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Forest Ecology and Management 173 (2003) 235–250

Forest Ecology
and
Management

www.elsevier.com/locate/foreco

Consequences of plantation harvest during tropical forest restoration in Uganda

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Received 1 June 2001; accepted 26 December 2001

Abstract

Timber plantations have recently received considerable attention as a forest restoration strategy for heavily degraded lands in the humid tropics. Plantations can facilitate secondary forest regrowth by providing an understory environment more favorable for native plant recruitment than unmanaged degraded habitats. To better understand secondary forest development and to evaluate plantation use as a restoration strategy, we studied natural forest succession after plantation harvest in Kibale National Park, Uganda. We asked four questions concerning using plantations as a restoration tool. How does timber harvest affect forest succession? How does initial on-site recruit availability (e.g., seeds, seedlings) after logging influence successional pathways? How easily can forest succession be enhanced through intervention? How does using exotic timber plantations to restore forests compare with fire exclusion as a restoration strategy? Timber harvesting killed or severely damaged many native stems, hindering subsequent forest regrowth. Despite this setback, native stem densities 4–6 years after logging equaled or surpassed native stem densities in unlogged plantations, suggesting timber removal accelerated forest succession. Successional habitats with high and low initial densities of on-site recruits first diverged in forest structure and composition, but then converged for many of these variables within 6 years of logging. Intervening to accelerate forest succession met with mixed results. Removing non-tree vegetation did not enhance tree establishment, growth, or survival after 2 years. However, leaving standing, dead timber trees as perches for seed-dispersing birds seemed to increase seedling establishment relative to control areas. Mortality and growth of seedlings planted into successional habitats 1–2 and 5–6 years after logging were similar, and predicting individual species responses based on successional status was unsuccessful. We compared succession in unlogged and logged (5–6 years after logging) plantations to a similar aged site where fires were excluded but no plantation species established. Our results suggest excluding fire is a better strategy for promoting forest succession than establishing then not harvesting plantations. Fire exclusion versus establishing then harvesting timber are comparable restoration strategies differentially enhancing tree sapling recruitment and growth, respectively. While forest regeneration was successful where fire was excluded long-term fire exclusion may be difficult, and several non-ecological challenges to using plantations exist (e.g., the conflict of managing for biodiversity versus timber production). In summary, our research suggests managers should carefully weigh the risks of using plantations or fire exclusion against other forest restoration strategies.

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Keywords: Arrested succession; Exotic tree plantations; Reforestation; Regeneration; Restoration ecology; Secondary forests; Tropical forest succession

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1. Introduction

Forest regrowth is often slow or absent on heavily degraded lands in the tropics (Lamb, 1998). However, restored forests on these lands can be useful for conservation, timber production, carbon-dioxide sequestration and can provide other ecosystem services (Brown and Lugo, 1990, 1994). Establishing timber plantations on degraded lands can facilitate forest succession by providing an understory environment favorable for native plant recruitment (Chapman and Chapman, 1996; Lugo, 1997; Parrotta et al., 1997). An additional benefit of plantations is that timber sales can help pay for the restoration. While plantations have attracted the attention of many tropical restoration ecologists, there has been little investigation of how best to use plantations in restoration (though see Ashton et al., 1998), how plantation harvest affects subsequent forest regrowth, or how using plantations compares to alternative restoration strategies. Furthermore, our understanding of forest succession is poor (Chazdon, 1994), thus limiting our ability to manage secondary forests after plantation harvest.

We studied forest regrowth after the harvesting of exotic timber plantations in Kibale National Park, Uganda. These plantations were planted in the 1950–1960s on previously forested lands that had been used by agriculturalists, but had been abandoned and were dominated by grasses (especially *Pennisetum purpureum*) (Osmaston, 1959). Active fire exclusion was important to protecting young plantations, but became less necessary as plantations matured. As plantations matured, native trees established in their understory and were not removed by managers (Chapman and Chapman, 1996; Fimbel and Fimbel, 1996). After logging, surviving native stems, the soil seed bank, and seed dispersal contributed to secondary forest growth. In contrast, similar areas not planted with exotics are still fire-maintained grassland, confirming plantations can facilitate forest regrowth on degraded lands (Duncan and Duncan, 2000; Zanne and Chapman, 2001).

Two successional trajectories emerge from plantation logging at Kibale. As a result of differential native stem recruitment before harvesting, logged pine plantations (*Pinus caribae* and *P. patula*) have greater initial on-site recruit densities (e.g., more native stems) than do logged cypress (*Cupressus lusitanica*)

plantations harvest (Fimbel and Fimbel, 1996). Logged pine plantations are dominated by early- and mid-successional trees, while logged cypress plantations are dominated by early- and mid-successional shrubs and trees. The juxtaposition of disturbed habitats with varying initial recruit availability is common among degraded lands in the tropics (Uhl et al., 1982, 1988a; Adedeji, 1984; De Rouw, 1994; Garciamontiel and Scatena, 1994; Fernandes and Sanford, 1995; Gillespie et al., 2000). Typically, more heavily degraded sites start succession with lower recruit availability than do less degraded sites (Brown and Lugo, 1990, 1994). Understanding how forest succession proceeds given different levels of initial recruit availability has not been well studied. However, such knowledge would assist managers choosing among disturbance types for investing restoration efforts. Or, this knowledge could be useful for managers predicting when successional forests will provide extractive resources or adequate habitat for species of interest.

We investigated four questions relevant to managing plantations for restoring forests and managing successional forests. First, how does initial on-site recruit availability influence subsequent successional trajectories? We compared the first 6 years of succession between logged cypress and pine plantations. Second, how does timber harvesting affect ensuing forest successions? We investigated how long a forest takes to regain the structure and composition present in the plantation understory before harvest, the damage inflicted on native stems during harvest, the effects of felling versus felling and timber removal, and the effect of variation in disturbance intensity within and among logged plantations. Third, how easily can forest succession be enhanced through intervention? In particular, we examined whether removal of all non-tree vegetation enhanced tree recruitment and growth, the success of planting tree and shrub seedlings into successional habitats of two ages, and leaving dead remnant trees in successional habitats to enhance seedling recruitment by attracting seed-dispersing birds. Fourth, how does using exotic timber plantations to restore forests compare with fire exclusion as a restoration strategy? Recurring fire is a severe impediment to forest succession on many degraded lands, especially grasslands (Uhl et al., 1988b; Uhl and Kauffman, 1990). We contrasted forest succession among logged and unlogged plantations and a former grassland where

fire was excluded since the time of plantation establishment. Finally, we evaluate pros and cons of using timber plantations to restore forests in light of our findings.

2. Methods

2.1. Study site

Located on the foothills of the Ruwenzori Mountains, Kibale National Park, Uganda, is dominated by moist-evergreen forest. Average elevation is 1500 m, and average rainfall is 1543 mm yr⁻¹ (1903–1999), though it has recently increased to 1765 mm yr⁻¹ (1991–1999). Softwood plantations in the park are surrounded by natural forest (plantation interiors are <250 m from natural forest) and have been described by Chapman and Chapman (1996), Fimbel and Fimbel (1996), and Zanne and Chapman (2001). Plantation logging began in 1993 when Kibale became a national park, and current management is allowing natural forests to grow in their place. From 1998 to 2000 we studied forest succession among plantations logged between 1993 and 2000.

Mature cypress plantations have lower mean basal areas than do pine plantations (44 m ha⁻¹ versus 71 m ha⁻¹, respectively), shorter canopies than pine plantations (23 m versus 33 m, respectively), but similar mean DBHs relative to pine plantations (36 cm for both; Fimbel and Fimbel, 1996). Timber is felled with chainsaws, and this results in many native stems being killed or damaged. Felled timber is rolled or winched to nearby portable sawmills, pitsawing stations, or roadsides. During this movement, more native stems are killed or damaged. When harvest is completed, few stems >1 m tall remain. However, many native seedlings survive, and root sprouts and coppice from native species are common. There is no regeneration of plantation species. Because unlogged pine plantations have more native stems than unlogged cypress plantations, hereafter, we refer to them as “high density” and “low density” plantations, respectively.

2.2. Influence of initial recruit availability

To assess how initial recruit availability influences succession, we quantified how vegetation differed

between the only remaining unlogged low density plantation and an adjacent unlogged high density plantation (thus attempting to control for site history and landscape position). Within each plantation, 15 plots were positioned by randomly choosing distances along a 100 m transect, then randomly choosing a distance of 1–60 m perpendicular to the transect. Plots were non-overlapping and within 100 m of natural forest edge. The openness of plantation understories allowed use of 25 m² circular plots, which were faster to survey than rectangular plots used in logged plantations. In these and all plots surveyed in logged plantations, we measured density, species richness, and height of tree stems ≥1.0 m tall. These stems were termed “saplings”, though a few early-successional trees were mature. Smaller (ht ≤ 0.25 m) and intermediate stems (ht = 0.26–0.99 m) were described in 4 m² subplots. We termed these smaller stems “seedlings” and intermediate stems as “large seedlings” though stems of both groups likely were no longer dependent on seed reserves or may have been sprouts. Plant identification was based on manuals by Eggeling and Dale (1952), Polhill (1952), Hamilton (1991) and Katende et al. (1995).

We established long-term plots in a low density and high density plantation within a year after logging (~9 and ~4 months, respectively). These were the only similarly aged recently logged high and low density plantations available. Thirty plots (5 m × 5 m) in each logged plantation were placed randomly (non-overlapping) along parallel transects (10 m apart) covering the logged area. All plots were within 100 m of natural forest edge. Plots were marked, surveyed, and surveyed again after 2 years; one plot in the logged high density plantation was destroyed by human activity before the second survey. During both surveys, vegetation was quantified as done in unlogged plantations. Some plots in the logged plantations were heavily disturbed areas (pit-sawing stations, portable sawmill locations and roads) where soil and vegetation were more disturbed than in moderately disturbed plots where the main activity was timber removal. Heavily disturbed plots were excluded from the present analyses, but are considered below. Many stems (23%) classified as seedlings in the first year after logging were actually root sprouts or coppice from stems buried during logging (Duncan, unpublished data).

Finally, we surveyed the oldest successional habitats in logged plantations at Kibale (4–6 years after logging). Vegetation plots were surveyed in two low density plantations, one logged 4 years before ($n = 30$ plots) and another logged 5 years before ($n = 28$ plots). Thirty plots were surveyed in a high density plantation logged 5–6 years before. In these plantations, plot positioning and sampling followed that of long-term plots. We used $2\text{ m} \times 10\text{ m}$ plots with 4 m^2 subplots for smaller stem to ease sampling of the dense vegetation. Plots from the two logged low density plantations were combined for comparisons to plots in the logged high density plantation.

We used non-parametric analyses for these and most comparisons because data were often highly zero-skewed. Consequently, we present medians to describe data, but also provide means to aid comparisons with other studies where means are reported.

2.3. Harvesting effects on forest succession

2.3.1. Native vegetation before and after timber harvest

From the surveys described above, we compared native vegetation present before logging to that at <1 and 4–6 years after logging in moderately disturbed plots. High and low density plantations were analyzed separately to control for differences in initial recruit availability.

2.3.2. Stem damage from logging

The process of felling and removing timber can severely damage native vegetation. During plot sampling in recently logged plantations, we classified surviving native stems into four categories: (1) undamaged, (2) bent stems pinned to or near the ground by felled trees, (3) broken stems broken near their base and (4) cut stems cut at their base. Some stems fell in multiple damage categories. We pooled all plots (heavily and moderately disturbed, high and low density plantations) and compared numbers of damaged stems to undamaged stems.

2.4. Damage from felling

When plantations are managed to promote natural forest regrowth, managers may want to develop harvest methods reducing damage to native stems and

encouraging regeneration. Thus, it is important to know how damage during felling compares to that during timber removal. For reasons unknown to us, one section of high density plantation (0.4 ha) was felled and the timber left in place. We sampled this area 3 years after timber was felled (felled-only plots). Plot positioning and sampling followed that in logged plantations. Plots ($n = 20$) were $2\text{ m} \times 10\text{ m}$, with 4 m^2 subplots for sampling stems <1 m tall. We compared this habitat to the high density plantation logged 5–6 years previously (only moderately disturbed plots), the most similarly aged logged high density plantation.

2.4.1. Within-site disturbance intensity

Damage to native stems during logging varies spatially, and depends on intensity of harvest activity (e.g., portable sawmill locations versus adjacent logged areas). We pooled plots from the two recently logged plantations and compared stems from plots on moderately ($n = 49$) and heavily disturbed ($n = 10$) sites.

We also compared effects on forest succession of two timber processing methods used in the same plantation. In one part of the high density plantation logged 6 years previously, felled timber had been winched or wheeled with a gurney to the roadside, then transported to a sawmill. In the other part of the plantation, timber was cut into logs and rolled to nearby pitsawing stations (each station $\sim 100\text{ m}^2$). At pitsawing stations, logs were rolled onto scaffolding above a pit. Then, two men, one on the log and the other in the pit, used a large handsaw to cut the log into boards. Where pitsawing was used to process timber, stations cover 20–30% of the area. Left at each station was a thick layer of sawdust, compacted soil, and timber scraps potentially hindering forest regrowth. In addition, scaffolding was often constructed from native tree saplings cut from the plantation or adjacent forest. On the other hand, while pitsawyers felled all timber, only larger stems were processed, potentially leaving patches where native stems were less damaged than where all stems were extracted for millsawing. We compared how these two timber processing methods affected forest regrowth by comparing vegetation plots that had been randomly placed into either pitsawn or millsawn parts of the plantation ($n = 16$ and 10 plots, respectively).

2.5. Facilitating forest succession

2.5.1. Vegetation removal experiment

Resource competition among plants in early-successional habitats can restrict recruitment, growth, and survival of tree species (Holl, 1998a). To evaluate this, we removed all non-tree vegetation from 5 m × 5 m randomly placed plots in the recently logged low density plantation (~5 months after logging; $n = 22$ plots) and high density plantation (~10 months after logging; $n = 20$ plots). Approximately, every 4 months (mean ± S.D. = 4.1 ± 1.5 months), plots were cleared of all aboveground non-tree vegetation. Larger stems (ht ≥ 0.5 m) were cut at the base, while smaller stems were pulled from the ground. Plots were surveyed when first cleared and 2 years later. During surveys, all tree saplings were identified, counted and their heights measured; stems <1 m tall were surveyed in a 2 m × 2 m subplot in the plot center. Vegetation removal plots were compared to the vegetation plots of the same successional age in the same plantation (control plots; no heavily disturbed plots included). Height, density, and species richness of trees were compared between removal and control plots for initial and final surveys. Seedling density and species richness were compared between treatments for final surveys.

2.5.2. Seedling planting experiment

Planting seedlings into successional habitats has been suggested to be an effective strategy for overcoming dispersal limitation and improving species composition or structure of developing forests (Tucker and Murphy, 1997; Parrotta and Knowles, 1999). We contrasted success of nursery-raised seedlings transplanted into two plantations logged 7–20 and 63–75 months earlier (hereafter, “younger” and “older” site, respectively). Initially, the younger site was open with little vegetation above 0.25 m, but by the end of the experiment, it had a dense, short canopy (~1.5 m), composed of herbaceous growth, shrubs and trees. Thus, light availability shifted from direct to indirect during the experiment. The older site’s canopy was 3–7 m tall, composed of tree saplings and shrubs; usually only indirect sunlight and sun flecks reached the understory.

Seedlings of local tree and shrub species were selected based on seed availability and nursery germination success. Seeds of each species were planted in

nursery beds in full sun or below a short (1 m) thatch roof. Planted seeds were watered every few days (unless it had rained) until seedlings were transplanted, usually 2–4 weeks after germination. Within a species, all transplanted seedlings had the same number of cotyledons and leaves, and were roughly the same height. After planting, seedlings were watered once (~200 ml) to reduce transplanting mortality. When possible, seedlings germinating in full sun were planted into the younger site (65% of the time), and seedlings germinating in shade were planted into the older site (94% of the time). When few seedlings germinated from one of the light treatments, seedlings were taken from the other light treatment for both successional habitats (e.g., shade-germinated seedlings planted into the younger site). The length of time that transplanted seedling species were monitored varied because timing of germination varied. Altogether, 33 species were planted into the nursery, only 17 species produced enough seedlings for experimentation.

Seedlings were planted into the two successional habitats during the same or subsequent days. In the younger site, seedlings were planted along ten 50 m transects placed parallel to one another and separated by 10 m; seedlings stations were at 1 m intervals. Conspecifics were planted with 15 m minimum distance between them on transects. A similar design was used in the old successional habitat, but we used variable length parallel transects, each separated by >5 m (shorter distance due to area constraints).

The proportion of seedlings surviving to the end of the experiment was calculated for each species, and the proportions from all species were compared between treatments. Within each species, proportions of surviving seedlings were also compared between treatments. For each seedling surviving to the end of the experiment, relative growth rate (RGR) was calculated by subtracting the natural log of initial seedling height from the natural log of final seedling height, and dividing this difference by the number of days growth was monitored. In this experiment, RGR is a product of intrinsic and extrinsic (e.g., herbivory) factors. To look for overall growth differences between treatments, a mean RGR was calculated for each species in each treatment, and means were compared between younger and older sites. Within each species, RGRs of individual seedlings were compared between treatments. Survival proportions and RGRs of

early- versus mid- and late-successional species were also compared (the latter two categories were lumped due to low numbers of late-successional species). These designations were based on Eggeling and Dale (1952), Polhill (1952), Hamilton (1991), Katende et al. (1995), and our own observations.

2.5.3. *Recruitment below snags*

In the recently logged low density plantation, many dead cypress trees remained standing (cause of death unknown). To determine whether these snags attracted seed dispersers and facilitated seedling recruitment, we surveyed seedlings below snags and in adjacent open areas at 17–19 months after logging. Twenty snags (ht \geq 10 m) were randomly chosen from \sim 50 candidates, and a 2 m² plot was placed below the tallest part of each. A control plot was placed in a random direction 10 m from the snag in an area without snags or emergent trees. In both plots, all tree and shrub stems were identified, counted and their heights were measured.

2.6. *Alternatives to plantations—fire exclusion*

Within one of the high density plantations is a section (1.7 ha) where pine seedlings were either not planted, or died soon after planting for unknown reasons (possibly drought or fire according to B. Kitembo (Uganda Forestry Department) who helped to establish and maintain the plantations, pers. comm.). We believe seedling death was not due to edaphic factors, since this plot was surrounded by mature high density plantation and mature forest. This site and the adjacent plantations were fire-maintained grasslands when the pines were planted in the 1950s and 1960s. As the surrounding plantations matured, the maintenance of fire-breaks and reduction of grasses (probably due to shading) in plantation understories reduced or stopped fire occurrence, and a well-developed successional forest now occupies the site. This represents a case study of succession following fire exclusion on degraded grassland (hereafter, “fire-excluded site”).

Twenty-five 10 m \times 2 m plots were placed and surveyed in the fire-excluded site following the methods used in surveys of logged plantations. The obvious control for assessing how effective fire exclusion is as a restoration tool are Kibale grasslands never planted with timber species. While not resurveyed for

this study, these areas are still fire-dominated grasslands where tree establishment and growth is minimal (Duncan and Duncan, 2000; Zanne and Chapman, 2001).

Given that fire exclusion appears to promote forest succession, we wanted to compare it to the use of plantations as a forest restoration tool. Because establishing plantations and not harvesting them is a possible restoration tool, we compared successional forest in the fire-excluded site to that in the unlogged high density plantation. Because establishing plantations and later harvesting them is another restoration strategy, we compared the fire-excluded site to the high density plantation surveyed 5–6 years after logging. Like many large-scale studies, the spatial replication of this design is limited. However, given that all three sites share a similar landscape position and site history prior to plantation establishment, this was a rare opportunity to compare fire exclusion and plantation establishment as restoration strategies.

3. Results

3.1. *Influence of initial recruit availability*

Tree seedling density and species richness were similar between the unlogged high and low density plantations, but the high density plantation had significantly greater tree sapling density (16 \times more) and species richness (7 \times more) than did the low density plantation (Table 1). In the high density plantation, tree saplings were significantly taller (3 \times more) than in the low density plantation.

Within a year of logging, tree seedling density and species richness were similar between the high and low density plantations (Table 1). The high density plantation still had significantly more tree saplings (16 \times more) and sapling species (8 \times more). Tree saplings were still significantly taller (2 \times more) in the high density than in the low density plantation.

Four to six years after logging, tree seedling density and species richness were still similar between plantation types (Table 1). Tree sapling density and species richness in the low density plantation were now not different from those in the high density plantation. Tree sapling heights in low density plantation were similar to those in the high density plantation.

Table 1
Median (top number) and mean (bottom number \pm S.D.) values for trees in unlogged and logged high and low density plantations^a

Stem variable	Unlogged		6 Months			4–6 Years			
	High density	<i>P</i>	Low density	High density	<i>P</i>	Low density	High density	<i>P</i>	Low density
Seedling density (stems m ⁻²)	0.3 0.4 (0.5)		0.3 0.3 (0.4)	0.8 1.3 (1.3)		0.8 1.7 (2.8)	1.5 2.4 (3.5)		1.0 2.4 (3.1)
Seedling species richness (species m ⁻²)	0.3 0.3 (0.3)		0.3 0.3 (0.3)	0.5 0.6 (0.6)		0.5 0.5 (0.5)	0.5 0.6 (0.5)		0.5 0.6 (0.4)
Sapling density (stems m ⁻²)	0.6 0.8 (0.4)	***	<0.1 0.1 (0.2)	0.3 0.5 (0.4)	***	<0.1 0.1 (0.2)	0.5 0.6 (0.4)		0.5 0.7 (0.6)
Sapling species richness (species m ⁻²)	0.3 0.3 (0.1)	***	<0.1 0.1 (0.1)	0.2 0.2 (0.1)	***	<0.1 0.1 (0.1)	0.3 0.3 (0.1)		0.3 0.3 (0.2)
Sapling height (m)	2.9 2.9 (0.4)	***	1.1 1.3 (1.5)	1.5 1.7 (0.6)	***	0 0.7 (1.1)	5.5 6.1 (3.3)		5.0 5.6 (4.0)

^a Results of Mann–Whitney contrasts of stem variables between the plantation types within a successional age are also presented: (***) $P \leq 0.01$, (*) $P \leq 0.05$, (+) $P \leq 0.10$.
*** $P \leq 0.001$.

3.2. Harvesting effects on forest succession

3.2.1. Native vegetation before and after timber harvest

In the high density plantation, trees were significantly or marginally more ($\sim 2\times$ more) dense, species-rich and taller before harvest than <1 year after harvest (Table 2). However, tree seedlings were significantly more dense and species-rich after logging than they were before. The high density plantation harvested 5–

6 years previously had significantly more tree seedlings ($6\times$ more) and tree seedling species ($2\times$ more) than did the unlogged high density plantation (Table 2). Tree saplings were significantly more dense ($13\times$ more) in the unlogged than in the logged plantation, though tree sapling species richness and height were similar between plantations.

In low density plantations, stem variables were similar before and within 1 year of logging, except tree seedlings were significantly more ($3\times$ more)

Table 2
Results (*P* values) of Mann–Whitney comparisons of stem variables between the plantation types within a successional age are also presented when significant: *** = $P \leq 0.001$ ^a

Stem variable	Unlogged versus 6 months after logging		Unlogged versus 6 years after logging	
	High density	Low density	High density	Low density
Seedling density (stems m ⁻²)	0.014 L > UL	0.042 L > UL	0.007 L > UL	<0.001 L > UL
Seedling species richness (species m ⁻²)	0.051 (L > UL)	0.112	0.041 L > UL	0.008 L > UL
Sapling density (stems m ⁻²)	0.010 UL > L	0.689	0.047 UL > L	<0.001 L > UL
Sapling species richness (species m ⁻²)	<0.001 UL > L	0.450	0.744	<0.001 L > UL
Sapling height (m)	<0.001 UL > L	0.390	0.001 L > UL	<0.001 L > UL

^a Directions of significant or marginally insignificant (in parentheses) differences are indicated below *P* values (see Table 1 for tree variable values).

Table 3

Percentages of trees with and without damage within 1 year of logging and extraction of timber species^a

Stem size class	Bent	Broken	Cut	% total damaged stems	% undamaged stems	Total stems	P
Seedling	3.2	3.5	0.3	7.0	93.0 (87.1)	345	<0.001
Large seedling	3.7	12.0	3.3	19.8	80.2 (74.6)	242	<0.001
Sapling	11.7	7.3	11.3	32.3	67.7 (57.6)	452	0.002

^a Three damage categories, total percent damaged, percent undamaged stems, and total stem number are presented. Also results of Wilcoxon signed rank tests comparing for each category, the total damaged and undamaged stems in plots are also presented. Percentages of undamaged stems when *T. orientalis* is excluded are presented in parentheses.

dense after than before logging (Table 2). Low density plantations logged 4–6 years before had significantly greater tree seedling density (4× greater) and species richness (2× greater), and tree sapling density (13× greater), species richness (8× greater), and height (4× greater), than did the unlogged low density plantation (Table 2).

3.2.2. Stem damage from logging

During surveys, it was impossible to determine whether stems had survived logging or established after logging. Significantly more stems found during initial surveys were undamaged than damaged for all size classes (80% and 20%, respectively; Table 3). Tree saplings generally had more damage than did smaller size classes, probably because many of the smaller stems recruited after logging. Nearly one-third of tree saplings found had been damaged during logging. One pioneer tree species, *Trema orientalis*, only established after logging. When these stems were excluded, nearly half of all tree saplings showed damage (Table 3).

3.2.3. Damage from felling

Tree seedling density and species richness were similar between the felled-only site and the high density site logged 5–6 years previously (Table 4). There were non-significant trends for greater tree sapling density and species richness in the felled-only than logged site, but there was a trend for taller tree saplings in the logged than felled-only site. These results suggests that most damage to native stems occurs during timber felling, rather than timber extraction.

3.2.4. Within-site disturbance intensity

Significantly more tree seedlings (2× more) and tree seedling species (2× more) were found in moderately disturbed than in heavily disturbed sites <1 year after logging (high density and low density

plantations; Table 5). For all other stem variables, no differences existed between plot types.

When we compared regeneration between millsawn and pitsawn areas in the 6-year-old logged low density plantation, significantly more tree saplings (3× more) and marginally more sapling species (1.5× more) were found in millsawn plots. Tree seedling density and species richness, and tree sapling height were similar between the two areas.

3.3. Facilitating forest succession

3.3.1. Vegetation removal experiment

Initial height, density, and species richness of tree saplings were similar between plots where all non-tree

Table 4

Median (top number) and mean (bottom number ± S.D.) for stem variables from felled-only plots^a

Stem variable	Median and mean (S.D.), felled-only	P
Seedling density (stems m ⁻²)	1.1 1.8 (2.6)	0.669
Seedling species richness (species m ⁻²)	0.5 0.5 (0.2)	0.362
Sapling density (stems m ⁻²)	0.7 0.7 (0.3)	0.081 (F > L)
Sapling species richness (species m ⁻²)	0.3 0.3 (0.1)	0.054 (F > L)
Sapling height (m)	3.8 4.3 (1.4)	0.068 (L > F)

^a Stem variables from felled-only plots (F) were compared to those from a logged plantations where timber was felled and extracted (L). Directions of significant or marginally insignificant (in parentheses) differences are indicated below P values (Mann-Whitney tests; see Table 1 for tree stem values for unlogged and logged plantations).

Table 5

Median (top number), mean (bottom number \pm S.D.) and Mann–Whitney comparisons of stem variables from plots in heavily and moderately disturbed areas in high density and low density plantations <1 year after logging, and from millsawn and pitsawn sites within one low density plantation 6 years after logging

Stem variable	Heavily disturbed plots	Moderately disturbed plots	<i>P</i>	Millsawn plots	Pitsawn plots	<i>P</i>
Seedling density (stems m ⁻²)	0 0.9 (2.3)	0.8 1.5 (2.2)	0.042	2.3 3.7 (3.7)	0.9 1.7 (2.3)	0.232
Seedling species richness (species m ⁻²)	0 0.25 (0.41)	0.50 0.57 (0.53)	0.037	0.8 0.7 (0.4)	0.5 0.6 (0.4)	0.771
Sapling density (stems m ⁻²)	0.2 0.2 (0.2)	0.2 0.3 (0.4)	0.984	0.6 0.7 (0.5)	0.2 0.3 (0.3)	0.014
Sapling species richness (species m ⁻²)	0.1 0.1 (0.1)	0.1 0.1 (0.1)	0.652	0.3 0.3 (0.1)	0.2 0.2 (0.2)	0.062
Sapling height (m)	1.4 1.6 (1.2)	1.3 1.3 (1.0)	0.489	3.4 4.2 (2.2)	8.5 7.9 (6.0)	0.280

vegetation were removed and control plots in both the logged low and high density plantations (Mann–Whitney test, $P > 0.10$ for all contrasts; Table 6). After 2 years, height, density, and species richness of tree sapling and seedling variables in experimental plots were still similar to those in control plots of either successional habitat (Table 6).

3.3.2. Seedling planting experiment

The average proportion of seedlings surviving to the end of the experiment was not significantly different between younger and older habitats (*t*-test paired by species, $t = 0.8$, $P = 0.458$, proportions were arcsine-square root transformed). Within either the older or younger habitat, there was no difference in proportion

Table 6

Median (top number) and mean (bottom number \pm S.D.) values for trees in plots cleared of all non-tree vegetation versus unmanipulated controls^a

Plantation and year	Tree size class	Height (m)		Density (stems m ⁻²)		Species richness (species m ⁻²)	
		Control	Cleared	Control	Cleared	Control	Cleared
<i>Logged low density plantation</i>							
Year 1	Saplings	0 0.7 (1.1)	0.5 0.8 (1.3)	<0.1 0.1 (0.2)	<0.1 0.1 (0.1)	<0.1 0.1 (0.1)	<0.1 0.1 (0.1)
Year 3	Seedlings	–	–	0 0.2 (0.2)	0 0.2 (0.5)	0 0.1 (0.2)	0 0.1 (0.2)
	Saplings	1.6 1.7 (1.1)	1.7 1.6 (1.0)	0.4 0.5 (0.5)	0.3 0.4 (0.3)	0.2 0.2 (0.1)	0.1 0.2 (0.1)
<i>Logged high density plantation</i>							
Year 1	Saplings	1.5 1.7 (0.6)	1.5 1.5 (0.4)	0.3 0.5 (0.4)	0.4 0.8 (1.2)	0.2 0.2 (0.1)	0.2 0.2 (0.1)
Year 3	Seedlings	–	–	0 0.2 (0.4)	0.1 0.4 (0.7)	0 0.2 (0.3)	0.1 0.2 (0.3)
	Saplings	3.6 3.7 (1.2)	3.2 3.3 (1.2)	0.8 0.9 (0.6)	0.7 1.0 (1.0)	0.3 0.3 (0.1)	0.2 0.2 (0.1)

^a Treatments were compared with Mann–Whitney tests, but no differences ($P > 0.10$ for all contrasts) were found.

of seedlings surviving between early-successional versus mid- and late-successional species (Mann–Whitney test, $P = 0.118$ and 0.590 , respectively; Table 7). When proportions surviving were compared within species, two early-successional species (*Mae-sopsis eminii* and *Polyscias fulva*) had significantly more seedlings surviving ($10\times$ and $5\times$ more, respectively) in the older habitat, and two others (*Securinega virosa* and *T. orientalis*) had significantly more seedlings surviving ($11\times$ more for *S. virosa*; no *T. orientalis* survived in the older habitat) in the younger habitat (Table 7). Two other early-successional species (*Erythrococca trichogyne* and *Maesa lanceolata*) showed trends for greater survival ($2\times$ and $7\times$ greater, respectively) in the younger habitat, while a third species (*Prunus africana*) showed a trend for greater survival ($1.1\times$ greater) in the older habitat. Two late-successional species (*Uvariopsis congensis* *Monodora myristica*) had significantly greater survival ($4\times$

greater) in the older than in the younger habitat, while another (*Monodora myristica*) showed a trend for greater survival ($2\times$ greater) in the older habitat.

With means from all species pooled, mean RGRs were similar between younger and older successional habitats (t -test paired by species, $t = -1.0$, $P = 0.351$). In the younger and older successional habitats, there was a trend for early-successional species to grow faster ($1.9\times$ and $1.7\times$, respectively) than mid- and late-successional species (Mann–Whitney test, $P = 0.079$ and 0.064 , respectively). RGRs within species differed significantly between the two successional habitats for several species (Table 7). For four early-successional species (*Erythrina excelsa*, *P. africana*, *P. fulva*, *E. trichogyne*), RGRs for seedlings in the younger successional habitat were significantly greater (1.4 – $1.6\times$ greater) than were those in the older successional habitat. For one late-successional species (*M. myristica*), RGRs were significantly greater in the

Table 7

RGRs and proportions surviving for seedlings transplanted into younger and older successional habitats (averaging 14 and 69 months after logging, respectively)^a

Seedling species	Guild	Number planted		Days observed	Mean height (cm)	RGR per 100 days			Proportion surviving		
		Young	Old			Young	Old	P	Young	Old	P
Trees											
<i>A. grandibracteata</i>	E	30	30	171	3.9	0.34	0.29	0.325	0.37	0.47	0.549
<i>Aphania senegalensis</i>	L	30	30	132	6.8	-0.01	0.09	0.325	0.60	0.70	0.631
<i>Bridelia micrantha</i>	E	15	12	358	4.1	0.41	0.40	0.624	0.27	0.25	0.933
<i>Celtis durandii</i>	M	30	23 ^b	374	3.6	0.37	0.27	0.302	0.20	0.30	0.440
<i>Cordia abyssinica</i>	M	30	30	41	3.0	1.53	1.18	0.588	0.20	0.37	0.225
<i>Diospyros abyssinica</i> ^c	M	15	15	359	5.3	0.07	0.06	0.837	0.60	0.67	0.819
<i>E. excelsa</i>	E	30	30	172	5.8	1.13	0.68	<0.001	0.67	0.70	0.876
<i>Fagaropsis angolensis</i> ^c	M	9	8	179	2.5	0.54	0.42	1.000	0.22	0.38	0.560
<i>M. lanceolata</i> ^c	E	30	30	443	0.5	0.87	0.83	0.617	0.20	0.03	0.059
<i>M. eminii</i>	E	30	30	384	6.1	0.24	0.35	0.384	0.03	0.30	0.011
<i>M. myristica</i> ^c	L	30	30	309	11.0	-0.37	0.02	<0.001	0.33	0.77	0.024
<i>P. fulva</i>	E	30	30	384	3.0	0.58	0.37	0.021	0.10	0.50	0.005
<i>P. africana</i> ^c	E	30	15	440	5.3	0.46	0.31	0.002	0.63	0.67	0.095
<i>T. orientalis</i>	E	30	30	440	1.3	0.40	-	-	0.27	0.00	-
<i>U. congensis</i> ^c	L	30	15	261	7.3	-0.04	0.07	0.230	0.13	0.53	0.006
Shrubs											
<i>E. trichogyne</i> ^c	E	30	30	447	1.7	0.60	0.43	0.005	0.73	0.40	0.086
<i>S. virosa</i>	E	30	30	434	0.8	0.64	0.82	0.752	0.33	0.03	0.007
Mean (all species)				313	4.2	0.46	0.41		0.35	0.42	
S.D. (all species)				129	2.7	0.46	0.32		0.22	0.25	

^a Provided for each species is successional guild (E: early-, M: mid- and L: late-successional), sample number, number of days seedlings were monitored, and mean initial height. P values for RGR and proportion surviving are from Mann–Whitney and chi-square tests, respectively.

^b 18 of these seedlings germinated in shade plots.

^c Species that germinated only from sun plots.

Table 8

Median (top number) and mean (bottom number \pm S.D.) density and species richness of seedlings and saplings below snags and in adjacent control areas within a low density plantation 1.5 years after logging^a

Stem variable	Density (stems m ⁻²)			Species richness (species m ⁻²)		
	Snag plots	<i>P</i>	Control plots	Snag plots	<i>P</i>	Control plots
Tree seedlings	0.8	*	0.3	0.5	*	0.3
	1.8 (2.4)		0.4 (0.5)	0.8 (0.9)		0.3 (0.4)
Shrub seedlings	0		0	0		0
	0.1 (0.3)		0.2 (0.3)	0.1 (0.2)		0.2 (0.3)
Tree saplings	0		0.5	0		0.5
	0.7 (1.1)		0.5 (0.5)	0.5 (0.6)		0.5 (0.4)
Shrub saplings	1.5	*	0.5	0.5	*	0.5
	1.3 (0.9)		0.8 (0.8)	0.7 (0.4)		0.5 (0.4)

^a Results of Wilcoxon Signed Rank comparisons of stem values are presented when significant: * = $P \leq 0.05$.

older successional habitat (*M. myristica* had negative growth in the young habitat). Four species showed trends or significant differences between treatments for both RGR and survival. Oddly, two of these had better growth but lower survival in the sun (*P. africana*, *P. fulva*).

3.3.3. Recruitment below snags

Density and species richness of animal-dispersed tree seedlings and shrub saplings were significantly greater (~ 1.5 – $3\times$ greater) below snags than control plots (Table 8). However, animal-dispersed shrub seedling and tree sapling density and species richness

were similar between snag and control plots. Densities and species richnesses of non-animal-dispersed trees and shrubs were similar between snag and control plots (Mann–Whitney tests, $P > 0.10$ for all contrasts) suggesting these areas were similar for other factors potentially affecting recruitment.

3.4. Alternatives to plantations—fire exclusion

The successional forest on the fire-excluded site had a tall (15–25 m), closed canopy of *Albizia grandibracteata* trees, with an open understory composed mainly of early- and mid-successional tree saplings

Table 9

Median and mean stem variables from the fire-excluded site, and results of Mann–Whitney contrasts of stem variables (*P* values) between this site and logged and unlogged high plantations^a

Stem variable	Contrasts with fire-excluded site (FES)		
	Median and mean (S.D.)	Unlogged high density (UHD)	Logged high density (LHD)
Seedling density (stems m ⁻²)	0.50	0.031	0.104
	0.93 (0.91)	FES > UHD	(LHD > FES)
Seedling species richness (species m ⁻²)	0.50	0.079	0.383
	0.47 (0.34)	(FES > UHD)	
Sapling density (stems m ⁻²)	1.00	0.240	0.003
	0.90 (0.38)		FES > LHD
Sapling species richness (species m ⁻²)	0.35	0.021	0.028
	0.35 (0.15)	FES > UHD	FES > LHD
Sapling height (m)	4.25	0.005	0.012
	3.93 (1.18)	FES > UHD	LHD > FES

^a The directions of significant differences are indicated below *P* values. Marginally insignificant trends are indicated within parentheses (see Table 1 for stem values for unlogged and logged plantations).

and few shrubs. When compared to the unlogged high density site, the fire-excluded site had significantly greater tree seedling density, tree sapling species richness, and tree sapling height ($\sim 2\times$ greater for all variables); there was a trend for greater tree seedling species richness in the fire-excluded site (Table 9). Tree sapling density was similar between the unlogged and fire-excluded site.

When the fire-excluded site was compared with the high density plantation at 5–6 years after logging, the former had greater tree sapling density ($2\times$ greater) and species richness ($1.5\times$ greater). In contrast, the logged plantation had taller tree saplings ($1.3\times$ taller). There was a trend for greater tree seedling density in the logged plantation, but tree seedling species richness was similar between sites.

4. Discussion

Our results provide insight into how exotic tree plantations can initiate forest regrowth in arrested successional habitats, a restoration strategy recently receiving much interest from researchers and managers. We relate our findings to four considerations for using plantations as a restoration tool; influence of initial recruit availability; effects of harvest on succession; facilitation of succession and alternatives to using plantations.

4.1. Influence of initial recruit availability

Differences in tree variables in unlogged high and low density plantations led to differences in postharvest succession between plantation types. Fimbel and Fimbel (1996) also found these disparities between unlogged high and low density plantations in Kibale. They concluded these differences were not soil-related, since pH and quantities of P, K, Ca, Mn and B were similar in soils from either plantation type. Concentrations of Mg, Cu and Zn were significantly lower in high density than low density plantations, a trend that should support more, not less, recruitment in low density than high density plantations. In addition, recruitment differences are probably not related to landscape position since planting sites for low density and high density plantations were similar (Osmaston, 1959; Fimbel and Fimbel, 1996). Fimbel and Fimbel (1996) hypothesized

recruitment differences arose from faster maturation rates of pines than cypress, more understory light in high density than low density plantations, and greater seed dispersal into high density than low density plantations. Studies elsewhere in the tropics have also found differences in native stem recruitment among plantation types (Parrotta, 1995, 1999; though see Geldenhuys, 1997; Keenan et al., 1997; Powers et al., 1997; Otsamo, 2000), suggesting succession will vary among logged plantations in other regions.

We expected forest regrowth in logged low density plantations to be slower than in logged high density plantations. While true during the first few years of succession, by 4–6 years tree sapling density and species richness were similar between the successional habitats. This rapid convergence suggests managers may not always need to invest resources into accelerating succession on sites with low initial recruit densities. For example, if managers at Kibale want forest structure in logged low density plantations to match that in logged high density plantations, they could intervene in the logged low density plantations (e.g., seedling planting), or wait 5–10 years for unassisted convergence with succession in the logged high density plantations. When management goals are more specific (e.g., obtaining a certain density for one species), waiting for convergence may not be sufficient. The difficulty will be predicting when convergence will happen naturally, or when intervention is needed.

Other studies of succession in the tropics have not found evidence for convergence. In Nigeria, Adedeji (1984) found no convergence in stem recruitment after 2 years of succession where slash was either burned or removed after forest clearance. Uhl et al. (1982, 1988a) found no convergence during the first 8 years of succession following light, moderate, or heavy pasture use in Amazonia. Rates of successional convergence probably depend on various factors, including variation in site degradation and availability of seed sources and seed dispersers. Managers should be cautious about assuming whether convergence will occur among different successional habitats.

4.2. Harvesting effects on forest succession

Logging in the low density plantations had no effect on most stem variables within the first year; the

exception was an increase in tree seedling density in the logged plantation. In contrast, logging in high density plantations led to declines for sapling variables and increases for seedling variables within the first year. This increase in seedling density in both plantation types after logging was likely due to increased resource availability (especially light) for seedlings in the understory. Perhaps saplings declined in the high density and not in the low density plantation because unlogged high density plantations have twice the density of timber stems and 16 times the density of native tree saplings than do unlogged low density plantations.

In the low density plantations, timber extraction seems to accelerate forest succession. Tree seedlings and saplings are more abundant and species-rich 4–6 years after logging than in unlogged plantations. In the high density plantations, timber extraction leads to increased seedling recruitment and taller saplings, but sapling density is still less than in unlogged high density plantations 5–6 years after logging. Once tree seedlings survive and recruit into larger size classes, sapling densities in logged high density plantations may match or exceed densities in unlogged high density plantations. Thus, despite heavy damage to native stems during logging, timber extraction accelerates natural forest succession in plantations. Thus, when managing for natural forest regrowth, it may be better to harvest timber than leaving plantations intact. This supports the findings of others that removal of tropical plantation species can promote native seedling growth (Ashton et al., 1998; Otsamo, 1998).

Many questions remain about using plantations for forest restoration. For example, how do we determine when to harvest during plantation maturity to maximize native tree recruitment? Because native stems become bigger and, initially, more dense as plantations age (Geldenhuys, 1997; Keenan et al., 1997; Parrotta, 1999), harvesting when plantations are older may be better. On the other hand, if logging destroys much of the native plant community, then logging when plantations are younger may reduce native stem loss. When plantations are harvested, managers could plan felling and extraction to limit native stem damage. For example, timber could be directionally felled to avoid patches of native vegetation or native species of particular interest. Or, selective logging could reduce damage to advanced regeneration, while still accelerating forest succession. Any extraction

techniques reducing damage to native stems will also benefit succession. For example, in one small plantation area at Kibale, timber was winched by cable to the road, thus sparing many native trees.

Variation in harvesting activities within logged forests can produce spatial variation in forest regrowth (Pinard et al., 2000). Our data suggest seedling recruitment is negatively related to disturbance intensity. However, tree saplings were similar in heavily and moderately disturbed areas, perhaps due to their overall low densities early in succession. When we compared millsawn and pitsawn areas within a single plantation, we found greater tree sapling densities in millsawn than pitsawn plots. One explanation for this is there is less damage to native stems in areas where timber is processed off-site (millsawn) than areas where timber is pitsawn on-site. However, tree seedling variables were similar between treatments, suggesting recruitment processes are similar between millsawn and pitsawn areas later in succession.

4.3. *Facilitating forest succession*

We quantified whether vegetation removal, direct plantings, and leaving snags helped facilitate forest succession in logged plantations. These interventions were within the first years of succession, when succession is potentially more tractable than it is later when it becomes more difficult to alter composition or structure of vegetation.

Previous studies have indicated that vegetation removal can have either positive or negative effects on forest succession. Successional habitats often have high densities of non-tree vegetation that may limit tree recruitment and growth (Tilman, 1990; Putz and Canham, 1992; Berkowitz et al., 1995; Sun and Dickinson, 1996; Holl, 1998a). On the other hand, harsh abiotic conditions of recently disturbed habitats can be unfavorable for tree establishment (Aide and Cavellier, 1994; Brown and Lugo, 1994), and shade provided by non-tree vegetation may help establishing trees (Nepstad et al., 1991; Vieira et al., 1994; Zahawi and Augspurger, 1999). Our removal of all non-tree vegetation did not effect recruitment, survival, or growth of trees at the community level. This suggests that any facilitative and inhibitive effects of non-tree vegetation on trees were weak, or that they offset each other. We conclude that removal of non-tree vegetation at

our site is not a beneficial management strategy in early-successional habitats, although it may be useful for particular tree species. Chapman et al. (in press) found that tree density declined in plots where most non-tree vegetation was removed in a 4–6-year-old forest on a logged low density plantation at Kibale. In contrast, others have found that thinning non-tree vegetation in successional habitats enhances tree recruitment and growth (Holl, 1998a; Otsamo, 1998; Guariguata, 1999). Until more is known, managers should be cautious about assuming that removing non-tree vegetation will benefit forest regrowth in early-successional habitats.

Less than half of seedlings we planted into successional habitats at Kibale survived, and successional habitat age was important for few species. Direct seedling planting into degraded habitats tends to be a successful strategy, although it is expensive, time consuming, and labor intensive (Tucker and Murphy, 1997; Parrotta and Knowles, 1999). Direct planting overcomes recruitment limitations due to limited seed dispersal, high seed predation, and poor germination. However, seedlings are vulnerable to other hazards in disturbed habitats including herbivory, water-stress, and shading from adjacent plants. Our limited success could be improved if plantings of individual species were timed to optimize survival and growth.

We found greater density of animal-dispersed tree seedlings and shrub saplings below snags than in adjacent areas without snags. However, animal-dispersed tree saplings and shrub seedlings were similar between snag and control plots. One explanation for this pattern is that optimal conditions for recruitment were early during succession for shrub species and later for tree species. Non-animal-dispersed species were similar between snag and control plots suggesting that above patterns were not due to differential recruitment conditions. Seed rain is usually greater below emergent perches than in adjacent open areas of other disturbed tropical systems (Holl, 1998b; Duncan and Chapman, 1999; Toh et al., 1999). Dispersal to emergent perches may be especially important where other vegetative resources for succession are absent. However, enhanced seed dispersal must be coupled with favorable conditions for seedling recruitment (Holl, 1998b), and the effectiveness of snags or other perches may be limited since they often cover only small portions of degraded areas.

4.4. Alternatives to plantations—fire exclusion

We compared forest succession in the fire-excluded site to that in an unlogged and logged (5–6 years after logging) high density plantation, each of which represents a strategy for managing plantations to promote natural forest regrowth. We found that tree sapling density in the fire-excluded site was similar to that in the unlogged high density plantation, and that sapling densities in both these treatments were greater than in the logged plantation. Tree sapling species richness followed a similar pattern. Thus, in terms of native tree sapling recruitment, fire exclusion was at least as good as plantations as a restoration strategy. Tree saplings were taller in the logged plantation than the fire-excluded site (though only by 1.3 m). Saplings were taller in the fire-excluded site than the unlogged plantation. This suggests that plantation establishment and logging may benefit forest succession more than fire exclusion for tree growth. However, the fire-excluded site had a taller canopy than the logged site due to the presence of tall pioneer trees not always found in plots. Seedling recruitment was greater in the logged plantation than the fire-excluded site, and both treatments had greater seedling recruitment than the unlogged plantation. Without seedlings survivorship data, it is difficult to predict whether the higher seedling density in the logged plantation would enable it to surpass the fire-excluded site for sapling densities. Whether fire exclusion or plantation establishment with subsequent logging is better for promoting forest succession may depend on which variables are emphasized (e.g., sapling growth versus recruitment). Since from this analysis neither strategy is clearly better, decisions on whether to exclude fire or plant timber in degraded fire-prone habitats may depend on other considerations. For example, the costs of fire exclusion versus plantations establishment and maintenance may differ, fire exclusion will not provide the financial benefits possible through timber production, and degraded areas may need many years of fire exclusion until they become less flammable.

4.5. Plantations as a restoration tool

If plantations are to be used for forest restoration, we need to know more about how to use them to maximize native tree recruitment and growth. Our

findings suggest harvesting timber promotes forest succession more than leaving timber intact, and fire exclusion seems to be just as or more effective than either of these strategies, although one must consider the costs of excluding fire and the timescale of the program. Approaches using a combination of strategies may be particularly useful (Lamb, 1998). For example, large degraded grasslands could be protected by fire-breaks. Smaller areas within the grassland could be planted with timber species to generate income, an extractable resource, and an alternative restoration strategy if fire exclusion fails. Potentially invasive species should be avoided. Using suitable native species would avoid this threat, and probably increase plantation visitation by native animals (Lamb, 1998).

Non-ecological challenges exist to using plantations as a restoration strategy (Lamb, 1998). Predicting profits from the sale of timber involves estimating the market value of timber several decades in advance. This may be difficult since market prices depend on fluctuating local, regional, and global timber supplies (Leslie, 1999). In addition, there can be a constant temptation to manage plantations for profit rather than for conservation priorities (Lamb, 1998). For example, because an understory of native species will slow timber growth, there will be temptation to clear colonizing native vegetation (Lamb, 1998). Or, managers may decide to replant harvested plantations with more timber species rather than allowing natural forest succession to proceed. For these reasons, managers and institutions backing them need to carefully consider how using plantations to restore biodiversity compares to alternative strategies.

Acknowledgements

We thank Kaoru Kitajima, Carmine Lanciani, Doug Levey, and Francis “Jack” Putz for helpful comments on this paper. We would also like to thank field assistants Swaibu Katusabe Amooti and Francis Katuramu Amooti for their dedication and hard work. Funding for this research was provided by the Ford Foundation, Wildlife Conservation Society, National Geographic Society, The Explorer’s Club, Lindbergh Foundation, University of Florida, and National Science Foundation (NSF Graduate Fellowship and

grant numbers SBR-9617664, SBR-990899). Permission to conduct this research was given by the Office of the President, Uganda, National Council for Science and Technology, Uganda, Uganda Wildlife Authority and Ugandan Forest Department.

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