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RESEARCH ARTICLE

Planting high-value timber species in log yards in a Central African rainforest

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Abstract

Introduction: Planting high-value timber species in logged forests can contribute to long-term wood availability. Although log yards, forest clearings where harvested logs are temporarily stored, may be suitable planting areas for light-demanding timber species, soil compaction by heavy vehicles may affect species performance.

Objectives: We evaluated the survival and growth of high-value native timber species planted in former log yards in Cameroon.

Methods: We planted seedlings of 11 species in 147 log yards in two forest concessions in south-east Cameroon, monitoring 4453 saplings during one to 6 years. We assessed the effects of species, time since planting, soil ripping with bulldozer claws down to 40 cm, and log yard area on sapling survival and growth using linear mixed-effects models.

Results: Survival rates varied among species and time, averaging 83.1% 1–2 years since planting and rising to 94.1% between 1 and 2 years and 3 and 6 years. Seven species maintained annual survival rates above 80% throughout. Survival rates did not vary with soil ripping or log yard area (108–990 m^2). Growth rates varied between species and time since planting with the highest increments in diameter observed for *Terminalia superba* (20.6 mm/year) and *Pterocarpus soyauxii* (12.2 mm/year). Sapling growth did not increase with the log yard area. Soil ripping affected the growth of a few species, such as *T. superba*, which grew faster in ripped log yards.

Conclusions: This study supports the planting of native tree species in Central Africa's former log yards and outlines key implementation challenges.

Implications for Practice: The consistently high survival rates recorded for most planted species over time indicate their capacity to establish within log yard environments. This research suggests enriching the log yards of southern Cameroon with *Terminalia superba* and *Pterocarpus soyauxii* may increase densities of these species over time. Soil ripping before planting did not improve sapling survival and had a minor influence on growth, despite its additional cost, highlighting that clearing competing vegetation is a more effective investment. Nevertheless, since soil ripping benefited *T. superba*, the fastest-growing species, it could be applied selectively to accelerate its growth and shorten the time to harvest.

Key words: Cameroon, enrichment planting, log landings, log yard ripping, logging concession, tree performance

Introduction

In Central Africa, legal logging must comply with management plans that include rotation of tree felling over time and space (Ruiz et al. 2005; Umunay et al. 2019). Forest concessions granted to private companies are divided into annual harvesting areas depending on the cutting cycle (25–30 years). Logging is highly selective, with only one to two trees felled per hectare per cycle (Karsenty & Gourlet-Fleury 2006; FRMi 2018), affecting 4–11% of the forest cover (Durrieu de Madron et al. 1998; Medjibe et al. 2011; Dupuis et al. 2025). Despite low canopy disturbance, the density of commercial species, most of which are light-demanding, decreases over time (Karsenty & Gourlet-Fleury 2006). These species, whose current presence in the canopy results from past disturbances, struggle to regenerate in those low-impacted dense forests (Bourland et al. 2012; Morin-rivat et al. 2016; Makemba et al. 2022). Consequently, tree planting is often suggested by national legislation and sustainable forest management (SFM) certification bodies such as Forest Stewardship Council (FSC) and Programme for the Endorsement of Forest Certification (PEFC) (Schulze 2008; Doucet et al. 2016; Brown et al. 2020). However, implementation is often constrained by the limited availability of open spaces for planting.

Between the four main sources of canopy disturbances in logged forests—logging gaps, roads, skid trails, and log yards (Dupuis

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et al. 2023)—log yards account for 0.1–1.9% of the annual cutting area (Asner et al. 2004; Medjibe et al. 2011; Gideon et al. 2014). Also known as log landings, landing sites, or log decks, these are temporary storage infrastructures for harvested logs (Bonnell et al. 2011; Karsten et al. 2014; de Carvalho et al. 2017).

Typically located along forest roads, they are created by clearing trees and compacting the soil with logs and heavy vehicles (Nussbaum et al. 1995; DeArmond et al. 2022). Their size varies, often smaller than 500 m^2 for Reduced Impact Logging (Daïnou et al. 2021) to 1000 m^2 for conventional logging (Durrieu de Madron et al. 1998). Their location depends on the density of trees harvested (Gideon et al. 2014). Once the annual cutting area has been harvested, the log yards are abandoned and the roads are closed for the duration of the cutting cycle (Veldman & Putz 2010).

As log yards can be reused during the next cutting cycle (Biwolé et al. 2019; Duhesme et al. 2021), questions arise about the relevance of planting seedlings at these sites. However, three main arguments support planting in log yards: (1) harvestable timber availability and location can vary between cutting cycles, which can lead operators to relocate log yards; (2) planting could support carbon sequestration (Cerullo & Edwards 2019; Philipson et al. 2020); and (3) reforestation of log yards can increase the density of timber species, increasing future forest productivity (Cerullo & Edwards 2019).

Tree species often struggle to recover and colonize abandoned log yards slowly (Malmer & Grip 1990) as a result of the topsoil removal and soil compaction (Craul 1994; Watson & Kelsey 2006). The time required for many tree species to reestablish in abandoned log yards can only very poorly be estimated (Malmer & Grip 1990; DeArmond et al. 2022) justifying that some forest companies are required to restore log yards as part of their SFM certification before leaving annual cutting areas.

In Central African countries, some SFM certified logging companies restore log yards with the planting of commercial species, although the species that can thrive at these degraded sites remain poorly identified. In some concessions, the soil is ripped to promote tree growth, but its effectiveness remains unclear. Soil ripping is a mechanical method of plowing before planting (Hartmann et al. 2008; Ning et al. 2022). Often the organic layer is pushed away before compacting the soil and brought back at the end of the operation. It is often carried out with a bulldozer and it aims to improve soil structure and fertility (Ning et al. 2022). However, it can stimulate competing vegetation, which can inhibit planted seedlings' growth (Toledo-Aceves & Swaine 2008; Rozendaal et al. 2020). Given the large number of log yards in a forest management unit (FMU), soil ripping is expensive and its efficiency should be tested (Kozlowski 1999).

In highly competitive environments, clearing competing vegetation can improve the performance of planted saplings (Doucet et al. 2009; Ouédraogo et al. 2014). However, clearing competing vegetation in log yards is difficult, especially in certified concessions, where roads are blocked and bridges fall down to limit access by poachers.

Finally, the size of the log yards can also influence the performance of the planted species. Depending on their guild, some species, such as pioneers, may thrive better in larger log yards, while others may perform well even in smaller ones.

This study aims to address the knowledge gap on the performance of seedlings planted in log yards in southeastern Cameroon. Specifically, we explore the following questions: (1) What are the survival and growth rates of native tree species planted in log yards, and which timber species are the most promising candidates for planting in log yards? (2) How do tree survival and growth of the planted trees vary through time? (3) Does soil ripping in log yards improve the survival and growth of planted seedlings? (4) Does the area of the log yard influence the performance of the species?

Methods

Study Area and Environmental Conditions

The sampled log yards were located within six FMU granted to two logging concessions in south-east Cameroon: Alpicam-Grumcam (between 3°N and 4.25°N and 14.2°E and 15°E) and Pallisco (between 3°N and 3.73°N and 13.2°E and 14.5°E) companies (Fig. 1). The FMUs were L_{10023}, L_{10026}, L_{10051} and L_{100053} for Alpicam-Grumcam and L_{10039} and L_{10044} for Pallisco. At the time of planting, Pallisco was FSC certified while Alpicam-Grumcam achieved certification as FSC later. In the study area, the mean annual temperature was 23.2°C and the mean rainfall was 1640 mm (Gorel et al. 2019), distributed in two distinct rainy seasons (August–December and March–June; Daïnou et al. 2011). The climate is Am-type according to Köppen classification (Beck et al. 2018).

The geological substrate consists of metamorphic rocks, including schistoquartzite, gneiss, migmatite, and micaschist (Ouédraogo et al. 2016; Vleminckx et al. 2017). The soils are weathered reddish or yellowish ferralsols (Fétéké et al. 2004), acidic (with pH values often <4) and have a very low cation exchange capacity (Vleminckx et al. 2017; Mantel et al. 2023).

The forest is mainly semi-deciduous, with the stand dominated by Fabaceae, Annonaceae, and Malvaceae (Letouzey 1985; Réjou-Méchain et al. 2021).

Study Species

We selected 11 native tree species that are commonly harvested in Central Africa (Table 1). *Entandrophragma cylindricum* (Sprague) Sprague and *Triplochiton scleroxylon* K.Schum. are among the top three most logged species in the Congo Basin, while *Afzelia bipindensis* Harms, *Pericopsis elata* (Harms) Meeuwen, and *Pterocarpus soyauxii* Taub are listed in the CITES Appendix II. This listing regulates their international trade and requires an export permit from the country of origin.

Log Yard Preparation, Planting, and Clearing Competing Vegetation

A total of 147 log yards were selected, with 89 and 58 located at Alpicam-Grumcam and Pallisco, respectively. The average area (mean ± standard deviation) of the log yards was 466.5 ± 208.8 m^2 (ranging from 134 to 990 m^2) at Alpicam-Grumcam, and 381.4 ± 120.9 m^2 (ranging from 108 to 750 m^2) at Pallisco. During the preparation of the log yards,

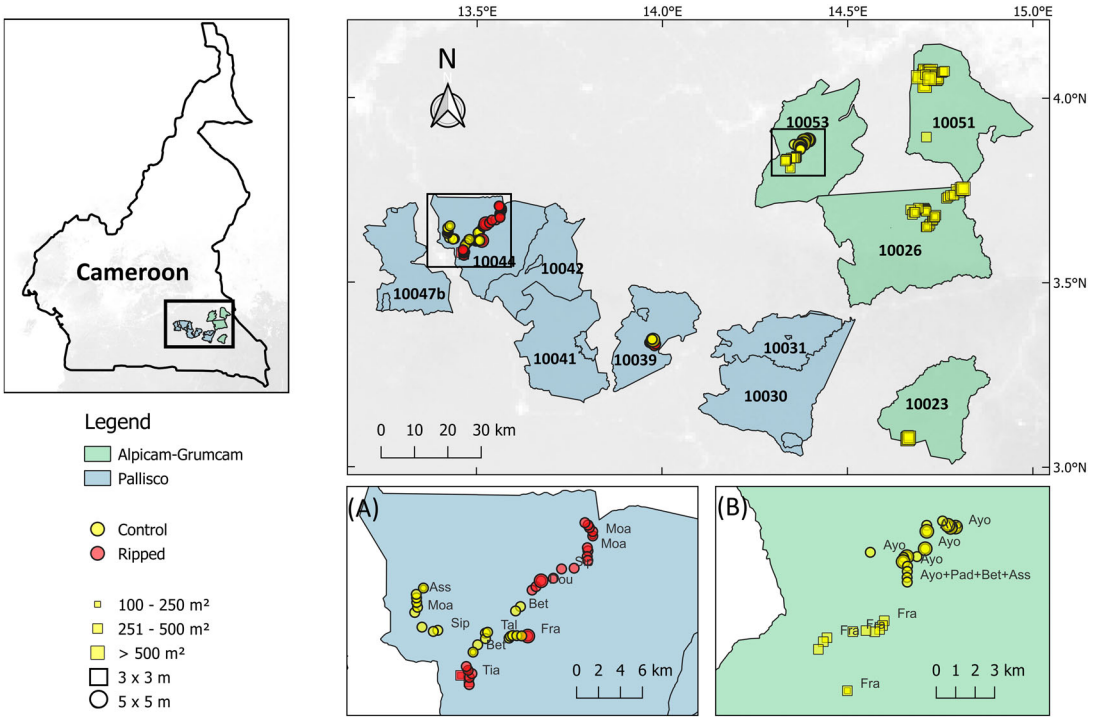

Figure 1. Location of the log yards studied. The numbered areas indicate the forest management units (FMUs) that are managed by the Alpicam-Grumcam and Pallisco companies. The tested variables are illustrated by (A) plots located in FMU 10044 and (B) plots located in FMU 10053. Some species are presented using their abbreviated pilot names (Table 1).

the litter and humus layers were pushed to the edges using a bulldozer. The topsoil was compacted with the logs and the movement of trucks and machinery. Two months after the end of the logging operations, all log yards at Alpicam-Grumcam and 29 at Pallisco did not undergo any treatment prior to planting. The soil of the 29 remaining log yards at Pallisco was ripped to a depth of 40 cm with bulldozer claws and the removed litter and humus layers were added prior to planting.

The planting was carried out between November 2016 and 2019. At Alpicam-Grumcam, 16 log yards were planted in 2017, 31 in 2018, and 42 in 2019. At Pallisco, two log yards were planted in 2016, 22 in 2017, 18 in 2018, and 16 in 2019. Only monospecific plantings were used at Pallisco, whereas 78 monospecific plantings and 11 mixed species were used at Alpicam-Grumcam (Table 1). The seeds were harvested from mature, well-formed trees in each concession and seeded in polyethylene bags in nurseries. When the seedlings reached 40 cm height (after 7–12 months), they were transplanted in log yards following a grid of 3 × 3 m or 5 × 5 m (Table 1). The planting holes were 35 cm deep, and a buffer zone of 2–5 m was cleared.

At Alpicam-Grumcam, log yard clearing of competing vegetation was carried out annually for up to 3 years, while at Pallisco, it was carried out only at the end of the first year, after which the log yards were abandoned.

Data Collection

Sapling survival, height, and diameter were recorded during three measurement campaigns after planting. The measurement included the diameter at 10 cm from the collar (D_{10}) for saplings smaller than 135 cm in height and the diameter at breast height (DBH, measured at 130 cm) for saplings larger than 135 cm in height. The first campaign was carried out immediately after planting, the second campaign was carried out 1–2 years after planting, and the third and last campaign was carried out 3–6 years after planting, from July 2023 to February 2024 (Fig. S1). All saplings were measured at the first and third campaigns but, unfortunately, 15% of them were not measured during the second campaign due to logistical issues. The height was measured from the collar to the terminal bud using a Vertex IV dendrometer (for saplings >2 m) or a ruler (for saplings <2 m). Diameter was measured with a caliper (for saplings with a diameter <8 cm) or girth tape (saplings with a diameter >8 cm).

Data Calculation

Given the limitations of our dataset (missing data at the second campaign), growth increments and survival rates were computed for two overlapping time periods. The first monitoring period, hereafter referred to as P1, spanned between the first and the second measurement campaign. The second monitoring period, hereafter referred to as P2, spanned between the second and third measurement campaign.

The survival rate S_t (%) was computed with Equation (1) and the annual survival rate S_{an} (%) was computed with Equation (2) (Sheil et al. 1995).

Table 1. Characteristics of study species and the number of study log yards (*N*) with the corresponding individuals (*n*). The species guild is either pioneer (P) or non-pioneer light-demanding (NPLD). The pilot names follow ATIBT (2016) nomenclature with additional and common names in brackets for some species. Alpicam-Grumcam had 78 monospecific (single) and 11 mixed plantations with *Pterocarpus soyauxii* + *Triplochiton scleroxylon* (5 log yards named ⅁); *Entandrophragma cylindricum* + *Erythrophleum suaveolens* (1 log yard, *); *Pericopsis elata* + *Baillonella toxisperma* (1 log yard, ‡), *Mansonia altissima* + *Terminalia superba* (1 log yard, ⅁), *M. altissima* + *P. soyauxii* (1 log yard, ⅂ↄ), *B. toxisperma* + *P. soyauxii* + *T. superba* (1 log yard, ↄ) and *M. altissima* + *P. elata* + *P. soyauxii* + *T. scleroxylon* (1 log yard, ⅄), while Pallisco was monospecific. The spacing used was 3 × 3 and 5 × 5 m.

				Alpicam-Grumcam						*Pallisco*			
				N			*Spacing*			N		*Spacing*	
Species	*Pilot names*	*Family*	*Guild*	*Single*	*Mixed*	n	*3 × 3 m*	*5 × 5 m*		*Single*	n	*3 × 3 m*	*5 × 5 m*
Afzelia bipindensis Harms	Doussié	Fabaceae	NPLD							6	103		6
B. toxisperma Pierre	Moabi	Sapotaceae	NPLD	6	1‡, 1ↄ	368	8			6	73		6
E. angolense (Welw.) C.DC.	Tiama	Meliaceae	NPLD							6	91		6
E. cylindricum (Sprague) Sprague	Sapelli	Meliaceae	NPLD	8	1*	288	9			5	88		5
E. utile (Dawe & Sprague) Sprague	Sipo	Meliaceae	NPLD							5	72		5
E. suaveolens (Guill. & Perr.) Brenan	Tali	Fabaceae	P	10	1*	360	11			4	66	1	3
M. altissima (A.Chev.) A.Chev.	Beté	Malvaceae	NPLD	5	1⅁, 1⅂ↄ, 1⅄	334	7	1		9	152		9
P. elata (Harms) Meeuwen	Afrormosia (Assamela)	Fabaceae	P	14	1‡, /⅄	475	15	1		5	68		5
P. soyauxii Taub.	Padouk	Fabaceae	NPLD		5⅁, 1⅂ↄ, 1ↄ, /⅄	134	2	6		5	87	1	4
T. superba Engl. & Diels	Limba (Fraké)	Combretaceae	P	23	/⅁, 1ↄ	1204	25			7	150		7
T. scleroxylon K.Schum.	Ayous	Malvaeae	P	12	5⅁, 1⅄	340		18					
Total				78	11	3503		58			950		

$$S_{\mathrm{t}} = \frac{N_{t_f} \times 100}{N_{t_i}}, \tag{1}$$

$$S_{\mathrm{an}} = 100 - \left(\left(1 - \left(\frac{N_{t_f}}{N_{t_i}} \right)^{1/\left(t_f - t_i\right)} \right) \times 100 \right), \tag{2}$$

where N_{t_i} and N_{t_f} were population counts at the censuses conducted at times t_i and t_f.

The annual increment in diameter (ΔD in mm/year) was calculated using D_{10} for the first monitoring period (P1) and DBH for the second monitoring period (P2) using Equation (3). Only 28% of the surviving saplings were measured at breast height during the second measurement campaign. Both increments in diameter (ΔD in mm/year) and height (ΔH in cm/year) were calculated for each sapling using Equation (3):

$$\Delta y = \frac{y_{t_f} - y_{t_{i(f>i)}}}{\left(t_f - t_i\right)/365.25} \tag{3}$$

with y the diameter (or height) measured in the censuses carried out at time t_i and t_f (in days).

We compared the diameter increments at D_{10} and DBH for individuals measured at both points simultaneously ($n = 1328$) and found a good relationship ($r^2 = 66\%$, $p < 0.001$; Fig. S2).

To assess the effect of the log yard area on survival and growth, the log yard area was classified into three classes (100–250, >250–500, and >500 m^2).

Statistical Analysis

Annual survival rate (S_{an}), diameter (ΔD) and height (ΔH) increments of tree i, of species j in plot k were modeled using linear mixed-effects models (LMMs), fitted with the *lmer* function of the "lme4" R package (version 1.1–30) (Bates et al. 2015). We include log yard (ρ_j) as a random effect to predict sapling survival. To predict the increase in diameter, we include the log yard and the sapling (θ_k) as random effects. The variance at sapling level in height increment was negligible ($<10^{-4}$), so θ_k was excluded from the height increment model to avoid a singular fit. Fixed effects included species (C; 11 levels), time since planting (P; 2 levels), their interaction, FMU (L; 6 levels), spacing (G; 2 levels), mixture (M; 2 levels), ripping (R; 2 levels), and log yard area (A; *3 levels*) (Equations 4 and 5).

$$\begin{aligned} S'_{\mathrm{an}_{ij}} = {} & \mu + b_{1,C_{ij}} + b_{2,P_{ij}} + b_{3,L_{ij}} + b_{4,G_{ij}} + b_{5,M_{ij}} + b_{6,A_{ij}} + b_{7,R_{ij}} \\ & + b_{8,C_{ij} \times P_{ij}} + \rho_j + \varepsilon_{ij}, \end{aligned} \tag{4}$$

$$\begin{aligned} \Delta y'_{ijk} = {} & \mu + b_{1,C_{ijk}} + b_{2,P_{ijk}} + b_{3,L_{ijk}} + b_{4,G_{ijk}} + b_{5,M_{ij}} + b_{6,A_{ij}} + b_{7,R_{ijk}} \\ & + b_{8,C_{ijk} \times P_{ijk}} + \rho_j + \theta_k + \varepsilon_{ijk} \end{aligned} \tag{5}$$

with $\rho_j \sim N\left(0, \sigma_\rho^2\right)$; $\theta_k \sim N\left(0, \sigma_\theta^2\right)$, and $\varepsilon_{ij\,(or\,k)} \sim N\left(0, \sigma_\varepsilon^2\right)$.

Parameters b_1–b_8 were fixed coefficients with μ as the model intercept. The model contrasts were estimated using *Terminalia superba* as a reference species ($b_1[C_{T.\ superba}] = 0$), due to its high survival and growth rates and FMU ($b_3[L_{10053}] = 0$) located at Alpicam-Grumcam.

The response variables S_{an} and Δy (ΔD or ΔH), were transformed using respectively: $S'_{an} = \sin^{-1}\sqrt{S_{an}}$, $\Delta y' = \log_e(\Delta D + 1)$, and $\Delta y' = \sqrt{\Delta H}$, to approximate normally distribution of the residuals.

Model selection was based on the lowest Akaike Information Criterion (AIC) and model parsimony. For significant interactions, we compared factor levels with a Tukey-adjusted post hoc test using the *emmeans* function of the "emmeans" R package (version 1.8.1-1). The selected model of annual survival rates was Equation (6), while the increments in diameter and height were modeled with Equation (7).

$$S'_{an_{ij}} = \mu + b_{1,C_{ij}} + b_{2,P_{ij}} + b_{3,C_{ijk}\times P_{ijk}} + \rho_j + \varepsilon_{ij}. \quad (6)$$

$$\Delta y'_{ijk} = \mu + b_{1,C_{ijk}} + b_{2,P_{ijk}} + b_{3,L_{ijk}} + b_{4,C_{ijk}\times P_{ijk}} + \rho_j + \theta_k + \varepsilon_{ijk}. \quad (7)$$

We assessed spatial autocorrelation among plots using Moran's I statistic (Moran 1948), setting the maximum neighborhood distance to 150 km. This corresponds to the greatest separation between the most distant plots. Analyses were performed with the "spdep" R package (version 1.3-3) (Bivand et al. 2011).

As ripping was only applied at Pallisco and the dataset was unbalanced between time since planting, we used an additional general linear model (Equation 8) to assess the effect of ripping by species and time since planting at Pallisco.

$$\Delta y'_{ijk} = \mu + b_{1,R_{ijk}} + \varepsilon_{ijk}. \quad (8)$$

Conditional and marginal r^2 values were calculated using the *r.squaredGLMM* function of the "MuMIn" R package (version 1.47.1) (Barton 2022) to assess the variance explained by the full model (with random effects, conditional r^2) and by fixed effects alone (marginal r^2), respectively. All analyses were performed in R version 4.2.1 (R Core Team 2022).

Results

Survival Rate

The average annual survival rate (S_{an}) was 83.1 ± 19.2% during P1. It increased slightly to 94.1 ± 13% during P2, although not significantly ($t = 0.91$, $p = 0.364$; Table S1; Fig. 2A). Seven species (*Pterocarpus soyauxii*, *Terminalia superba*, *Entandrophragma angolense*, *Erythrophleum suaveolens*, *Mansonia altissima*, *Triplochiton scleroxylon*, and *E. cylindricum*) maintained annual survival rates above 80% in both time since planting (Fig. 2A), while others ranged between 59 and 66% during P1 (see corresponding S_t values in Fig. 2B). During P2, most species exceeded 91% survival, except *Afzelia bipindensis* (75.1%) and *E. utile* (73.4%). Only *Baillonella toxisperma*, *Pericopsis elata*, and *T. scleroxylon* had differences in survival rates between times since planting (Fig. 2A). The model (Equation 6) explained 43.9% (marginal r^2) and 45.5% (conditional r^2) of the variance. Survival rates were not affected by ripping ($t = -0.36$, $p = 0.724$), spacing ($t = 1.52$, $p = 0.131$), planting composition (mixed or monospecific; $t = -0.71$, $p = 0.479$), and log yard area (Table S1). Tree survival rates were not spatially autocorrelated (Moran's $I = -0.004$, $p = 0.592$).

Sapling Growth

Annual Diameter Growth Over Monitoring Times Since Planting and Sites. The increment in diameter varied by species, time since planting, and FMUs (Table S2). The best model (Equation 7; Table S2) had marginal and conditional r^2 values of 40.5 and 59%, respectively. Overall, the mean diameter increment increased from 7.5 ± 5.9 mm/year during P1 to 13.1 ± 11.1 mm/year during P2 ($t = 27.4$, $p < 0.001$; Table S2). This pattern was observed across most species, except for *E. angolense* and *T. scleroxylon* (Fig. 3A). During P2, *T. superba* was the fastest-growing species with a mean diameter increment of 20.6 ± 12.1 mm/year, outperforming *P. soyauxii* (12.2 ± 8.1 mm/year; Table S4). In contrast, *E. angolense*, *T. scleroxylon*, *M. altissima*, *P. elata*, and *E. suaveolens* had increments between 5 and 8 mm/year, while all the other species remained below 4.1 mm/year (Fig. 3A). Trees' diameter increments were not spatially autocorrelated (Moran's $I = -2.1 \times 10^{-4}$, $p = 0.542$). During P1, diameter increments were generally lowest at FMU L_{10051} ($t = -2.9$, $p < 0.01$; Table S3). However, they became the highest during P2 ($t = 3.3$, $p < 0.01$; Table S3), and this pattern was observed across species (Fig. S3A & S3B).

The increment in diameter did not vary with the planting mixture ($t = -0.92$, $p = 0.358$), the spacing ($t = -1.14$, $p = 0.255$) nor the log yard area ($A_{250–500m}{}^2$: $t = 0.91$, $p = 0.365$; $A_{>500m}{}^2$: $t = -0.58$, $p = 0.562$; Table S2), showing a flat and non-significant relationship (Fig. 4A).

Annual Height Growth Over Monitoring Times Since Planting and Sites. The height increment depended on species, time since planting, and FMUs (Table S6). The best model (Equation 7; Table S5) had marginal and conditional r^2 values of 45 and 56%, respectively. As with diameter, the height increases from 49.4 ± 38.6 cm/year during P1 to 102.6 ± 82 cm/year during P2 ($t = 40.2$, $p < 0.001$; Table S5). This pattern was observed across all species, except for *T. scleroxylon* (Fig. 3B). During P2, *T. superba* had the greatest height increment (153.5 ± 87.5 cm/year), surpassing *P. soyauxii* (120.7 ± 65.8 cm/year; Table S4). *Erythrophleum suaveolens*, *T. scleroxylon*, *P. elata*, *M. altissima*, and *B. toxisperma* followed, with increments of 51–80 cm/year (Fig. 3B), while all other species remained below 50 cm/year (Fig. 3). Tree height increments were not spatially autocorrelated (Moran's $I = -1.2 \times 10^{-4}$, $p = 0.552$). During P1, height increments were generally lower at FMUs L_{10051} and L_{10044} (Table S6). However, they tended to be highest during P2 at FMU L_{10051}, although this was not significant ($t = 2.0$, $p = 0.05$;

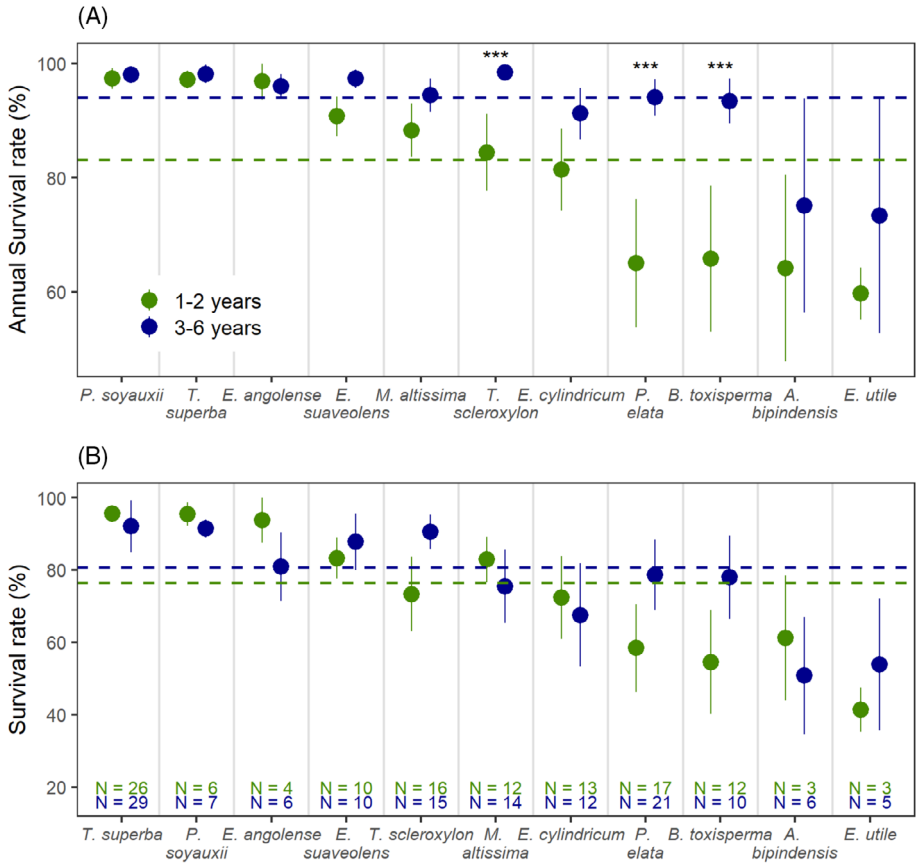

Figure 2. Species-specific mean (± SD) (A) annual survival rate (S_{an}), and (B) survival rate (S_t) during the two monitored times since planting. The colored dashed lines indicate the overall means between 1 and 2 years (P1) and 3 and 6 years (P2) after planting, respectively. *N* is the number of log yards in each monitoring time since planting. Significant differences are indicated by *** ($p < 0.001$, Tukey-adjusted post hoc test).

Table S6), and lowest at FMUs L_{10039} and L_{10044} (Table S6). This pattern was consistent across many species (Fig. S3C & S3D).

Height growth did not vary with the planting mixture ($t = -0.5$, $p = 0.621$), the spacing ($t = -0.23$, $p = 0.822$) nor the log yard area ($A_{250–500m}{}^2$: $t = 0.05$, $p = 0.958$; $A_{>500m}{}^2$: $t = -1.26$, $p = 0.212$; Table S6). Similar to diameter, height increment showed a flat, non-significant relationship with log yard area (Fig. 4B).

Effect of Log Yard Soil Ripping on Sapling Growth. When all species were included in the model, the ripping had a non-significant effect on the increases in diameter ($t = 0.47$, $p = 0.642$) or height ($t = 0.06$, $p = 0.953$) (Tables S2 & S5). However, species-specific models (Equation 8) showed some differences (Table S7; Fig. 5). The effect of soil ripping varied by species and time since planting, showing both positive and negative results, as other factors influenced sapling performance (Fig. 6). For example, *T. superba* grew faster in both diameter and height in the soil ripped log yards in both monitoring times since planting, whereas *B. toxisperma* reduced growth under the same conditions (Fig. 5).

Discussion

Tree planting can make a substantial contribution to forest enrichment (Laughlin & Clarkson 2018). To ensure the success of tree planting in the log yard, it is essential to identify factors that influence sapling performance. We investigated how species identity, tree age, soil ripping, log yard area, planting density (spacing), and species composition (monospecific vs. mixed) affected sapling survival and growth at six FMUs managed by two logging companies subjected to distinct clearing regimes.

Differences in Sapling Performance Between Species Over Time

The survival and growth of planted seedlings is expected to vary between species depending on their light requirements (Poorter et al. 2003; Makana & Thomas 2005; Schmitt et al. 2022). In our study, most species showed a strong ability to survive in the early years after planting in log yards. Only a few NPLD species (*Afzelia bipindensis* and *Entandrophragma utile*), appeared more sensitive and had relatively low annual survival rates (73–75%) during the second monitoring time since planting. On the contrary, the remaining nine species, comprising both pioneer and NPLD guilds, had annual survival rates greater than

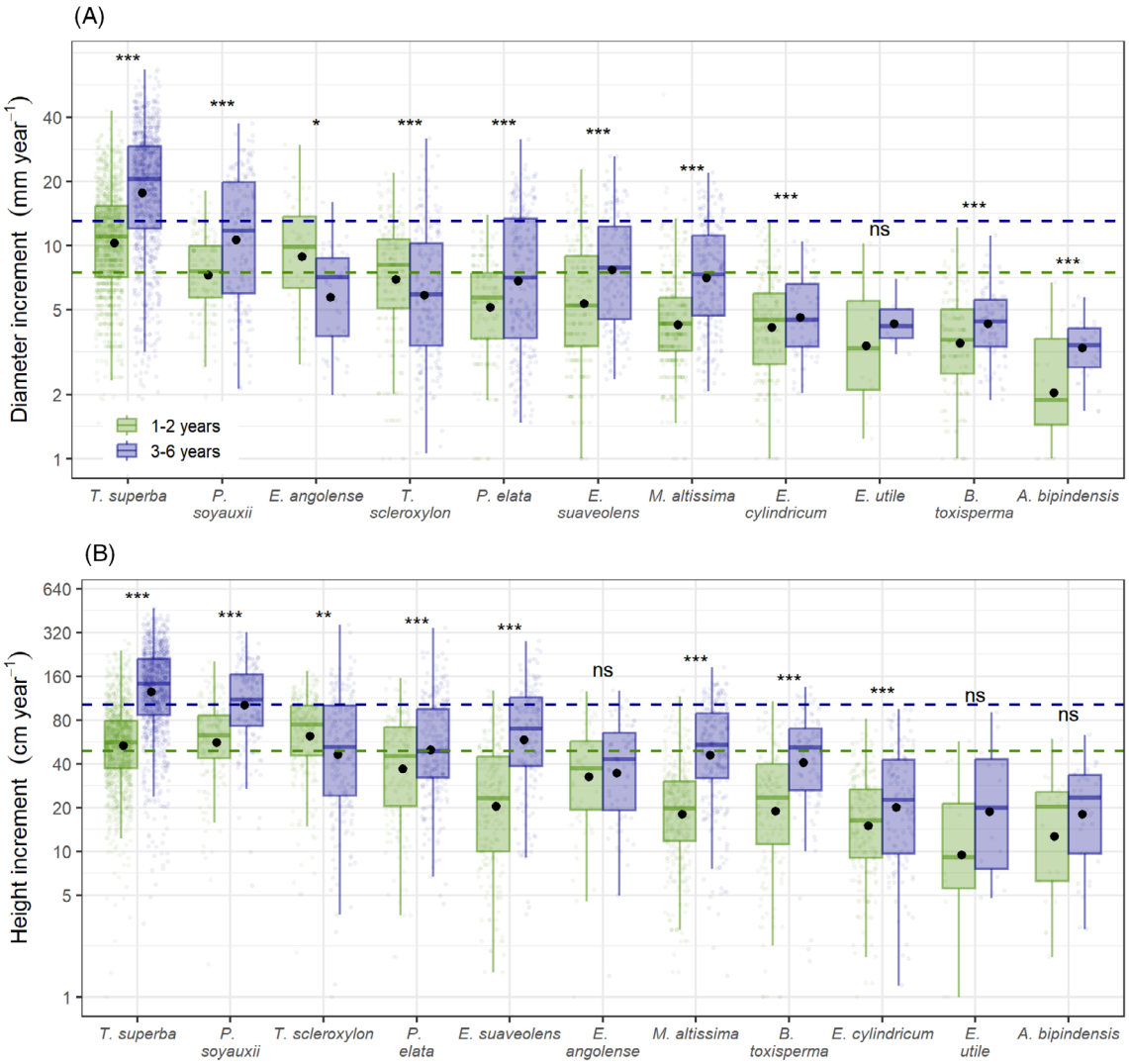

Figure 3. Diameter and height increments for each species planted in log yards between monitoring times since planting. Boxplots with distinct colors illustrate (A) diameter and (B) height increments for 1–2 years (P1) and 3–6 years (P2) since planting. The *y*-axis is plotted with a natural logarithmic scale using log(x + 1). The colored dots are the increments for each tree and time since planting. Black dots are the mean increments of each log yard at the corresponding time since planting. Line colors correspond to the mean values of the monitoring time since planting. The p values of the Tukey-adjusted post hoc test are shown above the boxplots (*$p < 0.05$; **$p < 0.01$; ***$p < 0.00B1$ and $^{ns}p \geq 0.05$).

90%. These results suggest that the species guild may not be a reliable predictor of planting success in log yards. In open environments, pioneer species are generally expected to grow faster than NPLD species. The two species with the highest growth rates, *Terminalia superba* and *Pterocarpus soyauxii*, showed similar growth patterns, despite belonging to different guilds. For example, *P. soyauxii*, which typically has diameter growth rates of 5–7 mm/year in degraded areas (Mumbanza 2015; Doucet et al. 2016; Daïnou et al. 2021), recorded 7 mm/year during P1, faster growth to 12 mm/year during P2. This finding suggests that species-specific traits within a given ecological guild may play a more important role in optimizing sapling performance than guild classification alone.

The species *E. cylindricum*, *Pericopsis elata*, *E. utile*, and *A. bipindensis* have previously been reported to have low survival rates when planted in forest gaps (Ilunga-Mulala et al. 2025a). In our study, these species had survival rates (S_t) ranging from 51% (S_{an} = 73%) to 79% (S_{an} = 94%) 3–6 years after planting in log yards. The relatively low survival observed in *E. cylindricum* plantings, especially in gap environments, has often been attributed to limited light availability (Hubert 2003; Makana & Thomas 2005).

Furthermore, the survival of planted seedlings depends on the planting environment and the methods used (Doucet et al. 2009; Laughlin & Clarkson 2018). For example, previous studies reported survival rates (S_t) of *E. angolense* in degraded areas between 34 and 56% (Hubert 2003; Doucet et al. 2016). In contrast, our study found a higher survival for *E. angolense*, reaching 81% 6 years after planting. Despite this encouraging result, *Entandrophragma* species are generally known to exhibit low

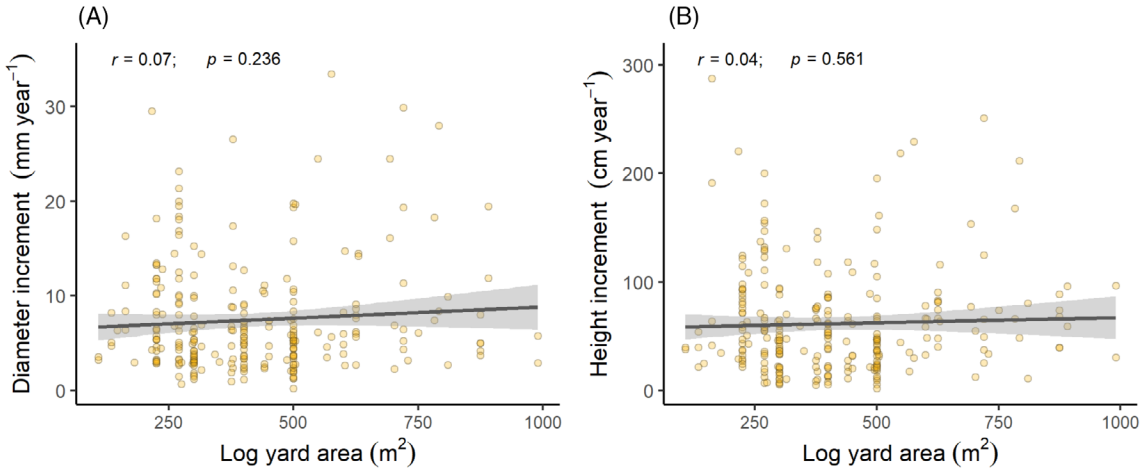

Figure 4. Relationships between log yard area and the increments in (A) diameter and (B) height. The dashed areas represent the 95% CI.

survival rates under low-light conditions, and individuals often die mainly due to fungal infections, insect attacks, or predation by small mammals (Makana & Thomas 2005; Hall 2008). It is possible that the higher survival observed in *E. angolense* in our study is due to reduced competition in log yards, where soil compaction is likely to have inhibited the growth of competing vegetation, in contrast to degraded or cleared areas.

Soil Ripping in Log Yard Has Little Influence on Species Performance

Soil ripping can affect soil structure, nutrient availability, and vegetation dynamics. Soil compaction alters physical properties by increasing bulk density and erosion risk, while reducing porosity, aeration, and water infiltration (Kozlowski 1999; Fredericksen & Pariona 2002). Additionally, removal of organic matter can lead to nutrient depletion, particularly of phosphorus,

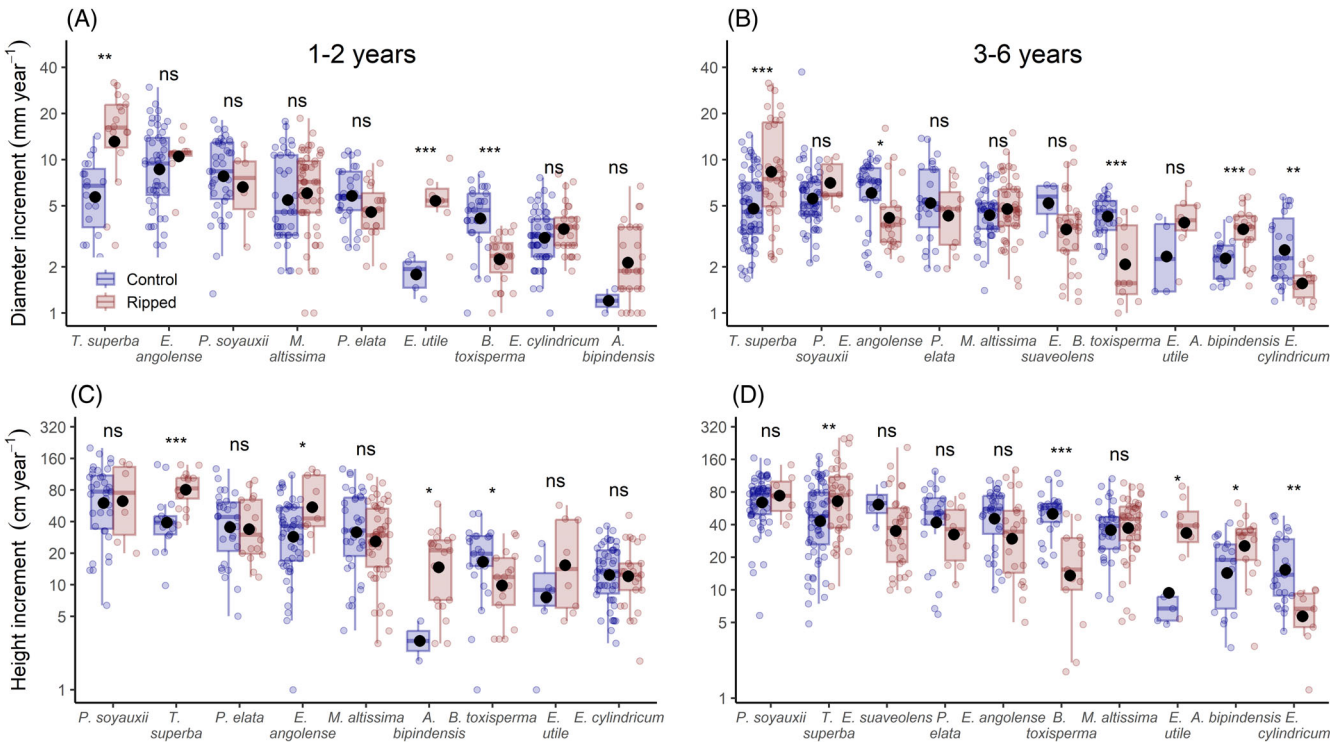

Figure 5. Effect of soil ripping on species growth in log yards. Colored boxplots show the diameter and height increments of planted trees in ripped (brown) and control (blue) log yards during (A, C) 1–2 years (P1) and (B, D) 3–6 years (P2) since planting. The *y*-axis is on a natural logarithmic scale using $\log(x + 1)$. The colored dots are individual tree increments. Black dots indicate species means per treatment and time since planting. The *p* values of the Tukey-adjusted post hoc test are shown above the boxplots ($^{*}p < 0.05$; $^{**}p < 0.01$; $^{***}p < 0.001$ and $^{ns}p \geq 0.05$).

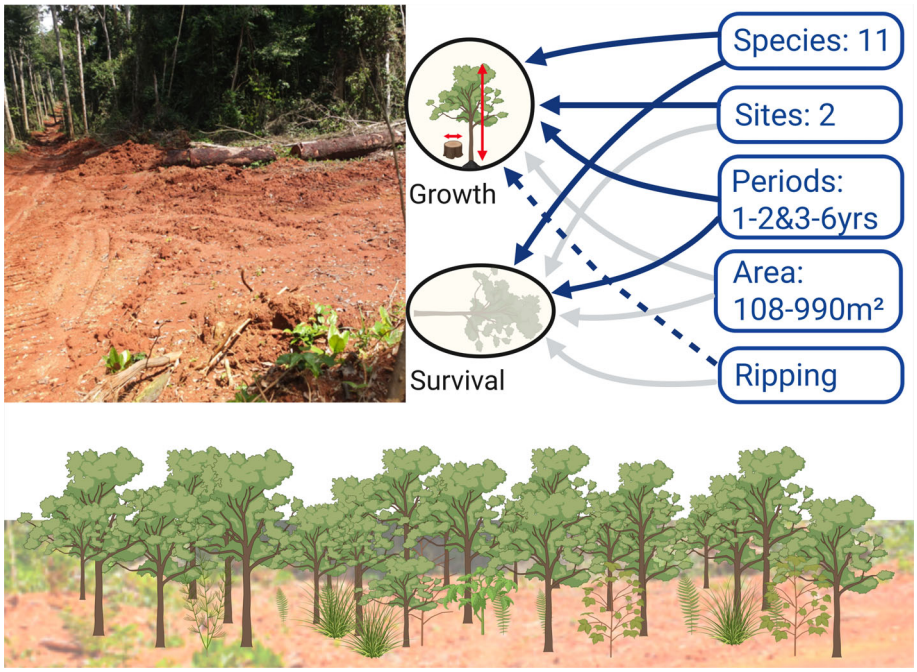

Figure 6. Conceptual diagram illustrating the effects of tested variables on tree survival, diameter and height growth. The upper-left image shows a log yard after logging, which will subsequently be planted with seedlings of high-value tree species to establish large trees, as illustrated in the image at the bottom. The study species (Table 1) were planted at Alpicam-Grumcam and Pallisco sites across six forest management units. Blue and gray arrows among presented variables indicate significant and non-significant relationships, respectively. The dashed line indicates a variable that was not significant overall, but for which some individual categories were significant. This is the case for *Terminalia superba*, which improved growth in log yards with ripped soils.

which can limit tree growth (Makana & Thomas 2005; Sanchez et al. 2006; Lyczak et al. 2021). Although litter replacement can help offset this phosphorus loss, our study did not show a consistent or general effect of soil ripping, which combines soil plowing with the replacement of litter and the organic soil layer. However, soil ripping may still be beneficial for optimizing *T. superba* growth, potentially accelerating its development and reducing the time to harvest. Under conditions of limited financial resources, avoiding soil ripping could free funds for alternative activities. Given that species respond differently to soil compaction (Kamaluddin et al. 2005; Lyczak et al. 2021), some species benefited from soil ripping, while others were negatively affected. This variability can indicate that soil conditions were not the main driver of growth, or that the limited sample size—soil ripping was only tested at two FMUs of one concession—limited our ability to detect broader patterns. However, in tropical forests, soil compaction by skidding has been associated with increased growth of commercial species (Fredericksen & Pariona 2002), possibly due to reduced competition from understorey vegetation. Compacted soils can, in some cases, provide a competitive advantage to planted trees by limiting the establishment of competing vegetation (Zhang et al. 2017).

Local Site Conditions Shaped by Soil Fertility and Competition Affect Sapling Performance

Soil fertility and microclimate are key drivers of differences in species performance and distribution, even among geographically close sites (Swaine 1996; Hall et al. 2003; Vleminckx et al. 2017). As data on soil composition were unavailable, we included "plot" as a random factor in our models and tested for spatial autocorrelation among plots to account for this source of variation. Plot-level variability was very low for survival, but it was substantial for diameter and height growth, although still lower than the residual variance. In *Entandrophragma* spp. for example, slight variations in soil fertility between plots influence the performance of trees (Hall et al. 2003, 2004). We posit that variation in species performance is also linked to differences in soil fertility. In addition, light availability influences seedling performance (Agyeman et al. 1999; Biwolé et al. 2015), and it is highly determined by local conditions influenced by competing vegetation.

Clearance through the removal of competing vegetation in plantations is expected to improve seedling survival and growth (Doucet et al. 2009; Ouédraogo et al. 2014), especially when clearing for a longer period (Ilunga-Mulala et al. 2025b). In fact, we found a slight difference in survival that could be associated with clearing regimes, with a higher survival rate at FMUs at Alpicam-Grumcam, where clearing was carried out over a longer period. This study did not thoroughly assess the influence of clearing competing vegetation on growth; however, it may have contributed to differences in tree performance. Clearing competing vegetation is costly, especially in logging concessions. At the study site, the cost of clearing competing vegetation in degraded areas has been estimated at approximately USD 1971 per hectare (Doucet et al. 2016), corresponding to the cost of clearing about 21–26 log yards, based on their average size. However, this figure is indicative and may vary

considerably depending on other factors, such as the distance between log yards.

Does the Size of the Log Yard Affect the Growth of the Saplings?

Competition is typically expected to be stronger in smaller gaps and to limit more severely seedling growth (Makana & Thomas 2005; Doucet et al. 2009). However, in this study, non-significant differences in growth were observed between log yard sizes (100–250, >250–500, and >500 m^2). The correlations between growth (both in diameter and height) and the size of the log yard were weak (approximately 7%) and non-significant. Unlike forest gaps, proximity to roads in log yards increased light availability, which may have mitigated the effects of opening size. While some species may grow more slowly in smaller gaps, this trend was not observed here. In degraded areas, diameter growth ranges from 2 to 5 mm/year for *E. cylindricum*, 5–6 mm/year for *A. bipindensis* and 4–10 mm/year for *Baillonella toxisperma* (Hubert 2003; Mumbanza 2015; Doucet et al. 2016). In comparison, their growth in log yards was slightly lower, with diameter increments between 2.5 and 4 mm/year. Despite the relatively open canopy of log yards, which allows sufficient light penetration, growth can still be constrained by competition for belowground resources, especially given the nutrient-poor soils at the study sites (Vleminckx et al. 2017). Overall, while competition for light is typically greater in forest gaps (Makana & Thomas 2005; Doucet et al. 2009), the size of log yards in this study appeared sufficient to support sapling growth, suggesting that larger log yards (>500 m^2) are not necessary to improve performance.

Planting in log yards with high-value timber species after harvesting an annual cutting area can play a crucial role in supporting the recovery of logged tropical forests. Among the 11 species studied, *T. superba* and *P. soyauxii* had the highest growth rates, with diameter increments of 20.6 and 12.2 mm/year, respectively, 3–6 years after planting.

Sustained clearing, was found to be essential to optimize sapling growth (Ilunga-Mulala et al. 2025a). On the contrary, soil ripping in log yards prior to planting did not significantly improve sapling survival and only improved growth for a few species, such as *T. superba*, *A. bipindensis*, and *E. utile*. Site-specific factors, such as soil characteristics can influence the results and should be investigated in future research. Therefore, further research is needed on more diverse sites, over longer monitoring periods, and with a wider range of species, while also assessing the implications of these practices for forest biodiversity. Such studies are necessary to better understand which species are best suited for planting in log yards.

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Supporting Information

The following information may be found in the online version of this article:

Figure S1. Number of measured trees across periods at (a) Alpicam-Grumcam and (b) Pallisco.

Figure S2. Relationships between the diameter increment at 10 cm and at 130 cm from the collar.

Figure S3. Diameter and height increments of the species planted in log yards across the forest management units (FMUs).

Table S1. Parameter estimates of the linear mixed models used to examine survival rates.

Table S2. Parameter estimates of the linear mixed models used to examine diameter increment.

Table S3. Parameter estimates of the linear mixed models (Equation 5) used to examine diameter increment across the two monitored periods P1 (1–2 years) and P2 (3–6 years).

Table S4. Diameter and height increments values for species across tested periods P1 (1–2 years) and P2 (3–6 years).

Table S5. Parameter estimates of the linear mixed models used to examine height increment.

Table S6. Parameter estimates of the linear mixed models used to examine height increment across the two monitored periods P1 (1–2 years) and P2 (3–6 years) (Equation S3).

Table S7. Parameter estimates of the generalized linear models (GLM; Equation 8) used to examine diameter and height increments across species.

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