Field testing of an example of biodiversity-related outcome monitoring requirements developed for the Forest Stewardship Standard for Plantations for Brazil.

Final Report

Forest Stewardship Council®

CASA DA FLORESTA

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INTRODUCTION

Forests are among the most biologically diverse ecosystems on Earth, providing habitats for countless species and playing a crucial role in maintaining global ecological balance. Conserving biodiversity is essential for sustaining ecosystem services such as carbon sequestration, water regulation, and soil health. These services are vital for environmental stability and human well-being.

In response to increasing deforestation and biodiversity loss worldwide, the Forest Stewardship Council (FSC) established the gold standard for responsible forest management that promotes healthy and resilient forests. To better demonstrate the positive outcomes achieved by FSC certification, FSC International, WWF and IKEA launched the "Fast-Tracking Biodiversity Assessment Project", which comprises two work packages and focuses on three countries: Brazil, Sweden and Finland.

The first work package evaluated the added value of FSC certification for biodiversity conservation across the three countries. The desk review found that the FSC Forest Stewardship Standards complement national legislation by introducing additional requirements, including biodiversity monitoring, the promotion of landscape connectivity, and the designation of set-aside areas. Survey evidence from key stakeholders supports these findings, indicating that FSC certification enhances knowledge of native species occurrence, strengthens connectivity among patches of native ecosystem, and contributes to the protection of habitats of rare and threatened species. However, significant differences in monitoring methodologies among certificate holders limit the comparability of data and constrain the ability to undertake robust quantitative assessments of the biodiversity outcomes of FSC certification.

Building on these findings, the main aim of the second work package is to develop and test an example of biodiversity-related outcome monitoring requirements for the FSC Forest Stewardship Standard for Plantations in Brazil (FSC-STD-BRA-1.1-2025-Plantations). These requirements, developed based on the FSC Guidance to Develop Outcome-Oriented Forest Stewardship Standards (FSC-GUI-60-006 V1-0), illustrate how outcome-oriented standards can ensure consistent, continuous, and replicable biodiversity monitoring across FSC-certified management units.

In this context, the present consultancy focuses on testing the implementation of an example of monitoring requirements developed to test the progress towards the following biodiversity-related intended outcomes of FSC certification:

1) *The biological diversity of rare and threatened species in the management unit is maintained over time (Key Intended Outcome under Criterion 6.4)*; and

2) *The biological diversity of representative sample areas of native ecosystems within the management unit resembles that of ecological reference conditions (Key Intended Outcome under Criterion 6.5)*.

Therefore, the consultancy comprises four main deliverables: 1) the Inception Report, establishing the planning, field guide, list of partners, and maps of the study areas; 2) the Field Report, documenting executed activities, costs, photos, and collected data; 3) the Presentation of Preliminary Results, sharing metrics and comparisons with reference values to obtain stakeholder feedback; and 4) the Final Report, a consolidated document summarizing activities, presenting final results at different levels, and providing recommendations to improve the monitoring framework.

Overview of Deliverable 4

Deliverable 4 (Final Report) consolidates the results and lessons learned from the field testing. This document presents a comprehensive summary of all activities carried out, from initial planning to the presentation and feedback sessions. It includes a detailed analysis of the indicator values at the Management Unit (MU) level, complemented by numerical and graphical comparisons with the established baselines. Finally, the report offers a critical reflection on the process and presents strategic recommendations to FSC International, FSC Brazil, and FSC stakeholders for the future improvement of the monitoring framework, its implementation, and the interpretation of the results obtained.

TESTED MONITORING REQUIREMENTS

Two distinct sets of monitoring requirements, designed for FSC Forest Stewardship Standard for Plantations in Brazil (FSC-STD-BRA-1.1-2025-Plantations), were tested. The first set was intended to evaluate progress toward the key intended outcome under Criterion 6.4, while the second targeted outcomes under Criterion 6.5. Although formulated independently, both sets employed comparable approaches to sampling design and data collection methodologies. See the summary of these monitoring requirements below; they are all listed in Annex 1.

For criterion 6.4.

Criterion 6.4 requires organizations to protect rare and threatened species and their habitats within the management unit through the implementation of conservation areas, ecological connectivity and other appropriate management measures. Therefore, the key

intended outcome associated with this criterion is "The biological diversity of rare and threatened species in the management unit is maintained over time ".

To evaluate this outcome, monitoring is based on the collection of phytosociological data in permanent sample plots distributed across representative vegetation areas. Remote sensing-related components support the identification of the level of connectivity between Representative Sample Areas of native ecosystems or RSA and the delimitation and spatial characterization of conservation areas. Field-based components include the measurement of species composition, density and dominance within these plots, with analyses focused exclusively on RTE species. These data allow the analysis of species composition and population structure through metrics related to floristic composition, density and dominance.

In the field testing, these metrics were synthesized into an index integrating structural variables derived from phytosociological analyses, allowing comparison between the current monitoring results and data from previous monitoring cycles.

For criterion 6.5.

Criterion 6.5 requires organizations to identify and protect representative sample areas of native ecosystems or restore them when such areas do not exist or are insufficient. Thus, the key intended outcome associated with this criterion is "The biological diversity of representative sample areas of native ecosystems within the management unit resembles that of ecological reference conditions".

The evaluation of this outcome is based on the collection of phytosociological data in standardized sampling plots, allowing the characterization of the floristic composition and structural attributes of the plant communities present in the assessed areas. Remote sensing-related components support the development of the Ecological Similarity Index. Field-based components include the collection of floristic and structural data (composition, density, dominance and diversity) within sampling plots. From these data, ecological metrics related to species composition, density, dominance and diversity are calculated.

To support the interpretation of these comparisons, a synthetic index was used to integrate normalized differences in species diversity and richness between the monitored areas and the reference ecosystems.

FIELD TESTING METHODS

Selection of Partner Certificate Holders and Management Units

Certificate Holders (CHs) were selected based on the location and characteristics of their management units. Priority was given to management units located across biomes that host most FSC-certified plantations. Additionally, only management units large enough to accommodate the minimum of four plots were considered. For each selected unit, the sampled area was calculated as Sampled Area = 0.03 × √ (Area of the management unit), and the number of plots was determined using Number of Plots = 10 × Sampled Area.

Management Units Selected for Field Testing

The study was conducted in three management units (MUs) located across three states and biomes. Management Unit 1 is located in the Pampa biome. Land use is composed of 63% eucalyptus plantations and 32% native vegetation at different stages of conservation or regeneration. The predominant native vegetation is Deciduous Seasonal Forest. Management Unit 2 is located in the Atlantic Forest biome. Land use is composed of 61% eucalyptus plantations and 39% native vegetation at different stages of conservation or regeneration. The predominant native vegetation is Mixed Ombrophilous Forest and Semideciduous Seasonal Forest. Management Unit 3 is located in the Cerrado biome. Land use is composed of 77% eucalyptus plantations and 23% native vegetation at different stages of conservation or regeneration. The predominant vegetation type is Cerrado *sensu stricto* (typical woody savanna of the Brazilian Cerrado biome). The main characteristics of the management units selected for field testing is presented in Table 1.

Table 1. Overview of the assessed management units, including location, vegetation type, total certified area, subset included in the analyses, and area of set asides or representative sample areas of native ecosystems (RSA).

Management Unit	Biome	Vegetation type	Certified Area	Subset included in tests	Area of set asides or RSA (%)
MU1 from CH1	Pampa	Deciduous Seasonal Forest	454 692 ha	8 560 ha	2712 (32%)
MU2 from CH2	Atlantic Forest	Mixed Ombrophilous Forest	361 992 ha	120 285 ha	46566 (39%)
MU 3 from CH3	Cerrado	Cerrado *sensu stricto*	618 480 ha	16 783 ha	3942 (23%)

Stratification and sampling methods

Sampling design and plot allocation

Initially, the management units were stratified according to ecological characteristics, including vegetation type, based on the IBGE (2012) classification, conservation stage (initial, intermediate, advanced, and primary forest, according to CONAMA), and land use class, such

as native vegetation, restoration areas, and managed native forest. This stratification was based on information provided by Certificate Holders (CHs) or derived from MapBiomas.

A grid composed of 100 × 100 m cells was overlaid on the total area of each management unit, and only cells fully contained within native vegetation remnants were retained for sampling. From these filtered cells, sampling units were selected and proportionally distributed according to the area occupied by each stratum.

In a subsequent stage, the preliminary selection of plot locations was refined in coordination with the FSC International Monitoring and Evaluation (M&E) team, based on analyses of landscape metrics related to structural connectivity and ecological similarity among native vegetation fragments. This refinement aimed to ensure that plot allocation captured not only variation in vegetation characteristics, but also the spatial configuration and ecological integrity of the landscape.

Field validation

Within the selected cells, the exact location of each plot was defined through field verification to ensure ecological integrity and representativeness of the respective strata. Terrain accessibility and the presence of atypical environments, such as rocky outcrops, watercourses, flooded areas, and trails, were assessed prior to plot establishment. Permanent 0.1 ha plots (20 × 50 m) were established only in suitable locations, that is, areas where the entire plot could be safely installed and sampled without including atypical or physically restrictive conditions that could compromise the feasibility, consistency, or representativeness of the vegetation survey.

Sampling intensity

The sampled area in each management unit was calculated as the square root of the total area of representative native ecosystem fragments (in hectares), multiplied by 0.03. The number of sampling plots was determined by multiplying the sampled area by 10, with a maximum of 50 plots established per management unit.

Final plot distribution

The final distribution of sampling plots across the selected management units is presented in Table 2 and Annex 2, which summarize the strata represented in each plot, their proportional area within the set-aside areas, and the main ecological and landscape attributes considered during plot allocation. It is worth noting that the sum of the sampled ecosystems does not necessarily add up to 100% of the set-aside areas, because some

fragments and/or vegetation types that are less common and less representative were not sampled.

Table 2. Distribution of sampling plots across the selected management units, indicating the represented strata, the area and percentage of each stratum within the set-aside area, and the main ecological and landscape characteristics considered in plot allocation. Abbreviations: SSF = Seasonal Semideciduous Forest; MOF = Mixed Ombrophilous Forest; HCVA = High Conservation Value Area; ESI = Ecological Similarity Index.

Management Unit	Plot	Stratum	Area (ha)	% of set-aside area	Main characteristics
MU 1	Plot 1	Grassland	124	4,50%	Steppe grassland; good connectivity; negative ESI
	Plot 2	Native Forest	2577	95%	SSF; mixed land use; good connectivity; ESI near 0
	Plot 3				SSF; natural forest; good connectivity; positive ESI
	Plot 4				SSF; natural forest; good connectivity; positive ESI
	Plot 5				SSF; natural forest; low connectivity; ESI near 0
MU 2	Plot 1	Native Forest	1546	3%	MOF
	Plot 2		3269	7%	MOF; HCVA
	Plot 4		449	1%	MOF; HCVA
	Plot 3	Restoration area	31	0,10%	MOF; ecological restoration since 2013
MU 3	Plot 1	Forest formation	1236	31%	Typical Cerrado; HCVA Matão
	Plot 2	Savanna	187	5%	Typical Cerrado; wooded savanna
	Plot 3	Flooded Grassland	445	11%	Cerrado; buriti palm swamp
	Plot 4	Restoration area	785	20%	Cerrado; natural regeneration (20 years); land-use mosaic

Data collection methods

Plot establishment

Field data collection began with the establishment of sampling units within each selected grid cell. Initially, the most suitable area for plot installation was identified, prioritizing locations that allowed safe access and adequately represented the vegetation. Areas considered unsuitable included sites with conditions that could compromise sampling feasibility or representativeness, such as waterlogged ground, excessively dense vegetation, physically hazardous locations, or portions of the grid cell where plots could not be properly established.

Once the final position of the plot had been defined, the four vertices were located using a GPS. The field team then navigated to each point to measure distances and install PVC pipes marking the corners of the main plot. This procedure was necessary because the position of the 50 m plot boundary could not always be predicted accurately under field conditions, which could otherwise result in the plot extending into unsuitable portions of the selected grid cell. After plot establishment, the boundaries of the main plot were marked with nylon string.

Within each main plot, two 50 × 1 m subplots were established for sampling the shrub and regenerating strata. These subplots were delimited with PVC pipes and nylon string and subdivided at 10 m intervals to improve field visibility. In addition, 1 × 1 m subplots were established for sampling the herbaceous stratum, using PVC frames installed exclusively on one side of the main plot. The sampling design adopted is illustrated in Figure 1.

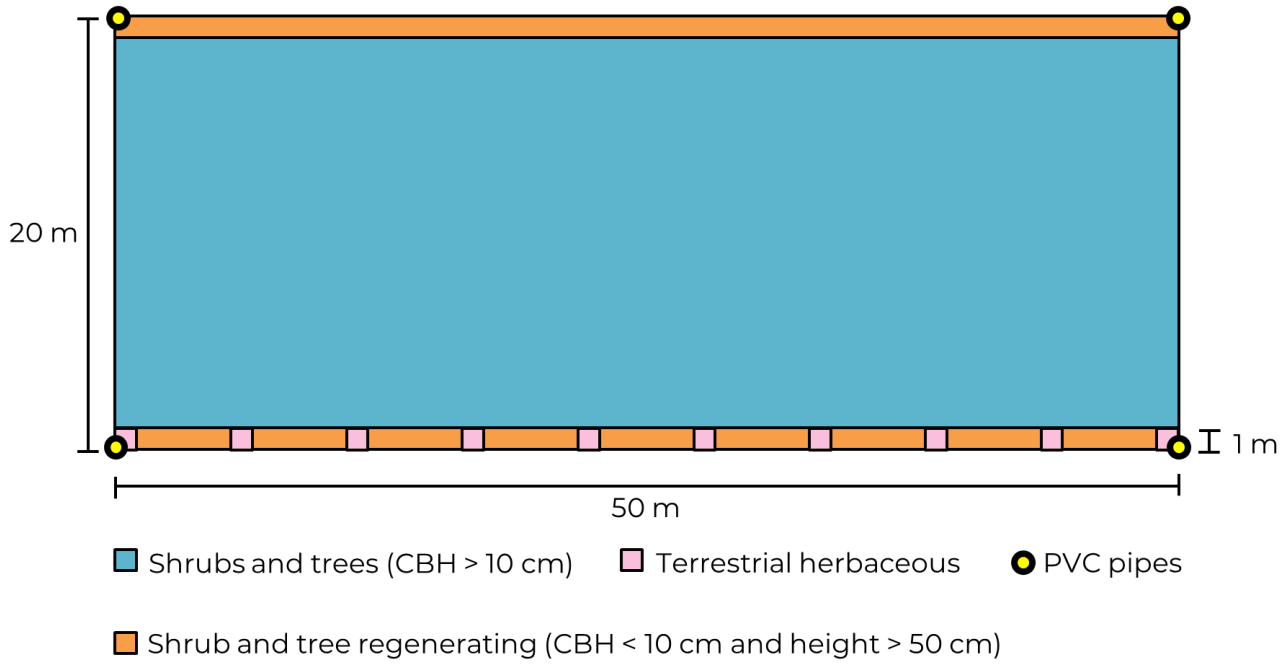

Figure 1. Layout of the sampling design, showing the main plot and nested subplots used to assess the arboreal, shrub, and herbaceous strata.

Vegetation sampling

Data were recorded using three field forms corresponding to the arboreal, shrub, and herbaceous strata. In the arboreal stratum, all individuals with circumference at breast height (CBH) > 10 cm and height > 1.3 m within the 20 × 50 m plot were identified to species, measured for CBH, had their total height estimated, and were permanently tagged. When bamboo clumps with multiple culms and CBH > 10 cm occurred within the plot, mean CBH and height were estimated from a subset of culms and extrapolated to the total number of culms counted.

In the shrub and regenerating strata, all individuals with CBH < 10 cm and height > 50 cm were recorded, including species identification and crown diameter, while bamboo individuals were quantified by counting culms per species. In the herbaceous stratum, the percentage cover of each herbaceous species was estimated within the 1 × 1 m subplots.

Field records and support materials

Field measurements were recorded using a mobile device to store plot coordinates and sampling data. A measuring tape was used to obtain CBH values in the arboreal stratum,

crown diameter in the shrub stratum, and species cover in the herbaceous subplots. Height measurements were assisted by a graduated pole, and a pruning pole was used to collect samples from taller individuals when necessary. Photographs were taken to support taxonomic identification. Arboreal individuals were marked with numbered aluminium tags fixed with nails and a hammer, while botanical samples were stored in collection bags and labelled with masking tape for subsequent identification and coding.

The sampling design, including plot configuration, evaluated life forms, and inclusion criteria for each vegetation stratum, is summarized in Table 3.

Table 3. Description of the sampling design applied to each vegetation stratum, including plot configuration, evaluated life forms, and inclusion criteria based on size thresholds.

Strata	Arboreal	Shrub/ regenerant	Herbaceous
Nr of plots/Plot size	1/ 50m x 20m	2/ 1m x 50m	10/1mx1m
Life forms evaluated	Trees, shrubs and others arborescent forms of life (palms, arborescent ferns, bamboos, and dracenoids)	Trees, shrubs and others arborescent forms of life (palms, arborescent ferns, bamboos, and dracenoids)	Herbs (Forbs, graminoids, ferns and non-woody climbers)
Inclusion criteria	CBH ≥ 10 cm/ h> 1.3m	CBH < 10 cm/ 1.3m >h> 50 cm	All herbs (including seedlings, where identification is possible)

Data analysis methods (incl. search of baseline data and selection of reference areas).

Definition of Reference Values

Baseline values for rare and threatened plant species (C6.4.)

Collection of baseline data

Although the monitoring framework was designed to rely on repeated measurements of permanent plots and their comparison over time, the project's short duration limited this approach. Therefore, existing historical monitoring data—more closely aligned with the sampling design used in this study—were used for illustrative purposes. Whenever possible, historical data collected in the same fragments assessed during the current project were used to maximize spatial comparability between baseline and current conditions. When no previous data were available for the exact same fragment, data from a nearby fragment within the same management unit were used as a comparative reference. In such cases, fragments were selected based on the best available information, giving preference to areas with the same vegetation type and the most similar conservation status possible.

An important methodological difference between the current and historical monitoring datasets concerns both sampling extent and spatial coverage. In the present project, the total sampled area was 4,000 m² in each management unit, whereas the historical datasets differed between units, totaling 8,000 m² in MU2 and 2,000 m² in MU3. These differences were maintained in the analyses in order to preserve the original sampling designs and allow comparison between monitoring approaches as they were effectively implemented, rather than artificially standardizing sampling effort. In addition, unlike the present project, the historical dataset for MU2 also included fragment edge areas, a difference in spatial coverage that should be considered when interpreting contrasts between baseline and current results.

Management Unit 1

For MU 1, no previous monitoring data were available. Therefore, no historical baseline could be established for this management unit, and the parameters required for temporal comparison were not obtained. As a result, it was not possible to make comparisons regarding the effectiveness of the methodology in assessing outcomes related to Criterion 6.4, since no previous monitoring information was available to support such analyses.

Management Unit 2

For MU 2, baseline data were obtained from previous monitoring campaigns conducted by the certificate holder within the same management unit. Historical information was prioritized for the same fragments assessed in the present study in order to maximize comparability between sampling periods. Previous data were available for three of the four fragments evaluated in the current project. For the fourth fragment, corresponding to plot 4 (restoration area), no historical data were available for the exact same site; therefore, data from a nearby fragment within the same management unit were used as a comparative reference. Although vegetation type was used as the main criterion for selecting the comparative fragment, equivalence in conservation status was not possible in this case.

The most recent available historical data, collected between 2022 and 2025, were used in the analyses. In this management unit, historical sampling was conducted using 10 × 10 m plots, with 2 × 5 m subplots established for the shrub and regenerating strata, totaling 8,000 m² of sampled area, compared to 4,000 m² in the present project. The inclusion criteria adopted were compatible with those used in the present project: in the arboreal stratum, all individuals with CBH ≥ 10 cm and height > 1.3 m were recorded, whereas in the shrub and regenerating strata, all woody individuals with 0.5 m < height < 1.3 m were recorded. In addition, unlike the present project, the historical dataset also included fragment edge areas,

which should be considered when interpreting differences between baseline and current results.

Management Unit 3

For MU 3, baseline data were also obtained from previous monitoring conducted by the certificate holder within the same management unit, prioritizing data from the same fragments sampled in the current project whenever available. However, historical data were available only for the same fragment corresponding to Plot 1. To allow comparisons with the other sampled plots, data from nearby areas within the same management unit were used as comparative references, prioritizing sites with the same vegetation type and the most similar conservation status possible, based on the best available information. This approach was not fully achievable for Plot 3 (flooded area) and Plot 4 (restoration area), as these plots were located in more degraded areas for which no historical data from comparable conservation conditions were available.

The most recent consolidated phytosociological data available for the management unit, collected in 2020, were used as the baseline. In this management unit, historical sampling was also conducted using 10 × 10 m plots, with 2 × 5 m subplots established for the shrub and regenerating strata, totaling 2,000 m² of sampled area, compared to 4,000 m² in the present project. As with the baseline used for comparison in MU 2, the inclusion criteria adopted were compatible with those used in the present project: in the arboreal stratum, all individuals with CBH ≥ 10 cm and height > 1.3 m were recorded, whereas in the shrub and regenerating strata, all woody individuals with 0.5 m < height < 1.3 m were recorded.

Calculation of baseline values

For the calculation of baseline values, the original raw data from previous monitoring campaigns were used. Taxonomic names were first reviewed and updated in order to ensure consistency with the nomenclature adopted in the present study. Based on these updated raw datasets, the same parameters evaluated in the current project were then calculated. The formulas used to calculate these parameters are provided in the Annexes.

Ecological reference conditions of native plant species (C6.5)

Selection of reference ecosystems

Reference ecosystems used for comparative analyses associated with Criterion 6.5 were selected through a bibliographic review, following the principle of best available information as recommended by the FSC framework. The review aimed to identify studies providing ecologically relevant benchmarks for comparison with the evaluated

management units and was based on a search for phytosociological studies in the scientific literature, prioritizing peer-reviewed articles, technical reports, and other reliable sources containing quantitative vegetation data, including theses, dissertations, and other academic works.

Geographic and ecological screening

The selection process began with the identification of studies conducted in areas geographically close to the evaluated management units. Geographic proximity was adopted as an initial screening criterion because it increases the likelihood of similarity in climatic conditions, soil characteristics, and regional species pools. When more than one study was available for a given region, preference was given to those located closest to the respective management unit.

After this initial screening, studies were further evaluated according to vegetation type. Only studies conducted in the same vegetation type as the assessed area were considered, in order to ensure ecological comparability between the evaluated management units and the candidate reference ecosystems. In cases involving ecotonal regions or transitional environments, a more detailed evaluation was conducted to identify the vegetation type with the greatest floristic similarity, prioritizing studies that best matched the dominant ecological characteristics of the assessed area.

Assessment of conservation status and ecological context

Candidate reference areas were then evaluated according to their conservation status. Priority was given to areas recognized as being in good conservation condition, such as protected areas. However, areas with evident signs of anthropogenic disturbance were avoided, even when legally protected. This evaluation was based on the descriptions provided in the studies, including information on land-use history, disturbance regimes, and species composition. In the absence of suitable protected areas, studies reporting vegetation parameters consistent with well-conserved ecosystems were selected, including species richness, Shannon diversity index, basal area, tree density, and the absence or low occurrence of exotic species.

Because these structural and diversity-related parameters vary according to vegetation type, the assessment of conservation status considered the expected ecological patterns for each ecosystem. For example, well-conserved Cerrado *sensu stricto* areas typically present lower values of height, basal area, density, and Shannon diversity than dense ombrophilous forests. Accordingly, the selection process accounted for these inherent

ecological differences and avoided direct comparisons based solely on absolute values across distinct vegetation types.

The ecological and historical context described in each study was also considered during the selection process. Additional information, such as past management practices, restoration activities, and other ecological conditions potentially affecting vegetation structure, was examined in order to ensure that the selected reference areas represented natural or near-natural conditions rather than systems substantially altered by recent or intensive human intervention.

Data suitability, methodological compatibility, and selection outcome

The final stage of selection considered whether the candidate studies provided the phytosociological parameters required for the analysis and whether their methodological descriptions were sufficiently clear to support comparison. This assessment included the availability of key parameters such as species richness, diversity indices, density, and dominance, as well as the clarity of the sampling design and inclusion criteria. Studies with sampling area, design, and inclusion criteria more similar to those adopted in the evaluated management units were prioritized in order to improve comparability between datasets. Only studies providing sufficient information to support consistent comparison were included.

In some regions, the number of available studies was limited, which restricted the pool of potential reference ecosystems. In such cases, the final selection was based on the best available studies, even when they did not fully meet all ideal criteria, and this condition was considered during result interpretation. It is also acknowledged that, in some situations, the evaluated management units may present structural or diversity parameters equal to or higher than those reported for the selected reference ecosystems. These cases were interpreted with caution, considering both the ecological context and the characteristics of the available reference data.

Based on this selection process, the reference ecosystems chosen to represent the most ecologically comparable and data-relevant benchmarks available for each management unit, supporting the evaluation of biodiversity patterns and ecological conditions in relation to Criterion 6.5.

Calculation of reference values

For the estimation of reference ecosystem values used in the comparisons required to assess Criterion 6.5, the methodological procedure initially consisted of recalculating all parameters directly from the original raw data in order to ensure full consistency with the

analytical procedures adopted in the present study. However, because raw datasets were not available for the selected reference ecosystems, this step could not be carried out. Consequently, the values reported in the selected reference studies were used as the comparative basis for the analyses.

Calculation of composite Indices for the Interpretation of Criteria 6.4 and 6.5

For each assessed area, the data were collected, tabulated, and used to calculate phytosociological parameters according to the formulas below (Mueller-Dombois & Ellenberg, 1974). The phytosociological formulas traditionally used in vegetation studies are provided in Annex 1.

Two synthetic indices were developed to facilitate the interpretation and visualization of the parameters associated with FSC Criteria 6.4 and 6.5. The Density Dynamics Difference Index (DDDI) was designed to summarize changes in the structural parameters of threatened and rare species (RTE species), integrating variations in density and dominance between monitoring periods. The Integrated Shannon–Species Evenness Difference Index (ISSED), in turn, was developed to support comparisons between management units and reference ecosystems by combining normalized differences in diversity and species richness. Together, these indices provide a simplified representation of multiple ecological parameters, supporting a clearer and more transparent interpretation of the progress towards the key intended outcomes under the selected criteria of the FSC Forest Stewardship Standard.

Density Dynamics Difference Index (DDDI) – Criterion 6.4

The Density Dynamics Difference Index (DDDI) was developed to synthesize changes in the structural parameters of threatened and rare species (RTE species) between monitoring periods. The index integrates normalized differences in absolute density and absolute dominance, two widely used phytosociological metrics that describe both the abundance and the structural contribution of species within plant communities (Mueller-Dombois & Ellenberg, 1974; Magurran & McGill, 2011). By combining these parameters into a single standardized value, the index facilitates the interpretation of population trends while preserving information on both demographic presence (density) and structural relevance (dominance). The use of normalized differences allows the index to remain scale-independent and comparable across monitoring cycles, reducing the influence of differences in sampling intensity or plot number. Similar normalization approaches are commonly applied in ecological monitoring and change detection metrics, where proportional change is preferred over absolute change to enable consistent comparisons across datasets and monitoring periods (Magurran & McGill, 2011). In this context, the DDDI

provides a practical way to summarize structural changes in RTE species populations while maintaining direct links to well-established phytosociological parameters.

$$\Delta DEN_{RTE} = \frac{DEN_{RTEcurrent} - DEN_{RTEprevious}}{DEN_{RTEcurrent} + DEN_{RTEprevious}}$$

$$\Delta DOM_{RTE} = \frac{DOM_{RTEcurrent} - DOM_{RTEprevious}}{DOM_{RTEcurrent} + DOM_{RTEprevious}}$$

$$DDDI = \frac{\Delta DEN_{RTE} + \Delta DOM_{RTE}}{2}$$

Where:

$DEN_{RTEcurrent}$ = density of RTE species in the current monitoring cycle

$DEN_{RTEprevious}$ = density of RTE species in the previous monitoring cycle

$DOM_{RTEcurrent}$ = dominance of RTE species in the current monitoring cycle

$DOM_{RTEprevious}$ = dominance of RTE species in the previous monitoring cycle

Integrated Shannon–Species Evenness Difference Index (ISSED) – Criterion 6.5

The Integrated Shannon–Species Evenness Difference Index (ISSED) was developed to support comparisons between management units and reference ecosystems in the evaluation of Criterion 6.5. The index combines normalized differences in species richness and Shannon diversity; two complementary biodiversity metrics widely used in vegetation studies. Species richness reflects the number of species present in the community, while the Shannon index incorporates both richness and the absolute density of species, capturing aspects of ecological dominance and evenness within the community (Magurran, 1988; Magurran & McGill, 2011). Because each metric captures different dimensions of biodiversity, their combined interpretation provides a more comprehensive assessment of community structure than either metric alone. The use of normalized differences allows the index to express the relative distance between the observed community and the reference ecosystem on a standardized scale, facilitating interpretation across different management units. This approach is consistent with ecological monitoring frameworks that rely on multiple biodiversity indicators to evaluate ecosystem condition and conservation outcomes, particularly when comparisons with reference conditions are required.

$$NSD = \frac{H'_{current} - H'_{reference}}{H'_{current} + H'_{reference}}$$

$$NRD = \frac{S_{current} - S_{reference}}{S_{current} + S_{reference}}$$

$$ISSED = \frac{NSD + NRD}{2}$$

Where:

H^'= Shannon diversity index;

S= species richness.

RESULTS

Criterion 6.4

The assessment of the key intended outcomes associated with Criterion 6.4 revealed variation among management units in both the richness and structural representation of rare, threatened, and endangered (RTE) species (Annex 3). MU2 showed the highest number of RTE species, with nine species recorded in the current sampling, whereas MU1 and MU3 recorded four and three species, respectively. In all management units, RTE species were restricted to the arboreal and shrub/regenerating strata, and no RTE species were recorded in the herbaceous stratum.

For MU1, four threatened species were recorded considering all strata, with three species in the arboreal stratum and three in the shrub/regenerating stratum. Because no previous monitoring data were available for this management unit, temporal comparisons were not possible, and the Density Dynamics Difference Index (DDDI) could not be calculated.

For MU2, the current and previous samplings recorded the same total number of RTE species (nine), although species composition differed between monitoring periods. In the arboreal stratum, current density was higher than in the previous dataset (105 vs. 78.75 individuals·ha^{-1}), whereas current dominance was lower (2.2705 vs. 6.7026). This resulted in a slightly negative DDDI (-0.1755), indicating a small reduction in the structural contribution of arboreal RTE species. According to the original monitoring records, this pattern was mainly associated with the non-detection of large Araucaria individuals in the current plots. In the shrub/regenerating stratum, both richness and density increased in the current sampling, from four to five RTE species and from 187.5 to 350 individuals·ha^{-1}. However, due to the absence of previous monitoring data on shrubs dominance, the DDDI value could not be calculated for this stratum.

For MU3, the number of RTE species recorded increased from one in the previous dataset to three in the current sampling. In the arboreal stratum, current density was slightly lower than the reference value (12.5 vs. 15 individuals·ha^{-1}), but current dominance was higher (0.4336 vs. 0.2087), producing a positive DDDI (0.1296). In the shrub/regenerating stratum, current richness and density were also higher, increasing from one to two RTE species and

from 50 to 125 individuals·ha^{-1}. However, due to the absence of previous monitoring data on shrubs dominance, the DDDI value could not be calculated for this stratum (Table 4).

Table 4. Comparative summary of key intended outcomes related to Criterion 6.4 for rare, threatened, and endangered (RTE) species in the three assessed management units, by vegetation stratum. Values from current monitoring (cur.) are presented alongside previous monitoring data (prev.), when available. RTE Species (R+T) = total number of rare and threatened species, with the number of rare (R) and threatened (T) species shown in parentheses; DEN RTE = absolute density of RTE species; DOM RTE = absolute dominance of RTE species; DDDI = Density Dynamics Difference Index. N/D = no data.

Management Unit	Stratum	Current monitoring			Previous monitoring (reference value)			DDDI
		Nr of RTE Species (R+T)	DEN RTE species	DOM RTE species	Nr of RTE Species (R+T)	DEN RTE species	DOM RTE species	
1	**Arboreal**	3 (3+0)	8	0.0505	N/D	N/D	N/D	N/D
	Shrub	3 (3+0)	100	98.702	N/D	N/D	N/D	N/D
	Herbaceous	0	N/D	N/D	N/D	N/D	N/D	N/D
2	**Arboreal**	9 (7+2)	105	22.705	9 (7+2)	78.75	67.026	-0.1755
	Shrub	5 (4+1)	350	1.301.209	4 (4+0)	187.5	N/D	N/D
	Herbaceous	0	N/D	N/D	N/D	N/D	N/D	N/D
3	**Arboreal**	2 (0+2)	12,5	0.4336	1 (0+1)	15	0.2087	0.1296
	Shrub	2 (1+1)	125	25.97	1 (0+1)	50	N/D	N/D
	Herbaceous	0	N/D	N/D	N/D	N/D	N/D	N/D

Overall, the comparative results indicate distinct patterns among management units. MU3 showed the most consistent positive response, with positive DDDI values in the arboreal stratum and an increase in RTE richness relative to the previous monitoring. MU2 maintained the same total number of RTE species and showed a slight negative trend in the arboreal stratum. For MU1, the absence of historical data precluded temporal assessment. Taken together, these results show that the methodology was effective in detecting RTE plant species and in capturing differences in their population structure among management units and between monitoring periods where comparable baseline data were available (Figure 2).

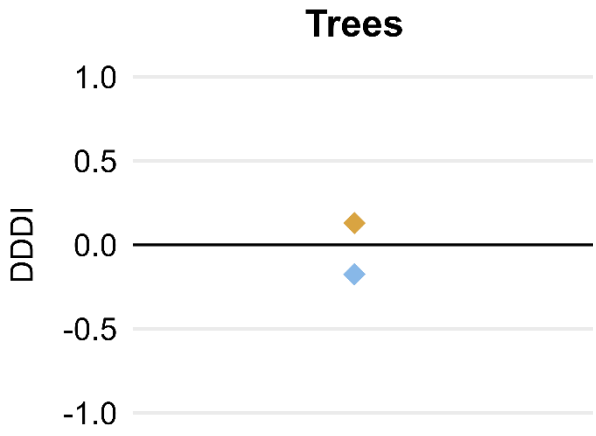

Figure 2. Values of the Density Dynamics Difference Index (DDDI) calculated for RTE species in the arboreal stratum.

Criterion 6.5.

The results obtained for Criterion 6.5 showed differences among management units in relation to their respective reference ecosystems (Annex 3). Comparisons with reference conditions were restricted to the arboreal stratum, since only for this stratum were comparable data on species richness and Shannon diversity available for all management units. For the shrub and herbaceous strata, the results are presented only as descriptors of current vegetation condition, providing a baseline for future monitoring.

Detailed information on the reference ecosystems used for each management unit is presented in Table 5, including the source study, vegetation type, geographic distance from the assessed management unit, vegetation stratum considered in the comparison, and the inclusion criteria adopted in each reference dataset.

Table 5. Main characteristics of the reference ecosystems used for comparison under Criterion 6.5 for each management unit, including source study, vegetation type, distance from the assessed area, evaluated stratum, and inclusion criteria adopted in the reference dataset.

	Management Unit 1	**Management Unit 2**	**Management Unit 3**
Reference Ecosystem	Parque Estadual do Turvo (Balbinot *et al.*, 2023)	Parque Estadual do Guartelá (Carmo & Assis, 2012)	Chapadão do Sul (Schardong *et al.*, 2020)
Vegetation type	Decidual Seasonal forest	Mixed Ombrophilous Dense Forest	*Stricto sensu* Cerrado
Distance	381 km	49 km	176 km
Strata assessed	Arboreal	Arboreal	Arboreal
Inclusion criteria	CBH > 15 cm	Height > 3 m	Diameter at ground level > 5

Within the arboreal stratum, MU1 showed the greatest divergence from the reference ecosystem. The management unit presented a Shannon index of 1.738, markedly lower than the 3.72 recorded for the reference area, indicating lower diversity and a less even distribution of individuals among species. This pattern is consistent with the lower ecological integrity observed for part of the vegetation remnants in the unit. In the shrub stratum, MU1 recorded 64 species and a Shannon index of 2.128, while the herbaceous stratum contained 37 species.

MU2 presented arboreal values below those of the reference ecosystem, but with a smaller difference than that observed for MU1. In this stratum, the management unit recorded 118 species and a Shannon index of 2.9975, compared to 140 species and 4.10 in the reference ecosystem. These values indicate an ecological condition closer to the benchmark than that observed for MU1, although still below reference levels. In the shrub stratum, MU2 recorded 90 species and a Shannon index of 2.865, while the herbaceous stratum contained 18 species.

MU3 showed the highest similarity to the reference ecosystem in the arboreal stratum. Species richness was practically equivalent between the management unit and the reference area, with 71 and 69 species, respectively. Shannon diversity, however, remained lower in the management unit, with a value of 2.25 compared to 3.33 in the reference, indicating that the main difference was related more to species distribution than to species richness itself. In the shrub stratum, MU3 recorded 70 species and a Shannon index of 2.5425, while the herbaceous stratum contained 8 species (Table 6).

Table 6. Comparative summary of biodiversity key intended outcomes related to Criterion 6.5 for the assessed management units, by vegetation stratum. Richness current = number of species recorded in the current monitoring; Richness reference = number of species recorded in the reference ecosystem; Shannon current = Shannon diversity index (H') recorded in the current monitoring; Shannon reference = Shannon diversity index (H') recorded in the reference ecosystem; ISSED = Integrated Shannon–Species Evenness Difference Index. N/D = no data.

Management Unit	Stratum	Current monitoring		Ecological reference conditions (reference value)		ISSED
		Richness current	Shannon current	Richness reference	Shannon reference	
1	**Arboreal**	45	1.738	37	3,72	-0.3300
	Shrub	64	2.128	N/D	N/D	N/D
	Herbaceous	37	N/D	N/D	N/D	N/D
2	**Arboreal**	118	29.975	140	4,1	-0.1203
	Shrub	90	2.865	N/D	N/D	N/D
	Herbaceous	18	N/D	N/D	N/D	N/D
3	**Arboreal**	71	2.25	69	3.33	-0.0923
	Shrub	70	25.425	N/D	N/D	N/D
	Herbaceous	8	N/D	N/D	N/D	N/D

The Integrated Shannon–Species Evenness Difference Index (ISSED), calculated for the arboreal stratum, reinforced this pattern. MU3 presented the value closest to zero (-0.0923), indicating the smallest divergence from the reference ecosystem. MU2 showed a very similar value (-0.1203), also indicating relatively high similarity, although slightly lower than that observed for MU3. In contrast, MU1 presented a markedly more negative value (-0.3300), indicating substantially greater dissimilarity from the benchmark ecosystem. These results show that MU2 and MU3 were more similar to each other than either was to MU1, with MU3 showing the best overall performance (Figure 3).

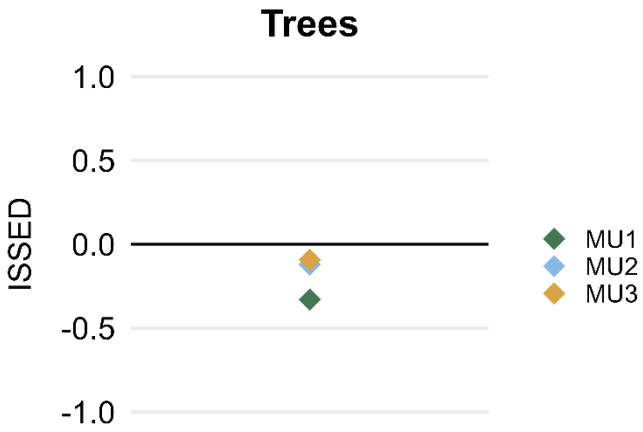

Figure 3. Values of Integrated Shannon–Species Evenness Difference index (ISSED) calculated for the management units.

Overall, the results indicate that the proposed methodology was effective in distinguishing management units according to their degree of similarity to reference ecosystems in the arboreal stratum, where direct comparison was possible. The greatest contrast was observed between MU1 and the other two management units, whereas MU2 and MU3 showed comparatively closer values, with a slight advantage for MU3.

DISCUSSION AND CONCLUSION

On the effectiveness of criteria 6.4.

Regarding the effectiveness of the proposed method in assessing the maintenance, conservation, or restoration of RTE species, it can be stated that, overall, the results were satisfactory, as the method was effective in recording these species and describing their respective population structures. Considering the two management units for which comparisons were possible (MU2 and MU3), species richness, as well as population density and dominance, were similar to or even higher than those obtained through the monitoring methodologies traditionally used by these companies, even though those approaches were based on greater sampling effort.

Since the sampled RTE species were recorded in the same strata evaluated in previous monitoring efforts, and the inclusion criteria were also consistent with those previously adopted, the only methodological difference in their sampling relates to the spatial distribution of the plots. One likely explanation for the satisfactory performance of the present methodology in detecting threatened species is the prior stratification used to select plot locations, particularly because this process favoured interior forest areas or core habitats. It is well established that rare and threatened species are often sensitive to disturbance; therefore, interior areas of forest fragments are more likely to harbour such species (Newton, 2007; Metzger, 2008). Conversely, the quantification of edge effects and their influence on

core areas is also highly relevant for the conservation of rare, disturbance-sensitive species, as well as for guiding management and restoration actions (Metzger, 2008).

Because the same plots could not be resampled, the results should be interpreted as illustrative rather than as direct evidence of ecological improvement or deterioration over time. This constraint, combined with the typically small population sizes of RTE species and the resulting sensitivity of the index to slight fluctuations in density, dominance, or composition, means that the patterns reported in the results section do not indicate better or worse ecological conditions. In MU2, for example, this variation was primarily associated with the non-detection of Araucaria angustifolia individuals in the current sampling, whereas the previous monitoring had recorded large Araucaria individuals derived from former cultivation. This effect is reinforced by the biogeographic pattern of tropical forests, where most tree species naturally occur at low densities, accounting for approximately 60–62% of the arboreal flora in both the Amazon (Ter Steege *et al.*, 2013) and the Atlantic Forest (Moreira *et al.*, 2024). In this context, indicators based on rare and low-density species are better interpreted as a tool for detecting potential ecological change when applied through the resampling of the same permanent plots over time.

In interpreting the effectiveness of the tested methodology for Criterion 6.4, sampling representativeness is an important consideration, since the assessment of temporal maintenance in RTE populations depends on adequately characterizing their richness and structural parameters under comparable sampling conditions. In this context, the tested methodology recorded substantially lower overall richness than the previous monitoring conducted by the Certificate Holder (118 vs. 198 species), suggesting lower sampling representativeness. This difference may be partially explained by the larger sampled area in the traditional monitoring, but also by its broader spatial distribution of plots within the fragment, which increases the likelihood of capturing species associated with a wider range of environmental conditions and microhabitats. Although the tested methodology recorded the same number of threatened species, the lower total richness indicates that this comparison should be interpreted with caution.

Another relevant aspect that may influence the interpretation of an ecosystem's conservation status is the heterogeneity of state-level threatened species lists, both in terms of their scope and the criteria adopted. In some states, official state-level lists are entirely absent, while in others, existing lists are more limited in scope or focus primarily on protected species rather than formally threatened taxa. In contrast, some states provide broader and more detailed assessments. Such discrepancies may distort the interpretation of the conservation relevance of specific forest fragments, particularly regarding their role as refuges for rare or threatened species. This heterogeneity may also help explain why MU3

yielded fewer RTE species records than MU1, despite MU1 comprising visibly more degraded and smaller fragments, with species associated with earlier successional stages.

Rather than reflecting a straightforward ecological contrast, this pattern likely stems, at least in part, from differences in the scope and robustness of state-level threat assessments. In addition, botanical knowledge remains unevenly distributed across Brazil, which may result in species from less-studied regions remaining underassessed due to limited occurrence and population data, even when they are rare or locally restricted. For example, Clethra scabra, recorded in MU1, is classified as Vulnerable in the state-level list despite its wide distribution, numerous records, and known tolerance to disturbed environments. In contrast, Aspidosperma nobilis, recorded in MU3, has far fewer records nationwide and within the state, yet it is not classified under any threat category, likely reflecting the absence of adequate population-level studies to support its formal assessment (SpeciesLink, 2026; Reflora, 2026; Bicalho *et al.*, 2021).

On the effectiveness of criteria 6.5.

Overall, the results indicate that the tested approach was effective in revealing how closely each management unit aligns with the intended outcomes associated with FSC Criterion 6.5, namely the maintenance or recovery of biodiversity conditions comparable to those found in reference ecosystems. By integrating richness and Shannon diversity into a single comparative signal, the methodology allowed a clear distinction among management units and made it possible to evaluate whether the observed vegetation condition was approaching the expected benchmark. In this sense, the approach was not only operationally useful, but also informative in relation to the ecological direction of each unit.

The results suggest that none of the assessed management units has yet reached biodiversity conditions equivalent to those of the selected reference ecosystems, since all ISSED values remained below zero. Because it is difficult to identify reference areas that are simultaneously comparable in vegetation type, geographic proximity, conservation status, and sampling methodology, these reference ecosystems should be interpreted as aspirational comparative benchmarks. Nevertheless, the ISSED adequately reflected the pattern observed in the field, with MU1 showing markedly lower conservation status and smaller fragment size than MU2 and MU3. This interpretation is reinforced by the Shannon values obtained for the arboreal stratum; MU1 showed particularly low diversity, consistent with its more degraded condition and smaller fragment size, and with values considered low when compared with phytosociological studies conducted in the state (Milanesi & Leite, 2014; Scipioni *et al.*, 2011; Balbinot *et al.*, 2016). MU2 showed a comparatively higher arboreal Shannon value, with values more consistent with better-preserved areas, although still below

those reported in studies carried out in the same state. (Dias *et al.*, 1998; Cordeiro & Rodrigues, 2007; Carmo & Assis, 2012). MU3, although intermediate, also remained below values reported for Cerrado *sensu stricto* (Schardong *et al.*, 2020; Lehn *et al.*, 2008).

Although the Shannon values obtained for the management units were lower than those of the reference ecosystems, species richness showed a different pattern. In MU1 and MU3, arboreal richness was higher than that recorded for the respective reference ecosystems, whereas in MU2 it was slightly lower. This contrast is explained by the sampling configuration, since richness was compiled across distinct fragments, whereas Shannon diversity was calculated separately for each fragment and then averaged, resulting in lower mean diversity values. In some cases, the evaluation combined distinct fragments represented by relatively small sampled areas, and the final comparison was based on the mean of these values. As a result, local spatial variation in species composition and absolute density was not fully captured, which likely led to conservative Shannon estimates for the management units (Magurran & McGill, 2011; Dušek & Popelková, 2017).

In management units composed of spatially separated fragments, a limited number of plots in each fragment may fail to capture the internal heterogeneity of the vegetation, particularly in tropical systems where diversity patterns are strongly influenced by small-scale spatial variation. If sampling effort had been greater and more evenly distributed within each fragment, the resulting Shannon values would likely have been higher, allowing a more precise comparison with reference ecosystems. Therefore, although the index was effective in identifying the relative position of the management units along a gradient of similarity to reference conditions, the absolute values obtained should be interpreted as conservative estimates of ecological similarity.

The results obtained for Criterion 6.5 suggest that the index adequately captured the ecological differences among management units, with MU2 and MU3 closer to the reference ecosystems than MU1. At the same time, the diversity components underlying this pattern should be interpreted carefully. Shannon values were likely conservative, as they were derived from multiple fragments represented by limited sampled area within each fragment, which may have reduced the ability to capture local variation in species composition and abundance. In contrast, species richness may have been inflated by compiling records from distinct fragments that differed in vegetation characteristics and were often spatially distant. Given the well-established importance of rare species in shaping diversity patterns in tropical plant communities (Ter Steege *et al.*, 2013; Moreira *et al.*, 2024), a larger sampled area within each fragment would likely provide more precise Shannon estimates and, consequently, a more appropriate comparison of ecological similarity to the reference ecosystems. In this sense, the negative values obtained in the present study likely

represent a conservative estimate of proximity to reference conditions, and broader within-fragment sampling could result in higher diversity values and a more robust assessment of Criterion 6.5.

Taken together, these findings indicate that the intended outcomes of Criterion 6.5 were met only partially across the assessed management units. The results do not support an interpretation of full equivalence to reference ecosystems, but they do indicate a gradient of progress, in which some certified units appear substantially closer to the desired ecological condition than others.

On the monitoring framework/methodological limitations

Plot distribution

From a statistical perspective, sampling based on randomly allocated plots is often regarded as the most appropriate approach for estimating population parameters, as it assumes random distributions of individuals (Felfili *et al.*, 2011; Batista *et al.*, 2014). However, under practical field conditions, full randomness is rarely achieved, and partially random or even subjective procedures are commonly adopted (Krebs, 1999). Moreover, each sampling strategy may perform better in answering specific ecological questions. Random allocation may better represent species richness in smaller sampled areas, whereas systematic sampling may perform better in larger sampled areas (Dalla-Côrte *et al.*, 2013). In addition, systematic allocation is particularly useful for recognizing ecological patterns and conservation-relevant attributes across heterogeneous landscapes.

For plots to adequately represent richness and diversity, sampling should be conducted in environmentally and floristically comparable areas, considering ecological conditions and floristic composition (Kersten & Galvão, 2011). In this context, systematic sampling may provide estimates comparable to those obtained through random sampling, provided that it is properly planned and consistent with the environmental heterogeneity of the study area (Krebs, 1999; Kersten & Galvão, 2011; Moro & Martins, 2011; Dalla-Côrte *et al.*, 2013; Magnusson *et al.*, 2013). Species rarefaction curves may also support decisions on sampling sufficiency, representativeness, and monitoring cost-effectiveness (Kersten & Galvão, 2011).

Plot configuration

Plot configurations adjusted to the vegetation type and operational conditions tend to provide more consistent results while also improving the practical feasibility of the method. Rectangular plots measuring 50 × 20 m (0.1 ha) are widely used and recommended for Cerrado formations, as they facilitate the structural characterization of vegetation in

environments with greater horizontal heterogeneity and more widely spaced individuals (Felfili *et al.*, 2011). This plot configuration is consistent with guidelines adopted in regulatory frameworks and inventory protocols for environmental licensing in some Brazilian states where Cerrado is the main vegetation type (SEMAC, 2008). In contrast, the application of 20 × 50 m plots in Atlantic Forest areas may impose relevant practical limitations. Rugged topography, dense and tangled vegetation, and the difficulty of anticipating where plot vertices will ultimately fall raise the likelihood of incorporating undesirable elements, such as watercourses, rocky outcrops, anthropogenic disturbances, or adjacent land uses, increasing installation time and operational efforts. This issue is particularly frequent because many vegetation remnants persist in areas of low agricultural suitability, including steep slopes, stony terrain, and riparian zones.

Plot size should be defined to ensure a minimum sampling effort capable of including at least 20–30 individuals, with species rarefaction curves used to assess representativeness and guide monitoring cost-effectiveness (IBGE, 1992; Felfili, 2005; Kersten & Galvão, 2011). Under these conditions, smaller and more easily replicable designs, such as 10 × 10 m plots, are often adopted to facilitate field implementation and enhance spatial replication. Assessing species richness generally benefits from a larger number of smaller and spatially independent plots rather than a smaller number of larger plots, as this reduces the concentration of sampling effort in a single continuous area, decreases spatial autocorrelation, and improves the representation of local heterogeneity, particularly in small fragments, heterogeneous management units, or landscapes with strong edge effects and patchy disturbance histories (Felfili *et al.*, 2005, 2011; Magurran & McGill, 2011). Applied studies in Atlantic Forest ecosystems have also demonstrated that 10 × 10 m sampling units perform well in representing arboreal and regenerating components, while improving sampling efficiency and reducing measurement costs (Moreira *et al.*, 2024; Silva *et al.*, 2020).

Sample efforts

With respect to sampling effort, the specialized literature converges in indicating that phytosociological surveys conducted in fragments of 100 hectares or more should aim for a sampled area close to 1 hectare to ensure adequate representativeness of the plant community (Moro & Martins, 2011; Newton, 2007; Kageyama *et al.*, 2021; Moreira *et al.*, 2024). In forest fragments smaller than 100 ha, sampling effort should be guided less by fixed area thresholds and more by representativeness and sampling sufficiency. In such contexts, increasing the proportional area sampled and ensuring adequate spatial coverage across internal environmental gradients become critical to avoid biased characterizations of the community (Felfili *et al.*, 2011). The use of species accumulation curves and rarefaction analyses is therefore recommended to assess whether additional plots contribute

meaningfully to species richness and structural parameters, allowing the identification of an effort at which estimates stabilize.

Complementarily, it is recommended that the total number of sampled individuals exceed 1,000, particularly when the objective is to estimate species richness and diversity (Magurran & McGill, 2011; Condit & Colwell, 2011; Kersten & Galvão, 2011). This recommendation is justified by the fact that rare species may occur at extremely low densities, sometimes with only one adult individual per 50 or even 100 hectares (Kageyama *et al.*, 2011). Although such levels of sampling effort are not always achievable under operational constraints, approaching these benchmarks helps reduce biases associated with the low density of rare species, which may represent a substantial proportion of the flora (Moreira *et al.*, 2024), and minimizes errors arising from the extrapolation of population density estimates. The interpretation of the results associated with Criterion 6.4 should also consider the absence of historical data across most management units, which limits the ability to assess whether the observed values reflect temporal changes in the populations of RTE species or represent baseline conditions.

As discussed previously, the index developed to assess Criterion 6.4 is highly sensitive, since small changes in the number or size of threatened-species individuals can lead to pronounced variations in its value. In this context, estimates derived from larger sampling areas and broader sample sizes tend to be more stable and reliable, as they reduce the influence of local or punctual events and increase the likelihood of capturing the actual distribution and structure of the populations under assessment. Consequently, increasing sampling effort contributes to a more representative depiction of the community, minimizing interpretations that are disproportionately influenced by fluctuations associated with limited sampling.

Inclusion thresholds

The tree inclusion criterion directly influences operational costs, comparability among studies, and the stability of phytosociological estimates. The threshold of DBH ≥ 5 cm (CBH ≈ 15.7 cm) is widely adopted in phytosociological surveys and facilitates comparisons across datasets. In this sense, increasing the minimum CBH relative to that adopted may represent a viable strategy to reduce field time and measurement costs, especially when the primary objective is to generate robust and comparable indicators at the management-unit scale, without the need to include very small individuals.

For the shrub/regenerating stratum, estimating dominance based on crown diameter is a methodological choice that tends to introduce high uncertainty. Crown architecture is highly plastic and responds to light availability, competition, water availability, leaf loss, and

crown overlap among individuals, which makes this measurement less reproducible and more sensitive to local variation. In addition, because this component rarely allows the remeasurement of the same individuals over time (in the absence of systematic individual tagging), the effort required to obtain comparable data across monitoring campaigns increases, and observed differences may reflect measurement noise rather than real structural changes. When temporal comparison is a priority, stem-based metrics tend to provide more consistent and reliable results. However, for Criterion 6.5, the limitations affecting the shrub and herbaceous strata are even broader, since the index could not be applied to these components due to the absence of compatible reference datasets, and the selection of appropriate reference ecosystems remains a major methodological challenge. In many cases, well-preserved sites that are geographically close and physiognomically similar do not provide sufficiently detailed phytosociological information. Moreover, when reference areas do not adequately represent near-optimal ecological conditions, the conservation status of the management unit may be misinterpreted, highlighting the need for more comprehensive and well-documented reference ecosystems to support reliable comparisons.

Although the inclusion of the herbaceous stratum represents a positive addition, given that this component has historically been underrepresented in vegetation monitoring surveys, its application in certain habitats, especially closed-canopy forests, may require substantial field effort with limited analytical return. This is particularly evident in environments where herbaceous occurrence or cover is low and strongly constrained by litter accumulation and shading. In such cases, more appropriate alternatives include conducting opportunistic floristic surveys in specific habitats or applying this methodology only in vegetation types where herbaceous species are more representative and contribute more meaningfully to ecological interpretation.

Diversity index

Computing the Shannon index for an entire management unit can be conceptually inappropriate, as the index is fundamentally a measure of diversity for a defined community sample or landscape. When heterogeneous fragments or habitats are pooled, richness signals may be artificially inflated and differences among communities (alpha and beta components) can be obscured (Dušek & Popelková, 2017), potentially leading to misleading summaries at the management-unit scale. In cases where a management unit includes spatially separated fragments, it is therefore more appropriate to calculate the Shannon index individually for each sampled fragment and then consider the mean value across fragments, rather than computing a single index for the entire management unit. Pooling all records into a single calculation would tend to overestimate diversity values. Despite these

caveats, the index proved useful as a screening indicator rather than a standalone diagnostic tool, as it consistently reflected the observed gap between management units and their respective reference ecosystems while supporting transparent and auditable interpretation for Criterion 6.5.

Additionally, the index developed to evaluate outcomes related to Criterion 6.5 also presents limitations when calculated from small sample sizes, as it integrates species richness and Shannon diversity index, ecological parameters that are sensitive to sampling effort. In reduced sampling sets, local variation in floristic composition or species abundances may disproportionately influence the final index value, making it difficult to distinguish consistent structural patterns from localized sampling effects. Considering the well-established importance of rare species in shaping species composition and diversity patterns in tropical forest communities (Ter Steege *et al.*, 2013; Moreira *et al.*, 2024), estimates based on larger sampled areas and a greater number of plots tend to provide more robust results, as they reduce the influence of local heterogeneity and improve the representativeness of the plant community as a whole. Consequently, index values derived from restricted sampling efforts should be interpreted considering these limitations, particularly when used for comparisons among management units or against reference ecosystems.

RECOMMENDATIONS

To FSC International on the deployment of outcome orientation

Composite indices (DDDI and ISSED) should be used as screening tools and always interpreted together with their underlying ecological variables

Synthetic indices that combine multiple ecological metrics, such as those developed in this study (DDDI and ISSED), require explicit methodological framing to guide their interpretation and appropriate use. These indices should be understood as integrative indicators designed to facilitate the visualization of general trends in vegetation structure and biodiversity, rather than as definitive diagnostic measures. Their interpretation should always be accompanied by the analysis of the underlying parameters used in their calculation, including species richness, absolute density, absolute dominance, Shannon diversity, and the environmental conditions described for each area. If index values suggest significant changes in threatened-species populations or biodiversity, additional phytosociological parameters, such as basal area, density, and species richness, should be evaluated alongside technical reports to provide a more comprehensive understanding of the actual condition of the area and to determine whether management actions are needed

to maintain its ecological integrity. Evaluating these parameters separately provides a more precise understanding of the ecological processes influencing the vegetation, as different combinations of these variables may generate similar index values while reflecting distinct ecological contexts.

Shannon diversity must be interpreted considering sampling effort to avoid underestimating biodiversity patterns

The Shannon index is sensitive to sampling effort; surveys with a low number of plots or limited representativeness per fragment tend to underestimate diversity. Its use in performance assessments should explicitly consider sampling intensity and analytical scale to avoid misinterpretation of biodiversity patterns. In practice, this means verifying whether the sampling effort adequately represents the local community. Species rarefaction analyses, for example, can help assess whether the observed diversity reflects the underlying ecological reality or whether it may be biased by insufficient sampling. When sampling effort is limited, the index should be interpreted cautiously, particularly when making comparisons with reference ecosystems or studies based on more robust sampling designs.

Faunal monitoring should complement vegetation assessments to strengthen biodiversity outcome evaluation

It is recommended that an applicability test be carried out for fauna, like the one conducted for vegetation, in order to assess how fauna monitoring responds in practice to Criteria 6.4 and 6.5, and how it can complement the information obtained from vegetation studies. Although vegetation indicates habitat structure and conservation status, it is not sufficient to confirm the presence and viability of animal populations, potentially masking the phenomenon of forest areas that appear conserved but have already lost part of their fauna. Including faunal groups such as birds and mammals in monitoring makes it possible to detect these silent losses of biodiversity, offering a more faithful picture of the ecological integrity of the management unit and strengthening the assessment of the intended outcomes under FSC Certification Criteria 6.4 and 6.5. Additionally, these groups harbor a considerable number of rare and threatened species (RTE), and their monitoring relies on well-established and widely applied methods, which favors data standardization and comparability across different units and over time.

To FSC Brazil and the Working Group on the revision of the FSS for plantations for Brazil

Reference Ecosystem Selection for Criterion 6.5

The selection of reference ecosystems should prioritize areas with high conservation status, floristic similarity, and sufficient data for analysis. When no area meets all three criteria, areas that meet the key ecological requirements should still be considered, even if raw data are unavailable or some of the parameters required for comparison are missing. In such cases, targeted sampling in the reference ecosystem can fill these gaps and support the comparative analysis with the same parameters and criteria.

This approach expands the range of potential reference areas and reduces dependence on pre-existing datasets, which are often scarce or inconsistent. Complementary sampling in ecologically representative remnants can strengthen the comparison and ensure that Criterion 6.5 analyses are based on current and methodologically consistent data. In addition, the comparison of structural parameters such as density and dominance may also be useful for assessing successional stage and further qualifying the ecological interpretation of the reference ecosystems.

Reference Ecosystem Continuity Across Monitoring Cycles

Consistency in the choice of reference ecosystem across monitoring cycles is essential to ensure temporal comparability of indicators. Once a reference area that meets the required ecological and analytical criteria has been selected, it is recommended that the same unit be retained for all subsequent Criterion 6.5 assessments, avoiding substitutions that may introduce variation unrelated to the management unit. In this way, any changes in results can be interpreted with greater confidence, distinguishing actual changes within the management unit from noise generated by shifting the reference ecosystem.

Threatened-species metrics should be interpreted cautiously as indicators, not as evidence of ecological change

The indices associated with Criterion 6.4, including DDDI, as well as the density and dominance of RTE species, should be interpreted as early-warning indicators rather than definitive diagnostic measures. Extreme values or abrupt variations should prompt further investigation and complementary analyses, rather than direct inferences regarding conservation loss or improvement. Due to their sensitivity to small changes, particularly under limited sampling effort, and considering the typically low densities of threatened species, minor variations in the number or size of individuals may result in substantial

fluctuations in index values. Accordingly, these results should be interpreted in conjunction with other ecological parameters, such as species richness, diversity, dominance, and overall vegetation structure. This integrated approach supports a more robust interpretation by distinguishing natural sampling variability from potential declines in environmental quality, particularly in the context of threatened species persistence.

Sampling design should consider vegetation type to adequately reflect local structural complexity

Vegetation types differ substantially in structure, density, stratification, and ecological dynamics, requiring context-specific methodological approaches. Large plots configurations, commonly used in Cerrado *sensu stricto* (Felfili *et al.*, 2005; SEMAC, 2008), rocky fields (Moura et al, 2011), and Amazonian formations (Condit, 1998), such as 50 × 20 m rectangular plots, may be inefficient in dense Atlantic Forest environments (Felfili *et al.*, 2011; Silva *et al.*, 2020; Moreira *et al.*, 2024), Pantanal (Damasceno-Junior & Pott, 2011), and other environments, where rugged terrain, flooded areas, high stem density, dense understory and others complicate plot establishment. The size of the sampling unit and the inclusion criteria should be consistent with the floristic structure of the community, being capable of representing the main life forms and the overall diversity of the environment; therefore, a plot size that includes at least 30 individuals, captures spatial variation in vegetation, and is aligned with the life forms under evaluation is recommended (Felfili *et al.* 2005; Moreira *et al.* 2024).

In structurally complex and heterogeneous tropical forests such as the Atlantic Forest, the use of multiple smaller sampling units distributed throughout the fragment can improve the representativeness of phytosociological surveys, as smaller plots allow greater spatial replication and greater degrees of freedom, improving statistical robustness and increasing the likelihood of capturing local environmental variation, which is particularly important in ecosystems where species composition may change over short distances (Felfili *et al.*, 2011; Moreira *et al.*, 2024). In addition, increasing the number of sampling units tends to improve the probability of detecting species with low local density or restricted spatial distribution, a common characteristic of tropical tree communities (Kersten & Galvão, 2011). From an operational perspective, the use of smaller and spatially distributed plots may also be more economically viable, as it can improve the coverage of floristic variation and generate higher-quality data when the sampling design would otherwise recommend only a single plot, while also reducing costs when many sampling units are required. In Atlantic Forest fragments, smaller plots have been shown to provide adequate sampling efficiency with lower operational costs, reinforcing their applicability in phytosociological surveys under complex field conditions (Silva *et al.*, 2020).

To local experts on the design of biodiversity monitoring frameworks for FSC certified forest plantations in Brazil

Systematic plot allocation is often more suitable than strict randomization in heterogeneous environments and change-detection studies

Although random sampling is often statistically supported, full randomness is rarely achieved under field conditions and may result in plots that are unrepresentative or operationally unfeasible. Random sampling is recommended in environments where populations are homogeneous across the landscape or where the area can be subdivided into homogeneous strata. However, in areas with strong environmental gradients or when the objective is to detect environmental changes, systematic sampling approaches are more appropriate (Quinn & Keough, 2002). Geospatial tools can support the preliminary selection of sampling points, but the choice of sampling strategy and the final placement of plots should remain flexible to ensure ecological representativeness and field feasibility.

Sampling effort should be allocated among multiple plots to improve the representation of internal environmental variation

When the sampling design results in the implementation of a small number of plots, concentrating effort in a few large units may lead to overrepresentation of local conditions. Distributing the same total sampled area across multiple smaller plots can improve spatial representation and better capture internal heterogeneity, reducing spatial autocorrelation and minimizing the influence of localized or punctual events that may distort the true ecological conditions of the fragment. In some areas, especially those with high environmental heterogeneity, subdividing a larger plot into smaller units may increase spatial representativeness without increasing the total sampling effort.

To improve richness estimates and enhance RTE species detection, complementary floristic surveys should be conducted alongside fixed plots

Fixed-area plots provide robust structural estimates but may underestimate total species richness, particularly for rare or low-density species. Rapid floristic assessments, standardized walk-through surveys, or active searches for threatened species can help determine whether recorded richness adequately represents the management unit as a whole. This complementary approach allows distinction between genuinely low richness and insufficient sampling effort, strengthening ecological interpretation without relying exclusively on fixed plots.

The use of CBH ≥ 15 cm as an inclusion threshold enhances comparability among studies

The adoption of widely recognized inclusion thresholds, such as CBH ≥ 15 cm, facilitates comparisons across studies and monitoring cycles. Methodological modifications should be carefully evaluated regarding their impact on historical comparability, especially when the objective is to assess temporal trends or evaluate conservation-related criteria. Consistency in sampling protocols enhances the reliability of longitudinal analyses and supports more robust decision-making frameworks.

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