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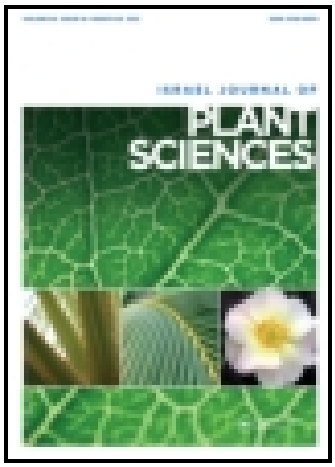

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REGENERATION OF AN AFROMONTANE FOREST FOLLOWING AGRICULTURAL ENCROACHMENT IN SOUTHWESTERN UGANDA

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REGENERATION OF AN AFROMONTANE FOREST FOLLOWING AGRICULTURAL ENCROACHMENT IN SOUTHWESTERN UGANDA

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ABSTRACT

A study of the regeneration of an Afromontane forest was carried out in Mgahinga Gorilla National Park (MGNP), southwestern Uganda, following agricultural encroachment in the last 50 years. The landscape was changed by terracing and removing the indigenous vegetation and replacing it with exotic tree species. This study also examined the soil nutrient status of the formerly encroached area in the park.

Species richness of indigenous trees was high in the formerly cultivated area. Twenty-six indigenous species were found in the formerly cultivated area, compared with 20 species in the natural forest and 12 species of indigenous trees found under exotic woodlots. There was a significant difference in species richness and density in the three habitat types. The natural forest supported the highest stem density (75%), and the lowest stem density (4%) was recorded under exotic woodlots. Seedlings (<2 cm diameter at breast height) accounted for the majority of juveniles in the three habitats. The natural forest had the highest density (24,625 seedlings/ha), and exotic woodlots supported the lowest stem density (1,350 seedlings/ha).

The level of regeneration in the formerly encroached area is influenced by the intensity of cultivation and soil nutrients. The advanced growth beneath the exotic woodlots, especially black wattle (*Acacia mearnsii*) and *Eucalyptus* sp. stands, is relatively impoverished. This condition beneath the exotic species suggests that a low diverse community of native species is able to exploit this environment.

INTRODUCTION

One of the major factors controlling the distribution of vegetation types and boundaries of forests is the effect of human interference such as grazing, burning, and cultivation (Grove, 1995). Radiocarbon dating suggests that in southwest Uganda, forest-clearing may have begun as early as 4,800 years ago (Hamilton et al., 1986; Cunningham et al., 1993). In the early 1900s, the tropical moist forests covered more than 6% of Uganda's land area (Butynski, 1984), but due to agricultural expansion, these forests have been reduced by more than half (Struhsaker, 1987; Butynski, 1988). FAO (1988) estimated that in the 1980s, there were 7,500 km^2 of closed forest in Uganda, and about 5,900 km^2 of these were tropical high forests and 1,500 km^2 montane catchment forests. However, the remaining forests are now widely separated from one another, forming ecological islands surrounded by other vegetation types (Hamilton, 1984).

The main cause of forest destruction is agricultural encroachment and increased demand for fuelwood (Struhsaker, 1987). Howard (1991) estimated that 12% of the forested land within Uganda's principal reserves had been affected by agricultural encroachment. The increased demand for agricultural land and forest products is attributed primarily to the high population growth rate of more than 3%, which has led to the doubling of population since 1960 (Hamilton, 1984). Extensive transformation of the Kigezi highlands landscape occurred since the 1900s as a result of natural

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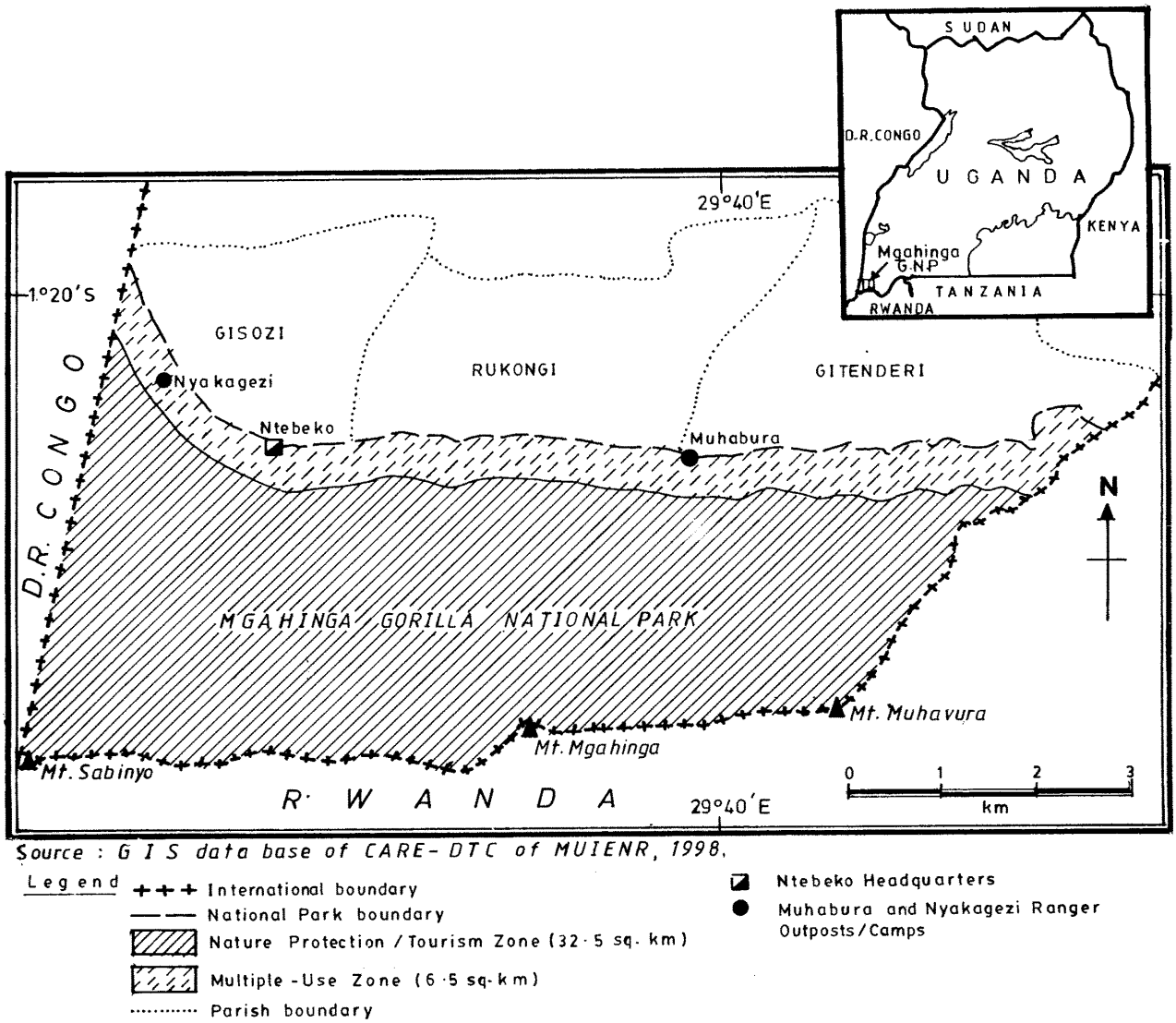

Fig.1. Location of Mgahinga Gorilla National Park and management zones.

population increase and migration from Rwanda (Cunningham et al., 1993).

Mgahinga Afromontane Forest, which is characterized by a great diversity of habitat types associated with its wide altitudinal range, was subjected to serious abuse and intensive illegal activities that led to forest degradation (Cunningham et al., 1993; Reynolds and Pomeroy, 1993). The forest reserve was cleared for agriculture and part of the degraded land was replaced by exotic plant species with the aim of reducing the degraded area (Kalina, 1993). However, in the early 1990s, the forest reserve was designated as a national park where the settlers were evicted and the area left to natural regeneration (Cunningham et al,. 1993). This study was therefore intended to determine the effect of agricultural encroachment on soil nutrient status and the extent of vegetation recovery in the formerly encroached upon areas.

STUDY AREA

Mgahinga Gorilla National Park (MGNP) is an Afromontane forest region situated in southwest Uganda (Fig. 1). The terrain ranges from gentle slope at lower elevation (2,227 m asl) to steep slope. Some very diverse, rugged slopes occur at high altitudes (4,127 m asl), and vertical cliffs and rock exposures are numerous at high elevations. The three volcanoes of MGNP are part of the Virunga Volcanic Range in East and Central Africa, which extends to the Albertine Rift on the border of Congo, Rwanda, and Uganda. Surface soils at MGNP are classified as Sabinyo complex (Harrop, 1960) with some zonation according to altitude (Kingston, 1967). The soils are all derived from volcanic activity and reflect the relatively short time between the most recent eruption and the present.

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MGNP is characterized by a great diversity of habitat types associated with its wide altitudinal range. The vegetation comprises both montane and afroalpine flora (Werikhe, 1991). The montane forest belt consists of the montane woodland zones, bamboo zone, and *Hagenia–Hypericum* zone. The montane woodland zone is the lowest zone of primary vegetation. It consists of open forest with dense ground flora, herbs and vines. The bamboo zone extends into a belt between 2,500 and 2,800 m asl. The subalpine/ericaceous belt comprises ericaceous and moorland zones. The ericaceous zone is dominated by *Phillipia johnstonii* (tree heath) and the subalpine belt is characterized by giant *Senecio* and *Lobelia*, which occurs above the ericaceous belt (Kalina, 1991).

MGNP is also a unique area for biodiversity and endemism in Africa. It is a part of the Albertine Rift Afromontane region which is currently believed to hold the richest montane fauna in Africa (Kingdon, 1990). The area is a habitat for the rare mountain gorilla (*Gorilla gorilla beringei*) and the rare golden monkey (*Cercopithecus mitis kanditi*), known only to occur in the Virungas and two other forests in central Africa (Kingdon, 1971). The park also supports a unique avifauna. Seventy-nine bird species have been recorded within the park including several species endemic to the East Congo montane region (Kalina, 1993). Grauer's rush warbler (*Bradypterus graueri*), listed in the IUCN/ICBP Red Data book as vulnerable to extinction, occurs in MGNP (UNP, 1996).

MATERIALS AND METHODS

VEGETATION SAMPLING

Transect lines were established in the exotic woodlots, degraded area, and less disturbed natural forest habitat. Four transect lines, each 1,000 m long, were established in the natural forest and twelve were established in the formerly cultivated area. The transect lines established in the formerly cultivated area ran from one edge to the other because the width of the formerly cultivated area was narrow in some parts, ranging from 700 m to 1300 m. The transects were positioned using a compass and marked with stakes and flagging tapes.

Systematic sampling which involved the location of sampling points at regular intervals was employed (Kent and Coker, 1996). In each transect, a series of 20 m × 25 m (0.05 ha) plots were established at regular intervals of 50 m. The sample plots were positioned on alternating sides of the transect line as in Kasenene, 1987.

A series of nested quadrats were established within the 20 m × 25 m plots. The size-classes of tree species that occupy different vegetation strata within the habitat were identified and enumerated. All larger and smaller trees with a diameter range of ≥ 15 cm dbh and 10 – <15cm dbh (dbh = diameter at breast height) were measured at 1.3 m height, identified, and enumerated. Poles with diameter range of 5 – <10 cm dbh were enumerated in a 10 m × 15 m quadrat. Tree saplings with diameter range of 2 – <5 cm dbh were sampled in a nested quadrat size of 5 m × 10 m, whereas tree seedlings with <2 cm dbh were recorded in a quadrat size of 2 m × 4 m. Tree seedlings in 2 m × 4 m plots were carefully searched in the understory vegetation and counted. The data were subjected to statistical analysis using Shannon diversity index and ANOVA.

SOIL SAMPLING

Within the exotic woodlots, degraded area, and natural forest, ten plots of 10 m × 10 m were randomly established. In the natural forest the plots were established along the transects used in earlier vegetation sampling. In the degraded area, the plots were also randomly established at 20 m away from each exotic woodlot. Three subplots were randomly located in each plot. Composite soil samples were systematically excavated from each quadrat to a depth of 20 cm, using a shovel as employed by Soedrason and Kuswata (1991). The soil samples were put in polythene bags and were later sundried to halt biological transformation. The dry soil samples were sieved through a 2-mm screen and subsamples from the same plots were combined. The samples were then packed in plastic bags and transported to the soil testing facilities at the Department of Soil Science, Faculty of Agriculture, Makerere University. Soil chemical analyses of pH, organic content, and exchangeable cations were carried out. Analyses of the nutrients were all based on methods of Okalebo et al. (1993). Data were analyzed by Mstat-c computer package.

RESULTS

SOIL CHEMICAL CHARACTERISTICS

Table 1 shows the mean values of the nutrients for the three habitat types. The mean pH values of these soils ranged from 4.8 in the natural forest to 5.2 in the formerly cultivated area. The mean pH values of all the three habitats were less than 5.5, an indication that these soils were strongly acidic. Generally, the mean pH values were not significantly different in the three habitat types ($F = 3.39$; $df = 2, 18$; $p > 0.05$, ANOVA). However, there was significant variation between the mean pH values of natural forest compared to that of degraded

Table 1
Soil nutrient concentrations (Mean values +S.E.) recorded in natural forest, formerly cultivated area, and exotic woodlots

Habitat type	Soil pH	% Org. matter	P ppm	Ca meq/100 g	K meq/100 g	Mg meq/100 g	Na meq/100 g	% N
Exotic woodlots	5.1 ± 00a	18.5 ± 1.4b	5.3 ± 0.7b	8.3 ± 1.7b	2.8 ± 0.4a	2.7 ± 0.4a	0.2 ± 0.0a	1.1 ± 0.1b
Natural forest	4.8 ± 0.9b	26.1 ± 1.4a	14.8 ± 3.6a	12.2 ± 2.2a	3.4 ± 0.6a	3.3 ± 0.6a	0.2 ± 0.0b	1.4 ± 0.1a
Formerly cultivated area	5.2 ± 0.1a	21.3 ± 1.7b	5.7 ± 0.6b	8.7 ± 1.7b	2.1 ± 0.4b	2.7 ± 0.5a	0.2 ± 0.0ab	1.1 ± 0.0b
Significance	ns	s	s	ns	ns	ns	ns	s
P*	**	*	*	**	**	**	**	*
LSD ($p < 0.05$)	0.30	3.33	6.07	ns	1.13	ns	0.03	0.13
CV (%)	6.4	16.1	75.4	46.1	44.0	37.4	19.2	11.9

Comparison was by One Factor Analysis of Variance. * = $p < 0.05$, ** = $p > 0.05$, LSD = Least Significant Difference, CV = Coefficient of Variation, s = significant, ns = not significant, a, b, and ab area ranks, Means with same ranks are not significantly different.

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area ($p < 0.05$, LSD). No significant difference was found between exotic woodlots and natural forest or formerly cultivated area ($p > 0.05$, LSD).

The mean soil organic content ranged from 18.5% in exotic woodlots to 26.1% in natural forest. These mean values of organic content were more than three times that of an average soil of 5% (Brook, 1983). Analysis of variance showed that there was a highly significant variation in organic content among the three habitats (F = 11.68; df = 2, 18; $p < 0.01$, ANOVA). However, there was no significant difference between the mean values in formerly cultivated area and exotic woodlots ($p > 0.05$, LSD).

The available phosphorus ranged from 5.3 ppm in exotic woodlots to 14.8 ppm in natural forest (Table 1). Two-way analysis of variance showed a highly significant variation in mean values of phosphorus in the three habitat types (F = 6.89; df = 2, 18; $p < 0.01$, ANOVA). However, there was no significant difference between the mean values of the formerly cultivated area and woodlots ($p > 0.05$, LSD).

Total nitrogen content ranged from 1.1% in the woodlots to 1.4% in the natural forest. These values are higher than average values reported (0.2%–0.5%) (Brook, 1983), and are indicators of a very rich soil. Analysis of variance showed a very highly significant variation of soil nitrogen in the three habitat types ($p < 0.001$, ANOVA). Ca, K, and Mg had highest values in natural forest, except for sodium which had the same mean values of 0.2 meq/100 g for the three habitat sites (Table 1). However, there was no significant difference in the mean values of calcium and magnesium in the three habitat sites ($p > 0.05$, LSD), but significant difference was observed for the mean values of potassium and sodium recorded in natural forest and woodlots ($p < 0.05$, LSD).

DIVERSITY OF INDIGENOUS TREE SPECIES

Diversity measures were calculated for all three habitat types. Table 2 shows diversity indices of all size-classes recorded in the three habitat types. Diversity index of seedlings (<2 cm dbh) ranged from 1.24 in exotic woodlots to 2.13 in the degraded area. Similarly, in saplings (2 – <5 cm dbh), the highest diversity index was recorded in formerly cultivated area and the lowest was observed in exotic woodlots ($p < 0.05$, ANOVA). However, highest diversity indices for smaller and larger tree size-classes were recorded in formerly cultivated area and the lowest in exotic woodlots ($p < 0.05$, ANOVA). Generally, there was significant variation in species diversity among habitat types (F = 6.50; df = 2, 8; $p < 0.05$, ANOVA).

Species diversity indices within each habitat type ranged from 0.99 for poles to 1.47 for saplings in exotic woodlots (Table 2). In formerly cultivated area, diversity indices ranged from 1.28 for smaller tree size-class to 2.46 for saplings. In natural forest, diversity indices ranged from 1.41 for seedlings to 2.34 for poles. Generally, diversity indices in each habitat type increased from seedlings to poles, but the trend declined in smaller and larger tree size-classes. However, this variation was not significantly different (F = 1.56, df = 4, 8; $p > 0.05$).

Species evenness was generally high in the three habitat types, indicating higher chances of occurrence of the species in each habitat. From the observations, equability/evenness of seedlings ranged from 0.57 in natural forest to 0.77 in formerly cultivated area (Table 2). In the sapling stage, evenness ranged from 0.73 in natural forest to 0.92 in exotic woodlots. However, the higher size-classes were more evenly distributed in exotic woodlots, compared to formerly cultivated area and natural forest. Within each habitat, evenness ranged from 0.69 for seedlings to 1.72 for smaller tree size-

Table 2
Diversity indices and evenness of tree species of different size-classes recorded in natural forest, formerly cultivated area (old farmland), and exotic woodlots

Habitat type	Diversity indices	Diameter size-classes (diameter at breast height, dbh)				
		<2 cm	2 – <5 cm	5 – <10 cm	10 – <15 cm	≥15 cm
Woodlot	r	12	5	3	2	3
	H^1	1.24	1.47	0.99	1.19	1.00
	E	0.69	0.92	0.91	1.72	0.91
Formerly cultivated area	r	16	22	22	18	11
	H^1	2.13	2.46	2.46	2.32	1.37
	E	0.77	0.80	0.80	0.83	0.57
Natural forest	r	12	18	18	14	11
	H^1	1.41	2.10	2.10	2.23	1.61
	E	0.57	0.73	0.73	0.84	0.67

r = species richness, H^1 = Shannon diversity index, E = species evenness.

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classes observed in exotic woodlots. While in natural forest, the range was from 0.57 in larger tree size-class to 0.80 for poles. In formerly cultivated area, seedlings had the least evenness and smaller trees had the highest evenness.

VEGETATION RECRUITMENT

Figure 2 shows the distribution of log density of different size-classes observed in each habitat type. From the observations, higher densities of seedlings than saplings were recorded in each habitat type. This ranged from 1,350 seedlings/ha in exotic woodlots to 24,625 seedlings/ha in natural forest. This variation was highly significant ($F = 74.79$, $df = 2, 12$; $p < 0.01$, ANOVA). Similarly, the highest recruitments of other size-classes were recorded in formerly cultivated area and the lowest densities in exotic woodlots. These variations in plant recruitment were also significantly different ($p < 0.05$, ANOVA). All three habitats showed a sharp decline in number of seedlings surviving to the smaller tree size-class with a slight increase in stem density in the larger tree size-class ($p < 0.05$, ANOVA).

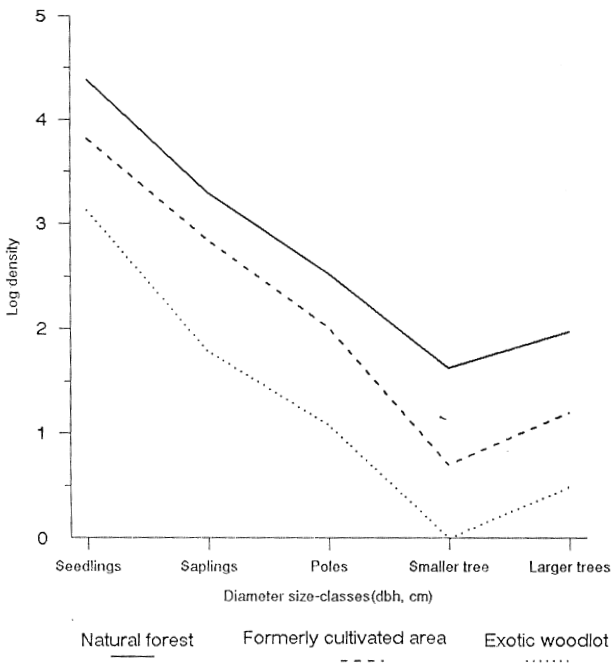

Fig. 2. Density stand curves for the various size-classes recorded in the natural forest, formerly cultivated area, and in exotic woodlots.

REGENERATION OF NATIVE TREE SPECIES

Table 3 shows the density, frequency, and dominance of canopy tree species recorded in the natural forest types. Most of the canopy species were well represented in formerly cultivated area; only *Lepidotrichilia volkensii* was not observed in this area. However, *Lepidotrichilia volkensii* was only recorded in natural forest. *Hypericum revolutum, Bersama abysinica, Nuxia congesta, Agaurea salicifolia,* and *Myrica salicifolia* do occur in all three habitats. Among these canopy tree species, *Hypericum revolutum* had the highest density of recruitment (2,220 stems/ha) recorded in formerly cultivated area ($p < 0.05$, ANOVA). This was followed by *Erica arborea*, with a density of 1,025 stems/ha.

Among the species recorded in natural forest, *Xymalos monospora* had the highest frequency of occurrence (137%). High frequencies of *Hypericum revolutum* and *Neoboutonia macrocalyx* were also recorded in natural forest. *Nuxia congesta* was the most dominant species recorded in both natural forest and formerly cultivated area with a value of 2.14 and 0.05,

Table 3
Density, frequency, and dominance of canopy tree species recorded in natural forest, formerly cultivated area, and exotic woodlot

	Density (No./ha)			% Frequency			Dominance (m^2/ha)		
Tree species	A	B	C	A	B	C	A	B	C
Hypericum revolutum	76	2220	644	104	33	4	0.03	0.31	0.80
Bersama abyssinica	188	167	0	40	9	2	0.26	0.07	0.02
Maesa lanceolata	43	167	0	58	12	0	0.35	0.07	0.00
Nuxia congesta	62	567	1	13	40	6	2.14	0.38	0.05
Ilex mitis	398	36	0	33	1	0	0.15	0.01	0.00
Agauria salicifolia	14	626	12	13	10	2	0.39	0.02	0.01
Dombeya goetzenii	0	75	26	0	1	0	0.00	0.01	0.00
Erica arborea	0	1025	12	0	28	4	0.00	0.17	0.01
Myrica salicifolia	0	708	26	35	16	12	0.00	0.15	0.02
Xymalos monospora	881	5	0	137	3	0	0.78	0.02	0.00
Neoboutonia macrocalyx	43	5	0	35	2	0	0.26	0.01	0.00
Dombeya kirkii	2	17	0	0	1	0	0.02	0.03	0.00
Hagenia abyssinica	0	5	6	0	3	6	0.00	0.01	0.03
Faurea saligna	0	0	0	0	2	0	0.00	0.01	0.00
Lepidotrichilia volkensii	453	0	0	108	0	0	0.43	0.00	0.00
Species richness	10	14	7	11	14	8	10	14	8
Total	2,160	5,623	727	576	161	36	4.81	1.27	0.94
Mean	216	433	104	38	12	5	0.48	0.09	0.12
Standard deviation	283	634	238	43	13	3	0.62	0.12	0.28
Standard error	90	176	90	11	4	1	0.20	0.03	0.10

Habitats: A = natural forest, B = formerly cultivated area, C = exotic woodlot.

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respectively, while *Hypericum revolutum* was the most dominant species observed in exotic woodlots.

DISCUSSION

Mgahinga Forest was greatly affected by human influence and agricultural clearing after mass settlement in the 1950s (Cunningham et al., 1993). This influence totally transformed the landscape of the formerly encroached area through terracing, and removal as well as replacement of most woody vegetation by crops and exotic tree species such as *Acacia mearnsii* and *Eucalyptus* sp. The levels of vegetation regeneration and patterns of distribution of exotic tree species in Mgahinga suggest the intensity of cultivation and effects of these plant species in the now protected area.

Results from this study indicate higher pH value in soils in exotic woodlots compared to that recorded in natural forest. This contradicts results obtained in studies carried out in India and South Africa (Poore and Fries, 1985). This therefore suggests that the lower pH value observed in the natural forest could be due to the presence of high organic matter recorded in this study site. Peat soil found in this area was encouraged by conditions of very poor soil drainage. This is due to the effect of relief upon the climate, which is well known for its depression of temperature and increase in rainfall (Brook, 1983). This leads to conditions favoring the development of thick organic horizons as a result of low temperatures retarding biological activities in the breakdown of organic matter. The significantly lower values of organic matter observed in the formerly cultivated area suggests high incidences of past human activities which occurred in this area. Forest clearing exposes the land to solar radiation, resulting in increased temperatures and rapid decomposition of organic matter (Sawyer, 1993).

Phosphorus content of the soil has often been used as a soil degradation index. Available phosphorus in topsoil decreases with increasing intensity of soil formation. This is generally true according to the results obtained in this study. Low mean values of phosphorus recorded in the exotic woodlots and formerly cultivated area indicate high intensity of past land use in this area. Bray method of phosphorus determination (Moukam and Nyakanou, 1997) assumes values greater than 35 ppm as high, between 15 and 35 ppm as medium, and those lower than 15 ppm as low. The intensive use of land in this area could be the cause of the extremely low values of phosphorus in these soils. Results obtained for total nitrogen were very high based on the ratings of Brook (1983). Brook (1983) also considers total

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nitrogen values to be low when below 0.2%, medium when between 0.2 and 0.5%, and high above 0.5%. So the high values of total nitrogen obtained in MGNP indicates a very rich soil.

Species richness refers simply to the number of species in a given area of a habitat or number of species found in a given community (Campbell, 1988). Species richness is a component of species diversity. The significantly high value of species richness recorded in the formerly cultivated area is an indication of a high level of forest regeneration following disturbance. Complete clearance of the forest in formerly cultivated area led to a change of microclimate, thus encouraging the colonization of the habitat by shade-intolerant species, leading to high species diversity. However, the lowest species richness recorded in exotic woodlots was due to the effect of suppression by exotic species. This agrees with results obtained by Poore and Fries (1985) in exotic plantations in Malawi. Some exotics such as *Eucalyptus* are known to suppress the growth of native trees (Evans, 1996), hence the reason for low species richness observed in exotic plantations.

Species diversity is a characteristic that pertains to a community level of organization and is an expression of community structure (Krebs, 1972; Brower and Zar, 1984). High species diversity reflects a highly complex community because the greater variety of species permits a larger array of species interactions and hence community stability. The highest diversity of seedlings and saplings recorded in the formerly cultivated area is an indication of vegetation recovery following intensive human disturbance. The regeneration of indigenous species in exotic woodlots was generally low compared to that recorded in the formerly cultivated area and natural forest. This agrees with the findings of Evans (1992) and Poore and Fries (1985). Evans (1992) stated that exotic species usually cause competition for nutrients and water that influence indigenous plant species. Similarly, human disturbance results in replacement of natural forest community with exotic plantations or woodlots that are usually poorer in species and contain different species than the natural forest they replace (Poore and Fries, 1985). The canopy cover of the exotic woodlots observed ranged from patchy to slightly dense. This was because most of the poles had been removed, leaving a patchy distribution of the trees. The patchy canopy supported a dense ground cover of grasses with some tree species, while the slightly dense canopy suppressed undergrowth of vegetation with almost no tree species. The colonizing characteristics of exotic species have on occasion enabled them to compete successfully against native vegetation for resources (Sawyer, 1993). This has been observed in South Africa and New Zealand, where exotic pines are suppressing indigenous forest species (Zobel et al., 1987). Large-scale disturbance due to agricultural practice in the formerly cultivated area encouraged the regeneration of secondary tree species which demand light for regrowth. A large proportion of the degraded area is, however, still covered by grasses and herbaceous plants.

The assessment of the size-class distribution is one of the central themes of plant community ecology. It provides knowledge about the population structure and function of a given population (Tsingalia, 1982). From the results of this study, heavy recruitment of juveniles into the population was obvious. However, the survival rate of these size classes into the larger size classes was low, leading to logarithmic decline in tree numbers with increased size. This formed a typical reverse J-shaped curve, which, according to Campbell (1988), indicates a growing cohort of trees that may have experienced a past disturbance. High densities of seedlings in all the habitat types highlight the importance of the presence of propagules in determining the composition of early successional communities and, indeed, their establishment. This also indicates the recovery potential of each habitat type. For instance, the natural forest had the highest potential of recovery, probably attributable to the fact that the forest had some patchy gap distribution created by dead trees.

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