirror adult density. Excluding plots where no *M. whitei* was observed, juvenile density was significantly higher where adult plants were absent (95.0 ± 45.6 plants/100 m² vs. 11.9 ± 3.4 plants/100 m²; Wilcoxon/Kruskal–Wallis normal approximation $S = 404.5$, $Z = 3.66$, $df = 1$, $n = 62$, $P = 0.0003$). Juvenile abundance did not exhibit any clear spatial patterns across the forest, but adult plants were most abundant in the eastern part of the forest.

We did not observe flowering or fruiting *M. whitei* in any of our plots, despite a previous researcher’s expectation that wild populations should be flowering during the timeframe of our fieldwork (Kavaka Mukonyi, personal communication). Flowering plants were observed on local farms throughout the duration of our fieldwork. The two harvesters we spoke with in January 2004 told us that the majority of *M. whitei* plants in the forest should be fruiting at that time of year. When we asked them to show us fruiting individuals in the forest, however, they were only able to find one fruiting plant in the course of a three-hour search; they pointed out over 30 non-fruiting *M. whitei* adults during the same time period. When asked to speculate about the observed rarity of fruiting plants, harvesters mentioned that charcoal burning can remove the large trees that *M. whitei* requires for support. They also attributed the absence of fruiting plants to harvesting itself. Specifically, they maintained that people generally harvest the plants that are big enough to produce pods, and this harvesting makes it hard to find fruiting plants in a harvested area.

3.2. Local abundance

Previous studies suggested that a tropical liana species can be considered locally rare if its density is consistently less than 1 plant/ha or, based conservatively on tree rarity and the average number of lianas per tree, 6.8 plants/ha (Pitman et al., 1999; Burnham, 2002; Alvira et al., 2004). In our plots, both adult (>1 m stem length) and juvenile (<1 m stem length) *M. whitei* densities exceeded these values. Average adult density was 1.01 plants/100 m², or 101 plants/ha (maximum density,

1500/ha). Average juvenile density was 805 plants/ha (maximum density, 38,000/ha). Although these density values suggested that *M. whitei* was not locally rare, both harvesters and vendors indicated that *M. whitei* is hard to find in the forest. Harvesters told us that it could take a full day to harvest 2–3 kg of wild *M. whitei*, and that many harvesters have to venture deep into the forest to find the plant.

Vendor interview results allowed us to assess the issue of scarcity both directly and via questions about price changes. Of the 10 vendors we interviewed, nine had been selling *M. whitei* for at least 1 year, and seven had been selling for at least 3 years. Seven of the ten vendors began selling *M. whitei* due to its income generating potential. Vendors estimated that they earned between 300 and 1800 Ksh (~US\$4–25) per week by selling *M. whitei*, and half of the vendors said they could make more than 1000 Ksh per week from *M. whitei*. Nine of the ten vendors said that they obtained their *M. whitei* either directly or indirectly from the forest.

When asked if wild *M. whitei* has become harder to find since they started selling, eight out of ten vendors said yes. They attributed the decline of wild populations to high demand and increasing numbers of vendors. None of the vendors said that the amount of *M. whitei* has increased since they started selling. When asked if the price of *M. whitei* has changed since they started selling, six out of ten vendors asserted that it has risen. All six of these attributed the rise in price at least partly to the increased scarcity of *M. whitei* in the forest. Two of them also pointed towards an increase in the number of *M. whitei* vendors, and one identified an increase in demand for the plant. None of the vendors said that the price of *M. whitei* has decreased since they started selling.

3.3. Habitat specificity

Land use, plot age, and the land use × plot age interaction were not significantly associated with the presence of adult or juvenile *M. whitei* ($P > 0.05$ in all cases). In plots where adults were present, however, adult densities differed based on land use type and the interaction between land use and plot age (Table 3; Fig. 2). These results were driven by young IFFD plots, which had adult densities 2- to 5-fold greater than any other combination of land use and plot age (Fig. 2). Juvenile densities were quite variable, but they also exhibited a trend towards larger values in young IFFD plots (Fig. 2; $P > 0.20$ for all effect tests). We did not ask harvesters where *M. whitei* grows best, but when they were asked to show us fruiting *M. whitei* in the forest, they took us to a young IFFD area with relatively high light and few canopy tiers. Adult plant size was similar across plot ages, but varied among land

Table 2 – Frequencies of occurrence for adults in 100 m² plots, juveniles in five 1 m² subplots per 100 m² plot, and adults in five 1 m² subplots per 100 m² plot

		Adults in 10 × 10 m plots		Adults in 5 1 × 1 m subplots	
		Present	Absent	Present	Absent
Juveniles in 5 1 × 1 m subplots	Present	16	8	7	17
	Absent	38	112	5	145

Table 3 – Results of ANOVA analyses for the effects of land use and plot age on adult density and mean adult size

	df	Sum of squares	F	P
Adult density (n = 50)				
Land use	3	12.52	11.02	<0.0001
Plot age	1	0.02	0.05	0.82
Land use × Plot age	3	4.07	3.58	0.022
Adult plant size (n = 54)				
Land use	4	3.31	3.05	0.026
Plot age	1	0.49	1.79	0.19

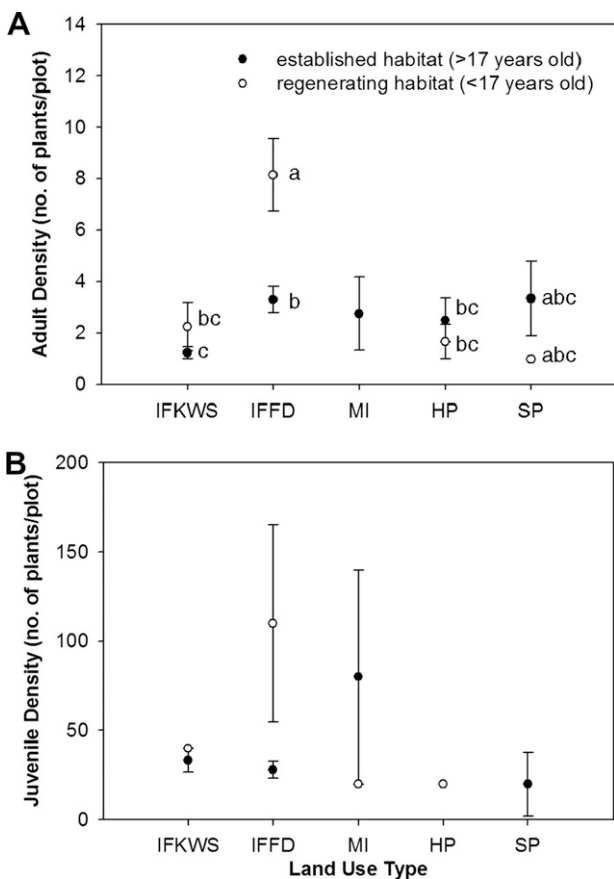


Fig. 2 – Relationship between land use type, plot age, and *M. whitei* density in plots where plants of a given size class were present. Data shown are untransformed, but analyses were performed on log-transformed data. (A) Adult density in different land use/plot age combinations (n = 54). No adult plants were found in young MI habitat. Letters correspond to significant Tukey groupings (MI land use excluded from analysis). (B) Juvenile density in different land use/plot age combinations (n = 24). The point for old HP habitat is hidden. No juvenile plants were found in young SP habitat. IFKWS = Indigenous Forest managed by the Kenya Wildlife Service, IFFD = Indigenous Forest managed by the Forest Department, MI = Mixed Indigenous Plantation, HP = Hardwood Plantation, SP = Softwood Plantation.

use types (Table 3; Fig. 3). Linear contrasts indicated that plants in IFKWS plots were significantly larger than plants in other land use types ($P = 0.003$; Fig. 3).

3.4. Habitat disturbance

The presence, density and size of *M. whitei* were not significantly associated with path presence or evidence of logging, fuelwood gathering, charcoal burning, hunting, grazing, grass collection, or cropland in or near our plots ($P > 0.05$ in all cases).

For path use variables, backwards stepwise logistic regression yielded a model in which bicycle travel (very strongly correlated with car travel and grass collection) and livestock travel were significant predictors of adult presence in plots adjacent to paths ($-\log\text{-likelihood} = 56.6$, $n = 99$, $P = 0.002$). Adult plants were less likely to occur near paths used for biking (effect likelihood ratio test $\chi^2 = 11.2$, $P = 0.0008$, odds ratio = 0.11) and more likely to occur near paths used for livestock (effect likelihood ratio test $\chi^2 = 7.3$, $P = 0.007$, odds ratio = 3.8; Fig. 4a). For juvenile presence, backwards stepwise logistic regression yielded a model in which only livestock use was included as a significant predictor ($-\log\text{-likelihood} = 38.1$, $n = 99$, $P = 0.03$). Juveniles were more likely to occur near paths used for livestock (effect likelihood ratio test $\chi^2 = 4.5$, $P = 0.03$, odds ratio = 3.6; Fig. 4a).

Where adults were present, adult densities were more than three times higher near paths used for livestock (effect test $F = 8.2$, $df = 1$, $n = 33$, $P = 0.008$; Fig. 4b). Similarly, where juveniles were present, juvenile densities were almost twice as high near paths used for livestock (effect test $F = 7.6$, $df = 1$, $n = 14$, $P = 0.025$; Fig. 4b). None of the path use variables were associated with adult plant size ($P > 0.05$ in all cases).

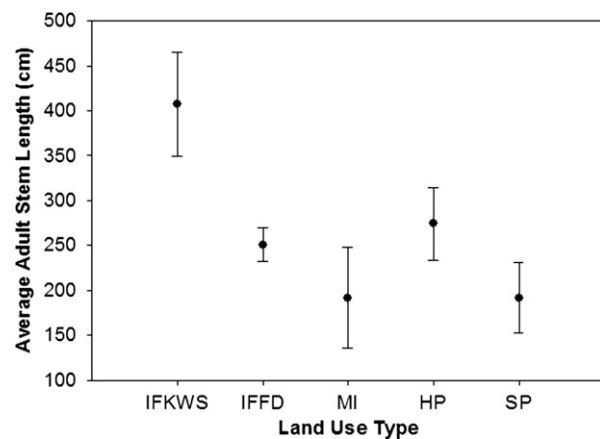


Fig. 3 – Relationship between land use type and adult *M. whitei* size (n = 54). Data shown are untransformed, but analysis was performed on log-transformed data. Plants in IFKWS plots were significantly larger than plants in other land use types (linear contrast $P = 0.0026$). IFKWS = Indigenous Forest managed by the Kenya Wildlife Service, IFFD = Indigenous Forest managed by the Forest Department, MI = Mixed Indigenous Plantation, HP = Hardwood Plantation, SP = Softwood Plantation.

3.5. Harvesting

Harvesting was observed in or near 11 of our 174 plots. The percentage of plots harvested was similar across different land use types and age classes ($7.4 \pm 1.5\%$). Seven harvested plots were adjacent to paths, and only two were more than 50 m from a path. Harvesting was not correlated with any other habitat disturbance variable ($P > 0.05$ in all cases). Adults were present in 64% of harvested plots, but only 29% of un-harvested plots (likelihood ratio $\chi^2 = 5.5$, $df = 1$, $n = 174$, $P = 0.02$, odds ratio = 5.2; Fig. 4c). In plots where adults were present, adult density was not associated with harvesting presence ($P = 0.36$). Juvenile presence and density did not differ significantly between harvested and un-harvested plots ($P > 0.40$ for both tests).

Harvesters and vendors told us that large plants are preferentially harvested, and that as a result, now they “only find the little ones.” They said a plant is considered harvestable if its stem diameter is larger than that of an index finger,

but that it is also worthwhile to dig around smaller plants, because these might be connected to large roots. During the harvesting procedure, the root is generally severed from the stem above the ground surface, leading to the death of above-ground tissue. We were told that some harvesters remove virtually the entire root during harvesting, while others leave part of the root. Harvesters asserted that a plant can regrow to harvestable size about 2–3 months after being harvested, and that a harvested individual can regrow to a height of 6 m in approximately 2 years. There appeared to be no internal coordination among harvesters regarding harvesting territories or communal management of *M. whitei* resources.

By measuring severed stems hanging above obvious harvesting holes, we were able to quantify the stem length of six harvested adults. Where harvested and un-harvested adults co-occurred, harvested individuals tended to be twice as large as un-harvested individuals (625 ± 131.5 cm vs. 329 ± 49.3 cm; harvesting effect test $F = 3.78$, $df = 1$, $n = 14$, $P = 0.08$; Fig. 4d).

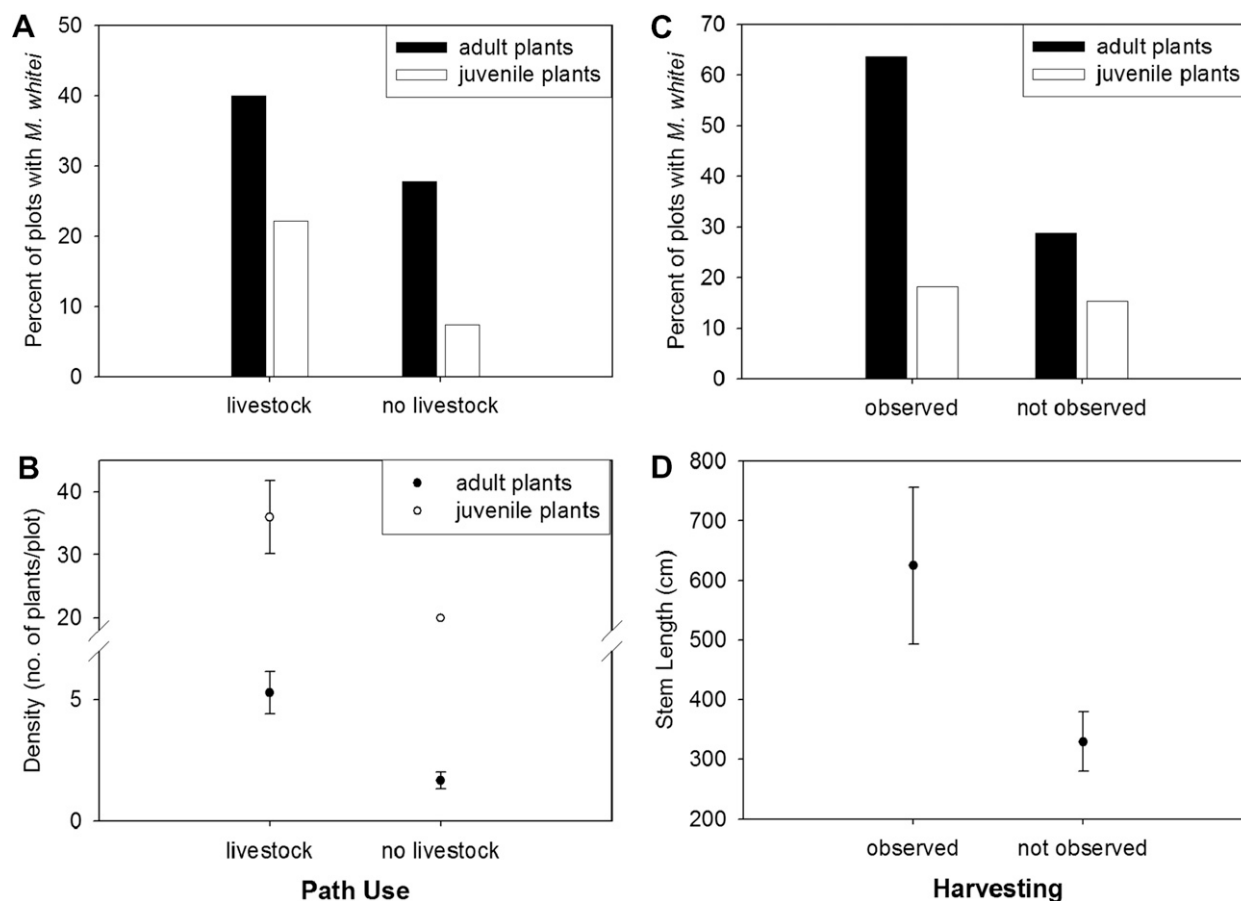


Fig. 4 – Relationships between anthropogenic disturbance and *M. whitei*. (A) Adult and juvenile occurrence near paths used by and not used by livestock ($n = 99$, adult likelihood ratio $\chi^2 = 7.3$, $P = 0.007$; juvenile likelihood ratio $\chi^2 = 4.5$, $P = 0.03$). (B) Adult and juvenile densities near paths used by and not used by livestock (adult effect test $F = 8.2$, $df = 1$, $n = 33$, $P = 0.008$; juvenile effect test $F = 7.6$, $df = 1$, $n = 14$, $P = 0.03$). Data shown are untransformed, but analysis was performed on log-transformed data. Open circle without error bars represents four plots with identical density values. (C) Adult and juvenile occurrence in plots with and without evidence of harvesting (adult likelihood ratio $\chi^2 = 5.5$, $df = 1$, $n = 174$, $P = 0.02$; juvenile likelihood ratio $\chi^2 = 0.18$, $df = 1$, $n = 174$, $P = 0.67$). (D) Mean size of adults in plots where both harvested and un-harvested plants were found. Data shown are untransformed, but analysis was performed on log-transformed data (effect test $F = 3.78$, $df = 1$, $n = 14$, $P = 0.08$).

4. Discussion

In this study, we investigated the status of and threats facing *M. whitei* in Kakamega Forest. To our knowledge, this represents one of the first ecological investigations of a harvested medicinal liana. Our results suggest that the plant is not currently rare in Kakamega, but may be declining. We found that habitat type, path use and harvesting have important, non-intuitive relationships with *M. whitei* in Kakamega.

4.1. Local abundance

Rabinowitz et al. (1986) explained that rarity can be conceptualized in three dimensions: geographic range, habitat specificity, and local population size. Though *M. whitei* clearly has a broad geographic range, spanning much of Sub-Saharan Africa, this study collected the first available data on the species' local abundance and habitat specificity. Our plot data suggested that *M. whitei* was not rare in Kakamega Forest during the time of our data collection, but vendors and harvesters viewed wild *M. whitei* as a scarce and declining resource. The absence of flowering and fruiting individuals during the year of our study, and of juveniles in the majority of plots where adults were present, suggests that seed production and recruitment from seeds may be rare or patchy in either space or time. We have no reason to believe that environmental conditions during the year of our study reduced *M. whitei* flowering or fruiting. Nevertheless, repeated surveys of the *M. whitei* population in Kakamega will be essential to distinguish between interannual variability and long-term trends in abundance, seed production and regeneration.

4.2. Distribution and size structure

On a forest-wide scale, juvenile *M. whitei* plants occurred at higher local densities than adults, suggesting either recent regeneration in the sites where juveniles were observed, or a population dominated by root-sprouts (Hall and Bawa, 1993; Bitariho et al., 2006). We observed an interesting spatial disconnect between adults and juveniles. High densities of adults and juveniles did not co-occur; 70.4% of plots with adults lacked evidence of juvenile regeneration; and although juveniles were more likely to occur in the presence of adults, 33.3% of plots with juveniles lacked adults. Since seedling density tends to be high within ten meters of a parent individual in tropical forests (Burkey, 1994; Hyatt et al., 2003), our results may indicate that most *M. whitei* adults are not regenerating sexually (as suggested by other observations), or that the plant only regenerates well in certain micro-environments (e.g., forest gaps, Mukonyi et al., 2002). Alternatively (or additionally), it suggests that many small root-sprouts are growing in areas where large ramets have been previously harvested.

The harvesters' frequent references to root-sprouting and our own observation of the phenomenon suggest that it may play an important role as an alternative form of *M. whitei* regeneration, especially in areas where harvesting occurs. Root-sprouting is a common reproductive strategy among tropical lianas (Putz, 1984; Gerwing, 2004), and the ability to root-sprout may enable some liana species to better tolerate

disturbance (Gerwing, 2006). It would be useful to quantify the relative importance of root-sprouting for *M. whitei* regeneration, and to assess the consequences (genetic and demographic) of a regeneration strategy that may be increasingly dominated by root-sprouting rather than sexual reproduction (Bond and Midgley, 2001). One important consequence is that root-sprouting, unlike seed dispersal, does not allow for colonization of new sites. If *M. whitei* is a colonizer of gap habitats, regeneration by root-sprouting in existing patches would not allow for escape from declining local habitat quality in sites undergoing succession.

4.3. Habitat specificity

Habitat specificity analyses suggested that although *M. whitei* adults or juveniles occur in all the habitat types we considered, young IFFD (indigenous forest managed by the Forest Department) plots had consistently high density values (Fig. 2). Both young IFFD and young IFKWS (indigenous forest managed by the Kenya Wildlife Service) plots were characterized by indigenous gap species, abundant light above the shrub layer, and fewer than three canopy tiers. Compared to their IFKWS counterparts, however, IFFD plots are more fragmented (Lung and Schaab, 2006) and experience more frequent and established human disturbance (Bleher et al., 2006). Our path use results suggest that more consistent livestock presence in IFFD areas may be a particularly important factor (see below). It is also possible that favorable environmental conditions (e.g., high soil fertility) happen to be located on Forest Department land but are not produced by Forest Department management practices. In particular, adult densities were highest in the eastern part of the forest, which is managed by the Forest Department.

Plants in IFKWS plots were significantly taller than plants in other land use types, regardless of the duration of the current land use practice (Fig. 3). Because IFKWS plots were all located in the northern part of the forest, it is possible that this result reflects differences in soil conditions or climate rather than management. On the other hand, we failed to find any *M. whitei* in the Malava forest fragment, managed by the Forest Department and located north of the Kenya Wildlife Service reserve. Given harvesters' preference for larger plants, we suspect that size differences are related to extractive use restrictions. Harvesters identified Kenya Wildlife Service wardens as one of the three most important obstacles to harvesting. They feared that the wardens would find them harvesting and arrest them. If, as these comments suggest, wardens are successfully deterring harvesters, our results parallel those of Ndangalasi et al. (2007), who found larger individuals of a different liana species in a section of Bwindi Impenetrable National Park, Uganda where harvesting pressure was lower.

Taken together, the results of our habitat specificity analyses may suggest a management tradeoff for *M. whitei*. While *M. whitei* plants were denser in areas managed by the Forest Department, larger individuals were found in areas managed by the Kenya Wildlife Service. In order to conserve the Kakamega *M. whitei* population, decision-makers will likely need to consider the relative importance of density and plant size for population sustainability.

4.4. Habitat disturbance

With the exception of harvesting, none of the anthropogenic disturbances we considered were significantly associated with the density or size of *M. whitei* plants in our study plots. Although several previous studies documented increased liana abundance in recently logged areas (e.g., Alvira et al., 2004; Schnitzer et al., 2004), logging was not significantly related to adult or juvenile density in our study. Moreover, though harvesters asserted that charcoal burning had negative impacts on *M. whitei*, our results showed no associations between charcoal burning and *M. whitei* density or size.

Path presence also failed to predict *M. whitei* density or mean size. This result is surprising, given *M. whitei*'s reputation as an edge colonizer (Hemp, 2001; Mukonyi et al., 2002). We expected paths to act as maintained edge environments whose high light availability would make them attractive to edge species. There are several possible explanations for the lack of association: (1) In contrast to local and published opinion, *M. whitei* may not actually grow better in edge environments. (2) Path-adjacent areas may not adequately epitomize the edge environments preferred by *M. whitei*. (3) Enhanced *M. whitei* growth along paths may be counteracted by increased competition from other plant species or increased harvesting pressure. (4) Different types of paths may have different effects on *M. whitei*. More research will be necessary to disentangle these possibilities, although our path use results provide support for the last two explanations.

Although overall path presence did not affect *M. whitei* densities, the liana occurred less frequently in plots adjacent to paths used for biking, car travel and grass collection. Paths used for these purposes were subject to frequent human use, and were generally wide enough (609 ± 110 cm) to create a large gap in the forest canopy. *M. whitei* on these paths may be subject to increased competition from shrubby edge specialists or increased harvesting pressure.

Livestock paths were associated with high occurrence and density of *M. whitei* (Fig. 4a and b). Livestock paths were moderately wide (450 ± 69 cm) and always covered in cow dung. A previous study in Kakamega Forest found that indigenous forest near livestock paths was characterized by high stem density, low mean tree diameter, and high densities of pioneer tree species (Fashing et al., 2004). Thus, enhanced soil fertility and the early successional state of forests near livestock paths may contribute to the abundance of *M. whitei* at their margins. Moreover, local people suggested that cows are likely to eat *M. whitei*; if livestock consume *M. whitei* pods or seeds, they may act as dispersal vectors for this species. It is also possible that herders are unable to harvest very much *M. whitei* while moving their livestock. Finally, harvesting by herders or consumption by cows may encourage root-sprouting, leading to a higher density of ramets rather than individual plants. It is important to note that although we found more plants along livestock paths, these plants were not significantly larger or smaller than plants along other types of paths.

4.5. Harvesting

Interestingly, we only observed evidence of harvesting in about 6% of our plots. This finding could indicate that har-

vesting is rare or patchy in Kakamega. Alternatively, evidence of harvesting may disintegrate quickly in this moist forest, or harvesters may work to minimize evidence of their activities by filling in holes and removing severed stems.

Like Ndangalasi et al. (2007), who studied an extracted liana in Bwindi Impenetrable National Park, Uganda, we found that adult plants are more common in harvested areas than un-harvested areas (Fig. 4c). We offer two explanations for this finding. First, harvesters are likely to concentrate their efforts in areas with many plants. Second, harvesting may lead to increased root-sprouting. These two alternatives have very different implications for the future of *M. whitei*, and further research will be necessary to distinguish their relative importance.

Our data supported harvesters' assertions that large plants are preferentially harvested (Fig. 4d). Moreover, our discussions with harvesters suggested that the preferential harvesting of large individuals may be limiting sexual reproduction of *M. whitei* in Kakamega. This process may lead to population decline, an increased reliance on root-sprouting, or selection favoring individuals that reproduce at smaller sizes (McGraw, 2001).

The wild population of *M. whitei* in Kakamega currently exists as an open access commons and is therefore vulnerable to continued overexploitation (Hardin, 1968). To combat overexploitation in Kakamega, education and community-based regulation may be more politically and socially sound solutions than a total ban on harvesting. Our observation of harvesting within areas managed by Kenya Wildlife Service, where the practice is technically illegal, makes it clear that outright bans do not completely prevent harvesting activities (see also Ndangalasi et al., 2007). Even if it were effectively enforced, a total ban would negatively impact local communities by reducing access to an important source of medicine and income. Given *M. whitei*'s apparent ability to root-sprout, the species would likely benefit from an educational campaign focused on sustainable harvesting techniques (e.g., leaving part of the root behind for regeneration, or leaving large plants alive for sexual reproduction). The creation of locally enforced management strategies, such as fallow periods or harvesting territories, could also benefit *M. whitei* while avoiding many negative consequences of a total ban (Hamilton et al., 2000).

5. Conclusions

This study represents the first comprehensive ecological assessment of *M. whitei* in Kakamega Forest. Using these data as a baseline, future studies will be able to determine whether or not *M. whitei* is declining in Kakamega and to evaluate the potential impacts of an on-farm cultivation project. More generally, our results emphasize that medicinal plant conservation cannot be accomplished using general assumptions. For example, restrictions on disturbance and extractive use may not automatically promote medicinal liana conservation. Some disturbances may be associated with increased plant abundance, and the net impact of extractive use restrictions may depend on tradeoffs between a population composed of many small plants and one composed of a few large plants.

Our results also suggest that some important consequences of harvesting may be overlooked by studies focused on numerical losses. A better understanding of genetic and morphological impacts, such as increased reliance on asexual reproduction or shifts towards smaller adult size, could help conservationists prevent, rather than react to, numerical declines. These insights illustrate the value of expanding ecological knowledge about important, but poorly understood medicinal plant species.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at [doi:10.1016/j.biocon.2008.06.021](https://doi.org/10.1016/j.biocon.2008.06.021).

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