Human impacts on forest structure and species richness on the edges of a protected mountain forest in Uganda

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ABSTRACT

We investigated how local scale variation in human impacts influenced forest structure and tree species richness within Mt Elgon National Park, Uganda. We assessed basal area (BA), stem density, diameter at breast height (dbh) and indicators of human activity in 343 plots in four study sites, on transects running inwards from the boundary of the park. Mt Elgon hosts the only remaining natural forest in a densely populated region (150–1000 p/km^2). All study sites suffered past encroachment for agriculture and were in various stages of recovery or renewed-clearing at the time of the study. Areas recovering from encroachment had lower mean BA (BA = 3–11 m^2/ha), dbh and often also lower stem densities than forest that had never been cleared (BA = 21–43 m^2/ha), even 35 years after abandonment and with restoration planting. Human impacts were found beyond 2 km into the park. Although most activities decreased with distance inside the boundary, their prevalence varied among sites. High coefficients of variation in BA (Cv = 0.8–1.1) and stem density (Cv = 1.0–2.2) within sites, together with the evidence of sustained human activities, suggest that forest use histories strongly influenced local forest structure. Mean BA increased with distance inside the boundary in all sites, but stem densities reflected more complex patterns. Large trees (dbh ⩾ 20 cm) were most affected by former clearing for agriculture. The collection of stems used as crop-supports reduced regeneration and the density of smaller stems at one site. In another site, charcoal making was associated with the smallest mean BA and marked variability in forest structure. Grazed forest consisted of large trees with very little regeneration. On forest margins in two sites grazing, generally together with fire and tree-cutting, had eroded the forest edge and prevented regeneration. Human impacts as well as natural gradients had major impacts on species richness patterns. Several areas in intermediate states of disturbance showed higher tree species richness than either old-growth forest or more severely degraded areas. This study illustrates the fine scale variation due to local impacts within one forest.

1. Introduction

Most tropical forests, even those in protected areas, are influenced by human activity (MacKenzie et al., 2012; Olupot et al., 2009). Harvesting of forest resources to meet livelihood needs can impact forest regeneration, structure and diversity (Fashing et al., 2004; Olupot, 2009), but there is scope for considerable variation with location, human activities and histories. To better manage forests for multiple local, regional and global values we need to understand human impacts and their variation at local scales. Such understanding is pertinent for forests managed for biodiversity conservation, catchment values and tourism that are increasingly considered in terms of their carbon stocks and the various benefits that they can provide to local people.

Different types and intensities of local resource extraction can lead to varying outcomes even within one forest (Thapa and Chapman, 2010). For example, forest grazing leads to different impacts than cutting timber or gathering other forest products (Fashing et al., 2004; Vadjunec and Rocheleau, 2009), and intensive extraction of certain highly valued species may have a greater impact on diversity than less intense forest uses (Ndangalasi et al., 2007).

Distance from settlements is often considered as a proxy for the extent of human impacts on forest (Boudreau et al., 2005), but preferred forest resources may not be evenly distributed and differ among groups of people. Environmental gradients like elevation, slope, substrate and moisture can confound results based on distance. For example elevation is known to affect tree size and species diversity (Ghazoul and Sheil, 2010), but human activities are

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also likely to be more intensive in lower elevation forest that is easier to access than on more remote, higher elevation sites.

In densely populated landscapes remaining natural forests have generally been subjected to multiple human impacts. The resulting complexity and the challenge of defining simple cause and effect relationships may explain why these patterns have seldom been studied in detail. Yet the diversity in human activities and their impacts call for different interventions. For instance, different approaches may be required where people have long used forest as a source of medicinal products or foods, or as a location for cultural activities, than in areas where people claim forest-land for agriculture. Historical factors such as conflicts over boundaries may also influence attitudes and behaviours towards forest management (Cernea and Schmidt-Soltau, 2006).

In this paper we investigate how use of the forest by communities on the edge of the Mt Elgon National Park (Uganda) has affected local forest structure and tree species richness. This is part of a linked series of studies that examine these forests and their relationship with local people. In a previous paper we have described the processes, contexts and drivers that led to localised episodes of forest loss and recovery over recent decades (Sassen et al., 2013). Here we look more closely at the nature of the resulting forests. We conducted a detailed comparative analysis of four study sites that vary in terms of the local land-use and the history of forest clearing and regeneration. We studied the variation of local activity and their ensuing impacts. We addressed the following questions: (1) How do indicators and measures of human activity vary within and among sites? and (2) How do forest structure and diversity vary with these indicators?

2. Site and methods

2.1. Mt Elgon

Mt Elgon (4321 m) is an extinct solitary shield volcano from the Miocene on the border between Uganda and Kenya. The top is an 8 km wide crater. The slopes are generally gentle until 2800–3000 m, with characteristic steep cliffs dropping down to the plains in the north, and some steeper slopes in the south-west. The mountain's slopes are cut by river and stream valleys that run down the mountain from the caldera (Fig. 1). A 20 km long ridge extends towards the west (Fig. 1). Dry north-easterly and moist south-westerly winds determine the climate. Rain falls year-round but peaks in April–May and September–November. Annual precipitation is between 1500 and 2000 mm. Rainfall is higher on the southern and western slopes than on the northern and eastern slopes and most rain falls at between 2000 and 3000 m above sea level (Dale, 1940; IUCN, 2005).

Mt Elgon is an important water catchment area for several million people in the surrounding districts, for the Nile and Victoria basins as well as Lake Rudolf through the Turkwell River (IUCN, 2005). A belt of bamboo and afromontane forest is found at on average between 2000 and 3000 m, followed by heathers and high elevation moorland (Dale, 1940; van Heist, 1994). Mt Elgon is valued for its global biodiversity values (Howard, 1991). It hosts 39 endemic higher plant species as well as many species with limited distributions (for details see Davenport et al., 1996; IUCN, 2005). Wildlife consists mainly of various monkeys, small ungulates and bush pig (*Potamochoerus larvatus*); rodents and birds are abundant (Davenport et al., 1996), but larger wildlife, in particular elephant (*Loxodonta africana*) and buffalo (*Syncerus caffer*), are found mainly on the Kenyan side of the mountain (van Heist, 1994).

Mt Elgon National Park in Uganda (1120 km^2) has a long history of human influence. As long as people remember, its forests provided a broad range of products and services such as fuelwood,

medicine, food, materials for construction, grazing for cattle, cultural sites (e.g. circumcision rituals, burial) and shelter against cattle raiding (Katto, 2004; Scott, 1994).

The slopes of Mt Elgon in Uganda are inhabited by two ethnic groups. The Bagisu, of Bantu origin, dominate the south and south-west since around 1500 AD. Their population density reaches over 1000 people/km^2 in places (UBOS, 2002c). They practice an intensive mixed agriculture dominated by coffee and banana (Kayiso, 1993; McMaster, 1962). Important forest products for the Bagisu include construction materials, bamboo stems and shoots and crop-supports (called "crop-stakes" from here-on) for bananas and for climbing beans (Sassen, unpublished data; Scott, 1994).

The second group is the Sabiny, a Nilo-Cushtic group of pastoralists, settled in the north and north-east from the 17th century. They lived on the edges of open grassy areas inside the forest (called "glades") on the higher slopes of Mt Elgon, until they were resettled down the mountain in the 1980s. Land for resettlement was allocated in an excision from the protected area, which was then still a forest reserve (van Heist, 1994). The forest in the excised area was rapidly converted to agricultural land (Scott, 1998) where people cultivate maize, potatoes, wheat and maintain pastures as cattle remain important. The Sabiny still use the forest and the glades (up to 3 km inside the park boundary) for (illegal) grazing, timber, medicine and wild foods (Norgrove, 2002; Scott, 1994).

Communities living near the park are poor and suffer land shortages; nearly all land directly surrounding the park is cultivated (IUCN, 2005; van Heist, 1994). There are no remnant forests within 20 km around the park and people are settled up to right next to the park boundary (Sassen et al., 2013). On the western and southern slopes trees are part of the agricultural system. They are found in combination with coffee and bananas, around homesteads and in valleys planted with *Eucalyptus* woodlots. In the north, where people are more recently settled, trees outside the park are scarce, particularly nearer to the park boundary. A few isolated former forest-canopy trees remain scattered amongst the fields (Sassen et al., 2013).

Political instability from 1971 until 1986 was associated with widespread encroachment of Uganda's forest reserves (Hamilton, 1985; Turyahabwe and Banana, 2008) and around 30% of Mt Elgon was cleared for agriculture (Sassen et al., 2013). From 1987, forest boundaries were reinstated and restoration activities were started on the western slopes (UWA, 2000). The forest on Mt Elgon was first gazetted as a reserve in 1938 and became a national park in 1993 (Scott, 1998). Since the late 1990s, Uganda Wildlife Authority (UWA), which manages the park, has initiated agreements with local people that allow regulated collection of non-timber products, fuelwood and crop-stakes from restricted non-tree species (*Mimulopsis arborea* and *Vernonia* spp.) (Scott, 1998; UWA, 2000). Although activities such as pit-sawing declined after the establishment of the national park (Scott, 1998), illegal resource extraction remained common at the time of our study. Law enforcement efforts were understaffed and overstretched but also felt that they could not always stop people from harvesting essential resources such as firewood (A. Bintoora, Conservation Area Manager, personal communication; personal observations).

2.2. Data collection

Four locations were selected to represent different elevation ranges and forest change histories. These locations are subsequently referred to as Sites 2, 9, 11 and 14 (see Table 1 for site codes and corresponding villages) – these numbers are the same as those used in Sassen et al. (2013). The communities near Sites 2 and 9 practise intensive coffee–banana based agriculture, while
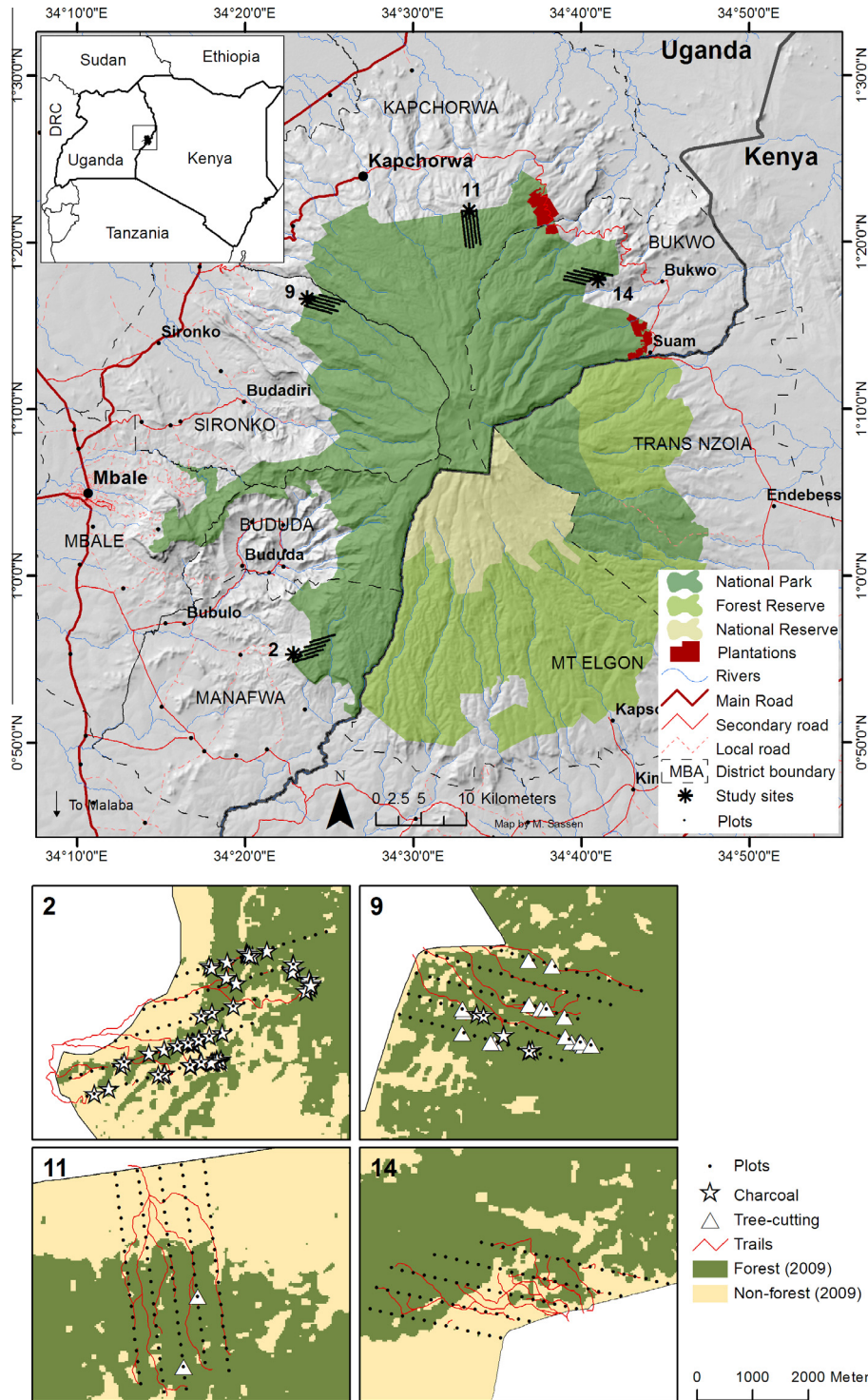
Fig. 1. Map of Mt Elgon, Kenya/Uganda and study sites. The small maps of the study sites show the location of the plots with a background of forest (dark)/non-forest (light) from the classification of a 2009 Landsat ETM+ image (Sassen et al., 2013). The locations of large (>40 cm) cut trees and charcoal-burning pits encountered in the field were marked by GPS and are shown on the maps.

those near Sites 11 and 14 grow mainly maize and potatoes. Sassen et al. (2013) assessed the patterns of forest loss and recovery around Mt Elgon, including in the four study areas. They found that people near Site 9 did not re-encroach the forest inside the park after boundaries were reinstated and forest restoration began around 1990. In the other sites however, renewed clearing and recovery took place at different times (Table 1). In Site 2 most regenerating vegetation and restoration planting was cleared again from 2006. Forest clearing in Site 11 started in the 1990s, when a

controversy arose about the boundary of the area excised for resettlement (see also Himmelfarb, 2006). In Site 14, encroachment was patchier than in the other sites (Sassen et al., 2013). At the time of our study, the parish neighbouring Site 9 had an agreement with UWA to collect resources such as fuelwood, green vegetables and medicinal plants in limited amounts, twice a week.

Fieldwork took place from November 2010 till April 2011. In each site, five parallel transects were laid out 400 m apart and as perpendicular as possible to the general orientation of the

Table 1
Study villages, land-use and history of encroachment.

	Site 2	Site 9	Site 11	Site 14
Village (2011)	Bukuwa	Kinyofu/Gibuzale	Korto/Kamatelon	Sindet/Kapsata
Sub-county (2011)	Bupoto	Masira	Kwosir	Kortek
Population density 2002[a]	631 p km^{-1}	712 p km^{-1}	448 p km^{-1}	374 p km^{-1}
Main cash crops[b] (% times listed)	Onions (34%), cabbages (34%), coffee (28%)	Coffee (69%), cabbages (35%)	Potatoes (37%), maize (28%)	Maize (98%)
Main food crops[b] (% times listed)	Maize (53%), banana (57%)	Banana (60%), maize (42%),	Maize (82%), potatoes (73%)	Maize (98%)
Mean number trees/hh	33	27	3	8
Resource use agreement	No	Yes	No	No
Main periods of encroachment	1979–1992, 2006–2008, 2010	1979–1992	1990–2008	1985–1997, 1991–1993, 2008 (patchy)
Restoration planting 1990s	+−	++	−−	−−

[a] UBOS (2002b,d,a).
[b] Based on number of people listing the crop as either their first or second cash crop (so total% >100).

boundary while ending in relatively intact forest (Fig. 1). The centre of the first plot on each transect was taken 50 m inside the park boundary and further plot-centres at 200 m intervals. We used a handheld GPS (Garmin 60CSx) to determine the position of the plot-centres along the transect line. We sampled 13–21 plots along each transect. The number depended on the travel time between each plot, which was influenced by terrain (e.g. obstacles) and vegetation (denser undergrowth required more clearing to discern the reference height of 1.30 m on tree stems). Each transect ended with at least two or three plots in areas that, according to our informants, were too far from the boundary to be used for poles and firewood. Some pit-sawing and/or charcoal burning would nevertheless still occur. We could not establish controls because any less disturbed site would not be similar, i.e. at higher elevation or less accessible. Hence we used distance effects to gauge impact intensity from assumed high to low.

We used a relascope (horizontal point sampling) approach for tree selection and direct basal area (BA) estimation (Bitterlich, 1984). This method allows quick sampling of many plots, with minimal accuracy differences compared with fixed-area plots (Piqué et al., 2011). During the 360° sweep from each plot-centre, a tree was counted as "in" if it was wider than the relascope notch; borderline trees were checked with their dbh and distance from the plot-centre. The BA of each plot was calculated by multiplying the number of "in" trees with the BAF (Bitterlich, 1984). Correction was later done for slope. Starting with the first plot, in every fifth plot after that we measured the diameter at breast height (dbh at 1.30 m) of each "in" tree. In these plots (referred to as "detailed-plots"), we also used a checklist to record whether branches or stems had been cut-off trees (termed "lopping"), whether the tree was alive and whether it had been planted. Sub-plots of 5 m diameter were used to measure stumps (<1.30 m in height) and count saplings smaller than 2.5 cm dbh and between 2.6 and 5 cm, but taller than 1.30 m and seedlings shorter than 1.3 m.

We scored each plot for signs of human activity such as trails and trampling, agriculture, fire (as evidenced from charring of stems and stumps), pit-sawing, charcoal burning (pits), pole cutting (cut stems) and other signs of wood harvesting. The collection of naturally fallen firewood does not leave obvious signs, but signs of wood splitting were found as waste or piles of split stems drying away from trails. Scores were assigned on a scale of 2 (absent and present) or 3 (absent, present, and severe) depending on the indicator (Table 3). Additionally, we recorded the location of any charcoal pits or signs of pit-sawing encountered while moving between plots (Fig. 1).

Tree species (standing trees and recently cut or coppicing stumps) were identified by cross-referencing names given by local informants and two knowledgeable rangers (one Bagisu and one Sabiny) with available references (Hamilton, 1991; Katende et al., 2000) and later at the Institute for Tropical Forest Conservation (ITFC), Uganda using photographed specimens.

2.3. Data analysis

Plots were classified into four categories according to encroachment history as reported by local informants: $c4$ = currently cleared and cultivated or grazed, $c3$ = cleared in the 1990s and 2000s now recovering, $c2$ = cleared in the 1970s and 1980s now recovering, $c1$ = not cultivated within living memory (called "old-growth" from here-on). In Site 11, $c2$ plots consisted mainly of plots in former settlement or grazing areas inside the forest from which people were relocated from halfway the 1980s.

Due to the irregular shape of the park boundary, plot position along a transect is not the same as distance inside the boundary, except in Site 11 (Fig. 1). The actual distance to the boundary was derived for each plot using a GIS (ArcGIS 10.0). We calculated BA and density of trees, seedlings, saplings and stumps per hectare for each plot. Human activities were gauged using stump density, lopping intensity and the activity indicators. From the indicators, we calculated mean scores per activity for each site but also for distance classes into the park (Fig. 2). Lopping intensity was calculated for each plot as the proportion of trees with signs of cutting (branches or stems).

We used two methods to calculate species richness correcting for the unequal numbers of stems per plot. We used "BiodiversityR" (Kindt and Coe, 2005), based on the "vegan" package in R (Oksanen et al., 2012; R Development Core Team, 2011) to calculate rarefied species richness for plots with a minimum of five stems (using Hurlbert's (1971) formulation). We also calculated Z = (species count)/log (stem count) per plot (all plots $\geqslant$ 2 stems) as a measure of species richness corrected for stem density, which allows for smaller sample sizes (Sheil et al., 1999).

Linear regression, analysis of (co-)variance (ANCOVA/ANOVA) and non-parametric tests (Kendall's tau-b rank correlation, Kruskal–Wallis (KW), Mann–Whitney U (U)) were used to investigate the relationship among forest structure, species richness, distance inside the boundary and indicators of human activity. There were no significant interactions among the activity indicators and we did not investigate how the covariates affected interactions between factors (Yzerbyt et al., 2004). Plots in areas that were cultivated at the time of the study (in Sites 2, 11 and 14), though within the national park, lacked much natural vegetation and we therefore also analysed the data without these areas. Statistical analysis was done using SPSS version 18.0 (SPSS Inc., Chicago IL).

Table 2
Summary of study site characteristics.

	Site 2	Site 9	Site 11	Site 14
All plots (detailed-plots)	76 (17)	84 (20)	101 (25)	52 (19)
Altitude range in masl	1911–2318	2152–2606	2478–2877	2238–2699
Recorded live stems	403	553	1139	627
Recorded tree species	39	32	17	30
Activity indicators:				
Plots with trails	46 (61%)	78 (93%)	67 (66%)	62 (76%)
Plots with cut stems	43 (57%)	60 (71%)	51 (50%)	35 (43%)
Plot with split wood	14 (18%)	34 (40%)	24 (24%)	11 (13%)
Plots with grazing	0 (0%)	12 (14%)	32 (32%)	26 (32%)
Plots with cultivation	19 (25%)	0 (0%)	30 (30%)	5 (6%)
Plots with fire signs	6 (8%)	7 (8%)	3 (3%)	26 (32%)
Plots with charcoal pits	12 (16%)	3 (4%)	0 (0%)	0 (0%)
Plots with pit-sawing	0 (0%)	4 (5%)	2 (2%)	0 (0%)

Table 3
Kruskall–Wallis or Mann–Whitney U tests for the relationship between distance from the boundary and activity indicator scores per plot (currently cultivated areas were omitted).

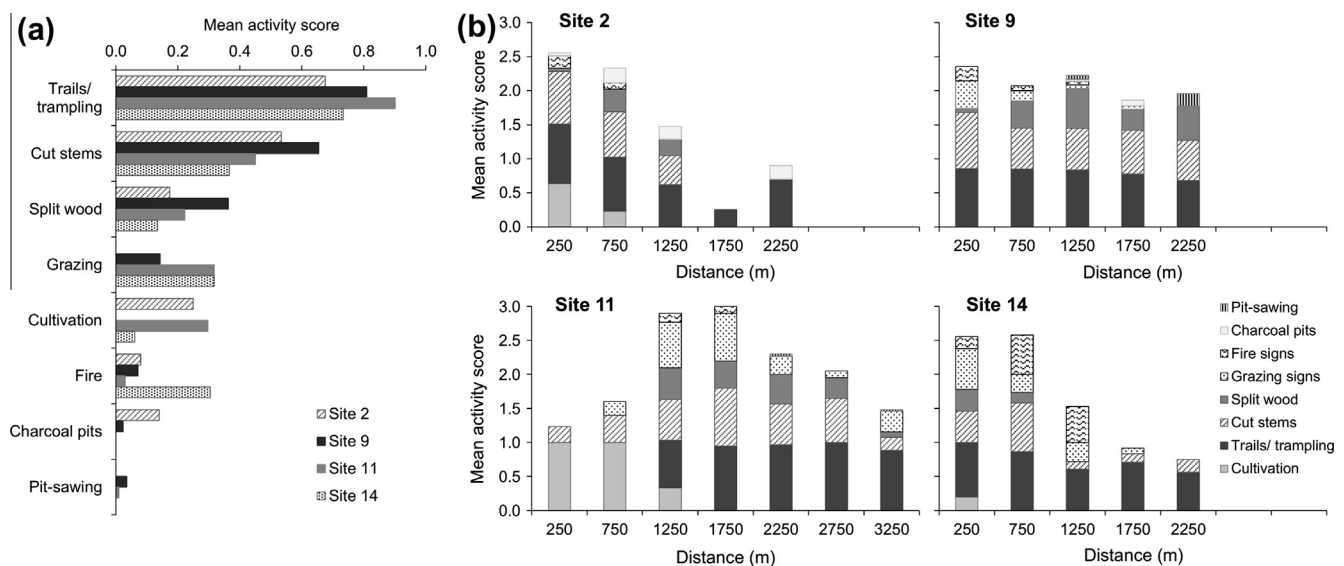|	Trails and trampling	Cut stems	Wood splitting	Grazing	Fire	Charcoal	Pit sawing
Site 2							
df	2	2	2		1	2	
KW/U^*	14.5	14.7	2.5		15[*]	0.06	
n	57	57	57		57	57	
p	0.001	0.001	0.286		0.001	0.973	
Site 9							
df	2	2	2	1	2	2	2
KW/U^*	7.5	7.0	4.1	171[*]	5.7	2.1	4.9
n	84	84	84	84	84	84	84
p	0.024	0.030	0.126	0.001	0.057	0.343	0.085
Site 11							
df	2	2	2	1	1		1
KW/U^*	13.4	16.3	13.0	316[*]	9[*]		85[*]
n	67	67	67	67	67		67
p	0.185	0.000	0.002	0.005	0.039		0.499
Site 14							
df	2	2	1	1	2		
KW/U^*	37.1	14.5	59[*]	129[*]	0.7		
n	61	61	61	61	61		
p	0.029	0.001	0.000	0.000	0.688		

Fig. 2. Mean scores per site for each activity indicator (a) and mean scores per site at distance class mid-points from the boundary (b). Note the different scale for Site 11 in (b) as transects were longer in this site. Trails and trampling signs were only counted in non-cultivated plots. In Site 11 the boundary was straight and all cultivated plots were situated in a 1000–1200 m wide strip where cultivation or pasture dominated all other possible disturbances.

3. Results

In total we assessed 343 plots, 76 in Site 2, 84 in Site 9, 101 in Site 11 and 52 in Site 14. Elevation ranged overall between 1911 and 2877 m above sea level but ranges also differed among the four sites (Table 2). We recorded 2722 live stems using the relascope method, of which 593 were measured for dbh in detailed-plots (Table 2). In total 61 species were recorded, although 3 of these were only identified to family and 6 remained unknown. The incidence of human activity indicators per site is summarised in Table 2.

3.1. Disturbance

3.1.1. Activity indicators

Fig. 2 summarises the scores for the human indicators per site (a) and with distance inside the boundary (b).

The types and degrees of disturbance varied within and among sites (Fig. 2a), and indicators of human activity extended more than 2 km from the boundary into the park in all study sites (Fig. 2b). Site 9 scored highest on cut stems and split wood (Fig. 2a). In Site 11, all plots in the first 1000 m along the transects were cultivated or grazed (Fig. 2b).

Signs of disturbance tended to decrease with distance inside the boundary, with variations per site and activity (Table 3). Plots that were cultivated at the time of the study were excluded from the test as agriculture dominated all other impacts. Trails continued into the forest beyond our last plots at all sites. Plots that scored high on trampling were closer to the boundary than plots with no trails or intermediate scores (Table 3). Except in Site 11 where wide and well-worn trails were found up to the former settlement and grazing areas at around 3000 m from the boundary (Fig. 3). Plots nearer the boundary tended to score higher for cut stems (Fig. 2b and Table 3). Split wood drying in the forest was widely observed in Site 9, but it decreased with distance inside the boundary in the other sites. This decrease was significant in Site 11 and 14 (Table 3). In Site 2 and 11, signs of fire occurred mostly on the edge between cultivated areas and forest at around 500 m and 1000 m inside the boundary respectively (Fig. 2b), whereas in Site 14, they were found in grassy areas scattered among degraded patches of forest or bush at varying distances up to 1500 m from the boundary (Table 3). In Site 2, charcoal-burning was an important activity and (old) pits were observed at almost all distances (Figs. 1 and 2b and Table 3). Pit-sawing tended to occur further away from the boundary, although the relationship was not significant ($p < 0.05$) (Figs. 1 and 2b and Table 3).

3.1.2. Lopping and stumps

Remnant trees in cultivated areas often showed signs of lopping: they had cut-off branches or stems. Overall, lopping decreased with distance inside the boundary (Kendall's tau-b = −0.472, $n = 49$, $p < 0.001$), and was most common in Sites 9 and 11 (Fig. 4).

No stumps were recorded within 400 m from the boundary (Fig. 5). In Site 9 we found at least one stump per plot in eight of our plots (40%), all more than 800 m from the boundary. This represented a mean density of 504 stumps per ha. In this site 45% of all stems had been cut, but 63% of the stumps (16) were re-sprouting. Stump density in Site 9 decreased with distance inside the boundary (Fig. 5), but the data were too sparse for this to be statistically significant. Stems with diameters commonly used as crop-stakes (3–15 cm) were cut more often than larger ones: their stumps represented 49% of all recorded stems in Site 9.

In the other sites, we found only one stump in each of 2 plots in Site 2 (12%), in 1 plot in Site 11 (4%), and in 2 plots in Site 14 (11%). This represented a proportion of cut stems of 16% in all three sites. The cut stems observed in Sites 2 and 11 were more than 15 cm in diameter. In Site 2 stumps were observed mostly in open places in the forest, associated with signs of past pit-sawing or charcoal-burning.

3.2. Impacts of human activities

Impacts of human activity were assessed for BA, stem density, tree regeneration and species richness.

3.2.1. Basal area

Mean BA in the most intact vegetation type was different among sites (Kruskal–Wallis: KW = 50.5, $n = 147$, $p < 0.001$). Pairwise comparisons revealed that BA was significantly larger in Site 11 compared with all other sites (Table 4). BA per hectare increased significantly with distance inside the boundary in all sites (linear regression: Site 2: adj $R^2 = 0.49$, $F_{1,74} = 71.4$; Site 9: adj $R^2 = 0.54$, $F_{1,82} = 99.4$; Site 11 adj $R^2 = 0.70$, $F_{1,99} = 127.7$; Site 14 adj $R^2 = 0.48$, $F_{1,80} = 75.3$; $p < 0.001$) (Fig. 6). As can be seen in Fig. 6, plots in cultivated areas (c4), with relatively few trees, influenced the relationship between distance inside the boundary and BA. Excluding these plots from the analyses weakened the

Fig. 3. Main trails in Site 9, 14 and 11. Note the scale in relation to the human figures in Sites 11 and 14. The photo in Site 9 is at a similar scale. The trails in Site 11 and 14 are used by cattle.
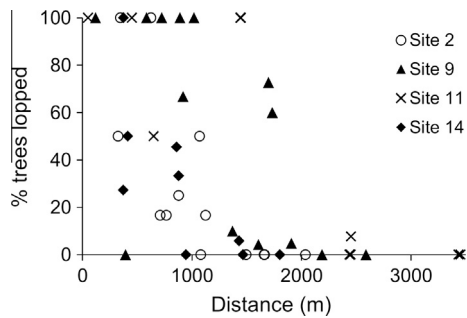
Fig. 4. Lopping intensity (% trees with cut-off branches or stems per plot) and distance inside the boundary.
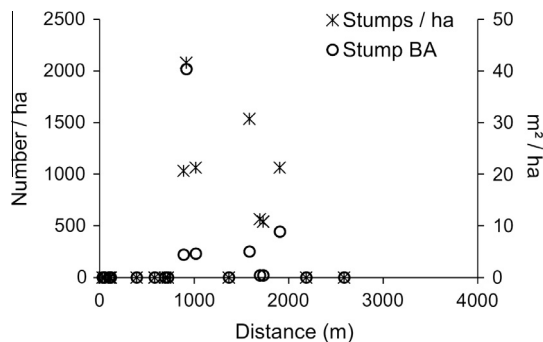

Fig. 5. Stump density (number ha⁻¹) and basal area (m² ha⁻¹) in Site 9.

increasing trend slightly, particularly in Site 11 (linear regression: Site 2: adj R^2 = 0.37, $F_{1,55}$ = 33.8; Site 9: adj R^2 = 0.54, $F_{1,82}$ = 99.4; Site 11 adj R^2 = 0.19, $F_{1,65}$ = 16.5; Site 14 adj R^2 = 0.43, $F_{1,59}$ = 46.9; $p < 0.001$). It made no difference in Site 9 where there was no cultivation at the time of the study. BA in glades (Site 11), (natural) tree fall areas or bamboo patches (Site 2 and 9) further inside the park was lower than in other plots at similar distances from the boundary (Fig. 6).

The proportion of explained variation increased in all sites when activity indicators were included as independent variables and distance, slope and elevation as covariates (Type II ANCOVA: Site 2: adj R^2 = 0.51, $F_{13,43}$ = 5.5; Site 9: adj R^2 = 0.66, $F_{17,66}$ = 40.6; Site 11 adj R^2 = 0.77, $F_{15,54}$ = 16.7; Site 14 adj R^2 = 0.63, $F_{14,60}$ = 10.1; $p < 0.001$). The encroachment category had significant influence on BA per hectare in all sites (p = 0.004 in Site 2 and $p < 0.001$ in the other sites) (see also Table 4), but the contribution of activity indicators varied per site. Pit-sawing (p = 0.002) and wood splitting (p = 0.004) were significantly related to BA in Site 9. Grazing ($p < 0.001$) contributed to explain BA whereas the influence of distance became insignificant.

In Sites 11 and 14 BA seemed to plateau-out at around 2000 m from the boundary, but in the other two sites effects continued further. Local variation of mean BA was high in Sites 2, 9 and 14, as shown by the standard deviations in Table 4.

3.2.2. Tree density

The data for tree density was not normally distributed. We therefore used rank correlations to analyse the relationship between tree density and distance inside the boundary and elevation.

The density of larger trees ($\geqslant 20$ cm dbh) increased with distance inside the boundary in all sites (Site 2: Kendall's tau-b = 0.547, n = 16, p = 0.004; Site 9: Kendall's tau-b = 0.526, n = 20, p = 0.002; Site 11: Kendall's tau-b = 0.596, n = 22, $p < 0.001$; Site 14: Kendall's tau-b = 0.556, n = 19, p = 0.001) (Fig. 7). The density of smaller stems (<20 cm dbh) increased with distance only in Sites

9 and 14 (Site 9: Kendall's tau-b = 0.445, n = 20, p = 0.009; Site 14: Kendall's tau-b = 0.396, n = 19, p = 0.031). When cultivated areas (c4) were excluded these relationships were no longer significant. They remain valid for Site 9, which had no c4 plots.

The density of larger stems was greater in areas that had been abandoned longer ago (Mann–Whitney U or Kruskal–Wallis tests for categories c1 till c3: Site 2: U = 8, n = 12, p = 0.048; Site 9: U = 10.5, n = 20, p = 0.039; Site 11: KW = 10.3, n = 14, p = 0.006; Site 14: KW = 13, n = 5.9, p = 0.052). In Site 11, we found more large trees in plots with signs of wood splitting than in plots without (Kruskal–Wallis: KW = 7.3, n = 14, p = 0.026). In Site 14, we found fewer large trees in plots with signs of fire than in plots without (Kruskal–Wallis: KW = 4.7, n = 13, p = 0.031). Relationships between tree density and other activity indicators were not significant.

We tested if tree density was correlated with elevation outside of currently cultivated plots and found a negative correlation with large tree density ($\geqslant 20$ cm dbh) in Site 2 and a positive correlation in Sites 9 and 11 (Site 2: Kendall's tau-b = −0.443, n = 12, p = 0.046; Site 9: Kendall's tau-b = 0.352, n = 20, p = 0.035; Site 11: Kendall's tau-b = 0.456, n = 14, p = 0.024; Site 14: Kendall's tau-b = 0.128 n = 13, p = 0.542).

3.2.3. Regeneration

Densities of saplings and seedlings varied strongly (Table 4) and were not significantly correlated with distance inside the boundary. We tested for the effect of shading by larger trees on regeneration but found no significant correlation with BA. There was least regeneration in Sites 9 and 11 (Table 4). In Site 9 saplings were recorded only in regenerating areas and 86% originated from coppicing stumps. In Sites 2, 11 and 14, respectively 52%, 33% and 29% of all measured saplings were coppices. The greatest density of seedlings in old-growth forest was found in Site 14 (Table 4).

3.2.4. Species richness

We recorded the largest number of species in Site 2, and the least in Site 11 (Fig. 8). We used two methods for calculating species richness, rarefaction in plots with a minimum of 5 stems and a Z-species richness score corrected for stem numbers that included plots with at least 2 stems. Differences among sites were significant both when using rarefied (5-stem sample) richness (Kruskal–Wallis: KW = 23.1, n = 178, $p < 0.001$) and Z-scores (all plots $\geqslant$ 2 stems) (Kruskal–Wallis: KW = 12.1, n = 243, p = 0.007). Pairwise comparisons revealed greater species richness in Site 2 compared with Site 11 for both methods. Using rarefaction, species richness was also greater in Site 14 compared with Site 11.

3.2.4.1. Rarefied species richness. Plots with a minimum of 5 trees *de facto* excluded the cultivated areas (c4) in Sites 2 and 11 and retained only 2–3 plots in encroachment categories c2, c3 and c4 in all sites, except in Site 9 (Fig. 9a). Rarefied species richness was not significantly correlated with BA or tree density in any of the sites. In Site 2 rarefied species richness was significantly greater in old-growth forest (c1) compared with regenerating areas (c3) (Kruskal–Wallis: KW = 4.7, n = 34, p = 0.031), but there was no correlation with elevation or distance inside the boundary. In Site 9 the only significant correlation was the increase in species richness with distance inside the boundary (Kendall's tau-b = 0.252, n = 44, p = 0.017) (Fig. 9). In Sites 11 and 14 rarefied species richness was correlated only with elevation (Site 11: Kendall's tau-b = 0.213, n = 54, p = 0.023; Site 14: Kendall's tau-b = −0.181, n = 60, p = 0.043).

3.2.4.2. Z-scores. More plots were included in the calculation of Z-scores as these require a minimum of only 2 plots (Fig. 9b). In Sites 9 and 14, Z-scores did not show any significant correlations with

Table 4
Characteristics of the study plots, per encroachment category. Basal area was measured in all plots, other forest structure data only in the detailed-plots. Forest structure data are given as means with standard deviation.

	Site 2				Site 9			
	Never cultivated	Cleared 1970s–1980s	Cleared 1990s–2000s	Currently cultivated	Never cultivated	Cleared 1970s–1980s	Cleared 1990s–2000s	Currently cultivated
General characteristics								
Number of plots, detailed in brackets	36 (8)	0 (0)	21 (4)	19 (5)	21 (4)	63 (16)	0 (0)	0 (0)
Maximum distance to boundary (m)	2619.0		1424.1	739.0	2588.7	1873.5		
Mean altitude (masl)	2173 ± 67		2231 ± 42	1995 ± 74	2427 ± 137	2276 ± 77		
Forest structure								
BA (m^2 ha^{-1})	21 ± 12		3 ± 4	1 ± 2	29 ± 11	10 ± 8		
Stems ha^{-1} > 5 cm	546 ± 1052		210 ± 244	56 ± 101	790 ± 604	433 ± 486		
Dbh of trees ⩾ 5 cm (cm)	57 ± 37		21 ± 17	24 ± 25	62 ± 48	27 ± 23		
Saplings ha^{-1} 2.6–5 cm > 1.3 m	212 ± 422		1508 ± 3017	0 ± 0	0 ± 0	164 ± 461		
Saplings ha^{-1} ⩽ 2.5 cm > 1.3 m	0 ± 0		2033 ± 2362	0 ± 0	0 ± 0	1465 ± 1870		
Seedlings ha^{-1} ⩽ 1.3 m	619 ± 1158		134 ± 267	0 ± 0	901 ± 1802	601 ± 782		
	Site 11				Site 14			
	Never cultivated	Cleared 1970s–1980s	Cleared 1990s–2000s	Currently cultivated[a]	Never cultivated	Cleared 1970s–1980s	Cleared 1990s–2000s	Currently cultivated[a]
General characteristics								
Number of plots, detailed in brackets	50 (8)	7 (2)	10 (4)	34 (11)	40 (8)	9 (2)	12 (3)	21 (6)
Maximum distance to boundary (m)	4046.0	4046.7	2041.5	1449.1	2448.5	1464.9	1135.8	1447.6
Mean altitude (masl)	2807 ± 36	2829 ± 43	2748 ± 14	2587 ± 70	2556 ± 85	2480 ± 124	2434 ± 133	2438 ± 109
Forest structure								
BA (m^2 ha^{-1})	43 ± 13	11 ± 8	7 ± 7	2 ± 2	28 ± 11	9 ± 9	7 ± 7	2 ± 5
Stems ha^{-1} > 5 cm	291 ± 137	79 ± 4	12 ± 5	0 ± 0	490 ± 507	526 ± 664	159 ± 248	1 ± 2
Dbh of trees ⩾ 5 cm (cm)	66 ± 31	70 ± 30	83 ± 27	25 ± 71	64 ± 38	52 ± 26	48 ± 34	14 ± 35
Saplings ha^{-1} 2.6–5 cm > 1.3 m	0 ± 0	0 ± 0	0 ± 0	0 ± 0	277 ± 515	0 ± 0	349 ± 302	0 ± 0
Saplings ha^{-1} ⩽ 2.5 cm > 1.3 m	128 ± 236	0 ± 0	130 ± 260	0 ± 0	1252 ± 1920	255 ± 360	519 ± 513	0 ± 0
Seedlings ha^{-1} ⩽ 1.3 m	64 ± 181	0 ± 0	1298 ± 1499	0 ± 0	1674 ± 2375	255 ± 360	0 ± 0	0 ± 0

[a] Includes pastures kept open through burning and grazing.

BA or tree density. In Sites 2 and 11, *Z*-species richness was negatively correlated with BA (Site 2: Kendall's tau-b = −0.288, *n* = 46, *p* = 0.005; Site 11: Kendall's tau-b = −0.415, *n* = 71, *p* < 0.001) and in Site 11 also with tree density (Kendall's tau-b = −0.617, *n* = 14, *p* = 0.003). *Z*-scores for Site 11 were significantly different between encroachment categories (Kruskal–Wallis: KW = 12.2, *n* = 71, *p* = 0.007). Pairwise comparisons revealed that *Z*-scores were smallest in old-growth forest (c1) and largest in regenerating areas (c3). These were located between cultivated land (c4) and old-growth forest (c1) in this site (Fig. 9b, Site 11). In Site 14, *Z*-species richness was significantly different between encroachment categories (Kruskal–Wallis: KW = 8.0, *n* = 60, *p* = 0.046), with more richness in intermediate encroachment categories (c3 and c2) than in old-growth (c1) or currently cultivated plots (c4) (Fig. 9b), but pairwise comparisons between categories were not significant. In Site 14 the correlations of *Z*-species richness with distance and with elevation were identical and negative (Kendall's tau-b = −0.180, *n* = 60, *p* = 0.043), whereas in the other sites correlations with distance or elevation were not significant.

4. Discussion

The legacies of human activities observed in this study reflected both past events and on-going processes. These showed marked local variations. Such variations have implications for management. Here we shall first discuss the variation in human activities and their impacts on forest structure. Before addressing site-specific impacts, we first discuss more general patterns and extents of human impacts. We then follow this with an evaluation of the impacts of human activities on species richness. Finally we discuss implications for management and propose a number of options to consider or further investigate that could contribute to forest conservation on Mt Elgon for both local and wider demands.

4.1. Activities and impacts on forest structure

Signs of human activities and their impacts remained visible over 2 km inside the park boundary, revealing a broader impact zone than the 900–1000 m estimated in Western Uganda by Olupot (2009). Not all signs of activity were equally prevalent in all sites or at all distances from the boundary, presumably reflecting varying local demands and practices, availability of the resource and concern to avoid detection. Some trails remained wide and well-worn far into the park. Such paths are important indicators of human activity, as was also observed on the Kenyan side of Mt Elgon, where the size of local trails was significantly correlated with stem and grass harvesting (Hitimana et al., 2010). The relationship of paths with human activities was also highlighted in a study in Kakamega, Kenya, where evidence of pole-cutting in sites where research and management trails provided access (Fashing et al., 2004).

Past episodes of cultivation within the park still affect forest structure gradients. BA and often stem densities were lower in formerly-cultivated than in old-growth forest areas, even where forest had been "recovering" for 35 years (Sites 2 and 9). In old-growth forest plots from our study we found comparable BA and
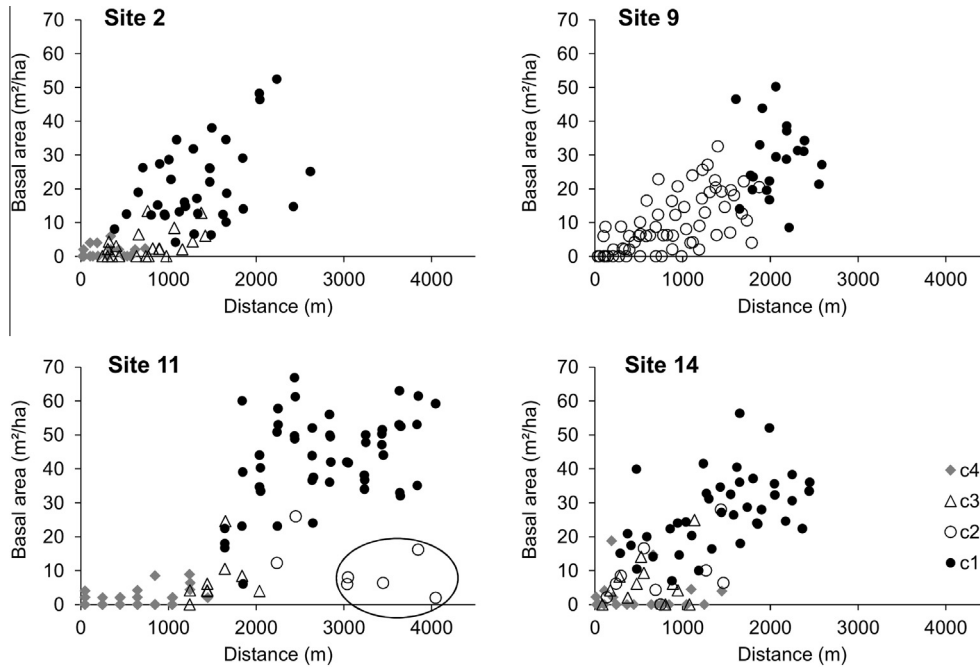
Fig. 6. Basal area of all plots and distance inside the boundary per study site, coded per encroachment category. c1 = never cultivated, c2 = cleared in the 1970s and 1980s now recovering, c3 = cleared in the 1990s and 2000s now recovering, c4 = currently cleared and cultivated (2011). In Site 2, c3 plots nearer the boundary were situated on top of a wide rocky cliff that is kept "open" by fire (to harvest ground honey). In Site 11, the encircled plots were in or on the edge of traditional grazing areas inside the forest. In Site 14, grasslands, regeneration and more intact forest form a patchwork up to 1500 m from the boundary.
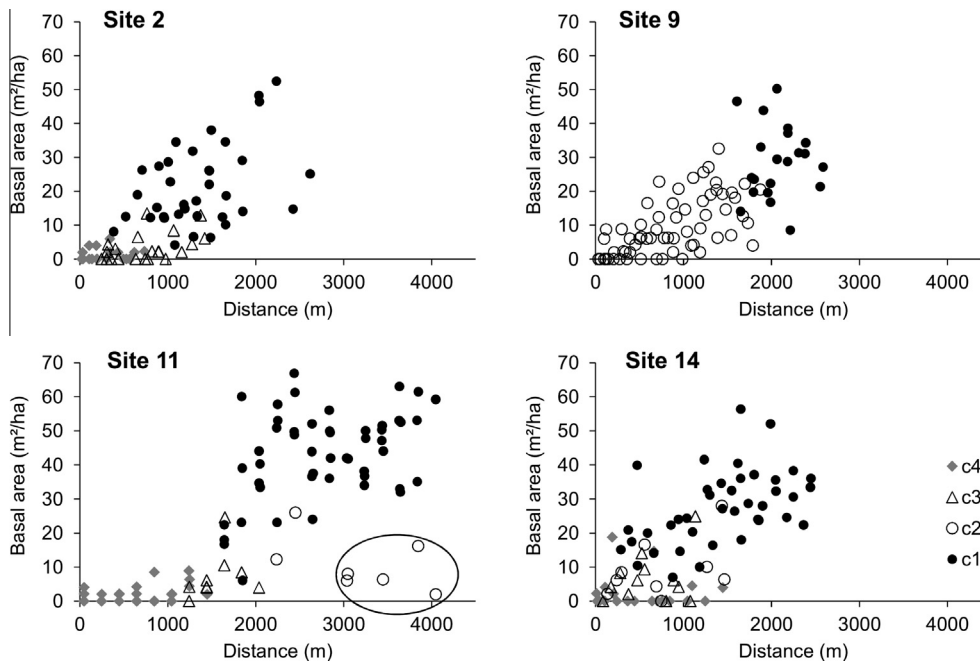
Fig. 7. Stem densities (stems ⩾ 5 cm) and distance inside the boundary per study site (detailed plots). Density of stems <20 cm and ⩾20 cm in diameter per plot, coded per encroachment category. The furthest plots in Site 11 were near or in traditional grazing areas.

stem densities to those found in previously selectively logged sites on the Kenyan side of Mt Elgon (Hitimana et al., 2004; Ongugo et al., 2008). Unfortunately there are no studies of recognisably "untouched" forest on Mt Elgon, if such sites exist, to compare with.

Structural diversity and patchiness is characteristic for montane forests in East Africa (Hamilton and Perrott, 1981). This means that

the high local variations in BA and stem densities that we found in our sites (as expressed in high standard deviations, see Table 4) could be natural. It is also plausible that the patterns seen by Hamilton and Perrott (1981) rather reflect extended human presence on most mountain forests rather than any natural pattern (Hamilton et al., 1986). In our sites, the known and inferred history of human presence, the extent and the intensity of the signs of human
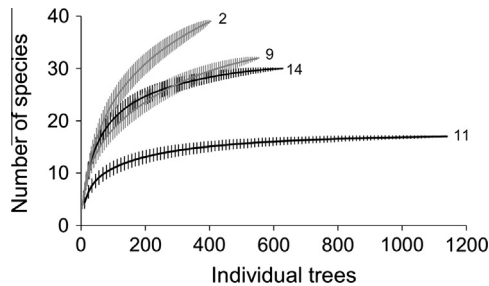
Fig. 8. Rarefied species accumulation curves per study site (in BiodiversityR, with 100 permutations) with standard deviation.

activity and the observed overall increase in BA (all stems) and stem density (stem size ≥ 20 cm dbh) with distance inside the boundary, suggest that human impacts played a major role.

Hamilton and Perrott (1981) also observed that, overall, tree density (stems ≥ 15 cm dbh) on Mt Elgon decreased up to about 2700 m elevation, which is the elevation spanned by our transects (except in Site 11 where they reach almost 2900 m). We found the opposite or no pattern in our study sites, which was likely related to previous disturbance: in all four sites trees with stem size ≥ 20 cm dbh were negatively affected by the concentration of previous clearing nearer to the park boundary. Trees on the lower rather than the higher slopes were therefore more affected. Logging as in Site 9 and charcoal burring as in Site 2 generally affect larger trees. Any such artificial reduction of large trees is a concern as large trees dominate forest structure and micro-climate, store considerable amounts of carbon and are important as habitat and for forest regeneration (Clark and Clark, 1996).

4.2. Site-specific impacts

Site-specific histories and resource use patterns can help us to interpret our results. Sites 2, 11 and 14 had areas in different stages of regeneration or degradation more patchily distributed. Therefore forest structure gradients - beyond strips of current cultivation – were less clearly related to distance inside the boundary in Sites 2, 11 and 14 than in Site 9 (Figs. 6 and 7).

In Site 9, the relationship between BA, tree density and distance inside the boundary (largest R^2, see also Figs. 6 and 7) reflected a gradual change from bushy regeneration to more advanced regrowth to fairly intact forest away from the boundary. The dominant crops–bananas, coffee and climbing beans – near this site require crop-stakes. The area closest to the boundary in Site 9 appears to be kept in an early succession state due to continuous harvesting of coppice shoots and small trees to meet the demand for small poles. Contrary to previous observations suggesting that local forest use was concentrated in the regenerating areas (Scott, 1998), our study revealed that impacts on regeneration and small stems also occurred in old-growth forest.

Resource use agreements include the monitoring of resource off-take by a local resource use committee (UWA, 2000). Yet, in Site 9, the incidence of split wood, the signs of cut stems, the numbers of stumps and lopped trees indicated that this was likely not entirely effective. Local informants in this site (Site 9) confirmed that the split fuelwood and crop-stakes for banana that we found drying at further distances in the forest were for commercial purposes (local informants and community conservation ranger, personal communication).

The smallest mean BA and stem sizes in old-growth and regenerating forest were found in Site 2 (Table 4). The patchy structural variation of the forest was clearly seen in the high standard deviations of stems densities and BA (see Table 4). There were many clearings in the forest, some were likely natural, but charcoal pits

were found in 16% of the plots while many others were encountered while moving between the plots (Fig. 1). Charcoal production removes larger trees than are taken for firewood or for poles (Girard, 2002) – this creates larger openings. The local importance of charcoal-burning in this area is consistent with previous findings (Scott, 1994). Conflicts between local people and park management were high in this area of the park (Site 2), which may have encouraged destructive behaviours and illegal activities (Sassen et al., 2013).

In Sites 11 and 14, cattle appeared the main control of regeneration. Cattle are important to the local economy is these sites (Table 1) and law enforcement has been unable to prevent grazing inside the park. In Site 11, forest structure data exhibited patterns typical for a grazed forest: open forest with little regeneration and the maintenance of grassy glades (Reed and Clokie, 2000). Grazing on either side of the forest edge in combination with tree-cutting for firewood appear the main forces keeping these areas open and eroding the forest edge.

In Site 14, the combination of fire and grazing hindered regeneration in formerly cultivated areas. Fire is also used to help harvest ground honey, an important forest product. However, we found fewer signs of grazing in old-growth forest (although sometimes burn scars) in Site 14 than in Site 11 which could help explain why regeneration seemed better in this site.

Aside from grazing and fire other signs of human use in Sites 11 and 14 were less prevalent than in Sites 2 and 9, possibly because of lower demands due to lower population densities. We note that the crops grown in Sites 11 and 14 (cereals and potatoes) do not require crop-stakes – thus limiting demand for small poles. However, the human population is growing fast (2.5–4.3% (UBOS, 2002a,b,d)) which may lead to increased demand for wood products such as poles for construction and firewood. People in Sites 11 and 14 had fewer trees planted on their land that could provide alternatives for wood resources from the park (see trees per household in Table 1).
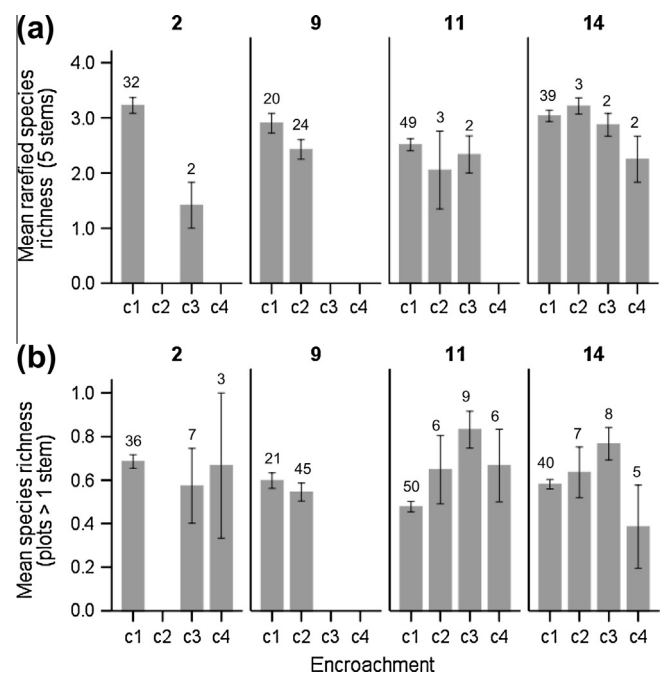

Fig. 9. Mean species richness (with ±1 standard error) per plot per encroachment category: c1 = never cultivated, c2 = cleared in the 1970s and 1980s now recovering, c3 = cleared in the 1990s and 2000s now recovering, c4 = currently cleared and cultivated. (a) rarefied species richness (samples of 5 stems) and (b) species richness in plots with more than 2 stems (Z = log(species count)/log(stem count)). n values are at the top of each bar.

4.3. Patterns of species richness

Our results suggest human impacts have affected tree species richness on Mt Elgon. We discuss our results in light of existing theories of natural patterns of species richness on mountains. Among our four study sites, and consistent with previous observations (Hamilton and Perrott, 1981; Rahbek, 1995), overall plot level tree species richness decreased with elevation. The highest mean number of tree species per plot was found in Site 2 and the lowest in Site 11 – the lowest and highest elevation sites respectively. Looking only at plots in old-growth forest Site 11 was also less diverse than the other sites (Fig. 9).

There are multiple theories concerning natural patterns of species richness on elevational gradients and their determinants (Ghazoul and Sheil, 2010). On mountains, area effects have been shown to affect species richness patterns (McCain, 2009; Romdal and Grytnes, 2007). Many studies indicate a so called "mid-domain effect" (MDE) in which species richness increases and then declines with increasing elevation and that can be seen as a natural consequence of species range patterns and elevation limits (Cardelús et al., 2006; Colwell and Hurtt, 1994). This theory has been challenged by empirical research (e.g. Kessler et al., 2011). The relative importance of different factors such as the MDE, climatic and topographic factors likely depends on climatic histories, taxa (Acharya et al., 2011; Grytnes and Beaman, 2006) and on spatial scale (Colwell et al., 2004).

As in our case, localised studies may only span a portion of the available elevation gradient. Our plots started above 1900 m and the highest elevation 2877 m was well below the treeline at around 3200 m. At our four sites, we sampled only a section of the forest-range so it is unremarkable that we did not see the humped-shaped elevational gradients of species richness either within the individual study sites or among them. Historical disturbances might in any case strongly influence these theoretical patterns. This indeed seems to be reflected by the complex site-specific patterns at each of the four study sites.

There has been much work on the impacts and influences of disturbance and disturbance regimes on the diversity of tropical forests. For example the intermediate disturbance hypothesis suggests that disturbance of old-growth forest can lead to enrichment through the addition of early successional species, but that excessive disturbance can lead to a decline (Sheil and Burslem, 2003).

We discuss how patterns of species richness were affected by human activity in each site and the evidence for the intermediate disturbance hypothesis, which varied among sites.

In Site 2 there were no plots that had been regenerating for more than 20 years (c2). Because only plots with at least 2 or 5 stems were included there were few plots in encroached areas (c3), which made it difficult to reveal clear patterns (Fig. 9). Z-species richness was negatively correlated with BA, which was strongly affected by intensive charcoal production in this site. Charcoal production led to many large openings at various distances into the park, which explains why we found no patterns with distance inside the boundary or elevation. In Site 9, there were no significant gradients. Selective harvesting may have affected species richness in the formerly encroached (c2) areas. The intermediate disturbance does not cover such selective processes (Site 2 and 9). In Site 9 replanting with a mix of native species will also have affected associated diversity patterns to an unknown degree.

We found evidence for the intermediate disturbance hypothesis In Site 11 and 14. In both sites plots in areas that were recovering or begin degraded (c2 and c3) were richer in species than old growth forest. In Site 11, rarefied species richness was slightly higher at higher elevations. This is because in this Site formerly

encroached (c2 and c3) plots with at least 5 stems were found on ridges further inside forest. Z-species richness was greater in areas with less BA, as most old-growth forest with large BA in this site was dominated by few species (*Cornus volkensii, Schefflera, volkensii, Hagenia abyssinica* and *Podocarpus Milianjianius*) (Sassen, unpublished results). A previous study in a site near Site 11 also found that formerly grazed or settled locations had greater species richness than old-growth forest (Reed and Clokie, 2000).

In Site 14 species richness decreased with elevation and distance inside the boundary, because there were no formerly encroached plots (c2 and c3) further into the forest as was the case in Site 11. In general terms it appeared that in these two Sites (11 and 14) human impacts, through fire, grazing and wood harvesting at intermediate levels (c2 and c3) often led to some enrichment, as might be predicted from the intermediate disturbance theory (Connell, 1978; Sheil and Burslem, 2003).

4.4. Implications for management

As is the case for many parks in the tropics, even if it were deemed ethical and necessary, it would be impossible to stop people from entering Mt Elgon National Park. Our results highlight various concerns. For instance, even in a site (Site 9) where local forest use is formally regulated and monitored, the intensity of pole harvesting raises concern over the sustainability of such activities. However, further research should investigate whether the coppicing ability of tree species that are illegally used for poles and crop-stakes (e.g. *Neoboutonia macrocalyx*) could be harnessed to provide a sustainable legal source of such materials. In addition, alternatives need to be found outside the forest. Bamboo-cultivation is currently being promoted in some areas around the park (community conservation ranger, personal communication; personal observation). Shade-trees in coffee can be further promoted as they have shown to benefit coffee production in sub-optimal smallholder systems such as those on Mt Elgon (DaMatta, 2004). Charcoal-burning (Site 2) is a commercial activity and unless brought under control will likely lead to continued forest degradation.

Grazing cattle inside the forest is traditionally important to the communities in the north and north-east. A strategy to avoid conflicts could include a system of periodic grazing, e.g. during the dry season when there is less fodder available outside the forest. Or a system where people restrict grazing to the open grassland inside the forest where this has a long history, without hampering regeneration in the surrounding forest (Reed and Clokie, 2000). A more in-depth study of the phenology and life history strategies of the plants in these glades may help development of a better understanding of the effects of grazing on the longer term.

Cause and effect of human activities on species richness on Mt Elgon were harder to determine than impacts on forest structure. But in all sites, species richness was affected by past and present disturbance (see also Huang et al., 2003). The effect of fire on the forest community on Mt Elgon needs to be studied in more detail, as some communities on the edges of grasslands may be fire-dependent (van Heist, 1994).

Approaches are required that balance conservation and local demands, and that can adapt interventions to local contexts. Mt Elgon has important values for local communities directly neighbouring the park and is also an important water-catchment area for more than a million people in the wider region. Most mountain forests have been able to support or recover from extended human influence over time (Taylor et al., 1999). Therefore, opportunities for creating and maintaining resource use areas in intermediary states of succession – balancing minimum conservation needs while meeting local needs – should be explored (see Hutton and

Leader-Williams, 2003). Particularly in areas with high population densities with strong claims on forest resources.

Despite some weaknesses, resource use agreements seemed to lead to better outcomes for forest conservation (see also Sassen et al., 2013). Developing capacity for collaborative management and monitoring, sharing responsibilities and rights between park management and local communities can help achieve better conservation outcomes, while taking into account local needs (Vermeulen and Sheil, 2007). Developments relating to payments to local communities for avoided deforestation or forest restoration, such as in the context of REDD+ are currently being developed on Mt Elgon as potential win–win solutions for conservation and local development. Practical implementation of such schemes have yet to prove successful in achieving better forest protection and benefits for local people.

Options depend on local needs and preferences and their impacts on conservation (biodiversity, hydrological, carbon) and other values. We can only have informed discussions about these complex trade-offs once we better understand them – and only if we begin to look more carefully at the diversity of local consequences.

Acknowledgements

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