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Title: **Mining expansion as new driver of deforestation in Côte d'Ivoire**

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Abstract

Mining is a rapidly expanding driver of tropical deforestation, yet the scale of its offsite impacts on deforestation and degradation at biome scale remain poorly quantified, especially across Sub-Saharan Africa. We focus on understanding these national-scale mining impacts in Côte d'Ivoire, which is a global biodiversity hotspot within the Upper Guinean Forests that has lost >80% of its forest cover since 1960. We combine high-resolution land-use maps with a staggered Difference-in-Differences approach, to assess deforestation and degradation dynamics after mine openings in the Côte d'Ivoire tropical moist forest zone between 2001–2020. We identify 446 mining clusters, which together directly caused the loss of 5,631 ha of forest. We estimate that the total effects extended far beyond mine sites, causing an additional 4.8 percentage-point (PP) increase in deforestation within 5 km over 10 years compared to unmined areas. Crucially, for every hectare of deforested land directly cleared for mining, 163 ha were deforested for other, indirect end uses — primarily agricultural and urban expansion linked to mining activity. There was little evidence of significant increases in forest degradation triggered by mining. Our results demonstrate that the true environmental impact of mining vastly exceeds the mine footprint itself and underscores the grave threat that mining poses to moist tropical forests. We recommend the urgent development of national policies and revisions to the mining code to acknowledge the wide area of effect that mining creates, and embedding mitigation to minimise spillover impacts into adjacent forests.

Key words: Tropical deforestation, forest degradation, Mining impacts, Indirect land-use change, biodiversity extinction crisis

1. Introduction

Tropical forests play a vital role in sustaining life on Earth ((Barlow *et al* 2018, Pillay *et al* 2022). Although they cover less than 20% of the world's land area, these forests provide habitats for ~60% of all terrestrial vertebrate species and account for more than half of the world's forest carbon stock (Dinerstein *et al* 2017, Pillay *et al* 2022, Pan *et al* 2024). Despite their ecological importance, tropical moist forests are disappearing at an alarming rate with >219 Mha (17% of the remaining tropical moist forest area) lost between 1990 and 2019 (Vancutsem *et al* 2021). This rapid deforestation is primarily driven by agriculture (Pendrill *et al* 2022), as well as infrastructure, mining, and urban expansion (Kissinger *et al* 2012).

Historically, mining has contributed less to deforestation than other drivers. However, the rapid expansion of mining in biodiversity hotspots, including the Amazon (Alvarez-Berríos and Aide 2015, Asner and Tupayachi 2017, González-González *et al* 2021),, India (Ranjan 2019), and Africa (Edwards *et al* 2014, Ladewig *et al* 2024) has become a growing concern. Globally increasing demand for minerals, driven by consumer goods, infrastructure, and renewable energy technologies, has triggered unprecedented growth in global mining activities (Haberl *et al* 2019, Wiedmann *et al* 2020). Since 2000, global raw material extraction has doubled (Krausmann *et al* 2017, Wiedmann *et al* 2020), with projections indicating further increases in the coming decades (Schandl *et al* 2016, UNEP 2019). This surge in mining disrupts natural ecosystems, accelerates biodiversity loss, reduces freshwater availability, and exacerbates environmental pollution, with impacts often extending into protected areas (Farrington 2005, Kobayashi *et al* 2014, Dezécache *et al* 2017, Northey *et al* 2017, Sonter *et al* 2017). Mining-driven deforestation is increasing particularly fast in tropical forests (Bebbington *et al* 2018),with industrial mining concessions directly responsible for over 326,400 hectares of tropical deforestation between 2000 and 2019 (Giljum *et al* 2022). However, whilst direct deforestation within mineral extraction sites is easy to monitor, it likely underestimates the true impact of mining on tropical forests because it does not account for additional deforestation indirectly caused by mining activities.

The indirect effects of mining beyond mine boundaries are more complex to assess (Butt *et al* 2013, Ferreira *et al* 2014, Sonter *et al* 2014). Mining can indirectly drive deforestation, including via ancillary activities such as road construction to access remote sites (Bebbington *et al* 2018). However, the most significant mechanisms for indirect effects are often urban expansion (including secondary settlements), agricultural encroachment (both cropland and grazing land), and other land-use changes spurred by the presence of mines (Mwitwa *et al* 2012, Werner *et al* 2019). Concerningly, indirect deforestation from mining can be far reaching and of a magnitude many times the direct deforestation. For example, in the Democratic Republic of the Congo, indirect deforestation from small-scale artisanal mines persists up to 5 km away from mines (Ladewig *et al* 2024). For large-scale industrial mines, these impacts can extend beyond 50 km (Giljum *et al* 2022) and in some cases up to 70 km from mine boundaries (Sonter *et al* 2017). Critically, indirect deforestation associated with mining can be up to 12-fold (Sonter *et al* 2017) or even 28-fold (Ladewig *et al* 2024) greater than the direct deforestation caused by mining infrastructure alone. While mining impacts can extend up to tens of kilometres beyond concession boundaries, most conservation measures implemented by governments and companies remain limited to the immediate mining sites. This gap is particularly concerning where mining expansion coincides with weak monitoring systems and limited enforcement capacity (Laurance *et al* 2012).

Mines are expanding rapidly across sub-Saharan Africa, and understanding mining-driven deforestation requires nationwide analyses to inform public policy effectively. Focusing on mines spanning 25 countries, Ahmed *et al* (2025) revealed that the average annual rate of deforestation rose from 1,665 hectares before the mines were created to 4,314 hectares after their installation, representing a more than twofold increase. However, their analysis did not provide complete national-level assessments, which is vital for policy making, governance and enforcement.

To quantify impacts at national-scale, we focus on Côte d'Ivoire, an emerging mining hotspot encompassing significant portions of West Africa's mineral-rich greenstone belts (Assie *et al* 2024, Ouattara *et al* 2021), with substantial deposits of gold, manganese, bauxite, nickel, coltan, and diamond (Bermúdez-Lugo 2011, Shaw *et al* 2022). The Ivorian mining sector is dominated by gold. Between 2015 and 2022, 93% of the 174 exploration permits issued were for gold, and in 2023 gold still accounted for 41% of all mining titles (PND 2021, TFE 2025).. Likewise, in 2021, gold represented 83.19% of the total volume of mineral ore extracted, followed by nickel (9.66%), manganese (5.90%), and bauxite (1.25%) (WU Vienna 2023). National gold production has increased almost 4-fold from 2014 to 2023 (from 13 to 51 tonnes) (TFE 2025), positioning Côte d'Ivoire as Africa's seventh-largest producer (World Gold Council 2025a). The sector consists of a small number of industrial mines (4%), but is largely dominated by smaller semi-industrial (58%) and artisanal mines (38%) with a high prevalence of illegal mining (PND 2021, TFE 2025).

We combine a new dataset that maps drivers of deforestation, including mining, across Sub-Saharan Africa (Masolele *et al* 2024) with national-scale maps obtained by this study and then use a robust staggered Difference-in-Difference (DiD) design accounting for variation in mine opening year. We do so to tackle 3 key objectives: (1) quantify direct deforestation caused by mining; (2) estimate the deforestation and (3) degradation around mining sites that is attributable to mining operations commencing.

2. Methods

2.1. Data

To capture the spatial extent of mining activities across Côte d'Ivoire, we used a new dataset from (Masolele *et al* 2024) that predicts post-deforestation land use across the whole of Sub-Saharan Africa annually between 2001-2020 at 30m resolution. The dataset first identifies areas of deforestation between 2001-2020 using global forest change data (Hansen *et al* 2013), before combining high resolution (5m) Planet–NICFI (Norway's International Climate and Forests Initiative) imagery with an active learning framework to train a deep learning model to predict post-deforestation land-use. The model assigns post-deforestation land-use to one of 15 different classes, one of which is mining, which is mapped with a 98% User's accuracy (see paper for original accuracy metrics) and defined as "Land used for extractive subsurface and surface mining activities (e.g. underground and strip mines, quarries and gravel pits), including all associated surface infrastructure". We used all mapped instances of mining by Masolele *et al* (2024) to represent our mining areas in this analysis.

Since the map of mining as a post-deforestation land-use in Africa is at the 30 m pixel resolution, it was important to group nearby mining pixels together into distinctive 'mining clusters' to be used in our analysis. To do this, we used distance-based density clustering in ArcGis to group together into single clusters all mining pixels that were within 1km of another mining pixel, only retaining individual mining clusters that consisted of at least 5 pixels. Thus, clusters represent spatially distinct areas of mining activity. This method does not require a pre-defined cluster size or shape, allowing clusters to be created that reflect the staggered growth of mining activities that often spread in particular directions (e.g. along riverbanks). This clustering step created 1115 mining clusters across Cote d'Ivoire (omitting 0.8% of mapped mining activity from Masolele *et al.*), however, since we were interested in mining-related deforestation and degradation of tropical moist forest, which accounts for the majority of the Côte d'Ivoire's forest cover, we discarded mining clusters that were either not located in the tropical moist forest biome (Dinerstein *et al* 2017), or had less than one third of tropical moist forest cover in the year 2001 (the start of the analysis) within a 5km buffer around the mine (Vancutsem *et al* 2021). These restrictions left us with a final dataset of 446 mining clusters in forested areas inside the tropical moist forest (TMF) biome.

In this study we consider deforestation (permanent conversion from forest to another land cover) and degradation (short term disturbance in tree cover canopy visible for <2.5 years) of tropical moist forest as defined by the Joint Research Council Tropical Moist Forest dataset (v1_2023; Vancutsem *et al* 2021). This pan-tropical dataset maps and monitors changes to the Tropical Moist Forest biome between 1990 to present. TMFs in this data include all closed humid forests (>90% canopy cover) and are based on satellite observations from Landsat throughout the period. The dataset characterises forests into three categories relevant to this study: undisturbed forest (no disturbance event observed throughout the whole period), degraded forest (visible tree cover disturbances that are apparent for <2.5 years), and deforestation events (visible tree cover disturbances that last >2.5 years). Disturbances in this dataset are defined as absence of tree cover in pixels that have previously been classified as TMF, with disturbance accuracy mapped at a reported 94.6% for Africa (Vancutsem *et al* 2021).

Our outcome variables consisted of deforestation or degradation of tropical moist forest between 2001-2020. To assess the spatial extent of any detected mining-related deforestation and degradation around the mining clusters, we calculated cumulative deforestation and degradation as the proportion of the number of forest pixels present in 2000 that were subsequently deforestation or degraded (Vancutsem *et al* 2021), inside concentric rings surrounding the mining clusters of the following sizes: 0-1 km, 1-2.5 km, 2.5-5 km and 5-10 km (see Figure 1 for example cluster and buffers). These metrics for deforestation and degradation were carried forward as the response variable.

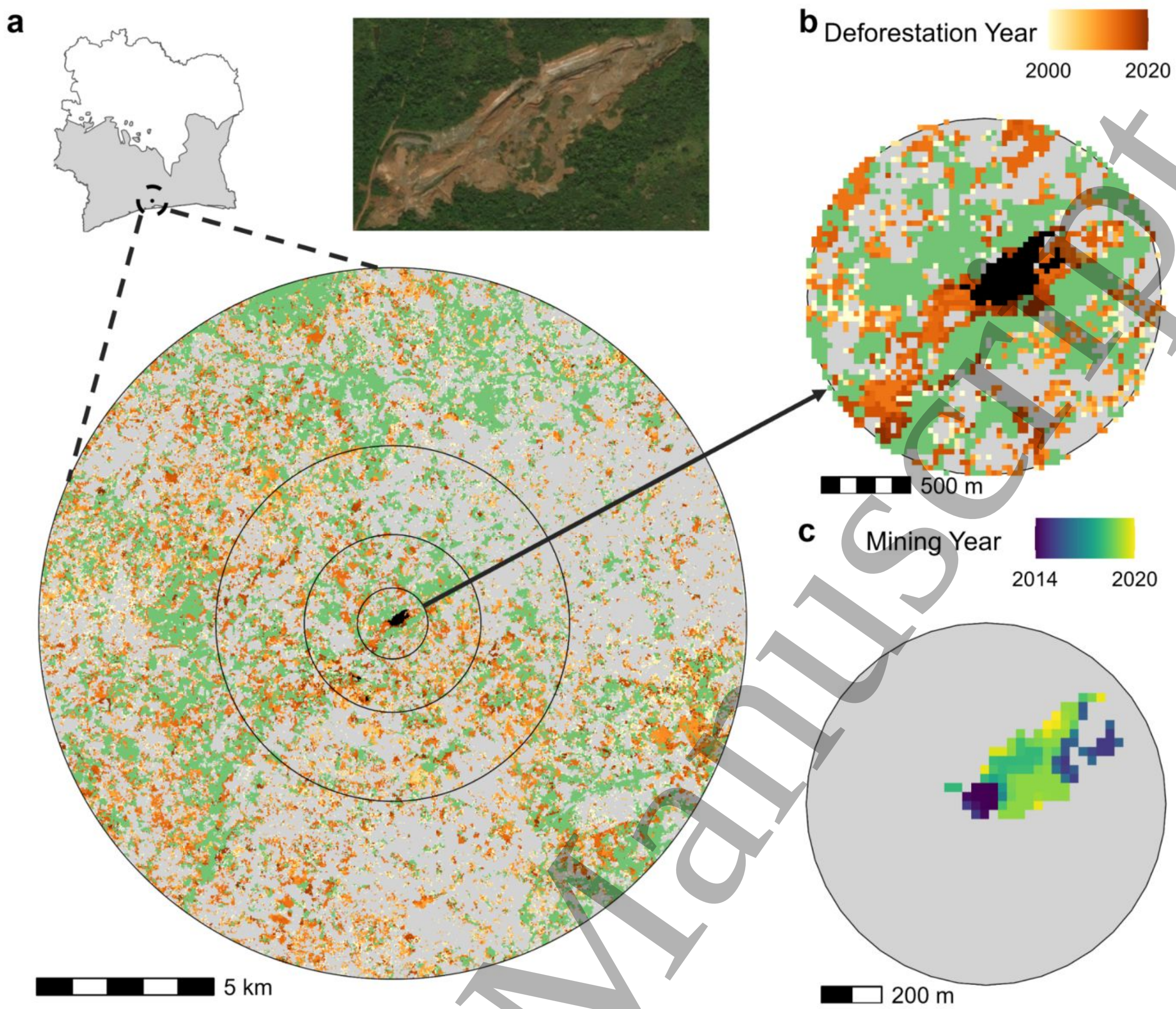

Figure 1. Schematic of our quasi-experimental design. a. Shows an example mine (central black pixels), where green denotes remaining tropical moist forest cover in 2000, yellow to red shading denotes deforestation (of any form) between 2001 and 2020, and grey denotes non-forested areas. Black concentric circles highlight the 1 km, 2.5 km, 5 km and 10 km buffer rings. b. A close up view of the 1 km buffer and the year of deforestation of the surrounding forest cover for the example mine. c. The year of forest loss for areas of forest that underwent direct mining-driven deforestation between 2001 and 2020. Satellite image of the mine in (a) is sourced from ESRI Wayback.

2.2. Difference in differences (DiD) approach

To assess the total impact of mine openings on the surrounding forest (including both the direct and indirect pathways shown in Supplementary Figure 2), we opted to use a staggered difference-in-differences quasi-experimental design, using not-yet-treated units as controls. This utilised the temporal variation in mine establishment where mine clusters are classed as treated from the year they first operate (defined as the year of the earliest appearance of deforestation due to mining in the mining cluster), and mines that are not yet established (but become treated at a subsequent time point) serve as controls. An alternate approach would be matching based and typically compares never treated sites to treated sites after employing statistical matching to balance covariates that drive variation in either the outcome or selection into the treatment group. However, even after matching on quantifiable covariates, never-treated sites may differ systematically from treated sites in ways that influence both treatment assignment and outcomes (Supplementary Figure 1; Callaway and Sant'Anna 2021, de Chaisemartin and D'Haultfœuille 2023).

Recent work has highlighted that while conventional two-way fixed effect (TWFE) estimators commonly used to implement DiD, and their extensions such as generalized TWFE DiD models, can be robust in the presence of staggered treatments (e.g. mines opening in different years), they can be biased in the presence of treatment effect heterogeneity across either time or groups (Baker *et al* 2022). We thus elected to use two recently proposed DiD estimators that have been demonstrated to be robust in the presence of heterogeneous and staggered treatment effects (Callaway and Sant'Anna 2021, Baker *et al* 2022, Gardner 2022).

The first estimator is a two-stage imputation based DiD estimator (Gardner 2022), where an initial model separately identifies cluster and year specific fixed effects that would occur in the absence of any treatment from the not-yet-treated observations. Additional covariates likely to affect the outcome can be incorporated in this first model, see *Sensitivity Analyses* for examples of this. Thus, the untreated outcome (cumulative deforestation or degradation) accounting for cluster and year fixed effects can then be imputed and removed from the observed treated outcomes. The second stage model then regresses this residualized outcome on the time since mining operations commence. Standard errors were clustered at the mine cluster level. We present the results of this approach in the main text.

The second estimator (Callaway and Sant'Anna 2021) identifies the group-time-specific average treatment effect on the treated (the $ATT\ (g,t)$ as defined in Eq. 1), where a group is defined as the year when clusters are first treated (g, mining operations commence) and time refers to the calendar year (t) this is observed. Thus, for mining operation commencing in year g, observed in year t the estimand is the difference in Y (cumulative deforestation or degradation) in year $g-1$ and t across mines that commence in year g, minus the same difference for mines that are not-yet treated (defined by D_t).

$$ATT\ (g,t) = E[Y_t - Y_{g-1} \mid G = g] - E[Y_t - Y_{g-1} \mid D_t = 0, G \neq g] \qquad \text{Eq.1}$$

These group-time ATTs thus do not assume or enforce homogenous treatment effects across all time periods or groups (mine opening years). Group-time ATTs where then aggregated into dynamic treatment effects relative to the year mining operations commenced. Standard errors were clustered at the mine cluster level. We present the results of this alternate approach in the Supplementary Information, crucially we note high agreement between the two methods.

Using the Masolele *et al* (2024) dataset we also categorized the deforested pixels around mines based on whether the post-deforestation land use was assigned to mining, or to any other classes. We respectively term these direct (e.g. mining pits, tailing ponds) and indirect deforestation (e.g. roads, infrastructure, agriculture) around mines, and assume that geographic barriers preventing access around mines are limited. To assess the relative scale of direct and indirect deforestation attributable to mining operations we estimated the ATT for mining operations on direct and indirect deforestation using the two previously described DiD estimators.

2.3. Sensitivity analyses

In addition to the alternate DiD estimator we conducted several sensitivity analyses to assess the robustness of our analytical choices. Firstly, we repeated our DiD estimations incorporating a suite of additional covariates that could plausibly influence mine effects on forest and wider deforestation trends. Time-invariant covariate values were used to prevent overcontrol bias. These covariates were travel time to the nearest settlement with a population >5000 (Nelson *et al* 2019), population density (GPWv4), distance to the nearest road (WorldPop 2016), distance to the nearest river (Lehner and Grill 2013), elevation and slope (NASADEM). Secondly, while the cut-off we use for identifying mining clusters within forest is commonly used (a third of pixels within 5 km must be TMF), we acknowledge that stricter criteria exist. Thus, we repeated the analysis considering only clusters with at least 50% TMF cover within 5 km. Thirdly, as the mining dataset used only commences in 2001, mine clusters labelled as starting in 2001 could potentially have been established before 2001 and predate this dataset, thus we repeated our DiD estimations using both estimators after removing all mines that commenced operation in the first year of the dataset.

Finally, recent work highlights the wide area of effect where mining can exacerbate deforestation and degradation (Sonter *et al* 2017, Eckert *et al* 2024), thus risking potential spillover effects between nearby mining sites. If spillover effects are present it can lead to an overestimation of a mining cluster's impact, as its own effect will be compounded by the effect of other mines in the vicinity. To address this, we use a recently proposed extension of the two-stage imputation based DiD estimator used in the main analyses (Butts 2023). For each mining cluster, we identify whether the cluster's 5km buffer intersects with another mine's 5km buffer, and thus the deforestation attributed to one mine may inflate the effect of another mine. This extension changes the first stage imputations of the outcome in not-yet treated mines to impute the outcomes for mines that are both not-yet-treated and are also not exposed to potential spillover from nearby mines. Subsequently, by including both the main treatment year (the year the mine became operational) and the spillover treatment years (the year the mines buffer first intersected another mines) in the second-stage regression we isolate both the direct effects of mining operations and the additional spillover effect attributable to other nearby mines.

3. Results

3.1. Direct mining deforestation

Between 2001 and 2020, 446 individual mine clusters were detected across the tropical moist forest zone of Côte d'Ivoire (Figure 2a). Detected mines were particularly clustered in the South-East of the country and along the southern coast. Combining post land-use deforestation data (Masolele *et al* 2024) with TMF data (Vancutsem *et al* 2021) reveals that mining infrastructure (pits, tailings, dams, etc) was directly responsible for 5631 hectares of tropical moist forest deforestation in Côte d'Ivoire between 2001-2020 (0.17% of the total national loss of tropical moist forest in this period). The number of individual mining clusters has grown steadily through time (Figure 2b), whilst direct annual deforestation through mining activity showed a strong increasing trend (Mann–Kendall test, $p < 0.01$), with the greatest annual losses seen in the most recent years (Figure 2c).

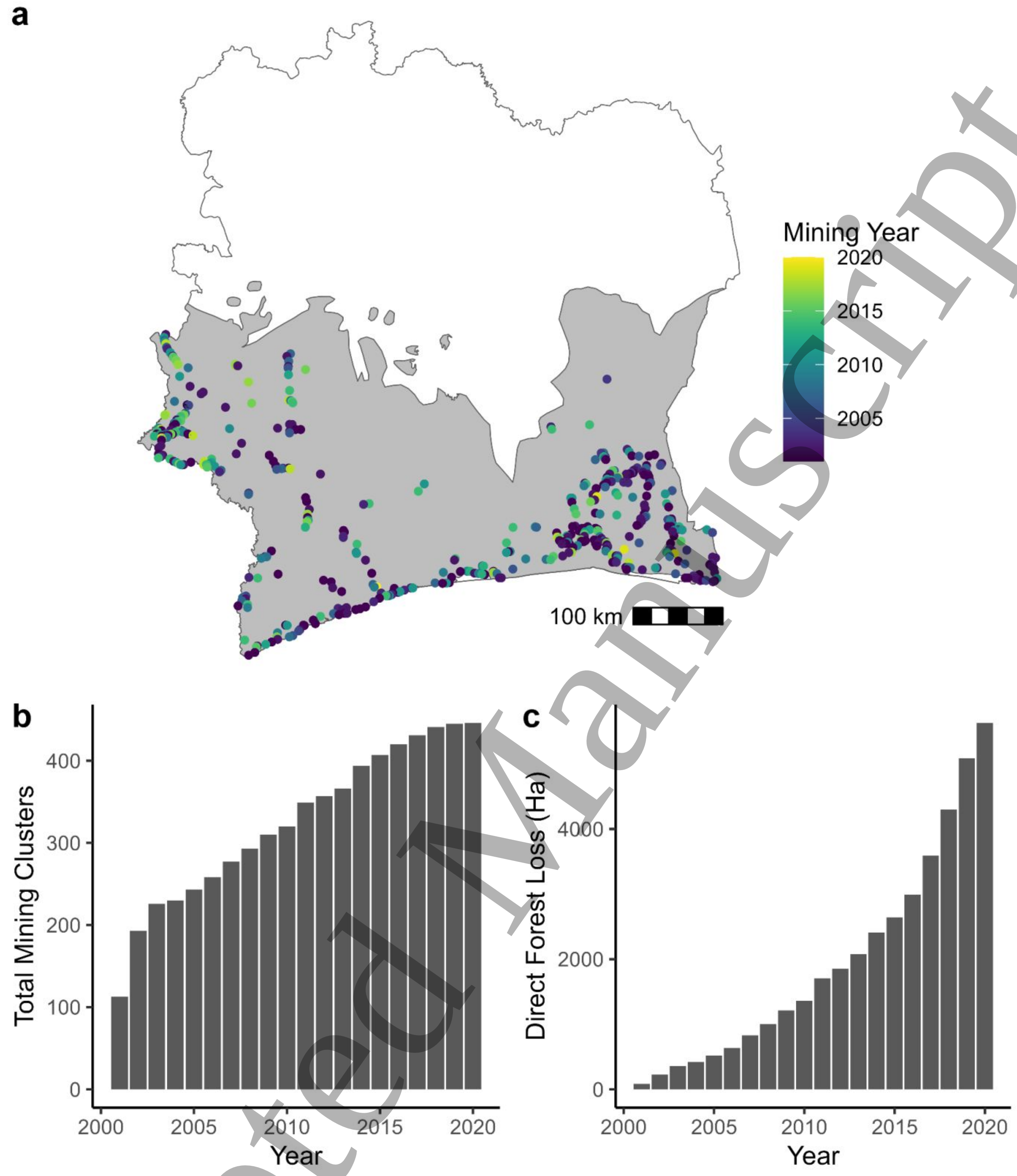

Figure 2. Distribution of mining activity across the tropical moist forest biome of Côte d'Ivoire between 2001-2020. Shown are the 446 independent mining clusters and their 5 km buffer (a), the cumulative number of active mining clusters per year (b), and the cumulative deforestation as a direct result of mining infrastructure (c). Grey area in (a) represents the extent of the tropical moist forest biome in the country.

3.2. Indirect mining-driven deforestation

Additional deforestation caused by mining was greatest in close proximity to mining sites and was greatest in the years immediately following mine opening (Figure 3). Across all distance rings tested (0 - 1, 1 - 2.5, 2.5 - 5 and 5 - 10 km), there is significant additional deforestation accumulated up to 5-years after mine opening (Figure 3a-d). This is clearest and most persistent 0-1 km from mining sites, with an additional 2.5pp of deforestation (95% CI: 1.54 - 3.47pp) occurring the year mining first starts and accumulating rapidly to 6.28pp (95% CI: 3.53 - 9.03pp) after 5-years (Figure 3a). Additional mining driven deforestation declined with increasing distance, with only an additional 1.53pp (95% CI: 0.72 - 2.33pp) detected 5 - 10 km from mines the year operations started (Figure 3d). Likewise, at this distance, a significant impact was only apparent in the first 5-years after mining, suggesting mining effects at greater distances (>5 km) waned with time.

Overall, mining triggered an immediate 2.14 percentage points (pp) (95% CI: 1.34 - 2.94pp) increase in deforestation in the year mining first starts up to 5 km from the site (Figure 3e). Deforestation continued to accumulate in mined areas in subsequent years, before plateauing ~5 years after mining began. However, adverse effects of mining remained 10-years later, with an overall 4.79pp (95% CI: 1.03 - 8.55pp) increase in accumulated deforestation within 5 km. Using a disaggregated data set of deforestation directly attributable to mining and deforestation due to other drivers, we further estimate that for each hectare of direct deforestation caused by mining, 163.0 hectares (95% CI: 158.1 - 165.9, Supplementary Figure 2) of additional deforestation is caused indirectly by mining (within the 0 - 5 km buffer), with mining activity instigating forest lost through other direct drivers (e.g. settlements, infrastructure, agriculture etc.).

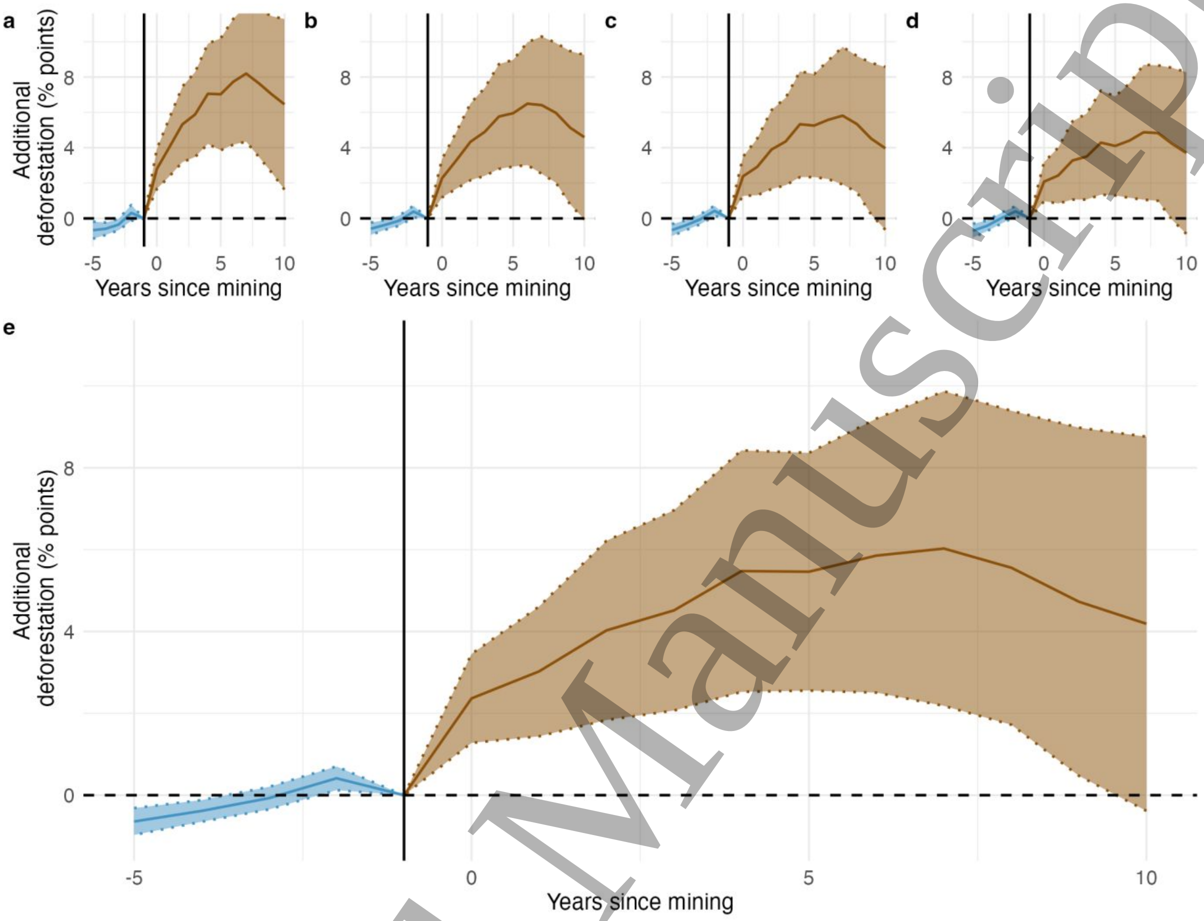

Figure 3. Impact of mining activity on cumulative deforestation. Deforestation coefficients for 0 - 1 km (a), 1 - 2.5 km (b), 2.5 - 5 km (c) and 5 - 10 km (d) buffers around mining clusters on the percentage point change scale. Panel (e) shows the deforestation coefficients for the 0 - 5 km buffer. Central solid lines are means and shaded ribbons and dotted lines are the 95% confidence interval. Values represent the additional cumulative deforestation (in percentage points) attributable to mining activities compared to not-yet-mined control sites. Deforestation before the start of mining operations is shown in blue, and deforestation after the start of mining operations is shown in brown.

3.3. Off-Site forest degradation

Forest degradation had a weaker and more uncertain response to mining operations compared to deforestation. Increases in degradation predominantly occurred in close proximity to mining sites (0-1 km, Figure 4a), with a significant increase only observed from 2-years post-establishment and peaking at a 2.29pp (95% CI: 0.41 - 4.17pp) increase in accumulated degradation after 7-years. This effect becomes both weaker and increasingly uncertain with distance from mines, with significant increases in accumulated degradation only occurring 4-7-years after mine establishment at 1 - 2.5 and 2.5 - 5 km and no impacts discernible beyond 5 km (Figure 4b-d). We note no long term (10-years post establishment) substantial increases in accumulated degradation attributable to mining at any distance from mines (Figure 4a-d). The total effect of mining on forest degradation up to 5 km from mining sites is likewise small compared to deforestation and largely uncertain (Figure 4e).

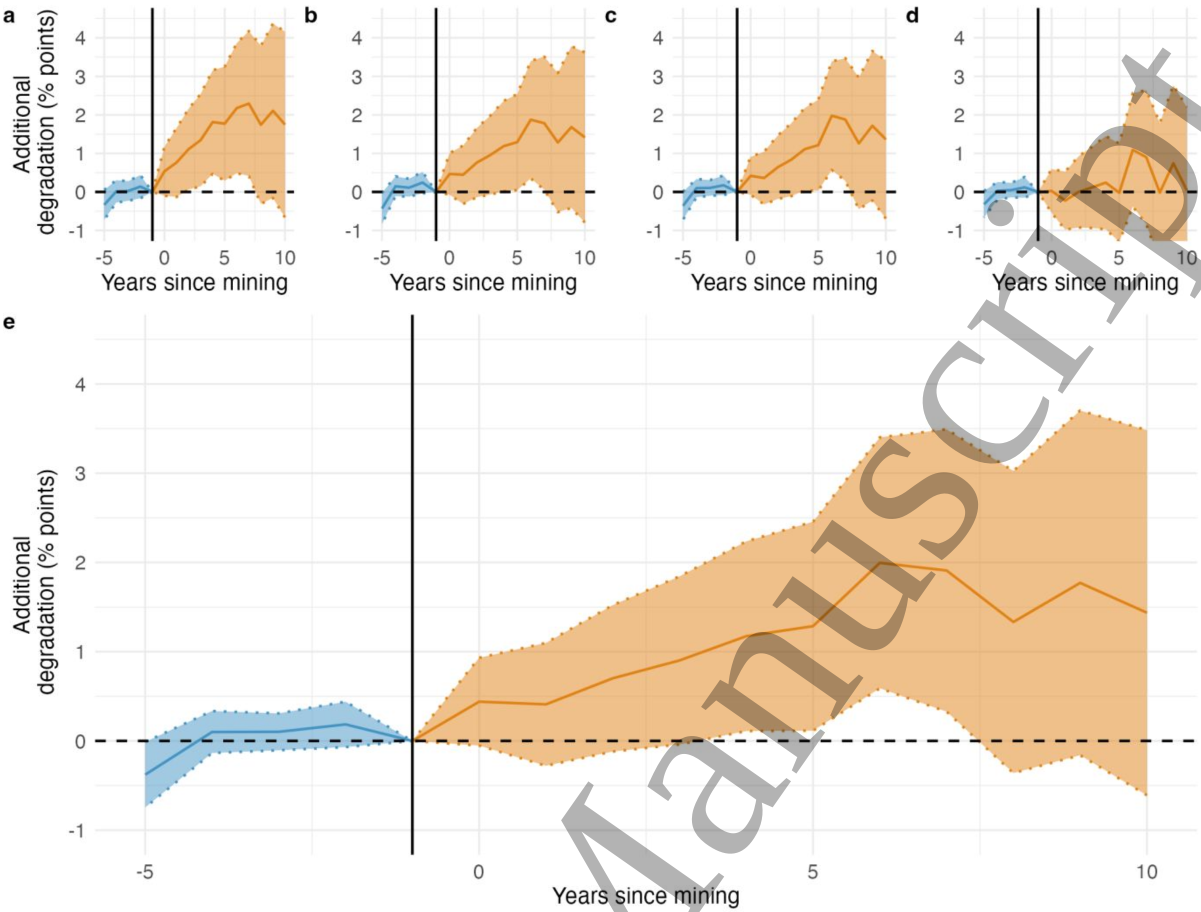

Figure 4. Effect of mining in cumulative forest degradation. Degradation coefficients for 0 - 1 km (a), 1 - 2.5 km (b), 2.5 - 5 km (c) and 5 - 10 km (d) buffers around mining clusters on the percentage point change scale. Panel (e) shows the degradation coefficients for the 0 - 5 km buffer. Central solid lines are means and shaded ribbons and dotted lines are the 95% confidence interval. Values represent the additional cumulative forest degradation (in percentage points) attributable to mining activities compared to not-yet-mined control sites. Degradation before the start of mining operations is shown in blue, and degradation after the start of mining operations is shown in orange.

3.4. Stability of our estimates

To assess the robustness of our approach, we considered a range of justifiable variations in modelling approach. Firstly, we refit all buffer-distance models using an alternate group-time-specific DiD estimator (Callaway and Sant'Anna, 2021) which yielded almost identical mean deforestation estimates with only slightly increased uncertainty noted in later years (Supplementary Figure 3). For degradation, the alternate estimator suggested greater, longer lasting and more spatially extensive increases in forest degradation (Supplementary Figure 4). We also refit all models incorporating a suite of covariates likely to influence deforestation and degradation as fixed effects, the results of which also align closely with the main text results (Supplementary Figure 5 and 6). Altering our definition of forest mining sites to require 50% (as opposed to a third) of pixels being classed as moist forest within 5km of a mining cluster in order for a mine to be classed as a forest mining site led to increased estimates of cumulative deforestation and degradation at all distances and post-mining years (Supplementary Figure 7 and 8). Since the mining dataset only commences in 2001, many of the mine clusters labelled as starting in 2001 could have been established before 2001 and predate this dataset. To account for this, we also refit the deforestation and degradation models using both DiD estimation approaches with these mines removed, the results of which remain largely unchanged (Supplementary Figure 9). Finally, due to the spatial clustering of mine sites, there is substantial risk of spatial spillover from multiple mines establishing in different years affecting the same area (e.g. a given mine may exist within the 5 km buffer of another mine). Utilising an extended spillover-robust version of the 2-stage DiD estimator, we find the direct effects comparable with previous estimates, and in almost all years post-mining there is no significant indirect effect of nearby mines on deforestation or degradation suggesting spatial spillover is minimal in this context (Supplementary Figure 10). The congruency across reanalyses indicates a high-level of confidence the estimates presented in the main text are robust and potentially conservative.

4. Discussion

Using 20-years of forest cover data and a staggered difference-in-differences with not-yet-treated units as controls design, we show that, between 2001 and 2020, mining activities directly contributed to the loss of 5,631 hectares of Ivorian tropical moist forest, accounting for 0.17% of the country's total deforestation. We found that alongside direct deforestation, mining indirectly drives significant increases in deforestation up to 5 km from mining sites. The strongest effects were observed nearest to mines and in the years immediately after establishment. Furthermore, for every hectare directly lost to mining, we estimate that more than a hundred hectares are indirectly lost, mainly due to related activities. Mining had less impact on forest degradation, with highly uncertain and statistically insignificant impacts across buffer rings. These results suggest that we should revise the way we assess the environmental costs of mining.

4.1. Mining has substantially larger indirect effects on deforestation than direct

Combining post land-use data (Masolele *et al* 2024) with TMF forest loss data (Vancutsem *et al* 2021) reveals over 5600 hectares of direct deforestation due to mining in Côte d'Ivoire between 2001 and 2020, exceeding the 1109 hectares estimated by Giljum *et al* (2022) in a pantropical assessment of deforestation caused by industrial mining between 2000 and 2019. This difference is likely explained by the fact that Giljum *et al* (2022) only considered industrial mines (4%), while we included all mine types in Côte d'Ivoire i.e., industrial, semi-industrial (58%) and artisanal mines (38%). Beyond direct deforestation, we demonstrate substantial additional indirect deforestation attributed to mining, which at the national scale will far exceed the 5600 hectares of deforestation that occurred as a direct result of mining activities.

Considering all scales of mining in Côte d'Ivoire and focusing only in the TMF biome of the country we found a high ratio of 1:163 hectares of indirect forest loss. In contrast, Giljum *et al* (2022) who examined a similar period but focused exclusively on industrial mines nationwide, did not capture the extent of the indirect amplification effects revealed in our analysis. This suggests that, in the context of Côte d'Ivoire, small-scale mining may drive more indirect deforestation than industrial mining. This is likely due to the dispersed nature of artisanal and semi-industrial operations and the fact that mining expansion in developing countries often occurs under weak regulatory oversight and limited enforcement capacity (Laurance *et al* 2012). Although industrial mining is well-known for its significant impact on deforestation, artisanal mining can also contribute to forest loss through different mechanisms, such as localised clearance rather than extensive infrastructure development. Enforcement of environmental regulations and social protections is significantly more challenging for artisanal mines due to their diffuse and informal nature (Espin and Perz 2021, Armah *et al* 2013). Land invasions, increased hunting, and the dumping of tailings can damage already precarious livelihoods of farmers in areas impacted by artisanal mining expansion (Adranyi *et al* 2023, Ofosu *et al* 2020, Spira *et al* 2019). Further research is needed to quantify the distinct deforestation patterns of artisanal mining and better understand the best approaches to reduce harmful land use decisions and negative outcomes for nature and people.

Our estimate of 163 hectares of tropical moist forests indirectly lost for each hectare directly exploited is higher than the ratio of 28:1 estimated for artisanal mining in the DRC (Ladewig *et al* 2024) and the ratio of 12:1 for industrial mining in Brazil (Sonter *et al* 2017). However, these comparisons do not mean that Côte d'Ivoire is directly comparable to the DRC or Brazil in terms of forest loss. Indeed, deforestation has been much more extensive in those countries: 73.3 Mha in Brazil and 21.1 Mha in the DRC, compared to 3.99 Mha in Côte d'Ivoire between 2002 and 2024. The gigantic forests of Brazil and the DRC are subject to massive losses, whereas those of Côte d'Ivoire, though smaller in absolute terms, are under proportionally much higher pressure. Thus, although the absolute area of forests lost between 2002 and 2024 was far greater in Brazil and the DRC than in Côte d'Ivoire, the proportional loss relative to national forest cover was much higher in Côte d'Ivoire (27%) than in Brazil (14%) and the DRC (11%) (Global Forest Watch 2025). These figures, however, should be interpreted with caution and should not justify a direct comparison between countries. For example, in the Brazilian state of Rondônia, relative forest loss reached 28% of its 2000 forest cover, equivalent to 5.3 Mha lost between 2001 and 2024 (Global Forest Watch 2025), which is slightly higher than the case of Côte d'Ivoire.

The difference highlighted in our study concerns mainly the multiplication factor of indirect forest losses as a function of the rate of losses directly linked to mining. These differences can likely be explained by two key elements: (i) the composition of mining activities, and (ii) the baseline pressure on deforestation. Unlike studies in Brazil and the DRC, which focused on specific mining sectors, our analysis considers the synergistic presence of industrial, artisanal, and semi-industrial operations, within a context where relative deforestation pressure is already very high in Côte d'Ivoire. This underlines the importance of assessing mining impacts at a more local scale, where broader deforestation trends may significantly amplify the indirect impacts of mining on forests. The higher ratio observed in CIV implies that the indirect footprint of mining grows more than proportionally relative to its direct forest loss, which may be amplified in settings where baseline deforestation dynamics are already severe. The 2014 Mining Code reforms, spurred a 54.66% rise in tax revenues (2016–2020) leading to a 90.26% increase in mining jobs (PND 2021), and heightened interest in the gold sector—potentially escalating indirect mining-related deforestation. As a major global producer of cocoa, cashew, and rubber, Côte d'Ivoire has already experienced vast forest areas being converted into agricultural land (Cuny *et al* 2023, Aké Assi 1990). Increased fragmentation and human presence driven by mining activities are likely to further intensify deforestation (Lewis et al., 2015).

4.2. Indirect effects on degradation are limited

While deforestation implies permanent land-use conversion detectable by remote sensing (Hansen *et al* 2013), degradation refers to partial, often reversible disturbances (Vancutsem *et al* 2021, Souza *et al* 2024). Our results suggest that mining drives no or limited additional degradation. There are two likely explanations for this. Firstly, remote sensing detects forest degradation due to selective logging and fires (Hosonuma *et al* 2012, Matricardi *et al* 2013) but can struggle to quantify diffuse forest degradation from shifting cultivation and fuelwood collection (Bullock *et al* 2020), especially in humid tropical forests where regrowth is fast (Poorter *et al* 2021). Secondly, high levels of already degraded forest likely limit the additional impact mining can have. These biases, combined with the importance of timber (46%) and fuelwood (23%) as drivers of degradation in Côte d'Ivoire (Cuny *et al* 2023), highlight the need to integrate advanced technologies for more accurate assessments of mining-related forest degradation.

4.3. Repercussions for conservation on biodiversity, livelihoods, and human health

The scale of mining-driven deforestation in Côte d'Ivoire poses a severe threat to biodiversity (Lamb *et al* 2024). Côte d'Ivoire hosts the Upper Guinea Forest (UGF) global conservation hotspot, including a range of endemic fauna and flora already under pressure from agricultural expansion (Cuny *et al* 2023). Several species already face threats from habitat loss and exploitation. The increasing mining footprint further exacerbates these threats (Spira *et al* 2019) particularly when we consider the illegal encroachment into and in the periphery of protected areas, where most of the threatened species' populations are confined (Campbell *et al* 2008, Fischer 2005, Ouattara *et al* 2018). The larger indirect effects of mining on deforestation that we observed in Côte d'Ivoire could also be observed in other countries hosting the Upper

Guinean Forest (Togo, Ghana, Liberia, Sierra Leone, and Guinea), where the mining sector is developing under relatively weak regulatory frameworks (Laurance *et al* 2012).

Chemical pollution from mining also highly threatens biodiversity (González-González *et al* 2021, Ranjan 2019), particularly in wetlands with sensitive river systems, as is the case in Côte d'Ivoire (FAO 2005, Rebello *et al* 2021). As the seventh-largest gold producer in Africa (World Gold Council 2025a), the country is largely dominated by semi-industrial and artisanal gold mines (TFE 2025). Given that artisanal mines generate 37% of global mercury (Hg) emissions (Seccatore *et al* 2014) it is thus subject to significant mercury pollution. These contaminants affect the air, water and soil, posing a threat to health (Patiño Ropero et al 2016, Junge et al 2017, Zhao et al 2019) and ecosystems. Hg, particularly in its organic form as methylmercury (MeHg), is a potent neurotoxin that accumulates in human tissues mainly through fish consumption. It affects the nervous, renal, and cardiovascular systems, with foetuses and infants being the most vulnerable (Dorea and Donangelo 2006, Genchi *et al* 2017, Shinoda *et al* 2023). In ecosystems, MeHg bioaccumulates and biomagnifies in aquatic environments along food chains. This process impairs reproduction and survival in fish, birds, and mammals, and alters microbial activity and nutrient cycling, thereby destabilizing ecosystem functions (Scheuhammer *et al* 2007, Driscoll *et al* 2013). Its impact is associated with its persistent nature and long-range transport. In Côte d'Ivoire, artisanal gold mining results in significant mercury exposure, posing serious risks to human health in communities where these activities are concentrated (Mason *et al* 2019). This threat is exacerbated by the growing demand for gold, which has seen its price surge by over 40% since the end of 2023, reaching $3,000 per ounce by mid-March 2025 (Collyns and Klein 2025), with a further 26% increase in US dollar terms during the first half of the year (Mason *et al* 2019). Considering the large extend of indirect mining effect (Sonter et al 2017, Giljum et al 2022), the repercussions on biodiversity, livelihoods, and human health are likely to be higher than expected particularly in countries with low enforcement capacity (Laurance et al 2012).

4.4. Indirect effects of mining challenge policy solutions

Although mining companies implement mitigation measures for direct impacts, few address their responsibilities regarding indirect deforestation (NYDF Assessment Partners 2020). In Côte d'Ivoire, environmental impact assessments (EIAs) focus on direct deforestation, while indirect impacts are primarily assessed for water pollution yet it is clear that the effects of mining on deforestation extend far beyond the boundaries of extraction sites (Sonter *et al* 2017, Giljum *et al* 2022). On another side, a broader view may be warranted, since we cannot rule out that some of the additional deforestation and degradation we detect may have occurred elsewhere in the absence of mining. At a regional or national scale, mining may shift pressure on forests rather than increase it. This point was made by Devenish *et al* (2024) in Madagascar, where a sapphire rush did not result in additional forest loss at the watershed scale, highlighting the importance of multi-scale assessments of mining impacts. This reinforces the need to integrate indirect land-use changes into public policy, particularly in national reforestation strategies.

Given that the impacts of mining can extend up to 10 km from the mining site, there is urgent need for revision of the regulatory framework governing mining in Cote d’ Ivoire, especially the buffer zone of only 100 m around protected areas established in Article 113 of the mining code (Law No. 2014-138 of March 24, 2014 establishing the Mining Code of the Republic of Côte d'Ivoire). To account for indirect impacts of mining, buffer zones around protected areas should instead be at least 1 km and extended further according to the ecological sensitivity of the area. In many sub-Saharan countries including Côte d'Ivoire, mining reforms have attracted multinational companies, while neglecting the needs of artisanal miners (Sauerwein, 2020) who are not subject to environmental and social impact assessments. For artisanal mining in particular, appropriate policies are needed to address its localised impact while supporting sustainable livelihoods. As shown in Ghana, poor institutional coordination, insufficient human and logistical resources, and a lack of political will tend to limit the effectiveness of environmental policies (Tuokuu et al., 2018). The balance between conservation and socio-economic development remains a key challenge for the sustainable management of forest resources and habitats.

5. Conclusions and recommendations

Côte d'Ivoire's mining sector is set for substantial growth, fuelled by investor-friendly reforms such as the 2014 Mining Code. This boom threatens to accelerate deforestation in a country which has already lost 80% of its primary forest since 1960, as mining concessions increasingly encroach on remaining forest ecosystems. We highlighted the role of the mining sector as a major driver of significant and spatially extensive deforestation in Côte d'Ivoire. Having already lost vast forest areas to agriculture, the country now faces a new threat from mining, which is increasing fragmentation and human presence to further intensify deforestation.

There is a need the need to adopt evidence-based policies on mining that address both direct and indirect impacts. Market-based conservation mechanisms, which rely on compensation approaches, have limitations when it comes to the mining sector. Indeed, given that it is difficult to measure the indirect impacts of mines on deforestation and forest degradation, it would therefore be difficult to reliably offset the indirect impacts associated with mining. If such policies are adopted, they must incorporate much larger compensation areas to reflect the scale of indirect deforestation. Conversely, regulatory landscape approaches—which plan land use on a larger scale and take ecological connectivity into account—appear to be more effective, as they simultaneously integrate direct and indirect impacts into continuous spaces.

Our study finally raises critical questions about how biodiversity responds to fragmentation and pollution in areas surrounding mining sites. By restricting new mining operations in biologically and socio-culturally significant areas and fostering alternative conservation-compatible activities, governments and conservation actors can help protect West Africa's fragile landscapes from further degradation while promoting more sustainable development trajectories.

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Data Availability

The data used in this study is open access and available for download online. The mining dataset from Masolele *et al.* is available at https://zenodo.org/records/11065705, and the TMF data is available at https://forobs.jrc.ec.europa.eu/TMF/data#downloads.

Justification statement

Mining is a major driver of tropical deforestation, with effects that extend far beyond the mine site. In Côte d'Ivoire, a biodiversity hotspot that has lost over 80% of its forests since 1960, mining has directly caused 5,631 ha of deforestation but has indirectly triggered 163 ha of off-site forest loss for every hectare mined, mainly due to agricultural and urban expansion. We show that mining has indirectly led to a 4.8 percentage point increase in deforestation within a 5 km radius over the space of a decade. These results reveal that the true environmental toll of mining is far greater than its immediate footprint, and that it poses a serious threat to tropical forests. We urge policymakers, mining industries and conservation organizations to implement mitigation strategies that take account of these large-scale spillover effects to preserve the remaining forests.

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