Regeneration of the East African timber tree *Ocotea usambarensis* in relation to historical logging

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ABSTRACT

East African montane forests have been subjected to heavy logging, particularly of *Ocotea usambarensis* Engl., formerly one of the dominant tree species of moist mid-altitude forests. At Mt. Kenya, logging was suspended in 2000 after a conspicuous decline in population size, but the success of this conservation measure has not yet been evaluated. Given that a management scheme of *O. usambarensis* forests based on vegetative regeneration has been suggested, we hypothesized that natural regeneration mainly by root suckers would be sufficient for a recovery of this species. Demography and regeneration (both sexual and vegetative) of *O. usambarensis* were studied in 45 study plots between 1700 and 2500 m asl along a gradient of historical logging intensity, while taking altitude and light incidence into account as predictor variables. The diameter distribution showed a high percentage of old individuals and rather low recruitment in *O. usambarensis*. In heavily logged areas (removed basal area >25 m²ha⁻¹), smaller trees (<50 cm DBH, >130 cm high) were completely absent. The number of seedlings was low and independent of logging intensity. It increased with higher light incidence. The number of root suckers was 5.6-fold the number of seedlings, underscoring the importance of vegetative reproduction. However, number of root suckers and logging intensity were negatively correlated. We conclude that regeneration of *O. usambarensis* at Mt. Kenya is generally low and negatively influenced by historical logging. Therefore, natural regeneration is inadequate for the recovery of this valuable timber species, and additional conservation measures such as enrichment planting should be considered.

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1. Introduction

Despite improved conservation strategies for many tropical forest ecosystems in the last decades, post-logging recovery of tropical timber tree species remains a matter of concern. Regeneration failure of commercial timber species has been reported from several tropical forests (e.g. Hall et al., 2003; Makana and Thomas, 2005). Many tree species require high light incidence at the seedling stage for survival and growth. However, canopy opening through logging does not necessarily lead to increased recruitment as local environmental conditions and population structure influence the number of safe sites for germination and establishment (Bazzaz, 1991; Makana and Thomas, 2005; Ouedraogo et al., 2011). Most tropical tree species are generally seed-limited as seeds only rarely reach potential regeneration sites (Muller-Landau, 2002). Vegetative regeneration by root suckers is an additional strategy in some species (Putz and Brokaw, 1989). Particularly in montane forests, logging may cause severe changes

in the natural regeneration cycle (e.g. Bussmann, 2001). However, the assessment of tree recruitment after logging in the tropics mainly relies on short-term observation, and little is known about long-term effects of logging (Struhsaker, 1997) and the requirements of individual species in all stages of their development (Gómez-Pompa and Burley, 1991).

In Kenya, approximately 8% of the indigenous closed canopy forest area was deforested between 1990 and 2010 (FAO, 2010). According to Akotsi and Gachanja (2006), only 1.7% of the total country area is covered by closed canopy forest. For many years, exploitation of the forests and their wildlife was not regulated, which led to degradation and destruction on a large scale (Beentje, 1990). The forests of Mt. Kenya still cover an estimated area of 180,000 ha and thus represent the largest continuous forest block in Kenya (Beentje, 1991). The water catchment of Mt. Kenya forests is estimated to provide more than 40% of the country's water needs (Akotsi and Gachanja, 2006).

The East African camphorwood (*Ocotea usambarensis*) used to be the dominant canopy tree species of moist East African montane forests below 2500 m (Lind and Morrison, 1974). Since the 1940s, it has been extensively exploited for its valuable timber (Nsolomo and Venn, 2000). Selective logging has impacted and destroyed an

alarmingly high proportion of the *O. usambarensis* forest (Bussmann, 1996; Gathaara, 1999). This occurred not only at Mt. Kenya, but also at many other sites in Kenya (e.g. Aberdare Mountains, Taita Hills; Lambrechts et al., 2003; Aerts et al., 2011) and in Tanzania (e.g. Eastern Arc Mountains, Mt. Kilimanjaro; Hemp, 2006; Persha and Blomley, 2009). Similar to other tropical timber species, *O. usambarensis* is known to experience low sexual regeneration through seedlings (Newmark, 2002). However, vegetative regeneration through root suckers is common (Dale and Greenway, 1961), and in Tanzania this characteristic of *O. usambarensis* has been used to devise a management scheme for timber production (Willan, 1965). Bussmann (1994, 2001) and Bussmann and Beck (1995) developed a scenario of the *O. usambarensis* regeneration cycle suggesting that, after heavy logging, the cycle is interrupted by the establishment of a secondary forest community dominated by the pioneer tree *Macaranga kilimandscharica*. However, this scenario has not yet been confirmed as the safe sites for germination and establishment of *O. usambarensis* have not been identified and the relative importance of vegetative regeneration through root suckers has not been assessed.

In 2000, the Kenya Wildlife Service took over the management responsibility of Mt. Kenya forest area as a whole. Since then, it has been managed as a National Reserve, which resulted in a 96% reduction in logging of *O. usambarensis* on Mt. Kenya due to improved patrolling and control (Vanleeuwe et al., 2003). In this study we examined the demography and current regeneration patterns of the *O. usambarensis* population at Mt. Kenya 10 years after logging suspension. In particular, we addressed the following questions: (1) Is the diameter distribution of *O. usambarensis* population linked to past logging intensity? (2) Are there significant differences between the extent of sexual and vegetative recruitment? (3) Are these regeneration strategies related to past logging rate and to other environmental conditions such as light incidence or altitude? This analysis allowed us to assess if the improved protection since 2000 alone has been sufficient for the recovery of *O. usambarensis* in areas affected by logging or if further silvicultural measures are necessary.

2. Materials and methods

2.1. Study area

The study was carried out at the eastern slope of Mt. Kenya (Fig. 1) in the first quarter of 2010. Due to their biodiversity and important status as a source for water, timber, firewood, and wildlife, the Mt. Kenya forests have been declared a UNESCO World Heritage site (UNEP, 2005). They belong to the catchment areas of the two major rivers in the country, the Tana and the Ewaso Ngiro, which are crucial for many livelihoods (Gichuki, 1999) and for the production of hydroelectric power (Akotsi and Gachanja, 2004). The fertile and humid uplands on the southeastern slopes of Mt. Kenya have a high population density of up to 800 people per km² (Imbernon, 1999). The study area comprised 6772 ha on an altitudinal gradient between 1778 and 2416 m asl and covered approximately 25% of the remaining *O. usambarensis* forest at Mt. Kenya (Beentje, 1991). The area belongs to the Meru South district in the Eastern Province of Kenya and is administered by Chogoria Forest Station (0°14′23″ S, 37°35′32″ E). The climate of the area is characterized by a mean annual precipitation of about 2200 mm, with two rainy seasons, from March to June and from October to November (Chogoria Forest Station, 1615 m asl, February 1973–March 2010; Schöning et al., 2010). Temperatures are relatively stable year round (mean annual temperature 18 °C at Meru Meteorological Station, about 33 km north of Chogoria Forest Station; Bussmann, 1994) and decrease by about 0.56 °C per 100 m altitude with frost being expected above 2500 m (Bussmann, 1994).

2.2. Study species

Ocotea usambarensis Engl. (Lauraceae) is an evergreen emergent tree species in East African moist montane forests at altitudes between 1220 and 2440 m (Dale and Greenway, 1961) and precipitation levels between 1600 and 2450 mm year⁻¹ (Beentje, 1991). It grows on deep, fertile soils with good drainage. *Ocotea usambarensis* is considered shade-tolerant and late-successional (Bussmann, 2001; Babaasa et al., 2004). With heights up to 45 m and trunk diameters up to 3 m, *O. usambarensis* is one of the largest trees in Kenya. It is a highly valued hardwood for furniture and joinery (Lind and Morrison, 1974). The tree is gynodioecious as both hermaphrodite and female flowers occur (Verdcourt, 1996). Seeds show a short-term viability that precludes the formation of a seed bank (Bussmann, 1994; Baskin and Baskin, 2005). Sexual regeneration is considered to occur mainly in mast years, approximately every 10 (Kigomo, 1987) or every 5–7 (Bussmann, 2001) years. However, the categorization of *O. usambarensis* as a mast-fruiting species needs to be confirmed by long-term data (see Kelly, 1994). Asexual regeneration through root suckers has been reported both without disturbance (Dale and Greenway, 1961) and as a response to felling (Willan, 1965; Lind and Morrison, 1974).

The area of Mt. Kenya that can support moist montane *O. usambarensis* forest is 51,000 ha (Kindt et al., 2005). However, the species has been subjected to heavy logging in the past, which led to a decrease to 27,500 ha of *O. usambarensis* forest (Beentje, 1991). In an aerial survey, Gathaara (1999) counted about 6700 individuals of selectively logged trees with the heaviest impact being visible over an area of 5600 ha. This overexploitation led to a change in the operating conservation authorities (Vanleeuwe et al., 2003).

Macaranga kilimandscharica Pax (Euphorbiaceae) is a short-lived pioneer tree species that colonizes and often dominates logging gaps in *O. usambarensis* forests (Bussmann, 1994, 2001; Bitariho and McNeilage, 2008).

2.3. Sampling design

Sampling was carried out in a stratified, randomized design following Kindt and Coe (2005). We divided the altitudinal belt where *O. usambarensis* occurs into three strata: (1) 1775–2000 m, (2) 2000–2200 m, and (3) 2200–2425 m. To include a wide range of past logging intensities, we differentiated three classes of logging intensity within each altitudinal stratum for a first orientation using data from the aerial logging survey of Gathaara (1999) in a geographical information system (ESRI ArcGIS, Redlands, CA) and uploaded the results on a GPS (Garmin Etrex Legend Cx, Kansas City, KS). This approach enabled us to find the scarce forest remnants that had not experienced any logging impact. For final data analysis, logging intensity was again measured on the plot level.

In each of the nine combinations of altitude and logging intensity categories, we randomly selected five replicate study plots. Each plot comprised a circular subplot of 20 m in diameter (314 m²) with one living adult *O. usambarensis* tree (diameter at breast height >25 cm) in the center and two transects (each 40 m long and 10 m wide) extending along the contour line in both directions from the circular subplot (Fig. 2). The prerequisite of at least one living *O. usambarensis* tree per plot was necessary as we aimed at analyzing the impact of stand disturbance through logging on the regeneration pattern of the remaining trees. We assessed the population structure and logging intensity considering both living trees and remaining stumps of logged *O. usambarensis* trees in the entire plot including the circular subplot and transects. The data on regeneration parameters (seedling and root sucker numbers) were collected exclusively in the central subplot adjacent to the living tree. Overall, this approach led to 45 study plots with an area of 314 m² + 2 × 400 m² = 1114 m² each (Fig. 1). To
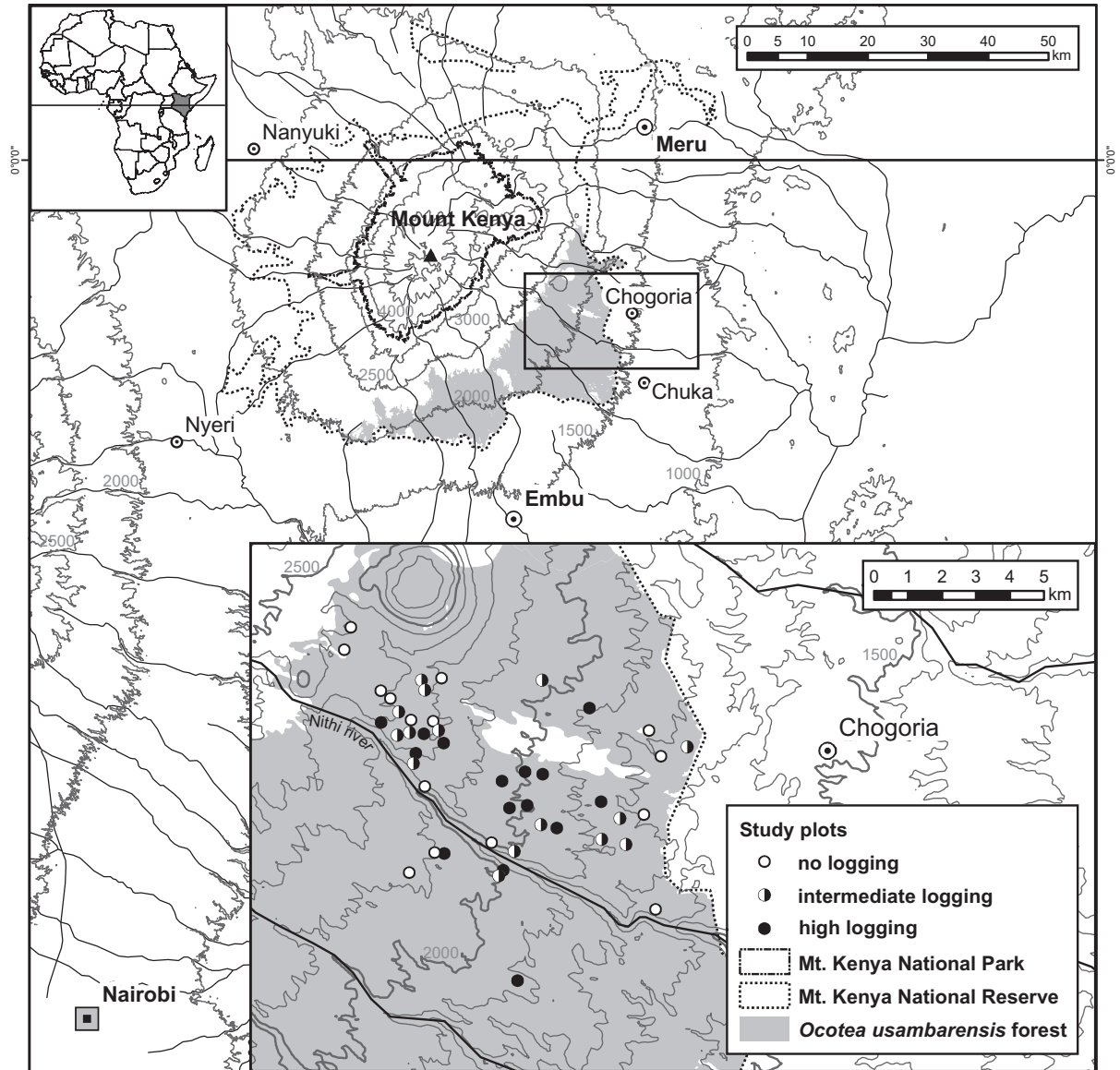
Fig. 1. Study area in Eastern Province, Kenya. Contour lines in 500 m increments in the overview map and 100 m increments in the detail map. Extent of the *O. usambarensis* forest, the boundaries of protected areas, and the location of study plots. Visualization of logging intensities as indicated by basal area removed: no logging: 0 m^2 ha^{-1}; intermediate logging: <25 m^2 ha^{-1}; heavy logging: >25 m^2 ha^{-1}.
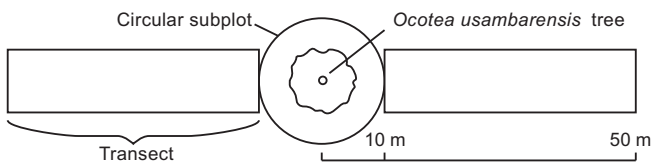

Fig. 2. Plot design: circular subplot of 20 m in diameter (314 m^2) around one living adult *O. usambarensis* tree (diameter at breast height >25 cm) in the center and two transects (each 40 m long and 10 m wide), extending along the contour line in both directions from the circular subplot.

ensure independence, we kept a minimum distance of 300 m between the study plots.

2.4. Population structure and logging intensity

Within each plot (circular subplots and two transects), we measured diameter at breast height (DBH) of all *O. usambarensis* trees >1.3 m in height. Furthermore, we counted all stems of *M. kili-*

mandscharica with DBH >15 cm. To exclude buttress roots, we measured their average depth and subtracted it from the overall diameter. We determined the diameter of stumps of logged *O. usambarensis* trees as the average of two perpendicular measurements on the saw plane, again excluding buttress roots (Rijks et al., 1998). As the decay of stumps is slow in the area, we estimate that their number indicates historical logging during approximately the last 25–35 years, which coincides with the time of heavy logging intensity in the area.

Past logging intensity was measured as the basal area of logged *O. usambarensis* trees per hectare and cross-checked with the number of logged stems of *O. usambarensis* per hectare. Based on the information that large-scale logging was effectively prevented from the year 2000 onwards, we assumed that all logging took place prior to this date. Wood decay of stumps and moss cover confirmed this assumption. In the whole area, we found only two trees that had obviously been logged more recently, both of which were not included in the data.

To visualize the impact of logging on the DBH distribution, we calculated the overall stem density and the density per size class (using 25-cm-wide size classes) and compared three different logging classes (no logging, <25 m² ha⁻¹ and >25 m² ha⁻¹ of basal area removed).

2.5. Sexual and vegetative regeneration

Within each circular subplot, we counted all seedlings (height <1.3 m) and root suckers (height <1.3 m) of *O. usambarensis*. A pilot study had revealed that it was not necessary to completely uproot putative seedlings to distinguish them from root suckers. A slight pull upward clearly showed whether the plant was connected to an adult tree's root system. Thus, destruction of the rare seedlings could be avoided. In all doubtful cases during subsequent data collection, we nevertheless excavated the roots completely to guarantee unambiguous categorization. To exclude any influence of the number of adult trees on *O. usambarensis* seedlings and root suckers, we used the numbers per adult *O. usambarensis* stem (DBH >25 cm, which is the minimum size where we found fruiting) in the central subplot as response variable for further statistical modeling.

2.6. Environmental variables

As environmental variables, we included altitude asl and light incidence. Altitude was measured in the central part of each circular subplot using a Garmin GPS Etrex Legend Cx. In addition, we estimated total light incidence by subtracting the estimated leaf cover from a 180° half globe around the center of the subplot (modified from Renison et al., 2005) as not only vertical direct light but also lateral direct and indirect light may be relevant for plant growth in the steep and furrowed terrain of montane areas (Clark, 1990).

2.7. Statistical analyses

We used multiple regression in generalized linear models (GLMs) with logging intensity (expressed as both basal area and stem number removed per hectare) as explanatory variable. For sexual and vegetative regeneration counts, we additionally included altitude asl and light incidence as covariables to obtain additional insights into the regeneration ecology of *O. usambarensis*. For measured data we assumed a Gaussian distribution, for count data a Poisson distribution. In cases where we detected overdispersion (indicated by the ratio of the residual deviance and the degrees of freedom being larger than 1) due to excess zero-counts, we used hurdle regression models (Zeileis et al., 2008; Zuur et al., 2009). The following response variables were tested in separate models: stem number of adult *O. usambarensis* with DBH >25 cm, stem number of young *O. usambarensis* with DBH <25 cm, stem number of *M. kilimandscharica* per hectare based on the data from the entire study plots; numbers of seedlings and root suckers per adult tree based on the data of the central subplots.

To compare abundances of seedlings and root suckers with each other, we used the Wilcoxon test. All statistical analysis was done using R version 2.12 applying the packages "vegan", "stat," and "pscl" (R Development Core Team, 2010).

3. Results

3.1. Population structure

Overall, the *O. usambarensis* population was characterized by a high proportion of large trees with DBH >100 cm and few smaller

trees (>1.3 m in height and DBH <75 cm) irrespective of logging (Fig. 3a–c). Trees of the diameter class <25 cm were found in only 4% of the plots.

Logging led to a mean removal of 16.76 m² basal area and 12.37 stems ha⁻¹ in all study plots. The mean diameter of logged tree stumps was 127 cm. Consequently, the number of adult *O. usambarensis* stems (DBH >25 cm) decreased marginally significantly with basal area removed and significantly with stems removed (Table 1). In contrast, we found no logged trees in diameter classes below 75 cm (Fig. 3a–c).

In heavily logged areas (removed basal area of *O. usambarensis* >25 m² ha⁻¹), diameter classes <50 cm were completely absent, as were their stumps (Fig. 3c). In accordance, the hurdle regression model revealed that the number of small trees (DBH <25 cm) decreased marginally significantly with the removed basal area and significantly with the number of logged stems (Table 1 and Fig. 4a). On the other hand, the stem number of *M. kilimandscharica* was positively correlated with both measurements of logging (Table 1 and Fig. 4b).

3.2. Sexual regeneration

Seedlings were present in approximately half of the study plots; their mean density in all plots was 124 ha⁻¹, and 253 ha⁻¹ in plots
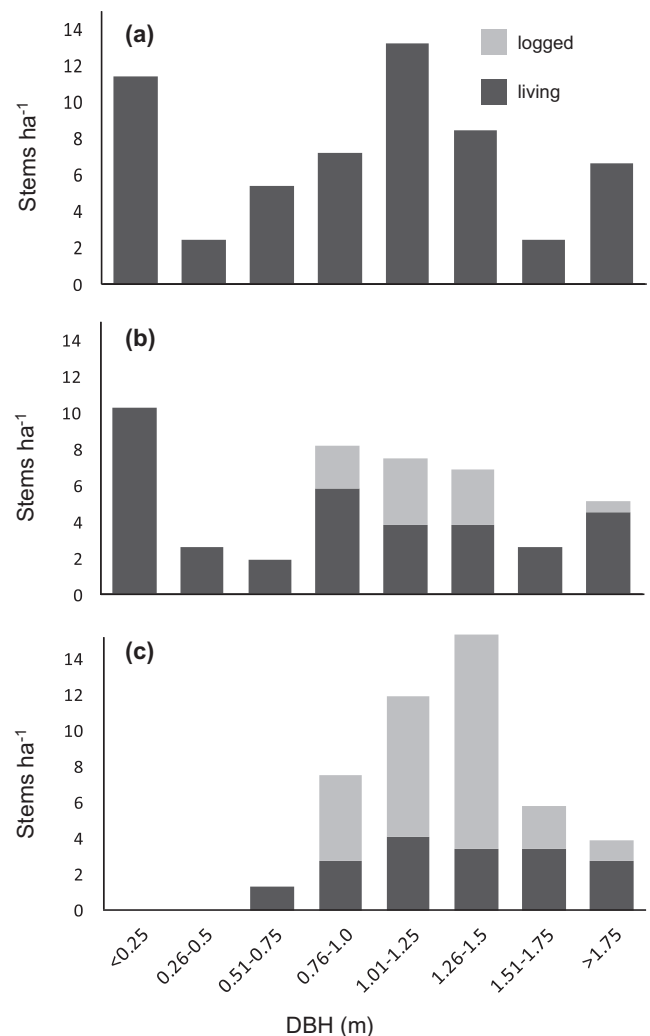

Fig. 3. Diameter distribution of logged and living *O. usambarensis* stems comparing different historical logging intensities as indicated by basal area of *O. usambarensis* removed: (a) no logging: 0 m² ha⁻¹, (b) intermediate logging: <25 m² ha⁻¹, (c) heavy logging: >25 m² ha⁻¹.

Table 1
Population structure and regeneration (number of seedlings and root suckers) of O. usambarensis in relation to past logging. Means with standard errors in parentheses, n = 45; P-values <0.05 indicate significant correlations in regressions using GLM (assumed distributions for measured data: Gaussian, count data: Poisson, *overdispersed count data: hurdle zero-inflated models).

|	Mean (SE)	Historical logging intensity			
		Basal area removed m^2 ha^{-1}		Stems removed N ha^{-1}	
		Coefficient	P-value	Coefficient	P-value
Population parameters (measured in entire 1114 m^2 plots)					
Stem number of adult O. usambarensis (DBH >0.25 m) ha^{-1}	28.72 (4.04)	−2.001	0.052	−2.129	0.039
Stem number of small O. usambarensis (height >130 cm, DBH <0.25 m) ha^{-1}*	6.98 (3.21)	−1.705	0.088	−1.980	0.048
Stem number of Macaranga kilimandscharica (DBH >15 cm) ha^{-1}	138.24 (25.6)	6.114	<0.001	7.575	<0.001
Regeneration (measured in circular 314 m^2 subplot)					
Seedlings (height <130 cm) per adult tree*	2.72 (0.97)	0.489	0.625	−0.063	0.950
Root suckers (height <130 cm) per adult tree*	12.99 (3.20)	−2.168	0.03	−2.521	0.012
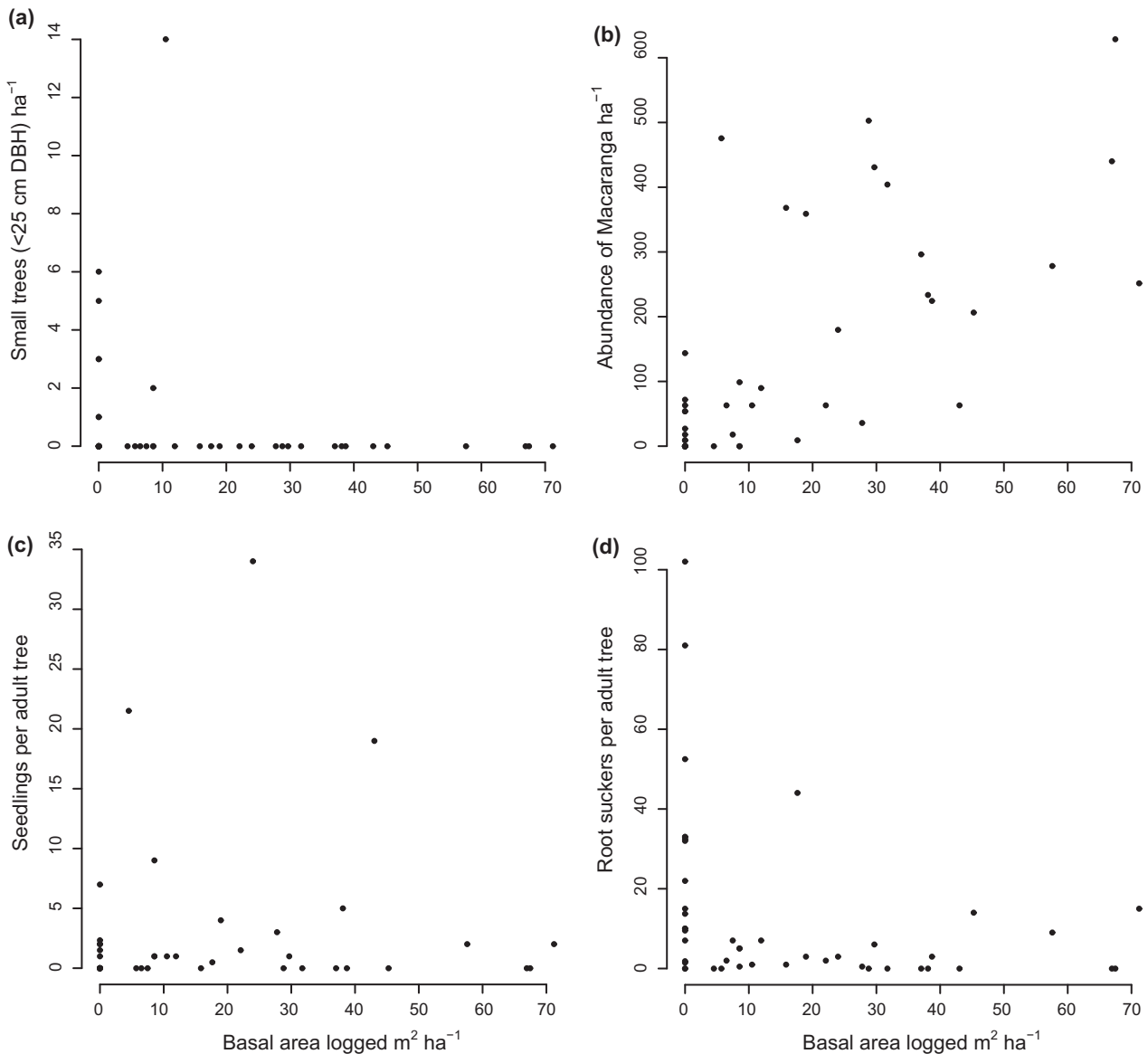
Fig. 4. Numbers of (a) small trees (<25 cm DBH) of O. usambarensis ha^{-1}, (b) M. kilimandscharica stems ha^{-1}, (c) seedlings of O. usambarensis per adult tree, and (d) root suckers of O. usambarensis per adult tree in relation to the basal area of logged O. usambarensis ha^{-1}.

with seedlings. In the hurdle regression model, neither logging (Table 1 and Fig 4c) nor altitude was significant as a predictor for seedling number, but total light incidence proved to correlate positively (P <0.001). Light intensity correlated neither with basal area removed (P = 0.83) nor with stem number removed (P = 0.449).

3.3. Asexual regeneration

Root suckers occurred in 78% of the study plots (mean of 691 ha^{-1}), and total numbers were significantly (5.6 times) higher than those of seedlings ($W = 1447$, $P < 0.001$); density and occurrence of the two reproduction forms were not correlated with each other. The hurdle regression model showed that root suckers significantly decreased with both measurements of logging intensity (Table 1 and Fig. 4d).

4. Discussion

4.1. Population structure

The overall diameter class distribution with a high percentage of larger trees along with few trees in lower diameter classes indicates generally low recruitment despite the fact that the species is considered shade-tolerant and may regenerate below canopy (Bussmann, 2001; Babaasa et al., 2004). Similar findings on the size distributions in populations of tropical tree species have been attributed to the irregular distribution of suitable regeneration sites in space and time (Poorter et al., 1996).

Ten years after the suspension of logging activities, the population structure of O. usambarensis at Mt. Kenya still clearly shows past disturbance. The overall decrease in the number of adult trees with logging is obviously a direct consequence of stem removal by selective logging. This is supported by the fact that the sum of living and logged stems >75 cm in DBH is similar across all logging classes. In contrast, we found no logged trees in diameter classes <75 cm. However, in areas with logging rates of >25 m^2 of logged basal area, we observed the complete absence of trees that were taller than 1.3 m and had diameters <50 cm. Accordingly, the number of small trees (<25 cm in DBH) was negatively correlated with logging intensity. A high logging intensity (>25 m^2 ha^{-1}) seems to have prevented seedlings and root suckers from establishing and growing beyond 1.3 m in height.

These data provide evidence that heavy logging in the past has led to much slower population recovery of O. usambarensis compared to intermediate logging. Since most of the study area was logged heavily (Gathaara, 1999; Vanleeuwe et al., 2003), our study species seems to require additional conservation measures. Hall et al. (2003) similarly found a general lack of seedling establishment for commercially logged timber species, when no additional silvicultural treatment was applied. Lind and Morrison (1974) hypothesized that present day conditions are no longer favorable for O. usambarensis to maintain itself by regeneration. However, at Bwindi (Uganda), where logging of O. usambarensis had ended 14 years before data collection, the diameter class distribution showed juvenile trees as most frequent, while seedlings and adults were rare (Bitariho et al., 2006). Here, the fast population recovery may be related to less severe logging and a higher light availability due to the absence of the shade of old growth trees (see Section 4.2).

The abundance of M. kilimandscharica trees showed a positive correlation with logging intensity. This finding is generally in line with the "arrested succession" hypothesis (Gómez-Pompa and Burley, 1991; Paul et al., 2004), proposed for O. usambarensis by Bussmann (2001). He suggested that this pioneer species directly prevents regeneration of O. usambarensis after heavy logging. Here, we only show that regeneration of the two species responds in opposite ways (but possibly independently) to logging intensity.

4.2. Sexual and vegetative regeneration

The number of seedlings was independent of logging intensity. However, we found a strong positive correlation between total light incidence and seedling numbers—although O. usambarensis is considered shade tolerant (Bussmann, 2001; Babaasa et al., 2004). An initially increased light availability subsequent to logging usually enhances seedling establishment of both pioneers and late successional species (Mwavu and Witkowski, 2009). Consequently, late successional species like O. usambarensis may profit from high light availability (Makana and Thomas, 2005) as long as the competition with faster growing pioneers for light and other factors does not counteract the effect. Seedlings of late successional species establish more frequently in small gaps, while pioneer species usually dominate in large gaps that result from logging (Chapman and Chapman, 1997). This theory is supported by Babaasa et al. (2004), who only found saplings of O. usambarensis in natural gaps within undisturbed forests and not in formerly logged areas. Ten years after logging cessation, gaps are dominated by pioneers and light incidence is not correlated with logging intensity anymore. According to Lind and Morrison (1974), O. usambarensis grows most on the top and sides of narrow ridges, where light incidence is naturally higher. Altogether, it seems to be the high light demand, among other as yet unknown factors, that hampers seedling establishment of O. usambarensis both after logging and below canopy in undisturbed stands. Now further research is needed to identify the safe sites for germination and establishment.

Vegetative regeneration through root suckers was more important than sexual regeneration. But in contrast to our initial hypothesis, we found a negative correlation between logging impact and root sucker number. Vegetative regeneration, especially through root suckers, is a widespread trait in tropical tree species. It is considered to be an adaptation to the low radiation environment in the understory of mature forests, where the established root system provides sufficient assimilates for the successful growth of resprouts (Kammesheidt, 1999). Root suckers found in undisturbed sites may serve as a "ramet bank." Such vegetative sprouts can persist many years in the understory with reduced growth rates and may later have an advantage in stand regeneration once an opening appears in the tree canopy (Kowarik, 1995). Vegetative regeneration is also a common response to anthropogenic disturbance (Rijks et al., 1998; Del Tredici, 2001).

Vegetative regeneration through root suckers is generally such a typical trait for O. usambarensis that it has been suggested as a way of managing timber plantations (Willan, 1965). Yet in heavily logged areas, the number of root suckers was reduced and small trees (DBH <0.5 m) were missing. This may be explained by two effects of former logging. (i) Soil compaction after mechanized logging with heavy machines has been hypothesized to hamper vegetative regeneration of tree species in logged neotropical forest stands (Kammesheidt, 1999). At Mt. Kenya logging spread to more remote areas during the 1990s, after new roads and tracks were established (Bussmann, 1996). Consequently, improved access for more people and heavier equipment may have similarly impacted soil and root sucker growth in O. usambarensis. (ii) Moreover, direct destruction of roots and root suckers before and during logging is likely to have played a role. Although a presidential ban on harvesting of live indigenous trees was enacted in 1986, dead wood was still allowed to be removed. Bussmann (1996) described a timber harvesting method that involved killing trees by digging up the roots, then leaving the tree for some time, and returning to collect the "naturally fallen" material later. This method apparently destroys root suckers (Bussmann, 1994). Moreover, root suckers that were still alive at the time of timber removal were probably prone to destruction by trampling during sawing and timber transport. These particular circumstances at Mt. Kenya may thus have prevented the otherwise typical vegetative regeneration by root suckers.

5. Conclusions

Overall, our results show that natural recovery of the *O. usambarensis* population at Mt. Kenya is slow in heavily logged areas and these areas represent by far the largest part of the remaining forest. Even 10 years after logging suspension, neither seedlings nor root suckers have grown to bigger size classes in these areas. Sexual regeneration appears to require particular site conditions that are generally not readily met in this forest. As already observed by Bussmann (1994, 2001), there still seems to be a negative legacy effect of logging on the number of root suckers. Consequently, the better forest protection enacted in 2000 is not sufficient. Instead the *O. usambarensis* population requires adapted restoration schemes.

Other authors have assumed that it will presumably be difficult to restore pristine *O. usambarensis* forest quickly (Aerts et al., 2011). The alteration of the forest structure through logging by removing the best adult stems may have weakened the population and accelerated an already ongoing decline caused by other unknown factors. Therefore, enrichment planting should be considered in heavily degraded areas. *Ocotea usambarensis* was successfully planted for traditional forestry with rotations of 75 years (Kigomo, 1987), but management experience from the few remaining plantations in Kenya does not give a clear picture of successful management options (B.N. Kigomo, personal communication). In the remaining forest areas, enrichment planting may be necessary to increase population size (Montagnini et al., 1997). As an additional measure, Bussmann (1994) proposed encouraging the local communities to establish tree nurseries and tree planting schemes in areas below the forest. These measures may contribute to counteracting the severe lack of regeneration in the remaining forest. However, future research is needed to assess the long-term development of the whole forest ecosystem.

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