RESEARCH ARTICLE

Diversity and Distributions    WILEY

# Multiple facets of biodiversity are threatened by mining-induced land-use change in the Brazilian Amazon

Thomas J. Lloyd[1,2] ⬡  |  Ubirajara Oliveira[3]  |  Britaldo S. Soares-Filho[3]  |  Richard A. Fuller[2,4]  |
Nathalie Butt[1,2]  |  John S. Ascher[5]  |  João Paulo Peixoto Pena Barbosa[6]  |
João Aguiar Nogueira Batista[7]  |  Antonio D. Brescovit[6]  |  Claudio J. B. de Carvalho[8]  |
Paulo De Marco[9]  |  Viviane Gianluppi Ferro[9]  |  Felipe Sá Fortes Leite[10]  |
Peter Löwenberg-Neto[11]  |  Adriano Pereira Paglia[12]  |  Daniella Teixeira de Rezende[13]  |
Adalberto J. Santos[14]  |  Daniel Paiva Silva[15] ⬡  |  Marcelo Ferreira de Vasconcelos[16]  |
Laura J. Sonter[1,2]

[1]School of Earth and Environmental Sciences, The University of Queensland, St Lucia, Queensland, Australia

[2]Centre for Biodiversity & Conservation Science, The University of Queensland, St Lucia, Queensland, Australia

[3]Centro de Sensoriamento Remoto, Instituto de Geociências, Universidade Federal de Minas Gerais, Belo Horizonte, Brazil

[4]School of Biological Sciences, The University of Queensland, St Lucia, Queensland, Australia

[5]Department of Biological Sciences, National University of Singapore, Singapore, Singapore

[6]Laboratório Especial de Coleções Zoológicas, Instituto Butantan, São Paulo, Brazil

[7]Departamento de Botânica, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Belo Horizonte, Brazil

[8]Departamento de Zoologia, Universidade Federal do Paraná, Curitiba, Brazil

[9]Departamento de Ecologia, Instituto de Ciências Biológicas, Universidade Federal de Goiás, Goiania, Brazil

[10]Laboratório Sagarana, Instituto de Ciências Biológicas e da Saúde, Universidade Federal de Viçosa, Florestal, Brazil

[11]Universidade Federal da Integração Latino-Americana, Foz do Iguaçu, Brazil

[12]Departamento de Biologia Geral, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Belo Horizonte, Brazil

[13]Sección Paleontología de Vertebrados, Museo Argentino de Ciencias Naturales Bernardino Rivadavia, Buenos Aires, Argentina

[14]Departamento de Zoologia, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Belo Horizonte, Brazil

[15]Conservation Biogeography and Macroecology Lab, Departamento de Ciências Biológicas, Instituto Federal Goiano, Urutaí, Brazil

[16]Museu de Ciências Naturais, Pontifícia Universidade Católica de Minas Gerais, Belo Horizonte, Brazil

**Correspondence**
Thomas J. Lloyd, School of Earth and Environmental Sciences, The University of Queensland, St Lucia, 4072, Australia.
Email: thomas.lloyd@uq.edu.au

## Abstract

**Aim:** Mining is increasingly pressuring areas of critical importance for biodiversity conservation, such as the Brazilian Amazon. Biodiversity data are limited in the tropics, restricting the scope for risks to be appropriately estimated before mineral licensing decisions are made. As the distributions and range sizes of other taxa differ markedly from those of vertebrates—the common proxy for analysis of risk to biodiversity from mining—whether mining threatens lesser-studied taxonomic groups differentially at a regional scale is unclear.

**Location:** Brazilian Amazon.

**Methods:** We assess risks to several facets of biodiversity from industrial mining by comparing mining areas (within 70 km of an active mining lease) and areas unaffected by mining, employing species richness, species endemism, phylogenetic diversity and phylogenetic endemism metrics calculated for angiosperms, arthropods and vertebrates.

**Results:** Mining areas contained higher densities of species occurrence records than the unaffected landscape, and we accounted for this sampling bias in our analyses. None of the four biodiversity metrics differed between mining and nonmining areas for vertebrates. For arthropods, species endemism was greater in mined areas. Mined areas also had greater angiosperm species richness, phylogenetic diversity and phylogenetic endemism, although less species endemism than unmined areas.

**Main Conclusions:** Unlike for vertebrates, facets of angiosperm and arthropod diversity are relatively higher in areas of mining activity, underscoring the need to consider multiple taxonomic groups and biodiversity facets when assessing risk and evaluating management options for mining threats. Particularly concerning is the proximity of mining to areas supporting deep evolutionary history, which may be impossible to recover or replace. As pressures to expand mining in the Amazon grow, impact assessments with broader taxonomic reach and metric focus will be vital to conserving biodiversity in mining regions.

**KEYWORDS**
Endemism, evolutionary potential, extractive industries, habitat loss, indirect impacts, mineral resource governance, phylogenetic diversity, phylogeography, species richness

## 1 | INTRODUCTION

Mining is an important driver of tropical deforestation worldwide, both directly (Edwards et al., 2014; Harfoot et al., 2018; Laurance et al., 2001) and indirectly (Curtis et al., 2018; Giljum et al., 2022; Laurance et al., 2015; Sloan et al., 2019; Weng et al., 2013). In the Brazilian Amazon, one of the largest and most biodiverse intact bioregions on Earth (Potapov et al., 2017), industrial-scale mineral mining drove nearly 12,000 km²—or an area the size of Qatar—of forest loss between 2005 and 2015 (Sonter et al., 2017). Such losses result directly from mineral extraction (i.e. due to land-use change at mine sites) and extensive indirect effects associated with establishing infrastructure for mineral extraction, processing and transportation (Sonter, Moran, et al., 2014). In the Brazilian Amazon, mining-induced forest loss occurs up to 70 km from lease boundaries (Sonter et al., 2017). Nonetheless, research examining the extent of mining-induced land-use change has been at odds with an outdated yet perpetuated concept of mining affecting only small geographic areas (Bridge, 2004; Marsh, 1864). This perception is despite a large body of research now demonstrating the extent of mining-induced land-use change, with much evidence emanating from Brazil explicitly (Alvarez-Berríos & Aide, 2015; Bebbington et al., 2018; Hänggli et al., 2023; Kamino et al., 2020; Santos et al., 2020; Siqueira-Gay, Sonter, & Sánchez, 2020; Sonter et al., 2015; Sonter, Barrett, et al., 2014; Souza-Filho et al., 2018) as well as globally (Giljum et al., 2022; Werner et al., 2019).

As mining activities are granted access to areas that are otherwise off-limits to industry, the quality and conservation importance of forests that can be impacted by mining make it a unique driver of forest loss (Grantham et al., 2021). By potentially enabling clearing in the undisturbed forest interior to radiate outwards via linear infrastructure, further deforestation beyond that necessary for mining-associated infrastructure can result if stringent protocols are not in place to restrict access to other industries, illegal loggers and miners, or spontaneous settlements (Bebbington et al., 2018; Edwards et al., 2014; Giljum et al., 2022; Laurance et al., 2014; Laurance & Balmford, 2013; Siqueira-Gay, Sonter, & Sánchez, 2020). Concerningly, mining claims in Brazil have the highest overlap with intact forest landscapes anywhere globally, covering approximately 370,000 km² (Grantham et al., 2021), risking considerable losses of the highest conservation value forest if they are ultimately developed for mineral extraction. However, even modest forest loss in these areas may be especially problematic for biodiversity, as evidenced by the disproportionate detrimental consequences for vertebrate diversity associated with intact forest loss (Betts et al., 2017).

Given the extensive deforestation footprints linked to mining, considering biodiversity risks beyond direct mine footprints is crucial for estimating the complete suite of potential impacts mining operations encompass. Several studies have begun to reveal the negative consequences of mining for biodiversity beyond lease boundaries at local scales, such as for primates and other medium-to-large-bodied

mammals in Ghana (Owusu et al., 2018), and above-ground carbon, stem density and tree and butterfly richness in Tanzania (Seki et al., 2022). At regional scales, factoring in the indirect impacts of mining, Sonter et al. (2022) show substantial mining regions in Brazil overlap with high diversity mammal habitats. Iron ore-rich areas also coincide with regions of comparatively high plant species richness in Brazil (Murguía et al., 2016). Areas of unsurpassed vertebrate richness in northern South America are similarly shown to be at increased risk of future fossil fuel exploitation (Butt et al., 2013), as are a variety of taxonomic groups and diversity facets incorporating range restrictions, with fine-scale areas of conflict between fossil fuel extraction and biodiversity having also been identified (Harfoot et al., 2018).

Globally, half of all mines are within 20 km of a protected area, with Brazil being a hotspot for mining growth in direct proximity to protected areas, and extraction volume is increasing within moist broadleaf forests more so than any other biome (Luckeneder et al., 2021). Bauxite, copper, iron and zinc mines specifically exhibit high overlap with protected areas in South America, with considerable activity within 1–5 km of protected areas in the Brazilian Amazon (Durán et al., 2013). At continental scales, infrastructure development driven by mining (Weng et al., 2013) has potentially far-reaching and damaging consequences for biodiversity (Laurance et al., 2009, 2015). The broader ecological theory also indicates that cumulative anthropogenic disturbances would substantially compound the biodiversity losses driven by mining-associated infrastructure in isolation (Barlow et al., 2016). Mining-induced deforestation is thus a substantial threat to forest biodiversity and valuable ecosystem services (Strand et al., 2018), especially where mineral-rich areas coincide with intact primary forests (Murguía et al., 2016; Siqueira-Gay, Soares-Filho, et al., 2020). Nonetheless, a more detailed examination is required to determine where and what biodiversity values are most at risk and what options are available and practicable for impact avoidance and mitigation.

Previous research on risks from mining to biodiversity at broad spatial scales has traditionally focussed on vertebrates (Edwards et al., 2014; Finer et al., 2008; Sonter et al., 2022), with just one study explicitly considering risks for plants (Murguía et al., 2016), and none examine arthropods. Although direct site-level mining impacts on plants are often explored, the only regional scale assessment of risk (Murguía et al., 2016) is limited to the examination of broad plant diversity zones (Barthlott et al., 2005, 2007), highlighting the need for more detailed investigation. These limitations are mainly due to the poor sampling of Brazilian Amazonian biodiversity in less easily accessible areas, as with much tropical diversity, leading to knowledge gaps and shortfalls (Oliveira et al., 2016; Oliveira, Soares-Filho, Santos, et al., 2019). Indeed, due to the large areas of botanically unsampled forests that have been cleared in the Brazilian Amazon, much of the unknown diversity has already been lost (Stropp et al., 2020). Concerningly, such losses have failed to generate the urgency necessary to explore threats to plant diversity with the best currently available data in areas at near-term risk of deforestation, the most pressing priorities of which are the lowland tropics (Corlett, 2016).

Although data paucity may affect assessments of conservation priorities (Bini et al., 2006), conservation decisions often need to be made urgently without the benefit of more extensive sampling (Grantham et al., 2008, 2009). Knowledge of conservation status for arthropods is the most poorly resolved of all macrobiota, with less than 0.5% of described species evaluated on the IUCN Red List (Cardoso et al., 2011). There is also evidence of significant declines in insect abundance and diversity in Brazil (Lewinsohn et al., 2022), suggesting substantial losses of undescribed diversity have already occurred in these taxa. Of the small proportion of insects that have been assessed for their representation in protected areas, only one-in-four are adequately covered (Chowdhury et al., 2022), despite their fundamental importance to ecosystem processes and functioning (Ollerton et al., 2011; Potts et al., 2010; Seibold et al., 2021; Wilson, 1987) and their contributions to people (Potts et al., 2016). Given new species are regularly identified in Amazonia (Valsecchi et al., 2017)—even among well-studied vertebrate orders as taxonomies are resolved through molecular techniques (e.g. Costa-Araújo et al., 2021; Ennes Silva et al., 2022)—a substantial proportion of arthropod diversity in the Brazilian Amazon remains undiscovered. Examination of imminent threats to understudied arthropod diversity thus cannot be delayed for want of more complete data to avoid putting further unknown diversity at risk. Delaying action will also likely make conservation actions less effective and more costly in remote areas susceptible to mineral exploitation (Cimon-Morin et al., 2016).

In addition to the narrow taxonomic restriction in assessments of regional-scale mining risks to biodiversity, the metrics used are also often limited. For example, species or taxon richness is the most fundamental, easily scalable and reproducible measure of biodiversity applied to impact assessments. Although relatively few studies examine broad-scale risks to biodiversity from mining relative to other threats, earlier research also mostly explored species richness values in proximity to areas targeted for large-scale mineral exploitation (e.g. Durán et al., 2013; Murguía et al., 2016). Yet, richness alone does not capture the rarity of organisms found in a region, or restrictions in their distribution, leading some resource extraction-focussed studies to apply metrics such as range rarity and species richness aggregates (Harfoot et al., 2018), or areas of species endemism (Edwards et al., 2014) to provide more holistic assessments of biodiversity values at risk. Furthermore, phylogenetic diversity is rarely evaluated, despite its importance for maintaining evolutionary lineages (Mace et al., 2003), providing a basis for future speciation in response to changing environmental conditions (Sgrò et al., 2011), and capturing unknown facets of diversity (Faith, 2017). The overlooking of phylogenetic diversity measures may initially have resulted from assertions that species richness is an effective surrogate of phylogenetic diversity for some vertebrate groups in conservation planning studies, particularly birds and mammals (Brooks et al., 2006; Rodrigues et al., 2005; Rodrigues & Gaston, 2002), despite evidence to the contrary at least for plants (Forest et al., 2007). However, an expanding body of research (Brooks et al., 2015; González-Orozco

et al., 2016; Jetz et al., 2014) signals the growing acceptance of the importance of considering threats to evolutionary history explicitly.

Here, we examine the potential risks of mining-induced deforestation to biodiversity across the Brazilian Amazon. We expand on previous broad-scale studies of mining threats to biodiversity by considering the combined risks of direct and indirect deforestation and employing diversity parameters beyond species richness and absolute endemism. To do so, we utilise area-weighted metrics and phylogenetic approaches to determine risks to biodiversity facets posed by the complete land-use footprint of mining previously unexplored hitherto. Specifically, we (1) investigate the biodiversity sampling effort within mining leases and mining areas (defined to occur within 70 km of mining leases); (2) determine whether mining areas contain more or less biodiversity than areas unaffected by mining using species richness, species endemism, phylogenetic diversity and phylogenetic endemism metrics; and (3) compare differences among three taxonomic groups: angiosperms, arthropods and vertebrates.

## 2 | METHODS

### 2.1 | Database assembly

#### 2.1.1 | Mapping mining areas

We obtained spatial information on mineral prospecting and mineral mining leases within the Brazilian Amazon from SIGMINE (Sistema de Informações Geográficas da Mineração; DNPM, 2012). This database catalogues all registered legal mining activities within Brazil, detailing the extent of each activity, dates of operation and mined commodities. To map 'mining leases' of industrial-scale mineral mines, we selected records greater than 100 hectares in area and classified as mining concessions (*Concessão de Lavra*) and omitted leases extracting water or those classified as small-scale artisanal operations (*Lavra Garimpeira*). This resulted in 411 polygons (including active leases and adjacent extensions of such leases) of 15,750 km$^2$ in total area, with mining start dates ranging from 1944 to 2017 (mean = 1978, sd = 11.9; Figure 1). To map 'mining areas', which include the direct (i.e. immediate land-use change resulting from mineral extraction) and indirect (i.e. extensive land-use change associated with mineral extraction, processing and transportation) impacts of mining on forests (Sonter et al., 2017), we created a 70 km buffer surrounding each mining lease. 'Non-mining areas' (i.e. areas unaffected by industrial mining) were mapped by extracting our mapped 'mining areas' and an additional layer representing all other legal mining leases excluded from our analyses (i.e. inactive leases, those targeting water, or operations smaller than 100 hectares in area; shown in white in Figure 1) from the Brazilian Amazon (Figure 1).

For interpolation analyses, hexagons are the most logical sampling unit shape as their centroids are equidistant, the distance of points from the edges to the centroid is the closest, and sampling biases are reduced due to their lower perimeter-area ratio compared with squares or triangles (Birch et al., 2007). Hexagons of approximately 0.5° with equal area were assigned to one of two study areas—mining areas or nonmining areas—based on where their centroid was located (Figure 1). Hexagons were omitted from our analyses if they contained fewer than 20 occurrence records per taxonomic group or their centroid was located outside the Brazilian Amazon. We used 0.5° hexagon sampling units as sensitivity analyses conducted in previous studies utilising the same dataset indicated reduced variation in results for hexagon areas of 0.5° and above (Oliveira, Soares-Filho, et al., 2017; Strand et al., 2018), and so any fine-scale georeferencing inaccuracies remaining in the dataset after filtering are minimised (Oliveira, Brescovit, & Santos, 2017). This sampling unit area also ensured sufficient sample sizes would be assigned within and among mining-induced deforestation-affected areas to enable robust comparisons across the study area for all taxonomic groups, particularly arthropods, while reducing the amount of area hexagon interpolations may sample from outside their respective study area polygons.

#### 2.1.2 | Assembling biodiversity data

Data on species occurrences were obtained from (Oliveira, Soares-Filho, et al., 2017) and (Oliveira, Soares-Filho, Santos, et al., 2019) and represent the most comprehensive dataset of species occurrences in Brazil to date. These data were assembled from online databases spanning GBIF (gbif.org); CRIA (specieslink.net); Birdlife International (birdlife.org), Herpnet (herpnet.org), Nature Serve (natureserve.org); and Orthoptera Species File (orthoptera.speciesfile.org). These data were also supplemented with occurrence records obtained from taxonomic literature and biodiversity inventories (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019). All species occurrence records were filtered to determine whether they lacked geographic coordinates or exhibited location errors using a map of Brazilian municipalities (mapas.ibge.gov.br; Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019). Taxonomic validity for all occurrence records was confirmed using taxon-specific catalogues and expert reviews for each taxonomic group (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019). After filtering for geographic and taxonomic accuracy, the final dataset comprised 113,790 occurrence records for all the Brazilian Amazon. The dataset contained 44,660 records of angiosperms (6899 species of families Asteraceae, Bromeliaceae, Fabaceae, Melastomataceae, Myrtaceae, Orchidaceae, Poaceae and Rubiaceae), 24,374 records of arthropods (4630 species of bees, spiders, millipedes, Orthoptera, dragonflies, moths and Diptera) and 44,756 records of vertebrates (1584 species of birds, mammals and anurans). Spatial distributions of occurrence record densities for each taxonomic group are provided in the Figure S1.
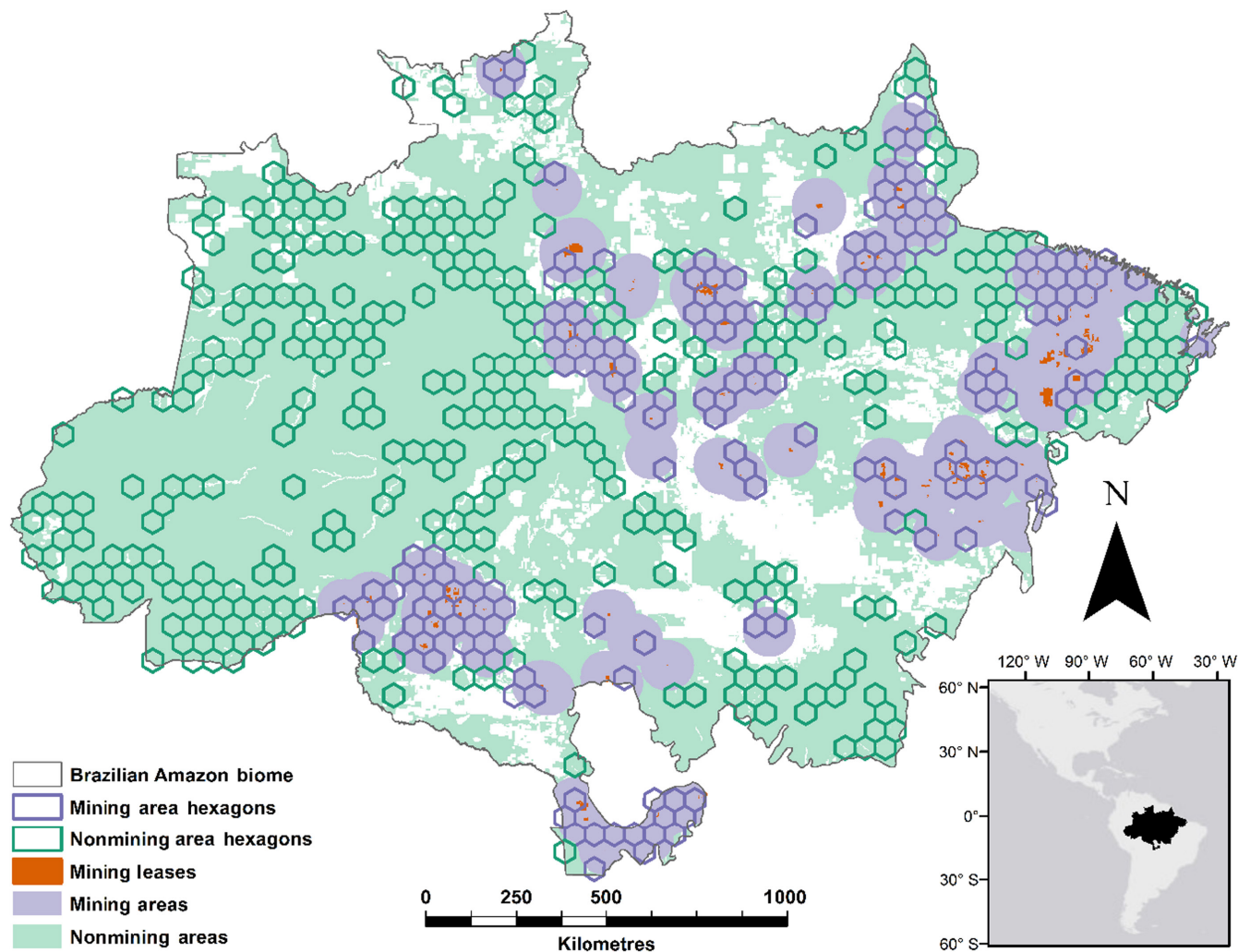
**FIGURE 1** Study area map showing legal mining activities across the Brazilian Amazon biome. Active mining leases (>100 ha) are shaded orange (total area: 15,750 km$^2$). Mining areas, which extend 70 km around active mining leases, are shaded purple (977,738 km$^2$); nonmining areas, free of any legal mining activity, are shaded green (2,540,188 km$^2$). Hexagons are outlined in dark purple for mining areas ($n=178$ with ≥20 species occurrence records for all taxonomic groups) and dark green for nonmining areas ($n=332$ with ≥20 species occurrence records for all taxonomic groups; see Table S1 for breakdowns by taxonomic group and metric).

Phylogenetic trees were constructed from published figures into Newick code with TreeSnatcherPlus (Laubach & Von Haeseler, 2007) and supplemented with data from empirical phylogenetic studies synthesised by The Open Tree of Life project (Hinchliff et al., 2015). As branch lengths, when available, are not directly comparable between trees, all branch lengths were considered equal to one (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019). Phylogenetic trees were compiled into a supertree using matrix representation with parsimony (Baum, 1992) and pruned to represent species restricted to Brazil. Our dataset represents the most extensive collection of species occurrence records and phylogenetic trees compiled in Brazil for this purpose to date (Oliveira, Soares-Filho, et al., 2017). However, data collected for environmental impact assessments that are not published online will inevitably be missing from our database, and rare, threatened or range-restricted organisms may also not be included due to limited sampling.

## 2.2 | Calculation of biodiversity facets

### 2.2.1 | Sampling effort

We first intersected mining lease and mining area polygons with species occurrence records to provide a coarse estimate of the proportion of occurrence records within mining leases and their more expansive impact areas from the total contained in our database. An equal-area measure was calculated through the 'Sampling Effort' functor of the BioDinamica plug-in (Oliveira, Soares-Filho, Leitão, & Rodrigues, 2019) of Dinamica EGO (Ferreira et al., 2019), which was set with a 10 km search radius due to limited and sporadic biodiversity sampling in the Brazilian Amazon (Oliveira et al., 2016; Oliveira, Soares-Filho, et al., 2017). We then converted the output raster to points and summed the mean sample effort index values across 0.5° radius hexagon sampling units (Figure 1). The 'Sampling Effort' functor in BioDinamica employs a Gaussian

kernel density index function. For all analyses using BioDinamica, 0.5° hexagon sampling units were only created where ≥20 species occurrence records existed.

## 2.2.2 | Biodiversity metrics

We calculated four sampling-effort-corrected biodiversity metrics for each of the three taxonomic groups: species richness, species endemism, phylogenetic diversity and phylogenetic endemism, since measuring biodiversity with species richness alone does not capture values pertinent to conservation at the landscape scale, such as endemism or evolutionary history (Faith, 1992; Faith et al., 2004). Indeed, the loss of species is not equivalent to the loss of evolutionary history (Vane-Wright et al., 1991), and conservation priority areas can differ when using species richness and phylogenetic diversity (Forest et al., 2007; Rodrigues et al., 2005). Furthermore, phylogenetic measures may capture the quantity and distribution of diversity better than species-based measures, especially when data are limited, but both are representative of different diversity components (Rosauer & Mooers, 2013; Tucker et al., 2017). Thus, here we employ a variety of biodiversity metrics for comparison between mining and nonmining areas in the Brazilian Amazon.

## 2.2.3 | Species-based metrics

Species richness (per unit area) is the most sensitive biodiversity measure to variation in sampling effort (Oliveira et al., 2016). To quantify species richness, we used a resampled species richness index to account for variation in sampling effort. The 'Resample Species Richness' functor (Oliveira, Soares-Filho, Leitão, & Rodrigues, 2019) operates by spatially resampling species occurrences. We set this functor to a minimum of 20 species occurrences per hexagon sampling unit, taking a random 25% subsample with 1000 iterations. This tolerance level retained the most variation in species richness while maintaining an adequate sample size to compare mining and nonmining areas. The output represents the mean resampled species richness per hexagon (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019). This method provides a relative measure of species richness, simulating uniform sampling throughout the study area, thus addressing variation in sampling effort (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019).

For comparisons of endemism, the level of geographic restriction among species, we used the weighted endemism index, a relative measure of endemism as opposed to an absolute measure (Williams & Humphries, 1994). We computed this using the 'Weighted Endemism' functor (Ferreira et al., 2019), which calculates the inverse of a species distribution area and sums it across hexagon sampling units (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Santos, et al., 2019). To control for variation in sampling effort and uncertainty in species distribution estimations, the functor

generates a sampling effort-corrected and area-weighted endemism index using the equation:

$$\frac{A*B}{((A*B) + ((1-A)*(1-C)))}$$

where 'A' is the weighted endemism as determined by the inverse of a species distribution area, 'B' is the product of weighted endemism and sampling effort (expressed as the mean kernel density index of species occurrence records within a 50 km radius of each occurrence point for each species analysed), and C is the total number of species sampled, with 150 species occurrences set as the maximum (0.999) as the frequency distribution of species records reaches an asymptote at this value (Oliveira, Soares-Filho, et al., 2017). Species with fewer records are assigned values beginning at 0.00001 for a single record upwards linearly (Oliveira, Soares-Filho, et al., 2017).

## 2.2.4 | Phylogenetic metrics

We considered two measures of spatial phylogenetic variation important for the maintenance and persistence of biodiversity, phylogenetic diversity (PD), a measure of divergence in phylogenetic relationships between species in an area (Faith, 1994), and phylogenetic endemism (PE), a measure of the restriction of phylogenetic lineages between given areas (Rosauer et al., 2009). Phylogenetic diversity is estimated by comparing summed distances between phylogenetic branches among species in a given area (Faith, 1992). Phylogenetic endemism is estimated by comparing the relative rarity of evolutionary lines among taxa between areas, with fewer branches at higher taxonomic classifications being afforded greater weighting in contributing to the endemism of a species' evolutionary lineage (Rosauer et al., 2009).

We compared phylogenetic diversity using branch lengths as a surrogate for the uniqueness or similarity in features of species within a phylogenetic tree (Faith, 1992). The 'Phylogenetic Diversity' functor sums branch lengths through the root of phylogenetic trees using the shortest path between species connected within a sampling unit (Faith, 1992, 1994; Oliveira, Soares-Filho, et al., 2017). Due to the paucity of phylogenetic information for Brazilian species and hence within our database, despite it being the most comprehensive dataset compiled in Brazil to date (Oliveira, Soares-Filho, et al., 2017), branches were assigned equal lengths under an assumption of equivalent rates of feature descent across all phylogenetic pathways (Faith, 1992). In comparing phylogenetic measures of biodiversity, we used phylogenetic trees for taxa geographically restricted to Brazil (Oliveira, Soares-Filho, et al., 2017).

To compare phylogenetic endemism, a measure of geographic restriction of phylogenetic diversity and hence evolutionary history, we used the phylogenetic weighted endemism index (Rosauer et al., 2009). This index employs a relative measure of endemism rather than an absolute measure to compare geographic concentrations of evolutionary history, to address the sensitivity to spatial scale

apparent in absolute endemism measures (Rosauer et al., 2009). The 'Phylogenetic Endemism' functor (Ferreira et al., 2019) interpolates phylogenetic weighted endemism indices across sampling hexagons by summing the phylogenetic branch lengths between species (Oliveira, Soares-Filho, et al., 2017; Oliveira, Soares-Filho, Leitão, & Rodrigues, 2019).

## 2.3 | Data processing and analysis

Sampling effort and biodiversity indices were generated in the BioDinamica plug-in extension (Oliveira, Soares-Filho, Leitão, & Rodrigues, 2019) for the freely available Dinamica EGO software (Ferreira et al., 2019). The outputs from Dinamica were processed and assigned to study areas in ArcMap 10.7 (ESRI, 2018). Due to the amount of species occurrence records present in a sampling unit for each taxonomic group and the availability of phylogenetic tree data for each occurrence, the number of groups compared between mining and nonmining areas varied, as hexagon sampling units were removed from analyses if they contained <20 occurrence records per taxonomic group (see N column in Table S1). Statistical comparisons between mining and nonmining areas for all metrics and all taxonomic groups were made using two-tailed Wilcoxon rank-sum tests, with all graphical representations created in R (R Core Team, 2018). For improved visualisation, the four biodiversity metrics were rescaled between 0 and 1 to facilitate interpretation using the scales package in R (Wickham & Seidel, 2020), which maintains identical data spread. Sensitivity analyses showing the small and largely inconsequential variation in results when using alternative potential impact buffers of 20 and 50 km are included in the Supporting Information, noting that the largest difference in findings is observed for arthropod phylogenetic endemism where the sample size is substantially reduced (mining area hexagon $n=17$) when using a 20 km buffer versus a 70 km buffer (mining area hexagon $n=53$; Figures S2 and S3). Effect sizes are reported (Table S1) and compared in sensitivity analyses of mining risk buffers (Figure S3) to show the magnitude of difference in metrics between study areas and to support comparisons with future analyses. However, they are not to be interpreted as measures of mining impact on biodiversity due to the abstract complexity of their biological interpretation as they relate to differences in interpolated biodiversity metrics.

## 3 | RESULTS

Of the 411 polygons representing active mining leases and neighbouring lease extensions, 80% had no sampling whatsoever, 2% had low levels of sampling (0–0.01 records/km²), 17% were moderately sampled (0.01–10 records/km²), and none were well sampled (>10 records/km²). Mining leases covered 0.4% of the Brazilian Amazon and contained 1.3% of species records, while mining areas (i.e. within 70 km of an active mining lease) contained 37.4% of all species records in our database, despite covering only 23.4% of the study area. Mean sampling effort was significantly greater in mining areas than in nonmining areas for all taxonomic groups; however, effect sizes were notably small (Angiosperms: $W=104{,}351$,
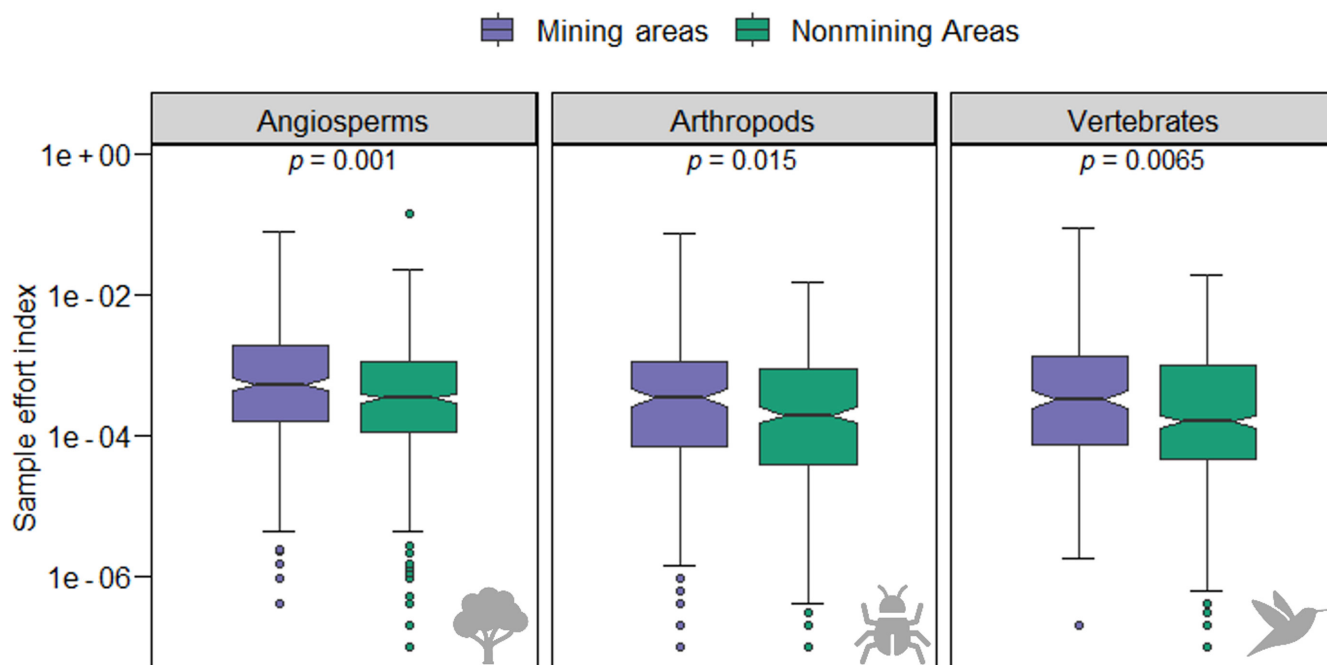


**FIGURE 2** Mean sample effort index per 0.5° hexagon containing 20 species occurrence records or more for Angiosperms, Arthropods and Vertebrates among mining and nonmining areas. Boxplots represent median, interquartile range (IQR) and limits (Q1 − IQR*1.5; Q3 + IQR*1.5). Sample effort index values plotted on a log scale; all p-values generated from two-tailed Wilcoxon rank-sum tests (see Table S1 for statistical summary).

$p=0.0011$, $r=0.108$; Arthropods: $W=48,741$, $p=0.01537$, $r=0.098$; Vertebrates: $W=79,543$, $p=0.00646$, $r=0.096$; Figure 2; Table S1).

Resampled species richness (corrected for differences in sampling effort) was significantly greater in mining areas for angiosperms, and while the effect size was relatively larger compared with other taxonomic groups, it was smaller than observed for other metrics ($W=13,058$, $p=0.0324$, $r=0.121$, Table S1, Figure S4). There was no significant difference in resampled species richness indices for arthropods ($W=3362$, $p=0.6592$, $r=0.034$) or vertebrates ($W=5384$, $p=0.2058$, $r=0.084$; Figure 3a). Mining areas had greater weighted endemism for arthropods and with the largest effect size of any taxonomic group ($W=4218$, $p=0.0120$, $r=0.193$). Mining areas contained less angiosperm endemism although the effect size was marginally smaller than for arthropods ($W=9372$, $p=0.0101$, $r=0.146$, Figure S4), and no observable differences existed between study areas for vertebrates ($W=5948$, $p=0.9267$, $r=0.006$; Figure 3b). Angiosperm phylogenetic diversity was significantly greater in mining areas with a relatively larger effect size compared with other taxonomic groups ($W=13,088$, $p=0.00008$, $r=0.230$, Figure S4), while no differences were observed for arthropods ($W=2683$, $p=0.9969$, $r=0.000$) or vertebrates ($W=6432$, $p=0.3391$, $r=0.064$; Figure 3c). Phylogenetic endemism was significantly higher in mining areas for angiosperms and with a relatively larger effect size compared with other taxonomic groups ($W=9385$, $p=0.0009$, $r=0.209$, Figure S4), while no observable differences existed among arthropods ($W=2056$, $p=0.5996$, $r=0.045$) or vertebrates ($W=6309.5$, $p=0.4847$, $r=0.046$; Figure 3d; see Table S1 for statistical summaries of all index comparisons).

## 4 | DISCUSSION

Mining in the Brazilian Amazon is at risk of dramatic increase due to substantial political support for pro-mining policy changes (Rorato et al., 2020; Villén-Pérez et al., 2018, 2022). Earlier studies have examined the extent of historical mining-induced deforestation in the Brazilian Amazon (Sonter et al., 2017), and here, we illustrate the potential risks of this land-use change for biodiversity. First, we find that many mining areas are poorly sampled—80% of mining leases have no species occurrence records whatsoever—and other rare, undiscovered or threatened species may also occur in these areas. These sampling shortfalls demonstrate a considerable opportunity for mining companies to improve national biodiversity inventories by openly sharing their impact assessment data. Accounting for sampling biases, our results indicate equal-area biodiversity values rank higher within wider mining-affected areas than unmined areas across the Brazilian Amazon. However, important differences exist

among biodiversity metrics and taxonomic groups, and the lack of biodiversity data in Brazil overall means that further research is required to validate these patterns. Our findings emphasise the importance of considering a variety of metrics and taxonomic groups when examining risks from mining or other similar threats to biodiversity in the Amazon, even under data limitation constraints. While more detailed studies are needed to explore the ecological relationships underpinning these diversity patterns, our results suggest any unmitigated impacts of mining may have had or will have greater consequences for angiosperm evolutionary history and range-restricted arthropods.

Based on our analyses utilising the most comprehensive compilation of species occurrence records for Brazil to date, mining areas appear to be more biodiverse than areas unaffected by industrial mining, at least for some metrics and taxonomic groups. For example, mining areas contain greater angiosperm species richness, higher arthropod endemism and greater angiosperm phylogenetic diversity and endemism (Figure 3, Figure S4). These results indicate that mining-induced deforestation may have more substantial consequences for some forms of biodiversity and may reflect an underlying association between minerals and some diversity facets. While more comprehensive analyses are required to explore these relationships, similar interactions have been observed previously in Brazil, where highly endemic and beta-diverse flora communities of endangered ironstone outcrops represent 26% of all vascular plant families (Jacobi et al., 2007; Salles et al., 2018). Understanding the conditions that explain these associations could help predict where mining and conservation conflicts may emerge in future. Knowledge of whether similar relationships occur in ecosystems elsewhere would also greatly benefit conservation planning in mining regions (Sonter et al., 2018). Regardless, mitigating the impacts of mining on plant communities ecologically associated with the mined commodity will be challenging as opportunities to secure their habitat elsewhere will likely conflict with similar economic pressures to extract minerals (Sonter et al., 2020).

Our findings also reveal differences between biodiversity metrics and taxonomic groups used to assess the conservation implications of mining-induced deforestation. Prior analyses consider a narrow set of metrics, typically species richness (Murguía et al., 2016), with some expanding to consider restrictions in species distributions (Harfoot et al., 2018; Lessmann et al., 2016), or focus solely on threats to vertebrate species (Edwards et al., 2014; Finer et al., 2008; Sonter et al., 2022). However, we find mining in the Brazilian Amazon may pose more substantial threats to phylogenetic diversity, specifically for angiosperms (Figure 3c), and higher concentrations of range-restricted arthropod species

**FIGURE 3** Biodiversity metrics per 0.5° hexagon containing 20 species occurrence records or more for Angiosperms, Arthropods and Vertebrates. Boxplots represent median, interquartile range (IQR) and limits (Q1 – IQR*1.5; Q3 + IQR*1.5), of (a) resampled species richness indices (25% subsamples); (b) weighted endemism indices; (c) phylogenetic diversity indices; and (d) phylogenetic endemism indices between mining and nonmining areas. Weighted endemism, phylogenetic diversity and phylogenetic endemism indices plotted on a log scale. All p-values generated from two-tailed Wilcoxon rank-sum tests (see Table S1 for statistical summary).
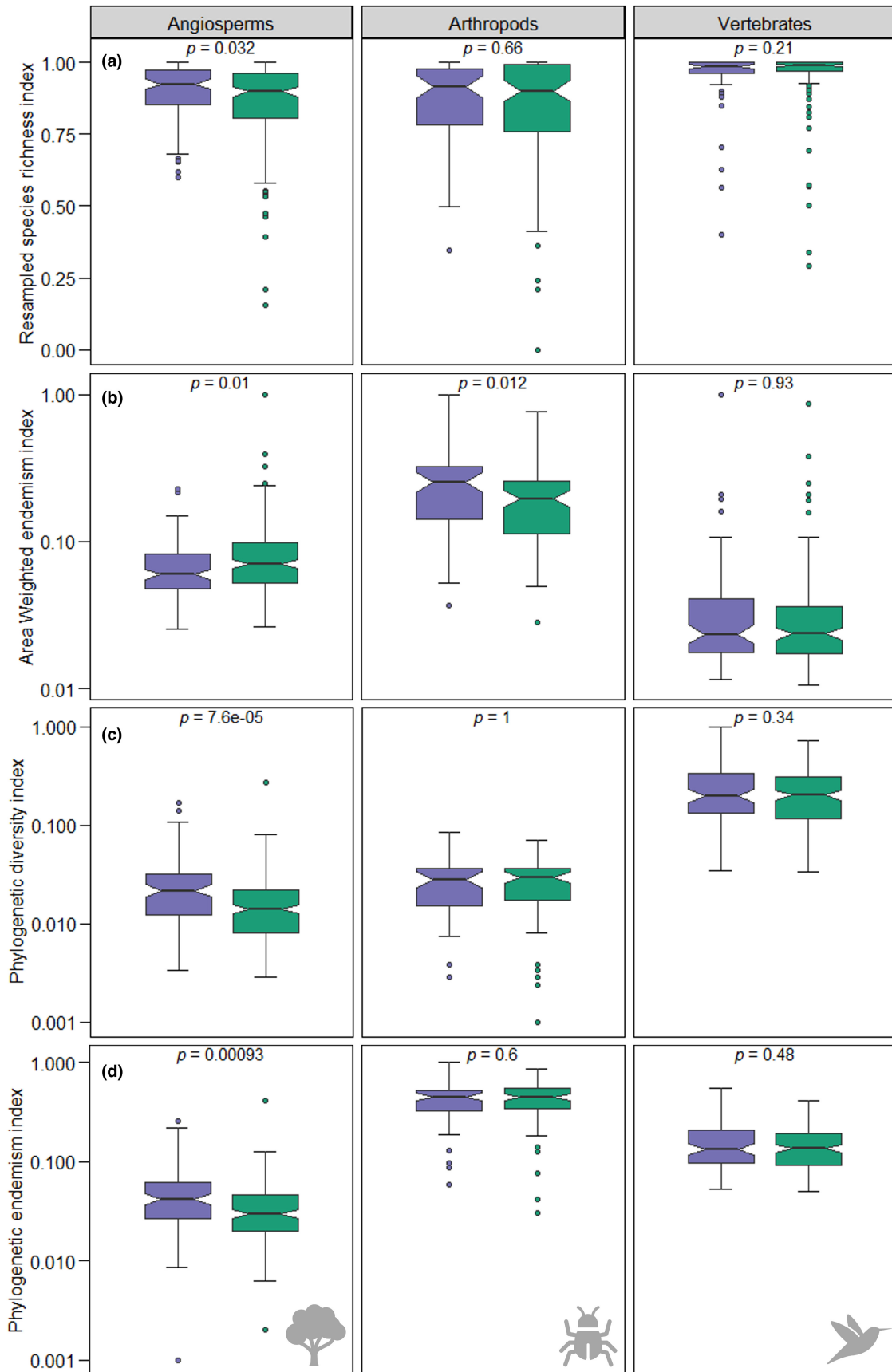
(Figure 3b). Mining may also pose greater risks to spatially restricted angiosperm evolutionary lineages (Figure 3d), with much of this variation being irreplaceable (Rosauer et al., 2009). To address these risks, environmental impact assessments and conservation planning exercises should focus on a complete range of taxa potentially affected by mining operations on- and off-site, at local and regional scales, and consider implications for biodiversity that extend beyond species richness. Failing to do so will not only perpetuate persistent biases within conservation (Di Marco et al., 2017; Troudet et al., 2017) but may also unduly risk biodiversity losses of lesser-studied yet ecologically important taxa (Cardoso et al., 2011; Lawler et al., 2003; Mouillot et al., 2013) and inordinately threaten evolutionary history, which is already being lost at rates faster than predicted by species extinction (Purvis et al., 2000). The loss of such diversity could severely hinder the ability of ecological communities and their components to adapt to changing environmental conditions (Sgrò et al., 2011), threaten the maintenance of evolutionary lineages that have adapted to current conditions (Mace et al., 2003) and risk losses of facets of biodiversity which are as yet unknown (Faith, 2017) that may provide relatively more option value, that is, a greater pool of biological resources from which people may derive benefit in future (Faith, 1992).

We find that mining may threaten phylogenetic facets more than species-based diversity and that discernible differences in risk to biodiversity would not have been captured if considering only vertebrates, underscoring the importance of comprehensive evaluation of diversity values when examining biodiversity risks of development in the Brazilian Amazon. Although diversity data are limited in the region, as with most tropical forests, our results indicate that the frequently applied proxies of richness for phylogenetic diversity and of vertebrates for other taxonomic groups to address these limitations may not be sufficiently representative to employ in decision-making at regional scales if the objective is to retain maximal phylogenetic diversity or diversity of nontarget taxonomic groups. Conservation research exploring links between mining and biodiversity should therefore examine an array of diversity metrics and taxonomic groups and consider the indirect footprint of mining. Without doing so, conservation scientists and decision-makers may be oblivious to the potentially undiscovered, unique and irreplaceable biodiversity within mining leases and their rapidly changing surrounding landscapes and risk substantial losses of diversity, much of which may never be known.

It is important to note that these results do not indicate that mining poses no threat to vertebrate diversity. Conversely, several approaches have demonstrated such threats at various scales (Edwards et al., 2014; Finer et al., 2008; Harfoot et al., 2018; Lessmann et al., 2016; Sonter et al., 2022). Nonetheless, should conservation planning scenarios be explored in Amazonian mining landscapes, vertebrate diversity may not be a reliable surrogate for the diversity of other taxonomic groups. Evidence suggests that animal diversity is positively associated with plant diversity at broad spatial scales (Castagneyrol & Jactel, 2012). However, the observed associations for diversity between taxonomic groups are primarily driven by environmental factors (Jetz et al., 2009), which may be effective planning surrogates for species (Beier et al., 2015) and genetic variation (Hanson et al., 2017) if chosen carefully. There may also be opportunities to reconcile broad spatial patterns of mineral ore deposits and plant diversity for conservation planning in mining regions, for example, using soil pH (Crespo-Mendes et al., 2019). Regardless, as biodiversity impact and risk assessments do not share the same objectives as conservation planning exercises, and given that broad-scale threats have also been shown to impact taxonomic groups differently (Warren et al., 2018), we caution that risks to taxonomic groups should be assessed separately.

We find overall greater levels of biodiversity in mining areas for some metrics and taxa. If these values occur exclusively within mining areas, are forest-dependent (and hence negatively affected by mining-induced deforestation) and lack suitable habitat or conditions to persist elsewhere, mining could pose serious extinction risks, particularly if sufficient opportunities to mitigate adverse impacts through ecological compensation do not exist (Sonter et al., 2020). Furthermore, threats driven by processes indirectly related to mining, such as land-use change for infrastructure establishment (Geist & Lambin, 2002), may pose more widespread risks to biodiversity. Indeed, many taxa found exclusively within mining areas may be threatened by activities indirectly related to the operation of the mining sector, such as linear infrastructure (Bebbington et al., 2018). Ensuring the persistence of biodiversity explicitly affected by mining, particularly in the face of increasing pressures and easing conservation regulations, will require an improved understanding of how multiple threats may interact and how they can be addressed concurrently. Infrastructure sharing, in particular, is one promising solution to minimise cumulative forest loss associated with informal industry proliferation following mining-driven infrastructure development (Runge et al., 2017), with potentially far-reaching outcomes for mitigating biodiversity loss (Barlow et al., 2016). Where new infrastructure is planned to facilitate mining, rigorous application of the Mitigation Hierarchy, where impacts are first avoided wherever possible (CSBI, 2015), and careful planning of infrastructure networks (Laurance et al., 2014) will be crucial to ensuring the long-term persistence of mining-threatened diversity.

Due to the often considerable temporal mismatch between the onset of mining operations and the development of infrastructure enabling resource extraction, this study set out to estimate the risks mining poses to a broader baseline of biodiversity that is likely to have been present before mining was initiated. Given that the Brazilian Amazon is a relatively recent frontier for industrial mining, most of the biodiversity records in our database likely predate the onset of mining. As such, the findings presented here are not intended to be interpreted as an assessment of the impacts of industrial mining on biodiversity, but rather an assessment of potential risk to biodiversity in mining-affected regions. In addition,

while a concerted attempt has been made to address bias in species occurrence records used in this study, given the large areas with no sampling whatsoever (Figure S1), a more thorough and spatially even sample of biodiversity across the Amazon may yield different results. Indeed, the dearth of data available for arthropods may result in markedly different diversity patterns compared with the true distributions. Remedying these biases by conducting traditional field inventories capable of informing decision-making at regional scales would require substantial capital inputs, which are notably limited in plant and arthropod conservation especially (Balding & Williams, 2016; Cardoso et al., 2011). Such limitations thus offer opportunities for the mining sector to improve their social licence to operate by contributing to the conservation and research of underresourced taxa, particularly in an area urgently requiring such attention as the Brazilian Amazon's highly diverse yet poorly inventoried lowland tropical forests (Corlett, 2016).

Mining areas had considerably higher sampling effort than areas more than 70 km from a mining lease (Figure 2), and these biases were directly addressed in analysing biodiversity metrics (Figure 3, Figure S1). Although mining companies are required to report on environmental impact assessments, the results are rarely made public, meaning biodiversity data were not expected to be any more or less abundant in mining areas. However, greater accessibility to mining areas (due to infrastructure development, potentially driven by mining) may explain sampling biases (Oliveira et al., 2016). While this underlines the importance of addressing such biases in analysing mining risk to biodiversity, it has also been shown that diversity values are more similar in areas closer to access routes (Oliveira et al., 2016), somewhat correcting for bias in regional diversity estimates. Nonetheless, mining areas, particularly in the eastern Amazon, are poorly sampled overall, with many containing less than 20 occurrence records per sampling unit (Figure 1). Furthermore, no mining leases had more than 10 records per km$^2$, and our database lacked species occurrence records for 80% of all leases. Consequently, many other unknown species may be at risk from mining (Stropp et al., 2020).

Low sampling effort in mining leases may reflect the inadequacies of environmental impact assessments in collecting and assessing biodiversity data within proposed mining sites (Dias et al., 2017; Ritter et al., 2017), particularly for angiosperms and, to a greater degree, arthropods. However, it may also be partly due to a lack of reporting on mining company survey data for more commonly surveyed taxonomic groups such as vertebrates. If the latter is the case, minor policy changes requiring impact assessors to upload their data from comprehensive surveys to online open-access repositories prior to leasing decisions could substantially expand biodiversity inventories. Such a change would aid conservation efforts as well as improve the transparency of mining impact assessments and licensing decisions. Given the relatively higher arthropod endemism, plant diversity and phylogenetic uniqueness found in mining areas, regional planning, impact and risk assessments should consider these diversity components explicitly and urgently. Updating evidence-based guidelines on selecting sites for mineral exploitation and

managing risk throughout the mining life cycle—as well as beyond lease boundaries—will be crucial to ensuring no further diversity is lost. Mining companies and financiers are thus in a unique and critical position to influence leadership on maintaining and regenerating biodiversity values in one of the last ecologically intact strongholds of terrestrial biodiversity globally.

## CONFLICT OF INTEREST STATEMENT

The authors have no conflicts of interest to declare.

## PEER REVIEW

The peer review history for this article is available at https://www.webofscience.com/api/gateway/wos/peer-review/10.1111/ddi.13753.

## DATA AVAILABILITY STATEMENT

Data used in this study are available in online materials associated with Oliveira, Brescovit, and Santos (2017) and Oliveira, Soares-Filho, Leitão, and Rodrigues (2019), as well as on DRYAD at https://doi.org/10.5061/dryad.s1rn8pkcm.

## ORCID

*Thomas J. Lloyd* https://orcid.org/0000-0001-9940-8767
*Daniel Paiva Silva* https://orcid.org/0000-0002-2457-6245

## REFERENCES

Alvarez-Berríos, N. L., & Aide, T. M. (2015). Global demand for gold is another threat for tropical forests. *Environmental Research Letters*, *10*(1), 014006.

Balding, M., & Williams, K. J. H. (2016). Plant blindness and the implications for plant conservation. *Conservation Biology*, *30*(6), 1192–1199. https://doi.org/10.1111/cobi.12738

Barlow, J., Lennox, G. D., Ferreira, J., Berenguer, E., Lees, A. C., Mac Nally, R., Thomson, J. R., de Barros Ferraz, S. F., Louzada, J., Oliveira, V. H. F., Parry, L., de Castro Solar, R. R., Vieira, I. C. G., Aragão, L. E. O. C., Begotti, R. A., Braga, R. F., Cardoso, T. M., de Oliveira, R. C., Jr., Souza, C. M., Jr., … Gardner, T. A. (2016). Anthropogenic disturbance

in tropical forests can double biodiversity loss from deforestation. *Nature*, *535*(7610), 144–147. https://doi.org/10.1038/nature18326

Barthlott, W., Hostert, A., Kier, G., Küper, W., Kreft, H., Mutke, J., Daud Rafiqpoor, M., & Sommer, J. H. (2007). Geographic patterns of vascular plant diversity at continental to global scales. *Erdkunde*, *61*, 305–315.

Barthlott, W., Mutke, J., Rafiqpoor, D., Kier, G., & Kreft, H. (2005). Global centers of vascular plant diversity. *Nova Acta Leopoldina NF*, *92*(342), 61–83.

Baum, B. R. (1992). Combining trees as a way of combining data sets for phylogenetic inference, and the desirability of combining gene trees. *Taxon*, *41*(1), 3–10. https://doi.org/10.2307/1222480

Bebbington, A. J., Bebbington, D. H., Sauls, L. A., Rogan, J., Agrawal, S., Gamboa, C., Imhof, A., Johnson, K., Rosa, H., Royo, A., Toumbourou, T., & Verdum, R. (2018). Resource extraction and infrastructure threaten forest cover and community rights. *Proceedings of the National Academy of Sciences*, *115*(52), 13164–13173.

Beier, P., Sutcliffe, P., Hjort, J., Faith, D. P., Pressey, R. L., & Albuquerque, F. (2015). A review of selection-based tests of abiotic surrogates for species representation. *Conservation Biology*, *29*(3), 668–679. https://doi.org/10.1111/cobi.12509

Betts, M. G., Wolf, C., Ripple, W. J., Phalan, B., Millers, K. A., Duarte, A., Butchart, S. H. M., & Levi, T. (2017). Global forest loss disproportionately erodes biodiversity in intact landscapes. *Nature*, *547*(7664), 441–444. https://doi.org/10.1038/nature23285

Bini, L. M., Diniz-Filho, J. A. F., Rangel, T. F., Bastos, R. P., & Pinto, M. P. (2006). Challenging Wallacean and Linnean shortfalls: Knowledge gradients and conservation planning in a biodiversity hotspot. *Diversity and Distributions*, *12*(5), 475–482.

Birch, C. P., Oom, S. P., & Beecham, J. A. (2007). Rectangular and hexagonal grids used for observation, experiment and simulation in ecology. *Ecological Modelling*, *206*(3–4), 347–359.

Bridge, G. (2004). Contested terrain: Mining and the environment. *Annual Review of Environment and Resources*, *29*(1), 205–259. https://doi.org/10.1146/annurev.energy.28.011503.163434

Brooks, T., Cuttelod, A., Faith, D., Garcia-Moreno, J., Langhammer, P., & Pérez-Espona, S. (2015). Why and how might genetic and phylogenetic diversity be reflected in the identification of key biodiversity areas? *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *370*(1662), 20140019.

Brooks, T. M., Mittermeier, R. A., Da Fonseca, G. A., Gerlach, J., Hoffmann, M., Lamoreux, J. F., Mittermeier, C. G., Pilgrim, J. D., & Rodrigues, A. S. (2006). Global biodiversity conservation priorities. *Science*, *313*(5783), 58–61.

Butt, N., Beyer, H., Bennett, J., Biggs, D., Maggini, R., Mills, M., Renwick, A. R., Seabrook, L. M., & Possingham, H. (2013). Biodiversity risks from fossil fuel extraction. *Science*, *342*(6157), 425–426.

Cardoso, P., Erwin, T. L., Borges, P. A., & New, T. R. (2011). The seven impediments in invertebrate conservation and how to overcome them. *Biological Conservation*, *144*(11), 2647–2655.

Castagneyrol, B., & Jactel, H. (2012). Unraveling plant–animal diversity relationships: A meta-regression analysis. *Ecology*, *93*(9), 2115–2124. https://doi.org/10.1890/11-1300.1

Chowdhury, S., Zalucki, M., Hanson, J., Tiatragul, S., Green, D., Watson, J., & Fuller, R. (2022). Three quarters of insects are insufficiently covered by protected areas.

Cimon-Morin, J., Darveau, M., & Poulin, M. (2016). Consequences of delaying conservation of ecosystem services in remote landscapes prone to natural resource exploitation. *Landscape Ecology*, *31*(4), 825–842. https://doi.org/10.1007/s10980-015-0291-4

Corlett, R. T. (2016). Plant diversity in a changing world: Status, trends, and conservation needs. *Plant Diversity*, *38*(1), 10–16. https://doi.org/10.1016/j.pld.2016.01.001

Costa-Araújo, R., Silva-Jr, J. S., Boubli, J. P., Rossi, R. V., Canale, G. R., Melo, F. R., Bertuol, F., da Silva, F. E., Silva, D. A., Nash, S., Sampaio, I., Farias, I. P., & Hrbek, T. (2021). An integrative analysis uncovers a new, pseudo-cryptic species of Amazonian marmoset (primates: Callitrichidae: Mico) from the arc of deforestation. *Scientific Reports*, *11*(1), 15665. https://doi.org/10.1038/s41598-021-93943-w

Crespo-Mendes, N., Laurent, A., Bruun, H. H., & Hauschild, M. Z. (2019). Relationships between plant species richness and soil pH at the level of biome and ecoregion in Brazil. *Ecological Indicators*, *98*, 266–275. https://doi.org/10.1016/j.ecolind.2018.11.004

CSBI. (2015). A cross-sector guide for implementing the mitigation hierarchy: Prepared by the Biodiversity Consultancy on behalf of IPIECA, ICMM and the Equator Principles Association: Cambridge, UK.

Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A., & Hansen, M. C. (2018). Classifying drivers of global forest loss. *Science*, *361*(6407), 1108–1111.

Di Marco, M., Chapman, S., Althor, G., Kearney, S., Besancon, C., Butt, N., Maina, J. M., Possingham, H. P., von Bieberstein, K. R., Venter, O., & Watson, J. E. M. (2017). Changing trends and persisting biases in three decades of conservation science. *Global Ecology and Conservation*, *10*, 32–42.

Dias, A. M. D. S., Fonseca, A., & Paglia, A. P. (2017). Biodiversity monitoring in the environmental impact assessment of mining projects: A (persistent) waste of time and money? *Perspectives in Ecology and Conservation*, *15*(3), 206–208. https://doi.org/10.1016/j.pecon.2017.06.001

DNPM. (2012). Processos Minerarios: Sistema de Informações Geográficas da Mineração (SIGMINE). Departamento Nacional de Produção Mineral (DNPM).

Durán, A. P., Rauch, J., & Gaston, K. J. (2013). Global spatial coincidence between protected areas and metal mining activities. *Biological Conservation*, *160*, 272–278. https://doi.org/10.1016/j.biocon.2013.02.003

Edwards, D. P., Sloan, S., Weng, L., Dirks, P., Sayer, J., & Laurance, W. F. (2014). Mining and the African environment. *Conservation Letters*, *7*(3), 302–311.

Ennes Silva, F., Valsecchi do Amaral, J., Roos, C., Bowler, M., Röhe, F., Sampaio, R., Janiak, M. C., Bertuol, F., Santana, M. I., de Souza Silva Júnior, J., Rylands, A. B., Gubili, C., Hrbek, T., McDevitt, A. D., & Boubli, J. P. (2022). Molecular phylogeny and systematics of bald uakaris, genus Cacajao lesson, 1840 (primates: Pitheciidae), with the description of a new species. *Molecular Phylogenetics and Evolution*, *173*, 107509. https://doi.org/10.1016/j.ympev.2022.107509

ESRI. (2018). Environmental Systems Research Institute (ESRI), ArcGIS release 10.7. Redlands, CA.

Faith, D. P. (1992). Conservation evaluation and phylogenetic diversity. *Biological Conservation*, *61*, 1–10.

Faith, D. P. (1994). Phylogenetic diversity: A general framework for the prediction of feature diversity. In P. L. Forey, C. J. Humphries, & R. I. Vane-Wright (Eds.), *Systematics and conservation evaluation* (pp. 251–268). Clarendon Press.

Faith, D. P. (2017). A general model for biodiversity and its value. In J. Garson, A. Plutynski, & S. Sarkar (Eds.), *Routledge handbook of philosophy of biodiversity*. Routledge.

Faith, D. P., Reid, C., & Hunter, J. (2004). Integrating phylogenetic diversity, complementarity, and endemism for conservation assessment. *Conservation Biology*, *18*(1), 255–261.

Ferreira, B. M., Soares-Filho, B. S., & Pereira, F. M. Q. (2019). The Dinamica EGO virtual machine. *Science of Computer Programming*, *173*, 3–20. https://doi.org/10.1016/j.scico.2018.02.002

Finer, M., Jenkins, C. N., Pimm, S. L., Keane, B., & Ross, C. (2008). Oil and gas projects in the western Amazon: Threats to wilderness, biodiversity, and indigenous peoples. *PLoS One*, *3*(8), e2932.

Forest, F., Grenyer, R., Rouget, M., Davies, T. J., Cowling, R. M., Faith, D. P., Balmford, A., Manning, J. C., Procheş, Ş., van der Bank, M., Reeves, G., Hedderson, T. A. J., & Savolainen, V. (2007). Preserving

the evolutionary potential of floras in biodiversity hotspots. *Nature*, 445(7129), 757–760. https://doi.org/10.1038/nature05587

Geist, H. J., & Lambin, E. F. (2002). Proximate causes and underlying driving forces of tropical deforestation: Tropical forests are disappearing as the result of many pressures, both local and regional, acting in various combinations in different geographical locations. *Bioscience*, 52(2), 143–150.

Giljum, S., Maus, V., Kuschnig, N., Luckeneder, S., Tost, M., Sonter, L. J., & Bebbington, A. J. (2022). A pantropical assessment of deforestation caused by industrial mining. *Proceedings of the National Academy of Sciences*, 119(38), e2118273119.

González-Orozco, C. E., Pollock, L. J., Thornhill, A. H., Mishler, B. D., Knerr, N., Laffan, S. W., Miller, J. T., Rosauer, D. F., Faith, D. P., Nipperess, D. A., Kujala, H., Linke, S., Butt, N., Külheim, C., Crisp, M. D., & Gruber, B. (2016). Phylogenetic approaches reveal biodiversity threats under climate change. *Nature Climate Change*, 6(12), 1110–1114. https://doi.org/10.1038/nclimate3126

Grantham, H. S., Moilanen, A., Wilson, K. A., Pressey, R. L., Rebelo, T. G., & Possingham, H. P. (2008). Diminishing return on investment for biodiversity data in conservation planning. *Conservation Letters*, 1(4), 190–198. https://doi.org/10.1111/j.1755-263X.2008.00029.x

Grantham, H. S., Tibaldeschi, P., Izquierdo, P., Mo, K., Patterson, D. J., Rainey, H., Watson, J. E. M., & Jones, K. R. (2021). The emerging threat of extractives sector to intact Forest landscapes. *Frontiers in Forests and Global Change*, 4(72), 692338. https://doi.org/10.3389/ffgc.2021.692338

Grantham, H. S., Wilson, K. A., Moilanen, A., Rebelo, T., & Possingham, H. P. (2009). Delaying conservation actions for improved knowledge: How long should we wait? *Ecology Letters*, 12(4), 293–301.

Hänggli, A., Levy, S., Armenteras Pascual, D., Bovolo, I., Brandao, J., Rueda, X., & Garrett, R. D. (2023). A systematic comparison of deforestation drivers and policy effectiveness across the Amazon biome. *Environmental Research Letters*, 18(7), 073001. https://doi.org/10.1088/1748-9326/acd408

Hanson, J. O., Rhodes, J. R., Riginos, C., & Fuller, R. A. (2017). Environmental and geographic variables are effective surrogates for genetic variation in conservation planning. *Proceedings of the National Academy of Sciences*, 114(48), 12755–12760.

Harfoot, M. B., Tittensor, D. P., Knight, S., Arnell, A. P., Blyth, S., Brooks, S., Butchart, S. H. M., Hutton, J., Jones, M. I., Kapos, V., Scharlemann, J. P. W., & Kapos, V. (2018). Present and future biodiversity risks from fossil fuel exploitation. *Conservation Letters*, 11(4), e12448.

Hinchliff, C. E., Smith, S. A., Allman, J. F., Burleigh, J. G., Chaudhary, R., Coghill, L. M., Crandall, K. A., Deng, J., Drew, B. T., Gazis, R., Gude, K., Hibbett, D. S., Katz, L. A., Dail Laughinghouse, H., IV, McTavish, E. J., Midford, P. E., Owen, C. L., Ree, R. H., Rees, J. A., ... Gazis, R. (2015). Synthesis of phylogeny and taxonomy into a comprehensive tree of life. *Proceedings of the National Academy of Sciences*, 112(41), 12764–12769.

Jacobi, C. M., Carmo, F. F. D., Vincent, R. C., & Stehmann, J. R. (2007). Plant communities on ironstone outcrops: A diverse and endangered Brazilian ecosystem. *Biodiversity and Conservation*, 16(7), 2185–2200. https://doi.org/10.1007/s10531-007-9156-8

Jetz, W., Kreft, H., Ceballos, G., & Mutke, J. (2009). Global associations between terrestrial producer and vertebrate consumer diversity. *Proceedings of the Royal Society B: Biological Sciences*, 276(1655), 269–278.

Jetz, W., Thomas, G. H., Joy, J. B., Redding, D. W., Hartmann, K., & Mooers, A. O. (2014). Global distribution and conservation of evolutionary distinctness in birds. *Current Biology*, 24(9), 919–930. https://doi.org/10.1016/j.cub.2014.03.011

Kamino, L. H. Y., Pereira, E. O., & do Carmo, F. F. (2020). Conservation paradox: Large-scale mining waste in protected areas in two global hotspots, southeastern Brazil. *Ambio*, 49, 1–10.

Laubach, T., & Von Haeseler, A. (2007). TreeSnatcher: Coding trees from images. *Bioinformatics*, 23(24), 3384–3385.

Laurance, W. F., & Balmford, A. (2013). A global map for road building. *Nature*, 495(7441), 308–309. https://doi.org/10.1038/495308a

Laurance, W. F., Clements, G. R., Sloan, S., O'connell, C. S., Mueller, N. D., Goosem, M., Venter, O., Edwards, D. P., Phalan, B., Balmford, A., Van Der Ree, R., & Arrea, I. B. (2014). A global strategy for road building. *Nature*, 513(7517), 229–232.

Laurance, W. F., Cochrane, M. A., Bergen, S., Fearnside, P. M., Delamônica, P., Barber, C., D'Angelo, S., & Fernandes, T. (2001). The future of the Brazilian Amazon. *Science*, 291(5503), 438–439.

Laurance, W. F., Goosem, M., & Laurance, S. G. (2009). Impacts of roads and linear clearings on tropical forests. *Trends in Ecology & Evolution*, 24(12), 659–669.

Laurance, W. F., Sloan, S., Weng, L., & Sayer, J. A. (2015). Estimating the environmental costs of Africa's massive "development corridors". *Current Biology*, 25(24), 3202–3208. https://doi.org/10.1016/j.cub.2015.10.046

Lawler, J. J., White, D., Sifneos, J. C., & Master, L. L. (2003). Rare species and the use of indicator groups for conservation planning. *Conservation Biology*, 17(3), 875–882.

Lessmann, J., Fajardo, J., Munoz, J., & Bonaccorso, E. (2016). Large expansion of oil industry in the Ecuadorian Amazon: Biodiversity vulnerability and conservation alternatives. *Ecology and Evolution*, 6(14), 4997–5012.

Lewinsohn, T. M., Agostini, K., Lucci Freitas, A. V., & Melo, A. S. (2022). Insect decline in Brazil: An appraisal of current evidence. *Biology Letters*, 18(8), 20220219.

Luckeneder, S., Giljum, S., Schaffartzik, A., Maus, V., & Tost, M. (2021). Surge in global metal mining threatens vulnerable ecosystems. *Global Environmental Change*, 69, 102303.

Mace, G. M., Gittleman, J. L., & Purvis, A. (2003). Preserving the tree of life. *Science*, 300(5626), 1707–1709.

Marsh, G. P. (1864). Man and nature; or physical geography as modified by human action: Sampson Low, Son and Marston.

Mouillot, D., Bellwood, D. R., Baraloto, C., Chave, J., Galzin, R., Harmelin-Vivien, M., Kulbicki, M., Lavergne, S., Lavorel, S., Mouquet, N., Paine, C. E. T., Renaud, J., & Thuiller, W. (2013). Rare species support vulnerable functions in high-diversity ecosystems. *PLoS Biology*, 11(5), e1001569.

Murguía, D. I., Bringezu, S., & Schaldach, R. (2016). Global direct pressures on biodiversity by large-scale metal mining: Spatial distribution and implications for conservation. *Journal of Environmental Management*, 180, 409–420. https://doi.org/10.1016/j.jenvman.2016.05.040

Oliveira, U., Brescovit, A. D., & Santos, A. J. (2017). Sampling effort and species richness assessment: A case study on Brazilian spiders. *Biodiversity and Conservation*, 26(6), 1481–1493.

Oliveira, U., Paglia, A. P., Brescovit, A. D., de Carvalho, C. J. B., Silva, D. P., Rezende, D. T., Leite, F. S. F., Batista, J. A. N., Barbosa, J. P. P. P., Stehmann, J. R., Ascher, J. S., de Vasconcelos, M. F., Paulo De Marco, P. L.-N., Jr., Dias, P. G., Ferro, V. G., & Santos, A. J. (2016). The strong influence of collection bias on biodiversity knowledge shortfalls of Brazilian terrestrial biodiversity. *Diversity and Distributions*, 22(12), 1232–1244. https://doi.org/10.1111/ddi.12489

Oliveira, U., Soares-Filho, B., Leitão, R. F. M., & Rodrigues, H. O. (2019). BioDinamica: A toolkit for analyses of biodiversity and biogeography on the Dinamica-EGO modelling platform. *PeerJ*, 7, e7213.

Oliveira, U., Soares-Filho, B. S., Paglia, A. P., Brescovit, A. D., de Carvalho, C. J. B., Silva, D. P., Rezende, D. T., Leite, F. S. F., Batista, J. A. N., Barbosa, J. P. P. P., Stehmann, J. R., Ascher, J. S., de Vasconcelos, M. F., de Marco, P., Löwenberg-Neto, P., Ferro, V. G., & Santos, A. J. (2017). Biodiversity conservation gaps in the Brazilian protected areas. *Scientific Reports*, 7(1), 9141. https://doi.org/10.1038/s41598-017-08707-2

Oliveira, U., Soares-Filho, B. S., Santos, A. J., Paglia, A. P., Brescovit, A. D., de Carvalho, C. J. B., Silva, D. P., Rezende, D. T., Leite, F. S. F., Batista, J. A. N., Barbosa, J. P. P. P., Stehmann, J. R.,

Ascher, J. S., Vasconcelos, M. F., Marco, P. D., Löwenberg-Neto, P., & Ferro, V. G. (2019). Modelling highly biodiverse areas in Brazil. *Scientific Reports*, 9(1), 6355. https://doi.org/10.1038/s41598-019-42881-9

Ollerton, J., Winfree, R., & Tarrant, S. (2011). How many flowering plants are pollinated by animals? *Oikos*, 120(3), 321–326. https://doi.org/10.1111/j.1600-0706.2010.18644.x

Owusu, E. H., Ofori, B. Y., & Attuquayefio, D. K. (2018). The secondary impact of mining on primates and other medium to large mammals in forest reserves in southwestern Ghana. *The Extractive Industries and Society*, 5(1), 114–121. https://doi.org/10.1016/j.exis.2017.11.007

Potapov, P., Hansen, M. C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., Smith, W., Zhuravleva, I., Komarova, A., Minnemeyer, S., & Esipova, E. (2017). The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. *Science Advances*, 3(1), e1600821. https://doi.org/10.1126/sciadv.1600821

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: Trends, impacts and drivers. *Trends in Ecology & Evolution*, 25(6), 345–353. https://doi.org/10.1016/j.tree.2010.01.007

Potts, S. G., Imperatriz-Fonseca, V., Ngo, H. T., Aizen, M. A., Biesmeijer, J. C., Breeze, T. D., Dicks, L. V., Garibaldi, L. A., Hill, R., Settele, J., & Vanbergen, A. J. (2016). Safeguarding pollinators and their values to human well-being. *Nature*, 540(7632), 220–229.

Purvis, A., Agapow, P.-M., Gittleman, J. L., & Mace, G. M. (2000). Nonrandom extinction and the loss of evolutionary history. *Science*, 288(5464), 328–330.

R Core Team. (2018). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. Retrieved from R-project.org

Ritter, C. D., McCrate, G., Nilsson, R. H., Fearnside, P. M., Palme, U., & Antonelli, A. (2017). Environmental impact assessment in Brazilian Amazonia: Challenges and prospects to assess biodiversity. *Biological Conservation*, 206, 161–168. https://doi.org/10.1016/j.biocon.2016.12.031

Rodrigues, A. S. L., Brooks, T. M., & Gaston, K. J. (2005). Integrating phylogenetic diversity in the selection of priority areas for conservation: Does it make a difference? In J. G. Purvis & T. Brooks (Eds.), *Phylogeny and conservation* (Vol. 8, pp. 101–119). Cambridge University Press.

Rodrigues, A. S. L., & Gaston, K. J. (2002). Maximising phylogenetic diversity in the selection of networks of conservation areas. *Biological Conservation*, 105(1), 103–111.

Rorato, A. C., Camara, G., Escada, M. I. S., Picoli, M. C., Moreira, T., & Verstegen, J. A. (2020). Brazilian amazon indigenous peoples threatened by mining bill. *Environmental Research Letters*, 15(10), 1040–1043.

Rosauer, D. F., Laffan, S. W., Crisp, M. D., Donnellan, S. C., & Cook, L. G. (2009). Phylogenetic endemism: A new approach for identifying geographical concentrations of evolutionary history. *Molecular Ecology*, 18(19), 4061–4072. https://doi.org/10.1111/j.1365-294X.2009.04311.x

Rosauer, D. F., & Mooers, A. O. (2013). Nurturing the use of evolutionary diversity in nature conservation. *Trends in Ecology & Evolution*, 28(6), 322–323.

Runge, C. A., Tulloch, A. I. T., Gordon, A., & Rhodes, J. R. (2017). Quantifying the conservation gains from shared access to linear infrastructure. *Conservation Biology*, 31(6), 1428–1438. https://doi.org/10.1111/cobi.12952

Salles, D. M., Carmo, F. F. D., & Jacobi, C. M. (2018). Habitat loss challenges the conservation of endemic plants in mining-targeted Brazilian mountains. *Environmental Conservation*, 46(2), 140–146. https://doi.org/10.1017/s0376892918000401

Santos, D. C., Souza-Filho, P. W. M., Rocha Nascimento, W., Cardoso, G. F., & Santos, J. F. (2020). Land cover change, landscape degradation, and restoration along a railway line in the Amazon biome, Brazil. *Land Degradation & Development*, 31(15), 2033–2046. https://doi.org/10.1002/ldr.3514

Seibold, S., Rammer, W., Hothorn, T., Seidl, R., Ulyshen, M. D., Lorz, J., Cadotte, M. W., Lindenmayer, D. B., Adhikari, Y. P., Aragón, R., Bae, S., Baldrian, P., Barimani Varandi, H., Barlow, J., Bässler, C., Beauchêne, J., Berenguer, E., Bergamin, R. S., Birkemoe, T., ... Müller, J. (2021). The contribution of insects to global forest deadwood decomposition. *Nature*, 597(7874), 77–81. https://doi.org/10.1038/s41586-021-03740-8

Seki, H. A., Thorn, J. P., Platts, P. J., Shirima, D. D., Marchant, R. A., Abeid, Y., Baker, N., Annandale, M., & Marshall, A. R. (2022). Indirect impacts of commercial gold mining on adjacent ecosystems. *Biological Conservation*, 275, 109782.

Sgrò, C. M., Lowe, A. J., & Hoffmann, A. A. (2011). Building evolutionary resilience for conserving biodiversity under climate change. *Evolutionary Applications*, 4(2), 326–337.

Siqueira-Gay, J., Soares-Filho, B., Sanchez, L. E., Oviedo, A., & Sonter, L. J. (2020). Proposed legislation to mine Brazil's indigenous lands will threaten Amazon forests and their valuable ecosystem services. *One Earth*, 3(3), 356–362. https://doi.org/10.1016/j.oneear.2020.08.008

Siqueira-Gay, J., Sonter, L. J., & Sánchez, L. E. (2020). Exploring potential impacts of mining on forest loss and fragmentation within a biodiverse region of Brazil's northeastern Amazon. *Resources Policy*, 67, 101662.

Sloan, S., Campbell, M. J., Alamgir, M., Engert, J., Ishida, F. Y., Senn, N., Huther, J., & Laurance, W. F. (2019). Hidden challenges for conservation and development along the trans-Papuan economic corridor. *Environmental Science & Policy*, 92, 98–106. https://doi.org/10.1016/j.envsci.2018.11.011

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*, 285(1892), 20181926. https://doi.org/10.1098/rspb.2018.1926

Sonter, L. J., Barrett, D. J., Moran, C. J., & Soares-Filho, B. S. (2015). A land system science meta-analysis suggests we underestimate intensive land uses in land use change dynamics. *Journal of Land Use Science*, 10(2), 191–204. https://doi.org/10.1080/1747423x.2013.871356

Sonter, L. J., Barrett, D. J., Soares-Filho, B. S., & Moran, C. J. (2014). Global demand for steel drives extensive land-use change in Brazil's iron quadrangle. *Global Environmental Change*, 26, 63–72. https://doi.org/10.1016/j.gloenvcha.2014.03.014

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(1), 1013. https://doi.org/10.1038/s41467-017-00557-w

Sonter, L. J., Lloyd, T. J., Kearney, S. G., Di Marco, M., O'Bryan, C. J., Valenta, R. K., & Watson, J. E. M. (2022). Conservation implications and opportunities of mining activities for terrestrial mammal habitat. *Conservation Science and Practice*, 4, e12806. https://doi.org/10.1111/csp2.12806

Sonter, L. J., Moran, C. J., Barrett, D. J., & Soares-Filho, B. S. (2014). Processes of land use change in mining regions. *Journal of Cleaner Production*, 84, 494–501. https://doi.org/10.1016/j.jclepro.2014.03.084

Sonter, L. J., Simmonds, J. S., Watson, J. E., Jones, J. P., Kiesecker, J. M., Costa, H. M., Bennun, L., Edwards, S., Grantham, H. S., Griffiths, V. F., Jones, K., Sochi, K., Puydarrieux, P., Quétier, F., Rainer, H., Rainey, H., Roe, D., Satar, M., Soares-Filho, B. S., ... Maron, M. (2020). Local conditions and policy design determine whether ecological compensation can achieve No net loss goals. *Nature Communications*, 11(1), 1–11.

Souza-Filho, P. W. M., Nascimento, W. R., Santos, D. C., Weber, E. J., Silva, R. O., & Siqueira, J. O. (2018). A GEOBIA approach for multitemporal land-cover and land-use change analysis in a tropical watershed in the southeastern Amazon. *Remote Sensing*, 10(11), 1683.

Strand, J., Soares-Filho, B., Costa, M. H., Oliveira, U., Ribeiro, S. C., Pires, G. F., Oliveira, A., Rajão, R., May, P., van der Hoff, R., Siikamäki, J., da Motta, R. S., & Toman, M. (2018). Spatially explicit valuation of the Brazilian Amazon forest's ecosystem services. *Nature Sustainability*, *1*(11), 657–664.

Stropp, J., Umbelino, B., Correia, R., Campos-Silva, J., Ladle, R., & Malhado, A. (2020). The ghosts of forests past and future: Deforestation and botanical sampling in the Brazilian Amazon. *Ecography*, *43*(7), 979–989.

Troudet, J., Grandcolas, P., Blin, A., Vignes-Lebbe, R., & Legendre, F. (2017). Taxonomic bias in biodiversity data and societal preferences. *Scientific Reports*, *7*(1), 1–14.

Tucker, C. M., Cadotte, M. W., Carvalho, S. B., Davies, T. J., Ferrier, S., Fritz, S. A., Grenyer, R., Helmus, M. R., Jin, L. S., Mooers, A. O., Pavoine, S., Purschke, O., Redding, D. W., Rosauer, D. F., Winter, M., & Mazel, F. (2017). A guide to phylogenetic metrics for conservation, community ecology and macroecology. *Biological Reviews*, *92*(2), 698–715.

Valsecchi, J., Marmontel, M., Franco, C. L. B., Cavalcante, D. P., Cobra, I. V. D., Lima, I. J., Lanna, J. M., Ferreira, M. T. M., Nassar, P. M., Botero-Arias, R., & Monteiro, V. (2017). Update and compilation of the list untold treasures: New species discoveries in the Amazon 2014-15. WWF Living Amazon Initiative, WWF-Brazil, and Mamiraua Institute of Sustainable Development.

Vane-Wright, R. I., Humphries, C. J., & Williams, P. H. (1991). What to protect?—Systematics and the agony of choice. *Biological Conservation*, *55*(3), 235–254.

Villén-Pérez, S., Anaya-Valenzuela, L., da Cruz, D. C., & Fearnside, P. M. (2022). Mining threatens isolated indigenous peoples in the Brazilian Amazon. *Global Environmental Change*, *72*, 102398.

Villén-Pérez, S., Mendes, P., Nóbrega, C., Gomes Córtes, L., & De Marco, P. (2018). Mining code changes undermine biodiversity conservation in Brazil. *Environmental Conservation*, *45*(1), 96–99. https://doi.org/10.1017/S0376892917000376

Warren, R., Price, J., Graham, E., Forstenhaeusler, N., & VanDerWal, J. (2018). The projected effect on insects, vertebrates, and plants of limiting global warming to 1.5°C rather than 2°C. *Science*, *360*(6390), 791–795. https://doi.org/10.1126/science.aar3646

Weng, L., Boedhihartono, A. K., Dirks, P. H. G. M., Dixon, J., Lubis, M. I., & Sayer, J. A. (2013). Mineral industries, growth corridors and agricultural development in Africa. *Global Food Security*, *2*(3), 195–202. https://doi.org/10.1016/j.gfs.2013.07.003

Werner, T. T., Bebbington, A., & Gregory, G. (2019). Assessing impacts of mining: Recent contributions from GIS and remote sensing. *The Extractive Industries and Society*, *6*(3), 993–1012. https://doi.org/10.1016/j.exis.2019.06.011

Wickham, H., & Seidel, D. (2020). Scales: Scale functions for visualization. *R package version*, *1.1*, 678.

Williams, P. H., & Humphries, C. J. (1994). Biodiversity, taxonomic relatedness, and endemism in conservation. In P. L. Forey, C. J. Humphries, & R. I. Vane-Wright (Eds.), *Systematics and conservation evaluation* (pp. 269–287). Oxford University Press.

Wilson, E. O. (1987). The little things that run the world (the importance and conservation of invertebrates). *Conservation Biology*, *1*(4), 344–346.

**BIOSKETCH**

Thomas Lloyd is a PhD candidate at The University of Queensland interested in conservation biogeography and planning. His current research focusses on the unique land-use impacts associated with mining and infrastructure on biodiversity, and the safeguards intended to protect it.

Author contributions: TL, UO, BS-F, and LS designed the study. JA - MV contributed to data collection and collation. Analyses were conducted by TL and LS. TL wrote the manuscript with guidance from RF, NB, and LS. All authors read, provided comments on, and approved the manuscript for submission.

## SUPPORTING INFORMATION

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