Increasing pressure on protected areas in the DR Congo over the last 20 years

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Abstract Protected Areas (PAs) are an essential element in strategies to conserve the Congo Basin rainforest. In the past, PAs in the Democratic Republic of Congo (DRC) have been able to maintain most of their forest cover, primarily due to their remote locations. As this passive protection is starting to fade with the progression of the deforestation frontier, the future role of PAs in conserving the Congo Basin rainforest is uncertain. Using a geographic regression discontinuity design and a novel discontinuity-based typology to categorise protection by its spatio-temporal context, the study investigates dynamics of forest loss at PA boundaries and assesses their potential to withstand deforestation as anthropogenic pressures rise. On average, PAs have deforestation sprawling in at 18% of their boundaries in 2022, and only 9% have been able to actively contain it. In expectation of a rapidly growing population and a renewed interest in the region's resources from industrial actors, more evidence on what works in conserving the rainforest of the DRC without compromising local livelihoods is needed.

Keywords Protected areas, Democratic Republic of Congo, deforestation, resource frontiers.

1 Introduction

The Congo Basin rainforest is a biodiversity hotspot, has important regulatory functions for the regional and global climate and directly provides for the livelihoods of millions of people (Eba'a Atyi et al. 2022b). Yet its extent continues to decrease, mostly driven by the expansion of small-scale agriculture under a growing population (Tyukavina et al. 2018; Masolele et al. 2024; Vancutsem et al. 2021). Conservation hopes have largely relied on the extension of Protected Areas (PAs) coupled to the ambition to designate 30% of land to conservation by 2030 (Hughes and Grumbine 2023). Today, already 15% of the global terrestrial area, and 14% of the Democratic Republic of Congo (DRC), are under some form of protection (UNEP-WCMC 2022). Assessing whether PAs are effective in protecting forests can be challenging, given their non-random locations (Joppa and Pfaff 2009; Joppa and Pfaff 2010). PAs tend to have higher forest cover not necessarily as a result of effective protection, but rather their remoteness. This has been particularly the case for the DRC. Forest conservation areas were often established in locations with low deforestation pressure due to their inaccessibility (Joppa, Loarie, and Pimm 2008; Pfaff et al. 2014), and thus have little additionality in avoiding deforestation.

Since conservation efforts in the DRC rely extensively on PAs, a crucial question concerns their mitigation potential once remoteness fades and deforestation pressure increases. For the last 20 years, scientists have been calling for better conservation impact evaluation to understand what works in halting forest loss and what does not (Baylis et al. 2016; Börner et al. 2020; Ferraro and Hanauer 2014; Sutherland et al. 2004). The call is emphasized by the apparent gap between available and required conservation funding and a closing time window to act, making efficient allocation of the resources available even more important (Ferraro and Pattanayak 2006). However, a systematic understanding of what works when remains thin, especially in the Congo Basin (White et al. 2021).

In experimental settings, treatment can be assigned randomly to make treated and controlled units comparable and thereby allow to infer causal effects. Given the non-random location of PAs, it is necessary to control for confounding factors that influence both, deforestation outcomes and protection status. Quasi-experimental methods can implicitly control for confounders through the choice of the appropriate empirical design (Jones and Shreedhar 2024). Which design is appropriate depends on the research question, the context and the available data. For PAs, the mainstream strategy to evaluate conservation effectiveness has become propensity score matching, where protected areas are matched to non-protected areas based on a set of *ex ante* and *a priori* defined observable characteristics to control for non-random locations (Andam et al. 2008; Geldmann et al. 2019; Joppa and Pfaff 2010; McNicol et al. 2023).

The findings of matching-type studies of PA effectiveness in the DRC are ambiguous. Sze et al. (2021), Shah et al. (2021) and Abman (2018) found only marginally lower deforestation within PAs compared to matched areas outside, while Bowker et al. (2017) found substantially less forest loss under protection. However, matching does not overcome bias from non-random location if variables that explain both deforestation and protection status are omitted, a condition which essentially cannot be tested (Smith and Todd 2005). It can further run into problems when the common support, i.e. the overlap between propensity scores of treatment and control groups, is poor (Börner et al. 2020).

This study takes a different approach to evaluating PA effectiveness by focussing on frontier processes at their boundaries. Previous research has shown that deforestation outside of PAs is a strong predictor of forest loss inside (Burivalová et al. 2021). Most of the deforestation, in the DRC and beyond, occurs in the form of expanding land use frontiers as a result of growing populations and resource extraction (Meyfroidt et al. 2024; Molinario et al. 2020; Shapiro et al. 2023). Frontiers can be defined as "places or regions with specific land-use dynamics, leading to the rapid development of the exploitation of some land or resource, and that experience marked social-ecological transformation accompanying and resulting from resource exploitation." (Meyfroidt et al. 2024). Agricultural expansion, mining and logging are examples that can bring about such dynamics, but also the territorialisation of conservation has been framed as a frontier process connected to fundamental social and ecological transformation (Buchadas et al. 2022a; Meyfroidt et al. 2024). As unexploited natural resources often coincide with critical ecosystems, PA boundaries can be locations of friction between conservation and other frontier dynamics (Buchadas et al. 2022a; Luckeneder 2021; Simpson and Zirhumana 2021; Vuola 2022). On the one hand, the establishment of PAs can be a response to the expansion of deforestation frontiers - or at least a precaution against their future emergence (Buchadas et al. 2022a). On the other hand, conflicting land uses may also lead to degazettement and downgrading of existing conservation areas under the pressure of other actors and interests, seen for instance in the cases of Virunga National Park and Salonga National Park for oil and gas explorations, and Itombwe Nature Reserve following conflicts with local communities (Kujirakwinja et al. 2019; Qin et al. 2019; Tesfaw et al. 2018).

Where land use rents and population densities are low, PAs maintain high forest cover by virtue of their remote location (Pfaff et al. 2014). Protection in such contexts has been described as *passive* protection (Joppa, Loarie, and Pimm 2008). As the opportunity costs of forest cover or the need for more land under a growing population increase, *passive* protection may turn into *active* protection in case a PA is successfully holding back deforestation. While the distinction between *passive* and *active* protection is a useful simplification, frontier processes often follow much more complex spatio-temporal patterns that can give useful insights into the state of forest conservation.

This study proposes a novel metric-based typology to assess the status of protection at PA boundaries of the DRC. In a geographic regression discontinuity (GRD) framework, forest cover and deforestation on either side of PA boundaries are taken into consideration to assess their potential of withstanding deforestation. It is further analysed how this potential changes with the presence of other land use frontiers in the form of industrial mining and logging concessions. Given that the remoteness of PAs will keep on fading and that signs of emerging commodity frontiers are already revealing, it is concluded that conservation in the DRC is at a crucial point in time with important implications for the future of the Congo Basin rainforest.

2 Conservation and other land use frontiers in the DR Congo

The political turmoil the DRC has been going through over the last decades, including two wars and persisting instability especially in the eastern part of the country, has given rise to a fragile state with limited capacity of planning (Karsenty and Ongolo 2012). This has also impeded conservation efforts in the country.

Market mechanisms aim at making it more profitable for land users to keep forest rather than converting it. However, especially in settings where deforestation follows to a large share subsistence needs and not profit maximisation, the effectiveness of such instruments can be low (Angelsen et al. 1999). Pantropically implemented conservation initiatives that work with incentives, such as REDD+ (Reducing Emissions from Deforestation and Degradation + other forest-based activities), have not fully unfolded in the DRC, despite the formal joining of the program in 2009, as institutional shortcomings hinder the implementation and the capability to engage into long-term anti-deforestation commitments (Karsenty and Ongolo 2012; Kengoum, Pham, and Mihigo 2024; Pham et al. 2021).

Most forest conservation efforts in the DRC have concentrated on the establishment of PAs. In line with the Kunming-Montreal Global Biodiversity Framework, the ambition has been formulated to designate 30% of terrestrial land as protected by 2030 (Convention on Biological Diversity 2022). The long history of PAs in the DRC reflects the notion that the removal of people is essential to conserve forest. Under colonial rule, authorities declared all unoccupied land as vacant and hence-

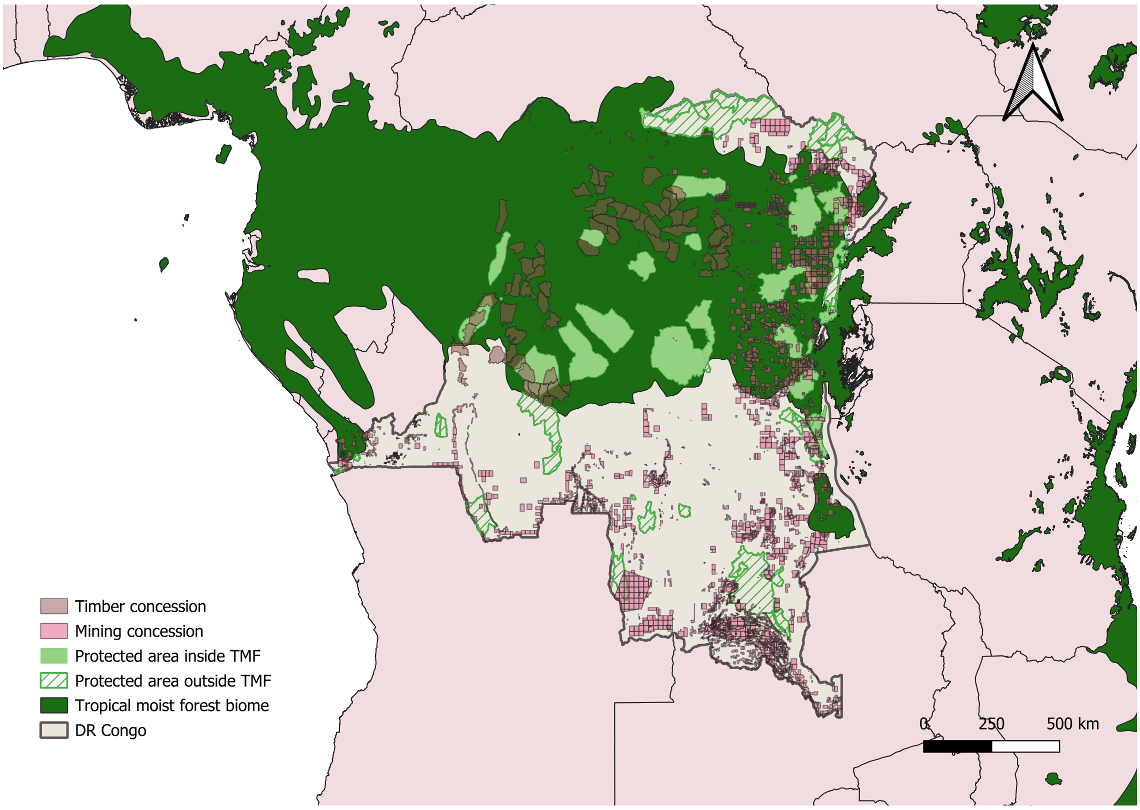

Figure 1: *Tropical moist forest biome coverage of the DRC and locations of PAs and mining and timber concessions. Note that most of the concessions are inactive (see Section 3)*

forth property of the state, on which they began to establish the first national parks (Van Acker 2005; Inogwabini 2014). In 1973, the law was augmented under the presidency of Mobutu who rendered all land, regardless of its tenure status, as state property (Van Acker 2005). These laws provided the basis for the establishment of numerous PAs, now counting a total of 60 and covering 15% of the country (World Database on Protected Areas 2024).

In the process, communities were often displaced and prohibited from accessing the forest, where necessary under the use of military force and without compensation (Flummerfelt 2022; Marijnen and Verweijen 2016; Domínguez and Luoma 2020; Inogwabini 2014; Simpson and Geenen 2021). Local communities and indigenous people who had lived inside the forests for centuries were removed from their ancestral lands under the legitimisation of forest protection, as happened in the case of the Batwa in Kahuzi-Biega National Park. (Barume 2000; Simpson and Geenen 2021).

Whereas protection exists on paper, conservation impacts on the ground are not obvious. Insufficient government funding and low legitimacy of PAs among communities impede the implementation (Inogwabini 2014). Programs to support communities and provide alternative livelihoods are also missing (Barume 2000; Spira et al. 2019). One way to compensate for the lack of funding and management capacities of the state has been the installation of co-management schemes, in which international NGOs partnered with government agencies. Co-managed parks have been found to increase avoided deforestation in the face of deforestation pressure (Desbureaux et al. 2025). However, only six PAs are currently managed collaboratively in the country.

Additionally, political priorisation appears to be a challenge, since conservation status has not prevented the allocation of protected land to industrial mining and logging concessions (Cirimwami, Baguma, and Mushagalusa 2021; Simpson and Pellegrini 2022). The implications of industrial mining operations on conservation areas are unreported this far (Cirimwami, Baguma, and Mushagalusa 2021). The eastern part of the country, where many of the mineral deposits are located (Figure 1), has been suffering from insecurity for a long time, largely preventing mining industry from setting foot and leaving most of the exploitation to artisanal and small-scale producers (Draulans and Krunkelsven 2002; Kilosho Buraye, Stoop, and Verpoorten 2017; Simpson and Pellegrini 2022). However, the industry's interest in the region re-surges (Geenen 2014; Kilosho Buraye, Stoop, and Verpoorten 2017; Radley 2020), and concessions for exploration and exploitation activities have been granted inside and outside of PAs alike.

Industrial logging concessions are only located in the western part of the country, where the Congo river is used to transport logs to the port (Ferrari and Cerutti 2023) (Figure 1). Compared to the neighbouring Congo Basin countries, timber production in the DRC has been low (Eba'a Atyi et al. 2022b). A moratorium issued in 2002 prohibited the signing of new logging titles while aligning the sector to principles of good governance, although concessions were assigned illegally while the moratorium was in force (Majambu, Demaze, and Ongolo 2021). The moratorium was accompanied by the adoption of a new forest code which obliged logging companies to adhere to social and environmental principles in their operations, as laid down in obligatory management plans (Majambu, Demaze, and Ongolo 2021). While the government has signaled intentions of lifting the moratorium in the near future, the implementation of management plans is still lagging behind (Chervier et al. 2024; Global Witness 2018a; Karsenty et al. 2017).

Given the anticipated expansion of conservation, mining and logging frontiers in the DRC, and the large overlap they have, the question of how these interact is crucial, but not evident. On the one hand, extractive industry and conservation actors may have synergistic interests of restricting territorial access for other actors, especially small-holder farmers and artisanal producers (Buchadas et al. 2022a; Geenen 2014; Tritsch et al. 2020; Vuola 2022). On the other hand, extraction itself requires the removal of vegetation and the construction of infrastructure, thereby triggering even more deforestation by providing access to other actors and undermining conservation agendas (Giljum et al. 2022; Kleinschroth et al. 2019).

3 Data

Forest disturbance data and data on remaining undisturbed forest was extracted from the Tropical Moist Forest (TMF) dataset of Vancutsem et al. (2021). The uniqueness of the TMF data is that it stores information on both deforestation and degradation for the years 1990-2023. In the data, forest cover is determined with the help of a procedural sequential decision tree. In comparison with other commonly used data sets, such as Hansen et al. (2013), TMF data does not rely on tree cover quantification and loss thereof, but instead distinguishes undisturbed and disturbed tropical moist forest, and other (non-forested) land cover. Undisturbed forest is classified as that which has neither been degraded nor deforested over the entire Landsat time series (i.e., since 1982), and forest disturbance marks the loss of canopy cover. Disturbances of high intensity or of a duration of more than 2.5 years are classified as deforestation events, whereas low intensity and short duration disturbances are classified as degradation. The focus of the TMF data on undisturbed forest is especially useful in the DRC context, where fallow land during cycles of shifting cultivation can quickly restore forest cover and easily be mistaken for forest (Potapov et al. 2012). Further, TMF data reportedly performs better in detecting disturbances than other frequently used datasets, such as that of Hansen et al. (2013) (Vancutsem et al. 2021).

As an irreversible, binary outcome variable, pixel-level analysis of deforestation can lead to bias in the estimation. The data was therefore aggregated from 30m to 500m resolution to obtain variation in the outcome variables (Garcia and Heilmayr 2024).

Shapefiles with the location of protected areas, mining and timber concessions were accessed through the DRC Forest Atlas (Bélanger and Mertens, n.d.). All data were clipped to the tropical moist forest biome, thereby excluding the Miombo dry forest in the southern part of the DRC (see Figure 1). 36 of the 54 PAs were covered at least partially by this area of which 6 did not have an IUCN classification reported in the data. After consulting with experts from the DRC, one PA was dropped for not having been established yet (*Kibali-Ituri*) and 3 missing IUCN categories were assigned as category VI, leaving only two small-sized PAs unclassified (*Kwada* and *Mont Homas*). For all PA boundaries within the tropical moist forest biome, boundary points were placed every 15km, which were then used to estimate local deforestation and forest cover discontinuities (see Methods section).

PAs with overlapping land use allocations were identified by overlaying mining and timber concessions with PA boundaries. For mining concessions, 144 overlapping boundary segements of more than 5km length were identified, of which 41 were exploitation permits covering a total length of 639 km[1]. However, only one of these mining concessions was hosting an operational mine: the *Twangiza* mine, operative between 2012 and 2020 (Maus et al. 2022; Radley 2020). This mine was used as a case study to see how forest cover discontinuities changed with the start of mining operations.

Also logging concessions shown in the Forest Atlas were not all operative. A moratorium on new logging titles in 2002 demanded concessions to have management plans approved to get concession rights validated (Chervier et al. 2024; Majambu, Demaze, and Ongolo 2021). Among all validated and active concession signed since then, four titles overlapping with the Oshwe Hunting Reserve and the Tumba-Lediima Reserve were used as a case study for mining-logging frontier dynamics.

[1]The data on mining concessions from the DRC Forest Atlas (Bélanger and Mertens, n.d.) was substantially more conservative than other data provided via Global Forest Watch. A comparison with official maps displayed on the DRC Mining Cadastre portal suggested that the former was more accurate and is therefore depicted in Figure 1.

4 Methods

4.1 A protection typology

Theories on deforestation processes highlight the importance of the spatio-temporal context in which a landscape is located to understand its land use trajectory (Meyfroidt et al. 2024; Barbier, Burgess, and Grainger 2010; Angelsen 2007; Lambin and Meyfroidt 2010). Once an area is deforested, forest has to regrow over a long time period before it is fully recovered, which makes deforestation a quasi-irreversible event (Garcia and Heilmayr 2024). Due to this absorbing property, the stage of land use transformation a PA is situated in has decisive implications on conservation.

Boundary discontinuities alone cannot fully disclose frontier dynamics. The absence of discontinuities in deforestation across PA boundaries, for instance, can lead to three different conclusions on protection effectiveness, depending on remaining forest cover and previous dynamics. In the first case, discontinuities can be absent in case the deforestation frontier has not yet reached PA boundaries. Forest cover would be high and deforestation low on either side of the boundary, and protection can be described as passive. In the second case, when deforestation has already reached the PA, discontinuities would be absent if deforestation spreads across boundaries and does not stop at PAs. Protection is then not able to stop deforestation at the boundary and hence inefficient. A third case could arise when forest is already scarce around PA boundaries and the deforestation frontier has moved inside the PA. Again, no discontinuity in deforestation or forest cover would be visible, but the context would be very different from the previous two cases.

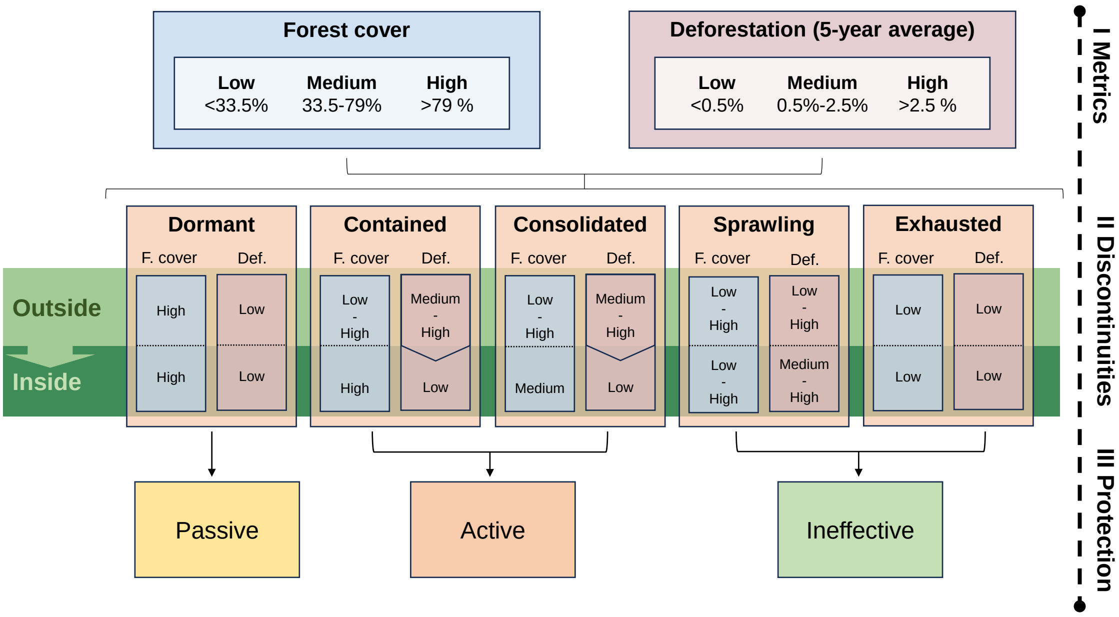

Figure 2: *Metric-based typology of forest protection used to classify PA boundary segments.*

To account for the variety of contexts in which discontinuities may or may not appear, this analysis develops a novel typology of forest protection is conceptually related to the forest transition theory (Barbier, Delacote, and Wolfersberger 2017; Mather 1992), and to landscape-level typologies of land use change developed in Buchadas et al. (2022b) and De Sy et al. (n.d.) (Figure 2). Data on remaining undisturbed forest cover and the annual rate of forest disturbance on either side of PA boundaries were used to identify the deforestation setting in which a PA is situated in. Forest cover and deforestation metrics were determined in a data-driven approach following interval classification methods adopted in Jamaludin et al. (2022), categorising each side of PA boundaries according to low, medium and high forest cover (<33.5%, 33.5-79%, >79%) as well as low, medium and high annual deforestation (<0.5%, 0.5-2.5% and >2.5%), where deforestation is estimated as the share of pixels that were cleared in a given year. Boundaries that already had low forest cover in 1990 were discarded from the analysis, as forest cover might be naturally low at these points. This also implies that points are excluded that experienced extensive deforestation dynamics before 1982, i.e. the year of the first available Landsat images. The metrics were then combined to analyse the type of conservation frontier that was present at different PA boundaries. Alternative threshold specifications were tested and are reported in the appendix, but did not lead to notable differences in the results.

Based on the estimated discontinuities, PA boundaries were distinguished into five different stages of protection. As PAs tend to be placed in remote locations, protection in the first stage can be described as *dormant*. Similar to the first stage in the forest transition theory (Mather 1992), forest cover is high and deforestation low on either side of the boundary. When the deforestation frontier progresses and reaches PA boundaries, the passive protection in the dormant stage fades, and deforestation is either *contained* by PA boundaries with high deforestation outside but low inside, or *sprawling* in case forest loss enters PAs, shown by high loss on either side of the boundary. If deforestation enters PAs, it may be stopped before the forest cover has fully disappeared, in which case deforestation has been *consolidated*. Ultimately, if forest cover has already vanished both inside and outside the park, protection is *exhausted* and has become locally redundant.

In the classification, forest loss is averaged over five years to smooth out annual fluctuations. The categorisation ultimately allowed to distinguish the mechanisms through which protected areas are associated with deforestation as either passive due to remote location, active by enforcement of protection, or ineffective in cases where protection does not contain deforestation.

Temporal information was used to determine how PA protection changed over time. Under a functioning PA system, it would be expected that *passive* protection from remoteness turns into *active* protection as the PA boundaries stop the forest edge from moving inwards and resist anthropogenic pressures. However, if protection was flawed, this would result in deforestation *sprawling* across boundaries and make protection exist on paper, but without any conservation effects.

4.2 Geographic regression discontinuity design

To account for the non-randomness of PA locations (Joppa and Pfaff 2009), this study used a geographic regression discontinuity (GRD) design for estimating discontinuities of forest cover and forest disturbance along PA boundaries. The identification of causal effects in this GRD relies on the assumption that under the same treatment status, potential outcomes on either side of the treatment cutoff are continuous, i.e. that deforestation would not show a discontinuous jump across PA boundaries in absence of protection (Keele and Titiunik 2015). Hence, all discontinuities in relevant covariates across PA boundaries should be a result of treatment itself for the design to be valid. Since the units of analysis are grid cells, and given that geophysical features are often continuous across space (Figure 1 in the appendix), the main concern for the validity of the empirical strategy is that other policy parameters change with protection status, so that the effect of protection and that of other policies can no longer be separated. This is known as the problem of compound treatment (Keele and Titiunik 2015). To avoid compound treatment, PA boundaries overlapping with country borders were removed from the analysis, as these have been shown to create discontinuities in forest cover due to differences in policy environments (Wuepper et al. 2024).

The empirical strategy followed Keele and Titiunik (2015) by sampling points along PA boundaries in the tropical moist forest biome of the DRC and non-parametrically estimating local discontinuities for each point separately. Instead of estimating only one treatment effect coefficient, as in the parametric case, a treatment effect curve along PA boundaries is estimated. The individual boundary point effects can then be aggregated into parameters of interest, such as effects by IUCN category or average effects over all boundary points. In addition to its flexibility, non-parametric regression has the advantage over linear regression that it is less sensitive to the choice of bandwidth around treatment cutoffs (Wuepper and Finger 2022) and that it explicitly shows location-specific treatment responses (Keele and Titiunik 2015). For this analysis, sampled points were placed every 15km along PA boundaries, resulting in a total sample of 810 points. For each of these points and all years between 2000-2022, local effects on forest cover, deforestation and forest degradation were estimated.

Formally, the effect on a forest outcome Y at boundary point p in time t can be written as:

$$\tau_t(p) = \lim_{x \to 0^+} E[Y_t|X = x] - \lim_{x \to 0^-} E[Y_t|X = x] \tag{1}$$

with x indicating the distance to boundary point p and Y_t being the observed forest outcome at time t. The limits from above and below the treatment cutoff at point p are then estimates in a local linear regression following Keele and Titiunik (2015) as follows:

$$\begin{aligned} \mu_t^+(p) &= arg\, min \sum_{i \in N^+} (Y_{it} - \alpha_p^+ - \beta_p^+ f(i,p))^2 w_{ip} \\ \mu_t^-(p) &= arg\, min \sum_{i \in N^-} (Y_{it} - \alpha_{pt}^- - \beta_{pt}^- f(i,p))^2 w_{ip} \end{aligned} \tag{2}$$

N^+ and N^- refer to the neighbourhoods within the chosen bandwidth around boundary point p, $f(i,p)$ is a function to indicate the euclidean distance between observation i and p, and $w_{i,p}$ are spatial weights determined by a triangular Kernel weighting function, with higher weights on observations in closer proximity to p.

The optimal bandwidth around p was calculated following the mean-square-error optimal bandwidth selection of Calonico, Cattaneo, and Titiunik (2014) for each point, and robust bias-corrected standard errors were reported as proposed in

Calonico, Cattaneo, and Farrell (2021).

4.3 Difference-in-Discontinuities

In addition to PA boundary effects on forest loss, it was tested how mining and logging concessions interact with the protection provided by PAs, in cases where new titles overlapped with boundaries. This was done in a geographic difference-in-discontinuities design in which the timing of a new land allocation was used to compare discontinuities before and after the placement. Under the assumption that the protection-impact remains constant over time, it can be estimated how mining and logging concessions affected protection (Butts 2021; Grembi, Nannicini, and Troiano 2019). Butts (2021) showed that, in absence of compound treatment and sorting, the treatment effect can be recovered from the regression discontinuity of the differenced outcomes as

$$\tau_D = (Y_{t+s} - Y_{t-1})^+ - (Y_{t+s} - Y_{t-1})^- \tag{3}$$

where t is the time in which a concession was placed on PA boundaries and $s > 0$.

Instead of the non-parametric approach from equation 2, a semi-parametric estimation for the different PA boundary segments with overlapping concessions was implemented, similar to that proposed in Dell (2010). The diff-in-disc resembles Wuepper et al. (2024) and Baragwanath and Bayi (2020) in its functional form in estimating the following regression separately for each year:

$$\tilde{Y}_i = \alpha + \beta \mathrm{d}_i + \tau \mathrm{D}_i + \gamma_b + \delta_i + \epsilon_i \tag{4}$$

where $\tilde{Y}_i$ is the differenced outcome of cell i as specified in equation 3, d_i is a cells distance to the PA boundary, D_i is a treatment dummy indicating whether or not cell i is inside a PA, γ_b are boundary segment fixed effects and δ_i controls for longitude and latitude of a cell.

5 Results

The non-parametric GRD estimates for different boundary points were aggregated in different ways to compute treatment effect parameters of protection. This section first reports the aggregate results on the country level, before documenting heterogeneities by IUCN categories and by individual PAs. Finally, results at overlapping mining and logging frontiers are reported.

5.1 Conservation performance of PAs in the DRC

5.1.1 Country-level discontinuities

Aggregated over all PAs in the DRC, estimates showed positive, statistically significant and increasing discontinuities in forest cover across boundaries over time (Figure 3a). Hence, forest cover outside of PAs generally declined at a faster rate. While there was 3 percentage points higher undisturbed forest cover on the inside of PA boundaries in 2000, this difference stabilised at around 4.2 percentage points between 2015 and 2022, suggesting that on the aggregate level protection has some effect, but also that pressure on forests inside PAs has risen noticeably over the past two decades.

In comparison, discontinuities in the annual rate of deforestation was negative in 16 out of the 23 years, but only statistically significant in the year 2016 (Figure 3b). Meanwhile, discontinuities in annual forest degradation was significantly lower in 6 years and showed an overall larger discontinuity compared to deforestation results.

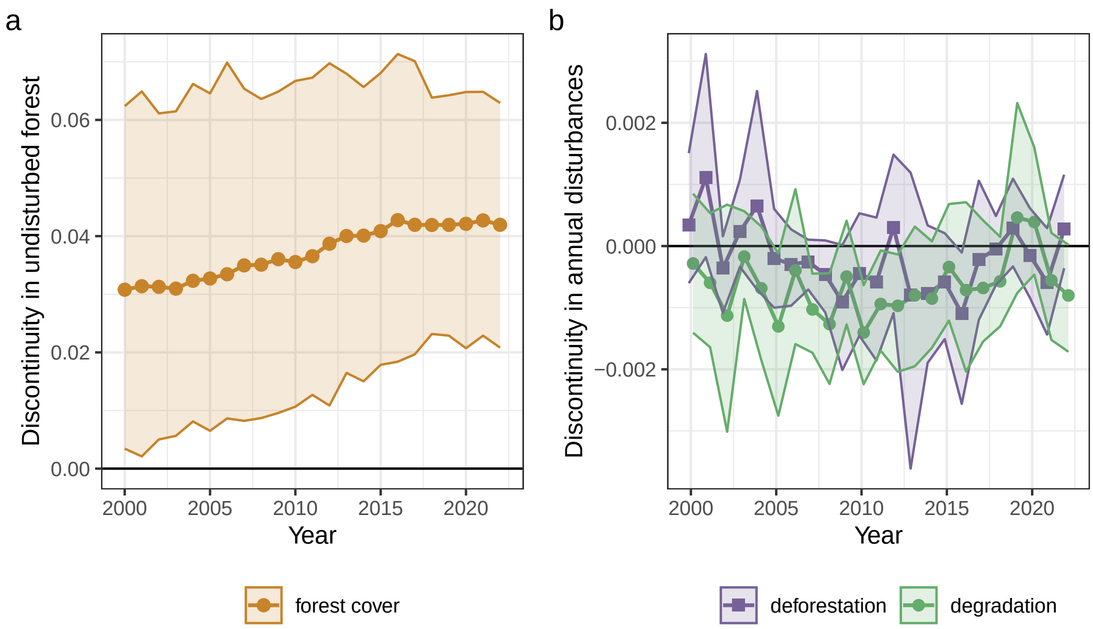

Figure 3: *Average regression discontinuities across PA borders aggregated from 811 non-parametric border point estimates within the tropical moist foret biome for* ***a*** *undisturbed forest cover and* ***b*** *deforestation and forest degradation. Positive estimates indicate higher incidence within PAs. 95% confidence intervals from blocked bootstraps displayed.*

Although the presence or absence of discontinuities can inform about the anthropogenic pressure that rests on PA boundaries, interpreting the effectiveness of PAs in resisting these pressures requires additional context. A classification of PAs according to the typology shown in Figure 2 gives more insights into the past and present dynamics of forest loss in the immediate surroundings of PA boundaries. It shows that the share of *dormant* protection, i.e. boundaries protected by remoteness, has decreased from 66% in 2002 to 55% in 2022 (Figure 4). Over the same time period, the share of boundaries with *contained* deforestation increased from 3% to 9%, and boundaries that have already exhausted their forest cover have increased only slightly from 8% to 9%. The highest increase was seen for share of boundaries with *sprawling* deforestation, i.e. where protection did not stop at the PA boundary, as it increased from 7% to 18%.

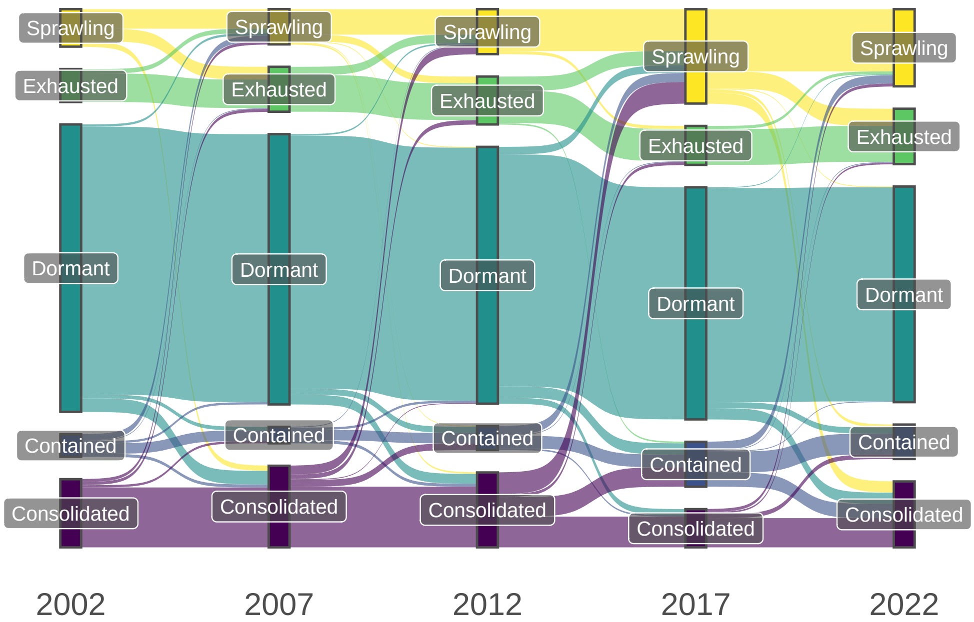

Figure 4: *Classification of forest protection according to the typology presented in Figure 2.*

Conclusively, aggregated country-level effects suggest that the remoteness of PAs in the DRC is gradually fading. As the deforestation frontier moves closer and pressure on PA boundaries rises, only a minor share is able to contain pressure in keeping deforestation out of the PAs. Most of the analysed boundaries were found to give in to deforestation as it arrived at the boundary.

5.1.2 Heterogeneities across PAs

When aggregating forest cover discontinuities by PA, heterogeneities become apparent (Fig 6.5). Focusing only on PAs that are not adjacent to country borders, 5 of the 19 investigated PAs had statistically significant discontinuities in undisturbed forest cover in 2022. The highest discontinuity in 2022 was observed for the Tumba-Lediima Reserve, with 10.8 percentage points higher forest cover inside, followed by Kahuzi-Biega National Park with 8% and Tanya Nature Reserve with 6.9 percentage points. The largest increases in forest cover discontinuity between 2012 and 2022 were found in Mangai, whereas the largest decrease was observed in Kahuzi-Biega National Park.

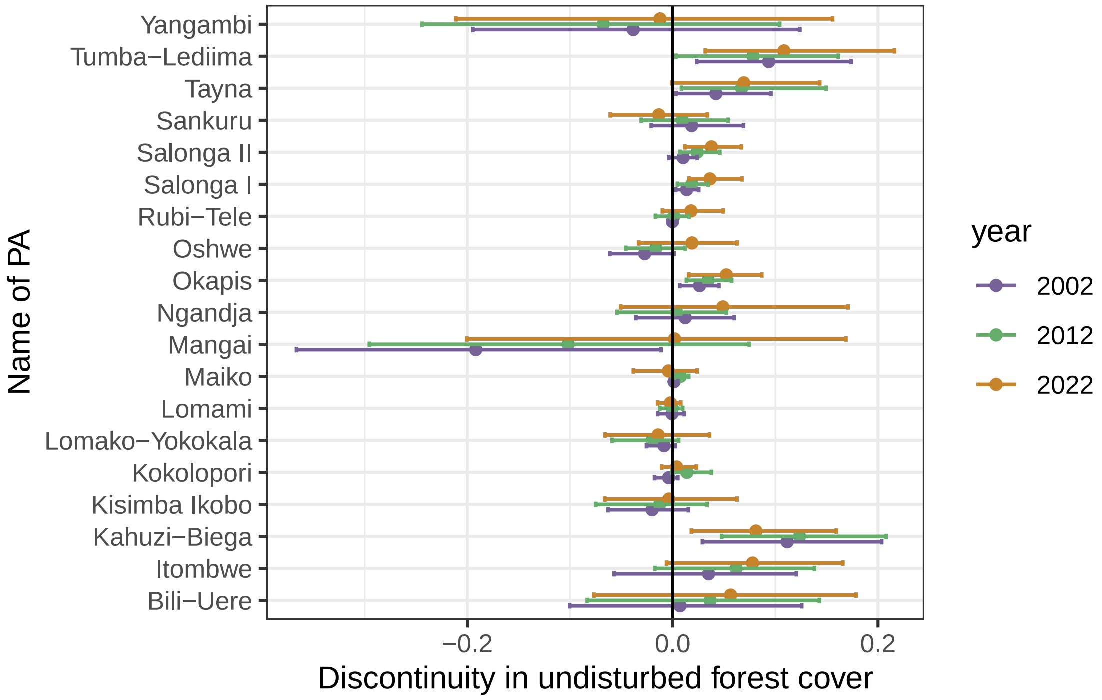

Figure 5: *Discontinuities in forest cover across PA boundaries, by PA. Positive numbers mean higher forest cover inside the PA. Confidence interval displayed at 95%. Inflated confidence intervals may be result of a low number of boundary points, and PAs with less than 10 border point observations were entirely dropped. See Table 7.1 for details.*

Applying the protection typology to the individual border points of each PA shows more of these heterogeneities (Table 1 in the appendix). While some PAs are not experiencing much deforestation pressure and have predominantly dormant boundaries due to their remote location, others have already been deforested substantially around PA boundaries or are currently seeing extensive deforestation sprawling inside. Of the 21 assessed PAs, 15 had at least some border point estimates with deforestation sprawling across boundaries, and 8 of them had at least a quarter of all border points classified as sprawling.

On average, PAs with more than 10 boundary points had 17.5% of their boundaries classified as sprawling, with the highest share found in the Sankuru Nature Reserve, followed by Yangambi and Virunga National Park [2]. The larger National Parks Maiko, Salonga and Lomami were almost exclusively characterised by dormant protection. The highest share of contained deforestation was observed for the Mangai Hunting Domain (27%), and the average across PAs was at 8.7% of border points categorised as contained.

5.1.3 Heterogeneity by strictness of protection and other characteristics

PAs differ in the human activities they allow within their boundaries, reaching from strictly scientific use to permitted resource extraction and land uses under certain conditions. The strictness is indicated by the IUCN categorisation of a PA

[2]Since Virunga National is partially located outside of the tropcial moist forest biome and is located adjacent to the Ugandan and Rwandan border in the east (Figure 1), only a fraction of its boundary is assessed in this analysis to avoid compound treatment effects.

(Stolton, Shadie, and Dudley 2013), although enforcement on the ground may differ. The tropical moist forest biome of the DRC overlaps with 4 strict nature reserves (IUCN category Ia), 1 wilderness area (Ib), 6 national parks (II), 12 habitat/species management areas (IV) and 10 protected areas with sustainable use of natural resources (VI).

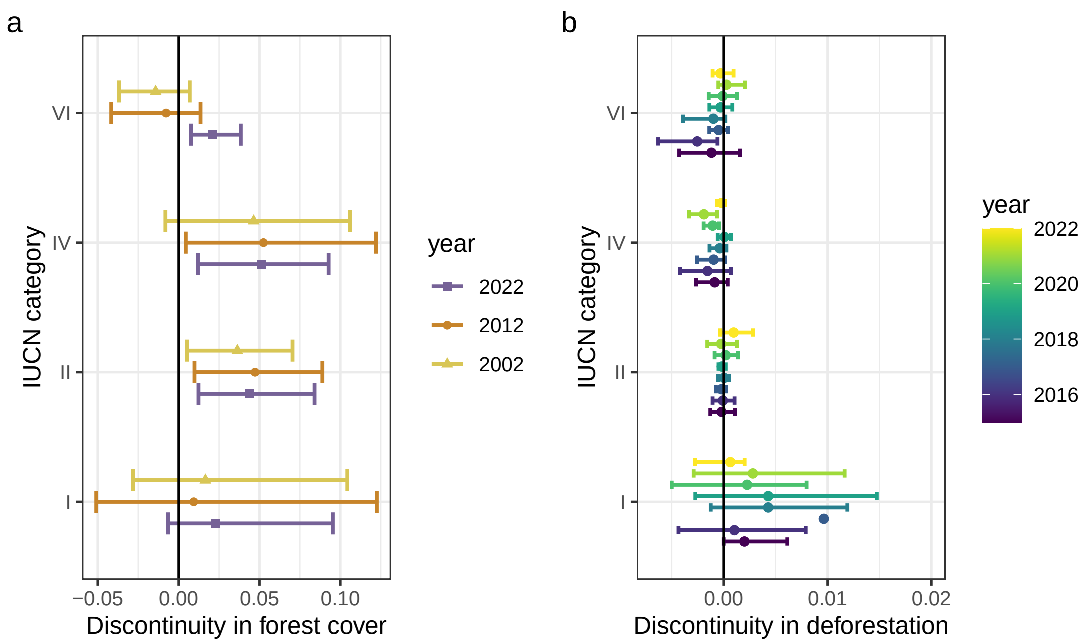

Figure 6: *Discontinuities in* ***a*** *undisturbed forest cover and* ***b*** *deforestation, by IUCN classification. All effects were aggregated from 809 individual boundary point estimates as specified in Keele and Titiunik (2015), and standard errors derived from blocked bootstrap procedure. Confidence interval displayed at 95%. Note that category Ia and Ib were merged into one category to obtain more points, but still remained the smallest of the categories, which might explain the large confidence intervals.*

Comparing the discontinuities for PAs with different IUCN categories between 2012 and 2022, the largest forest cover discontinuities are found for PAs in categories II and IV (Figure 6a). National parks in category II are large areas that are assigned to conserve large-scale ecosystem functionings and have frequently lead to displacement among local communities during the establishment, as they foresee strict prohibition of land-use and resource extraction from within park boundaries (Stolton, Shadie, and Dudley 2013; Inogwabini 2014). Category IV PAs are particulary targetting the preservation of critically endangered or threatened species and their habitats through interventions that are considered necessary to this end (Stolton, Shadie, and Dudley 2013).

The largest change in discontinuity over the last 10 years was visible for category VI PAs. These are explicitly allowing for traditional and cultural resource use practises and aim to promote sustainable ways of forest use. Consequently, they are also more likely to be established in areas where human-nature interactions are more prevalent, which may explain increasing deforestation pressure in the form of rising forest cover discontinuities. Discontinuities in deforestation over the last years were less evident. On average, discontinuities in stricter PAs of category I were larger than in others, but estimates were statistically insignificant for the most part throughout the different categories (Figure 5b).

In addition to forest cover and deforestation discontinuities for different IUCN categories, it was tested how the typology categorisation varied by the strictness of a PA (Table 7.3 in the appendix). With 77% of *dormant* border points, i.e. low forest disturbance on either side, national parks of category II had the largest share of this type in 2022. The lowest share was found for boundaries of nature reserves in IUCN category IV, with 53%. Effectiveness of protection under deforestation pressure was identified most frequently for strict nature reserves of category Ia and category IV nature reserves, with 7.4% and 6.3% of *contained* boundaries, respectively. The highest shares of boundaries with *sprawling* deforestation were found in strict nature reserves (Ia, 19%) and category IV and VI PAs (both 14%).

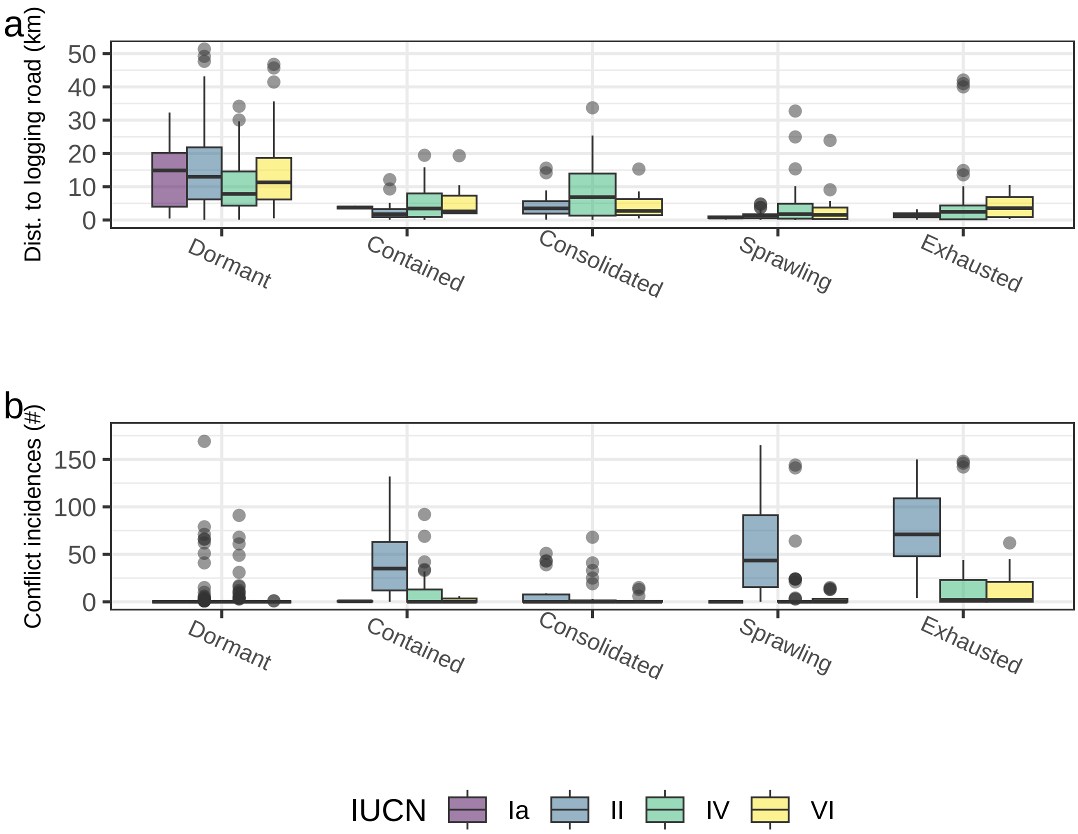

Figure 7: *Covariate distribution by protection type and IUCN category for **a** distance to nearest forest roads (Kleinschroth et al. 2019) and **b** number of conflict events within 10km of PA boundary since 2010 (Raleigh et al. 2010). Other covariate distributions are reported in Figure 3 in the appendix.*

To understand better how the protection performances of the different categories related to geographic characteristics of their location, Figure 6 and Figure 3 in the appendix depict covariate distributions by type of protection and IUCN category. Unsurprisingly, *dormant* PA boundary segments tend to be located away from forest roads (Kleinschroth et al. 2019) for all IUCN categories (Figure 6a). The weakest link with remoteness was observed for PAs of category VI, where deforestation outside of PA boundaries also occurred a few kilometres away from roads. Across all

IUCN categories, PA boundaries categorised as *sprawling* and *exhausted* were in closest proximity to forest roads. These findings were mostly confirmed by using data on accessibility (Weiss et al. 2018) and on roads (Meijer et al. 2018) (Figure 3 in appendix).

Further, remotely sensed data from Slagter et al. (2024) on newly developed forest roads since 2019 was used to show that 244km of new roads have been constructed within 5km of PA boundaries (Table 7.3 in the appendix). Of all PA boundaries in proximity of new forest roads, 59% were classified as *dormant* in 2022 and 22% already as *sprawling*. This shows that the establishment of new roads in the proximity of PAs continues to undermine protection efforts.

Another important factor in the assessment of PAs is the role of conflict. Conflict can be caused by land use restrictions imposed through conservation and result from the resistance against them (Inogwabini 2014; Pfaff et al. 2014). At the same time, conflict leads to displacement and hardship, and has people turn towards the forest in seek of shelter and resources (Merode et al. 2007; Nackoney et al. 2014). ACLED data (Raleigh et al. 2010) shows that armed conflicts were predominantly occurring close to the boundaries of category II PAs. Only dormant boundaries, where anthropogenic pressure was low, did not have any noteworthy occurrence of conflict events (Figure 7b).

The relationship between protection and collaborative governance of PAs was evaluated using data from Desbureaux et al. (2025), who mostly distinguished PAs governed in a co-managed or delegated way (Table 7.3 in the appendix). Among the sample of PAs, only three were co-managed in collaboration of public authorities and NGOs: Kahuzi-Biega National Park, Salonga National Park and Lomami National Park. Results confirm Desbureaux et al. (2025)'s findings that co-managed parks are predominantly located in remote areas with dormant protection. Only 4% of boundary points in co-managed PAs were categorised as sprawling, and none were found in PAs with delegated governance. While it is difficult to isolate the impact of co-management from other confounders in this setting, such as the IUCN category II that they all have in common or remoteness of their location, financial and technical support can have positive effects on PA outcomes in cases where the weak state capacity in the DRC hinders efficient PA management. Also other PAs not listed in the database of Desbureaux et al. (2025) received funding and support from both international and local NGOs to varying degrees, but the intransparency of the allocation makes a more detailed analysis on the role of support challenging.

5.2 Overlapping land allocations

A large part of PA boundaries in the DRC are overlapping with logging and mining titles. Although many of them are not yet productive, they have the potential of becoming extractive frontiers with both current and future implications for conservation, for instance by adding more stress on existing pressures or creating new pressures where none have existed previously. Contrarily, titles might as well ease pressure by helping to enforce land use restrictions imposed through PAs, especially under weak institutions. Insights on where titles have been allocated can help to contextualise the interactions of conservation-extraction frontiers and where they are most likely to occur.

The protection type varies with the category of overlapping concession title (Ta-

Table 1: *Overlapping land allocations by concession title and type of forest protection. Protection mechanisms are classified as displayed in Figure 2.*

Frontier	Dormant	Contained	Sprawling	Consolidated	Exhausted	N
Mining concessions						
Exploitation permit	46.67%	6.67%	20%	6.67%	13.33%	15
Research permit	59.62%	11.54%	11.54%	9.62%	7.69%	52
Logging concessions						
Active	42.86%	-	57.14%	-	-	7
Valid	66.67%	3.7%	14.81%	11.11%	-	27
In process	78.57%	7.14%	14.29%	-	-	14

ble 6.1). The share of PA boundaries with sprawling deforestation is highest for mining concessions under exploitation permit (27%) and active logging concessions (57%), while dormant PA boundaries are most commonly located within valid - but non-active - logging concessions (81%) and logging concessions that are still in process of validation (79%). Consolidated frontiers were most prevalent in overlaps with mining exploitation permits (20%), potentially a result of the exclusionary practises of extractive industry that drives out other actors. This has for instance been documented in the case of the Twangiza mine in the South Kivu province, the only operative industrial-scale mine in the DRC whose concession was overlapping with a PA in recent years.

5.2.1 Case study I: Twangiza mine

Despite the large area of mining concessions, only three industrial-scale mines in the eastern DRC were operative in recent years, of which only the Twangiza mine in South Kivu is located in a concession that overlaps with a PA, the Itombwe Nature Reserve (Radley 2020; Maus et al. 2022). To analyse the potential impact of industrial mining operations in proximity of PAs, a Diff-in-Disc model is estimated close to the Twangiza mine. The mine itself is located 6 km outside of the Itombwe Nature Reserve and was operative between 2012 and 2020, while the concession granted to the company also spans inside the reserve (Radley 2020; Maus et al. 2022) (see Figure 4 in the appendix).

Before applying for exploitation permits, mining actors are given exploration rights to scout the territory. For the Twangiza mine, the company Banro started exploration in 2005 that lasted five years. After receiving an exploitation title, Banro identified 2,000-2,500 people living inside the concession for relocation, and prohibited communities from building new constructions and from cultivating fields (Geenen and Claessens 2013). An agreement for replacement and compensation was signed in 2010, although concerns about the legitimacy of the agreement and its compliance were raised. The production of the mine started in 2012 (Geenen and Claessens 2013; Radley 2020).

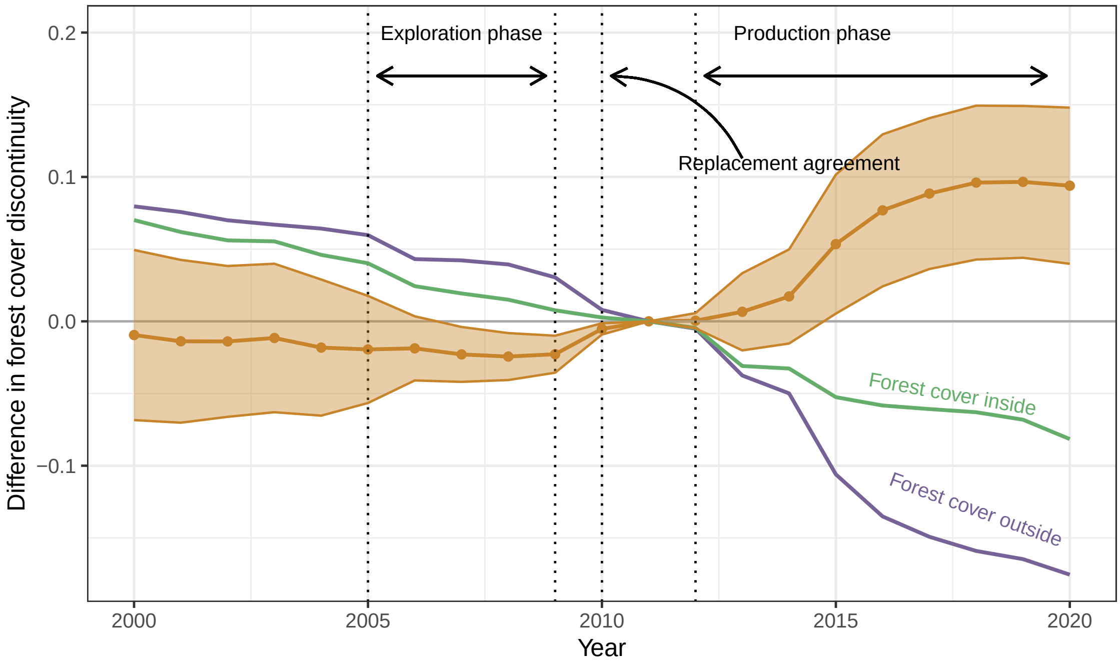

Figure 8: *Difference-in-Discontinuity estimates over time, normalised around the year before mining began. Discontinuities were estimated in a semi-parametric model around the affected PA boundary segment, with covariates controlling for longitude and latitude. 4634 observations to the left and to the right of the cutoff. Confidence interval displayed at 95%.*

Diff-in-Disc estimates for the PA boundary segment that lies within the mining concession have been relatively constant during the exploration phase for the Twangiza mine (Figure 8). However, after the replacement of communities and with the beginning of the production phase, forest cover outside - where the mine is located - started to diminish at a faster rate. The trend in forest loss inside the nature reserve has not changed in a meaningful way at any stage of the establishment of the mine.

Given that the Itombwe Nature Reserve entered a participatory process of regazetting its boundaries in 2010-2014 (Kujirakwinja et al. 2019), an additional analysis was conducted for the initial boundaries. When first established, the boundaries of the nature reserve were never fully recognised by local communities, and results accordingly do not show significant changes in the discontinuities over time (Figure 4 in the appendix). However, it cannot be excluded that the change in discontinuities at the redrawn boundaries during the production phase is influenced by the regazetting process.

5.2.2 Case study II: Logging concessions inside Tumba-Lediima Reserve and Oshwe Hunting Reserve

Despite the lasting 2002 moratorium on new logging concessions in the DRC, the government has been granting new logging titles starting in 2011. Under the forest code of 2002, all logging companies were obliged to submit management plans for their concessions the latest five years after issue. The plans involved commitments of sustainable harvesting and social benefits for residential communities, although their implementation is often flawed (Karsenty et al. 2017; Global Witness 2018a).

Overlap of concessions with PAs was observed particularly in three PAs: Tumba-Lediima (IUCN category IV), Oshwe (category VI) and Rubi-Tele (category VI). Given that, according to the data, Rubi-Tele only had concessions assigned in 2020, the analysis of overlapping logging-conservation frontier dynamics focuses on the former two.

As the earlier analysis in Figure 7 showed, the Oshwe Hunting Reserve and the Tumba-Lediima Reserve both indicated positive discontinuities in forest cover at their boundaries in 2022, although Oshwe Hunting Reserve only insignificantly. Each had two overlapping concession titles assigned in 2011 and one in 2014. However, one concession in each PA did not show signs of actual logging activities by 2018 (Global Witness 2018b), leaving a sample of four overlapping boundary segments (see Figure 5 in the appendix).

Among these overlapping concessions, an investigatory report by Global Witness showed that only one had their management plan approved by 2019 and did not harvest timber outside of the designated annual harvest area (concession 035/11 in Figure 9a). One concession had an approved management plan, but did not restrict itself to operating within the designated perimeter (concession 039/11, Figure 9b). Finally, two of the logging concessions had no approved management plan by 2019 but showed signs of logging activities regardless (concessions 020/11 and 015/11, Figure 9c and d).

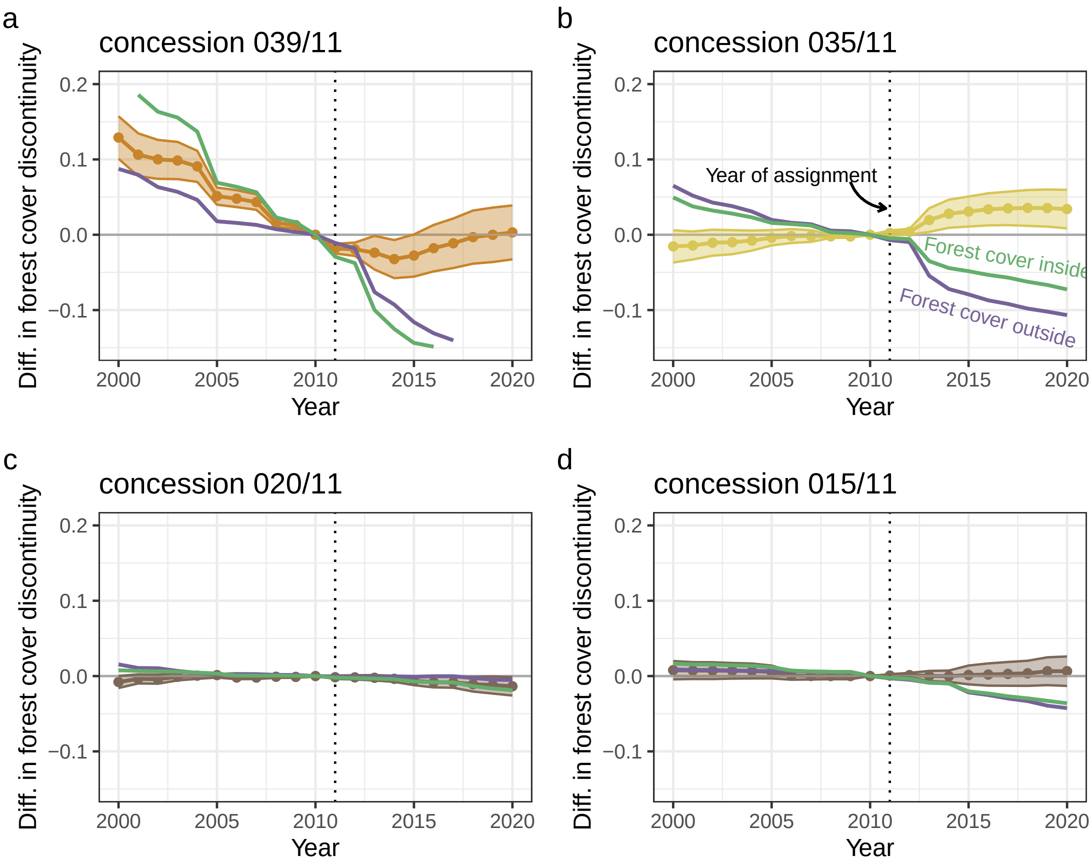

Figure 9: *Difference-in-Discontinuity estimates over time, normalised around the year prior to forest concession assignment. Confidence intervals displayed at the 95% level. Concessions displayed in **a** and **c** are located in Oshwe Hunting Reserve, **b** and **d** in Tumba-Lediima Reserve.*

The discontinuities in forest cover followed different trajectories for the four concessions (Figure 9). Concessions 020/11 and 015/11 did not reveal any noteworthy changes in forest cover discontinuity over time, regardless of the concession title assignment in 2011 (Fig 6.9c,d). The two concession with positive change in forest cover discontinuity after the title had been granted were concession 035/11 and 039/11, both of which had an approved management plan (Global Witness 2018b) and could thus indicate a positive interaction effect between well-managed logging concessions following the sustainable practises outlined in the management plans and conservation areas. However, when also considering the trajectory of forest cover extent inside and outside of PA boundaries, both concessions in fact show a drop in forest cover on either side as concessions were assigned, with a larger drop outside compared to inside. Hence, the increase in discontinuities was not associated with a decrease in deforestation inside the PAs due to better enforced protection, but rather a relatively stronger increase in forest loss outside.

6 Discussion

6.1 Remoteness protects, but is fading

Protected areas in the DRC stand at a crossroads. So far, protection has relied largely on remoteness (Joppa and Pfaff 2009). Although remoteness still plays an important part, the results presented in this study show that it has started to fade, and anthropogenic pressures on the PA system of the DRC are rising. Discontinuities in forest cover have been increasing to now 4% percent more undisturbed forest cover across boundaries on average, implying higher forest disturbance outside of PAs compared to inside over the last two decades.

A positive finding is that, even in a country with such fundamental institutional shortcomings as the DRC, not all PAs are conclusively inefficient. In 2022, 8% of investigated boundary points actively contained deforestation outside of the PA. However, twice as many have not succeeded in stopping deforestation at PA boundaries once the frontier arrived, casting doubt on this currently dominant conservation strategy. These results are coherent with findings by Burivalová et al. (2021) that, once the deforestation has arrived at PA boundaries, it is likely to spread inside.

In line with previous studies, results highlighted the role that forest roads play in attracting deforestation (Kleinschroth et al. 2019) and in weakening protection attempts. Most PA boundaries exposed to deforestation were located in close proximity to forest roads, and this exposure has further increased in the recent past as new forest roads are established (Slagter et al. 2024). With the anticipated growth in logging operations once the current moratorium on logging concessions ceases, pressure on PAs in the DRC will further increase.

6.1.1 Differences between IUCN categories

Previous studies have indicated that stricter protection does not necessarily lead to better conservation outcomes in terms of avoided deforestation (Ferraro et al. 2013; Pfaff et al. 2014; Elleason et al. 2021). One explanation for this observation is that stricter PAs tend to be located in more remote places where land use frictions and anthropogenic pressures are low (Ferraro et al. 2013; Pfaff et al. 2014). This is also in line with the finding that stricter category II PAs had the highest share of dormant protection among all IUCN categories.

Additionally, strict PAs are more likely to induce conflict over the use of land under protection and its resources (Ferraro et al. 2013). Acknowledgment by local communities is important for PAs to function, as for instance seen in the case of the indigenous Batwa in Kahuzi-Biega National Park. After initially having been forcefully removed from the park, the Batwa reoccupied a part of the park, with devastating impacts on its forest cover (Simpson et al. 2025). The findings in this study show that category II PAs were indeed associated with significantly more conflict incidences at their boundaries than other PAs that allow for certain forms of land use, underpinning this connection between strictness and conflict. Even among multiple use PAs, the degree to which communities were consulted in the implementation process can vary substantially, with implications on the social acceptance of installed land use rules (Pfaff et al. 2014).

Given the differences in location and the fact that GRD estimates are local to specific PA boundary segments, it is difficult to draw conclusions on comparative conservation performances of different IUCN categories when deforestation pressure rises. However, results presented in this study did show some interesting differences between the categories. One notable observation was that road access was particularly closely associated with sprawling boundaries of strictly protected category II national parks, although they also increased the likelihood of deforestation entering PAs with less stringent protection to a lower degree.

Co-management and delegated management of PAs have previously been found to increase the effectiveness in avoiding deforestation, but are found mostly in remotely located national parks under strict protection (Desbureaux et al. 2025). Given the dire need of funding and other kind of support for the PA system of the DRC, extending funding and collaborative management to less strict PAs with sustainable use could help to navigate land use frictions and offer support to the places facing the highest pressure (Buřivalová and Rakotonarivo 2025).

6.1.2 Looming mining and logging frontiers

Early signs of emerging land use frontiers in the form of concession title acquisitions represent another looming threat to protected forests in the DRC, with largely unknown implications at this point (Chervier et al. 2024; Meyfroidt et al. 2024; Radley 2020; Weng et al. 2013). So far, most of the granted mining titles are not operative, and only 55% of the timber concessions are productive with low harvest numbers compared to sectors in the neighbouring Congo Basin countries (Eba'a Atyi et al. 2022a). However, a substantial overlap between concession titles and PAs exists, frequently located at PA boundaries where deforestation has not reached yet. This coincidence of resource extraction and conservation frontiers will grow even stronger under the adopted Kunming-Montreal Global Biodiversity Framework aimed at declaring 30% of terrestrial area as protected by 2030 (Convention on Biological Diversity 2022). Given that the establishment of resource frontiers commonly requires infrastructural development (Meyfroidt et al. 2024; Slagter 2024), their emergence is likely to draw in other deforestation actors with potentially devastating impacts on intact forest landscapes and their biodiversity (Kleinschroth et al. 2019; Ladewig et al. 2024; Laurance, Sayer, and Cassman 2014; Weng et al. 2013).

Although first evidence on interaction effects between resource frontiers and land-based conservation in the DRC were provided in this study, more research is needed to generalise findings and account for the complex land use interactions in these processes (Meyfroidt et al. 2024). In principle, concessions can be understood as land titles, which have shown ambiguous effects on deforestation (Börner et al. 2020). Titles define access rights and can thereby exclude outside actors from engaging in deforestation, especially in the context of the DRC where institutional capacity to enforce conservation is low (Abman 2018). This has for instance been observed during the establishment of the Twangiza mine in eastern DRC, when artisanal miners and communities were removed from the mining concession area (Geenen 2014), although associated with counterproductive conservation outcomes at PA boundaries.

A similar logic was adopted to analyse logging concessions in other Congo Basin countries and in the Peruvian Amazon (Rico-Straffon et al. 2023; Tritsch et al. 2020). Findings from this study do not provide evidence for positive interaction effects of logging concessions and PAs on forest cover, with increased loss after concessions were established in two of the four analysed cases. Although the sample is too small to interpret findings as conclusive evidence, and a precise timeline of events in the establishment process of concessions is missing, they hint at the importance to insist on sustainable and lawful practises in the timber sector of the DRC. In neighbouring Congo Basin countries, certified concessions and those operating under management plans were found to have a much lower toll on forest cover (Tritsch et al. 2020). Further, the closing of abandoned logging roads is important, as the remaining infrastructure can leave scars in the forests that remain long after logging operations have ceased and is subsequently used by other actors to clear forest (Chervier et al. 2024; Kleinschroth et al. 2019; Potapov et al. 2017).

6.2 Limitations

A limitation in assessing PAs in a close environment around boundaries is the potential of leakage (Andam et al. 2008; Pfaff and Robalino 2017). Leakage may lead to to increased forest conversion outside of PAs as a consequence of restrictions posed on land use inside. Given that the GRD estimates are local to a close neighbourhood around PAs, they are more sensitive to be impacted by such spillovers than other methods. Although a relocation of deforestation from inside to the outside of PAs would inflate discontinuity estimates, it can also be interpreted as a consequence of effective protection. The typology developed in this study explicitly takes into account dynamics on either side of the boundaries and thereby helps to assess the effects of protection even if a part of the deforestation has been relocated to the outside of PAs. Quantifying leakage effects themselves is challenging, as it requires precise knowledge of their spatial extents to separate them from other dynamics (Pfaff and Robalino 2017), and is thus beyond the scope of this study.

A different problem concerns the use of PA boundaries as the treatment-defining cutoff in the regression discontinuity design. Although boundaries appear on maps, they may not always be visible on the ground, especially in remote locations. In such cases, no discontinuity in deforestation would be expected, and PAs would thus be assessed as inefficient in resisting deforestation pressure. A larger conceptual problem are shifting boundaries. Especially when frictions exist with other land uses, PA boundaries can get regazetted, as it happened for instance in the case of the Itombwe Nature Reserve after its initial establishment in 2006 (see Figure 4) (Kujirakwinja et al. 2019). In such cases, estimates for the wrong boundaries can make PAs appear inefficient although they were in fact only evaluated in the wrong locations. Besides the Itombwe Nature Reserve, no such instance since 2000 are known to the author, but information can be scarce and not readily available.

Finally, it should be noted that effectiveness of PAs is only evaluated based on forest cover loss in this study. It therefore does not account for other anthropogenic activities that cannot be as readily detected from satellite images but still compromise forest integrity. Defaunation, for instance, can be a disconnected and severe issue (Sagar et al. 2023). Also socio-economic implications of conservation were not considered in this analysis, but are important to take into account when assessing the eligibility of PAs to prevent deforestation given restrictions they pose on adjacent communities (Burivalová and Rakotonarivo 2025).

7 Concluding remarks

Previously, forest protection in the DRC has been known to rely largely on the remoteness of its PAs. In this study, a new typology of protection was developed to classify the activeness of the deforestation frontier at PA boundaries and their potential to resist it. The findings revealed that, in fact, the pressure on PAs along their boundaries has been increasing substantially over the past 20 years. While some PAs were able to withstand the pressure so far, twice as many did not succeed in keeping deforestation from entering inside. Especially in the later decade, these dynamics have further been accompanied by a renewed interest in the natural resources of the DRC by extractive industries, as seen through the acquisition of numerous concessions for timber and mineral extraction that coincide with PAs in many places. In the light of these developments, the conservation of the Congo Basin rainforest in the DRC is located at a crossroads: continuing on the path of protection-by-exclusion hoping to keep the deforestation front away from core forest areas, or adopting a new conservation paradigm in which communities are included in conservation planning and practise.

In the DRC, the large part of deforestation is not commodity-driven, but caused by small-scale rotational agriculture for subsistence consumption (Shapiro et al. 2023; Tyukavina et al. 2018). The increasing need for land is a result of a rapidly growing population that needs to support itself (Ernst et al. 2013; Molinario, Hansen, and Potapov 2015). Forest conservation strategies aimed at excluding communities from using the land may thus not be a lasting solution to halt deforestation. Instead, incorporating them in conservation strategies can provide more stable and inclusive ways to strengthen the resilience of both communities and conservation initiatives, especially in a context where the state does not have the capacity to provide alternatives (Berkes 2007; Hajjar et al. 2021). Attempts in recent years to give stronger agency to communities, for instance by legally enabling the creation of community forests since 2016 or allowing stronger community participation in PA establishments, are steps in the right direction, but also depend on the institutional setting (Lucungu et al. 2022; Kujirakwinja et al. 2019; Campos-Silva et al. 2021). Today, community forests in the DRC still face implementation challenges to become a viable alternative (Lescuyer et al. 2019), and further research on impacts and obstacles are urgently needed to understand the potential of creating win-win situations.

Data availability The data for the replication of the statistical analyses are available under `https://github.com/maladewig/PAs_DRC`.

Code availability The code for the replication of the statistical analyses is available for download under `https://github.com/maladewig/PAs_DRC`.

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8 Appendix

Continuity of covariates

A common test for the validity of the GRD is to assess whether geographic covariates are continuous across treatment thresholds, such that treatment was not assigned based on certain geographical features. As the figure beneath shows, there are no discontinuous changes for terrain slope, altitude, precipitation or temperature across PA boundaries.

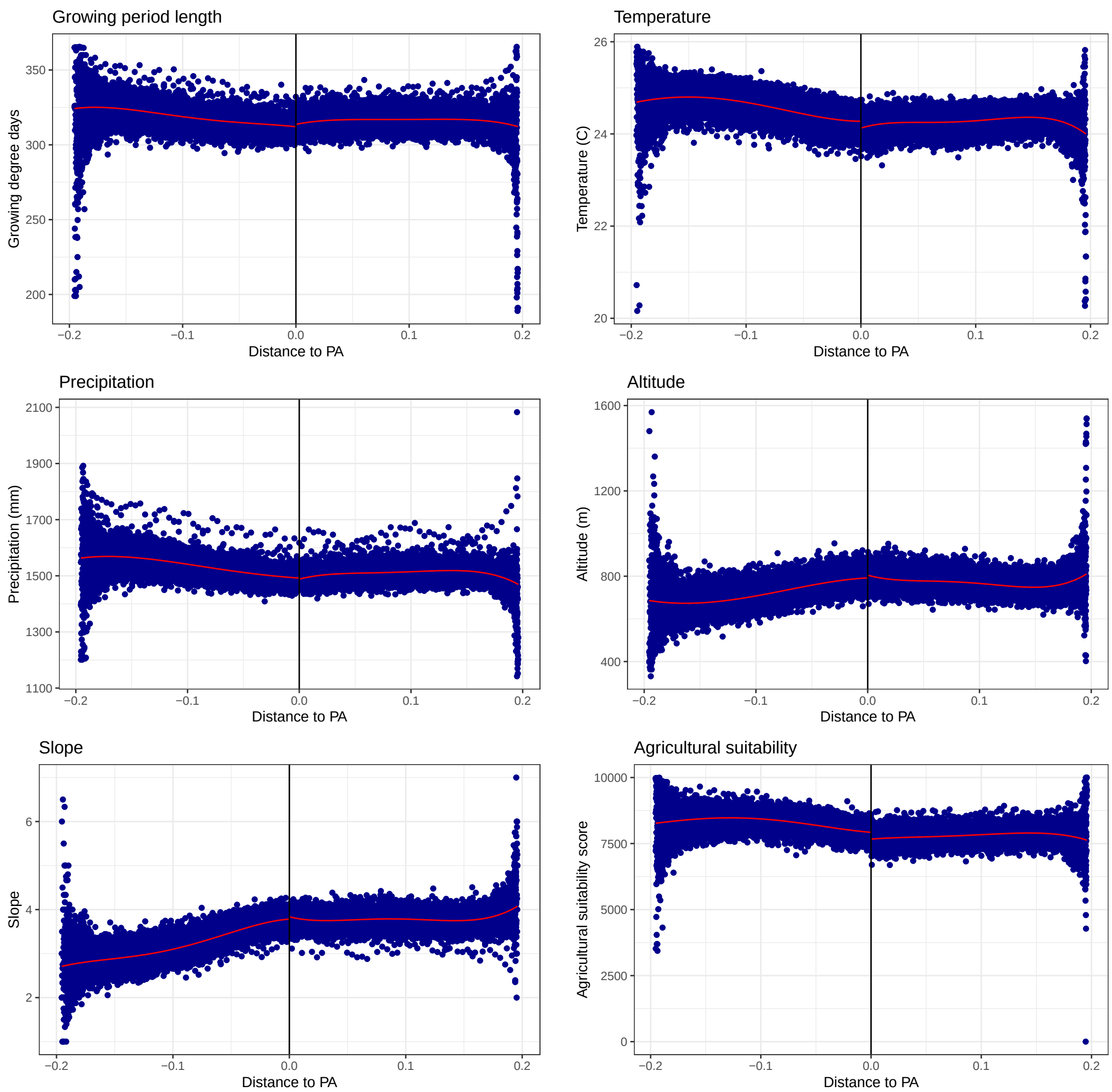

Figure 10: *Discontinuity plots around PA boundaries for different geophysical characteristics. All variables are taken from the Global Agro Ecological Zones (GAEZ) model (Fischer et al. 2021). Temperature, Precipitation and Growing Degree Days represent annual means.*

Pre-establishment discontinuity estimates

By testing for discontinuities existed prior to the establishment of PAs, it is possible to assess whether the observed differences in forest cover across thresholds have already existed before protection (Keele et al. 2017). Tanya Nature Reserve and Yangambi Biosphere Reservewere indicated in the data as established after 2000, but in fact had existed already before with different IUCN status and were therefore excluded. Of nine remaining PAs established after the year 2000, only Tumba-Lediima had a statistically significant discontinuity in forest cover already before PA establishment.

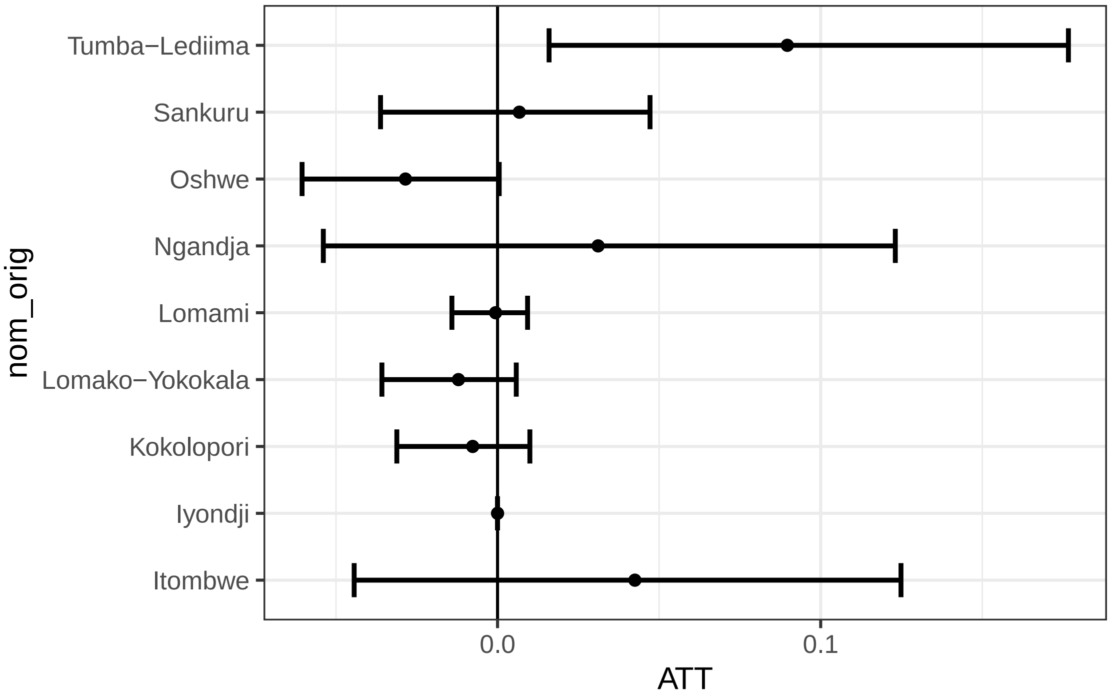

Figure 11: *Forest cover discontinuities across PA boundaries one year prior to PA establishment for PAs established after 2000. Confidence interval displayed at 95%.*

Protection type by PAs

Table 2: *Share of boundary points with different conservation frontier types by PAs in 2022 for PAs with more than 10 observations*

Name	IUCN	Area (km2)	Contained	Dormant	Exhausted	Sprawling	Consolidated	N
Domaine de chasse d'Oshwe	VI	1692.482	0.08	0.62	0.00	0.28	0.03	39
Domaine de chasse de Bili-Uéré	VI	3273.280	0.07	0.29	0.07	0.14	0.43	14
Domaine de chasse de Mangai	IV	1194.843	0.27	0.09	0.27	0.36	0.00	11
Parc national de Lomami	II	887.522	0.00	1.00	0.00	0.00	0.00	39
Parc national de la Maiko	II	1052.867	0.00	0.96	0.00	0.00	0.04	52
Parc national de la Salonga	II	1714.055	0.03	0.90	0.00	0.00	0.06	63
Parc national de la Salonga	II	1622.774	0.02	0.89	0.00	0.02	0.07	56
Parc national des Virunga	II	782.642	0.16	0.19	0.19	0.40	0.07	43
Parc national du Kahuzi-Biega	II	673.086	0.16	0.47	0.04	0.20	0.13	45
Réserve des primates de Kisimba-Ikobo	IV	97.041	0.19	0.56	0.00	0.00	0.25	16
Réserve naturelle d'Itombwe	IV	571.789	0.19	0.28	0.19	0.28	0.06	32
Réserve naturelle de Tayna	IV	89.967	0.07	0.86	0.00	0.00	0.07	14
Réserve Tumba-Lediima	IV	746.268	0.16	0.48	0.00	0.19	0.16	31
Réserve de biosphère de Yangambi	Ia	223.108	0.08	0.46	0.00	0.46	0.00	13
Réserve de chasse de Rubi-Télé	VI	1127.300	0.03	0.86	0.00	0.05	0.05	37
Réserve de faune à okapis	IV	1393.957	0.13	0.74	0.04	0.02	0.06	47
Réserve forestière de Lomako-Yokokala	IV	362.822	0.00	0.88	0.00	0.12	0.00	17
Réserve naturelle de Ngandja	IV	387.060	0.06	0.11	0.50	0.33	0.00	18
Réserve naturelle de bonobo de Kokolopori	IV	374.085	0.00	1.00	0.00	0.00	0.00	13
Réserve naturelle du Sankuru	IV	2664.194	0.05	0.33	0.05	0.53	0.05	43
Réserve naturelle du triangle de la Ngiri	IV	523.505	0.08	0.04	0.29	0.29	0.29	24

IUCN heterogeneity

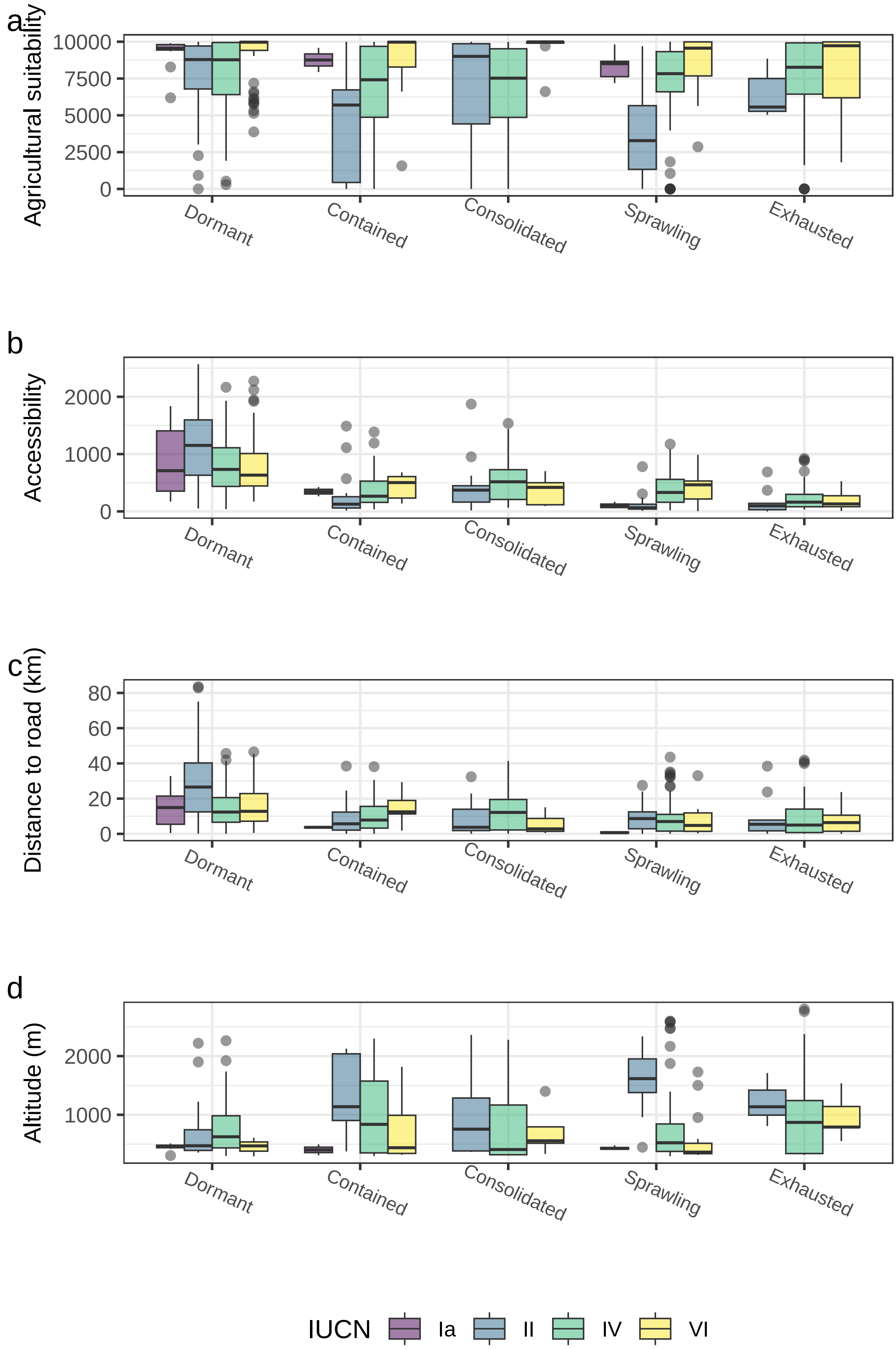

Figure 12: *Covariate distribution by IUCN category for* ***a*** *Agricultural suitability (Fischer et al. 2021),* ***b*** *travel time to nearest city (Weiss et al. 2018),* ***c*** *distance to roads (Meijer et al. 2018) and* ***d*** *altitude (Fischer et al. 2021).*

Metrics for protection typology

The classification into protection typology followed metrics derived from a jenks classification algorithm. The resulting threshold metrics were similar to a those obtained in Jamaludin et al. (2022). To test the sensitivity of the classification results to the specified metrics, thresholds used in Buchadas et al. (2022) and De Sy et al. (n.d.) were used. Buchadas et al. use 10% and 55% thresholds to distinguish low and high forest cover, but apply them to tropical dry woodlands which generally have lower forest cover density. De Sy et al. (n.d.) apply 15% and 50% thresholds in their application to tropical moist forests, but these thresholds are originating from a country-level study by Pendrill et al. (2019) and need to be treated with caution when applied to a finer landscape-scale.

For distinguishing high and low deforestation, Buchadas et al. (2022) used 0.6% of annual converted land cover as a threshold, and De Sy et al. used 0.37%.

Comparing classification of border points for the year 2022 for the different metrics, only minor differences are visible for consolidated and sprawling categories. Whereas Buchadas et al. (2022) metrics resulted in 8% consolidated boundary points and 12% with sprawling deforestation, a classification following De Sy et al. found 4% and 15%, respectively (Table 7.2).

Table 3: *Conservation frontier typology classification with threshold metrics of Buchadas et al. (2022) and De Sy et al. (in preparation) in comparison.*

Category shares in 2022	Buchadas et al. (2022)	De Sy et al. (in preparation)	Jenks clustering intervals
Dormant	0.66	0.65	0.61
Contained	0.06	0.07	0.08
Consolidated	0.08	0.04	0.08
Sprawling	0.12	0.15	0.16
Exhausted	0.08	0.08	0.07

Characteristics of protection types

Table 4: *Characteristics of different types of conservation frontiers in 2022 classified according to Figure 2 in the main document. Exposure to new logging roads established after 2019 was calculated based on data from Slager et al. (2024), and governance type was determined from Desbureaux et al. (2025). Row sums with shares do not add up to 1 due to unclassified boundary points.*

	Consolidated	Contained	Dormant	Exhausted	Sprawling	Points (#)
IUCN categories						
Ia	0.04	0.07	0.70	–	0.19	27
II	0.04	0.04	0.77	0.02	0.08	325
IV	0.11	0.06	0.53	0.10	0.14	288
VI	0.13	0.03	0.61	0.09	0.14	115
Accessibility (h to city)	7.11	4.75	15.85	4.6	4.21	–
Altitude (m)	757.62	1110.22	632.95	930.43	1053.5	–
Logging road distance (km)	6.02	3.91	12.96	6.73	2.24	–
New forest road w/in 5km						
No	0.09	0.05	0.65	0.06	0.11	714
Yes	0.06	0.04	0.59	0.02	0.22	51
Road distance (km)	8.06	9.32	15.1	9.3	6.36	–
PA governance type						
co-managed	0.04	0.03	0.85	0.01	0.04	221
delegated	0.15	0.02	0.81	–	–	48
other	0.10	0.06	0.53	0.09	0.16	496

Overlapping extractive frontiers

Given that the Itombwe Nature Reserve was first established in 2006 by ministerial decree, but entered a participatory mapping process for redrawing the boundaries in 2010-2014 in response to resistance by local communities and international NGOs (Kujirakwinja et al. 2019). Therefore, the analysis in Figure A8 was rerun for the former boundaries of the reserve, although they were never accepted by local communities.

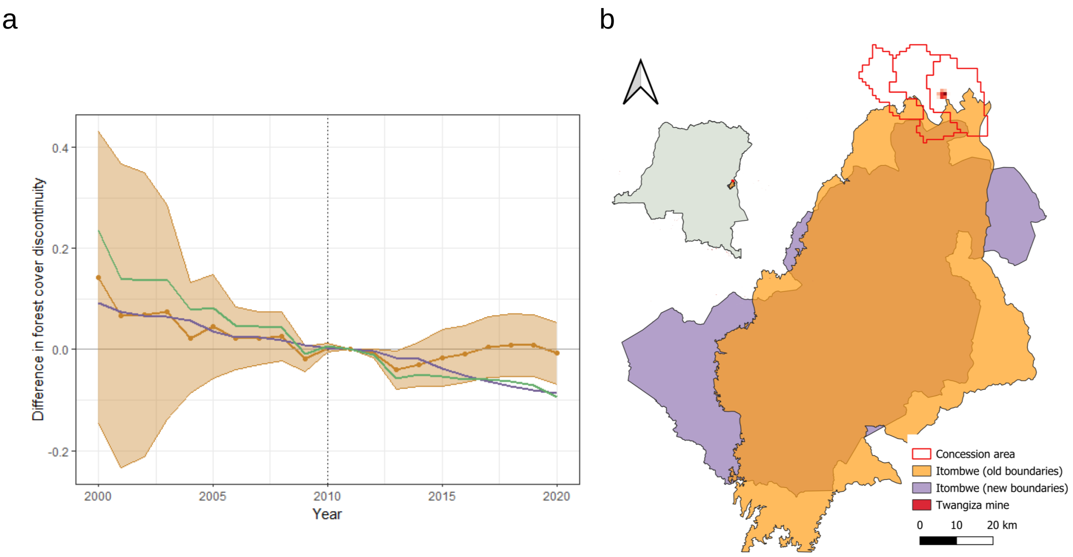

Figure 13: ***a*** *Dif-in-Disc estimates for previous Itombwe Nature Reserve boundaries. Purple line indicates forest cover outside, green inside.* ***b*** *Map with the location of the Twangiza mine and concession, as well as old and new boundaries of the Itombwe Nature Reserve.*

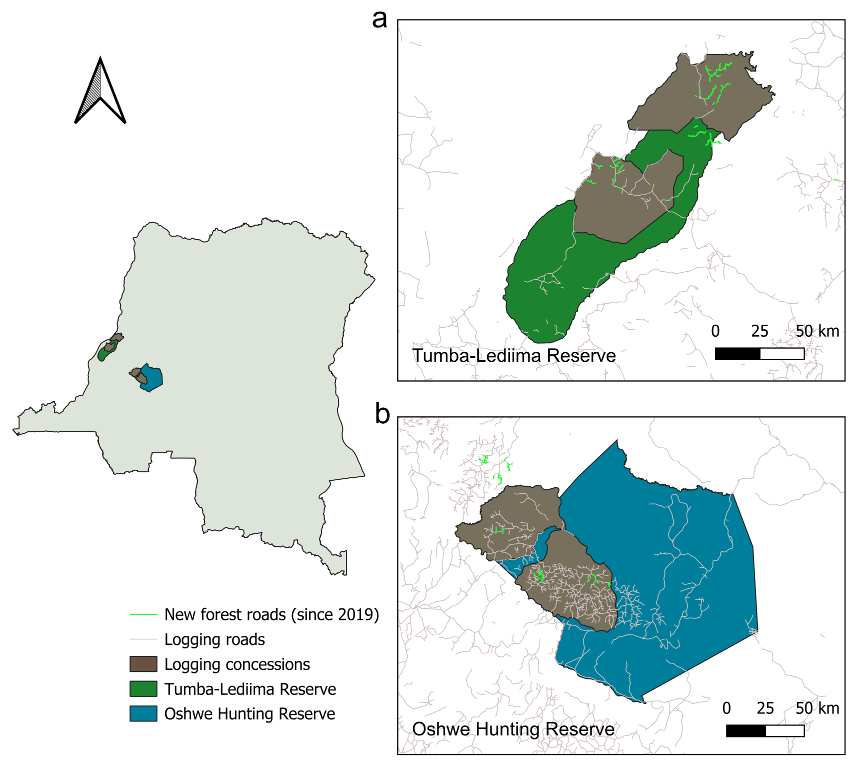

Figure 14: *Map with the location of selected logging concessions around* ***a*** *Tumba-Lediima Reserve and* ***b*** *Oshwe Hunting Reserve. Displayed logging roads from Kleinschroth et al. (2019) and newly constructed forest roads from Slagter et al. (2024).*

Software

The analysis was conducted in R version 4.4.2, using mainly the packages "tidyverse", "terra" and "rdrobust". Some adapted code chunks were used from the package "SpatialRDD", and jenks interval classification for deriving forest cover and deforestation thresholds was done with the package "classInt". The R script was partially executed on the ORION high performance cluster of the Norwegian University of Life Sciences.

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