


CONTRIBUTED PAPER

Short-term impacts of selective logging on forest elephants

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Abstract

Although forest elephants (*Loxodonta cyclotis*) are known to use logged forests, the impact of selective logging on this critically endangered species has never been well established. Considering the potential of some logged areas to serve as other effective area-based conservation measures, aligning with the Convention on Biological Diversity's 30/30 initiative, we aimed to assess the short-term impacts of logging on three population parameters: the presence, abundance, and activity of forest elephants. Combining camera traps and dung surveys before and after logging operations, we assessed the response of forest elephants in a certified timber concession in Gabon. Encouragingly, we found no negative effects of logging on the three population parameters studied. There was no discernible change in the presence of elephants after logging, and their abundance actually increased (relative abundance index from 1.56 to 2.59; p -value 0.0139) at one of the study sites. Activity patterns were also unaffected, showing sustained activity during daylight hours. We also identified the logging-associated factors that have significantly influenced forest elephant's abundance. Notably, logging intensity, as measured by the average number of trees harvested per hectare, and road density were positively correlated with the number of forest elephant observations (logging intensity effect: 0.2992, p -value 0.035; road density effect: 0.3628, p -value 0.060). As global conservation goals evolve, this research provides important insights into the coexistence of well-managed industrial activities and the conservation of endangered species, highlighting the need to include responsibly managed timber concessions in future conservation strategies. It also underscores the importance of wider adoption of sustainable practices such as low-impact logging, promoted by certification schemes such as FSC or PEFC, to secure the future of central African forests and their unique wildlife.

KEYWORDS

activity rhythm, habitat use, kilometric abundance index (KAI), logging concession, pre-post study, relative abundance index (RAI), skid trail, strip transect

1 | INTRODUCTION

The forest elephant (*Loxodonta cyclotis*) is listed as critically endangered by the IUCN (Gobush et al., 2021). Estimating its current population size remains challenging due to its elusive nature and the dense forest environment it inhabits (Hedges, 2012). Despite these challenges, specific studies have provided estimates on population trends over certain periods thanks to intensive data collection methods. For example, an alarming 62% decline in elephant population has been reported between 2002 and 2011, with three-quarters of the remaining individuals concentrated in northern Republic of Congo and Gabon (Maisels et al., 2013). Ivory poaching is the main cause of the decline of forest elephants and the greatest threat to their survival (Gobush et al., 2021). Additionally, the expansion of human activities, leading to increasing habitat loss and human–elephant conflicts, is another important threat (Gobush et al., 2021). Immediate and effective actions to combat poaching and address the underlying causes of habitat loss are needed to secure the long-term survival of forest elephants.

Although the increase in protected areas in central Africa over the past 20 years (Doumenge et al., 2021) has been a significant stride in forest elephant conservation, they cover only 14% of its potential habitat in the region (Eba'a Atyi et al., 2022), with considerable forest elephant populations living outside protected areas (Wall et al., 2021), notably in timber concessions (Scalbert et al., 2023b). In Gabon, one of the few remaining strongholds of forest elephants, it is estimated that about 65% of elephants live in forests allocated to timber production (Laguardia et al., 2021).

The timber sector represents a major economic activity in most central African countries, with more than 8 million m³ of wood produced in 2020 and generating more than 140,000 jobs across the Congo Basin (Eba'a Atyi et al., 2022). Legal logging in central Africa is generally highly selective, with only one to two trees harvested per hectare on average every 25–30 years (cutting cycle depending on the country), as defined in management plans validated by national authorities (Ruiz Pérez et al., 2005). Each year, specific areas known as annual allowable cuts (AACs) are harvested, which constitute a small portion of timber concessions. Among those production forests, 5.4 million hectares have been certified by the Forest Stewardship Council (FSC) or the Programme for the Endorsement of Forest Certification (PEFC), representing 10% of forests under selective logging (Eba'a Atyi et al., 2022). Such certification schemes attest that timber companies meet strict environmental and social standards, guaranteeing responsible logging operations, with a commitment to preserving forest

biodiversity and supporting local communities who rely on forest ecosystems. Unlike non-certified concessions, some of which are downright illegal, certified forest concessions are required to comply with national laws and international treaties and agreements to which the country is a signatory (Haurez et al., 2020). Certified logging must minimize waste and damage to other forest resources (Haurez et al., 2020). A management plan is established to ensure sustainable harvesting, with maximum harvesting rates set for each species and conservation zones where logging is prohibited (Pasquier & Forni, 2015; Tritsch et al., 2020). Law enforcement is ensured, and access to the concession is controlled to prevent any illegal activity such as poaching or illegal logging (Haurez et al., 2020).

Although reduced-impact practices might mitigate some effects, logging activity still impacts forest ecosystems in various ways. Logging alters the composition and structure of vegetation, often promoting the development of pioneer light-demanding species in logging gaps (Maicher et al., 2021). This habitat modification could lead to changes in spatial use for elephants, as they tend to favor secondary vegetation (Barnes et al., 1991). Furthermore, the creation of logging roads influences the movement of forest elephants, who use these roads extensively (Scalbert et al., 2023a). However, these roads also increase the forest's permeability to humans and their associated activities, such as hunting and slash-and-burn farming (Laurance, 2001). In addition to effects on spatial distribution, human disturbances often lead to several behavioral adaptations in wildlife, including changes in temporal activity patterns (Gaynor et al., 2018). Elephants, for instance, sometimes increase nocturnality in disturbed areas to minimize encounters with humans (Graham et al., 2009; Wrege et al., 2010). These responses and habitat changes can have significant implications for species survival and ecosystem dynamics, particularly in regions like Gabon where logging activities are prevalent (Eba'a Atyi et al., 2022). As questions arise as to whether certain areas of timber concessions will be eligible as Other Effective Area-Based Conservation Measures (OECMs) under the Convention on Biological Diversity's 30/30 goal, it is crucial to investigate more deeply the impacts of logging on iconic species such as forest elephants.

Few studies have been conducted on elephants in logged areas, mostly in the northern forests of the Republic of Congo (Clark et al., 2009; Poulsen et al., 2011; Stokes et al., 2010) and in Kibale National Park in Uganda (Omeja et al., 2014, 2016; Struhsaker et al., 1996), where forest and savanna (*Loxodonta africana*) elephants coexist (we therefore use the term elephant to refer to both species). These studies showed that elephants extensively

used logged areas, with indices of abundance 3–6 times higher in logged areas than in nearby protected areas or unlogged forests (Poulsen et al., 2011; Struhsaker et al., 1996), although the proximity of a protected area or unlogged forest appeared to be critical (Clark et al., 2009; Merz, 1981; Stokes et al., 2010). It also appeared that elephant abundance continuously increased after logging, although in some areas it started to slowly decrease after 15 or 20 years (but still reaching higher levels than before logging; Clark et al., 2009; Omeja et al., 2016; Stokes et al., 2010). While different methods and tools are available today for assessing species habitat preferences, all these studies have relied on dung surveys. However, this technique may be subject to numerous biases. First, it is highly dependent on the rate of dung decomposition. Yet, environmental conditions cause spatial and temporal variations in this rate, making comparisons of results challenging (Nchanji & Plumptre, 2001). Although estimates may include decomposition rates if available, determining them beforehand requires additional effort that is not always feasible prior to each study. Second, observer bias may also be important as not all observers have the same ability to detect dung, and an observer's performance can fluctuate over time due to varying weather conditions or fatigue levels (Olivier et al., 2009). Today, technological advances offer alternative methods for conducting wildlife monitoring, such as using camera traps, acoustic sensors, or genetic material (Houngbégnon et al., 2020; Laguardia et al., 2021; Wrege et al., 2012). Despite the booming use of camera traps (CTs) over the last three decades, we only found one study that has tested differences in both presence and abundance of elephants among different land uses (including timber concessions) using CTs (Djoko et al., 2022). This study, conducted in Cameroon, found fewer forest elephant detections in a timber concession than in community forests, but as many as in a protected area (Djoko et al., 2022). All the studies mentioned above have employed a space-for-time substitution approach, where they compared different land uses or areas with varying logging histories at the same moment. However, to our knowledge, no study has examined the impact of logging by comparing the same site before and after logging. Such an approach would help reduce bias linked to the inherent natural heterogeneity that might exist among distinct areas.

In this study, we aimed to assess the short-term impact of selective logging on forest elephants in a certified timber concession in eastern Gabon. From December 2019 to May 2020 and from January 2022 to June 2022, we used CTs to assess the effects of logging on forest elephant presence, abundance, and activity, as well as to identify whether logging-associated factors (i.e., logging intensity, road and skid trail network) have a significant

influence on forest elephant populations. We also conducted dung count on line transects to compare our results with our CTs assessment as well as other studies, this method being the most used in previous analyses. Prior research has indicated that elephants make heavy use of secondary forests and logging roads and that they can become more nocturnal in areas disturbed by human activities (Barnes et al., 1991; Scalbert et al., 2023a; Wrege et al., 2010). The following hypotheses are therefore proposed:

1. The presence and abundance of forest elephants do not decrease after logging.
2. Forest elephants become more nocturnal after logging.
3. The intensity of logging and the development of logging roads and tracks have a positive influence on forest elephant abundance.

By comparing the same site before (2020) and after logging (2022), our study provides new insights into the short-term impacts of logging on forest elephants.

2 | METHODS

2.1 | Study area

This study took place in Gabon, which is home to the majority of forest elephants, with an estimated population of 95,110 (95% CI 58,872–131,349) individuals (Laguardia et al., 2021). While 13% of Gabon's rainforest is dedicated to conservation, selective logging accounts for 60% of forest areas, making timber concessions the prevailing land use within its range in Gabon (Eba'a Atyi et al., 2022). The study area is encompassed within the concession managed by Precious Woods-CEB, which covers 600,000 ha and produces 240,000 m³ of timber per year (Martinoli, 2023). It has been FSC-certified since 2008 and was the first PEFC-certified concession in central Africa in 2018. Two distinct vegetation types occur within the concession: an old secondary evergreen moist forest dominated by *Aucoumea klaineana*, *Scyphocephalum mannii*, and *Julbernardia pellegriniana*, which is the predominant component, and a forest-savanna mosaic, which occupies about 15% of the total area (Terea, 2016). This mosaic is rich in young *A. klaineana* populations that colonize the savannas, which are typically kept open by human-caused fires (Terea, 2016). Although there is no estimate of the number of forest elephants living in the concession, it is within this forest-savanna mosaic that they are most abundant and where three adjacent annual allowable cuts (AACs) covering 4400 ha were

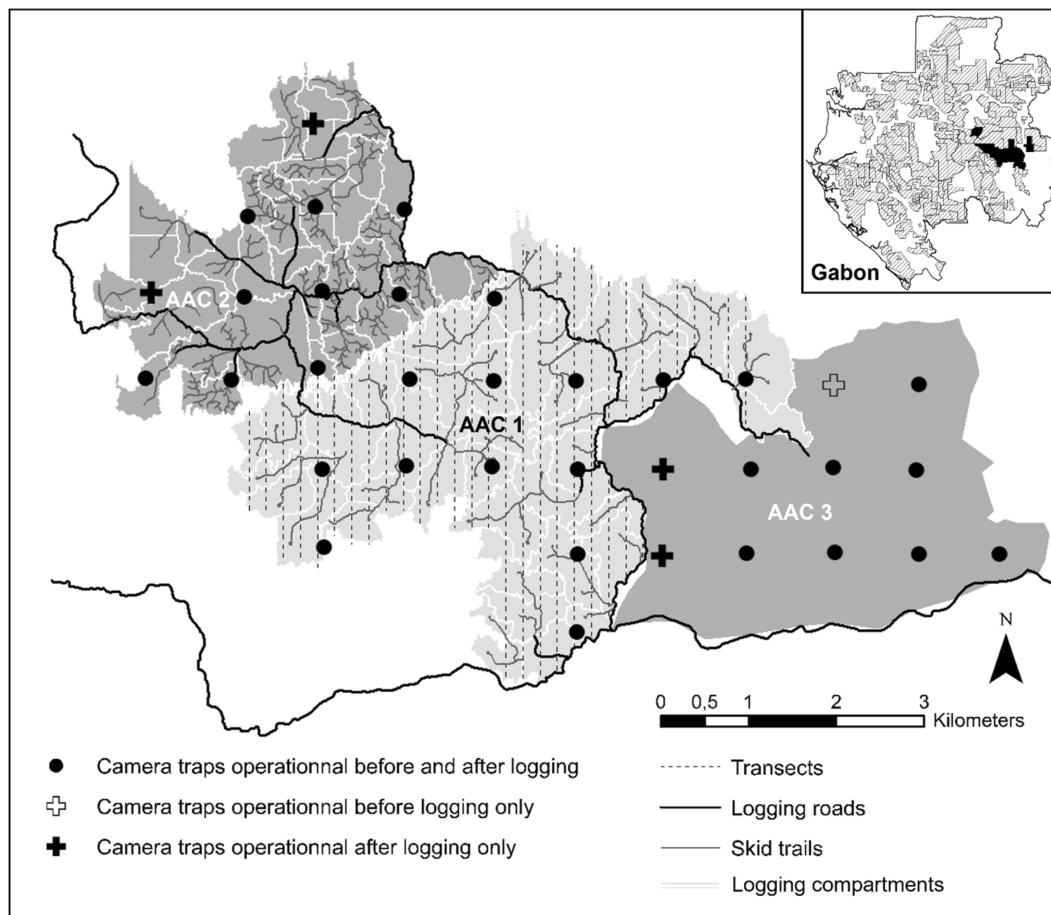


FIGURE 1 Study area. Location of camera traps and line transects used for forest elephant surveys in 2020 (pre-logging) and 2022 (post-logging). AAC: annual allowable cut. The inset map illustrates the forest license boundaries in Gabon (hatched; Precious Woods-CEB concession in black). Data on forest license boundaries are from Global Forest Watch.

surveyed in this study (Figure 1). These AACs are bordered by other AACs within the same concession. Most of the concession is surrounded by adjacent timber concessions, except to the south and east, which are characterized by open environments (forest-savanna mosaic) that are not designated for logging.

Forest elephants were inventoried in AAC 1, AAC 2, and AAC 3 in 2020 (our “before logging” treatment, although they all have been logged by the same company 25 years ago) and in 2022 (“after logging” treatment). Logging periods and intensity, as well as CT and dung survey periods, are shown for each AAC in Table 1. Although AAC 3 remained unlogged throughout the entire study period, its proximity with the two other AACs may have induced certain indirect side effects of logging and it is therefore also included in the “after logging” treatment when surveyed in 2022. Data from each AAC were analyzed together or separately, depending on the analysis performed (Table 2). The seasons have a strong influence on the movement of forest elephants, as they often undertake long

migrations to reach areas with abundant fruiting (White, 1994). Migrations are not documented within our study area. However, to minimize seasonality bias, the two surveys were conducted during the same season in 2020 and 2022, specifically from the beginning of the short dry season to the end of the short rainy season.

2.2 | Camera trap survey

For both 2020 and 2022 surveys, 35 CTs (Bolyguard SG 2060X, Boly, Victoriaville, QC, Canada) were deployed in the 3 AACs following a systematic grid with a 1-km spacing (Figure 1). Four and one cameras were retrieved from the dataset in 2020 and 2022, respectively, due to malfunction, resulting in 9–13 CTs in each AAC in 2020 and 10–13 CTs in each AAC in 2022. The CTs were attached to a tree at a height of 40–50 cm (Fonteyn et al., 2021) and oriented toward an elephant trail. They were set to record a 5-s video when triggered.

TABLE 1 Logging period, logging intensity, and periods of camera trap (CT) and dung surveys in the three annual allowable cuts (AAC).

| | AAC 1 | AAC 2 | AAC 3 |
|--------------------------------|---|--|----------------------|
| Logging period | Nov 2020 to Apr 2021 | Sep 2021 to Dec 2021 | No logging |
| Logging intensity ^a | 2.52 (0.48) | 2.67 (0.37) | No logging |
| 1st CT survey | Dec 2019 to Apr 2020 | Dec 2019 to Apr 2020 | Jan 2020 to Apr 2020 |
| 2nd CT survey | Jan 2022 to Jun 2022 (9 months after logging) | Jan 2022 to Jun 2022 (3–6 weeks after logging) | Jan 2022 to Jun 2022 |
| 1st dung survey | Jan 2020 to Feb 2020 | No dung survey | No dung survey |
| 2nd dung survey | Feb 2022 to Mar 2022 | No dung survey | No dung survey |

^aAverage number of trees harvested per hectare (SD).

All videos were screened using Timelapse 2.0 v2.2.3.8 (Greenberg, 2021). Two observations of forest elephants were considered independent if separated by a minimum of 30 min (Meek et al., 2014). The number of days of active functioning (camera.days) has been computed for each camera. As it is not uncommon for animals to turn the camera, batteries to fail, or vegetation to grow and obstruct the camera's field of view, the camera.days number was adjusted to reflect the last day that the camera was active, facing the monitored track, and species identification was possible.

The number of independent elephant detections and the number of camera.days were used to calculate a detection rate, called the relative abundance index (RAI) and calculated as follow:

$$\text{RAI} = \frac{\text{Number of independent events}}{\text{Number of camera.days}} \times 100.$$

Although the use of this index is controversial because it is highly dependent on detection probability (Martin-Garcia et al., 2022; Palmer et al., 2018), its use as a proxy for abundance in our study is justified due to the rigorous standardization of our protocol. Cameras were placed in the exact same locations when possible during both survey periods, and both surveys were conducted during the same season. This standardization helps to minimize variability in detection and allows for a more reliable comparison between years.

2.3 | Occupancy

Observing an animal on a camera trap video (i.e., the formal proof of its presence) depends on two parameters: the probability that the animal occupies the area (Ψ), and the probability of its detection by the camera (p) (MacKenzie et al., 2017). Therefore, observing a species requires both its presence and its detection, while

absence of detection could mean either actual absence or failure to be detected by the camera (i.e., a false negative). Occupancy models account for these false negatives and can be used to estimate occupancy rates (i.e., the proportion of sites that are occupied by the studied species; MacKenzie et al., 2017). As Davis et al. (2022), we interpreted Ψ as the probability of elephant presence within a site instead of the proportion of occupied sites. This is due to the large home range of forest elephants, which can span several hundred km² (Mills et al., 2018), making it challenging to satisfy the assumptions of occupancy modeling that assume a closed site not subject to changes in occupancy (MacKenzie et al., 2017).

First, we created the species detection history (i.e., the sequence of detections (1) and non-detections (0) for each CT; MacKenzie et al., 2017) dividing each camera survey period into 9-day sampling occasions. This time interval was defined to keep a maximum amount of data while minimizing the occurrence of missing values in the detection history matrix (i.e., sampling occasion where a camera trap was not operational for at least 1 day; MacKenzie et al., 2017; Tempel & Gutiérrez, 2013). Additionally, this interval closely aligns with those employed in other comparable surveys (Davis et al., 2022; Fonteyn et al., 2021).

Then, we modeled Ψ and p using several basic static occupancy models and covariates (MacKenzie et al., 2017). Ψ was modeled as a function of year and AAC and p was modeled in response to year and survey effort (i.e., number of camera.days). To select the relevant covariates (among year, AAC and survey effort), we fitted the models with different covariate sets (Table 2) and selected either the model with the lowest AIC or the most parsimonious model when the AIC difference was less than 2 (Burnham & Anderson, 2004).

Analyses were performed with R software (version 4.2.2; R Core Development Team, 2022) using the packages camtrapR (version 2.2.0; Niedballa et al., 2016) and RPresence (version 2.13.44; MacKenzie & Hines, 2023).

TABLE 2 Models tested for the occupancy, abundance, activity, and logging-associated factors analysis, using camera traps data.

| Type of model | Model | Variables to explain and fixed effects ^a | AIC | ΔAIC | Weight ^b |
|--|-----------------|--|--|-------|---------------------|
| Basic static occupancy models ^c | Occupancy | Data from the 3 AACs, both year | | | |
| | Occ 1 | Ψ (Year + AAC) p () | 588.75 | | 0.34 |
| | Occ 2 | Ψ (AAC) p () | 590.05 | 1.30 | 0.18 |
| | Occ 3 | Ψ (Year + AAC) p (effort) | 590.70 | 1.94 | 0.13 |
| | Occ 4 | Ψ (Year + AAC) p (Year) | 590.75 | 2.00 | 0.13 |
| | Occ 5 | Ψ (AAC) p (Year) | 591.69 | 2.95 | 0.08 |
| | Occ 6 | Ψ (AAC) p (effort) | 591.97 | 3.22 | 0.07 |
| | Occ 7 | Ψ (Year + AAC) p (Year + effort) | 592.69 | 3.94 | 0.05 |
| | Occ 8 | Ψ (AAC) p (Year + effort) | 593.64 | 4.89 | 0.03 |
| | Null model | Ψ () p () | 601.04 | 12.29 | 0.00 |
| Generalized linear mixed models ^d | Abundance | Data from the 3 AACs, both year | Dung count survey to confirm results from AAC1 | | |
| | GLMM 1 | $n \sim \text{AAC} + \text{Year} + \text{AAC} \times \text{Year}$ | | | |
| | Activity | Data from the 3 AACs, both year | | | |
| | GLMM 2 | $n \sim \text{Time} + \text{AAC} + \text{Year} + \text{Time} \times \text{Year}$ | | | |
| | Logging factors | Data from AACs 1 and 2, after logging | | | |
| | GLMM 3 | $n \sim \text{AAC} + \text{intens_log} + \text{dist_road} + \text{dist_skid} + \text{dens_road} + \text{dens_skid}$ | | | |

^a Ψ : probability of site used, p: detection probability, Year: year of the survey, AAC: annual allowable cut, effort: number of camera.days, n : number of forest elephant observations, Time: time of the day (day or night), intens_log: intensity of logging, dist_road: distance from the nearest road, dist_skid: distance from the nearest skid trail, dens_road: density of roads, and dens_skid: density of skid trails.

^bAkaike weight, or the probability that each model is the best model among those considered.

^cFor the occupancy analysis, only the null model and models with a ΔAIC less than 5 are presented.

^dThe camera trap ID was used as random effect in each general linear mixed model.

2.4 | Abundance

To assess the impact of logging on forest elephant abundance across the study area, we fitted a generalized linear mixed model (GLMM) including the year and AAC as fixed effects, and CT as a random effect (Table 2). The model was implemented using the “glmer” function from the lme4 package (version 1.1–34; Bates et al., 2015) and is hereafter referred to as GLMM 1. In this model, the number of independent elephant detections, following a Poisson distribution, was used as the response variable. To address variations in survey effort, considering that not all CTs were operational during the same duration, the model was fitted using an offset parameter representing the log-transformed camera.days. This approach assigned greater weight to cameras with more camera.days, ensuring a fair representation of their contribution to the analysis.

A comparison of means test with a Tukey-adjusted p -value was performed to specifically test differences between the means of factor levels or interactions found to have a significant effect in the model. This test was performed using the “emmeans” function from the eponymous R package (version 1.8.8; Lenth, 2023). Tests were considered statistically significant at p values < 0.05 .

2.5 | Activity

To investigate changes in forest elephant activity before and after logging, we fitted the GLMM 2 by adding a fixed time factor to the GLMM 1 with two levels: day and night (Table 2). These levels were defined as observations recorded between 6:00 a.m. and 6:00 p.m. and outside these hours, respectively.

Additionally, we examined the temporal overlap of forest elephant detections between 2020 (pre-logging) and 2022 (post-logging) using the methodology outlined by Houngbégnon et al. (2020). Specifically, we assessed the extent to which detections from both years coincided within the same time intervals. To quantify the overlap in activity patterns before and after logging, we utilized the overlap coefficient ($\hat{\Delta}_1$). This coefficient ranges from 0 (no overlap) to 1 (identical activity patterns) and was calculated as follows:

$$\hat{\Delta}_1 = \int_0^1 \min\{\hat{f}(t), \hat{g}(t)\} dt,$$

where $\hat{f}(t)$ is the probability density function before logging and $\hat{g}(t)$ is the probability density function after

logging. In simpler terms, $\hat{\Delta}_1$ is the area under the curve created by the minimum of the two density functions (Kernel density) at each point in time (Houngbégnon et al., 2020).

Furthermore, to explore potential distributional similarities, we employed a Watson test to determine whether the two sets of circular observations shared the same underlying distribution.

2.6 | Effects of logging-associated factors

Logging operations within an AAC are organized into logging compartments, based on various geographical features such as roads, rivers, and ridges, and harvested one by one during approximately 1 month. Data on logging intensity (the average number of trees harvested per hectare), road density, and skid trail density (the average number of meters of road or skid trail per hectare) were only available at this compartment scale. These variables were attributed to CTs depending on the compartment within which they were installed, along with two other variables derived from cartographic data, namely distance to the nearest road and distance to the nearest skid trail. They were scaled and incorporated into a third model (GLMM 3) to identify which component of logging activity could influence forest elephant abundance in the study area (Table 2). We ran this analysis using only CT data from AACs 1 and 2 in 2022 (after they were logged). The surface area of the compartments where the CTs were located varied from 12.05 to 62.86 ha, with a mean of 32.65 and a median of 32.16. The amount of data was too small to test the interactions, so we could not test whether responses to these factors varied between the two logged AACs.

2.7 | Dung count survey

Dung counts were carried out simultaneously with the CT surveys on 73 km of linear transects in one of the AACs (AAC 1), taking advantage of the existing forest inventory transects network. These are corridors of 1 m wide and spaced 200 m apart that cross the AACs in a north–south direction (Figure 1). They were manually opened by clearing all vegetation up to a height of 2 m using a machete to survey harvestable trees before logging. Transects were opened 2 months before the first dung count survey and were not reopened before the second one but were walked by following the old marks of machete or the GPS tracks from the first survey if no mark was discernible.

All dung seen from the transect was recorded. For each dung observation, the following information was

registered: GPS coordinates, perpendicular distance to the transect, and decay stage (according to Hedges & Lawson, 2006): five decay stages are defined, with dung from stages S1, S2, and S3 considered as “still visible” and from stages S4 and S5 considered as “decayed” (Hedges & Lawson, 2006). The analyses were run twice, the first time including all dung piles and the second time including only decay stages S1–S3 as stages S4 and S5 are generally not considered for elephant density estimates (CITES MIKE, 2012). The kilometer abundance index (KAI) was computed for each survey by dividing the observed number of dung piles along the transects by the total length of the transects (73 km).

A strip transect analysis was also conducted (Hedges, 2012), keeping only observations recorded within 0.5 m of the transect, on a total width of 1 m that allows to assume perfect detection of dung piles within this strip. The dung pile density was calculated as follows (Hedges, 2012):

$$D = \frac{N}{2wL} \times 10^4,$$

where D = dung pile density (number of dung piles per hectare), N = total number of observations (within a w distance of the transect), w = maximal distance from the transect to record a dung observation (0.5 m), and L = total length of all transects (73 km).

3 | RESULTS

3.1 | Camera trap survey

The camera trap surveys conducted before and after logging yielded a total of 2807 and 4531 camera.days, respectively. During the 2020 survey (before logging), 40 independent events of forest elephants were recorded across all AACs, resulting in an overall RAI of 1.43. After logging, the number of independent events increased to 88 across the three AACs, leading to a higher overall RAI of 1.94.

3.2 | Occupancy

The best fitting occupancy model (Occ 2) suggested that the probability of forest elephant presence (Ψ) varied across AACs but was not influenced by the year of inventory. This indicates that logging had no impact on the probability of occupation of a sampling site. The detection probability (p) remained constant at 0.17 (95% CI: [0.14–0.20]).

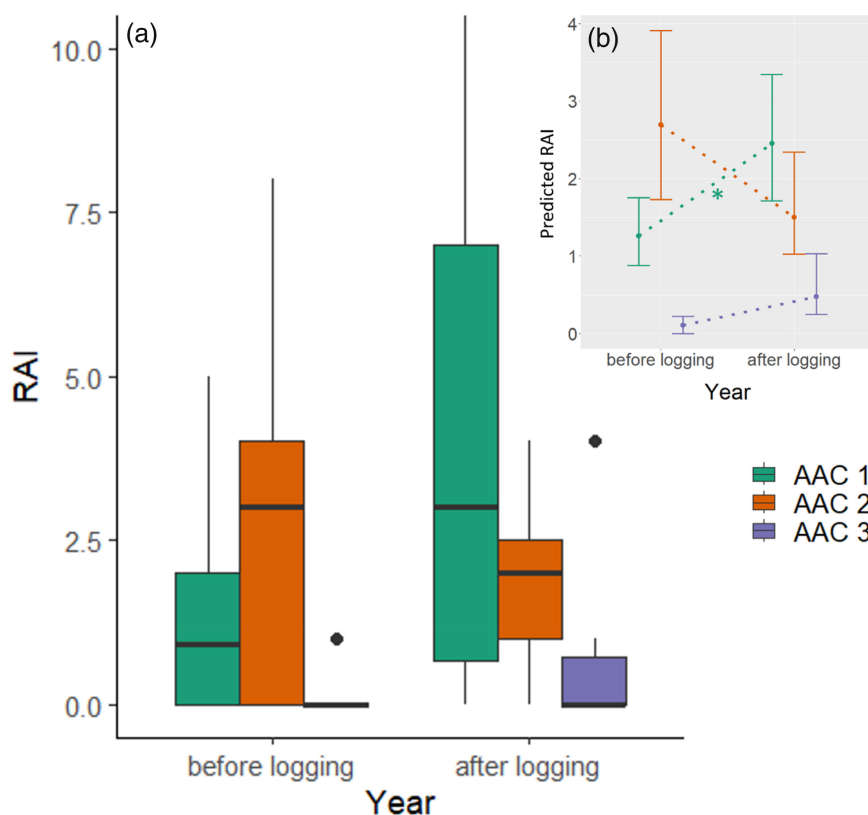


FIGURE 2 Changes in the elephant relative abundance index (RAI) between the two surveys (2020: before logging and 2022: after logging) for the three annual allowable cuts (AACs) studied (AAC 1, AAC 2, and AAC 3). (a) Boxplot illustrating the study data. The bold line represents the median, the lower and upper limits of the boxes are the first and third quartiles, respectively, the vertical lines represent the minimum and maximum values, and the points are outliers. (b) Model predictions of GLMM 1 ($n \sim \text{AAC} + \text{Year} + \text{AAC} \times \text{Year}$, where n is the number of forest elephant observations). The point represents the mean predicted value and the lines indicate the 95% bootstrap confidence interval. Dashed lines illustrate the change in the number of elephant observations after logging. The asterisk (*) indicates a significant change between the two surveys, observed only for AAC 1 (p -value = 0.0139).

3.3 | Abundance

GLMM 1 revealed a significant interaction between the AAC and year factors (p -value = 0.008), indicating that the variation in RAI between the two surveys differed across AACs. There were no significant differences in the RAI of elephants observed for AACs 2 (recently logged in 2022) and 3 (not yet logged in 2022) between the two surveys. However, in AAC 1, which was logged 9 months prior to the second survey, there was a notable increase in elephant abundance following logging (Figure 2).

3.4 | Activity

The GLMM 2 showed that forest elephants were more active during the day than at night in both surveys (p -value of time effect = 0.009). No effect of logging on the activity pattern of forest elephants could be detected (p -value time \times year = 0.29).

The overlap analysis (Figure S1, Supporting Information) yielded similar results, indicating no significant difference in activity rates between the 2 years, with an overlap coefficient of 0.88. Similarly, the Watson test showed no difference in the distribution of the two sets of circular observations ($p > 0.1$).

3.5 | Effect of logging-associated factors

The GLMM 3 showed a significant positive effect of logging intensity and, to a lesser extent, a positive effect of road density on forest elephant abundance (Table 3). The influence of the distance to the nearest road and skid trail as well as skid trail density was not significant.

3.6 | Dung count survey

Including all decay stages, the number of dung piles observed during the 2020 and 2022 dung count surveys in AAC 1 was 294 and 780, respectively, representing a 2.65-fold increase after logging. Considering only decay stages S1–S3 (still visible), the increase was slightly smaller (1.79-fold), with 91 dung piles recorded before logging, and 163 after logging. For the strip transect analysis, we kept the dung observed at a distance of max 0.5 m from the transect, that is 137 dung piles (31 from stages S1–S3) before logging and 322 (60 from stages S1–S3) after logging.

Both KAI and dung pile density were higher after logging than before, whether all decay stages were considered or only S1–S3 (still visible). Considering all decay stages, the KAI increased from 4.03 before logging to 10.69 after logging, accompanied by a corresponding

TABLE 3 Estimates of the effects of five logging-associated factors on forest elephant abundance.

| Logging-associated factors | Min | Max | Median | Mean (SD) | Effect | p-value |
|--|-------|---------|--------|-----------------|---------|---------|
| Logging intensity (harvested tree/ha) | 0.97 | 2.98 | 2.74 | 2.57 (0.50) | 0.2992 | 0.035* |
| Distance to the nearest road (m) | 22.91 | 1488.52 | 286.30 | 366.23 (331.24) | −0.1518 | 0.634 |
| Distance to the nearest skid trail (m) | 6.29 | 291.07 | 105.10 | 111.11 (86.23) | 0.1935 | 0.172 |
| Road density (m/ha) | 0 | 50.92 | 4.07 | 14.83 (18.51) | 0.3628 | 0.060** |
| Skid trail density (m/ha) | 0 | 206.49 | 27.97 | 37.08 (38.96) | −0.3918 | 0.151 |

Note: For logged annual allowable cuts only (AAC 1 and 2).

* $p < 0.05$. ** $p < 0.1$.

increase in dung pile density from 18.77 dung piles per hectare to 44.11 dung piles per hectare. Considering only the S1–S3 decay stages (still visible), the KAI increased from 1.25 to 2.23, while the dung pile density increased from 4.25 dung piles per hectare to 8.22 dung piles per hectare.

4 | DISCUSSION

Our study demonstrates that responsible reduced-impact logging did not lead to a reduction in the local occupancy and abundance of forest elephants, nor did it alter their activity patterns. Using camera trap surveys conducted before and after selective logging operations in the same area, we provided new insights into the short-term impacts of logging on forest elephants, moving beyond the limitations of the traditional space-for-time substitution approach (where different land uses or areas with different logging histories are simultaneously surveyed; Lhoest et al., 2020). Our findings confirm that elephants remained present and equally abundant in the study area post-logging as they were pre-logging, validating our initial hypothesis. In fact, in one specific site, AAC 1, which was logged 9 months prior to the second survey period, elephants' detection rates even increased—a trend further supported by dung survey data. Additionally, there was no discernible spillover effect into the unlogged AAC 3 area which could have act as a refugee from logging disturbances (Blitzer et al., 2012).

Although it is reassuring that the results of the CT and dung surveys converge, it is important to note that they are based solely on abundance indices, which are highly open to criticism. In fact, the RAI estimated from the CT data reflects not only the abundance of elephants, but also their detectability, which is directly related to other factors such as the environment in which the camera is located or the way the elephants move (Palmer et al., 2018). While logging could affect both factors, the analysis of occupancy revealed that logging did not influence the detection probability. This finding considerably strengthened our

confidence in the use of RAI. In addition to being dependent on dung detection, abundance indices derived from dung surveys depend on two additional parameters: the dung production and decay rates, which are site- and time-specific (Nchanji & Plumptre, 2001). These rates can be used in models to estimate elephant densities. Since the values of these rates are not available for our study site, and since we do not know how they may have changed due to logging, we chose to remain cautious and refrain from predicting forest elephant densities using the known rates from nearby sites, as such estimates could be highly biased. By increasing canopy openness, logging could affect the decomposition rates of dung in various ways. On the one hand, greater exposure to sunlight could dry out the dung, causing it to take longer to decompose. On the other hand, increased exposure to rain could wash it away more quickly. Since the impacts of logging on these rates have never been studied and could vary, we preferred not to make any assumptions about elephant densities.

The observed rise in forest elephant abundance 9 months after logging aligns with hypotheses put forth by other authors and suggests that it is the abundance of vegetation growing in secondary forests that attracts elephants (Merz, 1981; Stokes et al., 2010; Struhsaker et al., 1996). Vegetation recolonization after logging begins quickly, with pioneer species such as herbaceous, and fast-growing shrubs and trees establishing themselves within the first few months (pers. obs.). These early successional species are known to create dense undergrowth, which provides abundant forage for elephants (Struhsaker et al., 1996). Therefore, the absence of such an increase in forest elephant abundance in AAC 2, which was logged about a month before the second survey, may be attributed to insufficient time for vegetation to colonize the disturbed areas (Haurez et al., 2016). This highlights the importance of considering the time scale when evaluating the impacts of logging on forest elephant populations.

Forest elephants are generally more active at night (Wrege et al., 2012), and even more so in areas disturbed by human activity (Graham et al., 2009; Wrege

et al., 2010). Hunting particularly impacts the behavior of wildlife populations and may consequently alter their activity patterns (Croes et al., 2007). Human-induced changes in activity patterns can cause physiological stress, impacting reproduction, survival, and population growth (Bourgoin et al., 2011). In our study area, where hunting is minimal or non-existent as we are far from villages, elephants showed higher activity levels during the day, both before and after logging, supporting no effect of logging on their activity and refuting our second hypothesis. However, this may be an artifact related to the fact that elephants appear to prefer to use roads in recently logged forests at night (Scalbert et al., 2023a) and may have been less detectable by our cameras, which were placed in the undergrowth on an elephant track. It is also possible that only the presence of humans and the high levels of noise generated by logging operations, especially during the day, influence elephant activity (Buij et al., 2006). This effect would then be limited to the logging period and would have required continuous data collection throughout this timeframe to capture it effectively.

Logging intensity was identified as the most influential logging-associated factor and had a significant positive correlation with forest elephant abundance. This was also observed by Struhsaker et al. (1996) and supports the hypothesis that the creation of open areas with thriving vegetation attracts forest elephants. Road density also appeared to have a positive effect, although to a lesser extent (p -value = 0.060). This is not surprising as forest elephants are known to use roads extensively in timber concessions where human access to roads is controlled (Scalbert et al., 2023a).

This study was conducted in a well-managed logging concession, operating under the Forest Stewardship Council (FSC) and the Programme for the Endorsement of Forest Certification (PEFC) standards which are characterized by reduced-impact logging practices, robust law enforcement and strict control of road access. This high standard of management suggests caution when generalizing our findings to other timber concessions, as recently highlighted by Zwerts et al. (2024) who found a higher abundance of forest elephants in FSC-certified logging concessions compared to non-FSC concessions.

The average home range of forest elephants is 713 km², significantly larger than our study area (Mills et al., 2018). While elephants did not avoid our site post-logging, expanding surveys over larger areas and including distant control sites would provide clearer insights into potential spill-over and refuge dynamics.

Additionally, our study focused on presence, abundance, and activity, but further research should examine other effects of logging. These include potential stress

impacts on forest elephants and their reproductive performance (Munshi-south et al., 2008; Tang et al., 2020), as well as changes in group structure, which could be investigated through genetic analysis or camera trap images.

Lastly, a long-term survey covering multiple seasons and the post-logging period would improve understanding of logging's effects on forest dynamics and food availability for elephants. While reduced fruit tree availability has been linked to weight loss in elephants elsewhere (Bush et al., 2020), this is likely minimal in our area, where 99% of harvested trees are dispersed by wind, and two species that are important for forest elephants, *Bailonella toxisperma* and *Tieghemella africana*, are protected (Meunier et al., 2015; Precious Woods, 2021).

By highlighting that responsible logging practices can support consistent elephant presence, abundance, and activity patterns in logged areas, our findings demonstrate the value of integrating reduced-impact logging into forest elephant conservation strategies. Responsibly managed logging permits help maintain forest cover, thereby preventing agricultural conversion, which poses a much greater threat to forest elephants. Although we do not intend to romanticize logging practices, it is crucial to balance conservation and economic priorities when addressing global challenges. Therefore, considering the conservation potential of certified and responsibly managed timber concessions is essential for global biodiversity conservation strategies.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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