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Post-agreement acceleration of habitat loss and landscape connectivity decline in and around Colombian protected areas

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E-mail: kristian.rubiano@urosario.edu.co**Keywords:** Colombia, post-agreement, protected areas, land-use/land-cover change, deforestation, non-forest vegetation loss, landscape connectivitySupplementary material for this article is available [online](#)

Abstract

Protected areas (PAs) are increasingly exposed to land-use pressures that erode habitat and landscape connectivity. In Colombia, rapid socio-ecological changes following the 2016 peace agreement marked a major territorial and governance transition and have been associated with heightened habitat loss in and around PAs, yet the magnitude and spatial configuration of habitat and connectivity change across the Colombian System of National Natural Parks remain insufficiently quantified. In this work, we combined the MapBiomias Colombia v2.0 land-use/land-cover dataset with the equivalent connected area index to evaluate deforestation, loss of natural non-forest vegetation (NNFV), and landscape connectivity dynamics for 50 PAs and their immediate surroundings (0–10 km buffers), and broader surroundings (10–20 km buffers) across two periods bracketing the peace agreement (2009–2016 and 2016–2023). We found that deforestation and NNFV loss intensified after 2016 across most regions, with particularly pronounced signals in the Andes–Amazon transition. Overall, landscape connectivity declined across all zones and periods. Within PAs, reductions were comparatively modest, and most habitat remained connected by 2023, suggesting some persistence of internal structural integrity. In surrounding buffers, declines were stronger and not fully explained by habitat amount alone, as connectivity fell faster than area loss, indicating stronger configuration effects and increasing fragmentation of natural-vegetation networks. Taken together, these results suggest that the post-agreement period coincided with intensified habitat and disproportionate connectivity erosion in landscapes surrounding PAs. By extending national-scale monitoring through 2023 and, most importantly, by evaluating how habitat-loss trajectories translated into connectivity outcomes across 50 strictly PAs and two surrounding buffer bands, this study provides an updated basis for identifying where habitat conversion and isolation pressures are most strongly concentrated.

1. Introduction

Habitat loss and degradation caused by anthropogenic land use are widely recognized as the main drivers of global biodiversity decline (Pereira *et al* 2012, Newbold *et al* 2015). Tropical ecosystems are particularly vulnerable, as they harbor more than 60% of global biodiversity (Bradshaw *et al* 2009) and continue to experience some of the highest deforestation rates worldwide, with an average annual loss of 7.29 million ha of tropical moist forest between 1990 and 2019 (Vancutsem *et al* 2021). In response, the number and extent of protected areas (PAs), key strategies for biodiversity conservation, have increased substantially across the tropics over recent decades (Jenkins and Joppa 2009, Watson *et al* 2014). However, PAs are often exposed to multiple anthropogenic pressures that compromise their

effectiveness in conserving biodiversity and sustaining ecosystem services (Laurance *et al* 2012). Habitat and connectivity loss around PAs is of particular concern because these areas are linked to surrounding lands by key biotic and abiotic processes. Buffer areas play a crucial role in maintaining ecological flows, contributing strongly to PA integrity and counteracting ecological isolation (DeFries *et al* 2005, Hansen and DeFries 2007, Prugh *et al* 2008). Monitoring PAs and their surroundings jointly is thus essential for detecting threats, assessing the progress of conservation goals, and guiding management decisions (Willis 2015). Remote sensing-based land-use and land-cover (LULC) monitoring platforms provide critical spatial information for decision-makers and PA managers, offering valuable insights into habitat change, degradation, and connectivity loss (Nagendra *et al* 2013).

Colombia is one of the megadiverse countries, harboring around 10% of global biodiversity (Moreno *et al* 2018, Convention on Biological Diversity 2025), largely as a result of its geomorphological and climatic heterogeneity, which has fostered complex biogeographic diversification processes and a wide variety of ecosystems (Kattan *et al* 2004, Hazzi *et al* 2018). These features make Colombia a country with exceptional natural capital and of major global importance for biodiversity conservation (Ocampo-Peñuela and Winton 2017, Bax and Francesconi 2019). Currently, about 17% of the national territory is protected under the Colombian National System of PAs, encompassing about 1800 PAs (PNNC 2025) that harbor more than 70% of the country's bird species and approximately 39% of its vascular plants (Franco *et al* 2007, Mendoza-Cifuentes *et al* 2018). PAs with the most restrictive IUCN protection categories (IA, IB, II, and III) are grouped under the System of National Natural Parks (SPNN), which is managed by the national government. The SPNN includes 61 PAs covering nearly 146 200 km² of territory (PNNC 2025). PAs under these more restrictive conservation categories do not allow productive activities and are open only to tourism and recreation, while others serve as reference areas for strict protection and scientific research (Dudley 2008).

Despite this extensive PA network, Colombia's natural capital faces increasing anthropogenic pressures (Correa Ayram *et al* 2020), including illegal cattle ranching, agricultural expansion, illicit crop cultivation, and mining, all of which have been identified as important drivers of habitat loss and declining landscape connectivity in and around PAs (Dávalos *et al* 2011, González-González *et al* 2021). For instance, between 2023 and 2024, about 1930 km² of forest were lost in Colombia, of which around 8% occurred inside PAs (IDEAM and MADS 2025). These long-standing pressures have interacted with changing patterns of territorial control, governance, and accessibility, particularly in remote areas historically affected by armed conflict.

More broadly, transitions from armed conflict to ceasefire, demobilization, or peace-agreement settings can trigger major socioecological reorganization. Research framed within warfare ecology has shown that such transitions may alter territorial control, governance, access, migration, infrastructure expansion, and the intensity and geography of natural-resource use, with consequences that can either increase or reduce environmental degradation depending on local institutional and political conditions (Machlis and Hanson 2008, Hanson *et al* 2009, Gaynor *et al* 2016, Hanson 2018, Suarez *et al* 2018, Landholm *et al* 2019). In Colombia, the 2016 final peace agreement between the national government and the FARC-EP marked a major territorial and institutional transition that reshaped land-use dynamics in many remote regions. In this study, we use the term post-agreement to refer specifically to the period following the 2016 peace agreement as a temporal and political reference point. This terminology acknowledges that armed violence persisted and reconfigured after the accord and avoids implying that Colombia entered a fully settled post-conflict condition.

The Colombian armed conflict had diverging effects on LULC dynamics. On the one hand, some authors have described a form of *gunpoint conservation*, whereby the territorial presence of armed groups restricted access, limited extractive activities, and constrained infrastructure expansion in remote areas (Murillo-Sandoval *et al* 2020). Forced displacement from rural to urban areas also promoted ecosystem regeneration in many areas previously used for agriculture (Sánchez-Cuervo and Aide 2013a). On the other hand, illicit cropping, cattle ranching, and other activities used to finance armed groups also promoted deforestation in several regions (Dávalos *et al* 2011, Negret *et al* 2019, Mendoza 2020). These contrasting dynamics were particularly common in remote PAs and surrounding landscapes historically occupied by guerrilla groups (Canavire-Bacarreza *et al* 2018, Hoffmann *et al* 2018).

Remote-sensing studies have documented rapid forest loss after 2016, particularly in the Amazon and Andean–Amazon foothills, and have linked these patterns to land grabbing, agricultural expansion, weak governance, and shifting territorial control (Armenteras *et al* 2019, Negret *et al* 2019, Clerici *et al* 2020, Mendoza 2020, Murillo-Sandoval *et al* 2020, 2021). Illegal crop expansion has also been reported in several regions (Negret *et al* 2019, Mendoza 2020, UNODC-SIMCI 2021), and forest connectivity has declined sharply across the Colombian arc of deforestation (Murillo-Sandoval *et al* 2022, López *et al* 2024). These studies provide an essential basis for understanding environmental change after 2016.

However, most studies have focused primarily on deforestation, have evaluated a limited number of PAs, and have rarely integrated habitat-conversion trajectories with landscape connectivity outcomes across immediate and broader PA surroundings. While increases in forest loss after 2016 are already well documented, comparatively little is known about how habitat-loss patterns observed during the post-agreement period translated into connectivity outcomes for PAs and their surrounding landscapes at the national scale. Comparatively less attention has also been given to natural non-forest vegetation (NNFV), explicit transition destinations, and to whether connectivity decline tracked or exceeded habitat loss across different surrounding landscapes.

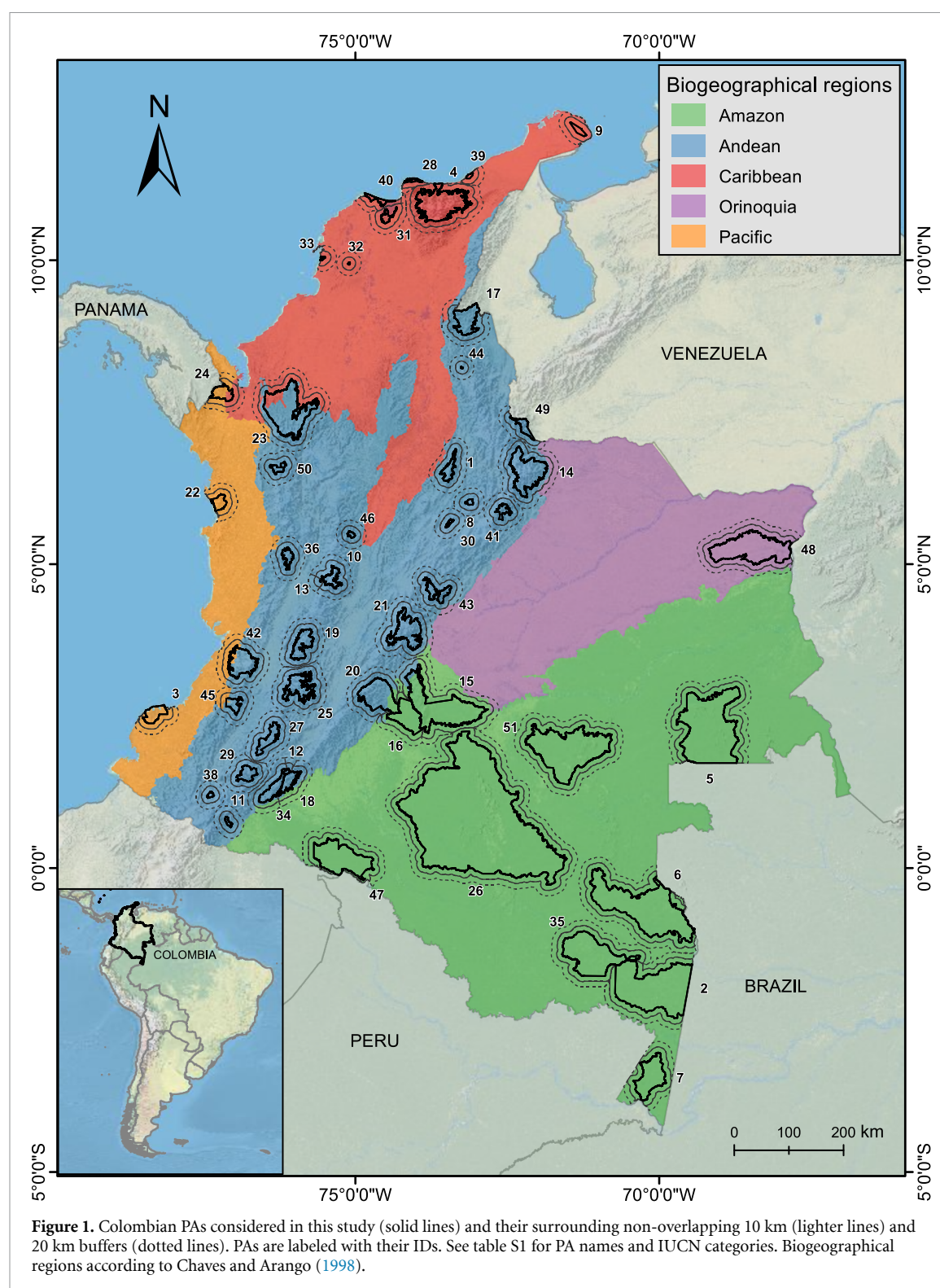
Accordingly, this study addresses three main gaps. First, it extends national-scale monitoring through 2023 across 50 strictly PAs and their surrounding landscapes. Second, it evaluates a broader set of habitat-conversion trajectories by jointly assessing deforestation, loss of NNFV, and combined natural-vegetation (NV) loss, together with their main destination land-use classes. Third, and most importantly, it integrates these habitat-loss trajectories with landscape connectivity metrics to evaluate how habitat loss during the post-agreement period translates into connectivity outcomes across PA cores, the immediate surroundings (0–10 km buffers), and the broader surroundings (10–20 km buffers). This is important because similar amounts of habitat loss can generate very different connectivity outcomes depending on where habitat is removed and how remaining NV is spatially configured. Together, these gaps limit the ability to identify which PAs and surrounding zones are most exposed not only to habitat loss but also to ecological isolation and fragmentation during the post-agreement period. Such information is crucial, as it can inform land-use planning and strengthen conservation strategies (Sierra *et al* 2017, Suarez *et al* 2018, Torres *et al* 2020).

Specifically, we ask: (1) how did deforestation, NNFV loss, and total NV loss differ between the conflict (2009–2016) and post-agreement (2016–2023) periods? (2) How did these changes differ among PAs, immediate surroundings (0–10 km buffers), and broader surroundings (10–20 km buffers)? (3) How did these habitat-loss trajectories translate into changes in landscape connectivity across PAs and their surroundings, and to what extent did connectivity decline track or exceed habitat loss? By addressing these questions, we provide an updated national-scale assessment of habitat loss and changes in connectivity in and around Colombian PAs and discuss the implications of these patterns for biodiversity conservation and land-use planning within the country's broader post-agreement context.

2. Materials and methods

2.1. Study area

This study included 50 of the 61 PAs in the SPNN (see table S1). The PAs excluded from the study (see table S2) were either predominantly covering water bodies (terrestrial area <5% of the extent of the PA), very small (<1 km²), or established after the beginning of the analysis time window (year of establishment > 2009). The size of 1 km² was set to minimize the effects of the 30 m spatial resolution of the LULC dataset (see section 2.2). Collectively, the 50 selected PAs cover about 144 000 km² of land. Also, to evaluate how land-use pressure and connectivity change with distance from PA boundaries, we included two non-overlapping buffers, 0–10 km and 10–20 km, around the 50 PAs (hereinafter, the 10 km and 20 km buffers, respectively; figure 1). These buffers were not intended to reproduce reserve-specific legal or management zones, which vary among PAs, but rather to provide standardized analytical units that distinguish the immediate from the broader surrounding landscape under a consistent national-scale framework. This design is supported by ecological theory, which shows that PA integrity depends on surrounding lands through effects on habitat area, ecological flows, crucial habitats outside reserve boundaries, and edge-related human pressures (Hansen and DeFries 2007, DeFries *et al* 2010). It is also consistent with classic buffer-based analyses of PA isolation in tropical forests (DeFries *et al* 2005) and with applied studies that have used 10 km surrounding buffers or broader multi-distance approaches to quantify conservation-relevant pressures around PAs (Geldmann *et al* 2013), including post-conflict deforestation in Colombia (Clerici *et al* 2020), built-up expansion around PAs globally (de la Fuente *et al* 2020), degradation of tropical buffer zones (de Almeida-rocha and Peres 2021), and heterogeneous spillover effects within 20 km surrounding landscapes (Shen *et al* 2022). PAs boundaries in vector format were obtained from the Colombian National Natural Parks institution (PNNC 2024). The dataset was filtered according to the study areas' selection criteria of this study, using the continental administrative boundaries of Colombia from the Colombian SIGOT system (Instituto Geográfico Agustín Codazzi 2019). The buffer areas around PAs were produced as non-overlapping surrounding bands for contiguous PAs by removing overlaps among adjacent buffers (see figure 1). A detailed description of the workflow is provided in Bottin (2022), and it is briefly summarized here for completeness. First,



influence zones were generated for each PA using a Voronoi/Thiessen approach based on points extracted along PA boundaries; these influence zones were then clipped to continental Colombia and the PA polygons themselves were removed. Finally, the resulting influence zones were intersected with the 0–10 km and 10–20 km buffers around each PA so that overlapping portions between adjacent parks were assigned to the nearest PA, yielding non-overlapping buffer bands. The workflow was implemented in PostgreSQL/PostGIS, with data handling and verification supported in R.

2.2. LULC data

We used data from the MapBiomass Colombia Collection 2.0 dataset (Souza *et al* 2020, MapBiomass Colombia 2024) as the reference to extract trajectory statistics and analyze connectivity changes. This

dataset consists of annual national LULC maps for the period 1985–2023 generated through cloud-based processing. Each annual map is a raster grid with a 30 m spatial resolution, produced from the classification of Landsat TM, ETM+, and OLI imagery mosaics. For every year, mosaics were created by spatial and temporal integration of multiple scenes representing optimal phenological conditions to minimize cloud and seasonal variability. Classification relies on random forest algorithms trained with more than 140 predictor variables, including spectral bands, indices, spectral fractions, topographic metrics, and ancillary variables (Souza *et al* 2020, Fundación Gaia Amazonas 2024).

The MapBiomias Colombia collection 2.0 product adopts a hierarchical LULC legend comprising 21 classes at level 2, aggregated into five level-1 categories. This legend is derived from the national land cover legend adapted from CORINE methodology, and the FAO land cover typologies (Di Gregorio and Jansen 2005, IDEAM 2010). Mean annual overall accuracy reported for MapBiomias Colombia is 97.9% ($\pm 0.5\%$ SD) at level 1 and 91.2% ($\pm 1.8\%$ SD) at level 2 (MapBiomias Colombia 2026). These values reflect overall product performance and should not be interpreted as class-specific validation of the aggregated categories used in this study. Table S3 summarizes the publicly accessible user's and producer's accuracies of the main Level 1 classes most relevant to this study for 2009 and afterward. To date, it constitutes one of the most comprehensive, spatially explicit, and temporally consistent sources of land use and land cover information available for Colombia, serving as the foundation for the subsequent LULC and connectivity analyzes (Fundación Gaia Amazonas 2024, Xu *et al* 2025).

2.3. Data processing

All processing of the MapBiomias Colombia collection 2.0 dataset was carried out on the cloud-based google earth engine platform (Gorelick *et al* 2017) to access, filter, and extract both raster and tabular data required for subsequent analyzes. First, the national LULC raster collection was filtered to select annual maps from 2009 to 2023, which represent land-cover conditions at the end of each year. This temporal range was chosen to encompass both the Colombian conflict (2009–2016) and post-agreement (2016–2023) periods of interest. To spatially constrain the analysis, we used the vector boundaries of PAs previously filtered according to the selection criteria described in section 2.1, together with their 10 km and 20 km buffers generated using a non-overlapping buffer algorithm (see section 2.1). All geometries were dissolved to create a single external boundary corresponding to the 20 km buffer limit, and the annual LULC rasters were clipped to this extent.

Subsequently, the original LULC legend of the dataset was reclassified into four level-1 categories to emphasize distinctions among habitat, anthropogenic land use, and non-vegetated natural areas (table 1). NV classes were grouped into two categories: Forest formation (hereafter Forest) and NNFV based on physiognomic and structural criteria that reflect major differences in vegetation form, such as canopy height, cover density, and the dominance of woody versus non-woody life forms (e.g. shrubs, grasses, or open herbaceous cover). Anthropogenic land-use classes were grouped into a single category, human-modified areas (HMA), encompassing all non-natural classes associated with landscape transformation and human activity. Finally, the Non-vegetated natural area and waterbody (NNW) category includes all non-vegetated natural areas and waterbodies. This reclassification served as the analytical framework for subsequent LULC change and landscape connectivity assessments.

2.4. LULC statistics extraction

We used the reclassified annual LULC maps, along with the PAs' boundaries and their 10 km and 20 km buffers, to extract statistics describing LULC change trajectories. As we aimed to assess transitions directly attributable to anthropogenic pressures, our analysis focused on two forms of anthropogenic NV conversion: deforestation, defined as the direct conversion of forest to HMA (table 1), consistent with the IPCC definition of direct human-induced conversion of forested land to non-forested land (IPCC 2006), and NNFV loss, defined analogously as the conversion of NNFV to HMA. Therefore, transitions between NV types (e.g. from forest to NNFV or vice versa) or toward NNW were not considered deforestation or NNFV loss events.

To quantify cumulative gross anthropogenic conversion within each period while avoiding multiple counting of the same pixel, we applied a monotonic non-increasing loss-accounting approach, following the logic of gross cover loss commonly used in large-scale remote-sensing assessments (Hansen *et al* 2013, Curtis *et al* 2018). Specifically, once a pixel classified as forest or NNFV at the beginning of a period underwent a transition to HMA, it was counted as a single loss event and remained flagged as lost for the remainder of that period for accounting purposes. This rule was not applied to transitions between NV classes, which were excluded from the loss analysis.

Annual estimates of deforestation and NNFV loss were generated for every year as first detected transitions from forest or NNFV to HMA, and cumulative totals for the conflict and post-agreement

Table 1. Level-1 reclassification scheme proposed for LULC change and landscape connectivity assessments, based on the original MapBiomias Colombia collection 2.0 LULC thematic legend (LU, land-use; LC, land-cover).

Habitat/No habitat	Level-1 reclassified class	Level-1 original MapBiomias class	Level-2 original MapBiomias class	LU/LC
Habitat	Forest	Forest formation	Forest	LC
			Mangrove	LC
			Floodable forest	LC
			Wooded sandbank vegetation	LC
	Natural non-forest vegetation (NNFV)	Natural non forest formation	Wetland	LC
			Grassland	LC
			Herbaceous sandbank vegetation	LC
			Other non forest formation	LC
	No habitat	Human-modified areas (HMA)	Forest plantation	LU
			Palm oil	LU
			Mosaic of agriculture and pasture	LU
		Non-vegetated area	Urban infrastructure	LU
			Mining	LU
			Other non-vegetated area	LU
		Non-vegetated natural area and waterbody (NNW)	Hypersaline tidal flat	LC
			Rocky outcrop	LC
		Non-vegetated area	Beach, dune and sand spot	LC
			Other natural non-vegetated area	LC
		Water body	River, lake or ocean	LC
			Aquaculture	LC
			Glacier	LC

periods were obtained by summing these annual first-loss events, with each pixel counted at most once per period. Accordingly, period totals represent cumulative gross anthropogenic loss rather than net end-of-period change. To better understand the trajectories of land-cover change, we identified the main destination land-use classes associated with deforestation and NNFV loss and evaluated their relative contributions between the two periods using both level-1 and disaggregated level-2 LULC classes.

Furthermore, to complement the separate assessments of deforestation and NNFV loss, we analyzed overall NV loss by combining both metrics into a unified indicator of total habitat conversion (hereafter referred to as NV loss). This allowed an objective comparison between predominantly forested and non-forested PAs and an evaluation of differences in habitat and connectivity outcomes between the conflict and post-agreement periods.

Temporal trend analyzes were then performed to evaluate changes in loss rates between the conflict and post-agreement periods. To quantify these changes, we calculated the percentage difference between periods for each PA and buffer zone as the relative change of post-agreement values with respect to conflict-period values (Clerici *et al* 2020). Percentage changes were also computed by aggregating PAs and buffer zones according to biogeographical regions, following the regionalization proposed by Chaves and Arango (1998).

2.5. Landscape connectivity analysis

We used the equivalent connected area (ECA) index (Saura *et al* 2011) to quantify the effects of habitat conversion during the conflict and post-agreement periods on the connectivity of NV networks within PAs and their surrounding buffer zones. ECA represents the area of a single, hypothetical, fully connected habitat patch that would provide the same probability of connectivity as the actual configuration of habitat patches in the landscape. In this sense, ECA expresses the effective amount of connected habitat, integrating both intra-patch area, and the probability of connection among patches under a defined dispersal-distance threshold into a single, ecologically meaningful metric (Saura and Pascual-Hortal 2007, Saura *et al* 2011). This makes ECA particularly useful for multitemporal comparisons, because it captures not only changes in habitat amount or spatial subdivision, as purely structural connectivity and fragmentation metrics do, but also their implications for potential functional connectivity (Tischendorf and Fahrig 2000, Calabrese and Fagan 2004, Oettel *et al* 2025).

To assess connectivity change between years within each zone type, we used the Makurhini v3.0 R package (R Core Team 2024, Godínez-Gómez *et al* 2025) to compute the ECA index for the aggregated

extent of all PAs combined, and separately for the 10 km and 20 km buffer zones. This zone-specific approach was intended to compare changes in the internal connectivity of NV networks within PAs and within their surrounding buffer landscapes. It was not designed to quantify connectivity between buffer habitats and PA cores. We reclassified the 2009, 2016, and 2023 LULC maps to a binary habitat map (Forest and NNFV classes combined; see table 1) and delineated habitat nodes as contiguous habitat patches. Because the connectivity analysis was based on a binary NV layer (Forest + NNFV) and Euclidean inter-patch distances, the results represent generic NV connectivity under a homogeneous-matrix assumption rather than species-specific habitat connectivity. Pairwise Euclidean distance matrices among habitat nodes were generated with the *Generate Near Table* tool in ArcMap 10.8. Accordingly, this analysis does not explicitly account for variation in movement costs or permeability among different HMA types.

To assess the sensitivity of connectivity patterns to alternative generic dispersal assumptions, we computed ECA using six median dispersal distances ($d = 1, 5, 10, 30, 50,$ and 70 km), spanning thresholds commonly used in Colombia (Castillo *et al* 2020, Murillo-Sandoval *et al* 2022, Correa-Ayram *et al* 2025). We adopted 10 km as the reference d for result interpretation, as it represents a central benchmark within the tested range of dispersal distances and has been widely used in large-scale connectivity analyses (see e.g. Martensen *et al* 2017, Murillo-Sandoval *et al* 2017, Saura *et al* 2017, Correa-Ayram *et al* 2025). ECA was calculated in absolute units (km^2) and later normalized by the total habitat area to obtain the normalized ECA (ECA_{norm}).

We also calculated other ECA-derived metrics helpful to assess spatiotemporal patterns of landscape connectivity. To assess trends in connectivity across periods and zone types, we computed the percentage change in the ECA (dECA) for the conflict and post-agreement periods as the difference between ECA values over time, expressed as a percentage of the initial value (Saura *et al* 2011). Negative dECA values indicate a loss of connectivity, whereas positive values indicate a gain in connectivity over the corresponding period. To identify how connectivity changes are related to changes in habitat area, we also estimated the relativized ECA (rECA), by dividing the dECA by the change in habitat area (dA) (Dilts *et al* 2016, Liang *et al* 2021). rECA values greater than 1 indicate that habitat changes result in a disproportionately large change in habitat connectivity, whereas values lower than 1 indicate lower impacts of habitat loss on connectivity (Saura *et al* 2011, Godínez-Gómez *et al* 2025).

Subsequently, the ECA and related connectivity metrics were also calculated for each individual PA and buffer zone, using 10 km as d value. The multi-distance analysis was used to assess sensitivity at the overall scale, whereas the PA-level analysis was performed only for the 10 km reference distance as a common benchmark for spatial prioritization. Finally, to compare connectivity outcomes between the post-agreement and conflict periods, we calculated the inter-period change in connectivity as:

$$\Delta\text{dECA} = \text{dECA}_{2009-2016} - \text{dECA}_{2016-2023}$$

where $\text{dECA}_{2009-2016}$ and $\text{dECA}_{2016-2023}$ correspond to the percentage change in ECA during the conflict and post-agreement periods, respectively. Negative ΔdECA values indicate that connectivity outcomes became more negative in the post-agreement period (i.e. connectivity loss intensified or connectivity gains weakened), whereas positive ΔdECA values indicate that connectivity outcomes became less negative or more positive in the post-agreement period (i.e. connectivity loss was reduced or connectivity gains increased).

Because identical ΔdECA values may arise from different combinations of period-specific dECA values, inter-period changes were interpreted jointly with the sign of dECA in each period. To facilitate interpretation, inter-period connectivity outcomes were also classified into six qualitative change categories based on the sign of dECA in each period and on the sign of ΔdECA : intensified loss, reduced loss, shift from loss to gain, shift from gain to loss, stronger gain, and weakened gain (table S4). These categories were used to identify the most vulnerable and most improved areas in terms of connectivity outcomes during the post-agreement period.

2.6. Statistical analysis

To evaluate temporal and spatial differences in patterns of habitat loss, we analyzed deforestation, NNFV loss, and NV loss for the conflict and post-agreement periods. For between-period comparisons within each zone type (PAs, 10 km buffers, and 20 km buffers), we used paired Wilcoxon signed-rank tests. Because our *a priori* expectation was that habitat loss would be greater during the post-agreement period than during the conflict period, we used a one-tailed test. This resulted in 9 one-tailed Wilcoxon tests for habitat-loss metrics (3 response variables $\times 3$ zone types). To assess differences among the three

zone types within each period, we applied the Friedman test to each response variable and period separately. When the omnibus Friedman test was significant, it was followed by three pairwise Wilcoxon signed-rank tests comparing PAs vs 10 km buffers, PAs vs 20 km buffers, and 10 km vs 20 km buffers. Holm-adjusted p -values were applied only within each response-variable $\times$ period family of post hoc comparisons. Nonparametric methods were selected because of non-normal residuals (Dytham 2011). These analyzes allowed us to identify both the magnitude and direction of habitat change across spatial contexts and time periods.

An equivalent statistical approach was applied to the dECA data to assess variations in connectivity change between the conflict and the post-agreement periods. For between-period comparisons within each zone type, we used three one-tailed paired Wilcoxon signed-rank tests to assess whether dECA values were more negative during the post-agreement than during the conflict period, indicating less favorable connectivity outcomes. To examine differences among zone types within each period, we applied Friedman tests, followed, when significant, by pairwise Wilcoxon signed-rank tests, with Holm-adjusted p -values applied within each period-specific post hoc family. All statistical analyzes were conducted in R (R Core Team 2024).

3. Results

3.1. Overall and regional trends of deforestation and NNFV loss

Within PAs, annual deforestation exhibited a slight increase during the conflict period, averaging $134 \pm 15 \text{ km}^2$ (figure 2(A)). After 2016, deforestation sharply increased, peaking at 316 km^2 in 2018. Similar increases in annual deforestation were observed in the 10 km (figure 2(B)) and 20 km buffers (figure 2(C)), where mean annual forest loss during the conflict ($463 \pm 41 \text{ km}^2$ and $523 \pm 57 \text{ km}^2$, respectively) increased after 2016, reaching maximum values of 834 km^2 and 828 km^2 in 2018.

Annual NNFV loss also intensified across periods, exhibiting spatial and temporal patterns similar to those of deforestation. Within PAs, maximum annual NNFV loss during the conflict was 21 km^2 in 2016 and grew nearly fourfold during the post-agreement period, reaching 81 km^2 in 2023 (figure 2(D)). In the 10 km buffers, losses decreased during the conflict and peaked in 2017 and 2023 (figure 2(E)), whereas in the 20 km buffers, they consistently declined during the conflict (from a maximum in 2010 to a minimum in 2016) but rapidly increased again after 2016 (figure 2(F)).

Deforestation increased across all biogeographical regions from the conflict to the post-agreement period, with the most pronounced growth occurring in the Amazon region (table 2; figures 2 and S1). Within PAs, deforestation in the Amazon rose from 431 km^2 to 1196 km^2 ($+765 \text{ km}^2$), accounting for the largest absolute increase among all regions. Similar increases in deforestation were recorded in the surrounding 10 km and 20 km buffers, where deforestation more than doubled between periods ($+1336 \text{ km}^2$ and $+803 \text{ km}^2$, respectively). In the Andean region, deforestation also increased notably within PAs ($+181 \text{ km}^2$) and in the 10 km and 20 km buffers ($+362 \text{ km}^2$ and $+204 \text{ km}^2$, respectively). Smaller yet consistent increases were detected in the Caribbean and Pacific regions, whereas the Orinoquia showed a marked relative rise despite its small absolute area (from 3.8 km^2 to 33.2 km^2 within PAs; see figure S1 for relative changes).

Patterns of NNFV loss differed among regions (table 2; figures 2 and S2). In the Amazon, NNFV loss intensified across all spatial zones, particularly within the 10 km buffer ($+63.9 \text{ km}^2$). The Caribbean and Orinoquia regions also exhibited substantial increases ($+74.8 \text{ km}^2$ and $+116.6 \text{ km}^2$ within PAs, respectively), while the Andean and Pacific regions showed slight declines or negligible changes. Notably, the Andean region displayed a reduction of NNFV loss across all spatial scales, suggesting conversion dynamics distinct from the deforestation trends observed in the same area. Relative changes of NNFV loss across regions are shown in figure S2.

3.2. Dynamics of LULC transitions

Analysis of land-use trajectories revealed that most deforestation and NNFV losses were driven by conversions to agricultural and pasture lands (table 3). During both periods, this class accounted for over 95% of all deforestation and more than 85% of NNFV loss across spatial zones. Within PAs, 96.4% of deforestation during 2009–2016 and 95.5% during 2016–2023 transitioned to this class while smaller proportions shifted toward forest plantations (0.1%–0.04%) or mining areas (0.1%–0.02%). A similar dominance was observed in the 10 km and 20 km buffers, where agricultural and pasture lands accounted for approximately 98% of total forest conversion. Oil palm expansion increased notably in the post-conflict period within PAs, rising from 0.2% to 2.3% of total deforestation, while remaining below 1% in surrounding buffers. Mining and infrastructure maintained low, relatively stable proportions ($<0.2\%$),

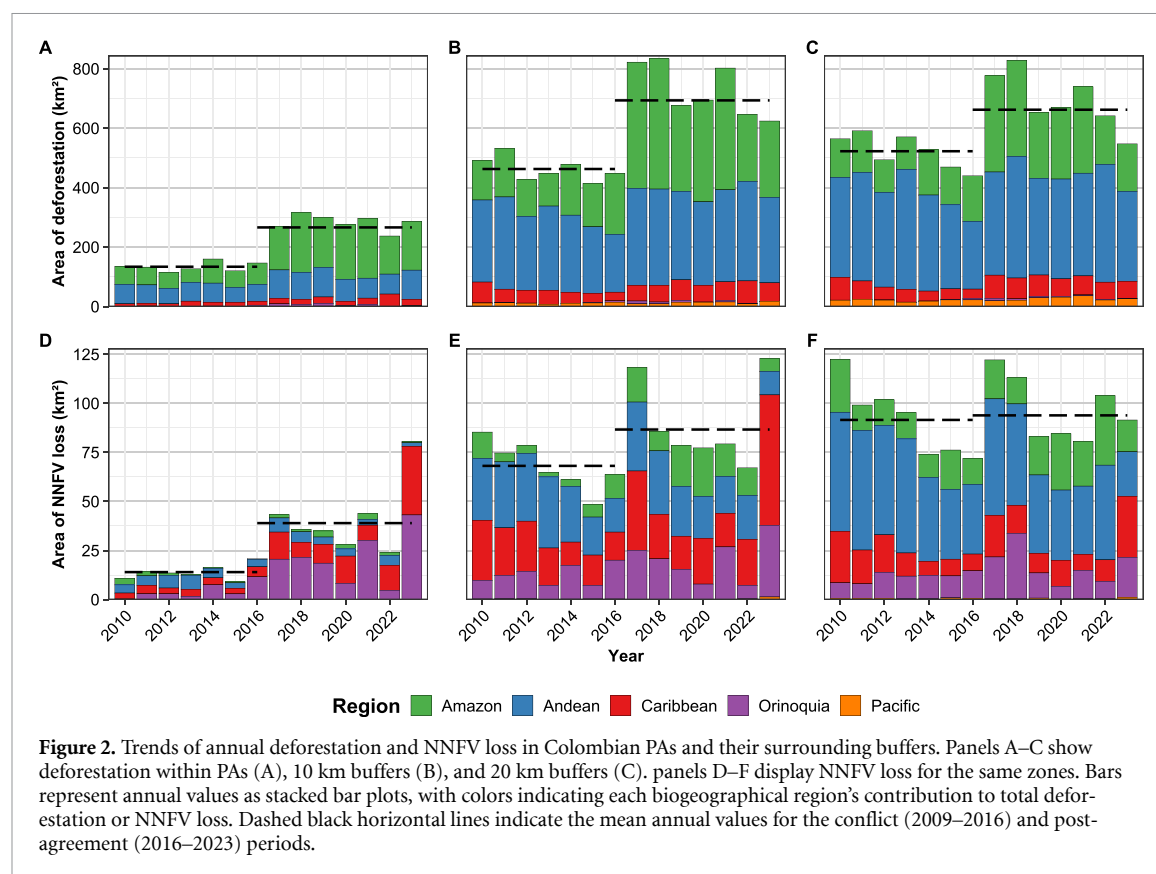


Table 2. Deforestation and NNFV loss within PAs and their surrounding 10 km and 20 km buffers, by biogeographical region, during the conflict (t_0) and post-agreement (t_1) periods, and their absolute difference (t_1-t_0). n indicates the number of PAs in every region.

Region	n	PA			10 km buffer			20 km buffer		
		t_0	t_1	t_1-t_0	t_0	t_1	t_1-t_0	t_0	t_1	t_1-t_0
Deforestation (km ²)										
Amazon	10	431.4	1196.2	764.8	1052.8	2388.3	1335.5	923.9	1726.9	802.9
Andean	28	412.6	593.4	180.8	1797.5	2159.9	362.4	2257.0	2460.6	203.6
Caribbean	8	82.9	150.6	67.7	302.5	435.4	132.9	326.0	461.9	135.8
Orinoquia	1	3.8	33.2	29.4	8.3	28.8	20.5	7.0	23.0	16.0
Pacific	3	7.4	10.6	3.2	79.4	89.6	10.2	145.0	186.5	41.5
NNFV loss (km ²)										
Amazon	10	8.9	13.7	4.8	46.8	110.8	63.9	111.2	156.0	44.8
Andean	28	33.5	30.0	−3.5	200.4	167.6	−32.8	349.6	293.0	−56.6
Caribbean	8	25.3	100.2	74.8	141.6	210.7	69.1	98.7	109.0	10.3
Orinoquia	1	29.8	146.5	116.6	86.0	138.1	52.1	77.2	117.4	40.2
Pacific	3	0.4	0.4	0.05	2.2	2.4	0.2	4.2	3.8	−0.5

suggesting limited influence on the direct processes of habitat loss. Losses of NNFV followed comparable trajectories, with the mosaic of agriculture and pasture remaining the primary destination (86–93%) across all zones. However, compared to the forest, a larger proportion of NNFV moved to other non-vegetated areas, with increases within PAs and declines in buffers between periods.

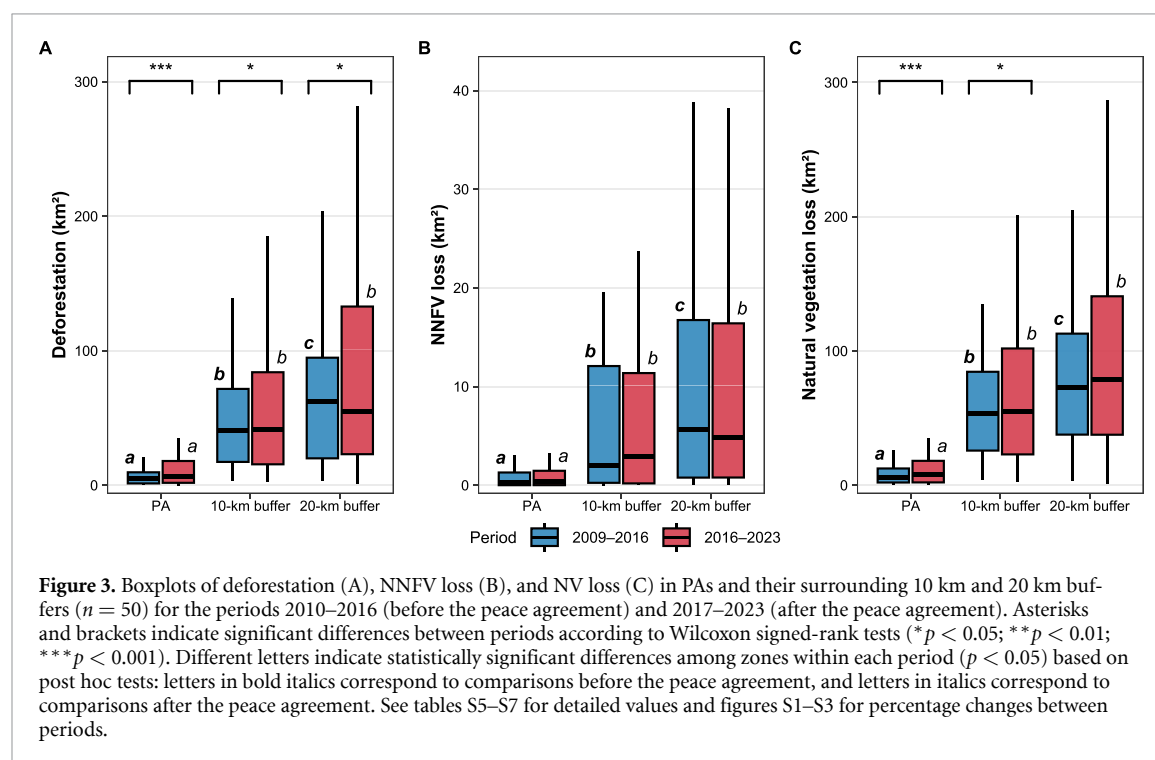
3.3. Changes in NV loss

Statistical analysis revealed significant increases in deforestation during the post-agreement period across PAs, and their surrounding 10 km and 20 km buffer, with moderate to large effect sizes (table 4; figure 3(A)). Overall, 34 of the 50 PAs (68%) experienced higher deforestation rates, representing an additional 1046 km² (+111%) of forest loss compared with the conflict period (table S5; figure S1). The largest absolute increases occurred in the Tinigua (447 km²), Sierra de la Macarena (222 km²), and Serranía de Chiribiquete (85 km²) national natural parks, all located in the Amazon region.

Table 3. Percentage of deforestation and NNFV loss in PAs and their surrounding 10 km and 20 km buffers during 2009–2016 and 2016–2023, disaggregated by destination land-use class.

Destination land-use class	2009–2016			2016–2023		
	Protected areas	10 km buffers	20 km buffers	Protected areas	10 km buffers	20 km buffers
Deforestation (%)						
Forest plantation	0.1	0.04	0.3	0.04	0.1	0.3
Infrastructure	0.001	0.02	0.01	0.001	0.004	0.01
Mining	0.1	0.1	0.1	0.02	0.1	0.2
Mosaic of agriculture and pasture	96.4	97.6	98.1	95.5	98.1	98.1
Oil palm	0.2	0.3	0.2	2.3	0.5	0.3
Other non-vegetated area ^a	3.2	1.9	1.2	2.1	1.2	1.0
Natural non-forest vegetation loss (%)						
Forest plantation	0.05	0.7	1.2	0.002	0.5	1.5
Infrastructure	0.001	0.3	0.1	0.01	0.2	0.1
Mining	0.002	0.1	0.3	0.005	0.1	0.2
Mosaic of agriculture and pasture	93.5	90.2	86.2	89.8	91.2	90.6
Oil palm	0.0	0.3	0.3	0.001	0.4	0.3
Other non-vegetated area ^a	6.5	8.4	12.0	10.2	7.63	7.2

^a Areas of anthropogenic origin (infrastructure, urban expansion, or mining) not mapped within their respective classes and soils devoid of vegetation or sparse vegetative cover. It also includes burned areas and agricultural lands under preparation or fallow (Fundación Gaia Amazonas 2024).



In the 10 km buffers, 29 areas (58%) showed increased deforestation, totaling an additional 1862 km² (+57%). The most affected were those surrounding Serranía de Chiribiquete (617 km²), Sierra de la Macarena (496 km²), and Cordillera de los Picachos (210 km²) NNPs. A comparable pattern emerged in the 20 km buffers, where 30 zones exhibited deforestation increases amounting to 1200 km² (+33%). Again, buffers around Serranía de Chiribiquete (700 km²), Sierra de la Macarena (471 km²), and cordillera de los Picachos (216 km²) NNPs recorded the highest increments.

Loss of NNFV also increased between periods within PAs and 10 km buffers, though changes were not statistically significant (table 4; figure 3(B)). Among the 50 PAs, 22 (44%) showed higher NNFV losses during the post-agreement period, totaling 193 km² (+197%), while 25 (50%) of the 10 km buffers exhibited an additional 153 km² (+32%) of loss (table S6; figure S2). In contrast, NNFV loss decreased by 20 (40%) of the 20 km buffers. Although not significant overall, these variations reveal

Table 4. Results of statistical tests comparing deforestation, NNFV loss, and NV loss between periods (2009–2016 vs 2016–2023; Wilcoxon signed-rank test) across zones and between zones (protected areas, 10 km buffers, 20 km buffers; Friedman test) across periods. Statistically significant results ($p < 0.05$) are highlighted in bold. r_{rb} = rank-biserial correlation (effect size for Wilcoxon test); W = Kendall's W (effect size for Friedman test).

LULC change trajectory	Between periods			Between zones	
	Protected areas	10 km buffers	20 km buffers	2009–2016	2016–2023
Deforestation	V = 993; p < 0.001; $r_{rb} = 0.56$	V = 853; p = 0.019; $r_{rb} = 0.34$	V = 836; p = 0.028; $r_{rb} = 0.31$	$\chi^2 = 59.1;$ p < 0.001; W = 0.59	$\chi^2 = 44.3;$ p < 0.001; W = 0.44
NNFV loss	V = 542; p = 0.204; $r_{rb} = 0.15$	V = 657; p = 0.241; $r_{rb} = 0.12$	V = 515; p = 0.882; $r_{rb} = -0.19$	$\chi^2 = 51.1;$ p < 0.001; W = 0.51	$\chi^2 = 42.71;$ p < 0.001; W = 0.43
NV loss	V = 996; p < 0.001; $r_{rb} = 0.56$	V = 821; p = 0.039; $r_{rb} = 0.29$	V = 786; p = 0.077; $r_{rb} = 0.23$	$\chi^2 = 59.1;$ p < 0.001; W = 0.59	$\chi^2 = 46.8;$ p < 0.001; W = 0.47

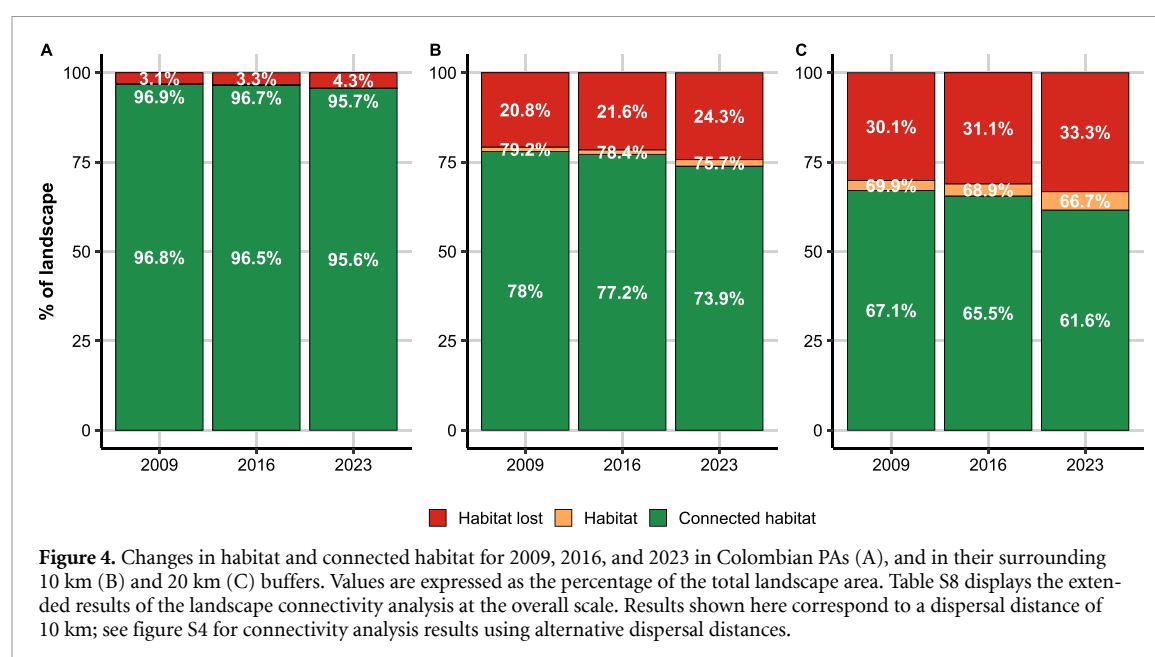


Figure 4. Changes in habitat and connected habitat for 2009, 2016, and 2023 in Colombian PAs (A), and in their surrounding 10 km (B) and 20 km (C) buffers. Values are expressed as the percentage of the total landscape area. Table S8 displays the extended results of the landscape connectivity analysis at the overall scale. Results shown here correspond to a dispersal distance of 10 km; see figure S4 for connectivity analysis results using alternative dispersal distances.

considerable spatial heterogeneity, with some PAs experiencing marked post-agreement increases in NNFV loss.

NV loss (combining deforestation and NNFV loss) showed significant increases between periods in both PAs (+1239 km²; 120%) and 10 km buffers (2014 km²; 54%), with large and moderate effect sizes, respectively (table 4). In 34 PAs (68%), NV loss rates intensified, with the steepest increases in Tinigua (447 km²; 397%), Sierra de la Macarena (222 km²; 166%) y El Tuparro (146 km²; 434%) NNPs (table S7, figure S3).

Comparisons between zones across periods (Friedman tests) revealed significant differences with moderate to large effect sizes for all three LULC change trajectories (table 4; figure 3(C)). Post-hoc analyzes showed consistent differences between PAs and their buffer zones in both periods, whereas contrasts between 10 km and 20 km buffers were evident only during the conflict period (figure 3).

3.4. Overall changes in landscape connectivity

PAs and their surrounding 10 km and 20 km buffer zones experienced overall declines in landscape connectivity during both the conflict and post-agreement periods (figure 4; table S8). Between 2009 and 2023, habitat connectivity within PAs decreased by 1.2% (1688 km²), with a sharper decline in ECA following the peace agreement (−0.9%) compared to the conflict period (−0.3%). In these zones, habitat loss closely matched connectivity loss ($rECA \approx 1$) during the entire analysis time window, but with variations in specific periods $rECA$ (table S8). The outcome by 2023 was 95.6% of the landscape covered by connected habitat (figure 4(A)).

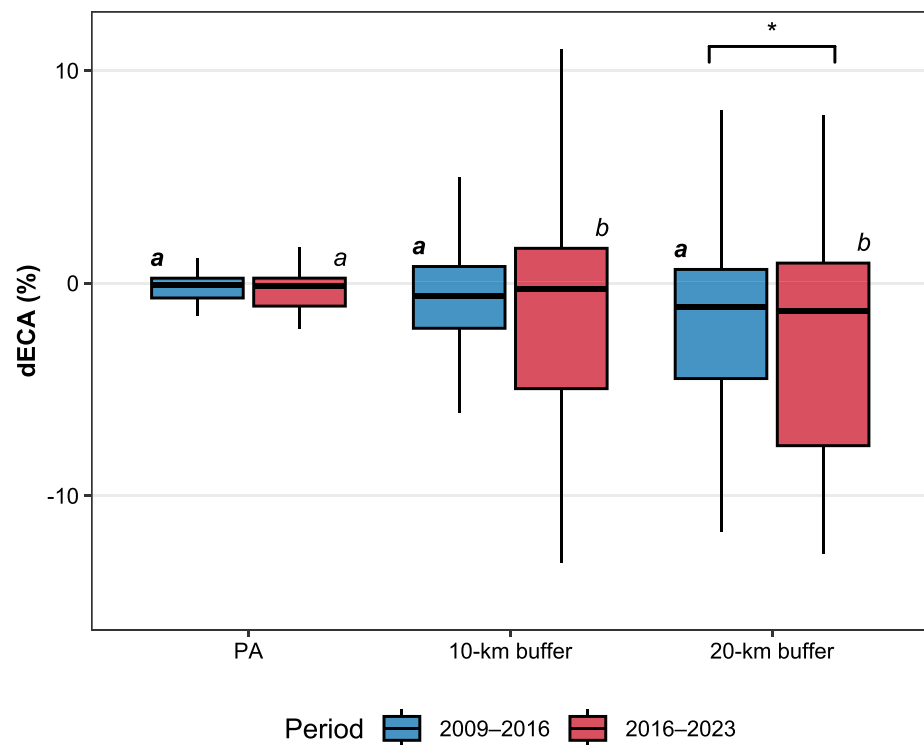


Figure 5. Boxplots of dECA in protected areas (PAs) and their surrounding 10 km and 20 km buffers ($n = 50$) for the periods 2009–2016 (before the peace agreement) and 2016–2023 (after the peace agreement). Asterisks and brackets indicate significant differences between periods according to Wilcoxon signed-rank tests ($*p < 0.05$). Different letters indicate statistically significant differences among zones within each period ($p < 0.05$) based on post hoc tests: letters in bold italics correspond to comparisons before the peace agreement, and letters in italics correspond to comparisons after the peace agreement. See tables S9–S10 for detailed values and figure S5 for the spatial distribution of dECA.

In contrast, buffer zones exhibited more pronounced spatial and temporal fluctuations in both habitat area and connectivity. By 2023, the 10 km buffers had lost 3875 km² of habitat and 4326 km² of connected habitat relative to 2009, representing reductions of 4.4% and 5.3%, respectively. Connected habitat loss intensified after the peace agreement, when 4.3% of connected areas disappeared compared to only 1% during the conflict period. By the end of the study period, connected habitat covered 74% of the 10 km buffer landscapes (figure 4(B)).

Similarly, the 20 km buffers lost 3473 km² of habitat (0.3%) and 6062 km² of connected habitat (8.9%) between 2009 and 2023. Connected habitat declined more rapidly after the peace agreement, with dECA values of 6% post-agreement and 2.3% during the conflict. Corresponding rECA values of 1.7 and 1.9 suggest that connected habitat loss exceeded the rate of habitat loss. By 2023, only 61.6% of the outer buffer landscapes remained as connected habitat (figure 4(C)).

Figure S4 shows the results of the overall landscape connectivity analysis using alternative dispersal values (d). Across these alternative dispersal distances, the broad direction of change remained consistent. Connectivity declines were comparatively small within PAs and stronger in the surrounding 10 km and 20 km buffers, although the absolute amount of connected habitat varied with d .

Although statistical differences were not significant (Wilcoxon $V = 525$, $p = 0.140$, $r_{tb} = -0.176$; figure 5), individual PAs exhibited heterogeneous inter-period connectivity outcomes (figure 6; tables S9 and S10). 22 of them showed positive Δ dECA values, indicating reductions in connectivity loss between periods, and whereas the remaining 28 showed negative Δ dECA values. The strongest deteriorations were observed in Tinigua NNP and Los Flamencos FFS (Δ dECA = -24.8% and -12.0% , respectively), whereas Los Estoraques and Isla de Salamanca showed the largest improvements relative to the conflict period (Δ dECA = 36.6% and 31.8% , respectively).

Regarding the 10 km buffers, no significant differences were detected between periods (Wilcoxon $V = 534$, $p = 0.160$, $r_{tb} = -0.162$; figure 5). In total, 29 buffers exhibited positive Δ dECA values, indicating less negative or more positive post-agreement connectivity outcomes, whereas 21 showed negative Δ dECA values, indicating more negative post-agreement outcomes. The strongest deteriorations were concentrated around the Tinigua and Sierra de la Macarena NNPS (Δ dECA = -28.9%

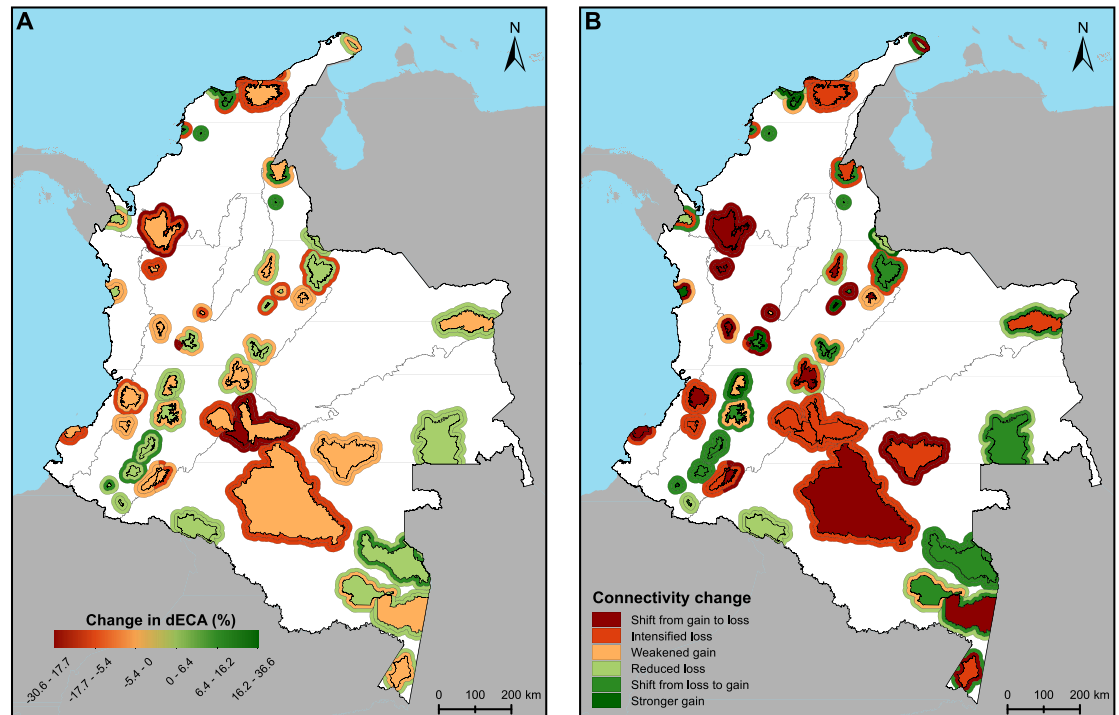


Figure 6. Inter-period change in landscape connectivity outcomes across PAs and their surrounding buffers. (A) Spatial distribution of $\Delta dECA$. Positive values indicate less negative or more positive connectivity outcomes in the post-agreement period relative to the conflict period, whereas negative values indicate more negative connectivity outcomes in the post-agreement period. (B) Qualitative categories of inter-period connectivity change derived from the sign of dECA in each period and the sign of $\Delta dECA$, distinguishing intensified loss, reduced loss, shift from loss to gain, shift from gain to loss, stronger gain, and weakened gain. See figure S5 for the spatial distribution of dECA among periods. Inter-period connectivity change category definitions are provided in table S4.

and -23.6% , respectively). In contrast, the buffers surrounding Los Estoraques and Galeras showed the largest improvements relative to the conflict period ($\Delta dECA = 15.3\%$ and 15.1% , respectively).

For the 20 km buffers, differences between periods were statistically significant (Wilcoxon $V = 416$, $p = 0.016$, $r_{tb} = -0.347$; figure 5). 29 buffers showed negative $\Delta dECA$ values and 21 showed positive $\Delta dECA$ values (figure 6; tables S9 and S10), confirming heterogeneous inter-period connectivity outcomes. The strongest deteriorations were observed around the Tinigua and Paramillo NNPs ($\Delta dECA = -30.6\%$ and -23.1% , respectively). In contrast, the buffers of Los Estoraques and Los Colorados registered notable connectivity gains ($+16.2\%$ and $+11.7\%$), reflecting localized recovery processes.

Differences between zones among periods were significant for both 2009–2016 (Friedman $\chi^2 = 7.96$, $p = 0.019$, $W = 0.08$) and 2016–2023 (Friedman $\chi^2 = 9.64$, $p = 0.008$, $W = 0.096$) periods, but with low size effects (figures 5 and S5). Nevertheless, post-hoc analysis showed significant differences between PAs and buffer zones only after the agreement.

4. Discussion

4.1. Regional patterns of vegetation loss

Our results showed different levels of change in NV loss across biogeographical regions, reflecting heterogeneous socioecological and institutional contexts that condition PAs' ability to prevent habitat conversion (González-González *et al* 2021). In general, previous studies have identified that Colombian PAs located closer to human settlements or within municipalities with better governance and public services tend to better mitigate deforestation, whereas remote areas are often more vulnerable to illegal ranching, coca cultivation, and gold mining, among other drivers of land-use change (Bonilla-Mejía and Higuera-Mendieta 2019).

The Amazon region experienced the most pronounced post-agreement surge in deforestation, with losses inside PAs more than doubling relative to the conflict period. This escalation was particularly severe around major PAs of the Andes Amazon Transition Belt (AATB), which were strategic territories

under FARC control for several years due to their remoteness and limited accessibility (Canavire-Bacarreza *et al* 2018, Hoffmann *et al* 2018). After the withdrawal of FARC, these landscapes faced rapid forest loss, particularly within PAs, associated with the reoccupation of previously restricted zones, weak institutions, the emergence of illegal land markets, and the expansion of cattle ranching, agriculture, illicit crops, and infrastructure (Armenteras *et al* 2019, Bonilla-Mejía and Higuera-Mendieta 2019, Negret *et al* 2019, Clerici *et al* 2020, Mendoza 2020, Murillo-Sandoval *et al* 2020, 2021). This pattern illustrates the dual role of armed conflict in shaping land-use trajectories, where conflict-driven isolation once constrained deforestation but subsequent peace-time accessibility accelerated landscape transformation (Suarez *et al* 2018, Landholm *et al* 2019).

Although other regions exhibited smaller absolute increases, the consistent upward trends within PAs and surrounding buffers indicate that forest pressures are spreading beyond traditional Amazonian hot-spots. For instance, the sharp deforestation relative increase in the Orinoquia likely reflects the growing pressures from oil palm cultivation, petroleum extraction, and illicit crops (Armenteras *et al* 2009, Romero-Ruiz *et al* 2012). Two PAs from the Caribbean region registered the most substantial relative increase in deforestation (see figure S1), which agrees with other studies that have reported PAs such as Los Colorados and Sierra Nevada de Santa Marta as the most transformed in the last two decades (Latorre Parra *et al* 2022), sharpening pressures on the last remnants of Colombia's tropical dry forest (García *et al* 2014). In the Andean region, the observed increases likely reflect the intensification of smallholder agriculture, cattle ranching, and urban expansion in peri-urban and montane areas where remaining forest remnants are highly fragmented and ecologically critical (Etter *et al* 2006).

Patterns of NNFV loss displayed greater heterogeneity among regions, suggesting that open vegetation formations respond differently to post-agreement land-use pressures. NNFV loss intensified in the Amazon, Orinoquia, and Caribbean regions, driven by the conversion of savannas and shrublands into pastures and mixed agricultural mosaics, while declining or stabilizing in the Andean and Pacific regions, possibly due to secondary succession, land abandonment, or stricter land-use regulation (Sánchez-Cuervo and Aide 2013b). These contrasts indicate that habitat degradation processes extend beyond forests, with Amazonian and Orinoquian PAs emerging as critical frontiers of NV loss.

4.2. Dominant land-use transitions and emerging pressures

The dominance of agricultural and pasture mosaics as the main destination of both deforestation and NNFV loss underscores the influence of extensive ranching and low-intensity agriculture as the leading proximate drivers of land-cover change across Colombia's PAs and their surroundings. These findings are consistent with previous national-scale assessments (Clerici *et al* 2020) but further reveal that conversion to productive land uses remains the primary mechanism of NV loss even within legally protected landscapes. The persistence of this trend across spatial scales and periods suggests that the peace process did not alter the proximate drivers of habitat transformation but rather sharpened them and enabled their spatial expansion into previously inaccessible areas by modifying underlying drivers (Landholm *et al* 2019). Beyond these dominant agricultural dynamics, our results also suggested localized post-agreement habitat conversions to oil palm and, to a lesser extent, mining, indicating an emerging diversification and intensification of land-use pressures near PAs.

Colombia is currently the world's fourth-largest oil palm producer, and further expansion of oil palm cultivation is expected, driven by rising global demand for this commodity and the country's extensive areas suitable for cultivation (García-Ulloa *et al* 2012, Castiblanco *et al* 2013). For instance, in the Amazon, oil palm has become the dominant industrial crop and an increasingly important driver of deforestation (Molina 2024). Assessments have identified South American forests as particularly vulnerable to oil-palm-related biodiversity loss, with regions such as the Serranía de la Macarena among the most at risk in Colombia (Vijay *et al* 2016, Ocampo-Peñuela *et al* 2018). Although our findings suggest emerging oil palm-related habitat loss within PAs, to the best of our knowledge, such processes have not been previously documented in Colombia. Therefore, these results should be interpreted cautiously, as they may partly reflect misclassification or thematic uncertainty in the LULC dataset.

4.3. Dynamics of NV loss in the post-agreement period

The significant increase in deforestation observed during the post-agreement period is consistent with multiple regional and national analyzes reporting accelerated forest loss after 2016 (Armenteras *et al* 2019, Negret *et al* 2019, Clerici *et al* 2020, Murillo-Sandoval *et al* 2020, 2021). As in those studies, the sharpest increases were concentrated in the Amazonian foothills and the AATB, where weak governance, land grabbing, and expanding agricultural frontiers have rapidly transformed landscapes previously constrained by armed territorial control. By extending the temporal window of analysis to seven years before

and after the 2016 peace agreement (2009–2023), increasing the number of PAs examined, and explicitly distinguishing PA cores from immediate and broader surrounding landscapes, our results provide an updated national-scale picture of how habitat conversion evolves after 2016. These patterns are consistent with a sustained intensification of deforestation within PAs and their buffers after 2016, but they should be interpreted as descriptive evidence within a broader post-agreement context rather than as a strict causal attribution to the agreement itself.

Results related to NNFV add an additional, but more uncertain, dimension to this picture. Although NNFV loss increased in several PAs and regions, and the relative increase within PAs was larger than that of deforestation, these patterns were more heterogeneous and should be interpreted with greater caution than deforestation results. Even so, they highlight that after 2016, habitat conversion pressures extended beyond forests into open natural ecosystems such as shrublands, savannas, páramos, and other non-forest formations, components of Colombia's protected landscapes that have received comparatively less attention in national post-2016 assessments. For example, this trend was evident in PAs such as El Tuparro PNN in the Orinoquia and Sierra Nevada de Santa Marta PNN in the Caribbean region, where NNFV loss increased sharply between periods, reflecting the growing vulnerability of open natural formations even within PAs.

These findings are particularly relevant for biodiversity conservation within PAs dominated by non-forest ecosystems, such as high-Andean páramos and Orinoquian savannas, which sustain unique species assemblages, regulate hydrology, store carbon, and maintain landscape connectivity. The intensification of NNFV loss suggests that these habitats are increasingly exposed to anthropogenic disturbance and land-use conversion following rapid socioecological shifts in the post-agreement period. Recognizing and incorporating NNFV dynamics into conservation monitoring frameworks is therefore essential to safeguard the integrity of these ecosystems and to ensure that post-agreement restoration and land-management policies effectively address the full spectrum of Colombia's NV types.

When considered together, the combined increase in NV loss provides a more integrated view of habitat transformation across Colombia's PAs. This metric captures both the rapid and extensive clearing of forests and the more diffuse transformation of open natural formations, revealing a cumulative erosion of ecological integrity that extends across all spatial zones. The significant rise in NV loss during the post-agreement period indicates that the effects of land-use expansion have transcended vegetation types and administrative boundaries, undermining the PA network's ability to maintain ecosystem functionality and connectivity. These results underscore the need for conservation and restoration strategies that operate at the landscape scale, addressing both forested and non-forested habitats within a unified framework capable of mitigating the diverse, interlinked drivers of NV loss in post-agreement Colombia.

4.4. Dynamics of landscape connectivity in the post-agreement period

Connected area declined across PAs and their surrounding buffers. Although PAs retained relatively high levels of connected habitat (>95%), dECA values were generally more negative in the post-agreement period, indicating less favorable connectivity outcomes after 2016. These patterns suggest that the post-2016 intensification of habitat conversion coincided with increasing fragmentation and isolation of NV networks around PAs, especially in broader surrounding landscapes. Spatially, the connected area decreased with increasing distance from PAs' boundaries, and losses in surrounding buffers were often disproportionate to habitat-area reduction alone. These patterns suggest that after 2016, the intensification of habitat conversion coincided with increasing fragmentation and isolation of NV networks around PAs, especially in broader surrounding landscapes. Such declines have the potential to promote ecological isolation and disrupt key ecological processes linking protected habitats with surrounding landscapes and other PAs, including species dispersal, gene flow, and biogeochemical exchanges (Hansen and DeFries 2007, DeFries *et al* 2010, Laurance *et al* 2012). This progressive disconnection indicates increasing spatial separation among NV patches surrounding PAs, weakening the role of buffers in maintaining habitat continuity and likely amplifying edge-related disturbances that further erode landscape integrity.

Beyond these overall patterns, changes in rECA values provide insights into the processes driving connectivity loss. The increase in rECA within PAs (from 0.9 to 1.3) indicates a shift from transformations primarily associated with habitat reduction (e.g. perforation or shrinkage) toward those linked to habitat isolation (e.g. fragmentation or attrition). In contrast, buffer zones exhibited more pronounced and heterogeneous declines, with connectivity losses exceeding habitat losses, particularly in the outer 20 km buffers (rECA > 1.7). This decoupling implies that land-use transitions not only reduced the amount of habitat but also disrupted its spatial configuration, increasing habitat isolation

and fragmentation. Such structural degradation likely intensifies edge effects and constrains species movement (Laurance *et al* 2011, Haddad *et al* 2015). These findings reinforce that maintaining habitat extent alone is insufficient to preserve connectivity within NV networks; spatial arrangement is also critical.

At finer spatial scales, the heterogeneous trajectories observed among individual PAs and buffers underscore the context-dependent nature of connectivity responses. Some landscapes exhibited localized recovery, while others continued to experience accelerated isolation. This variability likely reflects differences in land-use history, accessibility, governance capacity, and the scope of ongoing restoration or conservation actions. For instance, areas such as Los Estoraques, Los Colorados, and Isla de Salamanca showed positive Δ dECA values, likely associated with reduced NV loss or active restoration, whereas Paramillo, Otún Quimbaya, Sierra de la Macarena, and particularly Tinigua exhibited negative Δ dECA values consistent with advancing deforestation fronts. Recent analyzes have similarly identified the Eje Cafetero, where Otún Quimbaya SFF is located, as a hotspot of connectivity decline in Andean forests (Correa-Ayram *et al* 2025). Similarly, PAs within the AATB have been repeatedly reported as critical areas of connectivity loss during the post-agreement period (Murillo-Sandoval *et al* 2022, López *et al* 2024). Together, this evidence highlights the growing vulnerability of the northern Andes–Amazon biodiversity bridge (Clerici *et al* 2018, Murillo-Sandoval *et al* 2022) and underscores the urgent need to integrate connectivity restoration and landscape governance into post-agreement conservation and land-governance agendas (Calle-Rendón *et al* 2018, Negret *et al* 2021, Linero-Triana *et al* 2023).

4.5. Study limitations

This study is based on a descriptive before-and-after comparison and does not isolate the specific causal effect of the 2016 peace agreement from other concurrent processes, such as market dynamics, governance shifts, infrastructure expansion, enforcement changes, or climate variability. Accordingly, the observed patterns are best interpreted as changes occurring across the conflict and post-agreement periods within a broader socioecological transition, rather than as effects attributable solely to the agreement itself. In addition, potential residual temporal inconsistencies in annual LULC products arising, for example, from sensor transitions (TM, ETM+, and OLI), image availability, or differences in classification performance may influence apparent changes in time series as previously observed on the Global Forest Change dataset (Hansen *et al* 2013, Ceccherini *et al* 2021, Palahí *et al* 2021). However, the methodology used by MapBiomass to produce its annual LULC dataset includes spatiotemporal filtering rules designed to reduce implausible year-to-year transitions (Souza *et al* 2020, Fundación Gaia Amazonas 2024).

Results related to NNFV loss should be interpreted more cautiously than those for deforestation. The overall MapBiomass accuracy values reported here reflect product-wide performance rather than direct validation of the aggregated NNFV category used in this study, and the class-level accuracies indicate greater thematic uncertainty for Natural non-forest formation than for Forest formation MapBiomass classes (table S3). Finally, the connectivity analysis represents generic NV connectivity rather than species-specific or resistance-weighted connectivity. Habitat was defined as a binary NV layer (Forest + NNFV), and connectivity was estimated from Euclidean inter-patch distances under a homogeneous-matrix assumption. Therefore, the analysis captures changes associated with habitat amount and spatial configuration, but it does not model differential matrix permeability among HMA types, quantify connectivity between buffer habitats and PA cores, or support species-level inference (Saura and Pascual-Hortal 2007, Saura *et al* 2011).

5. Conclusion

The period following the 2016 peace agreement coincided with more intense habitat conversion in and around PAs, with the Andes–Amazon transition showing the strongest signals. By integrating habitat-loss trajectories and connectivity metrics across 50 strictly PAs and two non-overlapping buffer bands, this study shows that connectivity remained comparatively stable inside PA cores but declined much more strongly in surrounding landscapes, where configuration effects often outpaced habitat loss. These results indicate that safeguarding PA cores alone is insufficient to maintain ecological flows across the national network. Strengthening buffer governance, targeted restoration, and connectivity-aware land-use planning at ecologically meaningful landscape scales will be essential for reducing isolation of PA cores and enhancing the long-term functionality of Colombia's protected-area system.

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Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: <https://doi.org/10.5281/zenodo.19445808> (Rubiano et al 2026).

Supplementary data 1 available at <https://doi.org/10.1088/2515-7620/ae69a5/data1>.

Declaration of interests

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

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Acceso a los códigos de programación

Los algoritmos y/o códigos de programación utilizados para el desarrollo del trabajo se encuentran disponibles a través del siguiente enlace al repositorio:

<https://zenodo.org/records/19209402?token=eyJhbGciOiJIUzUxMiIsImIhdCI6MTc4MDA2NzQ1NSwiZXhwIjoxNzk4Njc1MTk5fQ.eyJpZCI6ImI2OGM1ZTg0LWMzN2ItNDA2MS05ZTM3LTliMjA4ZTFhYjUzZSIsImRhdGEiOnt9LCJyYW5kb20iOiI2NTRmZjBmMDkwZjEyMWQ2ZWJyYTMjIj5ZmFkMDg3YyJ9.V6tLWV9wVvGrw0iXu2wFfQEDjIiQTjFOx6lZMovEQQg3C9Vw6ZV2hR4M8gNGAW0RIqkVLyaDpurugVEoBGUKMg>