Research Square

Preprints are preliminary reports that have not undergone peer review.
They should not be considered conclusive, used to inform clinical practice,
or referenced by the media as validated information.

How representative is flora rescue? A 15-year analysis of species composition in iron ore mining in the Brazilian Amazon and the gap for endemic and rare species conservation

Fernando M. G. Santos[1]

fernandomarinog@gmail.com

Universidade Federal de Minas Gerais

Fernando A. O. Silveira[1]

Universidade Federal de Minas Gerais

Research Article

Keywords: Amazonian canga, Biodiversity monitoring, Compliance trap, Environmental licensing, Campo rupestre ferruginous, Mitigation translocation

Posted Date: November 19th, 2026

DOI: https://doi.org/10.21203/rs.3.rs-11580645/v1

License: This work is licensed under a Creative Commons Attribution 4.0 International License.
Read Full License

Additional Declarations: No competing interests reported.

How representative is flora rescue? A 15-year analysis of species composition in iron ore mining in the Brazilian Amazon and the gap for endemic and rare species conservation

Authors: Fernando M. G. Santos[1]* and Fernando A. O. Silveira[1]

ORCID: F.M.G. Santos https://orcid.org/0000-0002-8458-735X; F.A.O. Silveira https://orcid.org/0000-0001-9700-7521

[1]Center for Ecological Synthesis and Conservation, Universidade Federal de Minas Gerais, Av. Antônio Carlos 6627, Belo Horizonte, Minas Gerais, 31270-901, Brazil.

Corresponding Author: *Fernando Marino Gomes dos Santos

Email: fernandomarinog@gmail.com

Address: Center for Ecological Synthesis and Conservation, Universidade Federal de Minas Gerais, Av. Antônio Carlos 6627, Belo Horizonte, Minas Gerais, 31270-901, Brazil.

Acknowledgements We thank the Coordination for the Improvement of Higher Education Personnel (CAPES) for the scholarships granted to the first author. We are deeply grateful to Vale S.A. for providing the access and support with the RADA reports. We also thank Tais N. Fernandes, Cecilio F. Caldeira, Danielle Fernandes, Breno Chaves, Flavio Gontijo and Lourival Tyski for support during data curation, as well as for valuable suggestions and comments, which greatly contributed to the improvement of this manuscript. We thank comments by M Gastauer, R Lemes and JCF Cardoso which increased overall manuscript quality.

Abstract

Mitigation-driven plant translocations are widely used during environmental licensing, yet their effectiveness in protecting biodiversity remains uncertain. We evaluated a large-scale flora rescue and reintroduction program associated with iron ore and manganese mining on Amazonian rock outcrops (*canga*), Carajás, Brazil. Using data from the Annual Environmental Performance Reports (RADA) from 2010–2024, we assessed taxonomic uncertainty, species, functional and phylogenetic diversity, habit-based representation, species abundance and temporal trends relative to the reference *canga* flora. The database comprised 351,594 rescue records assigned to 347 registered taxonomic names, of which 277 (79.9%) were retained as accepted names. The validated assemblage shared 107 species with the 1,045-species reference flora (10.24% coverage; Jaccard = 0.0881). Restricting the reference flora to the 429 species documented for the two mined plateaus raised coverage to 14.5% but similarity was nearly unchanged (Jaccard = 0.0963). Among the 99 shared seed-plant species, coverage-standardized richness did not differ from the reference (95.7 vs 99.0 species), and null models showed that registered assemblages held less phylogenetic diversity than expected from their richness, which reflects rescue concentration in a few predominantly epiphytic clades. Abundance was highly uneven, with 37 species accounting for ~90% of validated individuals. Rescue yield per suppressed hectare increased over time, and was decoupled from vegetation suppressed area. Because the records do not support inference on post-translocation persistence, these results characterize representativeness rather than conservation outcomes. Embedding reference-based targets and independent verification of post-translocation performance into licensing conditions would allow programs to demonstrate ecological rather than documentary performance.

Keywords: Amazonian *canga*; Biodiversity monitoring; Compliance trap; Environmental licensing; *Campo rupestre* ferruginous; Mitigation translocation

Introduction

The global demand for minerals is driving the expansion of mining into megadiverse tropical regions and intensifying conflicts between economic development and biodiversity conservation (Sonter et al. 2018, 2023; Torres et al. 2024). This conflict is particularly evident in Amazonian ironstone outcrops, or *cangas*, where the geological features that sustain highly endemic biota also make these landscapes prime targets for mineral extraction and place irreplaceable biodiversity at risk (Mota et al. 2018; Giulietti et al. 2019; Souza-Filho et al. 2019). These ancient, montane and nutrient-impoverished ecosystems are characterized by high endemism, strong edaphic filtering and ecological specialization shaped by long-term environmental stability, substrate heterogeneity, metal-rich soils, water limitation and low nutrient availability (Zappi et al. 2019; Mota et al. 2018; Arruda et al. 2023).

To address the impacts associated with mineral extraction, environmental assessment and licensing systems aim to prevent, reduce, and manage environmental damage through legally enforceable mitigation measures (Sánchez and Duarte 2022). A recurring concern in impact assessment, however, is the gap between procedural compliance and effective ecological outcomes, particularly when monitoring programs prioritize auditable outputs rather than biologically meaningful biodiversity endpoints (da Silva Dias et al. 2017; Mallett et al. 2021; Serrano and Fonseca 2024). In parallel, regulatory frameworks worldwide have increasingly incorporated biodiversity offsetting policies intended to achieve no net loss, or in some cases net gain of biodiversity (Bull et al. 2013; Gardner et al. 2013; Maron et al. 2016). More recent approaches emphasize ecological equivalence and measurable

biodiversity outcomes and shift the focus from procedural actions to verifiable conservation performance (Bezombes et al. 2017, Borges-Matos et al. 2023).

In this regulatory context, the mitigation hierarchy provides the conceptual basis for environmental licensing, requiring that mitigation and compensation measures address the critical biodiversity attributes affected by development (Arlidge et al. 2018; IFC 2012 - Performance Standard 6). Plant rescue programs are positioned in this hierarchy as corrective measures intended to compensate for unavoidable impacts, with their effectiveness linked to ecological restoration outcomes and conservation priorities (Skirycz et al. 2014; Fernandes et al. 2022). Under this framework, mitigation translocation has been a strategy to compensate for habitat loss associated with activities such as mining and dam construction by establishing new populations in protected areas or areas not directly affected by the original enterprise (IUCN/SSC 2013; Germano et al. 2015; Julien et al. 2022; Doyle et al. 2023).

Biodiversity monitoring programs are expected to be grounded on standardized data collection protocols, robust sampling designs, and sufficient temporal and spatial replication to detect meaningful ecological changes (da Silva Dias et al. 2017). When these elements are absent or lacking, monitoring data has limited relevance for decision-making. In contrast, flora rescue programs are typically implemented under project-driven timelines, with limited ecological planning and incomplete knowledge of species ecology and supported by regulatory frameworks that lack clear metrics to assess post-translocation performance (Gardner and Howarth 2009; Albrecht et al. 2019; Doyle et al. 2023; Julien et al. 2023a). As a result, while monitoring aims to detect long-term ecological change, biodiversity rescue tend to prioritize short-term compliance, which constrains their effectiveness in mitigating biodiversity loss at the population level (Germano et al. 2015; Bradley et al. 2022; Julien et al. 2023b).

Under compliance-driven implementation and reporting, taxonomic uncertainty becomes a consequential source of error in species-rich tropical monitoring datasets, because field identifications may be uncertain, informal names may circulate in corporate records, and habitat misclassification may distort baselines and perceived outcomes (Scott and Hallam 2002; Bortolus 2008; Meyer et al. 2026). Such uncertainty can inflate apparent species richness, undermine assessments of representativeness, priority-taxon coverage and functional completeness, and constrain conservation decision-making (Murray et al. 2017). These issues are particularly relevant in *canga* landscapes, where floristic composition can change over short spatial scales and where rare, cryptic, or difficult to detect edaphic endemics may go undetected requiring targeted sampling and taxonomic expertise (Viana et al. 2016; Mota et al. 2018; Giulietti et al. 2019). Together with structural limitations in post-translocation monitoring that prevents inference about survival and establishment (Godefroid et al. 2011; IUCN/SSC 2013), these constraints motivate an evaluation focused on what is being rescued and translocated, and how this assemblage aligns with the target ecosystem.

The notion of a compliance trap originates in regulatory theory, where formal adherence to rules becomes an end in itself and displaces the substantive goals the rules were meant to serve (Parker 2006). In environmental governance, this trap arises when licensing and monitoring reward auditable outputs, such as documents delivered, individuals rescued or actions reported, rather than verified ecological outcomes, so that meeting the requirement is mistaken for achieving conservation (Thompson and Thompson 2023; Stoffels and Thompson 2026), resulting in a translocation debt that can be defined as the shortfall between the composition of the translocated assemblage and that of the flora it is meant to mitigate. This distinction is well illustrated by the difference between monitoring that merely documents whether an action was implemented and monitoring that demonstrates whether it achieved its ecological objective (Stoffels and Thompson 2026). Left unexamined, such

arrangements can settle into a self-reinforcing state, a social-ecological trap, in which documented effort persistently substitutes for measurable biodiversity gains (Tanguay et al. 2025).

We critically analyzed 15 years of data from a long-term flora rescue program associated with iron ore and manganese mining in the Carajás Mining Complex to assess whether documented mitigation effort translated into ecological representativeness. Our aim was to evaluate how the rescued assemblage aligned with the known reference *canga* flora using taxonomic validation, reference-based compositional metrics, abundance structure, habit-based functional representation, priority-species coverage, sample-based rarefaction and Faith's phylogenetic diversity. Because the monitoring structure of the program does not allow species-level inference on post-translocation survival, recruitment or population persistence, our analysis focused on the auditable dimensions consistently available across years. Specifically, we predicted (i) that taxonomic uncertainty affects representativeness estimates; (ii) that compositional overlap with the reference *canga* flora is low, given that the reference list includes many taxa that were never present in the concession areas; (iii) that rescued individuals and species are concentrated in a restricted subset of families, habits and abundant taxa; and (iv) that temporal increases in taxonomic and phylogenetic diversity indicate broader ecological representation.

Material and Methods

Study area and conservation context

The Carajás Mining Complex is located in southeastern Pará, Brazil, within the Brazilian Amazon biome. The region comprises a vegetation mosaic dominated by dense ombrophilous forest interspersed with ironstone outcrops (*canga*) and mining areas. Ironstone substrates support *canga* vegetation that is floristically distinct from the lowland rainforest, including 38 edaphic endemics restricted to ironstone substrates (Giulietti et al. 2019). Some

of these edaphic endemics have extremely restricted distributions which are confined to a few plateaus in the northern range within the study area (Viana et al. 2016). The northern ironstone outcrops also host the Carajás Mining Complex, which implements a suite of environmental programs aimed at mitigating impacts on Amazonian ironstone ecosystems.

Data sources and compilation

We compiled corporate records from Annual Environmental Performance Reports (RADAs – Relatório Anual de Desempenho Ambiental) covering 2010–2024. These reports document flora rescue activities, defined here as plant removal during vegetation suppression, and reintroduction actions, defined as outplanting to recipient areas within the Carajás Mining Complex. Each record included the activity date, operation identifier, action type (rescue or reintroduction), habitat category, taxonomic identification, life form, and individual count. For manganese operations, RADAs from 2010–2012 were unavailable, whereas reports from 2022–2024 indicated no rescue activities because no vegetation suppression occurred during those years. Together, these reports constitute the operational record of the RADA rescue program and provide the only continuous source for tracking rescued and translocated individuals across years. This study analyzed previously compiled corporate records; no plant collections were made by the authors. Voucher specimens associated with the RADA program flora surveys are deposited at the HCJS herbarium (Carajás, Brazil).

Taxonomic standardization and data quality screening

All species records were standardized to accepted scientific names and checked against Flora e Funga do Brasil via REFLORA (http://floradobrasil.jbrj.gov.br/; BFG 2021a, 2021b). Each record was classified as: (i) accepted, when identified to species level and

matching an accepted REFLORA name; (ii) non-valid, when flagged with qualifiers such as cf. or aff., identified as errors, or not registered for Brazil; and (iii) missing, when identified only to genus level, assigned as morphospecies, or otherwise indeterminate. One taxon recorded as Bromelia aff. lagopus was treated as accepted because a specialist confirmed it as a new species currently being described. Only accepted records were included in species-level representativeness analyses. Non-valid and missing records were excluded from species-level comparisons but retained to characterize data quality and potential sources of bias (Online Resource 1). This threshold follows recommended practice in biodiversity assessments by restricting similarity analyses to taxonomic identifications that are verifiable and directly comparable with the reference flora.

Annual summaries were produced by stratum and event type, and we distinguished true zeros, defined as documented reporting with no events, from missing data, defined as cases in which species-level information was unavailable. Years or categories lacking species-level reintroduction data were excluded from trend interpretation. A preliminary screening identified an outlier in the 2010 data with unusually high recorded abundance for *Sobralia liliastrum*, *Epidendrum purpurascens*, and *Pitcairnia caricifolia*, likely to reflect overestimation of vegetative structures during counting. These records were retained but flagged as potential errors. Because abundance was not used in program–reference similarity analyses, these flagged records did not affect the presence–absence-based representativeness metrics. From the species-by-year matrices, we computed cumulative annual richness and the number of new species per year (species whose first non-zero record occurred in that year), coding true zeros as zero and excluding year × category combinations lacking species-level data from trend interpretation. Absence of a rescue record was not interpreted as absence of the species. A species may be unrecorded because it was never exposed to suppression, because it was present but not detected, because it was genuinely absent from the cleared

area, or because a documented population had already been extirpated locally before suppression. Operational records do not distinguish among these, and coverage metrics therefore describe documented representation relative to the reference pool rather than detection rates.

Analytical framework

Our analytical framework was designed to evaluate the auditable dimensions of long-term flora rescue records. We therefore focused on four complementary dimensions: (1) taxonomic review of species records from the flora rescue program; (2) the representativeness of rescued and translocated species relative to the reference *canga* flora; (3) the concentration of records among taxonomic groups and growth forms; and (4) the representation of endemic and threatened species.

The Flora de Carajás was used as the best available reference for *canga* vegetation (Mota et al. 2018; Salino et al. 2018). Because the regional checklist includes species that were never present in the cleared areas, the regional pool is treated as an upper bound on what could have been rescued, and the plateau-restricted pools as progressively more realistic denominators. Coverage and similarity were therefore computed at three nested spatial scales: the regional canga flora, the Serra Norte landscape (plateaus N1–N8 and Serra Norte records without plateau assignment), and the mined plateaus from which the suppression data originate (N4 and N5). Species documented for Serra Norte but absent from N4 and N5 could not have been rescued there, and the three-scale comparison isolates this effect. Plateau-level occurrences were taken from the area records of Mota et al. (2018) for seed plants and from Salino et al. (2018) for ferns and lycophytes.

Taxonomic representativeness analyses

Because the reference flora is available as a reference checklist and area-based occurrence dataset, comparisons between RADA and the reference flora were restricted to presence–absence matrix. Abundance values in the RADA database represent operational counts of rescued or translocated individuals and are influenced by detectability, rescue feasibility, handling, nursery processing and destination allocation. They were therefore used only to describe operational effort within the program, not to estimate ecosystem-level compositional similarity.

For each RADA rescued species set, we calculated reference-flora coverage as:

$$Coverage = \frac{| S_{program} \cap S_{ref} |}{| S_{ref} |}$$

where S_program is the RADA rescued species set and S_ref is the reference flora. We also calculated Jaccard and Sørensen similarity indices as binary presence–absence metrics to quantify compositional overlap between the RADA rescued assemblage and the reference *canga* flora (Oksanen et al. 2022). These coverage and similarity metrics were interpreted as descriptive benchmarks against the regional reference pool and against the plateau-restricted pools (Serra Norte and N4–N5), not as expectations of full representation. Because vegetation clearing is spatially restricted to licensed areas within a few plateaus, only the subset of the reference flora that actually occurs in the cleared sites could ever be exposed to rescue year-by-year. The reference pool is thus an upper bound for contextualizing what the mitigation instrument documents, and the eligible-pool sensitivity analyses described below progressively restrict the comparison to more realistic denominators.

Rarefaction and phylogenetic analyses

To account for unequal sampling effort among years, operational groups and between the RADA rescued and reference-flora datasets, we applied sample-based rarefaction

analyses to species-by-sample presence–absence matrices. Rarefaction was implemented using random resampling without replacement, following the general reasoning of sample-based biodiversity standardization for incomplete and unevenly sampled assemblages (Gotelli and Colwell 2001; Chao et al. 2006).

For the RADA dataset, sampling units corresponded to operational records aggregated by year, mining group and locality. For the reference flora, sampling units corresponded to the distinct *canga* areas represented in the regional floristic database. Taxonomic rarefaction was calculated as sample-based gamma richness, defined as the number of unique species accumulated across randomly selected sampling units. For the full RADA dataset, rarefaction was performed using 277 validated species distributed across 164 samples.

A species-level phylogeny was generated for the validated RADA species set using V.PhyloMaker2 and the GBOTB.extended.LCVP megaphylogeny (Jin and Qian 2022). The resulting tree was pruned to match the species present in each analytical matrix using the *ape* package (Paradis and Schliep 2019). Species lacking in the megaphylogeny were added to the base of the genus as polytomies. Phylogenetic diversity was quantified using Faith's phylogenetic diversity, defined as the sum of branch lengths connecting all species present in an assemblage (Faith 1992). Faith's PD was calculated for individual samples, pooled assemblages and rarefied assemblages using the same resampling structure applied to taxonomic richness.

To evaluate whether phylogenetic diversity represents an independent dimension of diversity or primarily tracked species richness, we tested the relationship between taxonomic richness and Faith's PD using Spearman rank correlation and a linear model of Faith's PD as a function of richness. To test whether assemblages captured more or less phylogenetic diversity than expected given their species richness, we additionally computed the standardized effect size of Faith's PD (SES-PD) using the taxa.labels null model, which

randomizes phylogeny tip labels while holding sample richness constant (999 randomizations; picante v1.8.2; Kembel et al. 2010). SES-PD was calculated for each operational sample and for pooled annual assemblages (year × mining complex), with significance assessed against the two-tailed 95% null envelope.

Temporal and statistical analyses

From the species-by-year rescue matrices, we computed cumulative annual species richness and the number of new species rescued per year, defined as species whose first non-zero record occurred in a given year. Destination strata for the transplantation individuals as described in the reports followed the operational categories used in the program: canga, forest and ex situ. True zeros, corresponding to documented years without events, were coded as zero, whereas years or strata with missing species-level information were excluded from trend interpretation.

Temporal trends in annual richness and new species per year were tested using Kendall's rank correlation and negative binomial generalized linear models, with year as a continuous predictor and results reported as incidence rate ratios, 95% confidence intervals and P values (Kendall 1938; Venables and Ripley 2002). Temporal trends in rarefied gamma richness and rarefied Faith's PD were tested separately for the iron and manganese mining complexes using Spearman rank correlations between year and the mean rarefied metric (Spearman 1904). As an additional sensitivity analysis addressing unequal clearing effort, the annual suppressed area (ha) was compiled from the original RADA reports (12 of 15 years; area data are absent from the 2012, 2013 and 2017 reports) and related to annual registered diversity using Spearman correlations, and to rescue yield per hectare using a negative-

297 binomial GLM of observed annual richness with log(area) as an offset; residual adequacy
298 was checked by simulation using the DHARMa package (Hartig 2022).
299 We conducted a sensitivity analysis to assess the robustness of coverage and binary
300 similarity metrics across alternative taxonomic filtering scenarios, following
301 recommendations for biodiversity assessments affected by incomplete or uncertain species
302 identifications (Bortolus 2008; Murray et al. 2017). Analyses were performed in R 4.6.1 (R
303 Core Team 2023). Community dissimilarities were calculated using vegan (Oksanen et al.
304 2022), phylogenetic trees were manipulated using ape (Paradis and Schliep 2019), null-model
305 phylogenetic analyses were run in picante (Kembel et al. 2010), coverage-based
306 standardization in iNEXT (Hsieh et al. 2016), and negative-binomial models in MASS
307 (Venables and Ripley 2002).
308
309 *Coverage-standardized comparison between RADA and the reference flora list*
310 For the comparative analysis between RADA and the reference canga flora, we
311 reconstructed a species-level area-by-species matrix from the reference flora and restricted
312 both datasets to species shared by RADA and the reference flora. This shared-species
313 analysis included species present in both datasets and aligned with the phylogeny. Because
314 area-by-species matrices were available only for seed plants (Mota et al. 2018), this analysis
315 was restricted to angiosperms.
316 Total gamma richness and total Faith's PD were first calculated for the pooled
317 assemblages; we then rarefied both datasets to 14 sampling units (the number of reference-
318 flora areas containing shared species) to equalize sampling effort. Because RADA and the
319 reference flora have different sampling structures (temporal or locality-based operational
320 rescued records in RADA versus floristic areas in the reference flora), we standardized the
321 comparative analysis by sample coverage rather than by number of sampling units (Chao and

Jost 2012), using incidence data in iNEXT v3.0.2 (Hsieh et al. 2016) with 200 bootstrap replicates and comparing both datasets at the lowest observed coverage. A classical rarefaction of both datasets to a common number of 10 sampling units was retained as a complementary check. These contrasts were interpreted as descriptive measures of representativeness contrast between databases, not a strict equivalence of sampling design. As sensitivity analyses addressing the spatial scope of the comparison, we recalculated registered species coverage and Jaccard and Sørensen similarity after restricting the reference flora to the Serra Norte landscape (plateaus N1–N8 plus unassigned Serra Norte records) and to the mined plateaus only (N4 and N5).

Temporal trends within the RADA dataset were also evaluated for the full species set and for the shared-species subset. For these analyses, rarefied gamma richness and rarefied gamma Faith's PD were calculated for each year × group combination. Only combinations with at least three samples were retained for temporal inference, and all valid combinations were standardized to three samples, corresponding to the minimum sampling effort available across valid year × group combinations.

Two samples with zero recorded species were retained in the main analyses; excluding them changed rarefied estimates by at most 3.7 species and 188 PD units and did not affect temporal trends (Online Resource 2).

Functional representativeness

Functional representativeness was assessed using growth form or habit as the only trait consistently available in our database. Growth-form categories included tree, shrub, herb, vine/climber and epiphyte/other. Because only a coarse categorical trait was available, these results were interpreted as habit-based functional representativeness rather than as comprehensive multidimensional functional diversity.

We did not use Rao's quadratic entropy or other distance-based functional-diversity metrics as primary outputs because the available trait information did not support a robust multidimensional species-by-trait distance matrix (Rao 1982). Functional patterns were therefore assessed descriptively through the proportional representation of growth forms and used to identify operational concentration or underrepresentation among habits.

Conservation priority species

Threatened species were identified using three sources: Brazil (Portaria MMA nº 148/2022, updating Portaria nº 443/2014), Pará State (Resolução COEMA nº 54/2007), and the IUCN Red List. Only taxa classified as Critically Endangered (CR), Endangered (EN), or Vulnerable (VU) were treated as threatened. Threatened-species representativeness was evaluated by restricting both the reference and program species sets to threatened taxa and recalculating coverage and binary similarity metrics within that subset.

Edaphic endemics are species growing exclusively on ironstone (*canga*) substrates and absent from the surrounding habitats and soils (Giulietti et al. 2019). They were analyzed as a priority subset by reporting their occurrence in rescue records in the RADA reports and their proportional representation relative to the endemic pool in the reference flora (Online Resource 3). To test whether edaphic endemics were disproportionately represented in the program, we built a 2 × 2 contingency table crossing rescue status with endemism status within the reference flora and applied a chi-squared test of independence (Cressie and Read 1984).

Artificial intelligence tools assisted in organizing the RADA dataset, extracting information from PDF reports, correcting figure scripts and legends, and reviewing the manuscript language. All content was verified and approved by the authors.

Results

Taxonomic quality and data validation

The RADA dataset initially comprised 351,594 individuals or translocation records assigned to 347 operational taxonomic names. After taxonomic standardization against Flora e Funga do Brasil, 277 species distributed across 48 families, including 269 angiosperms and eight ferns or lycophytes (Fig. 1, Online Resource 1). Seventy (20.2%) taxa were excluded because they corresponded to unresolved identifications, non-valid names, morphospecies, qualifiers such as cf. or aff., or names not confirmed for Brazil. Coverage and similarity metrics were robust to the taxonomic filtering, where 14 species-level names not validated against REFLORA (11 flagged with qualifiers or identification errors and three not reported for Brazil) have no occurrence in the reference flora, so their inclusion left the shared-species count unchanged (107) and moved the similarity indices only marginally (coverage 10.24% in both scenarios; Jaccard 0.0881 versus 0.0871; Sørensen 0.1619 versus 0.1602; Online Resource 2).

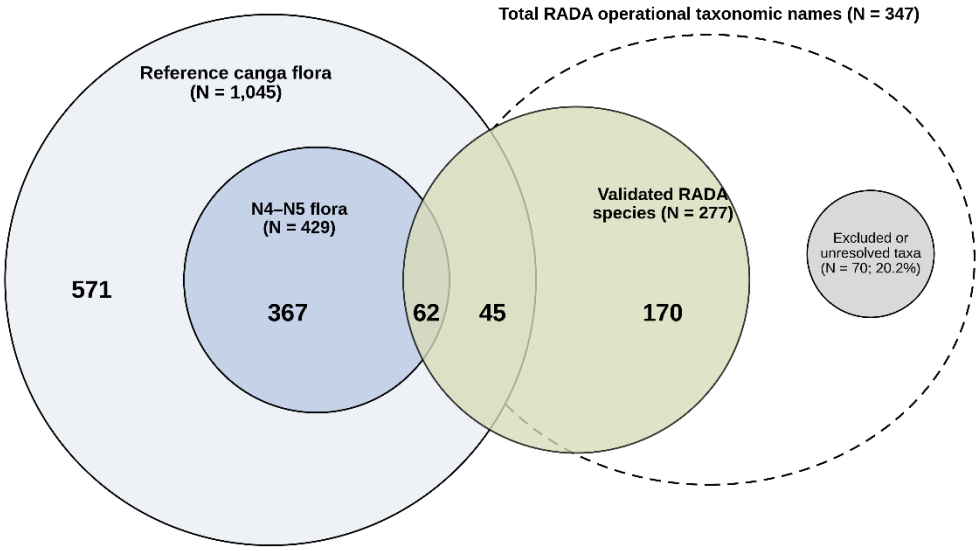


Fig. 1 Taxonomic overlap and identification in a long-term flora rescue program in the eastern Amazon. Nested diagram of the reference canga flora of Carajás (seed plants, Mota et al. 2018; ferns and lycophytes, Salino et al. 2018), the subset documented for the mined plateaus N4 and N5, and the taxonomically validated RADA species set (accepted names). Numbers are species counts in each region; region sizes are schematic. RADA species shared with the reference flora outside the N4–N5 circle include taxa without plateau-level records in the reference sources. The dashed ellipse encloses all RADA operational taxonomic names.

The validated RADA dataset shared 107 of its 277 species with the 1,045-species reference *canga* flora, whereas 170 validated species were absent from the reference list. This surplus is expected, because licensed targeted areas extend beyond *canga*, and 71% of all rescued individuals originated from *canga* vegetation while 24% came from surrounding ombrophilous rainforest, so a substantial share of the rescued pool comprises forest species. The reference flora, in turn, contained 938 species with no rescue or reintroduction record. The combined dataset therefore covered 10.24% of the reference canga flora, with low compositional similarity (Jaccard = 0.0881; Sørensen = 0.1619). Rescue records alone

included 275 species (106 shared; 10.14% coverage; Jaccard = 0.0873; Sørensen = 0.1606), and reintroduction records included 183 species (80 shared; 7.66% coverage; Jaccard = 0.0697; Sørensen = 0.1303).

The translocation debt persisted when the reference pool was restricted spatially (Fig. 1). The flora documented for the mined plateaus from which the suppression data originate (N4 and N5) comprised 429 species, of which 62 were recorded by the program (14.5% coverage; Jaccard = 0.0963; Sørensen = 0.1756). Restricting the reference pool therefore raised coverage modestly while leaving compositional similarity practically unchanged, and 86% of the species documented for N4 and N5 had no rescue record (Online Resource 2).

Family-level comparisons reinforced the translocation debt between RADA and the reference *canga* flora, where this comparison was restricted to validated RADA species documented rather than to the full rescued species set. In the reference flora, Poaceae (87 species), Fabaceae (65), Rubiaceae (46), Cyperaceae (45) and Melastomataceae (42) were the richest families, but their representation in the RADA rescue flora subset was limited. By contrast, Orchidaceae remained prominent in rescue records, with 14 rescued species compared with 26 in the reference flora (Fig. 2).

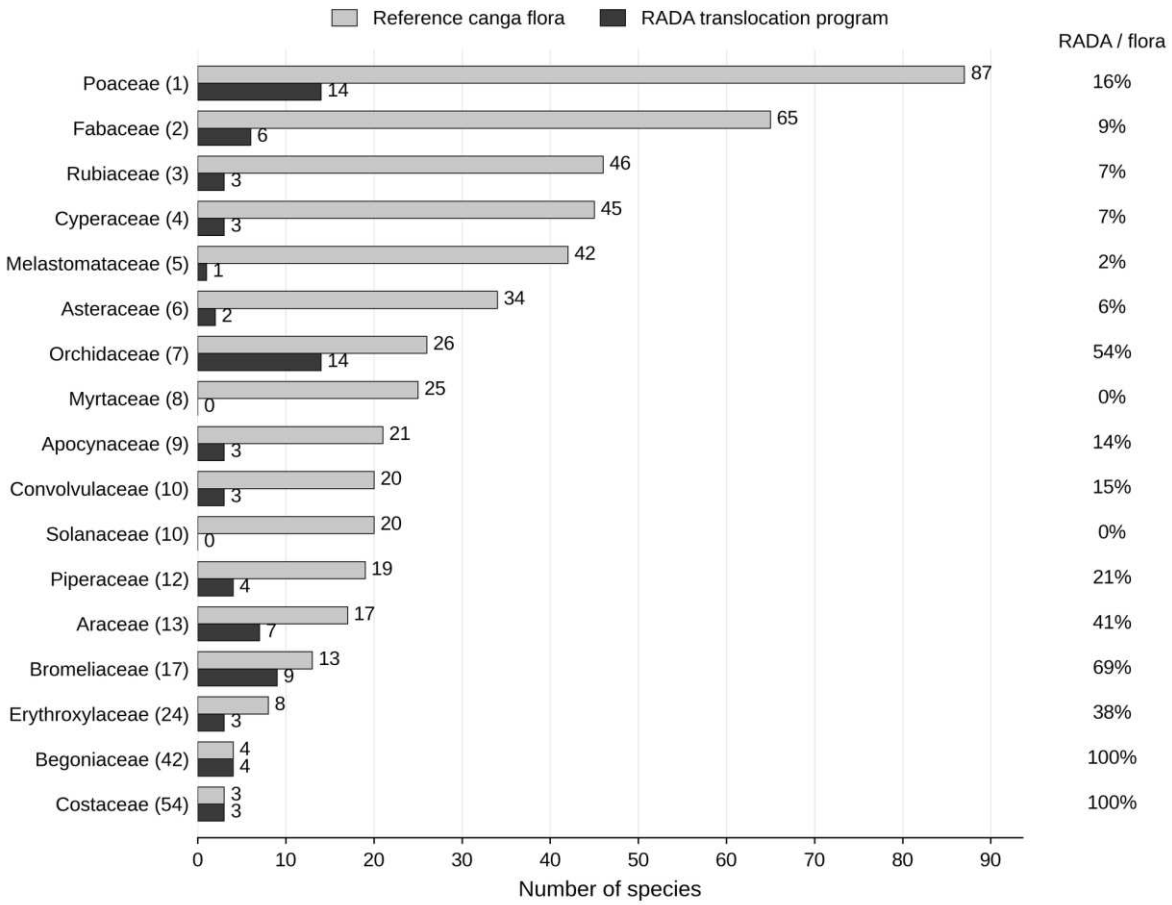


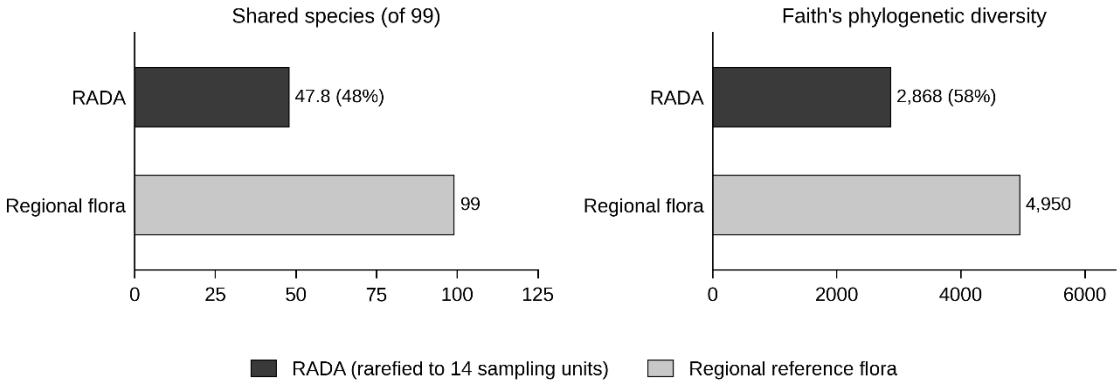
Fig. 2 Family-level richness in the reference canga flora of Carajás and in the RADA translocation program, eastern Amazon, 2010–2024. Bars show the number of species per family in the reference flora (light bars) and in the subset of taxonomically validated RADA species documented in the Flora de Carajás (dark bars), on a single shared scale. The number in parentheses after each family name is its richness rank within the reference flora; the right-hand column gives the percentage of the family's reference richness represented in the program.

Rarefaction and phylogenetic analyses

RADA and the reference flora shared 99 seed-plant species, all present in the phylogeny (Faith's PD = 4,950). Rarefied to the 14 reference-flora areas, RADA recovered 47.8 of these species (95% interval 32–72) and 58% of their Faith's PD (Fig. 3). Sample

435 coverage was 0.990 for RADA and 0.986 for the reference flora, and at equal coverage
436 RADA accumulated 95.7 shared species (95% CI 88.4–103.1) against 99.0 (90.7–107.3) in
437 the reference flora (Fig. 4). Shared species therefore accumulated slowly across operational
438 records but were almost completely sampled, so the compositional gap lies in the reference
439 species never recorded rather than in incomplete sampling of the shared pool. Temporal
440 trends differed by mining type, since iron-ore operations showed significant increases in
441 standardized richness and Faith's PD over time while manganese operations showed no
442 significant trends.

443 Faith's PD tracked richness closely across samples (Spearman ρ = 0.944). Once
444 richness was controlled through null models, however, rescued assemblages held
445 consistently less phylogenetic diversity than expected, with a mean sample-level SES-PD
446 of −1.22; one third of the samples and half of the annual assemblages were significantly
447 clustered, and none were overdispersed (Online Resource 4). Phylogenetic accumulation
448 in RADA therefore reflected species accumulation concentrated in a few related,
449 predominantly epiphytic clades rather than phylogenetically representative coverage.
450



451

452 **Fig. 3** RADA recovery of shared species and Faith's phylogenetic diversity relative to the
453 reference canga flora, for the 99 seed-plant species recorded in both datasets. Dark bars,

RADA rarefied to 14 sampling units (mean of 999 resamplings without replacement); light bars, reference flora (14 areas). Values in parentheses give the percentage of the reference value.

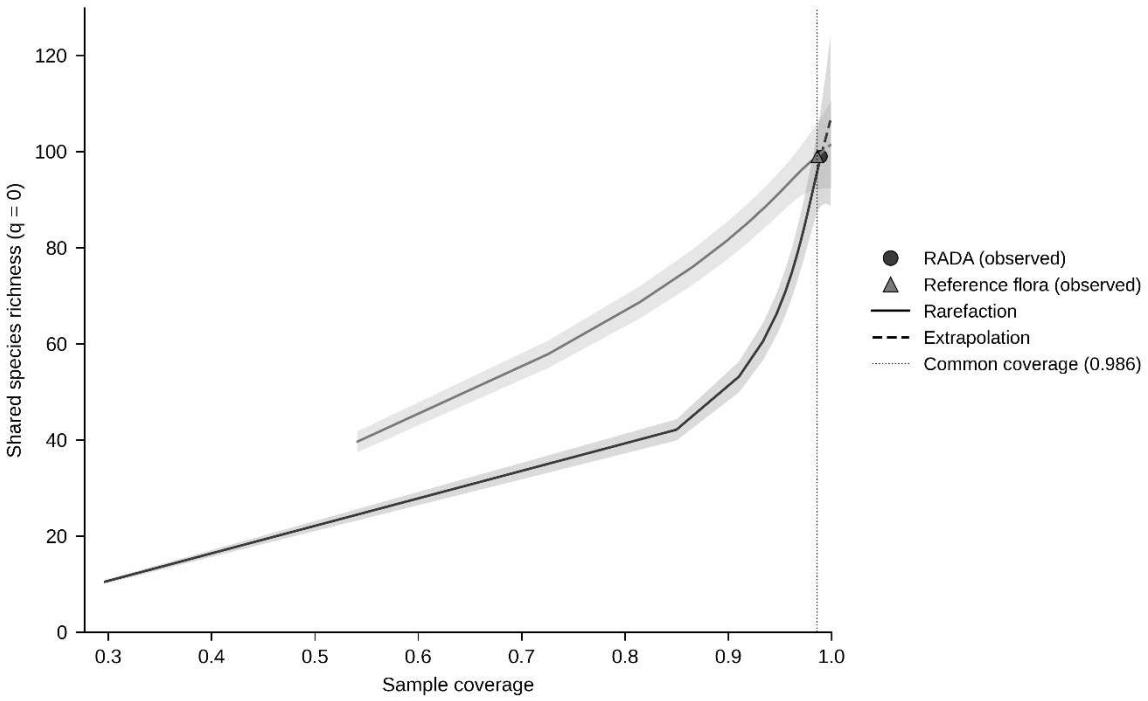


Fig. 4 Coverage-based rarefaction (solid lines) and extrapolation (dashed lines) of the 99 seed-plant species shared between RADA and the reference flora, from incidence data (200 bootstrap replicates). Curves show species richness as a function of sample coverage; symbols mark observed values, shaded areas indicate 95% confidence intervals and the dotted line marks the common coverage (0.986).

Temporal trends

The annual richness in the taxonomically validated RADA dataset varied markedly among years, whereas cumulative richness increased steadily across the study period. Annual richness ranged from 69 species in 2012 to 119 in 2019, with 107 species recorded in 2024,

and cumulative richness reached 277 validated species by 2024 (Online Resource 1). The cumulative number of RADA species documented in the Flora de Carajás also increased through time, reaching 107 species by 2024. Edaphic endemics accumulated more slowly and irregularly, reaching 17 species by 2024 (Fig. 5a). The three destination strata (canga, forest, *ex situ*) showed similar accumulation patterns (Fig. 5b).

Annual registered diversity was unrelated to the area cleared in the same year (n = 11 years with area data and at least three samples), while rescue yield per suppressed hectare increased by 15.3% per year ($P < 0.001$; Online Resource 5). Temporal trends in registered diversity therefore reflect changes in the program rather than variation in suppressed area.

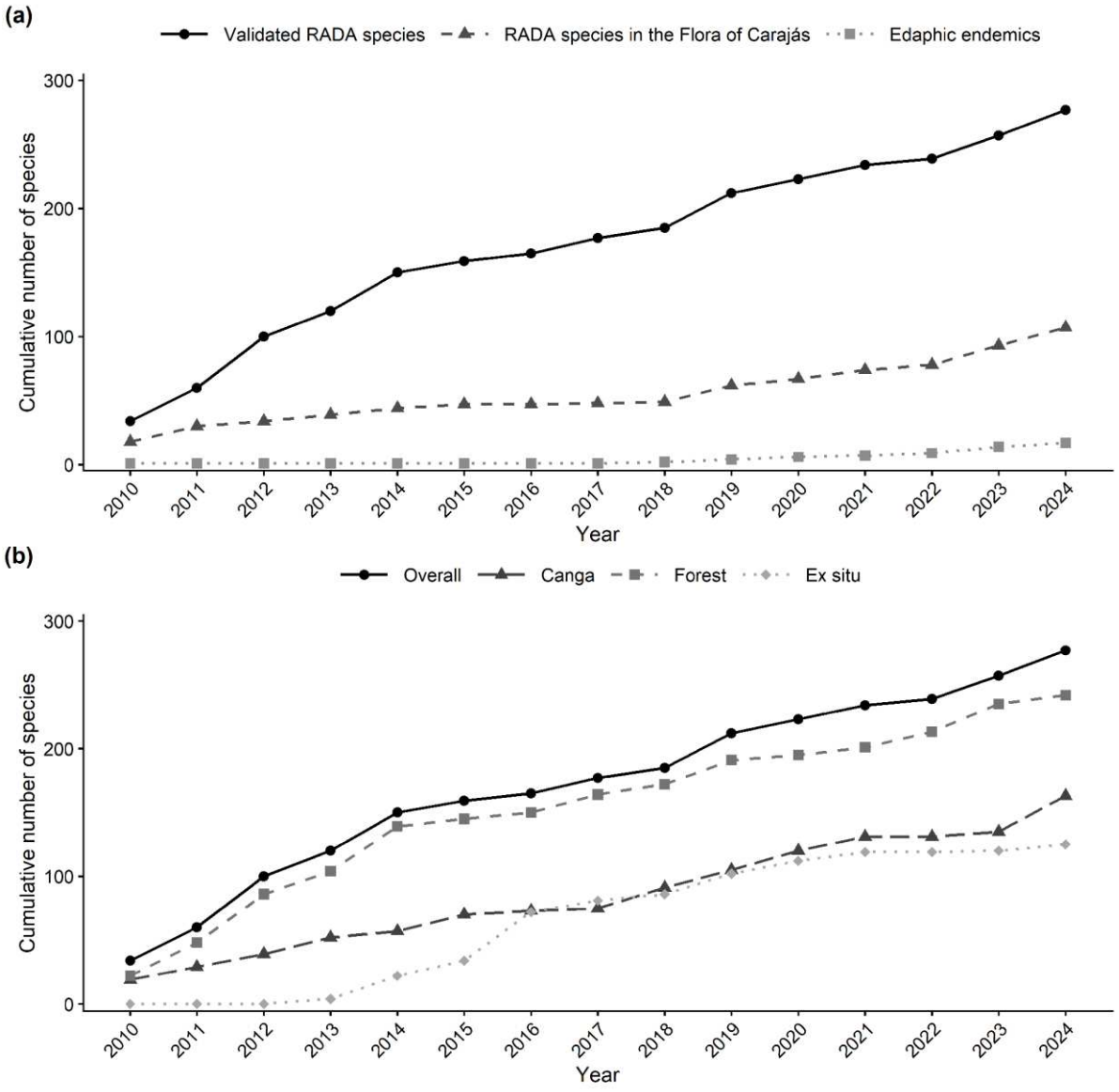


Fig. 5 Cumulative species accumulation in the RADA program, Carajás, eastern Amazon, 2010–2024. (a) Taxonomically validated RADA species (n = 277 by 2024), the subset documented in the Flora of Carajás (n = 107), and edaphic endemics sensu Giulietti et al. (2019) (n = 17). (b) Species accumulation across destination strata (canga, forest, ex situ) and overall. Curves show the cumulative number of unique species recorded up to each year; years with no new species contribute zero and the curve remains flat.

Species abundance distribution

Abundance across the 323,374 validated individuals was highly biased towards a small subset of species, with 37 species with more than 1,000 rescued individuals accounting for 289,704 individuals (89.6%). In turn, 161 species with 1-100 individuals accounted for only 3,692 individuals (1.14%). The ten most abundant species alone represented 224,422 individuals, or 69.4% of all validated individuals (Fig. 6; Online Resource 6).

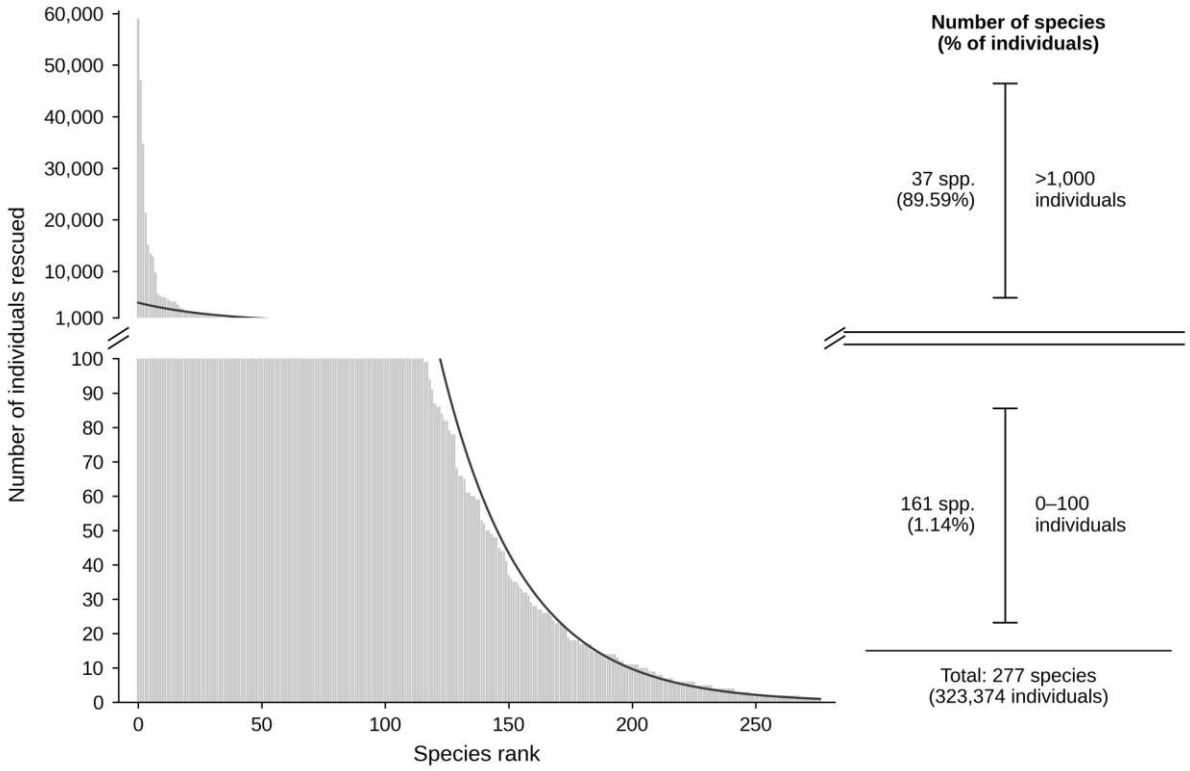


Fig. 6 Rank–abundance distribution of the 277 taxonomically validated RADA species (323,374 individuals). The y-axis is segmented to display the 0–100 and >1,000 abundance ranges; the 101–1,000 range is not displayed on the axis. The fitted relationship is shown ($\log_{10}(y) = -0.0130x + 3.5933$, $R^2 = 0.9740$).

Functional representation

Growth-form representation was uneven across the validated RADA species pool, which was dominated by herbaceous species (200 of 277; 72.2%), whereas woody species were comparatively scarce (trees 9.7%, shrubs 6.5%; combined 16.2%), with vines/climbers (9.7%) and palms (1.8%) making up the remaining (Online Resource 7). Herbaceous dominance was similar in the subset documented in the reference flora (76 of 107; 71%) and in the RADA-only fraction (124 of 170; 72.9%); the overall growth-form distribution nonetheless differed between the two fractions ($\chi^2 = 17.06$, df = 4, P = 0.002), driven mainly by a higher share of trees among RADA- only species (12.9% vs. 4.7%) and of shrubs within the *canga*-associated subset (13.1% vs. 2.4%) rather than by any difference in herbaceous representation (Fisher's exact test for herbs vs. woody taxa, P = 0.61).

Conservation priority species

Conservation priority taxa, here counted as threatened species and edaphic endemics, were underrepresented in RADA records throughout the study period. The rescue data covered 19 of 52 regionally listed threatened species (36.5%; Online Resource 8) and 17 of 34 possible edaphic endemics registered in Serra Norte (50%; Online Resource 3). Within the mined plateaus N4 and N5, 14 of the 27 documented edaphic endemics had rescue records (51.9%; Online Resource 2). Taxa subject to compulsory compensation under environmental licensing were more likely to be included, but translocation debts were substantial in both categories.

Discussion

This study explored whether a long-term flora rescue program fulfills its conservation goals, particularly for endemic and rare species in mined sites in eastern Amazon. We show three dimensions of biodiversity (species, phylogenetic and functional) and their similarity

526 with the reference *canga* flora, highlighting opportunities for improvements in rescue. The
527 program nonetheless delivers measurable results, beginning with an auditable record of more
528 than 351,000 rescue events. We also show that representativeness increased over time,
529 supporting the idea of improved adaptive management through time. Rescue yield per cleared
530 hectare also increased over time, and canga-associated and endemic species entered the
531 records at a growing rate after the publication of the reference floras. To better assess
532 conservation outcomes, reporting data on survival and establishment is required (Doyle et al.
533 2023).

534 The gap between the reference flora and the rescue records persisted even after spatial
535 restriction of the reference pool. Natural beta diversity among campo rupestre sites and
536 sampling-effort differences may inflate apparent dissimilarity (Rapini et al. 2008), and the
537 N4–N5 reference list does not reconstruct the flora present within the exact cleared areas.
538 These limitations qualify the magnitude of the gap but not its direction: registered coverage
539 reached only 14.5% of the flora documented for N4 and N5, the reference pool most
540 favorable to the program.

541 Limited floristic coverage should not be interpreted either as ecological equivalence
542 or as evidence that rescue lacks conservation value. At the Alegria mine, 79 of 333
543 inventoried species were rescued, while rescue and topsoil transfer together retained
544 germplasm from 107 species; nevertheless, more than 200 inventoried species were not
545 represented by either method (Santos 2010).

546 Our results show that a long-running flora rescue program can generate substantial
547 operational output relevant from the conservation perspective because mitigation
548 translocations are often evaluated through auditable delivery metrics, such as the number of
549 individuals rescued, propagated or planted, whereas conservation outcomes depend on
550 whether actions retain the taxonomic, functional, phylogenetic and priority-species attributes

of the impacted ecosystem (Doyle et al. 2023; Julien et al. 2023b; Wardil et al. 2025). Doyle et al. (2023) distinguishes process-oriented mitigation translocations from conservation-oriented projects with explicit objectives and measurable post-translocation outcomes. Thus, improving conservation outcomes relies on aligning agreements between governments and industries based on international standards.

The concentration of individuals in a limited number of botanical families in the RADA records suggests a manageability bias, in which species that are easier to detect, handle, transport or cultivate become overrepresented relative to the broader canga flora (Chen et al. 2013; Isaac and Pocock 2015; Holl et al. 2022). This pattern is consistent with the practical constraints of translocating plants from ancient, nutrient-poor and edaphically specialized systems, where species may depend on narrow microhabitats, low-fertility substrates, specialized interactions and local adaptation (Zappi et al. 2019; Arruda et al. 2023). The family-level comparison between the rescued and the reference species pools reinforces this interpretation: highly diverse reference families remain poorly represented among rescued species, whereas manageable groups remain prominent (Zappi et al. 2019; Andrino et al. 2020; Devecchi et al. 2020; da Fonseca-da-Silva et al. 2020; Massante et al. 2023). The phylogenetic clustering detected by the null-model analysis provides a formal signature of this bias.

Santos (2010) provides a useful comparison for interpreting this concentration without assuming that it results exclusively from operational bias. A comparable concentration was reported by Santos (2010), with Orchidaceae and Bromeliaceae accounting for 88.7% of rescued individuals at Alegria. Both families showed survival above 80% after one year, and Orchidaceae was also the

richest family in the local inventory. Thus, concentration in particular groups may reflect both local floristic composition and differential suitability for rescue, although it does not demonstrate broad community representativeness. The temporal increase in canga–associated and endemic records after 2018 likely reflects the publication of regional floristic references (Mota et al. 2018; Salino et al. 2018), the reduced taxonomic uncertainty that followed, the recent description of several endemics (Giulietti et al. 2019), and possibility of a better knowledge of suppression fronts into areas where these species occur. Taxa documented in the reference canga flora rose from 26.5% to 63.0% of the species entering the records between the two periods, whereas canga records rose only from 33.4% to 43.2% of rescue events, indicating that the program absorbed the newly available taxonomic information rather than merely encountering more canga. This pattern is consistent with field teams actively incorporating updated scientific knowledge into rescue practices. The phylogenetic signal points the same way without reaching significance, as mean SES–PD rose from −1.75 to −1.23 after 2018 and the proportion of significantly clustered assemblages fell from 9 of 15 to 3 of 9, but with 24 year–by–complex assemblages

the trend is not distinguishable from zero (Spearman ρ = 0.10, P = 0.65; Online Resource 9).

Furthermore, endemic representativeness was interpreted as a descriptive indicator, rather than a performance expectation, because many edaphic endemics are micro-endemics of plateaus not yet affected by clearing in the period evaluated (e.g., *Parapiqueria cavalcantei, Isoetes cangae*, *Carajasia cangae*), and several were formally described only during the study period (19 of the 38 edaphic endemics were described between 2010-2018 - e.g. *Paspalum cangarum, Buchnera carajasensis, Eleocharis pedrovianae, Isoetes serracarajensis and I. cangae*), so their absence from RADA records reflects lack of exposure or of an available name, rather than program failure (Online Resource 3).

Conservation priority taxa remained underrepresented throughout the study period, with 19 of 52 regionally threatened species (36.5%) appearing in RADA records and edaphic endemic coverage reaching 17 validated species despite improvement in *canga*-associated taxa after 2018 (Tyski et al. 2023). That post-2018 increase in representativeness indicates that refined taxonomic knowledge can partially alleviate taxonomic uncertainty, but insufficient to overcome the filters acting upon manageable edaphic endemics, with restricted phenology and functional specialization that are both most irreplaceable and least amenable to *ex situ* propagation (Arruda et al. 2023; Wardil et al. 2025; Godefroid et al. 2011). This preference for operationally manageable taxa over conservation priorities is a recurrent feature of translocation programs worldwide (Julien et al. 2022; Fenu et al. 2023; Diallo et al. 2023; Julien et al. 2023a; zu Ermgassen et al. 2019). Phenology and demographic features add further constraints, as many canga species are annual or seasonally dormant herbs, visible and collectable only during short temporal windows, and many endemics occur at naturally low densities. Altogether, these features reduce the probability of a rescue team encountering collectable individuals before the vegetation suppression schedule. Part of the under-

representation of these groups therefore may reflect the timing and demography of the species themselves rather than operational issues (Giulietti et al. 2019; da Costa et al. 2023; Schaefer et al. 2025; Wardil et al. 2025).

Once sampling completeness was standardized, RADA retrieved almost all species it shares with the reference flora (Chao and Jost 2012), which places the deficit in which species enter the records rather than in sampling effort, while raw Faith's phylogenetic diversity tracked species richness so closely (Spearman $\rho = 0.944$) that it added no conservation signal beyond species counts. A distinct signal emerged only after standardization against a null model: rescued assemblages were phylogenetically clustered relative to expectation (mean SES-PD = −1.22), with one third of samples and half of the annual assemblages significantly clustered and none overdispersed (Faith 1992; Sandel 2018; Tucker and Cadotte 2013).Without calibration against a regional reference, neither metric can distinguish a compositionally biased assemblage from a representative one, because indicators that do not account for species composition are systematically insensitive to the compositional failures that characterize rescue programs under operational constraints (Reid 2015; Noreika et al. 2020; Medeiros-Sarmento et al. 2024). The reference-based coverage metrics, compositional similarity indices and functional representation analyses of the kind developed here translate this diagnosis into auditable instruments that give companies and regulators the means to evaluate whether mitigation programs are achieving ecological rather than procedural targets (Sonter et al. 2018; Doyle et al. 2023).

Taxonomic validation distinguished registered names from accepted species-level records and identified unresolved names and nomenclatural inconsistencies that could affect richness and comparability with reference floras (Meyer et al. 2026). Consequently, unresolved or unverified identifications in species lists submitted under licensing requirements may reduce their comparability with accepted regional inventories (Scott and Hallam 2002). When taxonomic effort is documented without species-level verification, formal compliance may not provide sufficient evidence of ecological performance, creating a potential compliance trap (Parker 2006; da Silva Dias et al. 2017; Margarido et al. 2025). The results therefore suggest, without establishing, that individual or event counts alone are an inadequate metric for evaluating the ecological performance of programs, and suggest that taxonomic verification should be employed across translocation programs.

Environmental licensing agencies are central to this discussion, since government agencies define the flora-rescue obligation under environmental legislation, approve its scope, audit its delivery, and require compliance (Sánchez and Duarte 2022). The metrics analyzed here are therefore those the regulator required, and compliance does not necessarily mean achievement of positive conservation outcomes. The compliance trap is thus co-produced, as agencies specify auditable but ecologically silent deliverables while operators optimize toward them (Parker 2006; Stoffels and Thompson 2026). Regulators can therefore drive positive change by setting reference-based biodiversity targets at the outset, adopting outcome indicators such as compositional similarity to the reference flora and coverage of threatened and endemic species, requiring cumulative and auditable data across reporting years, and providing for independent verification of post-translocation performance over horizons compatible with canga recovery dynamics (Bezombes et al. 2017; zu Ermgassen et al. 2019; Doyle et al. 2023).

Ironstone outcrops carry a flora largely restricted to them (Mota et al. 2018; Giulietti et al. 2019; Zappi et al. 2019) and their ecological recovery is unlikely to occur within typical licensing timeframes (Arruda et al. 2023). These characteristics indicate a high degree of ecological irreplaceability and reinforce the priority of avoiding and minimizing impacts before considering restoration or offsetting measures (Gibson et al. 2011; Fernandes et al. 2022; Maron et al. 2026). Where clearing is authorized, flora rescue may retain part of the affected biodiversity, but it should not be interpreted as replacing the original community or compensating for the absence of verified avoidance and set-aside measures (zu Ermgassen et al. 2019); its value must then be judged by what it recovers from the community being cleared.

Ecological irreplaceability holds across rock outcrop ecosystems worldwide, from ironstone and serpentinite to quartzite, sandstone and limestone formations (de Paula et al. 2026). In such context, evaluating rescue programs only by counts of individuals may overestimate ecological performance if taxonomic composition and conservation-priority taxa are not considered. Rescue records from other regions show a comparable bias away from conservation priorities: of 436 plant translocations in France, only 14% involved threatened taxa (Diallo et al. 2023; see also Julien et al. 2023a; Fenu et al. 2023). Where a regional flora has been published and program records are taxonomically validated, the same framework turns a compliance record into a measure of what was actually recovered.

Conclusion

Fifteen years of flora rescue records revealed a translocation debt between the species documented by the program and the reference flora. This gap was expressed through limited taxonomic coverage, uneven representation of growth forms, abundance disparities and lower phylogenetic diversity than expected for the recorded species richness. However, rigorous

taxonomic validation reduced taxonomic uncertainty, and the representation of canga taxa increased over time, particularly after the publication of regional floristic references, a pattern consistent with the progressive incorporation of updated scientific knowledge into field practices. Taxonomic validation also improved dataset reliability and confirmed that the observed representativeness gap was not explained by unresolved or unvalidated names. The year-by-year format in which RADA data are reported also hinders integrative assessment of what the program has achieved over time, and its data structure, metrics and reporting should evolve accordingly, and integrated reporting across years would further support cumulative evaluation of the program. The reference-based coverage metrics, compositional similarity indices and rarefaction analyses developed here, together with the reorientation discussed above, provide programs with practical, low-cost instruments for demonstrating measurable ecological benefit rather than effort alone.

Statements and Declarations

Competing Interests: Financial interests: F.M.G. Santos was employed by Vale S.A. from 2018 to 2025, the company that owns the RADA database analyzed in this study. F.A.O. Silveira declares no financial interests. Non-financial interests: none.

Ethics approval Not applicable. The RADA reports analyzed here are official environmental licensing documents, formally filed by the operator with the competent environmental agency as a condition of the operating license, and the floristic reference data are previously published. The study did not involve human participants, animals, or the collection of biological material.

Author Contribution: **Fernando M. G. Santos** designed the study and the methodology, performed data curation, analyzed and interpreted the data, and wrote the original draft. **Fernando A. O. Silveira** contributed to study conception and design, methodology and investigation, writing – review and editing, and supervision.

Funding The first author has received a PhD grant from Coordination for the Improvement of Higher Education (CAPES). F.A.O. Silveira is supported by grants from FAPEMIG.

Data Availability The datasets analyzed in this study were compiled from Annual Environmental Performance Reports (RADA) filed with the competent environmental agency as part of the statutory licensing process. The compiled dataset is provided in Online Resource 1; the original reports are held by the licensing authority.

References

Albrecht MA, Osazuwa-Peters OL, Maschinski J, Bell TJ, Bowles ML, Brumback WE, Duquesnel J, Kunz M, Lange J, McCue KA et al (2019) Effects of life history and reproduction on recruitment time lags in reintroductions of rare plants. Conserv Biol 33:601–611. https://doi.org/10.1111/cobi.13255

Andrino CO, Barbosa-Silva RG, Lovo J, Viana PL, Moro MF, Zappi DC (2020) Iron islands in the Amazon: investigating plant beta diversity of *canga* outcrops. PhytoKeys 165:1–25. https://doi.org/10.3897/phytokeys.165.54819

Arlidge WNS, Bull JW, Addison PFE, Burgass MJ, Gianuca D, Gorham TM, Jacob C, Shumway N, Sinclair SP, Watson JEM, Wilcox C, Milner-Gulland EJ (2018) A global mitigation hierarchy for nature conservation. BioScience 68:336–347.

https://doi.org/10.1093/biosci/biy029

Arruda AJ, Medeiros NF, Fiorini CF, Ordóñez-Parra CA, Dayrell RL et al (2023) Ten principles for restoring campo rupestre, a threatened tropical, megadiverse, nutrient-impoverished montane grassland. Restor Ecol 31:e13924. https://doi.org/10.1111/rec.13924

Bezombes L, Gaucherand S, Kerbiriou C, Reinert ME, Spiegelberger T (2017) Ecological equivalence assessment methods: what trade-offs between operationality, scientific basis and comprehensiveness? Environ Manage 60:216–230. https://doi.org/10.1007/s00267-017-0877-5

BFG [The Brazil Flora Group] (2021a) Coleção Flora do Brasil 2020. Jardim Botânico do Rio de Janeiro, Rio de Janeiro. https://doi.org/10.47871/jbrj2021004

BFG [The Brazil Flora Group] (2021b) Flora do Brasil 2020. Jardim Botânico do Rio de Janeiro, Rio de Janeiro. https://doi.org/10.47871/jbrj2021001

Borges-Matos C, Maron M, Metzger JP (2023) A review of condition metrics used in biodiversity offsetting. Environ Manage 72:727–740. https://doi.org/10.1007/s00267-023-01858-1

Bortolus A (2008) Error cascades in the biological sciences: the unwanted consequences of using bad taxonomy in ecology. Ambio 37:114–118. https://doi.org/10.1579/0044-7447(2008)37[114:ECITBS]2.0.CO;2

Bradley HS, Tomlinson S, Craig MD, Cross AT, Bateman PW (2022) Mitigation translocation as a management tool. Conserv Biol 36:e13667. https://doi.org/10.1111/cobi.13667

Bull JW, Souter NJ, Gordon A, Singh NJ, Milner-Gulland EJ (2013) Biodiversity offsets in theory and practice. Oryx 47:369–380. https://doi.org/10.1017/S003060531200172X

Chao A, Chazdon RL, Colwell RK, Shen TJ (2006) Abundance-based similarity indices and their estimation when there are unseen species in samples. Biometrics 62:361–371. https://doi.org/10.1111/j.1541-0420.2005.00489.x

763 Chao A, Jost L (2012) Coverage-based rarefaction and extrapolation: standardizing samples
764 by completeness rather than size. Ecology 93:2533–2547. https://doi.org/10.1890/11-1952.1
765 Chen G, Kéry M, Plattner M, Ma K, Gardner B (2013) Imperfect detection is the rule rather
766 than the exception in plant distribution studies. J Ecol 101:183–191.
767 https://doi.org/10.1111/1365-2745.12021
768 Cressie N, Read TRC (1984) Multinomial goodness-of-fit tests. J R Stat Soc Series B Stat
769 Methodol 46:440–464. https://doi.org/10.1111/j.2517-6161.1984.tb01318.x
770 da Costa ACG, Vasconcelos LV, de Lima CT, Caldeira CF, Zappi DC, Giulietti AM,
771 Watanabe MTC (2023) Reproductive phenology of critical native plant species for mineland
772 restoration in the eastern Amazon. Plant Species Biol 38:131–143.
773 https://doi.org/10.1111/1442-1984.12406
774 da Fonseca-da-Silva TL, Lovo J, Zappi DC et al (2020) Plant species on Amazonian *canga*
775 habitats of Serra Arqueada: the contribution of an isolated outcrop to the floristic knowledge
776 of the Carajás region, Pará, Brazil. Braz J Bot 43:315–330. https://doi.org/10.1007/s40415-
777 020-00608-5
778 da Silva Dias AM, Fonseca A, Paglia AP (2017) Biodiversity monitoring in the
779 environmental impact assessment of mining projects: a (persistent) waste of time and money?
780 Perspect Ecol Conserv 15:206–208. https://doi.org/10.1016/j.pecon.2017.06.001
781 de Paula LFA, Perillo LN, Ottaviani G, Carmo FF, Datar MN, Enebeli-Ekwutoziam KC et al
782 (2026) Scientists' warning on the global destruction of rock outcrop ecosystems. Conserv
783 Biol e70316. https://doi.org/10.1111/cobi.70316
784 Devecchi MF, Lovo J, Moro MF et al (2020) Beyond forests in the Amazon: biogeography
785 and floristic relationships of the Amazonian savannas. Bot J Linn Soc 193:478–503.
786 https://doi.org/10.1093/botlinnean/boaa025

787 Diallo M, Mayeur A, Vaissière A-C et al (2023) The relevance of plant translocation as a
788 conservation tool in France. Plant Ecol 224:777–790. https://doi.org/10.1007/s11258-023-
789 01295-4
790 Doyle CAT, Abeli T, Albrecht MA et al (2023) Achieving conservation outcomes in plant
791 mitigation translocations: the need for global standards. Plant Ecol 224:745–763.
792 https://doi.org/10.1007/s11258-023-01310-8
793 Faith DP (1992) Conservation evaluation and phylogenetic diversity. Biol Conserv 61:1–10.
794 https://doi.org/10.1016/0006-3207(92)91201-3
795 Fenu G, Calderisi G, Boršić I et al (2023) Translocations of threatened plants in the
796 Mediterranean Basin: current status and future directions. Plant Ecol 224:765–775.
797 https://doi.org/10.1007/s11258-023-01303-7
798 Fernandes TN, Dos Santos FMG, Gontijo FD, da Silva Filho JA, Castilho AF, Sánchez LE
799 (2022) Mainstreaming flora conservation strategies into the mitigation hierarchy to strengthen
800 environmental impact assessment. Environ Manage 71:483–493.
801 https://doi.org/10.1007/s00267-022-01756-y
802 Gardner AS, Howarth B (2009) Urbanisation in the United Arab Emirates: the challenges for
803 ecological mitigation in a rapidly developing country. BioRisk 3:27–38.
804 https://doi.org/10.3897/biorisk.3.18
805 Gardner TA, Von Hase A, Brownlie S et al (2013) Biodiversity offsets and the challenge of
806 achieving No Net Loss. Conserv Biol 27:1254–1264. https://doi.org/10.1111/cobi.12118
807 Germano JM, Field KJ, Griffiths RA et al (2015) Mitigation-driven translocations: are we
808 moving wildlife in the right direction? Front Ecol Environ 13:100–105.
809 https://doi.org/10.1890/140137

Gibson L, Lee TM, Koh LP, Brook BW, Gardner TA, Barlow J, Peres CA, Bradshaw CJA, Laurance WF, Lovejoy TE, Sodhi NS (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478:378–381. https://doi.org/10.1038/nature10425

Giulietti AM, Giannini TC, Mota NFO et al (2019) Edaphic endemism in the Amazon: Vascular plants of the *canga* of Carajás, Brazil. Bot Rev 85:357–383. https://doi.org/10.1007/s12229-019-09214-x

Godefroid S, Piazza C, Rossi G et al (2011) How successful are plant species reintroductions? Biol Conserv 144:672–682. https://doi.org/10.1016/j.biocon.2010.10.003

Gotelli NJ, Colwell RK (2001) Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecol Lett 4:379–391. https://doi.org/10.1046/j.1461-0248.2001.00230.x

Hartig F (2022) DHARMa: residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.4.6. https://CRAN.R-project.org/package=DHARMa

Holl KD, Luong JC, Brancalion PHS (2022) Overcoming biotic homogenization in ecological restoration. Trends Ecol Evol 37:777–788. https://doi.org/10.1016/j.tree.2022.05.002

Hsieh TC, Ma KH, Chao A (2016) iNEXT: an R package for rarefaction and extrapolation of species diversity (Hill numbers). Methods Ecol Evol 7:1451–1456. https://doi.org/10.1111/2041-210X.12613

IFC (2012) Performance Standard 6: Biodiversity Conservation and Sustainable Management of Living Natural Resources. International Finance Corporation, Washington, DC. https://www.ifc.org/content/dam/ifc/doc/2010/2012-ifc-performance-standard-6-en.pdf

Isaac NJ, Pocock MJ (2015) Bias and information in biological records. Biol J Linn Soc 115:522–531. https://doi.org/10.1111/bij.12532

834 IUCN/SSC (2013) Guidelines for Reintroductions and Other Conservation Translocations.
835 Version 1.0. IUCN Species Survival Commission, Gland, Switzerland.
836 https://portals.iucn.org/library/efiles/documents/2013-009.pdf
837 Jin Y, Qian H (2022) V.PhyloMaker2: An updated and enlarged R package that can generate
838 very large phylogenies for vascular plants. Plant Diversity 44:335–339.
839 https://doi.org/10.1016/j.pld.2022.05.005
840 Julien M, Colas B, Muller S, Schatz B (2022) Quality assessment of mitigation translocation
841 protocols for protected plants in France. J Environ Manage 302:114064.
842 https://doi.org/10.1016/j.jenvman.2021.114064
843 Julien M, Delayance L, Cortes F, Dixon L, Filippi G, Schatz B (2023a) Anticipating plant
844 translocation in land-use projects: a case study of Helianthemum marifolium. Bot Lett
845 170:255–266. https://doi.org/10.1080/23818107.2023.2168751
846 Julien M, Schatz B, Robert A et al (2023b) Monitoring time of conservation-driven and
847 mitigation-driven plant translocations in Europe. Plant Ecol 224:791–802.
848 https://doi.org/10.1007/s11258-023-01311-7
849 Kembel SW, Cowan PD, Helmus MR, Cornwell WK, Morlon H, Ackerly DD, Blomberg SP,
850 Webb CO (2010) Picante: R tools for integrating phylogenies and ecology. Bioinformatics
851 26:1463–1464. https://doi.org/10.1093/bioinformatics/btq166
852 Kendall MG (1938) A new measure of rank correlation. Biometrika 30:81–93.
853 https://doi.org/10.1093/biomet/30.1-2.81
854 Mallett A, França ELB, Alves Í, Mills L (2021) Environmental impacts of mining in Brazil
855 and the environmental licensing process: changes needed for changing times? Extr Ind Soc
856 8:100952. https://doi.org/10.1016/j.exis.2021.100952

857 Margarido NT, Hanna P, Arts J, Souza LR, Sánchez LE (2025) Biodiversity monitoring in
858 impact assessment follow-up: insights from a large-scale mine in the Brazilian Amazon
859 rainforest. Environ Manage 75:2554–2570. https://doi.org/10.1007/s00267-025-02246-7
860 Maron M, Ives CD, Kujala H, Bull JW, Maseyk FJF, Bekessy S, Gordon A, Watson JEM,
861 Lentini PE, Gibbons P, Possingham HP, Hobbs RJ, Keith DA, Wintle BA, Evans MC (2016)
862 Taming a wicked problem: resolving controversies in biodiversity offsetting. BioScience
863 66:489–498. https://doi.org/10.1093/biosci/biw038
864 Maron M, Ward M, Simmonds JS et al (2026) Ecological irreplaceability in the era of Nature
865 Positive. Conserv Lett 19:e70073. https://doi.org/10.1111/con4.70073
866 Massante JC, Neri AV, Villa PM et al (2023) Looking similar but all different: phylogenetic
867 signature of Brazilian rocky outcrops and the influence of temperature variability on their
868 phylogenetic structure. J Ecol 111:1905–1920. https://doi.org/10.1111/1365-2745.14148
869 Medeiros-Sarmento PS, Castro AF, Penner FV, Gastauer M (2024) Indicators to quantify
870 biodiversity gains for compensation and mineland rehabilitation in the eastern Amazon. An
871 Acad Bras Cienc 96:e20230236. https://doi.org/10.1590/0001-3765202420230236
872 Meyer L, Ladle RJ, Trad RJ, Frazão A, Frateles L, Lessa T, Pinto RB, Freitas TMS, Tessarolo
873 G, Guedes JJM, Castro-Souza RA, Medeiros H, Jardim L, Neves DRM, Rosado BHP,
874 Lohmann LG, Hortal J, Diniz-Filho JAF, Stropp J (2026) Taxonomic uncertainty: causes,
875 consequences, and metrics. Trends Ecol Evol 41:299–308.
876 https://doi.org/10.1016/j.tree.2025.12.006
877 Mota NFO, Watanabe MTC, Zappi DC et al (2018) Amazon *canga*: the unique vegetation of
878 Carajás revealed by the list of seed plants. Rodriguésia 69:1435–1488.
879 https://doi.org/10.1590/2175-7860201869336

880 Murray BR, Martin LJ, Phillips ML, Pyšek P (2017) Taxonomic perils and pitfalls of dataset
881 assembly in ecology: a case study of the naturalized Asteraceae in Australia. NeoBiota 34.
882 https://doi.org/10.3897/neobiota.34.11139
883 Noreika N, Pärtel M, Öckinger E (2020) Community completeness as a measure of
884 restoration success: multiple-study comparisons across ecosystems and ecological groups.
885 Biodivers Conserv 29:3807–3827. https://doi.org/10.1007/s10531-020-02050-1
886 Oksanen J, Simpson GL, Blanchet FG et al (2022) vegan: Community Ecology Package. R
887 package version 2.6-4. https://CRAN.R-project.org/package=vegan
888 Paradis E, Schliep K (2019) ape 5.0: an environment for modern phylogenetics and
889 evolutionary analyses in R. Bioinformatics 35:526–528.
890 https://doi.org/10.1093/bioinformatics/bty633
891 Parker C (2006) The "compliance" trap: The moral message in responsive regulatory
892 enforcement. Law Soc Rev 40:591–622. https://doi.org/10.1111/j.1540-5893.2006.00282.x
893 Rapini A, Ribeiro PL, Lambert S, Pirani JR (2008) A flora dos campos rupestres da Cadeia
894 do Espinhaço. Megadiversidade 4:16–24
895 R Core Team (2023) R: A language and environment for statistical computing. R Foundation
896 for Statistical Computing, Vienna. https://www.R-project.org/
897 Rao CR (1982) Diversity and dissimilarity coefficients: a unified approach. Theor Popul Biol
898 21:24–43. https://doi.org/10.1016/0040-5809(82)90004-1
899 Reid JL (2015) Indicators of success should be sensitive to compositional failures: reply to
900 Suganuma and Durigan. Restor Ecol 23:519–520. https://doi.org/10.1111/rec.12254
901 Salino A, Arruda AJ, Almeida TE (2018) Ferns and lycophytes from Serra dos Carajás, an
902 eastern Amazonian mountain range. Rodriguésia 69:1417–1434.
903 https://doi.org/10.1590/2175-7860201869335

904 Sánchez LE, Duarte CG (2022) Environmental impact assessment in Brazil: a review of its
905 rise (and fall?). In: Fonseca A (ed) Handbook of Environmental Impact Assessment. Edward
906 Elgar Publishing, Cheltenham, pp 383–403. https://doi.org/10.4337/9781800889996.00027
907 Sandel B (2018) Richness-dependence of phylogenetic diversity indices. Ecography 41:837–
908 844. https://doi.org/10.1111/ecog.02967
909 Santos LM (2010) Restauração de campos ferruginosos mediante resgate de flora e uso de
910 topsoil no Quadrilátero Ferrífero, Minas Gerais. Dissertation, Universidade Federal de Minas
911 Gerais, Belo Horizonte. http://hdl.handle.net/1843/TJAS-8BQGLF
912 Schaefer CEGR, Michel RFM, Nunes JA et al (2025) Extreme stress environment:
913 unmatched temperature and water regimes of rupestrian grasslands on *canga* (ferricrete,
914 ironstone) at Carajás (eastern Amazonia, Brazil). Environ Monit Assess 197:1316.
915 https://doi.org/10.1007/s10661-025-14774-0
916 Scott WA, Hallam CJ (2002) Assessing species misidentification rates through quality
917 assurance of vegetation monitoring. Plant Ecol 165:101–115.
918 https://doi.org/10.1023/A:1021441331839
919 Serrano AM, Fonseca A (2024) License to pollute? A longitudinal analysis of environmental
920 performance, compliance, and enforcement in 232 licensed projects in Brazil. J Clean Prod
921 434:139977. https://doi.org/10.1016/j.jclepro.2023.139977
922 Skirycz A, Castilho A, Chaparro C, Carvalho N, Tzotzos G, Siqueira JO (2014) *canga*
923 biodiversity, a matter of mining. Front Plant Sci 5:653.
924 https://doi.org/10.3389/fpls.2014.00653
925 Sonter LJ, Ali SH, Watson JEM (2018) Mining and biodiversity: key issues and research
926 needs in conservation science. Proc R Soc B 285:20181926.
927 https://doi.org/10.1098/rspb.2018.1926

928 Sonter LJ, Maron M, Bull JW et al (2023) How to fuel an energy transition with ecologically
929 responsible mining. Proc Natl Acad Sci USA 120:e2307006120.
930 https://doi.org/10.1073/pnas.2307006120
931 Souza-Filho PWM, Giannini TC, Jaffé R et al (2019) Mapping and quantification of
932 ferruginous outcrop savannas in the Brazilian Amazon: A challenge for biodiversity
933 conservation. PLoS One 14:e0211095. https://doi.org/10.1371/journal.pone.0211095
934 Spearman C (1904) The proof and measurement of association between two things. Am J
935 Psychol 15:72–101. https://doi.org/10.2307/1412159
936 Stoffels RJ, Thompson RM (2026) Avoiding the 'one-size-fits-all' trap in policy-based
937 monitoring. River Res Appl 42:732–746. https://doi.org/10.1002/rra.70087
938 Tanguay L, Bissonnette J-F, Calmé S, Koutouki K, Turgeon K (2025) Avoiding an
939 anticipated social-ecological trap in biodiversity conservation on private lands in Quebec
940 province, Canada. Environ Sci Policy 170:104095.
941 https://doi.org/10.1016/j.envsci.2025.104095
942 Thompson GG, Thompson SA (2023) Compliance is not necessarily good science. The
943 Environmental Protection Authority's vertebrate fauna survey guidelines should be revised.
944 Pac Conserv Biol 30:PC23040. https://doi.org/10.1071/PC23040
945 Torres A, Zu Ermgassen SO, Navarro LM et al (2024) Mining threats in high-level
946 biodiversity conservation policies. Conserv Biol 38:e14261.
947 https://doi.org/10.1111/cobi.14261
948 Tucker CM, Cadotte MW (2013) Unifying measures of biodiversity: understanding when
949 richness and phylogenetic diversity should be congruent. Divers Distrib 19:845–854.
950 https://doi.org/10.1111/ddi.12087

951 Tyski L, Cardoso ARL, Neto CSC (2023) Taxons da Serra dos Carajás contidos em listas
952 oficiais de espécies ameaçadas de extinção. In: Giannini TC (ed) Capital Natural das
953 Florestas de Carajás. pp 75–85
954 Venables WN, Ripley BD (2002) Modern applied statistics with S, 4th edn. Springer, New
955 York
956 Viana PL, Mota NFO, Gil ASB et al (2016) Flora das *canga*s da Serra dos Carajás, Pará,
957 Brasil: história, área de estudos e metodologia. Rodriguésia 67:1107–1124.
958 https://doi.org/10.1590/2175-7860201667501
959 Wardil MV, Arruda AJ, Bueno ML, Santos FMG, Silveira FAO (2025) Environmental niche
960 modelling, population structure and reproductive phenology inform translocation in
961 threatened Amazon endemics. Plant Ecol 226:867–882. https://doi.org/10.1007/s11258-025-
962 01536-8
963 Zappi DC, Moro MF, Walker B et al (2019) Plotting a future for Amazonian *canga*
964 vegetation in a campo rupestre context. PLoS One 14:e0219753.
965 https://doi.org/10.1371/journal.pone.0219753
966 zu Ermgassen SOSE, Baker J, Griffiths RA et al (2019) The ecological outcomes of
967 biodiversity offsets under "no net loss" policies: A global review. Conserv Lett 12:e12664.
968 https://doi.org/10.1111/conl.12664

969

970

971 **Supplementary Material**

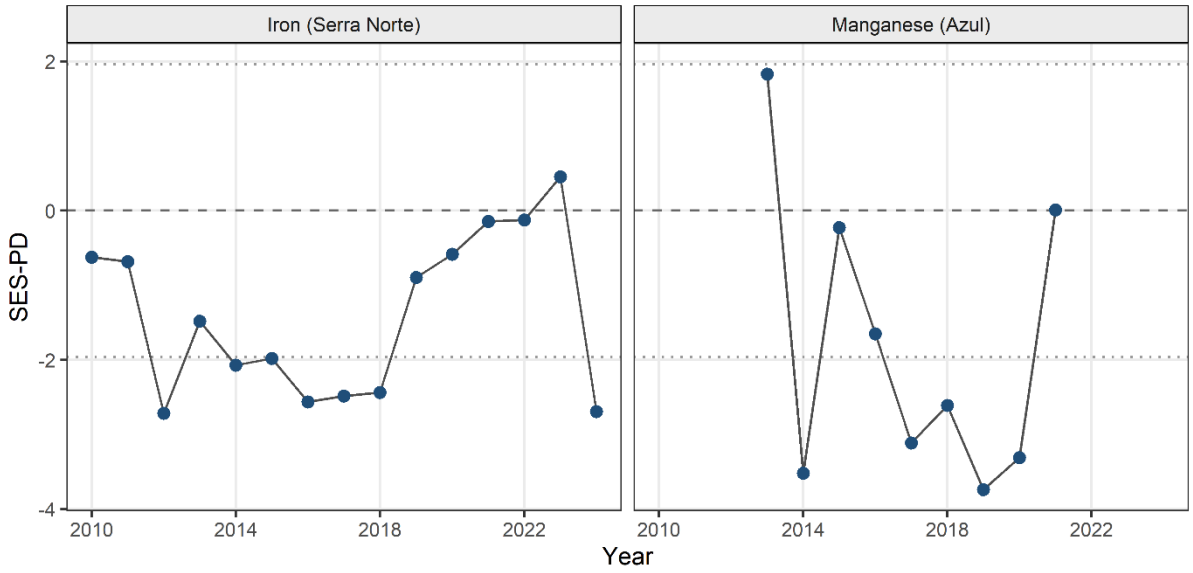
972

973 **Online Resource 1.** Complete list of species registered in RADA, North Range – Carajás and
974 their translocation status, distribution of registers year-by-year (2010–2024).

975 **Online Resource 2.** Additional analyses (spreadsheet): (a) registered coverage and
976 compositional similarity (Jaccard and Sørensen indices) under three reference-pool
977 definitions (regional flora; Serra Norte N1–N8; mined plateaus N4–N5); (b) SES-PD of
978 annual assemblages; (c) annual suppressed area versus registered diversity, with negative-
979 binomial GLM summary; (d) sensitivity of rarefied estimates to the exclusion of two zero-
980 richness samples; (e) coverage and similarity under two taxonomic filtering scenarios (277
981 accepted names versus 291 species-level names), (f) species-level list of the N4–N5 reference
982 pool (429 species) with plateau occurrence and RADA record; (g) membership of the 277
983 validated RADA species in each reference pool.

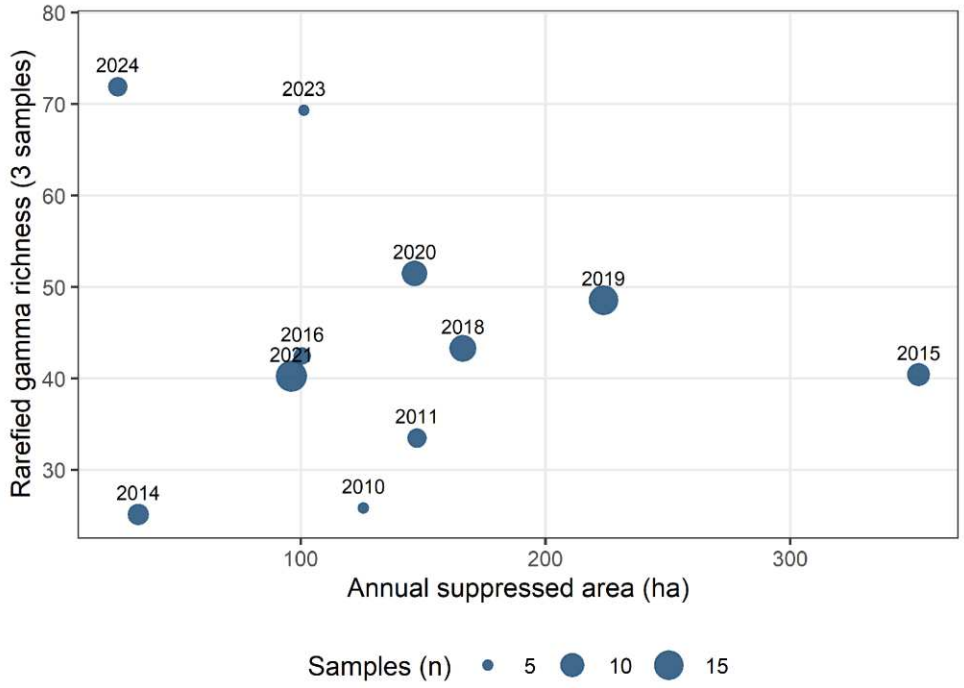
984 **Online Resource 3.** Complete list of edaphic endemic species (Giulietti et al. 2019) and their
985 rescue status.

986



987

988 **Online Resource** 4. Standardized effect size of Faith's phylogenetic diversity (SES-PD;
989 taxa.labels null model, 999 randomizations) for annual assemblages by mining complex.
990 Dashed lines indicate the null expectation (0) and the two-tailed 95% envelope (±1.96).

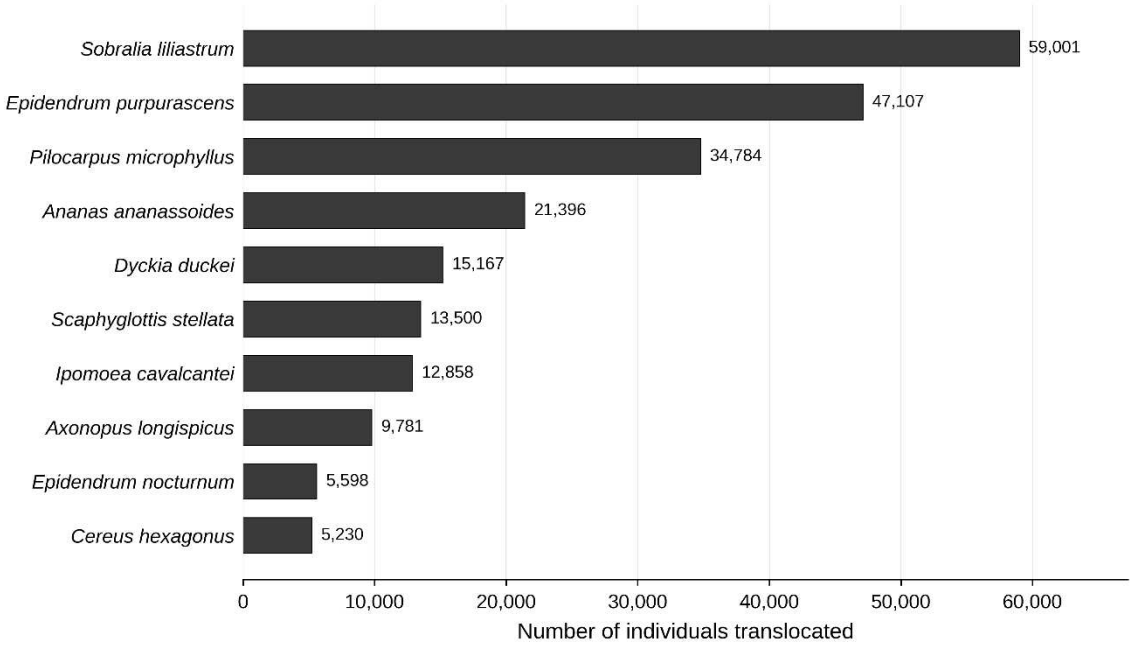


991

992 **Online Resource 5.** Annual suppressed area (ha), compiled from the original RADA reports,

993 versus rarefied gamma richness for the iron complex (2010–2024; 11 years with area data).

994 Point size indicates the number of operational samples per year.

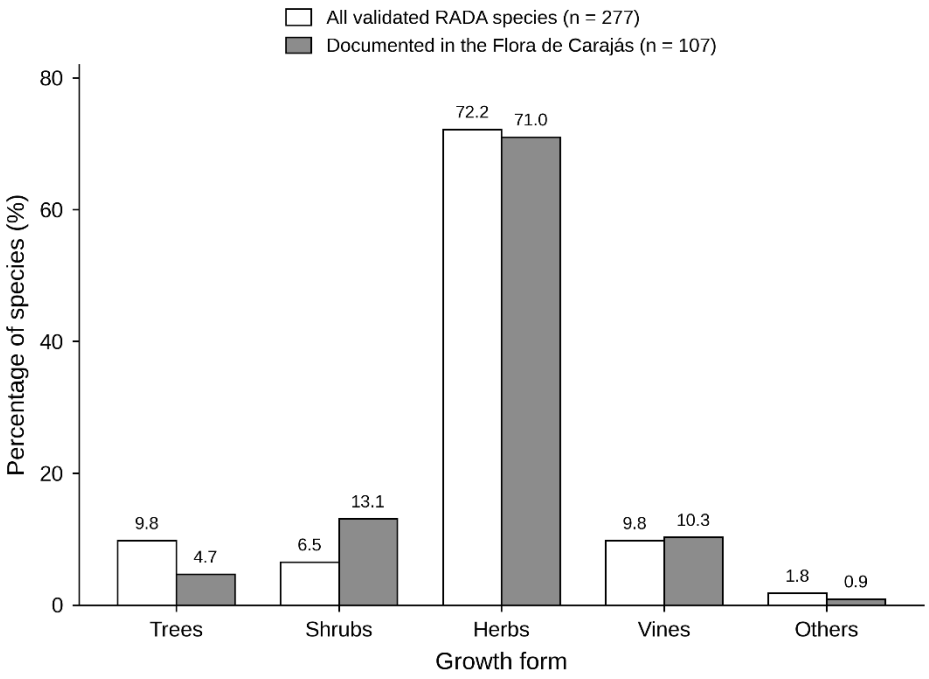


995

996 **Online Resource 6.** Ten most abundant plant species translocated by the RADA program in

997 Carajás, Brazil (2010–2024). Abundance is expressed as the total number of individuals per

998 species across the 15-year period.



999

1000 **Online Resource 7.** Habit-based functional representativeness of validated RADA species.
1001 Bars show the proportional distribution of growth forms in the full validated RADA species
1002 pool and in the subset of validated RADA species documented in the Flora de Carajás.

1003 **Online Resource 8**. Plant species registered in the Carajás Mining Complex RADA included
1004 in the lists: Endangered in Pará state (Coema-PA, 2007), Brazilian National Flora
1005 Endangered List (Portaria MMA 148/2022); Reference Flora (Mota et al. 2018 + Salino et al.
1006 2018); and Edaphic endemics (Giulietti et al. 2019). Ind refers to the number of rescued
1007 individuals.

1008 **Online Resource 9.** Temporal trend in the standardized effect size of Faith's phylogenetic
1009 diversity (SES-PD) across the 24 year-by-complex assemblages, 2010–2024. Sheets give the
1010 per-assemblage estimates, a comparison between the periods before and after the 2018
1011 publication of the regional floristic references, and the trend tests (Spearman rank correlation
1012 overall and within each mining complex; Mann-Whitney U between periods).

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Onlineresource1DatabaseRADA20102024.xlsx
- Onlineresource1SppListRADAYeartoyear.xlsx
- OnlineResource2additionalanalyses.xlsx
- OnlineResource3EndemicEdaphicrescuelist.xlsx
- OnlineResource8Plantspeciesconservation.xlsx
- OnlineResource9SESPDtemporaltrend.xlsx
- SupplementaryMaterial.docx