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ENVIRONMENTAL RESEARCH ECOLOGY

PAPER

Dung beetles as bioindicators of environmental change from artisanal small-scale gold mining

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Supplementary material for this article is available [online](#)

Abstract

The expansion of gold mining across Amazonia is a major driver of deforestation, yet its impacts on biodiversity are understudied. Dung beetles (Scarabaeinae) are excellent bioindicators and have been shown to be highly sensitive to habitat modification, especially within the tropics. Furthermore, they provide key ecosystem services. We quantify the impact of gold mining on dung beetle communities in 16 abandoned gold mines of different sizes and years since abandonment, in northwest Guyana. We surveyed dung beetles using baited pitfall traps at the centre of the mines, at the edges and 100 m into adjacent forest. We show that gold mining significantly reduces dung beetle species richness and abundance, with the most pronounced declines occurring in areas with a higher intensity of mining within a 500 m radius. We measured environmental covariates and found that while mine area had no notable effect on species richness, elevated temperatures and reduced canopy cover in the mine centres had negative consequences for dung beetle communities. Conversely, increasing canopy cover was positively associated with dung beetle richness and abundance. However, overall, we find no significant recovery of dung beetle communities up to two decades following mine abandonment. Our findings suggest that the expanding artisanal and small-scale gold mining in Amazonia could have lasting detrimental effects on dung beetle assemblages and the important ecosystem services they provide.

1. Introduction

As one of the world's most biodiverse ecosystems, the Amazon rainforest faces escalating threats from land-use change and climate change, leading to declines in ecosystem functioning and biodiversity (European Commission *et al* 2022, Lapola *et al* 2023). A growing driver of forest loss and degradation in the region is gold mining, both at industrial and small/artisanal scales (Kalamandeen *et al* 2020, Giljum *et al* 2022). While the localised impacts of gold mining on biodiversity are largely understudied, studies on other drivers of deforestation in Amazonia such as fragmentation and conversion to agriculture, have unequivocally found negative effects on biodiversity (Ochoa-Quintero *et al* 2015, Barlow *et al* 2016).

Mining expansion is driven by demand (Giljum *et al* 2022). The Guiana Shield, which comprises a quarter of the Amazonian rainforest, contains numerous greenstone formations which harbour high levels of gold deposits (Hammond *et al* 2007). The rise in the price of gold has increased the profitability of lower grade deposits, which are sometimes remote, or occur as fine particles that are harder to attain (Hammond *et al* 2007). In the Guiana Shield gold mining accounts for 90% of deforestation, and rates of mining associated forest loss are increasing (Dezécache *et al* 2017, Kalamandeen *et al* 2020).



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As such, in French Guiana, authorities have restricted artisanal small-scale gold mining activities since 2002 via environmental policies that are enforced by the military (Dezécache *et al* 2017). This has led to a spillover into the neighbouring countries of Suriname and Guyana where no such regulation exists (Dezécache *et al* 2017).

Tropical forests exhibit resilience to low-intensity disturbances and can naturally regenerate within 60–120 yr (Poorter *et al* 2021). However, mining is a high-intensity activity, meaning that recovery may be much slower. Indeed, recent studies of tropical forest biomass recovery rates have found that some of the slowest recovery rates are associated with gold mining (Kalamandeen *et al* 2020, Román-Dañobeytia *et al* 2021). This has significant implications for the potential of these ecosystems to store and sequester carbon (Giam 2017), and it reduces the availability of suitable habitats for biodiversity in the surrounding landscape.

Dung beetles (Scarabaeinae) are a globally distributed invertebrate group, with their highest diversity found in tropical forests and savannas (Hanski and Cambefort 1991). They are easily sampled using standardised methods (Larsen and Forsyth 2005, Mora-Aguilar *et al* 2023), making them an ideal taxon to rapidly assess changes in forest ecosystems due to habitat loss, degradation and disturbance (Quintero and Roslin 2005, Nichols *et al* 2007, Bicknell *et al* 2014). Dung beetles play key roles in nutrient cycling, seed dispersal and pollination (Hanski and Cambefort 1991, Torabian *et al* 2024), as well as providing other important ecosystem services (de Groot *et al* 2002). Furthermore, they have been shown to help restore soil nutrients and water infiltration rates after mining cessation in South Africa (Badenhorst *et al* 2018, Dabrowski *et al* 2019). Bioturbation from dung beetle activity increases soil porosity and aeration which, in turn, can improve soil quality for plants (Nichols *et al* 2008). By burying dung, they also help to enhance nitrogen levels, a key nutrient for plant growth, by accelerating mineralisation rates (Nichols *et al* 2008).

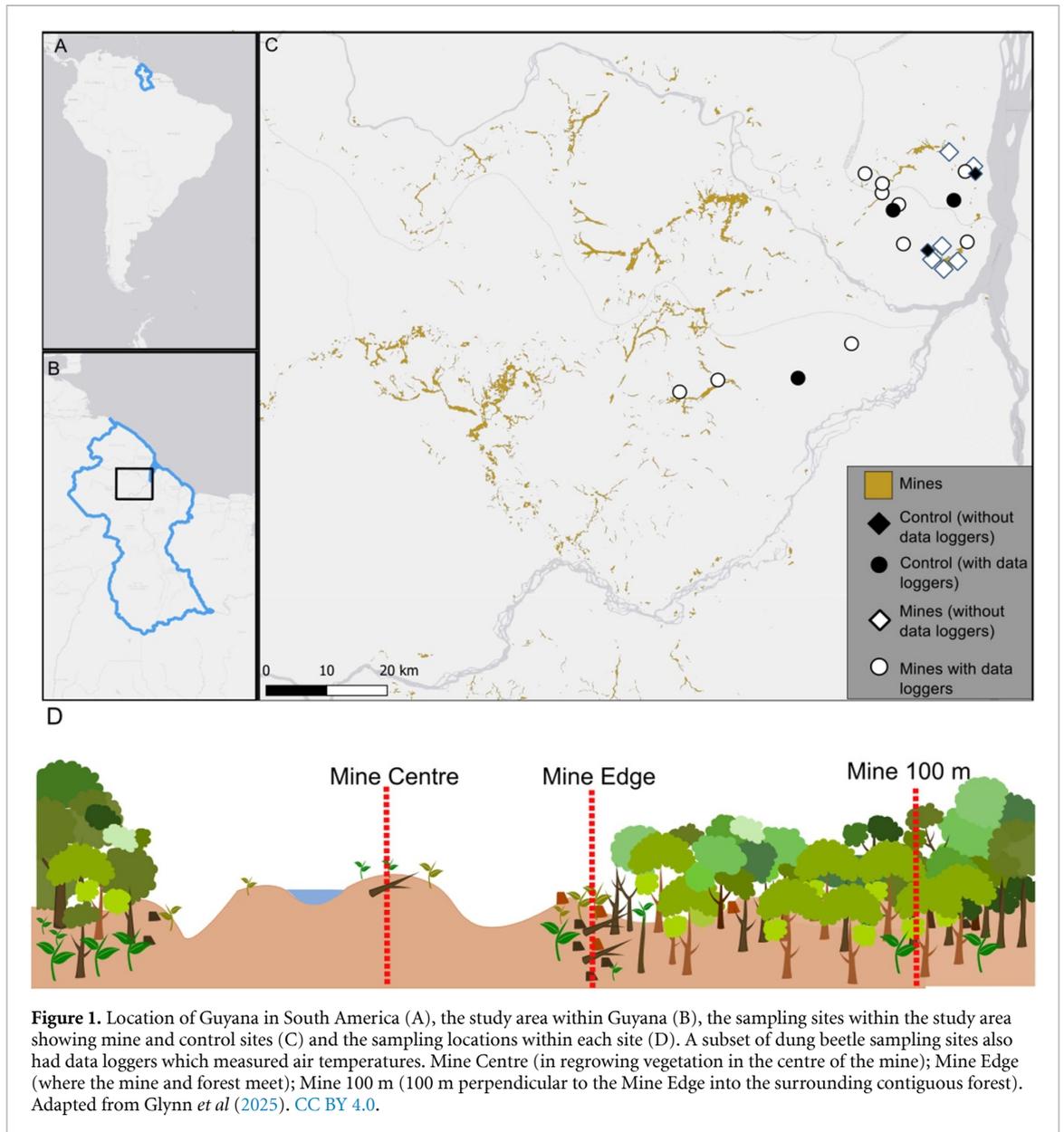
Despite their sensitivity to habitat disturbance, dung beetle communities have been found to recover over time. Quintero and Roslin (2005) demonstrated that within fifteen years, dung and carrion beetle assemblages in fragmented central Amazonian sites had largely recovered, closely resembling those in continuous forests due to the regrowth of secondary vegetation between fragments. However, the effects of artisanal small-scale gold mining on dung beetle communities remain poorly studied in Amazonia and cannot be directly inferred from forest fragmentation experiments. Although fragmentation studies such as the BDFFP are embedded within largely contiguous forest, they explicitly examine isolated forest patches, a spatial configuration that is fundamentally different from ASGM landscapes. In contrast to fragmentation, which isolates forest patches within an open matrix, ASGM creates discrete open clearings embedded within otherwise continuous forest, generating enclosed forest edges and extreme localized disturbance. These differences in spatial configuration and disturbance processes are therefore likely to produce distinct ecological responses. Under fragmentation, we know that edge effects impact many taxa negatively, including dung beetles, and this is often associated with microclimatic changes (Laurance *et al* 2002, Pfeifer *et al* 2017). Therefore, the influence of artisanal small-scale gold mines on dung beetle communities may be substantial. Mines often lack canopy cover, leading to elevated temperatures that penetrate nearby forest edges (Chaddad *et al* 2022), reducing habitat suitability for dung beetles that are unable to thermoregulate effectively at elevated temperatures (Verdú *et al* 2006, Giménez Gómez *et al* 2020). Combined with the slow rate of vegetation regeneration, these altered conditions have the potential to result in long-term negative impacts on dung beetle populations.

We studied dung beetle communities and potential covariates, in 16 abandoned gold mines and five nearby control forests in northwest Guyana, to assess their response to natural regeneration and edge effects, while also evaluating their efficacy as indicators of ecosystem recovery following the cessation of artisanal small-scale gold mining. We posed the following three research questions: (i) how do dung beetle communities in mined sites differ from those in nearby forests?; (ii) is there evidence of recovery in dung beetle communities in mines that have had more time to recover after abandonment?; and (iii) how do vegetation structure and microclimatic changes influence dung beetle communities?

2. Methods

2.1. Sampling sites

Sampling was undertaken in 16 abandoned artisanal small-scale gold mines and five nearby control forests, in northwest Guyana (figure 1), during dry seasons between February 2022 and May 2023. The region is a mixture of lowland forest on sand and laterite soils (Espinosa *et al* 2022). The average rainfall for Guyana ranges from 1778 mm and 2800 mm annually, with daily temperatures averaging 25.9 °C. Five control sites, located in contiguous forest, were a minimum of 1 km from any mining activity.



Mining sites were selected to cover a range of sizes (area) and time since abandonment, and therefore potential recovery. Information on the year of abandonment was obtained from local miners/workers who had existing knowledge of the area and its mining history supported by historical satellite imagery—examples of the mining landscapes within this study at different stages of recovery can be seen in figure S1. All sampled sites were separated by at least 1 km to maintain spatial independence and were located adjacent to contiguous forest. Sites were selected randomly from the subset of mining areas that were accessible and met these criteria. The five control sites were chosen to represent relatively intact forest ecosystems adjacent to the mining areas, providing a baseline for comparison of localised impacts. These control sites were located within the same contiguous forest that the mines were in and therefore represented forests that were directly comparable, and could potentially be converted into mines.

At each mined site, we sampled at three sampling locations: Mine Centre (in regrowing vegetation in the centre of the mine), at the Mine Edge (where the mine and forest meet) and Mine 100 m (100 m perpendicular to the mine into the surrounding contiguous forest (figure 1(D)).

2.2. Dung beetle sampling

Standard pitfall traps, 11 cm in diameter; 15 cm deep ($\sim 1500 \text{ cm}^3$), were used to sample dung beetles (Larsen and Forsyth 2005, Mora-Aguilar *et al* 2023). Traps were buried flush with the ground, partially filled with water mixed with unscented detergent, and baited with approximately 20 g of fresh human dung—the optimal bait for dung beetles (Larsen and Forsyth 2005, Mora-Aguilar *et al* 2023). Each

trap had a 30 cm diameter plastic plate above it to protect it from rainfall and debris, and from which the bait was suspended. At mine sites, four traps were placed at each of the sampling locations (Mine Centre, Mine Edge and Mine 100 m), resulting in 12 traps per mine site. Each trap was separated by at least 50 m. At control forest sites, all 12 traps were placed in a linear transect, spaced 50 m apart (Larsen and Forsyth 2005). Traps were deployed for 96 h, and bait was replaced after 48 h (Larsen and Forsyth 2005, Bicknell *et al* 2014). All beetles were identified (by J. Herce) to species level where possible, otherwise to genus morphospecies (e.g. *Aphodiinae* sp. 1) or morphotype (e.g. sp. A).

2.3. Air temperature variables

In a subset of 10 mines and three control sites, we recorded air temperatures (°C) between February and May 2023. At each site, we deployed nine data loggers (EasyLog USB, Lascar Electronics) with three per sampling location (Mine Centre, Mine Edge and Mine 100 m) close to the pitfall traps. The data loggers were suspended 1.5 m above ground covered by a polystyrene plate rain-cover (following Williamson *et al* 2020 except for at 1.5 m in height) for a period of 96 h, recording every 30 s. Due to equipment failure, only one site retained a complete dataset from all three data loggers at the Mine 100 m sampling location. Consequently, from the second week onward, each site had eight functioning data loggers: all three were operational at the Mine Edge and Mine Centre, but only two at the Mine 100 m location. At control sites, all nine data loggers were installed in a linear transect close to pitfall traps.

Given that dung beetles can be both endothermic and ectothermic (Verdú *et al* 2006, Giménez Gómez *et al* 2020), we quantify the effects of average and maximum air temperatures on dung beetle communities at both the site and sampling location levels. We calculated the mean temperature at the site level, averaging across all sampling locations, as well as the mean temperature at each individual sampling location. Additionally, we determined the maximum temperature recorded at each sampling location (but not at the site level because maximum temperatures were always recorded in the Mine Centre).

2.4. Vegetation structure variables

To understand whether there were differences in vegetation structure in and around mines, and whether these impacted dung beetle communities, we undertook tree plot surveys. We measured percentage canopy cover and number of trees in one 10 × 10 m quadrat at each sampling location. Canopy cover was measured by taking an iPhone 10 image, pointing directly up at a height of 1.5 m above the ground. We processed the image in ImageJ (Version 1.49b) to calculate percentage canopy cover following the method described by Xiong *et al* (2019), converting pixels to binary white or black and equating canopy cover to the percentage that were black. This method provides a replicable, low-cost estimate of canopy cover suitable for comparative ecological studies in remote field conditions (Xiong *et al* 2019). To better understand changes in tree community composition following mine abandonment, we classified each tree in the quadrats as either ≥30 cm diameter at breast height (DBH), representing mature individuals, or <30 cm DBH, representing younger or regenerating trees.

To consider the potential impact of mining intensity in the broader landscape, we also calculated the percentage area covered by mining activity within three radii distances (100 m, 500 m and 1 km) from the middle of the Mine Centre sampling locations, using Sentinel-2 satellite images from the year that each site was sampled (2022/2023). We did this by drawing polygons around each of the mines in QGIS, as mines stood out clearly from the surrounding forest, we were able to distinguish mines from the surrounding forest areas, and then used the package 'sf', with functions 'st_buffer' and 'st_intersection' to calculate percentage cover of mining in the buffer areas. This method was also used to calculate the area of the study mines to the nearest m² in QGIS.

2.5. Data analysis

All further data analyses were undertaken in R (R Core Team 2022). We used non-metric multi-dimensional scaling (NMDS), based on Bray–Curtis dissimilarity coefficients, to visualise dung beetle community composition at site (all sampling locations at each mine combined) and sampling location (Mine Centre, Mine Edge and Mine 100 m) levels. NMDS was performed using the 'metaMDS' function in the 'Vegan' package, with a maximum of 250 iterations to find a stable solution. ADONIS tests (Oksanen *et al* 2022) were used to test for differences between mined and control sites. Pairwise comparisons between the sampling locations within the mined sites and controls were performed using the 'pairwiseAdonis' package (Martinez Arbizu 2020).

Dung beetle species richness estimates for each site and sampling location were calculated using the 'iNEXT' (Interpolation and Extrapolation for Species Diversity) package in R (Hsieh *et al* 2016). We generated species accumulation curves (figure S2) at the site (Mine, Control) and sampling locations level

(Mine Edge, Mine Centre, Mine 100 m). We estimated species richness at two spatial scales: overall for each site, and separately for each sampling location within sites, using coverage-based estimates, this method estimates species richness by first determining the proportion of the community that has been sampled (sample coverage), before adjusting for unobserved species to predict the total species richness (Chao and Jost 2012). The 'estimateD()' function was used to quantify species diversity at a fixed coverage level of 95%. By setting this threshold, the approach interpolates species richness for well sampled sites and extrapolates for under-sampled sites, thereby generating a more accurate estimate of true species richness, accounting for uneven sampling efforts (Chao and Jost 2012). Independent t -tests were used to test for differences in estimated species richness and abundance at the site level, and paired t -tests were used at the sampling location level.

Potential relationships between estimated species richness with site variables (mine area, years since abandonment, percentage area of mining within each of the three buffers and mean air temperature) were investigated using generalised linear models with a Gaussian error distribution. As both the 100 m and 500 m percentage cover by mining activity radii were nested within the 1 km one, we ran generalised linear models individually to identify the most appropriate buffer for use within models with the lowest AIC value. At the sampling location level (including control data), we measured the microclimatic and vegetation structure variables at each of the sampling locations. All combinations of potential explanatory variables were assessed for collinearity using variance inflation factors (VIFs); any variables with a VIF score >5 were removed (at all site level we removed the mining buffers of 100 m and 1 km). AICc was used to assess explanatory power penalising for complexity in our models via the dredge function from the 'MuMin' package. A threshold of AICc < 4 was used for model plausibility, with the final model chosen being that with the lowest AICc value. As only a subset of Mine and control sites had a temperature dataset, models were run twice: first using all the sites without any temperature variables and, second, using only the 10 Mine and three control sites with temperature data.

3. Results

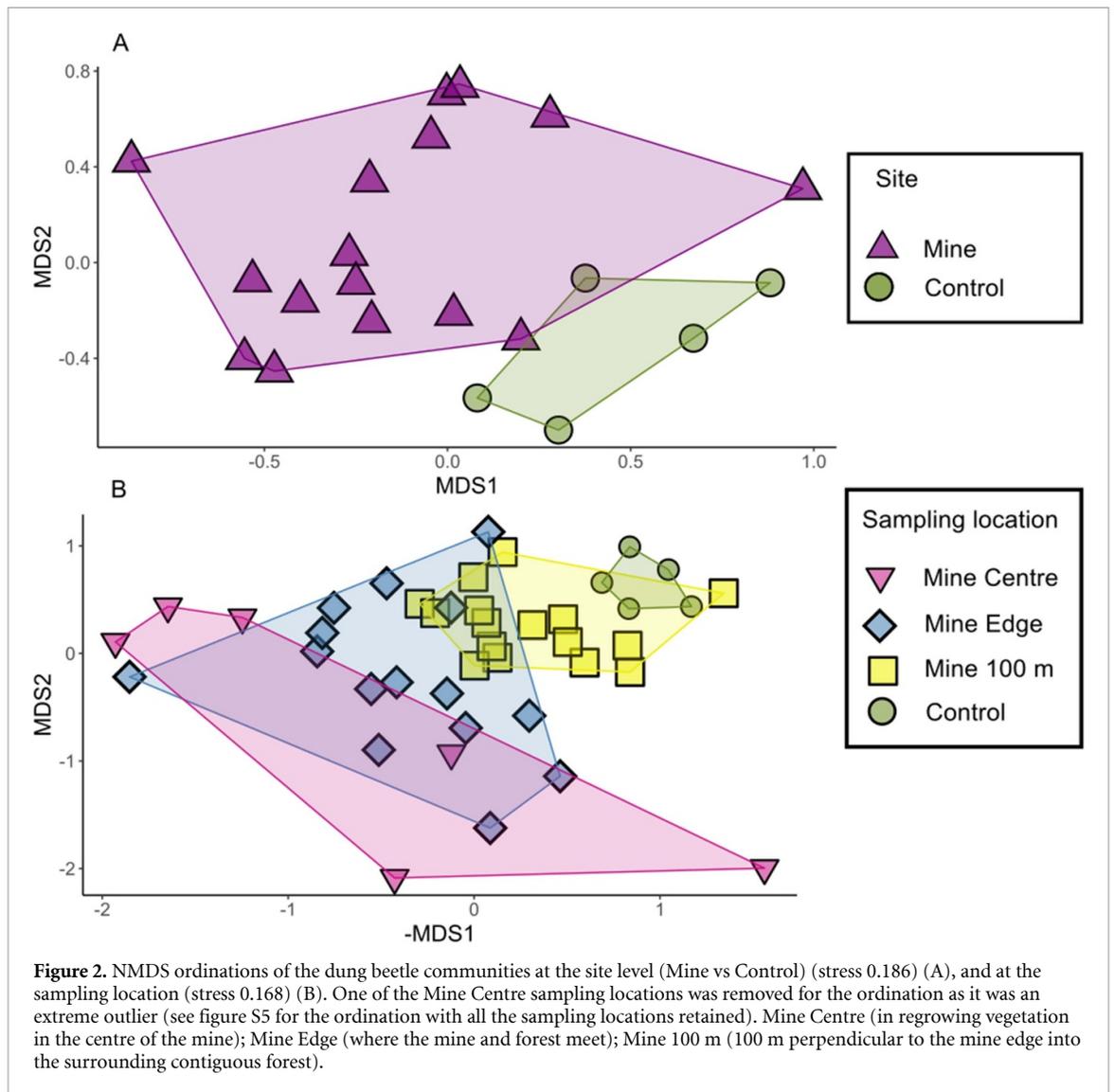
A total of 8187 dung beetles were captured from 44 species/morphospecies (table S1). *Oxysternon festivum* was the most abundant species with 1330 individuals, followed by *Onthophagus* sp. 1 with 1224 individuals (table S1). The Mine Centre had the fewest captures with only 86 beetles recorded, whereas the Control sites had 4016 individuals recorded (just under half of the total number of beetles captured) NMDS ordinations showed distinct communities amongst the sampling locations (figure 2; ADONIS $p = 0.003$ at the site level and $p = 0.001$ at the sampling location level). Pairwise ADONIS for each of the sampling locations revealed significant differences ($p < 0.05$) for all combinations, apart from Mine Centre vs Mine Edge ($p = 0.336$).

Number of mature trees and percentage of canopy cover increased from Mine Centre to the controls (figure S3). There were more young trees at the Mine Edge compared to other sampling locations, with the lowest numbers recorded in the Mine Centre (figure S3). Average and maximum air temperatures declined from the Mine Centre towards Mine 100 m (figure S4).

Species accumulation curves showed that estimated species richness was similar for Mine and control sites ($t = 0.79$, $p = 0.40$) (figure S2), despite there being a significant difference in abundance ($t = 6.02$, $p < 0.001$). Estimated species richness and abundance increased at sampling location level from Mine Centre to Mine 100 m (figure 3), with significant differences found between the Mine Centre and Mine 100 m and the Mine Edge ($p < 0.001$ and $p = 0.05$ respectively; table S2). Control sites were statistically different from both the Mine Centre and Mine Edge ($p < 0.001$, $p = 0.01$ respectively). No significant differences were found between the Mine Edge and Mine 100 m ($t = 1.57$, $p = 0.13$).

At the site level, the percentage of cover by mining activity within all three of the buffers negatively influenced estimated species richness (figure S6), but the 500 m radius was the best predictor ($\beta = -0.24$, CI = -0.40 – -0.08 ; table S3), explaining 38% of the variance. When years since abandonment and mine area were included in the final models, the percentage of mining within a 500 m buffer was still the best predictor for estimated species richness ($\beta = -8.42$, CI = -12.95 – -3.89), with the best model also including mine area but not years since abandonment. At the sampling location level, the only variable that best predicted estimated species richness was percentage canopy cover ($\beta = 7.3$, CI = 5.46 – 9.14 ; figure 4).

We observed similar results when temperatures were included in the models. Temperatures and percentage canopy cover were collinear (VIF < 5), but the former were retained in the models due to the lower VIF value. At the site level, mean air temperature was a predictor of estimated species richness ($\beta = 6.75$, CI = 2.04 – 11.46 ; table S3). At the sampling location level, the best model included

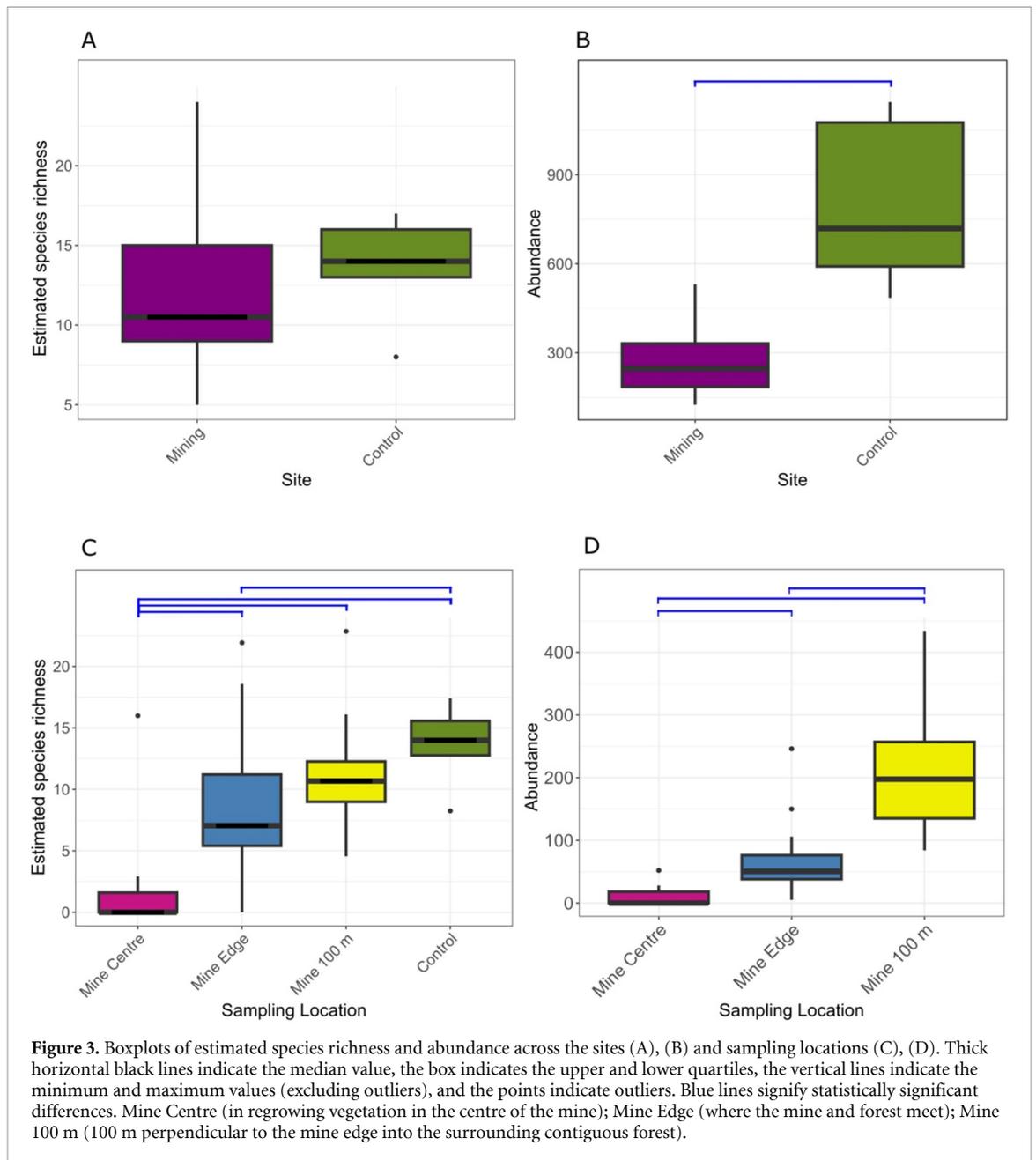


maximum air temperature (figure 5) and number of young trees as predictors of estimated species richness ($\beta = -2.42$ CI = -3.81 – -1.03 and $\beta = 1.66$, CI = 0.26 – 3.05 respectively).

4. Discussion

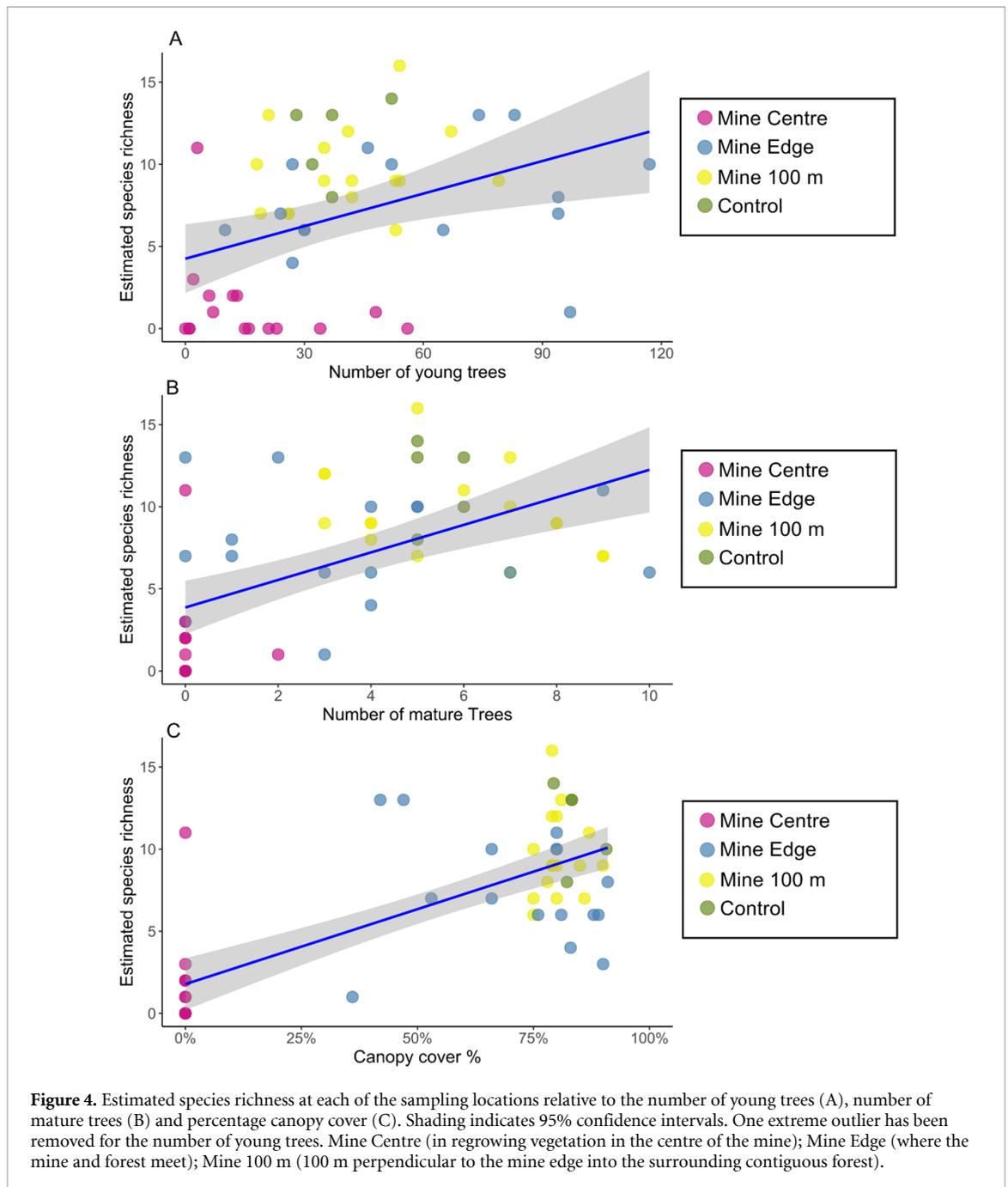
Assessing the resilience of tropical biodiversity to high-intensity land use is critical, yet the long-term ecological legacy of mining remains poorly understood. As such, we investigated the recovery of dung beetle assemblages and their response to edge effects and microclimatic changes in regenerating gold mine sites. Our findings demonstrate that artisanal small-scale gold mining in Guyana reduces dung beetle richness and abundance in both mining sites and surrounding edge forest. This form of mining is projected to reduce Guyana's total forest cover by 9% by 2043 (Hayes *et al* 2023) thereby intensifying pressure on dung beetle populations.

At a landscape level, there was a decrease in dung beetle diversity in areas with greater mining activity, while sites surrounded by less mining activity showed higher species richness. This suggests that the effects of artisanal small-scale gold mining penetrate further than the 100 m that we investigated. This supports the findings of Marsh *et al* (2018) that reported that edge effects could be detected in dung beetle communities up to 300 m from an edge. This contrasts with studies on vertebrate taxa, such as birds, where edge effects are most pronounced within the first 100 m (Harper *et al* 2005, Patten and Smith-Patten 2012). This study focused on mines surrounded by contiguous forest, representing scenarios where relatively intact dung beetle communities might reasonably be expected in the adjacent undisturbed habitat. In contrast, in other landscapes where the adjacent forest is already fragmented,



the impacts on dung beetle populations may be more severe. This is likely because these communities are already depleted by edge effects and isolation (Feer and Hingrat 2005).

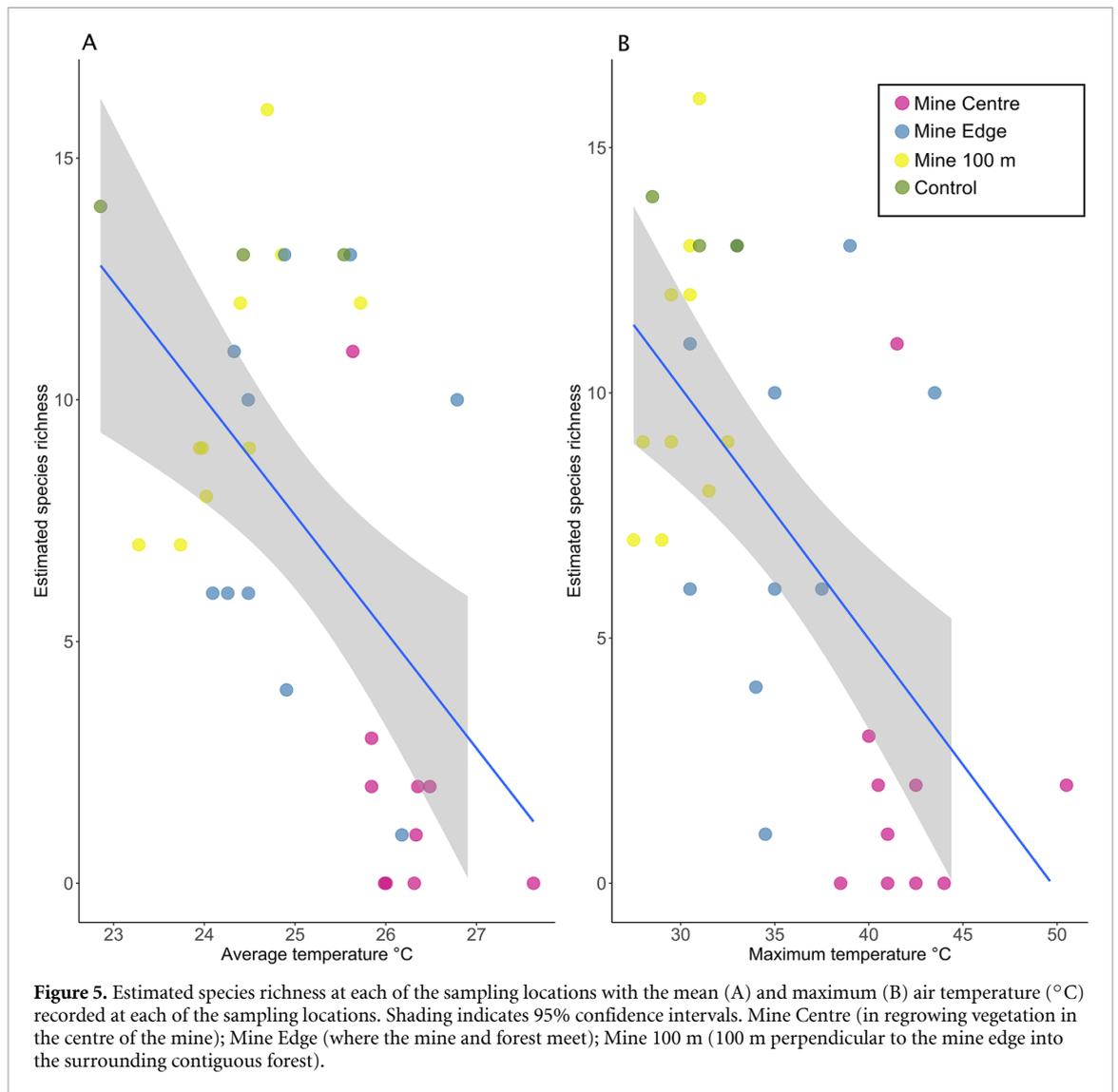
We might have expected to observe a recovery in dung beetle assemblages with time, as seen in earlier studies of forest fragmentation (e.g. Quintero and Roslin 2005). However, recent research highlights that recovery from high-intensity disturbances is often prolonged and incomplete. For example, Noriega *et al* (2021) found that Amazonian dung beetle assemblages required nearly 80 yr to return to pre-disturbance states following logging. Furthermore, specifically within Guyana, Stoll *et al* (2022) demonstrated that insect family richness remains significantly reduced in areas with high gold mining cover. Consistent with these recent findings, we found no significant relationship between the years since mine abandonment and either species richness or abundance. The lack of a significant relationship between years since abandonment and dung beetle recovery should be interpreted with caution, as the precise dates of mine cessation were based on local testimonies and may not be exact, and because variables other than time since abandonment may influence potential recovery. Nevertheless, our findings show no discernible recovery trend over two decades, aligning with other studies that have reported slow rates of vegetation regrowth in mined areas (Kalamandeen *et al* 2020), with little discernible increases in woody biomass regrowth even two decades after mining activity ceased (Chambi-Legoas *et al* 2021). Indeed, recent work in both the southeastern Peruvian Amazon (Garate-Quispe *et al* 2023) and



the Chocó region of Colombia (Abadía *et al* 2025) confirms that the recovery of aboveground biomass in post-mining forests is a slow, variable process, with little discernible regrowth even two decades after mining activity ceased (Chambi-Legoas *et al* 2021). This slow timeline is further supported by studies on different types of disturbance in the Amazon; Noriega *et al* (2021) found that in forests recovering from logging a full recovery of species richness, abundance, and functional group diversity may take more than 30 yr.

Our findings also indicate that dung beetle richness correlates with greater canopy cover, which is typically absent in the centre of mines. This aligns with Stoll *et al* (2022), who reported that structural degradation from mining in Guyana acts as a primary filter for insect biodiversity

Fewer dung beetles were captured in the centre of mines, comprising only 1% of the total number of individuals captured. This is likely to be because of the high temperatures and lack of canopy cover recorded within the centre of the mines, as temperature and vegetation structure are known to be primary drivers of dung beetle assemblage composition (Daniel *et al* 2022). However, the extreme maximum temperatures recorded in the Mine Centre (reaching up to 50.1 °C) should be interpreted accordingly, as they likely include the compounding effect of solar reflection. This is because the exposed



substrate in the Mine Centre likely increases radiative temperatures due to reflection from the surface onto the sensor. Indeed Teichmann *et al* (2024) found that high-albedo environments (such as light-coloured sand like that in mines) increase this effect. Studies such as Maclean *et al* (2021) and Teichmann *et al* (2024) additionally demonstrate that sensors exposed to direct solar radiation are susceptible to significant heating, where solar flux elevates the temperature around the sensor compared to the ambient air. Whilst the high temperatures seen in the Mine Centre occurred in the early afternoon when our data loggers would have been shaded by the plate rain-cover, Maclean *et al* (2021) specifically note that without proper shielding around the entire device, temperature differences between the air and a thermometer may reach 25°C due to radiative fluxes. Consequently, the peak values recorded in our dataset likely represent a combination of ambient heat and radiative loading.

The two beetle genera identified in this study with the largest body size, *Coprophanaeus* and *Deltochilum*, were primarily captured in traps 100 m from the edge of mines and at control sites (table S1), both of which were characterised by lower temperatures. This pattern suggests that microclimatic changes, especially increased temperatures and exposure, may exceed the physiological tolerance limits of larger-bodied Scarabaeinae species (Verdú *et al* 2006, Giménez Gómez *et al* 2020). Furthermore, many forest specialist dung beetle species are known to be highly edge-sensitive and reluctant to cross open areas (Nichols *et al* 2008, Noble *et al* 2023), which may further restrict recolonisation into disturbed zones. These findings highlight the ecological sensitivity of Scarabaeinae to canopy loss and thermal shifts and underscore the need for active habitat restoration to support their recovery.

Low capture rates within the centre of mines could be attributed to the disruption of biotic interactions, particularly the loss of food resources. Human disturbance can severely reduce mammal populations, thereby decreasing the availability of dung required by beetles (Culot *et al* 2013).

Correa-Cuadros *et al* (2022) found that this defaunation can impact dung beetle functional diversity as severely as deforestation, leading to the specific loss of large-bodied species and rollers that rely on mammal excrement. Furthermore, in these exposed open areas, elevated temperatures accelerate bait desiccation, potentially reducing trap effectiveness and resource quality compared to shaded, forested sites (Larsen and Forsyth 2005). Beyond resource constraints, persistent environmental thresholds appear to be limiting recolonisation. The incomplete temperature dataset, a known constraint due to equipment failure during our data collection, still provides evidence that high temperatures are likely to be a key driver of dung beetle decline. The absence of recovery, despite mine abandonment up to two decades ago, suggests that deeper landscape barriers are at play. Severely degraded soils, disrupted seed banks, and altered microclimates may prevent the reestablishment of the forest structure critical for dung beetle habitat. Additionally, areas with a higher density of mines in the surrounding landscape could act as a barrier to dispersal, especially for edge-sensitive or less mobile species. Together, these factors highlight that passive regeneration may be insufficient to restore dung beetle assemblages, and that active restoration interventions may be required to overcome these ecological thresholds.

Current artisanal small-scale gold mining expansion in Amazonian is likely to have far reaching implications for dung beetle communities, and other species that are sensitive to habitat and microclimatic changes, including edge effects. These habitat alterations could well compromise ecosystem services, such as nutrient cycling and seed dispersal, as well as have cascading effects on the wider ecosystem (de Groot *et al* 2002). Moreover, the sensitivity of dung beetle populations to the habitat disturbance caused by artisanal small-scale gold mining underscores their use as reliable bioindicators for assessing environmental impacts in Amazonian ecosystems. We acknowledge that the relatively small number of control sites may limit the broader generalisability of our findings. However, they provide a valuable comparison for the specific, localised effects of ASGM and highlight the stark differences between undisturbed forest and mined areas. These insights reinforce the need for targeted conservation and restoration efforts and sustainable mining practices to mitigate biodiversity loss across this highly biodiverse region.

5. Conclusion

Our findings highlight how artisanal small-scale gold mining alters the structure and function of tropical forest ecosystems, with measurable consequences for dung beetle communities. By identifying canopy cover and temperature as key environmental drivers, we provide a clearer understanding of the ecological mechanisms that limit dung beetle persistence and recovery. These insights strengthen the case for using dung beetles as effective bioindicators in post-disturbance monitoring. As mining continues to expand in Amazonia, integrating such faunal indicators into restoration planning will be critical to supporting ecosystem resilience and biodiversity recovery.

Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: <https://zenodo.org/records/17192092> (Glynn 2025).

Supplementary data 1 available at <http://doi.org/10.1088/2752-664X/ae43f2/data1>.

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Conflict of interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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