RESEARCH ARTICLE

Journal of Applied Ecology | BRITISH ECOLOGICAL SOCIETY

Limited biomass recovery from gold mining in Amazonian forests

Michelle Kalamandeen[1,2,3] | Emanuel Gloor[1] | Isaac Johnson[4] | Shenelle Agard[5] | Martin Katow[6] | Ashmore Vanbrooke[7] | David Ashley[1] | Sarah A. Batterman[1,8,9] | Guy Ziv[1] | Kaslyn Holder-Collins[10] | Oliver L. Phillips[1] | Eduardo S. Brondizio[11] | Ima Vieira[12] | David Galbraith[1]

[1]School of Geography, University of Leeds, Leeds, UK; [2]Department of Plant Sciences, University of Cambridge, Cambridge, UK; [3]Living with Lakes Centre, Laurentian University, Sudbury, ON, Canada; [4]Guyana School of Forestry, Annandale, Guyana; [5]Guyana Geology and Mines Commission, Georgetown, Guyana; [6]St Cuthbert's Village, Demerara-Mahaica, Guyana; [7]Guyana Forestry Commission, Georgetown, Guyana; [8]Cary Institute of Ecosystem Studies, Millbrook, NY, USA; [9]Smithsonian Tropical Research Institute, Ancon, Panama; [10]Department of Biology, University of Guyana, Georgetown, Guyana; [11]Department of Anthropology, Indiana University, Bloomington, IN, USA and [12]Museu Paraense Emilio Goeldi, Belém, Brazil

Correspondence
Michelle Kalamandeen
Email: mikaguyana@gmail.com

Abstract

1. Gold mining has rapidly increased across the Amazon Basin in recent years, especially in the Guiana shield, where it is responsible for >90% of total deforestation. However, the ability of forests to recover from gold mining activities remains largely unquantified.

2. Forest inventory plots were installed on recently abandoned mines in two major mining regions in Guyana, and re-censused 18 months later, to provide the first ground-based quantification of gold mining impacts on Amazon forest biomass recovery.

3. We found that woody biomass recovery rates on abandoned mining pits and tailing ponds are among the lowest ever recorded for tropical forests, with close to no woody biomass recovery after 3–4 years.

4. On the overburden sites (i.e. areas not mined but where excavated soil is deposited), however, above-ground biomass recovery rates (0.4–5.4 Mg ha^{-1} year^{-1}) were within the range of those recorded in other secondary forests across the Neotropics following abandonment of pastures and agricultural lands.

5. Our results suggest that forest recovery is more strongly limited by severe mining-induced depletion of soil nutrients, especially nitrogen, than by mercury contamination, due to slowing of growth in nutrient-stripped soils.

6. We estimate that the slow recovery rates in mining pits and ponds currently reduce carbon sequestration across Amazonian secondary forests by ~21,000 t C/year, compared to the carbon that would have accumulated following more traditional land uses such as agriculture or pasture.

7. *Synthesis and applications*. To achieve large-scale restoration targets, Guyana and other Amazonian countries will be challenged to remediate previously mined lands. The recovery process is highly dependent on nitrogen availability rather than mercury contamination, affecting woody biomass regrowth. The significant recovery in overburden zones indicates that one potential active remediation strategy to promote biomass recovery may be to backfill mining pits and ponds with excavated soil.

KEYWORDS

Amazon, biomass recovery, forest recovery, forest restoration, gold mining, land use change, secondary forests, tropical forest

1 | INTRODUCTION

Within the last decade, gold mining activity has become more prevalent across the Amazon Basin, to the extent that it is now the principal driver of deforestation in several northern Amazonian countries and a significant factor elsewhere (Alvarez-Berríos & Mitchell Aide, 2015; Asner, Llactayo, Tupayachi, & Luna, 2013; Guyana Forestry Commission & Indufor, 2012). In Guyana, for example, gold mining led to ~57,000 ha of forest lost between 2010 and 2017 (Guyana Forestry Commission, 2018), representing an area of similar magnitude to that in Peru, where gold mining-related deforestation was estimated at >60,000 ha within the same timeline (Espejo, Messinger, Román-Dañobeytia, Ascorra, & Luis, 2018). Much of this forest loss stems from artisanal and small-scale miners, who respond rapidly to increases in international gold prices (Howard et al., 2011). These mining activities invariably lead to widespread environmental damage and deforestation, resulting in substantial soil erosion and contamination, increased forest fragmentation and mercury pollution of rivers and streams (Castilhos et al., 2015; Dedieu, Allard, Vigouroux, Brosse, & Céréghino, 2014; Sonter et al., 2017; Veiga, Maxson, & Hylander, 2006). Mercury bioaccumulates along the aquatic food web and the consumption of contaminated fish, an integral part of human diet, can result in fatal damage to humans' central nervous system in affected regions (Gutiérrez-Mosquera et al., 2018). Further ecological impacts may arise from hunting and/or land clearance to support mining settlements (Sonter, Ali, & Watson, 2018).

The extent of soil physical damage and chemical contamination associated with gold mining activities sets it apart from other traditional deforestation drivers such as conversion to pasture or small-scale agriculture, which generally do not significantly alter soil structure or nutrients (e.g. nitrogen and phosphorus; Santos-Francés, García-Sánchez, Alonso-Rojo, Contreras, & Adams, 2011; Wantzen & Mol, 2013). The excavation of topsoil associated with mining renders the soil deficient in organic matter and nutrient content (Roman-Danobeytia et al., 2015). As such, gold mining is expected to arrest forest recovery rates compared to other land uses. Additionally, the toxicity from mercury applied during mining may be harmful to microbes, adversely affecting plant physiology and metabolism once threshold soil concentrations are breached (Beauford, Barber, & Barringer, 1977).

Despite the importance of mining as a major driver of tropical deforestation, its impacts on forest biomass accumulation remain largely unquantified. We are aware of only one previous field study that has attempted to evaluate forest recovery following deforestation from gold mining in the Amazon (Peterson & Heemskerk, 2001), but this study performed only visual assessments of vegetated area following 1–4 years of recovery with no quantitative measurements. With the increasing availability of high-resolution satellite imagery, other authors such as Novoa, Finer, and Román (2016) also visually assessed forest regeneration following gold mining, finding vegetation regrowth in previously mined areas in Peru. However, such studies have not incorporated field-based ground-truthing and do not consider separately the complex nature of mining sites, which typically consist of three distinct mining zones: (a) overburden: areas overlying the gold ore, including the topsoil, which are displaced during the mining process, (b) tailing pond: deposits of material remaining after the gold has been separated from the ore and (c) the mining pit. The intensity of the disturbance caused by mining is much higher in the tailing pond and mining pit. In these mining zones, recovery can be viewed as primary succession due to the negligible biological legacy that remains in the soil, resulting in a new surface for colonization that does not resemble the pre-disturbance conditions (Prach & Walker, 2019). Conditions for recovery are much more favourable in the overburden, however. Here, recovery is viewed as a secondary succession process, due to greater biological legacy remaining in the soil following the disturbance.

Several recent studies have quantified how recovering, secondary forests deliver a range of ecosystem services including maintenance of biodiversity and significant accumulation of carbon in their biomass (Anderson-Teixeira et al., 2013; Chazdon et al., 2016; Lohbeck, Poorter, Martínez-Ramos, & Bongers, 2015; Poorter et al., 2016). Indeed, across the tropics, secondary forests are thought to constitute a significant carbon sink, which in some estimates are even larger than that of intact primary forests (Pan et al., 2011). Recent syntheses of chronosequences from tropical secondary forests have concluded that secondary forest biomass recovery rates are (a) largely controlled by background rainfall conditions (Poorter et al., 2016) and (b) faster in forests previously under pasture than

agricultural production (Martin, Newton, & Bullock, 2013). Yet none of these syntheses contain chronosequences on previously mined areas, and thus we do not know how recovery of biomass and biodiversity following gold mining compares with recovery from more traditional land uses such as pasture and agriculture.

To address these critical knowledge gaps, we established forest plots in focal gold mining regions in Guyana to investigate the patterns of biomass recovery following the abandonment of gold mining activity, re-censusing them 18 months later. Here, we specifically consider (a) biomass accumulation between 3 and 4 years in recently abandoned gold mining plots, (b) how observed biomass recovery patterns from gold mining at our study sites compare with published tropical forest chronosequence data from other land uses (e.g. pasture, agriculture) and (c) which factors most limit biomass recovery in previously mined sites. Finally, having provided the first detailed field-based information on the regeneration of secondary forests after gold mining activity, we explore the implications of our findings for gold mining recovery in the wider Amazon context, and provide the first ground-based estimate of the secondary forest carbon sink foregone as a result of gold mining activities.

2 | MATERIALS AND METHODS

2.1 | Study sites and sampling design

Plots were installed in two central gold mining areas in Guyana, namely Mahdia (5°16′0.01″N 59°08′60.00″W) and Puruni (6°00′0.00″N 59°11′60.00″W, Figure 1), from January to March, 2016, and re-censused approximately 18 months (June–August 2017) later. Both sites are currently mined by artisanal and small-scale miners. At each study site, measurement plots were established on previously mined vegetation patches, ranging in age from 0.6 to 3 years since abandonment of mining activity at the time of establishment. Detailed description of the mining sites, plot selection and determination of date of abandonment is provided in Appendix S1.

Nine 0.25-ha (50 m × 50 m) plots and a control old-growth forest plot (100 m × 100 m) were established at each location. However, approximately half of all plots (five in Mahdia and four in Puruni) were re-mined before the 2017 re-census and thus were excluded in this study. Each plot was positioned to include
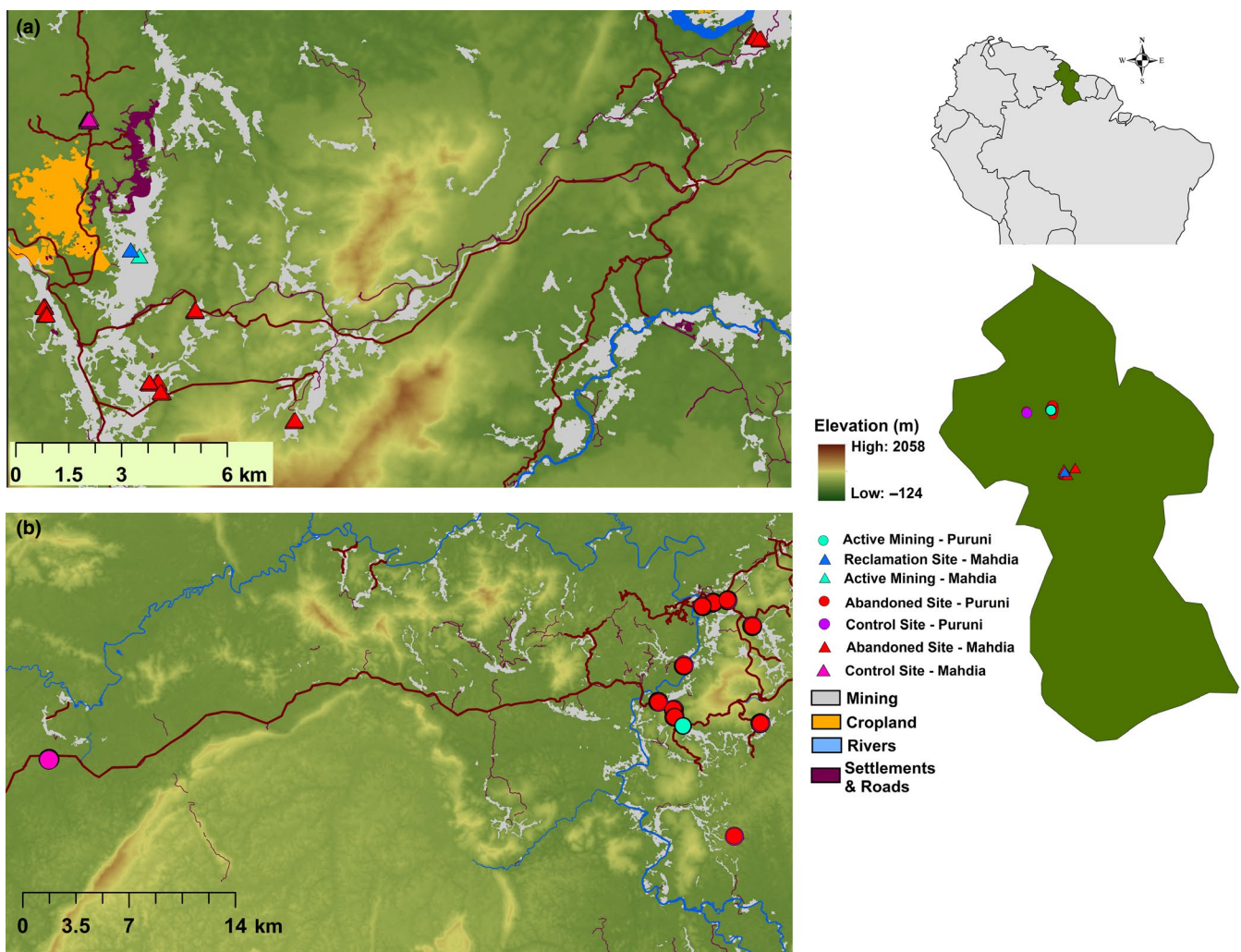
FIGURE 1 Map of field sites in Guyana: (a) Mahdia and (b) Puruni. Baseline map indicates mining, rivers, roads, cropland, and settlements from 2014 and STRM elevation (m)
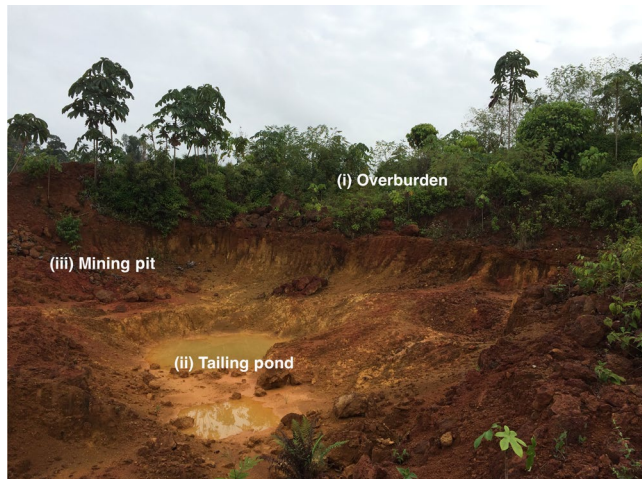
FIGURE 2 Zones of a small-scale mining process: (i) overburden: areas overlying the ore which are displaced during the mining process, (ii) tailing pond: deposits of material left over after the gold has been separated from the ore, and the (iii) mining pit. Size of entire plot is 0.25 ha. Area of zones differed from 0.05–0.18 ha for overburden, 0.03–0.13 ha for tailing pond and 0.03–0.16 ha for mining pit. Image © M. Kalamandeen (2016)

all three mining zones, that is, the mining pit, the tailing pond and the overburden (Figure 2). Within each plot and within each mining zone, three nested subplots of 3 m × 3 m and 1 m × 1 m were established. All trees >2 cm DBH (diameter at reference height of 1.3 m) were measured and tagged with a unique number in the larger 0.25-ha plot and identified to species level where possible. All tree saplings 25–200 cm tall and all tree seedlings 5–25 cm tall were identified and counted in the 3 m × 3 m and 1 m × 1 m subplots respectively. Nested subplots were also established in the 1-ha control plot, where all trees >10 cm DBH were measured and recorded and standard RAINFOR protocols applied to map, measure and identify trees (Phillips, Baker, Feldpausch, & Brienen, 2015). Tree species in all plots were identified by a local botanist. Heights of larger trees, saplings and seedlings were also measured (see Appendix S1 for details).

2.2 | Biomass calculations

To calculate above-ground biomass (AGB) of individual trees (in kg), we used the allometric model of Chave et al. (2014):

$$AGB = 0.0673 \times (\rho D^2 H)^{0.976},$$

where ρ is the wood density (g/cm^3), D is the DBH (cm) and H is the height (m). This is a generic equation for tropical forests, based on a broad set of trees including primary and secondary forest species, which has previously been used for estimating biomass in young secondary forests (Poorter et al., 2016; Rozendaal et al., 2017). Wood density values for species were obtained from the global wood density database available from the Dryad data repository (http://datadryad.org/).

Where species-specific wood densities were not available, genus or family mean wood densities were used (e.g. Baker et al., 2004). As the relative proportions of overburden, mining pit and tailing pond zones varied across plots, we rescaled our total biomass estimates for each zone so that all values are expressed on a 1-ha basis. For each plot, we calculated the dry AGB in Mg/ha, and the rate of net change in biomass between our two measurement censuses (ΔAGB), the biomass increase due to diameter growth (ΔAGB_{Growth}), the biomass increase due to recruitment of new trees ($\Delta AGB_{Recruitment}$) and the biomass loss due to mortality ($\Delta AGB_{Mortality}$), all in Mg/ha per year.

2.3 | Comparison with published secondary forest chronosequences

To place our study within the wider literature on tropical forest biomass recovery, we compared our AGB and ΔAGB values with published data for other tropical forests at 2–4 years since abandonment from chronosequence studies of recovery from prior pasture and agriculture land use. Specific criteria for inclusion of secondary forest biomass recovery studies in this analysis are provided in Appendix S1.

2.4 | Collections and analysis of soil samples

Soil samples were collected between January and March 2016 at the time of plot installation and determined for cation exchange capacity (CEC), total nitrogen (TN) content, total phosphorus (TP) content, mercury concentration and particle size distribution, following well-established protocols (see Appendix S1 for details of soil analysis).

2.5 | Statistical analysis

To examine the effects of mining regions and individual mining zones on AGB, ΔAGB, $AGB_{Mortality}$, $AGB_{Recruitment}$ and AGB_{Growth}, linear mixed-effects models were used, with a random intercept for each plot. For information on statistical treatment of biomass changes across plots, soil data and comparison with published estimates from other studies please refer Appendix S1.

3 | RESULTS

3.1 | Above-ground biomass dynamics during secondary succession

Although standing ABG biomass estimated in the control, old-growth plot in Mahdia (607 Mg/ha) was considerably higher than in the control plot in Puruni (409 Mg/ha), biomass accumulation following mining was markedly lower in Mahdia than in Puruni (Figure 3). In Puruni,

FIGURE 3 Mean above ground biomass (AGB) across mining zones (overburden, tailing pond and mining pit) in abandoned mining sites for Mahdia and Puruni. Inset maps indicate the mean AGB of tailing pond and mining pits at a larger scale. All abandoned sites were extrapolated to 1 ha
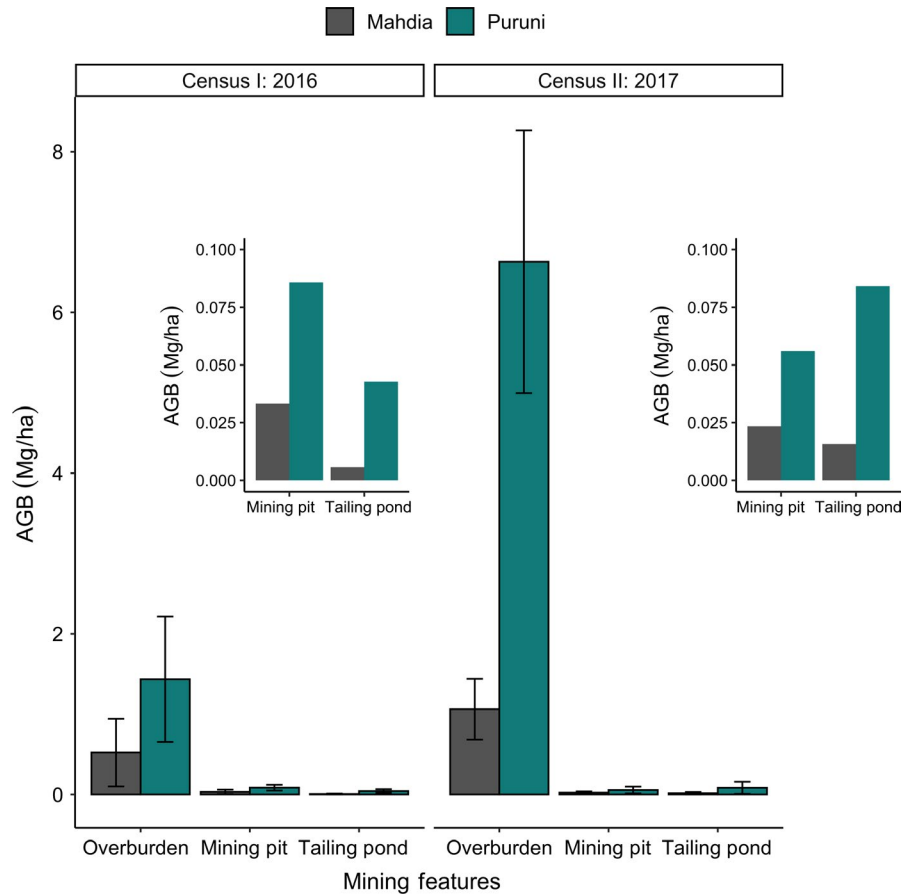
TABLE 1 Summary results from mixed-effects models for above-ground biomass (AGB), annual biomass change (ΔAGB), and biomass change resulting from recruitment ($\Delta AGB_{Recruitment}$), mortality ($\Delta AGB_{Mortality}$) and tree growth (ΔAGB_{Growth}). Zones include the tailing pit, mining pond and overburden zones at Mahdia and Puruni sites. Highlighted p-values are significant

	log($AGB_{CensusI}$)		log($AGB_{CensusII}$)		log(ΔAGB)		log($\Delta AGB_{Recruitment}$)		log($\Delta AGB_{Mortality}$)		log(ΔAGB_{Growth})	
	F-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value
Site	0.89	0.38	14.69	**0.006**	5.92	**0.04**	2.13	0.19	0.96	0.36	5.49	**0.05**
Zone	5.63	**0.02**	71.98	**0.0001**	18.59	**0.0001**	18.45	**0.0001**	1.48	0.26	8.42	**0.004**
Site × zone	0.58	0.57	14.54	**0.0004**	12.64	**0.0007**	2.37	0.13	1.85	0.19	3.98	**0.04**

overburden woody AGB following 3–4 years of forest recovery was over six times that in Mahdia (6.63 ± 1.63 Mg/ha vs. 1.06 ± 0.38 Mg/ha).

However, woody AGB was almost negligible in mining pits and tailing ponds (Figure 3). In fact, no woody species were observed in five of the nine mining pits and five of the nine tailing pond plots (i.e. woody AGB = 0) following 3–4 years of recovery and the maximum recorded woody AGB in Puruni and Mahdia was 0.39 and 0.29 Mg/ha respectively. Additionally, no trees >10 cm in height were found in any of our mining pit or tailing pond plots, even in the second census. In both sites, virtually all of the AGB in pits and ponds was accounted for by grasses and sedges, such as *Andropogon bicornis*. Based on destructive harvesting undertaken in Census II, herbaceous (non-woody) AGB was estimated at 0.03 ± 0.03 Mg/ha in Puruni and 0.17 ± 0.11 Mg/ha in Mahdia for mining pits and

0.09 ± 0.07 Mg/ha (Puruni) and 0.2 ± 0.08 Mg/ha (Mahdia) for tailing ponds.

Annual rates of overburden above-ground biomass change (ΔAGB) differed strongly between sites and zones (Table 1; Figure 4; $p < 0.0001$). ΔAGB was positive in all overburden plots except for one plot in Mahdia, indicating overall net AGB accumulation, with estimated annual ΔAGB almost 10-fold lower in Mahdia (0.36 ± 0.36 Mg ha^{-1} year^{-1}) than Puruni (3.46 ± 0.74 Mg ha^{-1} year^{-1}). ΔAGB for mining pit and pond zones was not significant in either site, remaining close to zero in both censuses. Tree recruitment ($AGB_{Recruitment}$) was the main contributor to biomass increase for overburden areas, responsible for 80% and 54% increase in Mahdia and Puruni, respectively, although $AGB_{Recruitment}$ was ~3.4 times greater ($p = 0.19$, Table 1) in Puruni (1.86 ± 0.77 Mg ha^{-1} year^{-1}) than in Mahdia
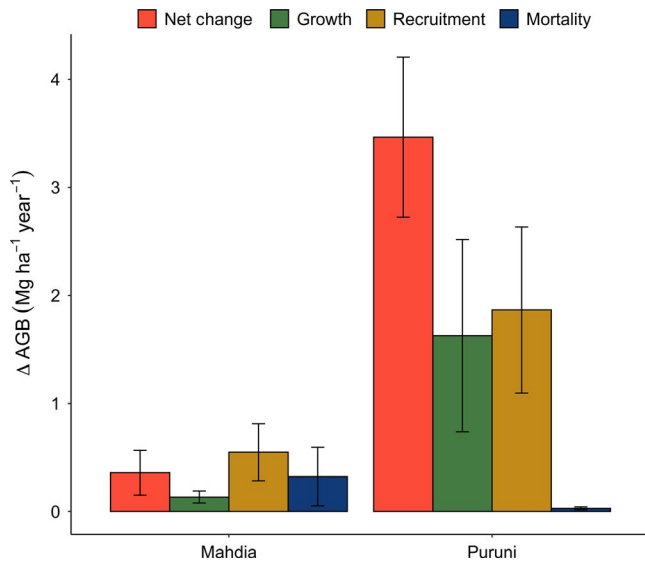
FIGURE 4 The contribution of tree growth, recruitment and mortality compared to annual biomass change on the overburden zone between Census I and Census II for abandoned mining plots in Mahdia and Puruni

(0.55 ± 0.26 Mg ha^{-1} year^{-1}). Furthermore, ΔAGB_{Growth} was ~12 times greater (p = 0.04, Table 1) in Puruni (1.63 ± 0.9 Mg ha^{-1} year^{-1}) than in Mahdia (0.13 ± 0.05 Mg ha^{-1} year^{-1}). Biomass mortality ($\Delta AGB_{Mortality}$), in contrast, were on average 11 times higher (p = 0.19, Table 1) in Mahdia (0.32 ± 0.27 Mg ha^{-1} year^{-1}) than in Puruni (0.03 ± 0.01 Mg ha^{-1} year^{-1}). ΔAGB_{Growth}, $AGB_{Recruitment}$ and $\Delta AGB_{Mortality}$ were close to zero in mining pit and tailing pond plots.

3.2 | Forest composition and structure

Overall, there was a clear difference in floristic composition between control and overburden plots (Figure S1). Fabaceae contributed most to AGB in the control plots, accounting for 63% of total AGB in both Mahdia and Puruni, followed by Lecythidaceae (16% and 26% of total AGB respectively). Differences in familial composition were prominent in overburden zones in Mahdia than Puruni. In the former, relatively small-sized individuals belonging to Ulmaceae (17 individuals) dominated and were responsible for 28% of total AGB across all plots in Census I. These were subsequently replaced by an abundance of pioneering species from Clusiaceae (106 individuals) and Urticaceae (71 individuals), accounting for 66% of total AGB in Census II. In Puruni, Fabaceae and Urticaceae were dominant and accounted for 77% of total AGB in Census I and 74% in Census II across overburden plots.

Species composition also varied considerably between sites and across plots. Important woody species present in Census I (~1–3 years post-abandonment) were *Tapirira* spp (1.3 Mg/ha), *Trema* spp (0.23 Mg/ha) and *Caryocar microcarpum* (0.2 Mg/ha) in Mahdia and *Zygia collina* (3.03 Mg/ha), *Cecropia surinamensis* (2.29 Mg/ha) and *Byrsonima aerugo* (0.7 Mg/ha) in Puruni. In Census II, species with the most biomass were *Zygia collina* (10.7 Mg/ha) and *Cecropia*

obtusa (9.09 Mg/ha) in Puruni, while *Vismia guianensis* (1.4 Mg/ha) and *Cecropia obtusa* (1.04 Mg/ha) dominated in Mahdia.

3.3 | Soil nutrient and mercury levels

To understand the differences in biomass and species composition, soil analyses were conducted, indicating significant differences across mining zones for mercury concentration (p < 0.0001), TN (p = 0.004) and exchangeable cations (p = 0.04; Table S3; Figure 5). Mercury concentrations in overburden zones were on average 55% and 72% lower than tailing ponds and pits in Mahdia and Puruni respectively (Table S3; Figure 5). Although mining pits and ponds were only abandoned for 6 months to 3 years, their mercury concentrations were two orders of magnitude lower than active site concentrations (Figure 5), suggesting that leaching of mercury from soils of mining areas happens very quickly. Further details on variation of TP, TN and soil bulk density across plots are provided in Appendix S2.

3.4 | Controls on biomass recovery

Among all predictors, TN had the strongest effect, featuring in all the best supported models for AGB, AGB_{Growth} and $AGB_{Mortality}$. It was
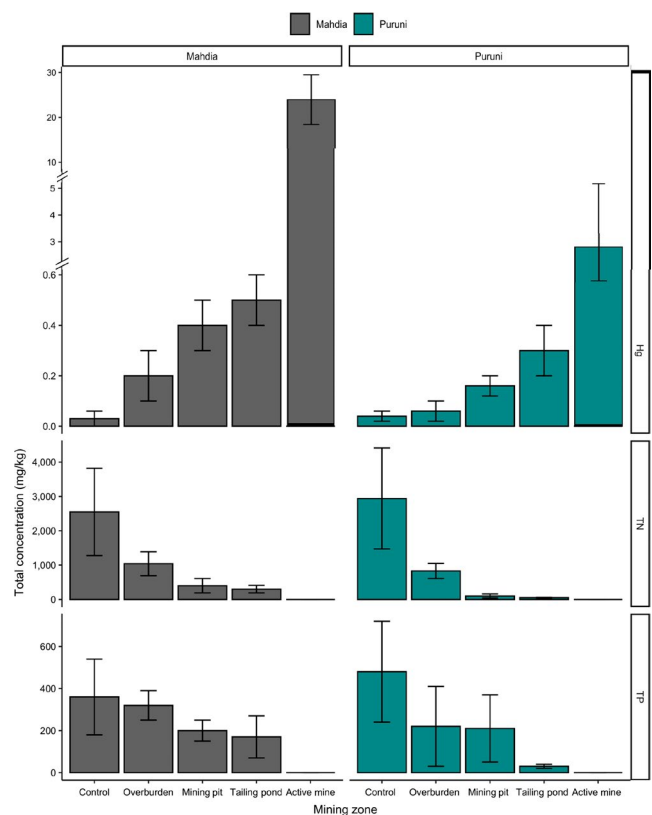

FIGURE 5 Mercury (Hg), total nitrogen (TN) and total phosphorus (TP) concentrations (mg/kg) in soils from mining zones (overburden, tailing pond, mining pit), in active mines and in old-growth forest control sites at Mahdia and Puruni

also the single most important predictor of mortality and explained 58% of the variations in AGB_{Growth}. Combinations of TN and TP, mercury and exchangeable cations had the strongest effect on $AGB_{CensusII}$ and ΔAGB (Table S6), with these models explaining 38%–43% of the observed variation in AGB. Overall predictive ability of the models was relatively weak for both $AGB_{Mortality}$ and $AGB_{Recruitment}$ where the best models explained only 20%–25% of the variation. Mercury content was found to be the most important predictor of $AGB_{Recruitment}$, implying an important role in limiting tree establishment. Phosphorus content was found to be less important than other predictors in driving variation in biomass recovery across plots.

4 | DISCUSSION

4.1 | Successional recovery post-mining

This analysis finds clear evidence that gold mining severely disrupts the succession process in former mining pit and tailing pond areas, but not on the overburden. While woody biomass recovery rates on mining pit and tailing ponds were among the lowest ever recorded, recovery rates on overburden areas (0.4–3.5 Mg ha^{-1} year^{-1}) were comparable to those from secondary forests recovering from agriculture (0.08–32.7 Mg ha^{-1} year^{-1}) and pasture (0.01–14 Mg ha^{-1} year^{-1}) in other Neotropical forest regions with similar climate and age (2–4 years post-abandonment, Figure 6). The annual rates of ΔAGB of the overburden in Puruni were also comparable to the only other multi-census data published for second-growth forests across the Neotropics (Figure 7). For instance, Puruni (3.5 Mg ha^{-1} year^{-1}) was similar in recovery to Chajul, Mexico (4.8 Mg ha^{-1} year^{-1}) while Mahdia (0.36 Mg ha^{-1} year^{-1}) was closest to Manaus, Brazil
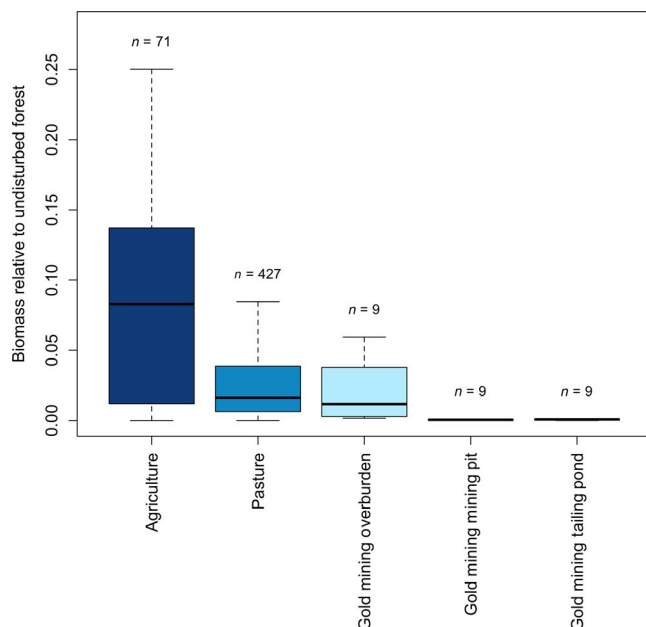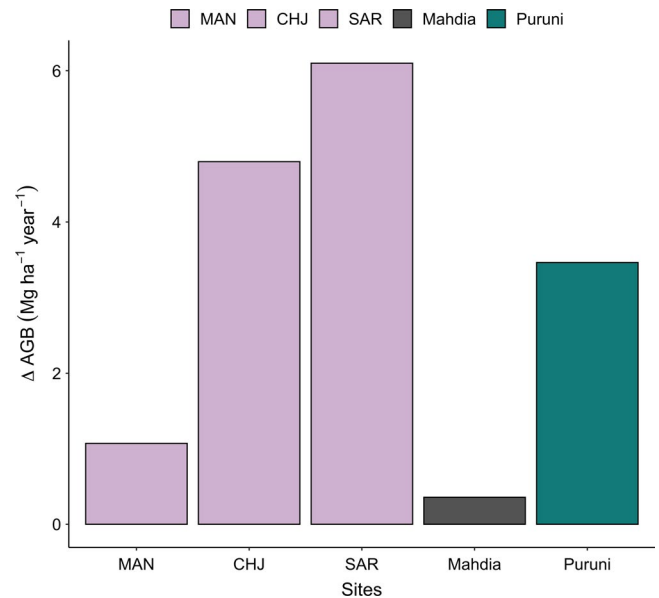
FIGURE 7 Comparison of annual biomass change of abandoned gold mining sites (overburden only) with three second-growth (5–15 years of age) wet forests across the Neotropics (MAN = Manaus, Brazil; CHJ = Chajul, Mexico; SAR = Sarapiqui, Costa Rica; Rozendaal et al., 2017)

(1.07 Mg ha^{-1} year^{-1}). Further, the biomass values for overburden plots in Mahdia and Puruni lie between the 20th and 35th percentile of plots of equivalent age (2–4 years) reported by Poorter et al. (2016) across the Neotropics.

Our results also show distinct shifts in floristic composition between the control plots and overburden zones, and across censuses in Mahdia. Tree diversity was dramatically lower in abandoned sites relative to control plots, indicating that mining may have a considerable negative effect on species diversity. In Mahdia, the observed increase in mortality rates in Census I may be driven by the shift in floristic composition from low biomass species to light-demanding fast-growing pioneer species (Gaui et al., 2019). The almost non-existent woody biomass recovery observed in the tailing ponds and mining pits is in line with *primary* succession conditions. Although, there are very few documented studies of primary succession following mining in moist forest ecosystems (Table S2), our results are qualitatively consistent with those of Peterson and Heemskerk (2001) in Suriname, who reported little woody cover in regenerating sites following gold mining activity. Beyond this study, we were unable to find other examples from moist tropical forest regions where natural regeneration was investigated following mining, as most studies related to this topic involved the examination of the effect of active restoration practices.

FIGURE 6 Recovery of biomass following different land use, relative to undisturbed reference forests

4.2 | Factors influencing biomass recovery

The lack of woody biomass recovery, particularly on the mining pit and tailing pond of gold mining sites, was found to be heavily

influenced by available TN and mercury concentrations (Table S6). Nitrogen content of soils across previously mined plots, especially the mining pits and tailing ponds, was much depleted relative to the control plots in both sites. Previously mined pits and tailing ponds contained only 10%–38% and 4%–6% nitrogen, respectively, compared to overburden plots (Table S4). Recovery of Neotropical secondary forests is known to be linked to nitrogen availability (e.g. Batterman et al., 2013; Davidson, Ishida, & Nepstad, 2004; Davidson, Reis de Carvalho, et al., 2004; Feldpausch, Rondon, Fernandes, Riha, & Wandelli, 2004; Wright et al., 2018). The stripping of topsoil from mining pits and tailing ponds substantially depletes nutrient concentrations, considerably retarding their recovery.

Although we did not find a strong role for phosphorus in driving biomass accumulation, fertilization experiments have shown that phosphorus additions to soil, when combined with nitrogen additions, can also boost biomass recovery in secondary forests (Davidson, Ishida, et al., 2004; Davidson, Reis de Carvalho, et al., 2004). In our study, the differences in phosphorus content between overburden plots, which recovered well, and plots on previously mined tailing ponds and mining pits were not as pronounced as for nitrogen. Previously mined tailing ponds and mining pits, which showed no recovery in both sites, were found to contain 64%–94% of the soil phosphorus content of the overburden plots.

While available nitrogen probably drives the differential recovery observed across mining zones, our analyses suggest that other factors likely drive the differences in accumulation rates observed between the two study sites. Indeed, overburden biomass accumulated approximately six times more slowly in Mahdia than in Puruni, despite actual nitrogen and phosphorus concentrations in overburden soils being on average higher in Mahdia than in Puruni (Table S4). Soil mercury content and exchangeable cations emerged as important secondary predictors of biomass accumulation metrics (Table S6) and may explain the observed between-site differences. Mercury concentrations on overburden soils in Mahdia were ~3.5 times greater than those in Puruni and were higher than mercury concentrations in previously mined pits and tailing ponds in Puruni. This may be due to higher mercury concentrations used in the mining process and/or the progressive frequency of mining activities occurring in Mahdia vs. Puruni.

Active mining sites were also substantially higher in mercury levels than the abandoned mining zones suggesting that the majority of the mercury is not retained locally in the soils and may be transported to surrounding soils and waterways. Other studies have shown that mercury-rich tailings have been detected hundreds of kilometres away from mining sites (Diringer et al., 2015). The potential rapid mercury leaching to neighbouring areas may lead to larger ecological impacts than previously estimated. Given the estimated level of mercury in Amazonian soils is between 0.02 and 0.4 mg/kg (Santos-Francés et al., 2011), mercury concentrations in active mines at Puruni and Mahdia were three to 240 times higher than reported levels respectively. Furthermore, only one overburden plot in Puruni was within the reported range of mercury concentrations seen elsewhere.

Our statistical analyses suggest an important role for mercury in inhibiting recruitment (Table S6). Mercury concentrations were previously found to strongly inhibit plant development and growth, but have differential impacts across taxa (Muddarisna & Krisnayanti, 2015). Comparison of plant composition data for overburden plots at both sites revealed that an important difference between the compositions of the overburden vegetation is the near absence of leguminous woody species with the capacity to fix nitrogen in Mahdia (Figure 8). While abundance of Fabaceae is similar in control plots in both sites, species capable of fixing nitrogen were significantly more abundant in Puruni. Nitrogen fixers are known to drive biomass accumulation in recovering secondary forests (Batterman et al., 2013) and their suppression in Mahdia may be associated with higher mercury loads inhibiting the activity of nitrogen-fixing bacteria (Nuraini, Arfarita, & Siswanto, 2015; Rafique, Amin, & Latif, 2015). Some studies on agricultural crops have shown that nitrogen-fixing bacteria from the genera *Staphylococcus*, *Pseudomonas*, *Escherichia* and *Bacillus* may be resistant to some levels of mercury contamination by converting mercury into a less toxic form, such as hydrogen sulphide (Nuraini et al., 2015; Rafique et al., 2015). We hypothesize that similar mechanisms may be limiting the establishment of nitrogen-fixing trees in Mahdia, yet we are aware of no study that has examined this. Future studies should directly test the hypothesis that germination of tropical nitrogen-fixing trees and association with nitrogen-fixing bacteria is reduced by soil mercury.

4.3 | Implications for policy and rehabilitation

Relative to other land uses, biomass recovery in previously mined pits and tailing ponds are among the lowest recorded in the literature (Figures 6 and 7). The carbon cost of mining can be computed in different ways, including (a) the standing forest carbon that is lost
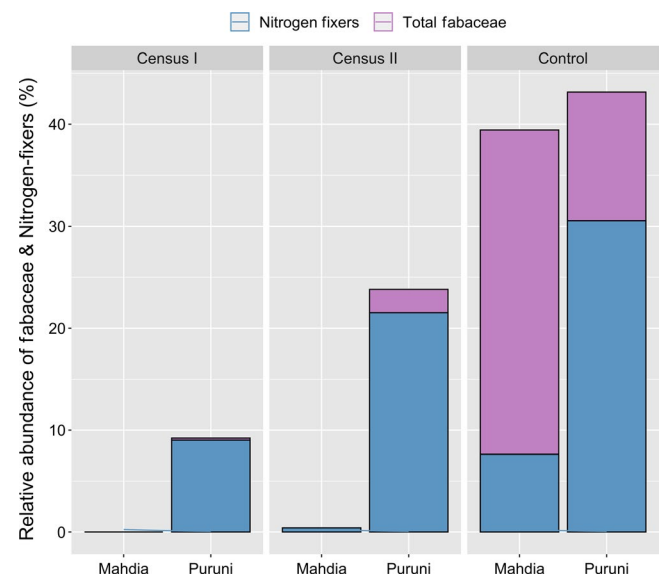
FIGURE 8 Relative abundance of Fabaceae and nitrogen-fixers for Mahdia and Puruni in Census I, Census II and Control

following mine clearing and (b) the lost carbon sequestration potential from mining in relation to carbon that would have been sequestered due to forest recovery from other land uses. For instance, an area of 570 km^2 of forest was deforested in Guyana due to gold mining between 2010 and 2017 (Guyana Forestry Commission, 2018), corresponding to an annual loss of ~71 km^2/year. Assuming that the mean AGB carbon of our two control sites is representative of Guyanese forests more broadly, mining-related forest clearing results in the loss of approximately 1.8 million tonnes of primary forest carbon annually. Furthermore, assuming that the pit/pond/overburden proportions observed in our plots apply to the mining sites across Guyana, we estimate that the arrest in recovery following gold mining resulted in ~11,500 t C/year less carbon being accumulated compared to that would have accumulated under recovery from agriculture/pasture. Although the magnitude of mining-induced deforestation across the Amazon is lower than deforestation caused by other drivers such as pasture and agriculture, it is important to note that currently ~370,000 km^2 of forests are found within active mining concessions across Amazonia (based on data from Alvarez-Berríos & Mitchell Aide, 2015; RAISG, 2012), with 1,680 km^2 lost to gold mining between 2001 and 2013 (Alvarez-Berríos & Mitchell Aide, 2015). This corresponds to an annual loss of ~130 km^2/year, leading to 21,216 t C/year less carbon accumulated compared to accumulation under agriculture/pasture over this period. With a further 1,300,000 km^2 of Amazonia currently under prospecting for mining (based on data from Alvarez-Berríos & Mitchell Aide, 2015; RAISG, 2012), the gold mining-induced carbon deficit in Amazonia is expected to increase greatly over the coming years.

Our results suggest that natural regeneration alone is not effective to rehabilitate abandoned mining sites. As nutrient (especially nitrogen) depletion may strongly influence the ability to recover biomass and that nitrogen concentrations on overburden soils can be up to 20 times higher than those in previously mined pits and tailing ponds, re-filling mining pits and tailing ponds with soils dumped on the overburden may accelerate forest recovery on those zones. The overburden should therefore be viewed as a strategic resource and its removal, storage and replacement to retain the physical and chemical properties should be protected during mining and prior to restoration. The stockpiling of the topsoil is mandatory according to Guyana's Mining (Amendment) Regulations 2005, with a 'close-as-you-go' process of reclamation. However, enforcement of this section of the amendment is rarely executed. Further, knowing that mercury contamination impedes tree recruitment, and may inhibit the establishment of nitrogen fixers, additional remediation treatments may be required to promote forest recovery where background mercury concentrations are high (e.g. in Mahdia).

Certain species can also play an important role in phytoremediation within post-mining sites (Ritchie & Raina, 2016), either through bioaccumulation (i.e. survival despite concentrated contaminants) or through bioexclusion (i.e. by restricting contaminants uptake into their biomass; Tangahu et al., 2011). Although we did not formally test for the tolerance of specific species to mercury, we note that

several tree taxa were found on the overburden plot in Mahdia, despite high soil mercury concentrations. These species, which include *Vismia guianensis* and *Cecropia obtusa*, may therefore represent candidate species for more focused phytoremediation testing. The optimal selection of tree taxa for restoration requires carefully planned field trials as the germination/establishment sensitivity of different taxa to high mercury concentrations remains largely unstudied. Given their central role in driving biomass recovery, phytoremediation strategies which promote the functional integrity of nitrogen fixation processes need to be further studied. Ultimately, however, large-scale remediation efforts will hinge on a balance of costs and effectiveness. Active restoration with additional fertilization can cost >US$3,000 ha^{-1} (Roman-Danobeytia et al., 2015), imposing very large financial challenges on Amazonian countries. Thus, the relative efficacy of different potential remediation treatments ranging from cheaper 'backfilling' options to more advanced targeted phytoremediation options needs to be carefully evaluated.

Our results have important implications for policy. Multiple international efforts have been launched with the goal of achieving large-scale restoration targets such as the Aichi Targets, Bonn Challenge and New York Declaration on Forests, restoring up to 3.5 million km^2 of degraded and deforested land to productive, functional and bio-diversity-friendly landscapes by 2030 (Chazdon et al., 2017). On 1 March 2019, the United Nations Environment Assembly declared 2021–2030 the Decade of Ecosystem Restoration, with the aim to massively scale up restoration of degraded and destroyed ecosystems as a measure to combat climate change, and enhance food security, water supply and biodiversity (United Nations Environment Programme, 2019). Guyana and other Amazonian countries will be challenged with remediating previously mined land as a means to achieve their global commitments. Guyana's Intended Nationally Determined Contributions (INDCs) explicitly highlight the implementation of policies to reclaim and re-forest previously mined land as an important element of its program for reducing carbon emissions. The slow recovery rates observed in our study suggest that active remediation approaches will be necessary for the success of such policies, along with the systematic monitoring of gold mining activities and enforcement to provide quantitatively robust evidence of change over time.

ORCID

Michelle Kalamandeen https://orcid.org/0000-0001-5385-7444
Sarah A. Batterman https://orcid.org/0000-0002-7703-9873
Guy Ziv https://orcid.org/0000-0002-6776-0763
Kaslyn Holder-Collins https://orcid.org/0000-0002-5855-4327
Oliver L. Phillips https://orcid.org/0000-0002-8993-6168
Eduardo S. Brondizio https://orcid.org/0000-0001-9376-8366

REFERENCES

Alvarez-Berríos, N. L., & Mitchell Aide, T. (2015). Global demand for gold is another threat for tropical forests. *Environmental Research Letters*, *10*(1), 014006. https://doi.org/10.1088/1748-9326/10/1/014006

Anderson-Teixeira, K. J., Miller, A. D., Mohan, J. E., Hudiburg, T. W., Duval, B. D., & DeLucia, E. H. (2013). Altered dynamics of forest recovery under a changing climate. *Global Change Biology*, *19*(7), 2001–2021. https://doi.org/10.1111/gcb.12194

Asner, G. P., Llactayo, W., Tupayachi, R., & Luna, E. R. (2013). Elevated rates of gold mining in the Amazon revealed through high-resolution monitoring. *Proceedings of the National Academy of Sciences of the United States of America*, *110*(46), 18454–18459. https://doi.org/10.1073/pnas.1318271110

Baker, T. R., Phillips, O. L., Malhi, Y., Almeida, S., Arroyo, L., Di Fiore, A., … Vásquez Martínez, R. (2004). Increasing biomass in Amazonian forest plots. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, *359*(1443), 353–365. https://doi.org/10.1098/rstb.2003.1422

Batterman, S. A., Hedin, L. O., van Breugel, M., Ransijn, J., Craven, D. J., & Hall, J. S. (2013). Key role of symbiotic dinitrogen fixation in tropical forest secondary succession. *Nature*, *502*(7470), 224–227. https://doi.org/10.1038/nature12525

Beauford, W., Barber, J., & Barringer, A. R. (1977). Uptake and distribution of mercury within higher plants. *Physiologia Plantarum*, *39*, 261–265. https://doi.org/10.1111/j.1399-3054.1977.tb01880.x

Castilhos, Z., Rodrigues-Filho, S., Cesar, R., Rodrigues, A. P., Villas-Bôas, R., de Jesus, I., … Santos, E. (2015). Human exposure and risk assessment associated with mercury contamination in artisanal gold mining areas in the Brazilian Amazon. *Environmental Science and Pollution Research*, *22*(15), 11255–11264. https://doi.org/10.1007/s11356-015-4340-y

Chave, J., Réjou-Méchain, M., Búrquez, A., Chidumayo, E., Colgan, M. S., Delitti, W. B. C., … Vieilledent, G. (2014). Improved allometric models to estimate the aboveground biomass of tropical trees. *Global Change Biology*, *20*(10), 3177–3190. https://doi.org/10.1111/gcb.12629

Chazdon, R. L., Brancalion, P. H. S., Lamb, D., Laestadius, L., Calmon, M., & Kumar, C. (2017). A policy-driven knowledge agenda for global forest and landscape restoration. *Conservation Letters*, *10*(1), 125–132. https://doi.org/10.1111/conl.12220

Chazdon, R. L., Broadbent, E. N., Rozendaal, D. M. A., Bongers, F., Zambrano, A. M. A., Aide, T. M., … Poorter, L. (2016). Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. *Science Advances*, *2*(5), e1501639. https://doi.org/10.1126/sciadv.1501639

Davidson, E. A., Ishida, F. Y., & Nepstad, D. C. (2004). Effects of an experimental drought on soil emissions of carbon dioxide, methane, nitrous oxide, and nitric oxide in a moist tropical forest. *Global Change Biology*, *10*(5), 718–730. https://doi.org/10.1111/j.1529-8817.2003.00762.x

Davidson, E. A., Reis de Carvalho, C. J., Vieira, I. C. G., Figueiredo, R. D. O., Moutinho, P., Yoko Ishida, F., … Tuma Sabá, R. (2004). Nitrogen and phosphorus limitation of biomass growth in a tropical secondary forest. *Ecological Applications*, *14*(4), 150–163. https://doi.org/10.1890/01-6006

Dedieu, N., Allard, L., Vigouroux, R., Brosse, S., & Céréghino, R. (2014). Physical habitat and water chemistry changes induced by logging and gold mining in French Guiana streams. *Knowledge and Management of Aquatic Ecosystems*, *415*, 2. https://doi.org/10.1051/kmae/2014026

Diringer, S. E., Feingold, B. J., Ortiz, E. J., Gallis, J. A., Araújo-Flores, J. M., Berky, A., … Hsu-Kim, H. (2015). River transport of mercury from artisanal and small-scale gold mining and risks for dietary mercury exposure in Madre de Dios, Peru. *Environmental Science: Processes & Impacts*, *17*(2), 478–487. https://doi.org/10.1039/C4EM00567H

Espejo, J. C., Messinger, M., Román-Dañobeytia, F., Ascorra, C., & Luis, E. (2018). Deforestation and forest degradation due to gold mining in the Peruvian Amazon: A 34-year perspective. *Preprint, November*, 1–17. https://doi.org/10.20944/preprints201811.0113.v1

Feldpausch, T. R., Rondon, M. A., Fernandes, E. C. M., Riha, S. J., & Wandelli, E. (2004). Carbon and nutrient accumulation in secondary forests regenerating on pastures in central Amazonia. *Ecological Applications*, *14*(4), 164–176. https://doi.org/10.1890/01-6015

Gaui, T. D., Costa, F. R. C., Coelho de Souza, F., Amaral, M. R. M., de Carvalho, D. C., Reis, F. Q., & Higuchi, N. (2019). Long-term effect of selective logging on floristic composition: A 25 year experiment in the Brazilian Amazon. *Forest Ecology and Management*, *440*(December 2018), 258–266. https://doi.org/10.1016/j.foreco.2019.02.033

Gutiérrez-Mosquera, H., Sujitha, S. B., Jonathan, M. P., Sarkar, S. K., Medina-Mosquera, F., Ayala-Mosquera, H., … Arreola-Mendoza, L. (2018). Mercury levels in human population from a mining district in Western Colombia. *Journal of Environmental Sciences (China)*, *68*, 83–90. https://doi.org/10.1016/j.jes.2017.12.007

Guyana Forestry Commission. (2018). *Guyana Forestry Commission Guyana REDD + Monitoring Reporting and Verification System (MRVS) Accuracy*. Assessment Final Report (November).

Guyana Forestry Commission, & Indufor. (2012). *Guyana Forestry Commission Guyana REDD + Monitoring Reporting & Verification System (MRVS) Interim Measures Report 01 October 2010–31 December 2011*. (December 2011).

Howard, J., Trotz, M. A., Thomas, K., Omisca, E., Chiu, H. T., Halfhide, T., … Stuart, A. L. (2011). Total mercury loadings in sediment from gold mining and conservation areas in Guyana. *Environmental Monitoring and Assessment*, *179*(1–4), 555–573. https://doi.org/10.1007/s10661-010-1762-3

Kalamandeen, M., Gloor, E., Johnson, I., Agard, S., Katow, M., Vanbrooke, V., … Galbraith, D. (2020). Data from: Limited biomass recovery from gold mining in Amazonian forests. *Dryad Digital Repository*, https://doi.org/10.5061/dryad.j6q573n9s

Lohbeck, M., Poorter, L., Martínez-Ramos, M., & Bongers, F. (2015). Biomass is the main driver of changes in ecosystem process rates during tropical forest succession. *Ecology*, *96*(5), 1242–1252. https://doi.org/10.1890/14-0472.1

Martin, P. A., Newton, A. C., & Bullock, J. M. (2013). Carbon pools recover more quickly than plant biodiversity in tropical secondary forests. *Proceedings of the Royal Society B: Biological Sciences*, *280*(1773), 20132236. https://doi.org/10.1098/rspb.2013.2236

Muddarisna, N., & Krisnayanti, B. D. (2015). Selection of mercury accumulator plants for gold mine tailing contaminated soils. *Journal of Degraded and Mining Land Management*, *2*(3), 341–346. https://doi.org/10.15243/jdmlm.2015.023.341

Novoa, S., Finer, M., & Román, F. (2016). Regeneration of vegetation in zone affected by gold mining in the Amarakaeri Communal Reserve. *MAAP*, *44*.

Nuraini, Y., Arfarita, N., & Siswanto, B. (2015). Isolation and characteristic of nitrogen-fixing bacteria and phosphate-solubilizing bacteria from soil high in mercury in tailings and compost areas of artisanal gold mine. *Agrivita*, *37*(1), 1–7. https://doi.org/10.17503/Agrivita-2015-37-1-p001-007

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., … Hayes, D. (2011). A large and persistent carbon sink in the world's forests. *Science*, *333*(6045), 988–993. https://doi.org/10.1126/science.1201609

Peterson, G. D., & Heemskerk, M. (2001). Deforestation and forest regeneration following small-scale gold mining in the Amazon: The case of Suriname. *Environmental Conservation*, *28*(2), 117–126. https://doi.org/10.1017/S0376892901000121

Phillips, O., Baker, T., Feldpausch, T., & Brienen, R. (2015). *Field manual for establishment and remeasurement (RAINFOR)*. Amazon Forest Inventory. Retrieved from http://amazonpire.org/PDF/fc2009/readings/1m1_RAINFOR_treeinventory.pdf

Poorter, L., Bongers, F., Aide, T. M., Almeyda Zambrano, A. M., Balvanera, P., Becknell, J. M., … Rozendaal, D. M. A. (2016). Biomass resilience of neotropical secondary forests. *Nature*, *530*(7589), 211–214. https://doi.org/10.1038/nature16512

Prach, K., & Walker, L. R. (2019). Differences between primary and secondary plant succession among biomes of the world. *Journal of Ecology*, *107*(2), 510–516. https://doi.org/10.1111/1365-2745.13078

Rafique, A., Amin, A., & Latif, Z. (2015). Screening and characterization of mercury-resistant nitrogen fixing bacteria and their use as biofertilizers and for mercury bioremediation. *Pakistan Journal of Zoology*, *47*(5), 1271–1277.

RAISG. (2012). Amazonia Mineria. Rede Amazonica de Informacion Socioambiental Georreferenciada (RAISG). 68 pp. Retrieved form https://www.amazoniasocioambiental.org/es/

Ritchie, M. E., & Raina, R. (2016). Effects of herbivores on nitrogen fixation by grass endophytes, legume symbionts and free-living soil surface bacteria in the Serengeti. *Pedobiologia*, *59*(5–6), 233–241. https://doi.org/10.1016/j.pedobi.2016.09.001

Román-Dañobeytia, F., Huayllani, M., Michi, A., Ibarra, F., Loayza-Muro, R., Vázquez, T., … García, M. (2015). Reforestation with four native tree species after abandoned gold mining in the Peruvian Amazon. *Ecological Engineering*, *85*, 39–46. https://doi.org/10.1016/j.ecoleng.2015.09.075

Rozendaal, D. M. A., Chazdon, R. L., Arreola-Villa, F., Balvanera, P., Bentos, T. V., Dupuy, J. M., … Bongers, F. (2017). Demographic drivers of aboveground biomass dynamics during secondary succession in neotropical dry and wet forests. *Ecosystems*, *20*(2), 340–353. https://doi.org/10.1007/s10021-016-0029-4

Santos-Francés, F., García-Sánchez, A., Alonso-Rojo, P., Contreras, F., & Adams, M. (2011). Distribution and mobility of mercury in soils of a gold mining region, Cuyuni river basin, Venezuela. *Journal of Environmental Management*, *92*(4), 1268–1276. https://doi.org/10.1016/j.jenvman.2010.12.003

Sonter, L. J., Ali, S. H., & Watson, J. E. M. (2018). Mining and biodiversity: Key issues and research needs in conservation science. *Proceedings of the Royal Society B: Biological Sciences*, *285*(1892), 20181926. https://doi.org/10.1098/rspb.2018.1926

Sonter, L. J., Herrera, D., Barrett, D. J., Galford, G. L., Moran, C. J., & Soares-Filho, B. S. (2017). Mining drives extensive deforestation in the Brazilian Amazon. *Nature Communications*, *8*, 1013. https://doi.org/10.1038/s41467-017-00557-w

Tangahu, B. V., Sheikh Abdullah, S. R., Basri, H., Idris, M., Anuar, N., & Mukhlisin, M. (2011). A review on heavy metals (As, Pb, and Hg) uptake by plants through phytoremediation. *International Journal of Chemical Engineering*, *2011*, 1–31. https://doi.org/10.1155/2011/939161

United Nations Environment Programme. (2019). New UN Decade on Ecosystem Restoration offers unparalleled opportunity for job creation, food security and addressing climate change. Press Release on 1 March 2019. Retrieved from https://www.unenvironment.org/news-and-stories/press-release/new-un-decade-ecosystem-restoration-offers-unparalleled-opportunity

Veiga, M. M., Maxson, P. A., & Hylander, L. D. (2006). Origin and consumption of mercury in small-scale gold mining. *Journal of Cleaner Production*, *14*(3–4), 436–447. https://doi.org/10.1016/j.jclepro.2004.08.010

Wantzen, K., & Mol, J. (2013). Soil erosion from agriculture and mining: A threat to tropical stream ecosystems. *Agriculture*, *3*(4), 660–683. https://doi.org/10.3390/agriculture3040660

Wright, S. J., Turner, B. L., Yavitt, J. B., Harms, K. E., Kaspari, M., Tanner, E. V. J., … Garcia, M. N. (2018). Plant responses to fertilization experiments in lowland, species-rich, tropical forests. *Ecology*, *99*(5), 1129–1138. https://doi.org/10.1002/ecy.2193

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.