Revista Brasileira de Ciências Ambientais
Brazilian Journal of Environmental Sciences

Spatial-temporal analysis of the landscape structure surrounding Cerrado conservation units without a buffer zone

Análise espaçotemporal da estrutura da paisagem no entorno de unidades de conservação do Cerrado sem zona de amortecimento

Ewilyn dos Santos Torquato[1], *Karine Borges Machado*[2], *Roniel Freitas-Oliveira*[3], *Patrick Thomaz de Aquino Martins*[4]

ABSTRACT

This study evaluated the landscape structure surrounding strictly protected conservation units (CUs) in the Cerrado biome that lack designated buffer zones, in comparison with landscapes outside these units across spatial and temporal scales. For the spatial analysis, 18 fully encompassed Cerrado CUs and 18 randomly selected external points within the biome were assessed for the year 2022. For temporal analysis, 9 CUs and 9 random points were analyzed for the years 2010 and 2022. In all cases, 10-km buffers were delineated, and land use as well as land cover maps were generated. Landscape structure was quantified using percentage of class (PLAND), edge density (ED), and effective mesh size (MESH) metrics. Comparisons within groups were performed using the Mann-Whitney, Student's t, and Wilcoxon tests. In the spatial analysis, areas surrounding CUs exhibited a low proportion of Cerrado cover (median≈29.8%) and did not differ significantly from external areas in PLAND or MESH. However, they presented significantly higher ED values, indicating greater fragmentation and increased exposure to edge effects around the CUs. In the temporal analysis, PLAND and MESH did not differ significantly between 2010 and 2022, whereas ED increased across all landscapes, reflecting the intensified fragmentation occurring near the CUs over time. These results demonstrate that the CUs are embedded in landscape contexts that are structurally unfavorable for conservation, and suggest that the absence of buffer zones, coupled with weakened legal instruments, may perpetuate and aggravate this situation.

Keywords: landscape metrics; protected area; fragmentation; spatial analysis; multitemporal analysis.

RESUMO

Este estudo avaliou a estrutura da paisagem no entorno de Unidades de Conservação (CU) de proteção integral do Cerrado sem zona de amortecimento, comparando-a com a estrutura da paisagem em áreas externas a essas unidades, num contexto espacial e temporal. Para a análise espacial, foram selecionadas 18 CU integralmente inseridas no Cerrado e 18 pontos externos a elas, distribuídos aleatoriamente pelo bioma no ano de 2022. Para a análise temporal, foram selecionadas nove CU e nove pontos aleatórios nos anos de 2010 e 2022. Para ambos os grupos foram delimitados *buffers* de 10 km, e foram gerados mapas de uso e cobertura da terra. A estrutura da paisagem foi quantificada pelas métricas de porcentagem da classe (PLAND), densidade de borda (ED) e tamanho efetivo da malha (MESH), comparadas dentro dos grupos por meio dos testes de Mann-Whitney, *t* de Student e Wilcoxon. No recorte espacial, o entorno das CU apresentou baixa proporção de Cerrado (mediana≈29,8%) e não diferiu significativamente de áreas externas em PLAND e MESH, mas exibiu ED significativamente maior, indicando maior fragmentação e exposição a efeitos de borda ao redor das CU. Na análise temporal, PLAND e MESH não apresentaram diferenças significativas entre 2010 e 2022, enquanto ED aumentou em todas as paisagens, evidenciando a intensificação da fragmentação no entorno das CU ao longo do período. Os resultados mostram que essas CU estão inseridas em contextos paisagísticos estruturalmente desfavoráveis à conservação e sugerem que a ausência de zona de amortecimento, associada ao enfraquecimento dos instrumentos legais, pode contribuir para a persistência e agravamento desse quadro.

Palavras-chave: métricas da paisagem; área protegida; fragmentação; análise espacial; análise multitemporal.

[1]Universidade Estadual de Goiás – Anápolis (GO), Brazil.
[2]Universidade Estadual de Goiás – Porangatu (GO), Brazil.
[3]Universidade Federal de Jataí – Jataí (GO), Brazil.
[4]Universidade Estadual de Goiás – Formosa (GO), Brazil.
Corresponding author: Patrick Thomaz de Aquino Martins – Av. Universitária, s/n – Nordeste – CEP: 73807-250 – Formosa (GO), Brazil.
E-mail: patrick.martins@ueg.br
Received on: 05/08/2025. Accepted on: 03/16/2026.
https://doi.org/10.5327/Z2176-94782588

Introduction

The intensification of anthropogenic pressures on natural ecosystems has led to unprecedented habitat loss and biodiversity decline on a global scale (Cheng et al., 2025). Land use and land cover changes, such as the conversion of natural vegetation to cropland and pasture, are among the primary drivers of this transformation, altering landscape structure, undermining ecosystem functioning, and reducing service provision (Jaureguiberry et al., 2022).

Within this context, the establishment of representative systems of protected areas is recognized as one of the pillars for curbing habitat loss. Protected areas play a central role in contemporary conservation strategies by combining the safeguarding of biodiversity with the preservation of ecosystem services and associated sociocultural values (Li et al., 2024). In Brazil, where land use and land cover have undergone significant changes, the consolidation and effective implementation of the protected area network have been identified as critical for curbing deforestation and habitat fragmentation (Strassburg et al., 2017).

Brazil holds the world's largest national network of terrestrial protected areas, encompassing approximately 2.47 million km^2 (UNEP-WCMC; ICUN, 2016), composed primarily of conservation units (CUs) (MMA, 2023). These CUs are classified into 12 categories, grouped into sustainable use units, where biodiversity conservation occurs in tandem with resource use within capacity limits (Silva et al., 2021), and strict protection units, which aim to maintain ecosystems free of human-induced changes, permitting only indirect use of natural attributes (Teixeira et al., 2021).

The presence, expansion, and intensification of anthropogenic land use near CUs can cause changes in biodiversity and ecological functions (Hansen and DeFries, 2007). It is common to establish buffer zones (BZs) around CUs to reduce or eliminate the harmful effects of human activities on biodiversity and the environments these areas protect (Guerra et al., 2025). In Brazil, these BZs were legally established in 1990 and termed the surrounding area (Vitalli et al., 2009), which encompasses a 10-kilometer radius around the CUs, where licensing is required for any activity that may impact the biota (Brasil, 1990).

Subsequently, with the enactment of the National System of Nature Conservation Units (SNCU) outlined by Federal Law No. 9,985/2000 (Brasil, 2000), the agency managing each CU was mandated to define the boundaries and specific regulations for land use and resource exploitation in the surrounding areas of CUs. This area (i.e., the BZ) became mandatory for all CU categories except for Environmental Protection Areas and Private Natural Heritage Reserves, both of which are designated for sustainable use. The BZ may be defined at the time of CU designation or thereafter and must be incorporated into its Management Plan, a technical document specifying the zoning, regulations, and guidelines for CU use and management (Brasil, 2000).

Paradoxically, over two decades after Law no. 9,985/2000 was enacted, most of Brazil's CUs still lack a Management Plan (MMA, 2023) and therefore remain without the regularization of their BZs. This situation was aggravated by Resolution no. 428/2010 (Brasil, 2010), which abolished the previously applicable surrounding area for CUs lacking management plans and, as of 2020 (Brasil, 2015), removed additional safeguards established by that resolution. The first mechanism to be revoked was the requirement to manage agency authorization for projects that require Environmental Impact Assessments and Environmental Impact Reports within 3 km of CUs without a BZ. The second pertained to the requirement to notify the managing agency in cases where Environmental Impact Assessments and Environmental Impact Reports were not required, within a 2 km radius of the CU (Guimarães et al., 2012). This 2 km extent is where the most severe conflict between development and conservation occurs, when anthropogenic changes in land use and land cover are observed (Li et al., 2024). As a result, CUs without a BZ went from having a legal guarantee of 10 km (surrounding area) in 1990 to a condition in which they lack any legal instrument to question or impose conditions on anthropic activities in their surroundings (Guimarães et al., 2012). This situation enables potentially impactful anthropogenic activities in their surroundings and, consequently, makes CUs without a BZ more susceptible to these impacts, especially in contexts of conversion of natural cover to anthropogenic uses.

Over recent decades, the Cerrado has undergone a transformation in which its natural cover has given way primarily to agribusiness-related activities (Souza Junior et al., 2020; Yang et al., 2025), and in recent years it has been the largest deforested area in the Brazilian biome (Del Lama et al., 2025). These factors demonstrate the fragility of this biome in terms of biodiversity protection, with high losses in cases of weakening legislation, which is particularly concerning for strictly protected CUs.

This scenario has increased scientific interest in understanding the pattern and dynamics of the landscape surrounding Cerrado CUs, both within BZs where they are delimited to and in areas adjacent to CUs without management plans (Bellón et al., 2020; Pinto et al., 2021; Ferreira, M. A. et al., 2022; Torgeski et al., 2025). For CUs without a BZ, most existing studies are spatially limited to one or a few CUs and/or focus mainly on land use and land cover characteristics. Consequently, they do not prioritize the assessment of the composition (number, type, and extent) and configuration (character, arrangement, spatial position, or orientation) of the elements constituting the landscape (Suárez-Castro et al., 2022).

In the Cerrado biome, this legislative fragility raises questions regarding the extent to which the surroundings of strictly protected CUs differ from landscapes in the biome not associated with protected areas in terms of spatial structure, habitat fragmentation, and the availability of continuous natural areas. This gap is particularly relevant after 2010, when the revocation of the surrounding area eliminated the only remaining legal safeguard for CUs without a Management

Plan. It also remains unclear whether this revocation coincided with detectable changes in the configuration of the surrounding landscape. Without this information, it cannot be determined whether strictly protected Cerrado CUs are in fact embedded in landscape contexts more favorable to nature conservation than the rest of the biome. Addressing this question is essential not only for evaluating the effectiveness of the current CU network, but also for prioritizing the conservation efforts in the Cerrado.

Nowosad and Hesselbarth (2025) stated that it is possible to gain a deeper understanding of environmental processes and changes by using landscape metrics, which allow one to quantify the composition and configuration of spatial landscape features. The application of these metrics has proven effective for describing various characteristics, such as fragmentation patterns, heterogeneity, and connectivity, using time series of land use and land cover for comparing landscapes subjected to different levels of anthropogenic pressure (Woodman et al., 2025). This approach provides an operational framework for characterizing changes in landscape structure and supporting assessments of their potential impacts on ecological processes and ecosystem services (Nowosad and Hesselbarth, 2025; Woodman et al., 2025).

This study aimed to determine whether the surroundings of strictly protected Cerrado conservation areas comprise landscapes whose structure supports ecological processes more favorable to nature conservation than locations outside these conservation contexts, and to measure whether landscape structure around these areas changed following the revocation of the surrounding area. This assessment was conducted both spatially, by comparing CUs and random points within the Cerrado in 2022, and temporally, by comparing the landscape structures surrounding CUs in 2010 and 2022, thereby evaluating potential changes associated with contemporary legislation. Given the period covered by relevant legislation regarding CU surrounding areas, we hypothesized that these areas would have maintained their landscape structure over time and would offer more favorable conditions for conservation than external locations.

Materials and Methods

Conservation unit selection

The vector file of Brazil's CUs was obtained from the Ministry of the Environment's online platform (http://mapas.mma.gov.br/i3geo/datadownload.htm), and the Cerrado's territorial boundary, available on the Brazilian Institute of Geography and Statistics (https://portaldemapas.ibge.gov.br/portal.php), both at a scale of 1:250,000. Based on these files and on online queries to environmental agencies at the municipal, state, and federal levels, the CUs were selected based on a strictly protected group without a BZ that are located entirely within the Cerrado, and are not inserted in or even bordered by another CU within a radius of at least 10 km. The definition of this extent was motivated by the concept of surrounding areas. Even after the publication of the SNCU and the guidelines for the establishment of BZs, the concept of a surrounding area was still valid (until 2010), especially in cases where the CU did not have a designated BZ (Guimarães et al., 2012), which is the situation considered in this study.

In spatial terms, the 18 strictly protected CUs that met the selection criteria (Figure 1) constituted a set of sampling units whose surrounding landscape composition and configuration were evaluated and compared with locations outside the CU context (Figure 2A). These external locations were defined by generating random points, in the same number as the selected CUs, across the entire extent of the Cerrado, with a minimum distance of 20 km between the points and relative to the boundaries of the CUs and of the Cerrado (Figure 1). This distance was defined to avoid: two points being close enough to share the same landscape; a point being generated within the surrounding area or BZ of a CU; and the landscape to be evaluated from being completely contained within the Cerrado.

In a temporal context, the samples were selected based on the year of revocation of the surrounding area. Among the 18 previously chosen CUs, those that were created after 2010 enabled an assessment of whether the landscape structure had undergone major changes. This year was adopted as the temporal reference as it coincides with the revocation of the surrounding area (Brasil, 2010). This filtering resulted in a total of nine CUs, whose surrounding landscapes were evaluated in 2010 and compared with their structure and configuration in 2022 (Figure 2B).

Definition of the sampling unit and its constituent land use/land cover classes

For each selected CU and each randomly generated point, a 10-km radius BZ was established (Figure 2A) to identify the land use and land cover classes present in these areas.

Land use and land cover data were sourced from the MapBiomas Project Collection 8 (Souza Junior et al., 2020) and accessed via Google Earth (Gorelick et al., 2017). Land use and land cover data were used due to this element often being employed to quantify and assess landscape structure (Istanbuly et al., 2022; Silveira et al., 2023).

The year 2022 was selected as it represented the most recent collection available during the execution of this study. For the BZs of the 9 CU that were subjected to temporal analysis, the classes for the year 2010 were also evaluated (Figure 2B). Natural vegetation cover classes identified in the surroundings of the CU (namely, flooded grassland and swamp area, rocky outcrop, and forest, savanna, and grassland formations) were grouped into a single class, termed "cerrado." Similarly, the remaining classes, which include anthropic uses (i.e., silviculture, pasture, sugarcane, agriculture-pasture mosaic, urbanized area, other non-vegetated areas, mining, soybean, temporary crops, coffee, citrus, and other perennial crops) and water bodies, were aggregated into a single class classified as "other."

All procedures involving the manipulation, processing, or presentation of georeferenced data were conducted using the long-term support release of QGIS software (QGIS Development Team, 2021). When necessary, such as during landscape cropping and quantification, datasets were reprojected to an appropriate coordinate reference system.

Landscape quantification

Landscape quantification was performed using landscape metrics. Metric selection prioritized those related to the quantification of habitat amount and fragmentation, applicable to landscapes (sampling units) of different sizes that did not generate redundant information (multicollinearity). Calculations were performed using the landscapemetrics package (Hesselbarth et al., 2019) in the R programming environment (R Core Team, 2021). The following metrics were selected: percentage of landscape of class (PLAND), which quantifies the proportional abundance of each class type present in the landscape; edge density (ED), defined as the sum of the total edge length of all segments involving the corresponding patch type, divided by the total landscape area (m^2) and multiplied by 10,000 (Russo et al., 2025); as well as effective mesh size (MESH), which calculates the mean patch size across the entire landscape, weighted by the areas of individual patches (Kubacka et al., 2025). It can be interpreted as the probability that two pixels chosen at random in the landscape are not located in the same patch (Mohammadi and Fatemizadeh, 2021).

Statistical analysis

The landscape structure metrics (PLAND, ED, MESH) were organized into two datasets for statistical analysis. The first dataset, which focused on the spatial aspect, consisted of the values corresponding to the surroundings of the 18 CUs and the 18 random points (Figure 2A).

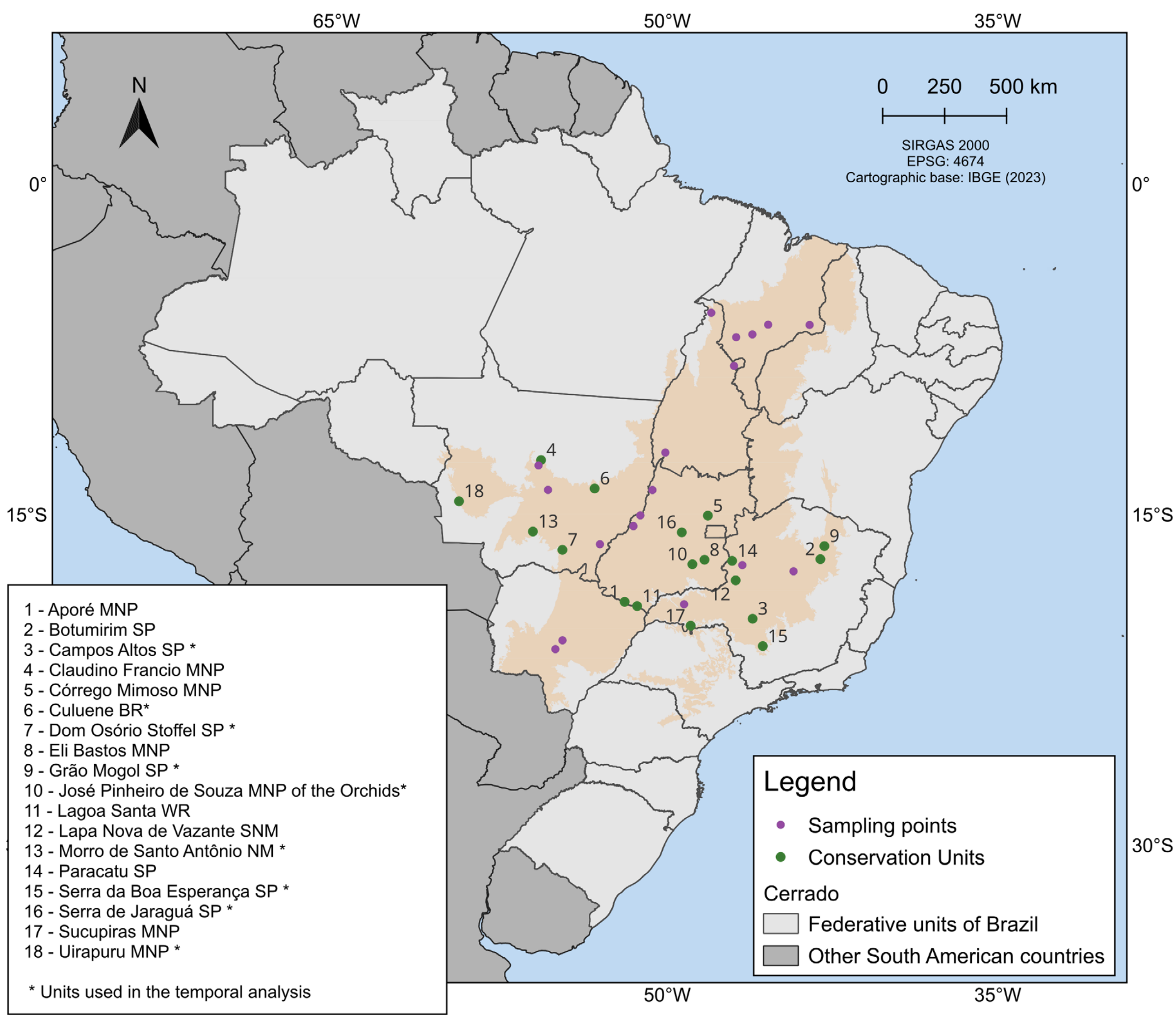

Figure 1 – Location of the landscapes (conservation units and sample points) used in this study.
MNP: Municipal Natural Park; SP: State Park; BR: Biological Reserve; WR: Wildlife Refuge; SNM: State Natural Monument; NM: Natural Monument.

This dataset was used to test whether there were statistically significant differences between these two groups and, if significant differences were found, which group presented parameters that were more consistent with ecological processes favorable to nature conservation.

The second dataset, which focused on temporal evaluation, comprised PLAND, ED, and MESH metrics for the surroundings of the 9 CUs throughout the two-year timeframe (Figure 2B). The main goal was to test whether there was a statistically significant difference in landscape structure between 2010 and 2022 and, similarly to the first dataset, which year presented more ecologically favorable characteristics, if significant differences were detected.

All data groups were subjected to the Shapiro-Wilk normality test to verify whether they presented a normal distribution, followed, by group comparison tests, in which a 95% confidence level was adopted ($p<0.05$). Landscape structure data were also described statistically, with mean and standard deviation computed when data were normally distributed, and median and quartiles when they were not. In addition to quantitative information, the data were represented graphically. All statistical analyses were performed in JAMOVI v. 2.3.26 (The Jamovi Project, 2022).

Study limitations

Despite efforts to identify all fully protected CU without BZs in the Cerrado, it is likely that some CU were not captured in the screening due to a lack of information in the search channels of the respective public agencies. The result of this potential information gap, combined with the other selection criteria for CU, is a reduced sample size (18 for spatial analysis and 9 for temporal analysis), which may restrict the statistical power of the tests and the generalization of the results to the entire biome. The selection criteria and the procedures for randomizing points likely introduced a spatial sampling bias, resulting in a noticeable sampling "void" of CU in the northern portion of the study area and an uneven distribution of the points across the Cerrado. These patterns may not represent the full landscape variability of the Cerrado and, con-

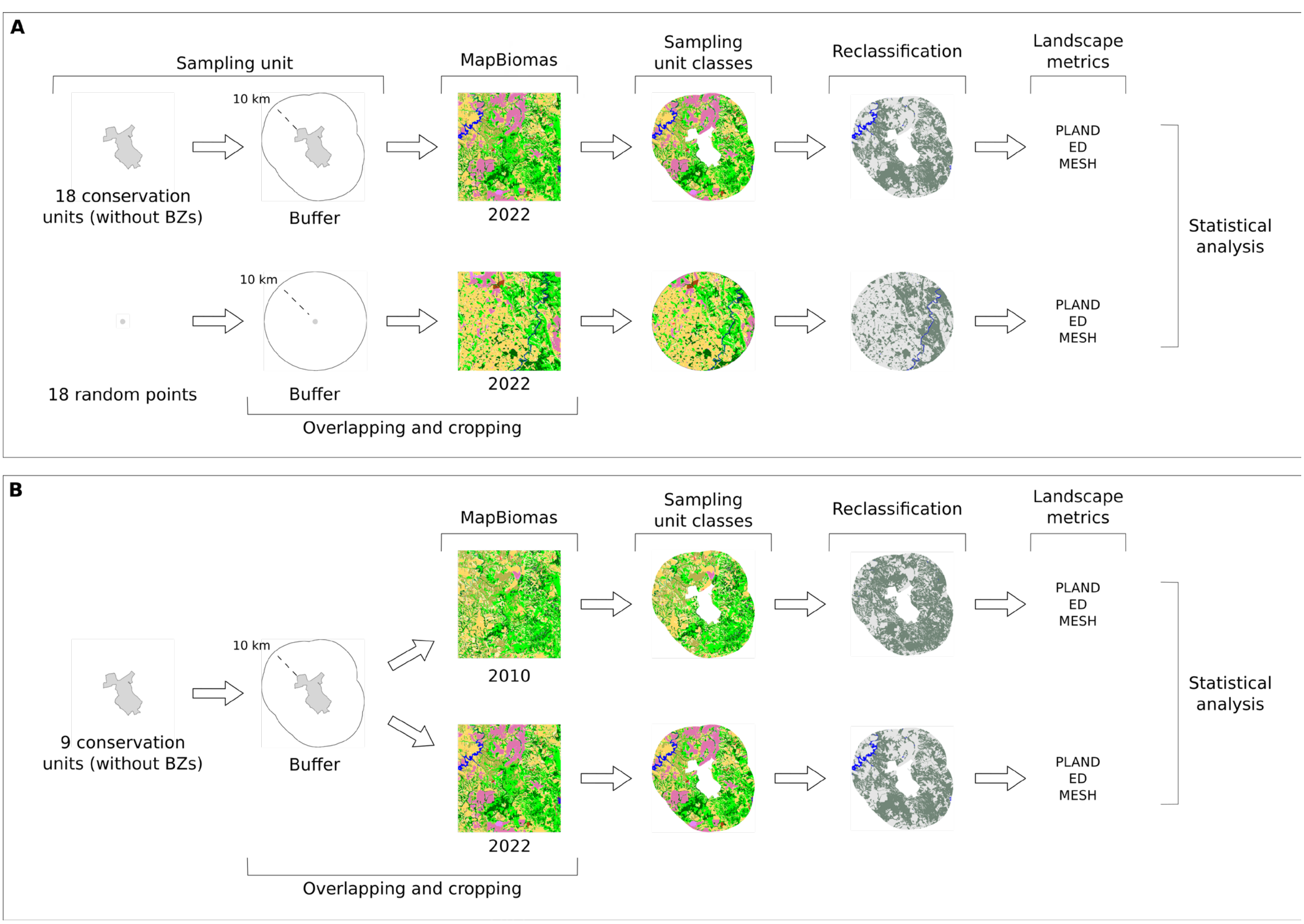

Figure 2 – Schematic representation of the methodological steps related to the (A) spatial and (B) temporal assessments.
PLAND: percentage of landscape class; ED: edge density; MESH: effective mesh size.

sequently, may limit the extrapolation of the findings. From a temporal perspective, although the benchmark was well grounded in criteria related to potential impacts on landscape transformation, the interval under analysis spans only 13 years (2010–2022), which may be insufficient to detect long-term or consolidated landscape change trends.

Results

The Shapiro-Wilk test indicated that the PLAND metric values across independent groups did not follow a normal distribution (p=0.008; W=0.913); therefore, the Mann-Whitney test was used in the statistical analysis. A low percentage of the Cerrado class was identified in most of the landscapes surrounding Cerrado strictly protected CUs without BZs: in 13 of the 18 CUs, less than 50% of this natural cover was present in their surroundings. This composition resulted in a median of 29.8% (1st quartile=20.1%; 3rd quartile=51.5%) (Figure 3), in addition to revealing Sucupiras Municipal Natural Park as the CU with the lowest proportional abundance of Cerrado in its surroundings, with just over 7%.

Similarly, at the random points defined as a sample of the Cerrado excluding CU surroundings, the class corresponding to native vegetation was predominant in only 8 landscapes, with a median of 40.4% (1st quartile=20.1%; 3rd quartile=75%). Although PLAND values were lower in the CU surroundings than in areas outside this spatial context, no significant difference was found between both groups (U=134; p=0.389).

For the ED metric, the result suggested that the data followed a normal distribution (W=0.956; p=0.159); therefore, Student's t-test was applied. As Student's t-test also assumes homogeneity of variances between groups, this assumption was tested and met using Levene's test (F=0.00107; p=0.974).

The t-test showed that the mean edge density of Cerrado in the CU surroundings differed from that observed at the random points (t=2.08; p=0.045), with the CU surroundings presenting a greater edge area (37.3 m/ha mean, standard deviation=14.4 m/ha) than the random points (27.3 m/ha, standard deviation=14.4 m/ha). Morro de Santo Antônio Natural Monument and Uirapuru Municipal Natural Park stand out as the CU with, respectively, the highest and lowest Cerrado edge density in their surroundings, with approximately 64.0 and 16.38 m/ha.

Regarding the size of Cerrado fragments in the spatially compared groups, the normality test yielded p<0.001 (W=0.610), indicating that the data did not follow a normal distribution; thus, the Mann-Whitney test was used to compare the groups. The CUs that stand out individually are Botumirim State Park, due to its MESH value of 86,366.24 ha in the surroundings (the highest among the evaluated landscapes) and Sucupiras Municipal Natural Park, with 4.60 ha, the lowest value. These CUs, despite belonging to the group with a lower MESH median of 349 ha (1st and 3rd quartiles=102 and 3,733 ha), did not constitute, according to the Mann-Whitney test (U=138; p=0.462), landscapes that differed from those found at the random sampling points, where a median of 1,955 ha (1st and 3rd quartiles=292 and 16,618 ha) was observed.

For the temporally measured datasets, statistical analyses were performed using the Wilcoxon test as the data were paired and did not follow a normal distribution, as indicated by the Shapiro-Wilk test (PLAND: W=0.807; p=0.025; ED: W=0.754; p=0.006; MESH: W=0.611; p<0.001).

For the PLAND metric, between 2010 and 2022 there was no significant change in the percentage of natural vegetation in any of the sampled landscapes (Figure 4A). Uirapuru Municipal Natural Park was the CU with the most notable change in PLAND values, with a reduction of approximately 8.4% in the proportion of the Cerrado class. The pattern observed for the set of CUs was confirmed by comparing the medians for the two years: 31.9% in 2010 (1st quartile=18.8%; 3rd quartile=56.6%) and 32% in 2022 (1st quartile=19.6%; 3rd quartile=53.3%). Based on the Wilcoxon test result (W=28.0; p=0.570), there was insufficient statistical evidence to conclude that the medians of these groups differed from one another.

The comparison between ED metrics across the research timeframe showed that edge density increased in Cerrado fragments in all landscapes studied. The discrepancy between years is illustrated in the graphical analysis (Figure 5). Based on the Wilcoxon test result

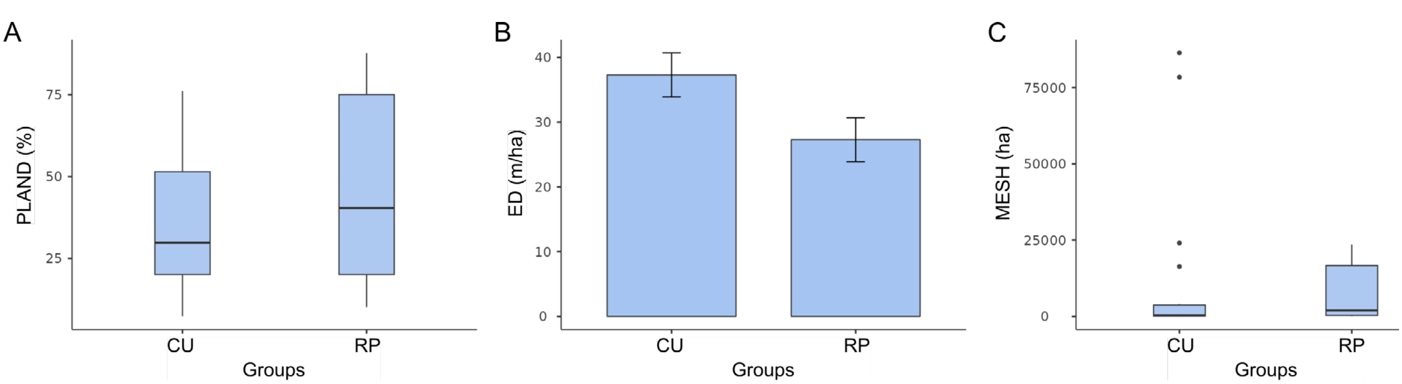

Figure 3 – Graphical representation of the distribution of PLAND, ED, and MESH values for Conservation Unit datasets and sampling points.
CU: Conservation Unit; RP: random point.

Figure 4 – Percentage variation in the values of the (A) PLAND, (B) ED, and (C) MESH values between 2010 and 2022.
SP: State Park; BR: Biological Reserve; MNP: Municipal Natural Park; NM: Natural Monument.

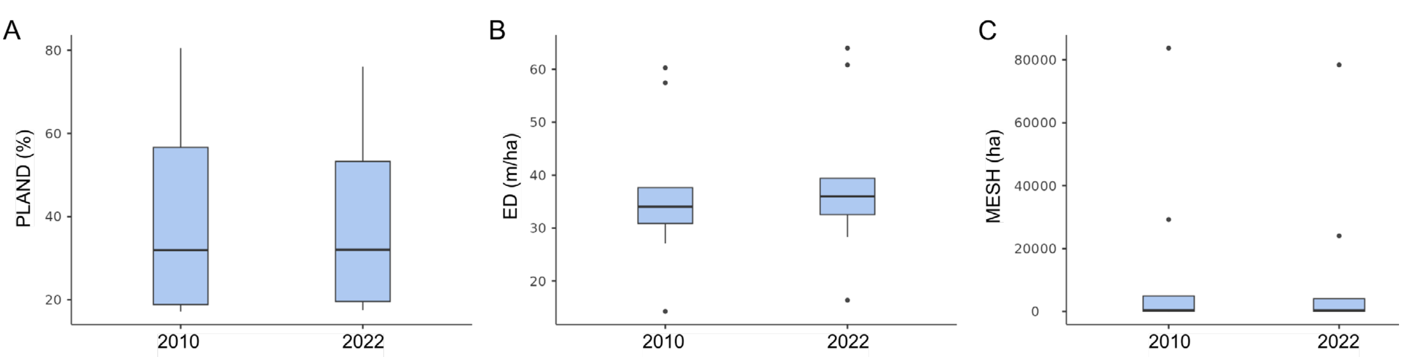

Figure 5 – Graphical representation of the distribution of PLAND, ED, and MESH values from the Conservation Unit datasets across two distinct years.
CU: Conservation Unit; RP: random point.

(W=0; p=0.004), one can infer that the edge density in 2010 was statistically different from that in 2022. Morro de Santo Antônio Natural Monument and Uirapuru Municipal Natural Park were the CUs that experienced the greatest variations in ED values, both with increases of 11.40% and 14.83, respectively (Figure 4B).

The measures of central tendency and dispersion for MESH values (Figure 4C) indicated a reduction in the mean size of fragments over recent years. In 2010, the median was 393 ha, with first and third quartiles of 110 ha and 4,951 ha, whereas in 2022 this measure decreased to 360 ha, with quartiles of 84.7 ha and 4,072 ha. Serra de Jaraguá State Park was the CU with the largest negative variation in MESH (Figure 4C), with a reduction of approximately 23% in the size of Cerrado fragments. In contrast, José Pinheiro de Souza Municipal Natural Park of the Orchids showed the greatest increase, from 28.53 ha in 2010 to 37.52 ha in 2022, an increment of 31.50% (Figure 4C). However, when all CU were compared, the reductions observed in median MESH values did not provide sufficient evidence to conclude that they represented different landscapes, as indicated by the Wilcoxon test (W=33.0; p=0.250).

Discussion

The matrix constitutes the dominant structural element of a landscape (i.e., it occupies over 50%) and, as such, generally plays a major role in its functioning (Fletcher et al., 2024). In the case of the landscapes evaluated herein, the matrix was composed of anthropogenic land use, which differs, even if slightly, from estimates reported in recent studies for the Cerrado (Souza Junior et al., 2020; Pompeu et al., 2024). These studies found that at least 51% of this biome is still composed of natural vegetation.

In terms of biodiversity conservation, the matrix provides habitat at smaller spatial scales, increasing the effectiveness of protected areas, and controlling landscape connectivity, including organism movements between protected areas (Franklin, 1993). Given the low percentage of natural vegetation found both in the landscapes surrounding the CUs and in the external areas, these fundamental functions may be compromised.

Various effects may arise from this situation, in which the reduction of the Cerrado area results in a loss of habitat. Confirming this perspective, Muylaert et al. (2016) demonstrated that a higher proportion of Cerrado cover in the landscape favors the occurrence of nectarivorous and omnivorous bats. The authors proposed that conservation strategies should prioritize increasing native vegetation in landscapes as an effective measure to promote higher bat species richness. Melo et al. (2017) conducted a study in the Cerrado and reported that habitat availability is the most important predictor of small mammal species richness. Also in the Cerrado, Sousa et al. (2022) confirmed, based on a study of Euglossini bees, that habitat availability is essential for maintaining this group. Assessing the diversity of reproductive traits in plants in the Cerrado landscapes, Martello et al. (2023) found that habitat availability is one of the variables (alongside fragment density and edge density) that best explained the density and richness of seed dispersal and pollination systems. The authors indicated that conserving native vegetation is essential to maintain the diversity of plant functional traits in agricultural landscapes of the Cerrado.

In light of these findings, it is reasonable to state that the landscapes evaluated in this study exhibit a critical status regarding Cerrado cover percentage (PLAND) and to the ecological processes and nature conservation functions that this variable can provide. Specifically for the CU, this context may imply reduced conservation capacity (Hansen and Rotella, 2001) and can be exacerbated by the size of the protected area (Wiersma et al., 2004), CU isolation (DeFries et al., 2005), or the duration of this ongoing situation.

The maintenance of Cerrado vegetation around strictly protected CU in the Cerrado without BZs for over a decade may indicate that the analysis period was not long enough to detect substantial landscape changes, that there is little remaining unprotected Cerrado vegetation available for conversion to anthropogenic use, that remaining areas are protected under other legal mechanisms, or even that SNCU discouraged landowners adjacent to CU from further clearing natural vegetation.

Although some of these premises are difficult to sustain, due to the severe alteration of permanent preservation areas (Gomes et al., 2022; Martins et al., 2023; Campos and Reichardt, 2025), which are legally protected areas where vegetation removal is prohibited (Ferreira, R. V. et al., 2022), others may be considered plausible. For instance, Melo and Martins (2020) found relative stability in Cerrado vegetation cover in 14 environmental protection areas. This CU category belongs to the sustainable use group, presumably more susceptible to human activities. In addition, it has been suggested that the global conversion rate of natural habitats is decreasing due to the scarcity of areas that can be easily converted for new agricultural uses (Laurance, 2010). Although this is not necessarily fully applicable to Brazil in the Cerrado, Zardini et al. (2016) observed that physical environmental characteristics, such as soils (sandy, shallow, and nutrient-poor) and landforms (predominantly undulating and mountainous), contributed to the preservation of natural vegetation in northern Goiás State by hindering agricultural development.

Habitat loss generally leads to increased habitat fragmentation, resulting in smaller patches, greater isolation, and an increase in the proportion of edge habitats in the landscape (Fletcher Junior, 2005). In these edge habitats, modifications occur in ecological patterns and processes, known as edge effects (Cui, 2025). The analysis of the areas in this study revealed greater vulnerability to edge effects around CU compared to areas that were not subjected to preservation commitments. This characterizes a less favorable condition to maintaining the ecological processes that are essential for nature conservation in the former. Considering that both groups did not differ significantly in Cerrado percentage, higher edge density indicates that the landscapes surrounding CU are more fragmented than those outside this spatial context.

A temporal assessment showed that this context has become consolidated in recent years, during which there was a significant increase in edge density around the analyzed CU, indicating intensification of the reduction in conservation values. According to Bennett and Saunders (2010), this results from modification of previously intact habitats. Since the Cerrado is composed of various phytophysiognomies distributed across different vegetation formations (Lira-Martins et al., 2022), potential impacts of edge effects will act with different intensities depending on the vegetation type. In forests, for example, because they are structurally denser and more vertically stratified formations than grasslands or savannas, the incidence of abiotic effects including temperature, humidity, and light tends to be more pronounced.

This difference means that the impacts of edge effects in forest ecosystems are well known but rarely studied in open-canopy areas, as found in many savannas (Mendonça et al., 2015; Franklin et al., 2021). Nevertheless, some studies have confirmed the existence of edge effects in the Cerrado. Mendonça et al. (2015), in savanna fragments, observed that the most significant edge effect threatening conservation of Cerrado vegetation is the widespread invasion by African grasses, which alter the structure and composition of native plant communities, thus compromising population dynamics and persistence of native species. In a *cerradão* area, Brasil et al. (2013) identified a positive and significant relationship between distance from the edge and litter layer depth, concluding that fragmentation can impair ecosystem functions and services, with possible future impacts on species diversity. In addition, in this forest phytophysiognomy, Lima-Ribeiro (2008) demonstrated a clear influence of edge effects on the structure and distribution of *Vernonia aurea* Mart. ex DC. (Asteraceae). Using a plant species as an indicator of edge effects, *Syagrus flexuosa* (Mart.) Becc. (Arecaceae), Ragusa-Netto (2024) compared the abundance and distribution of *S. flexuosa* seedlings between edges and interiors and obtained data suggesting reduced seed dispersal at edges. According to the author, because *S. flexuosa* is a widely distributed palm that serves as an important resource for several animals in Cerrado habitats, changes in its regeneration process due to edge effects may further impact frugivore populations. Regarding the fauna, Oliveira et al. (2012) found a positive relationship between the abundance of non-volant small mammals and distance from the edge in a study conducted in a forest-cerrado transition zone.

According to Turner (1996), the relative importance of habitat edges increases as fragment size decreases, a circumstance in which edge effects can become highly influential. Fragment size is also relevant for evaluating landscape fragmentation and, when reduced, together with a decrease in habitat amount, an increase in habitat isolation, and an increase in fragment number, forms the basis of most quantitative measures of habitat fragmentation (Fahrig, 2003).

In this study, the metric used to assess fragment size, MESH, did not show significant differences between values for CU surroundings and random points, nor between their medians. This result may reflect the absence of a BZ in CU management plans. Although the importance of large fragments for biodiversity maintenance and ecological processes is clear, it is difficult to establish the extent in which it can be considered large, since this can be understood as a functional variable that depends on the home range and space use of a given organism.

Using as reference the mean home range size of the jaguar (*Panthera onca*), a threatened species that occurs in the Cerrado and is recorded in the largest number of federal CU in Brazil (Nascimento and Campos, 2011), estimated at 16,100 ha (Silveira, 2004), it is evident that the median fragment size found is negligible. Even when considering a species with a much smaller home range, such as the giant armadillo (*Priodontes maximus*), estimated at 1,005 ha (Silveira et al., 2009), the fragment sizes obtained seem to be insufficient to support populations of these species.

Although a landscape without fragmentation, or at least with fragments large enough to meet the needs of present species, would be desirable, the fact that the current situation has persisted for more than a decade around strictly protected CU in the Cerrado without BZ suggests that attention should focus on preserving these fragments. This stance is necessary since the small remnants show fragile sustainability patterns over time and, without area expansion, they may disappear over the years (Pirovani et al., 2014). It is also worth emphasizing that small fragments play important roles in maintaining biodiversity, for instance, by functioning as links between larger fragments and serving as foraging areas (Han et al., 2022; Szangolies et al., 2022; Pompeu et al., 2024).

Despite being difficult to pinpoint, the factors that cause anthropogenic impact occurring in the Cerrado biome being reflected in the surrounding protected areas, the lack of management plans and the consequent absence of BZs can be considered an additional driver alongside the underlying forces of Cerrado conversion (i.e., financial incentives and infrastructure to expand agribusiness-related activities). Considering that there is a high proportion of CUs without any technical documentation, which contributes to poor administration and protection (Saleme and Costa, 2020), and that even the creation of CUs has occurred without ensuring that they will achieve their objectives (Lima et al., 2005), the data obtained in this study provide evidence of the need to enforce existing environmental protection laws in Brazil.

Conclusion

Our findings revealed that the surroundings of strictly protected conservation units in the Cerrado without buffer zones are composed of landscapes whose structure is associated with ecological processes less favorable to conservation than those in areas outside this spatial context. This scenario underscores the need for a better understanding of how species within these units may be affected.

Regarding temporal evaluation, we found that the landscape-structure circumstance around conservation units has deteriorated, even if slightly, over at least the past decade. This conclusion is supported by the degradation inherent in increased edge density and, together with the stability of both Cerrado percentage (PLAND) and mean fragment size (MESH), underscores the anthropogenic pressure experienced by the conservation unit.

These conclusions highlight the need for studies that assess landscape structure in buffer zones in the Cerrado conservation units, comparing the results with those reported herein. It is expected that such zoning may be an effective tool for providing conditions favorable to nature conservation. Another strategy that could be adopted, and which also requires validation of its effectiveness, would be to overlap conservation units, placing a strictly protected unit over a sustainable use unit so that the entire surrounding area would fall within the sustainable use conservation unit.

Conservation and management measures must be implemented to mitigate the effects of habitat fragmentation and promote connectivity among remaining fragments, thereby ensuring the long-term conservation of biodiversity and ecosystems in the Cerrado.

Acknowledgements

Work by RF-O e PTAM was developed in the context of the "Araguaia Vivo 2030" project, supported by an agreement between the Tropical Alliance Water Research (TWRA) and Fundação de Amparo à Pesquisa do Estado de Goiás (FAPEG) (grant 202210267000536).

Authors' Contributions: Torquato, E.S.: methodology; investigation; formal analysis; visualization; writing – original draft. Machado, K.B.: validation; writing – review & editing. Freitas-Oliveira, R.: validation; writing – review & editing. Martins, P.T.A.: conceptualization; data curation; writing – original draft; project administration; supervision.

Conflicts of interest: the authors declare no conflicts of interest.

Funding: Coordenação de Aperfeiçoamento de Pessoal de Nível Superior - Brasil (CAPES) - Finance Code 001; State University of Goiás / Pró-Programas — Edital no. 01/2023, no. 31-2023, process SEI no. 202300020012431; and PELD Araguaia (process 445733/2024-1/FAPEG 202510267001637).

References

Bellón, B.; Blanco, J.; De Vos, A.; Roque, F.O.; Pays, O. et al., 2020. Integrated landscape change analysis of protected areas and their surrounding landscapes: application in the Brazilian cerrado. Remote Sensing, v. 12, (9), 1413. https://doi.org/10.3390/rs12091413.

Bennett, A.F.; Sauders, D.A., 2010. Habitat fragmentation and landscape change. In: Sodhi, S.N.; Ehrlich (Eds), Conservation biology for all. Oxford, Oxford, pp. 88-106. https://doi.org/10.1093/acprof:oso/9780199554232.003.0006.

Brasil, 1990. Resolução no. 13/90, de 6 de dezembro de 1990. Conselho Nacional do Meio Ambiente – CONAMA (Accessed January 8, 2024) at:. https://conama.mma.gov.br/?option=com_sisconama&task=arquivo.download&id=110.

Brasil, 2000. Lei no. 9.985, de 18 de julho de 2000. Regulamenta o art. 225, § 1o, incisos I, II, III e VII da Constituição Federal, institui o Sistema Nacional de Unidades de Conservação da Natureza e dá outras providências (Accessed August 10, 2023) at:. http://www.planalto.gov.br/ccivil_03/leis/L9985.htm.

Brasil, 2010. Resolução no. 428, de 17 de dezembro de 2010. Conselho Nacional do Meio Ambiente – CONAMA (Accessed March 12, 2026) at:. https://conama.mma.gov.br/?option=com_sisconama&task=arquivo.download&id=622.

Brasil, 2015. Resolução no. 473, de 11 de novembro de 2015. Conselho Nacional do Meio Ambiente – CONAMA (Accessed December 23, 2023) at:. https://conama.mma.gov.br/?option=com_sisconama&task=arquivo.download&id=693.

Brasil, L.S.; Giehl, N.F.S.; Santos, J.O.; Santos, A.O.; Marimon, B.S. et al., 2013. Efeito de borda sobre a camada de serapilheira em áreas de cerradão no leste de Mato Grosso. Biotemas, v. 26, (3), 37-47. https://doi.org/10.5007/2175-7925.2013v26n3p37.

Campos, R.F.F.; Reichardt, L.G., 2025. Diagnóstico do uso e ocupação do solo da Área de Preservação Permanente (APP) do perímetro urbano do município de Ouro (Santa Catarina). Revista Brasileira de Sensoriamento Remoto, v. 6, (3), 002-012. https://doi.org/10.66205/rbsr.v6i3.211.

Cheng, Y.; Liu, H.; Du, J.; Yi, Y., 2025. Quantifying biodiversity's present and future: current potentials and SSP-RCP-Driven land use impacts. Earth's Future, v. 13, (2), e2024EF005191. https://doi.org/10.1029/2024EF005191.

Cui, D., 2025. Optimization of ecological landscape of urban human settlements under edge effect. Signal, Image and Video Processing, v. 19, 538. https://doi.org/10.1007/s11760-025-04129-0.

DeFries, R.; Hansen, A.; Newton, A.C.; Hansen, M.C., 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. Ecological Applications, v. 15, (1), 19-26. https://doi.org/10.1890/03-525.

Del Lama, C.; Shimbo, J.; Teixeira, L.; Oliveira, M.; Coelho-Junior, M.G. et al., 2025. RAD2024: Relatório Anual do Desmatamento no Brasil 2024. MapBiomas, São Paulo.

Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics, v. 34, 487-515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419.

Ferreira, M.A.; França, L.C.J.; Morais, M.S.; Gontijo, B.M.; Gorgens, E.B. et al., 2022. Conflitos do uso da terra em áreas protegidas por lei no Parque Estadual do Rio Preto e zona de amortecimento, Minas Gerais. Caderno de Geografia, v. 32, (68), 377-404. https://doi.org/10.5752/P.2318-2962.2022v32n68p377.

Ferreira, R.V.; Castro, S.C.; Gomes Júnior, J.A.; Alexandre, F.I.B.; Martines, M.R., 2022. Environmental Fragility in a Permanent Preservation Area. Floresta e Ambiente, 29, (4), e20220027. https://doi.org/10.1590/2179-8087-FLORAM-2022-0027.

Fletcher, R.J.; Smith, T.A.H.; Troy, S.; Kortessis, N.; Turner, E.C. et al., 2024. The prominent role of the matrix in ecology, evolution, and conservation. Annual Review of Ecology, Evolution, and Systematics, v. 55, 423-447. https://doi.org/10.1146/annurev-ecolsys-102722-025653.

Fletcher Junior, R.J., 2005. Multiple edge effects and their implications in fragmented landscapes. Journal of Animal Ecology, v. 74, (2), 342-352. https://doi.org/10.1111/j.1365-2656.2005.00930.x.

Franklin, C.M.A.; Harper, K.A.; Clarke, M.J., 2021. Trends in studies of edge influence on vegetation at human-created and natural forest edges across time and space. Canadian Journal of Forest Research, v. 51, (2), 274-282. https://doi.org/10.1139/cjfr-2020-0308.

Franklin, J.F., 1993. Preserving biodiversity: Species, ecosystems, or landscapes? Ecological Applications, v. 3, (2), 202-205. https://doi.org/10.2307/1941820.

Gomes, M.C.; Sousa, A.C.R.C.; Bayer, M.; Faria, K.M.S., 2022. Degradação da vegetação nativa e implicações sobre o regime hidrológico na bacia hidrográfica do Rio Claro, sub-bacia do Rio Araguaia (GO). Geociências, v. 41, (3), 559-568. https://doi.org/10.5016/geociencias.v41i03.15579.

Gorelick, N.; Hancher, M.; Dixon, M.; Ilyushchenko S.; Tchau, D. et al., 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote Sensing of Environment, v. 202, 18-27. https://doi.org/10.1016/j.rse.2017.06.031.

Guerra, B.R.; Abrantes, P.C.R.M.; Ranieri, V.E.L., 2025. Land cover changes in buffer zones and the effectiveness of this conservation strategy: Case study of Southeastern Brazil. Land Use Policy, v. 157, 107665. https://doi.org/10.1016/j.landusepol.2025.107665.

Guimarães, J.C.C.; Machado, F.S.; Borges, L.A.C.; Rezende, J.L.P.; Soares, A.A.V. et al., 2012. Aspectos legais do entorno das Unidades de Conservação brasileiras: área circundante e zona de amortecimento em face à Resolução CONAMA N. 428/2010. Espaço & Geografia, v. 15, (1), 1-20.

Han, L.; Wang, Z.; Wei, M.; Wang, M.; Shi, H. et al., 2022. Small patches play a critical role in the connectivity of the Western Tianshan landscape, Xinjiang, China. Ecological Indicators, v. 144, 109542. https://doi.org/10.1016/j.ecolind.2022.109542.

Hansen, A.J.; DeFries, R., 2007. Ecological mechanisms linking protected areas to surrounding lands. Ecological Applications, v. 17, (4), 974-988. https://doi.org/10.1890/05-1098.

Hansen, A.J.; Rotella, J.J., 2001. Nature reserves and land use: implications of the "place" principle. In: Dale, V.H.; Haeuber, R.A. (Eds.), Applying ecological principles to land management. Springer, New York, pp. 54-72.

Hesselbarth, M.H.K.; Sciaini, M.; With, K.A.; Wiegand, K.; Nowosad, J., 2019. landscape metrics: an open-source R tool to calculate landscape metrics. Ecography, v. 42, (10), 1648-1657. https://doi.org/10.1111/ecog.04617.

Istanbuly, M.N.; Kaboli, M.; Ahmadi, S.; Tian, G.; Michalak, M. et al., 2022. Landscape metrics explain the ecological susceptibility of terrestrial ecosystems. Landscape Online, v. 97, 1101. https://doi.org/10.3097/LO.2022.1101.

Jaureguiberry, P.; Titeux, N.; Wiemers, M.; Bowler, D.E.; Coscieme, L. et al., 2022. The direct drivers of recent global anthropogenic biodiversity loss. Science Advances, v. 8, (45), eabm9982. https://doi.org/10.1126/sciadv.abm9982.

Kubacka, M.; Piniarski, W.; Żywica, P. Frazier, A.E., 2025. Tracking spatio-temporal LULC changes in key ecological network elements using fragmentation metrics and a custom raster-based approach: A multi-scale study from Poland. Ecological Informatics, v. 90, 103369. https://doi.org/10.1016/j.ecoinf.2025.103369.

Laurance, W.F., 2010. Habitat destruction: death by a thousand cuts. In: Sodhi, N.S.; Ehrlich, P.R. (Eds.), Conservation biology for all. Oxford, Oxford, pp. 73-87. https://doi.org/10.1093/acprof:oso/9780199554232.003.0005.

Li, G.; Fang, C.; Watson, J.E.M.; Sun, S.; Qi, W. et al., 2024. Mixed effectiveness of global protected areas in resisting habitat loss. Nature Communications, v. 15, 8389. https://doi.org/10.1038/s41467-024-52693-9.

Lima, G.S.; Ribeiro, G.A.; Gonçalves, W., 2005. Avaliação da efetividade de manejo das unidades de conservação de proteção integral em Minas Gerais. Revista Árvore, v. 29, (4), 647-653. https://doi.org/10.1590/S0100-67622005000400017.

Lima-Ribeiro, M.S. 2008. Efeitos de borda sobre a vegetação e estruturação populacional em fragmentos de Cerradão no Sudoeste Goiano, Brasil. Acta Botanica Brasilica, v. 22, (2), 535-545. https://doi.org/10.1590/S0102-33062008000200020.

Lira-Martins, D.; Nascimento, D.L.; Abrahão, A.; Costa, P.B.; D'Angioli, A.M. et al., 2022. Soil properties and geomorphic processes influence vegetation composition, structure, and function in the Cerrado Domain. Plant Soil, v. 476, 549-588. https://doi.org/10.1007/s11104-022-05517-y.

Martello, F.; Santos, J.S.; Silva-Neto, C.M.; Cássia-Silva, C.; Siqueira, K.N. et al., 2023. Landscape structure shapes the diversity of plant reproductive traits in agricultural landscapes in the Brazilian Cerrado. Agriculture, Ecosystems & Environment, v. 341, 108216. https://doi.org/10.1016/j.agee.2022.108216.

Martins, W.R.; Silva, A.A.; Castro, J.D.B., 2023. Mapeamento da área de preservação permanente do rio Passa-Três: uma contribuição para a gestão territorial e preservação do manancial. Revista Cerrados, v. 21, (2), 283-301. https://doi.org/10.46551/rc24482692202328.

Melo, A.C.A.; Martins, P.T.A., 2020. Contribuição das áreas de proteção ambiental na conservação do cerrado. Revista de Geografia, v. 37, (2), 53-71. https://doi.org/10.51359/2238-6211.2020.242444.

Melo, G.L.; Sponchiado, J.; Cáceres, N.C.; Fahrig, L., 2017. Testing the habitat amount hypothesis for South American small mammals. Biological Conservation, v. 209, 304-314. https://doi.org/10.1016/j.biocon.2017.02.031.

Mendonça, A.H.; Russo, C.; Melo, A.C.G.; Durigan, G., 2015. Edge effects in savanna fragments: a case study in the cerrado. Plant Ecology & Diversity, v. 8, (4), 493-503. https://doi.org/10.1080/17550874.2015.1014068.

Ministério do Meio Ambiente (MMA). Painel de unidades de conservação brasileiras. MMA (Accessed August 18, 2023) at:. https://antigo.mma.gov.br/areas-protegidas/cadastro-nacional-de-ucs.html.

Mohammadi, A.; Fatemizadeh, F., 2021. Quantifying landscape degradation following construction of a highway using landscape metrics in Southern Iran. Frontiers in Ecology and Evolution, v. 9, 721313. https://doi.org/10.3389/fevo.2021.721313.

Muylaert, R.L.; Stevens, R.D.; Ribeiro, M.C., 2016. Threshold effect of habitat loss on bat richness in cerrado-forest landscapes. Ecological Applications, v. 26, (6), 1854-1867. https://doi.org/10.1890/15-1757.1.

Nascimento, J.L.; Campos, I.B., 2011. Atlas de fauna brasileira ameaçada de extinção em unidades de conservação federais. Instituto Chico Mendes de Conservação da Biodiversidade, Brasília, DF, 276 p.

Nowosad, J.; Hesselbarth, M.H.K., 2025. The landscapemetrics and motif packages for measuring landscape patterns and processes. In: Rocchini, D. (Ed.), R Coding for Ecology. Use R!. Springer, Cham, pp. 217-245. https://doi.org/10.1007/978-3-031-99665-8_10.

Oliveira, A.C.M.; Santos, P.G.P.; Carvalho Júnior, O.; Montag, L.F.A.; Lima, R.C.S. et al., 2012. Edge effects and the impact of wildfires on populations of small non-volant mammals in the forest-savanna transition zone in Southern Amazonia. Biota Neotropica, v. 12, (3), 57-63. https://doi.org/10.1590/S1676-06032012000300004.

Pinto, J.S.; Pinto, A.L.; Pinto, V.P.S.; Oppliger, E.A.; Oliveira, A.K.M., 2021. Análise das mudanças do uso e cobertura da terra em dois parques urbanos e seus entornos em Campo Grande, Mato Grosso do Sul. Boletim de Geografia, v. 39, e59194. https://doi.org/10.4025/bolgeogr.v39.a2021.e59194.

Pirovani, D.B.; Silva, A.G.; Santos, A.R.; Cecílio, R.A.; Gleriani, J.M. et al., 2014. Análise espacial de fragmentos florestais na Bacia do Rio Itapemirim, ES. Revista Árvore, v. 38, (2), 271-281. https://doi.org/10.1590/S0100-67622014000200007.

Pompeu, J.; Assis, T.O.; Ometto, J.P., 2024. Landscape changes in the Cerrado: Challenges of land clearing, fragmentation and land tenure for biological conservation. Science of the Total Environment, v. 906, 167581. https://doi.org/10.1016/j.scitotenv.2023.167581.

QGIS Development Team, 2021. QGIS Geographic Information System. Open Source Geospatial Foundation Project (Accessed August 10, 2023) at:. http://qgis.osgeo.org.

Ragusa-Netto, J., 2024. Syagrus flexuosa (Mart.) Becc. (Arecaceae) seedling recruitment at the edge and interior of Cerrado remnants. Brazilian Journal of Biology, v. 84, e259137. https://doi.org/10.1590/1519-6984.259137.

R Core Team, 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna (Accessed March 21, 2021) at:. http://www.R-project.org/.

Russo, L.F.; Andrello, M.; Giuliani, M.; Ancillotto, L.; Carboni, M. et al., 2025. Mapping the urban landscape at multiple ecological scales. Urban Forestry & Urban Greening, v. 110, 128849. https://doi.org/10.1016/j.ufug.2025.128849.

Saleme, E.R.; Costa, W.A., 2020. Planos de manejo como mecanismo defensivo permanente de planejamento em defesa do patrimônio natural. Revista Direito Ambiental e Sociedade, v. 10, (1), 29-53.

Silva, J.M.C.; Dias, T.C.A.C.; Cunha, A.C.; Cunha, H.F.A., 2021. Funding deficits of protected areas in Brazil. Land Use Policy, v. 100, 104926. https://doi.org/10.1016/j.landusepol.2020.104926.

Silveira, L., 2004. Ecologia comparada e conservação da onça-pintada (Panthera onça) e onça-parda (Puma concolor) no cerrado e pantanal. Tese de doutorado, Universidade de Brasília, Brasília, DF, 240 p.

Silveira, L.; Jácomo, A.T.A.; Furtado, M.M.; Torres, N.M.; Sollmann, R. et al., 2009. Ecology of the giant armadillo (Priodontes maximus) in the grasslands of central Brazil. Edentata, v. 10, 25-34. https://doi.org/10.1896/020.010.0112.

Silveira, P.; Iop, S.; Santos, J.S.; Lima, E.L.; Martello, F. et al., 2023. Amount and isolation of aquatic habitat drive anuran diversity in agricultural landscapes in the Brazilian Cerrado. Landscape Ecology, v. 28, 2261-2275. https://doi.org/10.1007/s10980-023-01693-z.

Sousa, F.G.; Santos, J.S.; Martello, F.; Diniz, M.F.; Bergamini, L.L. et al., 2022. Natural habitat cover and fragmentation per se influence orchid-bee species richness in agricultural landscapes in the Brazilian Cerrado. Apidologie, v. 53, 20. https://doi.org/10.1007/s13592-022-00925-6.

Souza Junior, C.M.; Shimbo, J.Z.; Rosa, M.R.; Parente, L.L.; Alencar, A.A. et al., 2020. Reconstructing three decades of land use and land cover changes in Brazilian biomes with landsat archive and earth engine. Remote Sensing, v. 12, (17), 2735. https://doi.org/10.3390/rs12172735.

Strassburg, B.B.N.; Brooks, T.; Feltran-Barbieri, R.; Iribarrem, A.; Crouzeilles, R. et al., 2017. Moment of truth for the Cerrado hotspot. Nature Ecology & Evolution, v. 1, 0099. https://doi.org/10.1038/s41559-017-0099.

Suárez-Castro, A.F.; Maron, M.; Mitchell, M.G.E.; Rhodes, J.R., 2022. Disentangling direct and indirect effects of landscape structure on urban bird richness and functional diversity. Ecological Applications, v. 32, (8), e2713. https://doi.org/10.1002/eap.2713.

Szangolies, L.; Rohwäder, M.; Jeltsch, F., 2022. Single large AND several small habitat patches: A community perspective on their importance for biodiversity. Basic and Applied Ecology, v. 65, 16-27. https://doi.org/10.1016/j.baae.2022.09.004.

Teixeira, M.G.; Venticinque, E.M.; Lion, M.B.; Pinto, M.P., 2021. The Brazilian Caatinga protected areas: an extremely unbalanced conservation system. Environmental Conservation, v. 48, (4), 287-294. https://doi.org/10.1017/S0376892921000308.

The Jamovi Project, 2022. Jamovi. (Version 2.3) [Computer Software] (Accessed on July 20, 2023) at:. https://www.jamovi.org.

Torgeski, M.R.; Carvalho, E.M.; Santos, A.M.; Lemke, A.P.; Fioratti, C.A.G. et al., 2025. Spatiotemporal analysis of land use and cover dynamics in protected areas of the Brazilian cerrado. Scientifica, v. 2025, (1), 7984448. https://doi.org/10.1155/sci5/7984448.

Turner, I.M., 1996. Species loss in fragments of tropical rain forest: a review of the evidence. Journal of Applied Ecology, v. 33, (2), 200-209. https://doi.org/10.2307/2404743.

UNEP-WCMC; ICUN, 2016. Protected Planet Report 2016. UNEP-WCMC and ICUN, Cambridge, UK, and Gland, Switzerland (Accessed February 18, 2023) at:. http://wdpa.s3.amazonaws.com/Protected_Planet_Reports/2445%20Global%20Protected%20Planet%202016_WEB.pdf.

Vitalli, P.L.; Zakia, M.J.B.; Durigan, G., 2009. Considerações sobre a legislação correlata à zona-tampão de unidades de conservação no Brasil. Ambiente & Sociedade, v. 12, (1), 67-82. https://doi.org/10.1590/S1414-753X2009000100006.

Zardini, F.P.F.; Sousa, J.C.; Martins, P.T.A., 2016. Meio físico e patrimonialização de áreas: elementos para a conservação do bioma cerrado no norte goiano? ACTA Geográfica, v. 10, (22), 1-16. https://doi.org/10.18227/2177-4307.acta.v10i22.2345.

Wiersma, Y.F.; Nudds, T.D.; Rivard, D.H., 2004. Models to distinguish effects of landscape patterns and human population pressures associated with species loss in Canadian national parks. Landscape Ecology, v. 19, 773-786. https://doi.org/10.1007/s10980-005-3989-y.

Woodman, T.L.; Alexander, P.; Burslem, D.F.R.P.; Travis, J.M.J.; Winkler, K. et al., 2025. Global assessment of landscape pattern changes from 1992 to 2020. Landscape Ecolology, v. 40, 196. https://doi.org/10.1007/s10980-025-02210-0.

Yang, C.; Dong, J.; Jenkins, C.N.; Zhang, X.; Li, Y. et al., 2025. Identifying conservation priorities in global Biodiversity Hotspots to protect small-ranged vertebrates from agricultural pressure. Communications Earth & Environment, v. 7, 78. https://doi.org/10.1038/s43247-025-03099-y.