



Forest structure and composition of previously selectively logged and non-logged montane forests at Mt. Kilimanjaro



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ABSTRACT

The montane forests of Mount Kilimanjaro in Tanzania have been subjected to a long history of selective logging. However, since 1984 logging of indigenous trees is prohibited. Today, these forests allow us to evaluate the long-term effects of selective logging. We mapped the height and diameter at breast height (DBH) of all trees >10 cm DBH on 10 sites of 0.25 ha. Five sites represent non-logged forests, another five selectively logged forests. We tested whether forests were still visibly affected 30–40 years after selective logging in terms of their forest structure and tree diversity. Additionally we compared tree densities of different species guilds, including disturbance-indicator species, late-successional species and main timber species. Furthermore, we specifically compared the community size distributions of selectively logged and non-logged forests, first across all species and then for the most important timber species, *Ocotea usambarensis*, alone. 30–40 years after selective logging forests still showed a higher overall stem density, mainly due to higher relative abundances of small trees (<50 cm DBH) in general, and higher densities of small size class stems of late-successional species specifically. For *O. usambarensis*, the selectively logged sites harboured higher relative abundances of small trees and lower relative abundances of harvestable trees. The higher relative abundance of small *O. usambarensis*-stems in selectively logged forests appears promising for future forest recovery. Thus, outside protected areas, selective logging may be a sustainable management option if logging cycles are considerably longer than 40 years, enough large source trees remain, and the recruiting *O. usambarensis* individuals find open space for their establishment.

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1. Introduction

Undisturbed tropical forests have become extremely rare (Gardner et al., 2009). Thus, forest management should focus on maximizing the conservation values of human-modified forest (Gardner et al., 2009). In this context, selective logging was proposed as a management option to maintain conservation values, in terms of carbon stocks and biodiversity, as well as the economic value of a once-logged forest (Putz et al., 2012).

Nowadays, a typical selective-logging cycle occurs at 30–40 years intervals even though selectively logged forest might still be degraded after 100 years (Bonnell et al., 2011). However, the effects of selective logging on the structure, dynamics and recovery of forests remain uncertain (Bonnell et al., 2011). This uncertainty calls for a thorough understanding of the effects of selective

logging on forest dynamics and regeneration, both in terms of structure and composition. A degraded forest may be recognised by changed recruitment resulting in smaller tree size and a higher stem density than in an undisturbed forest. Additionally, degraded forests are likely to have less abundant late-successional and more abundant disturbance-indicator species.

The montane forests of East Africa have been subjected to a long history of forest degradation, mostly due to selective logging (Ndangalasi et al., 2007; Bonnell et al., 2011; Hemp, 2006b; Bussmann, 1996; Kleinschroth et al., 2013; Persha and Blomley, 2009). The main target timber species for selective logging was the East-African camphor tree, *Ocotea usambarensis* (Bussmann, 1996; Kleinschroth et al., 2013), because it yields excellent timber which is resistant to fungal damage (Schulman et al., 1998). Harvesting stems between 50 and 90 cm DBH provided the most efficient wood extraction from forests as smaller trees yield little timber and larger trees require too much effort (Persha and Blomley, 2009). Selective logging was thus driven by the need for

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timber even before selective logging became a widely recommended management alternative to clear-cutting. The long history of selective logging in East African forests renders them ideal study systems when evaluating the long-term effects of selective logging and to inform future management decisions.

Kilimanjaro's montane forests have been subject to selective logging, especially before 1984, when severe forest destruction led to the banning of logging (Agrawala et al., 2003). To study the effect of past selective logging on these forests, we measured variables of vegetation structure, population structure and spatial arrangement of individual stems for non-logged forest sites and sites which were selectively logged 30–40 years ago. We expected formerly selectively logged forests to still have lower mean tree size, basal area, density of large trees and a higher overall stem density, as well as reduced tree species richness, Shannon diversity and pairwise Sørensen diversities compared with non-logged forests. Furthermore, we expected selective logging to result in higher densities of disturbance-indicator species but lower densities of late-successional species and main timber species. As a certain size class is subjected to selective logging we expected degraded tree size distributions in selectively logged forests compared with non-logged forests.

2. Methods

2.1. Study area and sites

Mt. Kilimanjaro, Tanzania, is located 300 km south of the equator and stretches from the savannas at 800 m above sea level (asl) to the snow-covered summit at 5895 m asl. Due to a wide precipitation gradient the Southern and Eastern slopes of Mt. Kilimanjaro have a wide spectrum of habitats in distinct vegetation zones. These include a distinct forest belt which ranges from about 1800 m to 3200 m asl (Hemp, 2006a). The forests in the middle montane zone (2100–2800 m asl) are dominated by *O. usambarensis* (Lauraceae; Hemp, 2006a), which due to its high commercial value was the main target for selective logging. Selective logging in our study area, the southern slope, was done from several sawmills located inside the forest belt. The number of stems extracted was usually quite low and very unlikely to have exceeded a dozen trees per hectare. However, on the drier eastern slope overexploitation has resulted in forests free of mature *Ocotea* with the same structure and otherwise the same species composition. These potential montane *Ocotea*-forests cover an area of about 110 km². This means that one third of the actual montane *Ocotea*-forest is already depleted of *Ocotea* (Hemp, 2006b). In 1984 logging was banned and in 2005 these forests were protected by inclusion in the National Park. Although there is still illegal selective logging in many areas of the mountain (Lambrechts et al., 2002), most selective logging along our study transects took place before 1984 (Agrawala et al., 2003). Our study sites were located in the montane zone at a mean annual temperature of 15–18 °C and mean annual precipitation of 2700 mm/year at 2200 m asl (Hemp, 2006a).

Between September 2011 and March 2012 we established 0.25 ha sites at five non-logged sites and at five sites which had been selectively logged 30–40 years ago. We chose the selectively logged sites for our study based on long-term expertise and local contacts in this specific area. While we are certain that these sites were selectively logged more than 30 years ago, the absence of young stumps indicated that there was no further logging since then. The sites were distributed over a 31 km east–west stretch along the southern, south-eastern slope and at elevations between 2120 m and 2750 m asl. The average distance between sites was 14 km with a minimum distance of 300 m.

2.2. Tree inventory

In each site we mapped all trees larger than >10 cm diameter at breast height (DBH) by measuring the distance and direction to known coordinates with an ultrasonic range finder (Haglöf Vertex IV Hypsometer, Langsele, Sweden) and a compass (Suunto KB-14 precision compass). For each tree we measured DBH, height and crown extension. We measured DBH with a tape measure and, for large trees with buttresses only, we used a laser dendrometer (Criterion RD 1000; TruPulse 200/200, Centennial, USA) to measure the diameter directly above the buttresses. To estimate tree heights and crown extensions in all four cardinal directions (N, E, S, W) we used ultrasonic rangefinders. Finally, we identified each tree to the species level, and assigned the species to disturbance-indicator species and late-successional species (Table S2) based on species occurrence in a large survey of several hundred vegetation records across the mountain (A. Hemp, unpublished data).

2.3. Analysis

We assessed the impact of past selective logging on plot-based measures of vegetation structure and diversity (assessed as mean DBH, mean tree height, stem density per ha, basal area per ha, crown area per square meter, tree species richness and tree species Shannon diversity and the species guilds) with one-way analysis of variance (ANOVA). We used Tukey's Honest Significant Difference post hoc tests within species guilds to test for differences between tree sizes.

To test the impact of logging on species composition, we calculated Shannon's diversity index and the pair-wise Sørensen's index using the diversity-function and the *vegdist*- and *betadisper*-functions in the 'Vegan 2.0-8' package (Oksanen et al., 2013), respectively. Then, we used an ordination analysis (with the *metaMDS*-function in the 'Vegan 2.0-8' package) to test whether logging affected community dissimilarity as assessed by the pair-wise Sørensen's index. In this analysis we used elevation, precipitation, UTM X- and Y-coordinates, slope, disturbance and a landscape index as predictor variables (Table S2). For each plot, slope was defined as the average inclination from the centre of the sites to all trees at distances between 5 and 15 m. The landscape index represents the proportion of man-made habitats (tree plantations, cropland, grasslands and traditional home gardens) within a 1.5 km diameter around the target site as assessed from satellite images (Nauss et al., 2014).

To assess differences in population structure, for all species and for *O. usambarensis* only, we classified the trees into small (<50 cm DBH), harvestable (50–89.9 cm DBH) and large (>90 cm DBH) trees (Persha and Blomley, 2009). We used a Chi-squared test (as implemented as *chisq.test*-function in R; R Development Core Team, 2014) to test for differences between logged and non-logged forests in numbers of stems per size class. Then we performed 2-sample tests for equality of proportions to test differences in numbers of stems between logged and non-logged forests separately for each size class (using the *prop.test*-function in R; R Development Core Team, 2014).

Additionally, we assessed the direct neighbourhood of large trees in order to reveal patterns at smaller spatial scales. We calculated the percentage of small trees, disturbance-indicator species and late-successional species (see above and Table S1) in the neighbourhood of each large tree, for which we could consider the full 5 m radius without interference by the site borders. The 5 m radius had been proposed earlier as the minimum distance relevant for neighbour effects (Stoll and Newbery, 2005). For each site, we grouped the points in each neighbourhood into two groups per pattern (small/large size, disturbance indicator/non-disturbance indicator and late-successional/non-late-successional).

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