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Original research article

Landscape connectivity loss after the de-escalation of armed conflict in the Colombian Amazon (2011–2021)

Jesica López[a,*], Yuyang Qian[b], Paulo José Murillo-Sandoval[c], Nicola Clerici[d], Lars Eklundh[b]

[a] *Centre of Environmental and Climate Science, Faculty of Science, Lund University, Sweden*
[b] *Department of Physical Geography and Ecosystem Science, Faculty of Science, Lund University, Sweden*
[c] *Department of Topography, Faculty of Habitat, Design and Infrastructure Sciences, Universidad del Tolima, Colombia*
[d] *Department of Biology, Faculty of Natural Sciences and Mathematics, Universidad del Rosario, Colombia*

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ABSTRACT

Rapid deforestation has been well-documented in Colombia after the 2016 peace agreement with FARC. While many analysis using remote sensing identify land cover change, structural connectivity variables are less studied for understanding landscape transformation. In this work, we used data from the Landsat archive from 2011 to 2021, the Continuous Change Detection and Classification algorithm (CCDC), and Morphological Spatial Pattern Analysis (MSPA) to analyze deforestation, land cover change, and landscape connectivity in northwestern Colombia's Amazon. We examined the spatial patterns in three specific subsets in the Colombian arc of deforestation, with a special focus on the surroundings of the National Natural Park Serranía de Chiribiquete. Our results confirm changes in structural connectivity linked to pasture expansion from the conversion *forest* to *pasture* during the analyzed period showing changes in the borders of Serranía de Chiribiquete National Park along rivers and roads, where cattle is transported. Before 2016, the average annual deforested area in the three study areas was 27.93 km^2; after 2016, this number increased to 73.36 km^2. The outcomes of our study contribute to an improved understanding of pasture dynamics. They can aid decision-making in areas that play a key role in ecological networks necessary for long-term conservation efforts. Our quantitative assessments have revealed a temporal shrinking of the core area, indicative of a decline in forest cover. Concurrently, we observed an expansion of the edge and background areas, which is consistent with the proliferation of pastures. This study presents methodology and data in support of policy-making for sustainable land use and ecological connectivity to mitigate further environmental degradation in the area.

1. Introduction

Pasture expansion for cattle is a significant driver of deforestation in many tropical regions around the world, notably in countries like Brazil, Indonesia, and Argentina (Ritchie and Roser, 2021). Large-scale clearing of forests to make way for cattle grazing not only results in the loss of valuable biodiversity and carbon sinks, but also contributes to climate change, alters the water cycle, and often

* Correspondence to: Centre of Environmental and Climate Science, Faculty of Science, Lund University, Sölvegatan 37, Lund 223 62, Sweden.
E-mail address: jesica.lopez@cec.lu.se (J. López).

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involves the displacement of indigenous and peasant communities, promoting land conflicts (Hansen et al., 2020; Curtis et al., 2018; Arias-Gaviria et al., 2021; Murillo-Sandoval et al., 2023). Despite several efforts to introduce sustainable cattle production practices, implement stricter land-use regulations, and promote other alternative land uses for production, halting deforestation remains a significant challenge with profound impacts on ecosystems, climate, and livelihoods (Pendrill et al., 2022).

The Amazon forests have been impacted by many deforestation drivers, i.e., irregular infrastructure development, illicit crop cultivation (i.e. coca farming), illegal logging, illegal mining, the expansion of agricultural activities into prohibited areas, and *praderization* [clearing land for pastures] (Pacheco et al., 2020). However, for the purpose of this study, we will focus on the agricultural expansion driven by praderization as an accelerating driver of deforestation (Krause, et al., 2022). The Amazon forests are being converted into pastures for beef and leather production, oil palm plantations, and fields of soybeans and other crops (Seymour and Busch, 2016). This is referred to as commodity-driven deforestation, which involves deforestation for the purpose of expanding any commodity primarily to meet consumer demands.

Distinctive problems of pasture-based systems for cattle ranching in the Colombian context are characterized by low productivity, the use of large areas for pasturelands to raise cattle (Zuluaga et al., 2021) and the involvement of land grabbing (Vergara et al., 2022). Other studies provide evidence that the conversion of forests to pasture has accelerated over the last four decades (Nepstad et al., 2006). In fact, the Colombian Amazon region accounts for 68.2 % of deforestation in the country, with an increasing trend until 2022; forest loss was 109,302 ha in 2020 and 112,899 ha in 2021 (FCDS, 2022). Critically, the northwestern *arc of deforestation* in the Amazon lost about 89,000 ha of forest in 2020 and 2021 (World Bank Group -WBG, 2023).

The arc of deforestation refers to a region in the Colombian Amazon where there are intense concentrations of deforestation. This region, while encompassing Protected Areas (PAs) such as National Natural Parks (NNPs) Tinigua, Serranía de la Macarena, Serranía de Chiribiquete, and La Paya, is not confined within these protected boundaries. Rather, it extends beyond them into the provinces of Putumayo, Caquetá, Meta, and Guaviare, with deforestation hotspots mainly outside these protected areas (Visión Amazonía, 2022).

Historically, competition over land and access to natural resources has helped to fuel armed conflict in Colombia. The consequences have led to a surge in deforestation, particularly after the ceasefire and peace accord in 2014 and 2016, respectively (ICG, 2021). The armed conflict instigated a forced and spontaneous mass migration of people, leading to large areas of abandoned land, which subsequently experienced significant forest regrowth (Sánchez Cuervo and Mitchell, 2013), as well as exerting a negative impact on forests, including increased deforestation rates and conflict-induced land use change that augmented smallholder land grabbing and cattle pasture expansion (Landholm et al., 2019). Moreover, armed groups, particularly the FARC (Fuerzas Armadas Revolucionarias de Colombia) played a complex role in forest conservation during the conflict period (1964–2016), they exerted intentional and unintentional conservation effects, and maintained forests to prevent clearance, serving to restrict access, obscure the militants' positions, and facilitate illicit trade of weapons (Murillo-Sandoval et al., 2020). As in numerous Colombian territories abundant in mineral resources or suitable for cattle ranching and coca cultivation, the cessation of environmental patrols during the conflict era has escalated deforestation and hunting activities (Bautista-Cespedes et al., 2021).

Pasture expansion for cattle ranching directly impacts landscape structural connectivity. The landscape structural connectivity is a concept linked to the spatial continuity of the ecosystems/habitats, it determines how the landscape aids or hinders biotic movements among resource patches (With, 2019). Changes in connectivity can negatively affect several ecological aspects. These include the maintenance of genetic exchanges, species dispersal, metapopulation resilience, and ecosystem service provisions (Clerici et al., 2018). Ecological connectivity is defined as "*the unimpeded movement of species and the flow of natural processes that sustain life on Earth*" (CMS, 1979). In our study we refer structural connectivity as a metric of habitat permeability. This metric is derived from the physical attributes and organization of habitat fragments, disturbances, and other elements of land and freshwater bodies in the landscape that are assumed to be crucial for the mobility of organisms within their environment (Hilty et al., 2019).

Monitoring landscape connectivity is key to developing proper integrated land use planning strategies for conservation and restoration. However, the consequences of forest clearing and its contributions to the loss of structural connectivity are still little known (Lienert and Burkart, 2023), particularly regarding the current and future dynamics of species populations in the region, and not the least after the 2016 Peace Agreement between FARC and the Colombian government.

Our objective is to comprehend the temporal and spatial dynamics of recent land transformations and its drivers in the northwest Colombian Amazon arc. We aim to achieve this by mapping deforestation using sub-annual change detection algorithms, making land cover maps and conducting a landscape connectivity analysis in key areas. Specifically, we analyze three study sites that display varied patterns of land alteration and are undergoing rapid landscape transformation. We use 2016 as a reference year, as it represents the year of the signing of the Peace Agreement, between FARC and the national government.

To accurately understand landscape connectivity, we combined the *Continuous Change Detection and Classification* (CCDC) (Zhu and Woodcock, 2014) algorithm to detect deforestation and rapid land cover changes, and the *Morphological Spatial Pattern Analysis* (MSPA) to analyze spatial patterns of deforestation. The Morphological Spatial Pattern Analysis (MSPA) is instrumental in identifying core habitat patches and ecological corridors, which play a pivotal role in landscape connectivity at the pixel level (Lin et al., 2021). As a spatial pattern analysis method, MSPA facilitates the evaluation of ecological networks, thereby enabling landscape connectivity analysis. It incorporates diverse morphological classes, including core habitat, bridges, and edges (Vogt et al., 2007; Soille and Vogt, 2009, 2017, 2022). As discussed in Vogt et al. (2007), these characteristics can be employed to describe structural and functional corridors in forest areas.

The combination of these geospatial techniques was used to monitor forest dynamics in terms of structural connectivity changes. We advocate for the mapping of morphological features (e.g., core, bridge, and edges classes) that help us to quantify the structural connectivity status of the landscape, rather than just categorical variables such as deforestation and land cover change.

2. Study area and data sources

2.1. Study area

The Amazon region in Colombia covers 403,348 km^2, which is equivalent to approximately 5.4 % of the total political-administrative Amazon basin and corresponds to 35.3 % of the continental national territory (MADS, 2020). This study encompasses three subset study areas. In Fig. 1, the southern (Ch-South) A subset region shows rapid, large-scale deforestation deeper into the Amazon and illegal land clearing. The northwestern (Ch-Northwest) B subset is part of a 'campesino reserve' as well as indigenous communities' lands in the Sabanas de Yarí. Lastly, the northern (Ch-North) C subset is primarily an area with a combined effect of pasture expansion and recent coca farming. The selection of the study areas is based on previous studies, i.e. Hansen and Loveland (2012), Dávalos et al. (2014), FCDS (2021) and Murillo-Sandoval et al. (2023). This includes the transformation of forests into pastures for cattle ranching, as cattle rearing through land grabbing is an inexpensive method for land speculation, leading to significant profits in the area (Clerici et al., 2020).

2.2. Data sources

2.2.1. Landsat data

We used Landsat 7 ETM+ and Landsat 8 OLI surface reflectance images (Collection 2, Tier 1) from 2000 to 2021, covering the study areas. The reason for selecting Landsat images beyond the chosen study period (2011–2021) is that the Change Detection algorithm (CCDC) requires a minimum of 15 clear observations for each image pixel to perform continuous change detection and classification (Zhu and Woodcock, 2014). If the number of clear observations is less than 15, the CCDC algorithm fails to initialize a time series model (Yang et al., 2023). In total, we used 1671, 916, and 1859 images in the study regions A, B, and C, respectively. The data were processed with the *CFMask* algorithm to produce masks for clouds and shadows (Foga et al., 2017) within the Google Earth Engine (GEE) geoprocessing platform, which allows for convenient and fast data processing (Arévalo et al., 2020). We also used PlanetScope satellite high-resolution imagery for validation.

2.2.2. Training data

The CCDC algorithm (see Methods, below) requires training data for land cover classification. Given that the primary objective of this paper is to detect the impact of pasture expansion on deforestation, and it is difficult to differentiate between pastures and grasslands, a precise land cover reference map is essential for classification purposes (Oliveira et al., 2020). In this study, the training

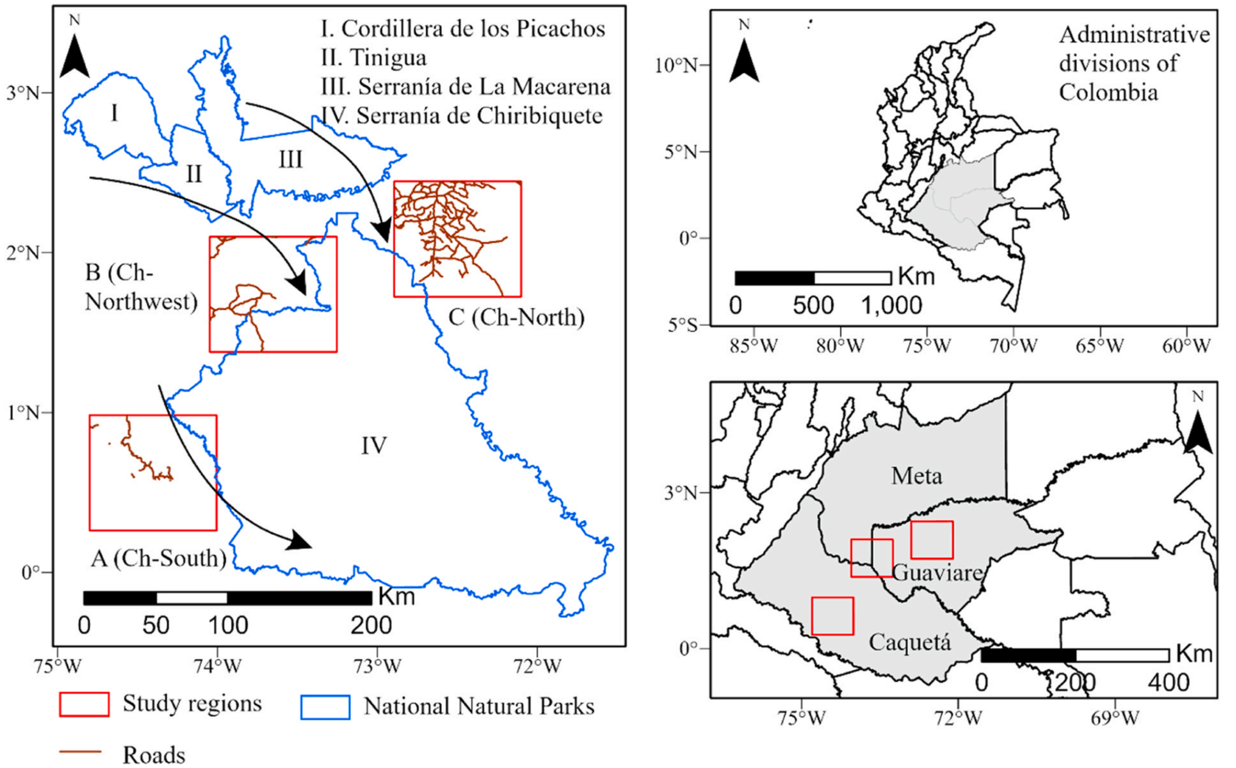

Fig. 1. Study regions around the Serranía de Chiribiquete NNP. (A) southern, (B) northwestern and (C) northern.

data were obtained from the land cover map generated in 2012 (https://siatac.co/simcoba/) by the Colombian Amazon Land Cover Monitoring System (SIMCOBA). The map distinguishes pastures from grasslands, is provided in vector format at a scale of 1:100,000 and has been prepared from visual interpretation of Landsat 5 TM and Landsat 7 ETM+ imagery (Agudelo-Hz et al., 2023). The land cover classes selected correspond to forest, pasture, grassland, and water bodies. Note that in the land cover map provided by SIMCOBA, some land cover types with very small proportions, such as built-up areas and wetlands, are ignored. Additionally, it is important to note that there is a significant amount of grasslands only in the B subset, Ch-Northwest.

A total of 3000 validation points were selected for all three subset study areas. The quantity of sample points was allocated randomly in proportion to the area of each land cover type within the study areas, as detailed in Appendix 1, Table 3.

3. Methodology

3.1. Deforestation detection

Deforestation events were detected using the CCDC algorithm. The method applies harmonic regression models to each spectral band present in the data time series to capture intra-annual, gradual inter-annual, and abrupt changes (Zhu and Woodcock, 2014). The Ordinary Least Squares (OLS) method is used for all bands in the time series, and the Root Mean Square Error (RMSE) is calculated for each spectral band. The deviation between the model predictions and observations for each band is normalized by three times the RMSE. This is done because when there is a change, the spectral signals often differ from the model prediction by more than three times the RMSE (Zhu and Woodcock, 2014). As a result, when the deviation of a new observation exceeds this, it is identified as a potential change, in our case, a deforested pixel. CCDC parameterization details for Landsat imagery are reported in Appendix 1 (Table 1).

A mask is created to identify all positions within the image array where breaks are detected within the study period and that satisfy the criterion of having a change probability of 1. This procedure helps to eliminate any false breaks that may have been detected.

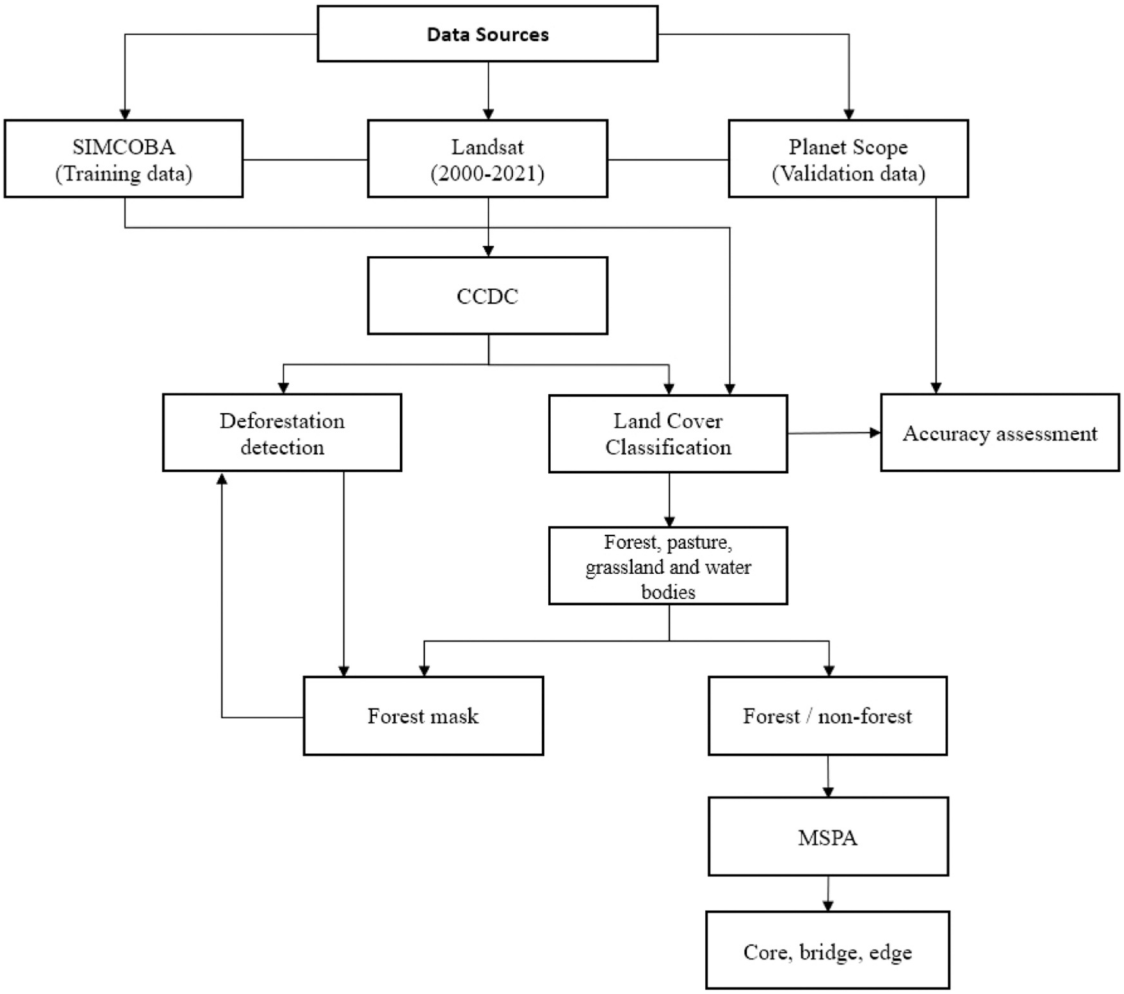

Fig. 2. Data sets and processing methodology to detect deforestation (CCDC) and land structural connectivity (MSPA).

Finally, the mask, which is described in Section 3.2, is used to filter false pixels which are identified as deforestation and detected as changes by CCDC. For example, some pixels could represent water bodies incorrectly identified as deforestation. The purpose of applying the mask is to prevent potential confusion between changes due to other reasons than deforestation, such as due to crop harvest (Hamunyela et al., 2020). It should be noted that to ensure uniformity in subsequent analysis, this analysis used the timing of the first occurrence of deforestation for each pixel (Li et al., 2023).

The 2016 Peace Agreement marks a pivotal institutional shift in the region, with our study spanning years 2011–2021. Therefore, deforestation areas within three subsets were categorized into three periods: pre-2016 (2011–2015), 2016, and post-2016 (2017–2021). The Shapiro-Wilk method, a powerful normality test (Milien et al., 2021), was first applied to the data as it checks if the data shows normal distribution. To evaluate significant differences in deforestation area before and after the peace agreement, we used the independent samples t-test and the Mann-Whitney U test (Kim, 2010), the latter not requiring normally distributed data. Based on the normality test results, one of the methods was selected to examine if deforestation values between groups were statistically significant to all study areas. The analysis was conducted using the Statistical Package for the Social Sciences software (SPSS).

3.2. Land cover classification

The output array image of CCDC incorporates the regression coefficients for every band within the input Landsat image collection, along with the RMSE of the model for each time segment and input band (Awty-Carroll et al., 2019). These variables (Appendix 1, Table 2) are useful for land cover classification (Awty-Carroll et al., 2019; Xie et al., 2022).

Once the training data has been prepared, a machine learning classifier can classify the generated change detection outcomes from CCDC. In this study, a Random Forest (RF) classifier (Breiman, 2001) was implemented in GEE. The choice of the RF classifier is due to its consistent accuracy across both small and large sample sets, and its fast-processing time (Ramezan et al., 2021). Annual land cover maps with classes forest, pasture, grassland, and water bodies were generated for the three subset study areas. We prepared a landcover binary mask, using 2010-year data that was reclassified into only two categories (forests/non-forests). This binary mask was used for the MSPA analysis. As per the time series analysis period, 2011 is the start year, 2016 is the year when the peace agreement was signed and 2021 is the last year in the analysis. We used the comprehensive road dataset provided by the Amazon Georreferenced Socio-Environmental Information Network (RAISG), aiming to discern the potential impacts of road expansion within our selected areas of study.

3.3. Accuracy assessment

To evaluate the accuracy of the land cover maps, we validate them all with an identical methodology from which we selected and generated the map in 2021, a random stratified sample design was implemented. The equation proposed by Cochran (1977) was utilized to determine the total number of validation samples. For the visual interpretation of the PlanetScope satellite high-resolution imagery produced in December 2021, a set of 250 validation points was selected for each of the three study areas (see Fig. 2). These samples were distributed based on the relative area of change in 2021 and were uniformly distributed within the research regions. This distribution considered the need to increase the sample size for less common classes, such as water bodies, to minimize the standard errors associated with accuracy estimates specific to these classes (Olofsson et al., 2014). Furthermore, overall accuracy, producer's accuracy and user's accuracy were calculated to evaluate the quality of the land cover map produced. The widely used Kappa coefficient was not employed to report accuracy because, while it is associated with overall accuracy, it can provide misleading or flawed information (Olofsson et al., 2014; Pontius and Millones, 2011).

3.4. Structural connectivity

Structural connectivity can serve as a comprehensive evaluation tool for the general assessment of landscape connectivity, thereby aiding in the strategic network planning for multiple ecosystem services (Butler et al., 2022, Rieb and Bennett, 2020). We used MSPA as it incorporates morphological classes, including core habitat, bridges, and edges (Vogt et al., 2007; Soille and Vogt, 2009) selected in our analysis. As discussed in Vogt et al. (2007), these characteristics can be employed to describe structural and functional corridors in forest areas. We leverage them to support our analysis of morphological variables in the landscape structural connectivity (Vogt et al., 2007, de Oliveira et al., 2017) due to deforestation changes in our study regions.

As applied e.g. in the study by Clerici and Vogt (2013), our study focuses on the morphological classes: core (core forest habitat), bridge (representing forest corridors), and edge (non-core habitat around the forest patch). The bridge pixels identify the connection between two or more core areas, representing biological corridors. We choose to focus specifically on these classes of the MSPA framework, leaving the other structural categories unexplored for specific reasons of the landscape mosaic in this work.

MSPA was performed using the software GUIDOS (http://forest.jrc.ec.europa.eu/download/software/guidos). As MSPA takes binary images as input, the land cover maps generated in Section 3.2 were reclassified into forest and non-forest categories. The edge distance is an essential parameter that influences the MSPA outputs (Clerici and Vogt, 2013). Increasing the edge width reduces the total area and number of patches belonging to the core class (de Oliveira et al., 2017). Edge width, the distance from a patch's edge to its core, is key in shaping the landscape's spatial layout (Vogt, 2023). Greater edge widths encompass more matrix, reducing core areas, and highlighting the portion of the habitat patch subject to the 'edge effect' (Young et al., 1996). Conversely, smaller widths can skew patch distribution, increasing fragment isolation (Rogan and Lacher, 2018), impacting species diversity. Due to this phenomenon, the edge width was set to a median value of 5 pixels (150 m), as the edge effect impacts the overall amount of core habitat in the landscape and influences species dynamics. Other parameters of MSPA were set as default (GUIDOS toolbox Manual, 2017).

3.4.1. Validation data

Due to the absence of in-situ information, PlanetScope high-resolution satellite imagery was employed to evaluate the land cover map produced for the year 2021. The land cover map developed by SIMCOBA in 2012 was also utilized as reference data to aid in distinguishing between grasslands and pastures. Norway's International Climate and Forest Initiative (NICFI) has recently released PlanetScope satellite images that cover tropical regions, with the aim of preserving tropical forests (Wagner et al., 2023). PlanetScope Visual Mosaics, one of the data products provided by NICFI, includes red, green, and blue bands at a spatial resolution of approximately 5 m for visual display and interpretation. Consequently, these images are suitable for creating reference data in tropical regions, making them applicable to our subset study areas. Since PlanetScope Visual Mosaics are composites of the best acquisitions, most images are free of clouds.

4. Results

4.1. Deforestation trends between 2011 and 2021

The CCDC algorithm provides a spatially explicit map of forest change in the northwestern Colombian Amazon from 2011 to 2021

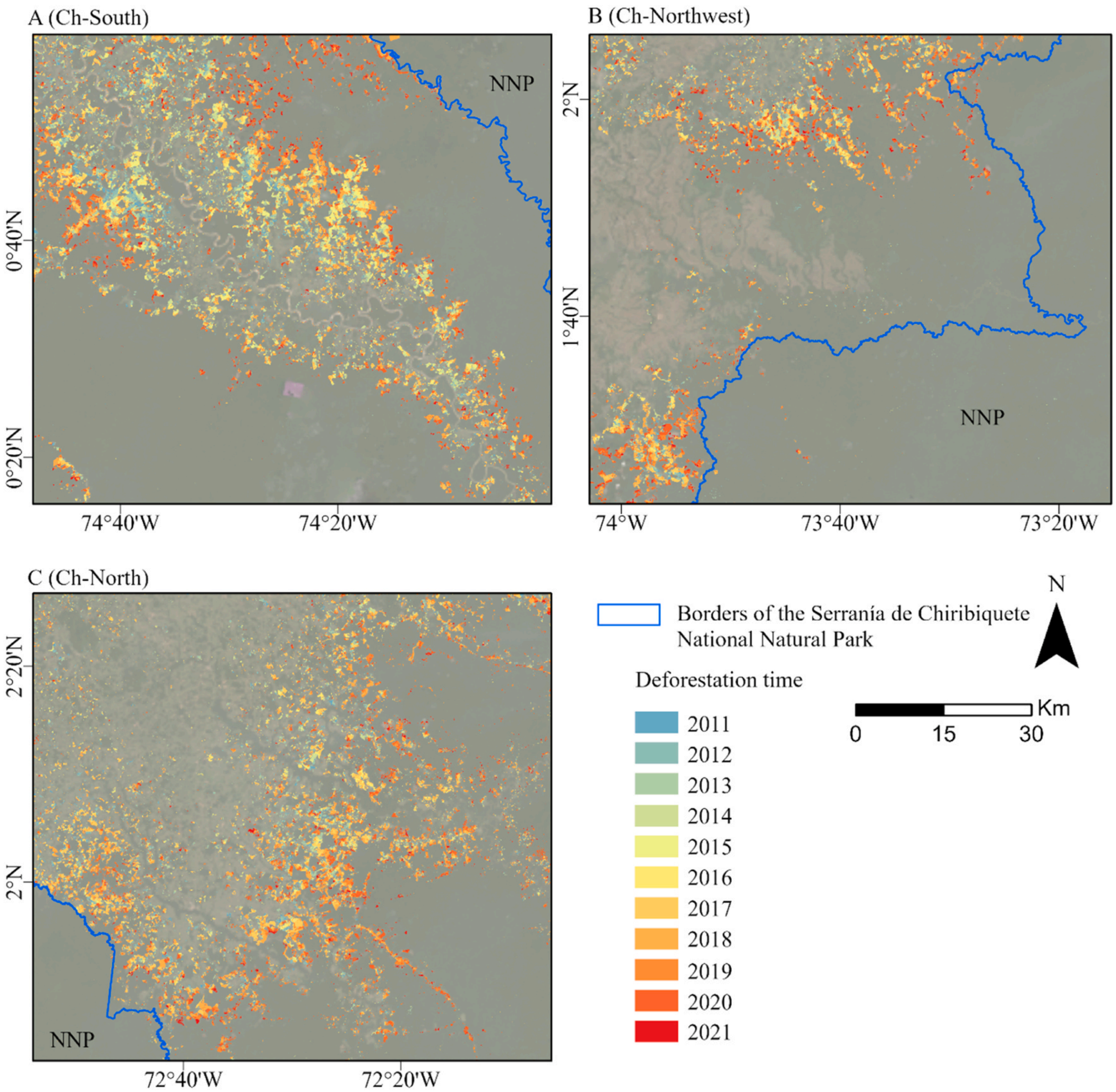

Fig. 3. Deforestation detected in study regions A (Ch-South), B (Ch-Northwest), C (Ch-North) from 2011 to 2021. The background is a PlanetScope visual monitoring mosaic image obtained in December 2021.

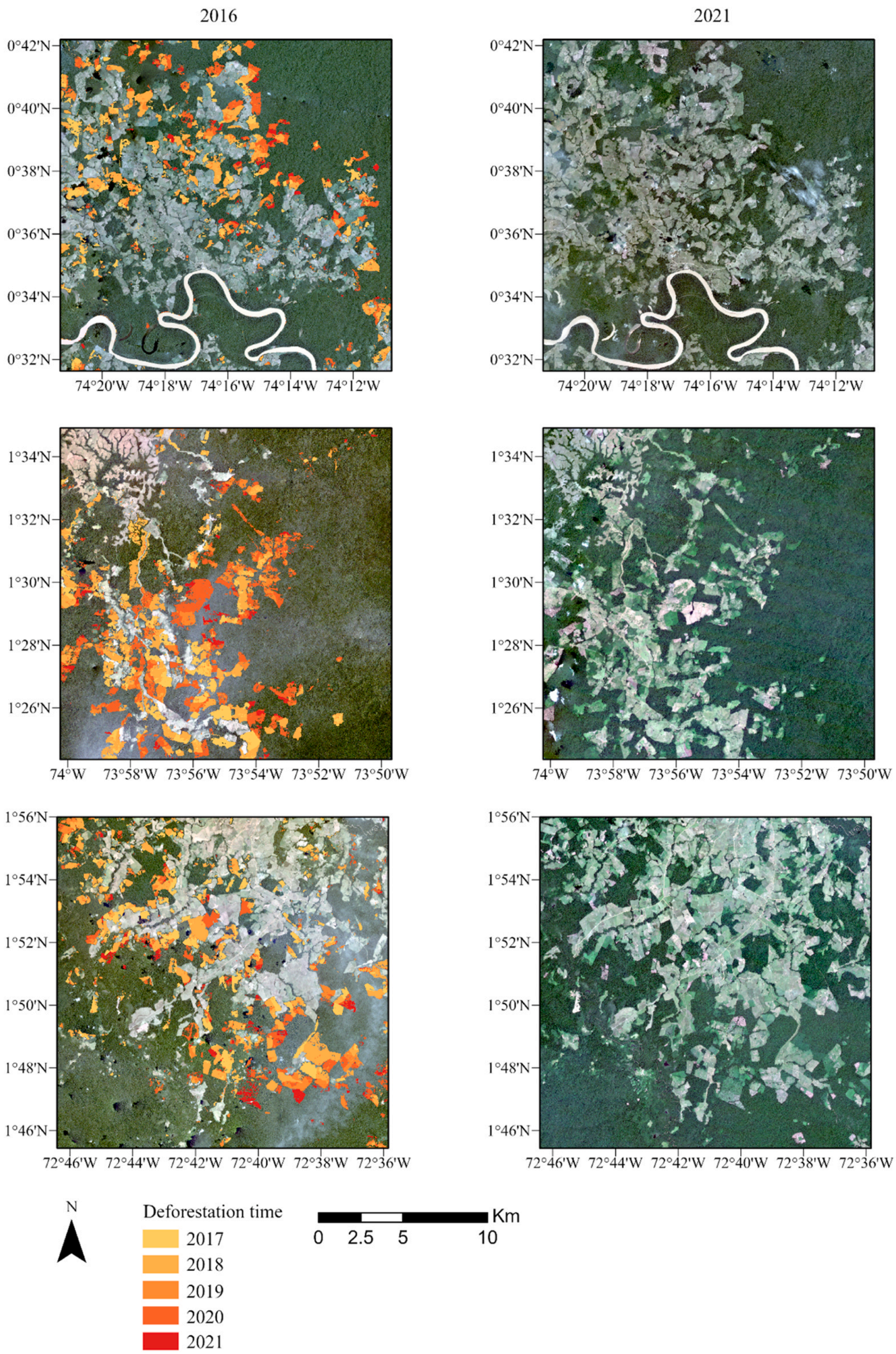

Fig. 4. Observed deforested areas across the three study areas in 2016 (left) and 2021 (right). In the limits and entering the borders of the Serranía de Chiribiquete National Park. Examples in order from top to bottom Sunciya river in Caquetá, Sabanas de Yarí in the area between Candilejas Este 2 (Caquetá) and La Tunia Suroeste 2 (Meta) and South of Calamar in Guaviare. Images © 2016 and 2021 Planet Labs PBC.

(Fig. 3). The results indicate that deforestation detected during this period occurred predominantly near the boundaries of the National Natural Parks (NNPs) and within the protected area in two of the subsets (B (Ch-Northwest), and C (Ch-North)). As observed in Fig. 3, concentrated deforestation can be observed.

In Ch-South (A), patches of illegal deforestation are evident inside the protected area towards the Sabanas deYarí (74.2942ºW 0.9372ºN), along the lower basin of the Caguán river in Caquetá (74.1120ºW 0.3695ºN) and the Sunciya river (74.7070ºW 0.6365ºN). This is where cattle are transported across these rivers to be sold in the municipality of Cartagena del Chairá. In the study region Ch-Northwest (B), the deforested patches were identified in the southern area of Caño los Esteros in Guaviare towards the border of the Chiribiquete protected area (73.4794ºW 2.0232ºN), and to the south of the Yaguará II indigenous reserve. This area is between *Candilejas* Este 2 and *La Tunia Suroeste* 2 in Caquetá along the road *La Tunia-Camuya* (73.9565ºW 1.4666ºN). Lastly, in the study region Ch-North (C), the concentration of deforestation caused by pastures, e.g. for cattle, and new coca farming determined from satellite imagery in an area between La Lindosa towards Las Juntas and Santa Ines (72.2936ºW 2.0448ºN) bordering the Inirida river, and to the south of the settlement of Calamar towards the northern border of NNP Chiribiquete (72.6729ºW 1.8028ºN), both located in the Guaviare province. Fig. 4 shows examples of the land use changes using Planet's imaginery.

The deforested regions within the three subset study areas exhibited a growing pattern of land clearing from 2011 to 2016 (Fig. 5). During this period, the average annual deforested area across the three study areas was 40.01 km^2. However, a marked escalation was observed from 2016 to 2018, with peak years of deforestation recorded in 2018 and 2020. A significant reduction in the deforested area was observed in 2019 and 2021 across all three subset areas.

The Shapiro-Wilk test results for the two groups suggest that deforestation data before and after 2016 adhere to a normal distribution ($p = 0.1$ and $p = 0.62$). Subsequent independent samples t-test reveal a significant difference in deforestation areas before and after the signing of the peace agreement in 2016 ($p = 0.002$).

In 2012 the negotiation started, showing an increase already from 2011. Particularly in study region A in southern Caquetá, land cover changes were detected, with pastures increasing from 9.91 % in 2011 to 19.94 % in 2021. Similarly, in region C in Guaviare, pastures increased from 32.2 % in 2011 to 41.6 % in 2021 (Table 4). In all three subset study areas, the trend was notably significant during the post-peace accord years, especially 2018 and 2020, where the highest rates of deforestation have been reported by official entities as well (MADS and IDEAM, 2022).

4.2. Land cover changes between 2011 and 2021

Land cover classes underwent significant transformations throughout the analysis period (Fig. 6 and Table 4). The collective forest area within the three subset study areas experienced a decrease of 24.19 % from 2011 to 2021, while the pasture area witnessed an increase totaling 23.58 %. The remaining land cover types exhibited minimal changes. Over the entire period, study area A (Ch-South) encountered the most substantial decrease in forest area (10.11 %) and the largest increase in pasture area (10.03 %), followed by study regions C (Ch-Northwest) and B (Ch-North), respectively.

Moreover, in all study areas, the rate of forest area declines, and pasture area increase was notably more pronounced during the post-conflict period of 2016–2021, compared to the conflict period of 2011–2016.

4.3. Land cover accuracy assessment

The overall accuracy of the land cover classification (Appendix 2, Table 4) in the three subset study areas in 2021 consistently exceeded 90.0 %. Remarkably, the Ch-North study area achieved an overall accuracy of 99.2 %. Both producer's and user's accuracies demonstrated relatively high values in identifying different land cover types, predominantly around 90.0 %. The classification performance for water bodies was the most satisfactory, with no misclassifications or omissions observed in any of the three subset study

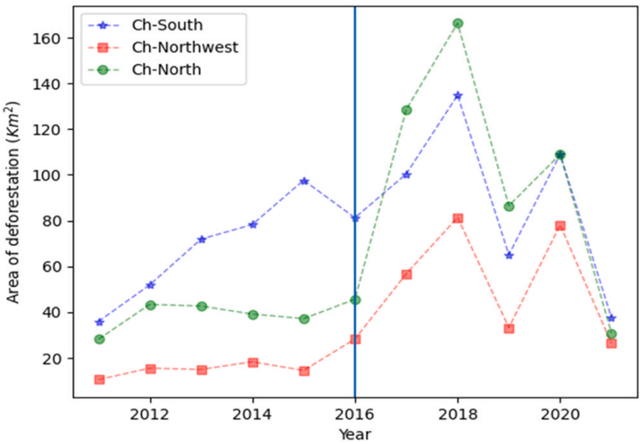

Fig. 5. Area of deforestation (km^2) in the study regions from 2011 to 2021. The blue vertical line in 2016 marks the year of the Peace Agreement.

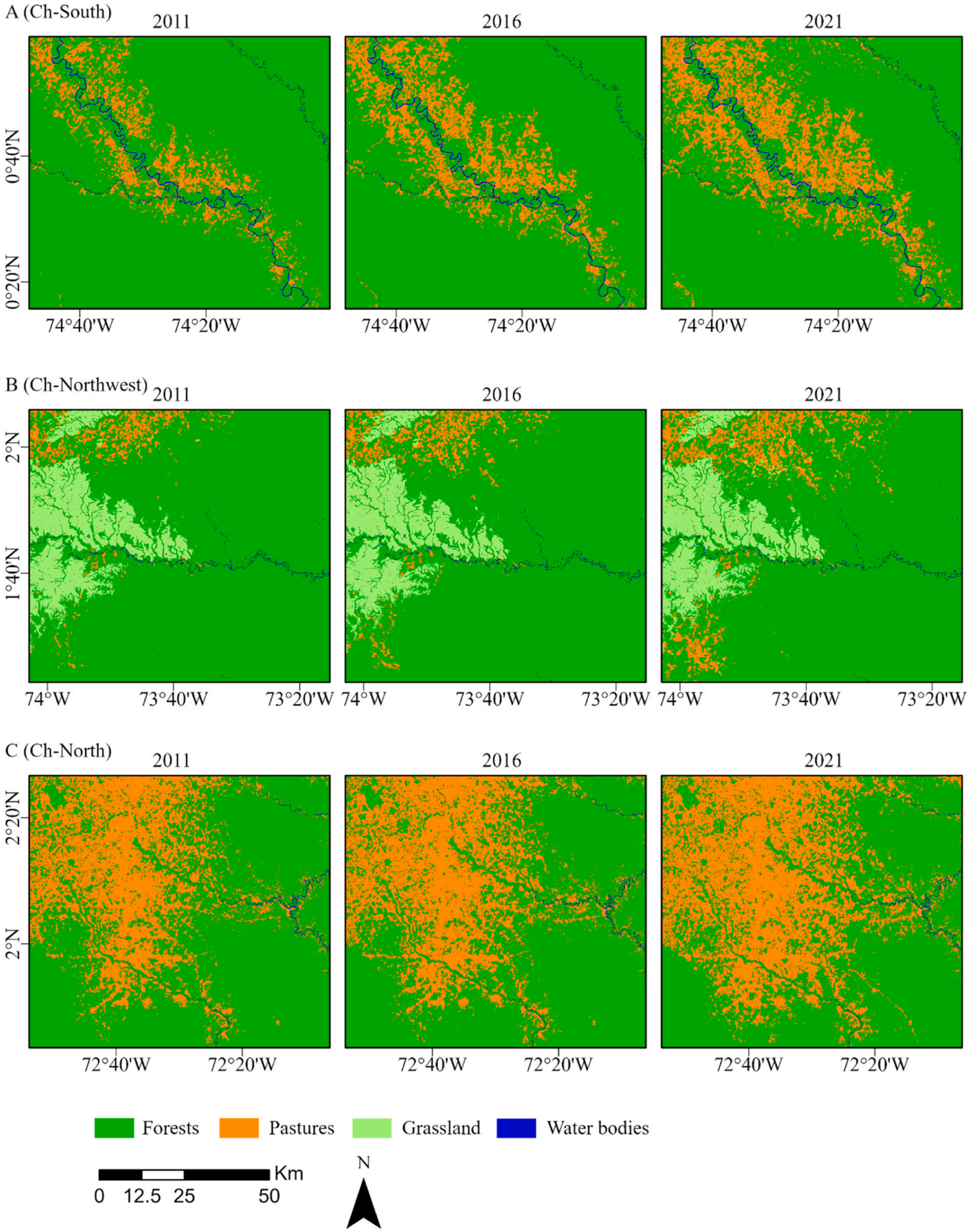

Fig. 6. Land cover classes in study regions A (Ch-South), B (Ch-Northwest), C (Ch-North) in 2011, 2016 and 2021.

Table 4
Land cover change from 2011 to 2021 (changes in percentage of total area).

Study regions	*Land cover types*	*2011 (%)*	*2016 (%)*	*2021 (%)*
Study region A (Ch-South)	Forests	88.91	84.02	78.80
	Pastures	9.91	14.78	19.94
	Water bodies	1.18	1.20	1.26
Study region B (Ch-Norhtwest)	Forests	81.90	80.61	77.24
	Pastures	3.88	5.13	8.01
	Grassland	13.87	13.91	14.40
	Water bodies	0.35	0.35	0.35
Study region C (Ch-North)	Forests	67.42	64.36	58.00
	Pastures	32.21	35.27	41.63
	Water bodies	0.37	0.37	0.37

areas. The forest class always presented a higher accuracy whereas pastures were more susceptible to confusion, due to Natural pastures (i.e., grasslands), coca, and forest regrowth, might have a similar spectro-temporal signature that introduces some errors during the CCDC classification process, particularly in the Ch-Northwest study region where the user's accuracy for pastures was only 62.0 %.

4.4. Structural connectivity analysis

The MSPA analysis (Fig. 7) concentrated on three classes: core, edge, and bridge. The core habitat is considered to as the source of a variety of ecological processes. The edge signifies the transition zone between the marginal zone of the core area and the transitional areas located at the external non-core regions, typically the peripheral forest. Bridges correspond to the narrow and elongated areas that connect patches of different core areas, embodying the characteristics of ecological corridors and linking the landscape to at least two separate core areas (Clerici and Vogt, 2013). The other structural categories identified by the MSPA (Islet, Perforation, Loop, and Branch) were not directly exploited for the purpose of this work. The proportion of the area for these three categories, in addition to the background, was computed (Appendix 1, Table 5). Also, we analyzed the background class, i.e. the class merging all non-forest land cover. If expanding, this shows decreasing trends of forest habitat. All MSPA segmentation algorithms are explained in detail in Soille and Vogt (2009).

Our findings indicate that the core was the most representative category of MSPA across all study regions. The distribution by core and all classes quantity and area size is delineated in Table 5, see Appendix. In region A, the forest habitat comprised 76.67 % of the core in 2011, which decreased to 63.3 % in 2021. Similarly, region B exhibited a core of 72.91 % in 2011, which declined to 65.11 % in 2021. In study region C, the core constituted 46.62 % in 2011 and 37.14 % in 2021. The core area in subset study area A (Ch-South) experienced the most substantial decline.

Over time, the core area diminished, corresponding to the reduction in forest area, while the edge and background areas expanded. No significant change was observed in the bridge category. Moreover, the rate of core area declines during 2016–2021 was higher compared to the period of 2011–2016 in all three subset study areas. This suggests that landscape fragmentation began to accelerate post-2016.

5. Discussion

Evidence suggests that the expansion of cattle ranching in the Colombian Amazon escalated with the introduction of nearly 1 million heads of cattle between 2016 and 2021, culminating in an estimated total of 2091,912 cattle heads in 2021 (FCDS, 2021a). During 2016–2019, it was reported that 300,415 ha were deforested in the municipalities of San Vicente del Caguán, Cartagena del Chairá, La Macarena, San José del Guaviare, El Retorno, Calamar, Miraflores, and Solano. These areas correspond to regions where cattle numbers are on the rise in the Colombian Amazon region (Valenzuela, 2021). Our analysis identified that the most severe land use change from forest to pasture transpired in the Guaviare province (study region C), and the colonization front in the Caquetá province (study regions A and B) (Fig. 6). While cattle ranching is a significant factor, other land use changes might also occur in the area of study e.g. coca farming, pasture expansion, irregular infrastructure development, logging, and mining. However, the provinces of Miraflores, Calamar (Guaviare), and San Vicente del Caguán and Cartagena del Chairá (Caquetá) have registered the highest number of cattle herds since 2016 (FCDS, 2022).

Our findings indicate that pastures are encroaching upon the borders of the protected area in all three subset study regions A (Ch-South), B (Ch-Northwest), and C (Ch-North), thereby augmenting the arc of deforestation since 2011.This corresponding trend is observed across the majority of the Amazon region, where pastures have often been associated with speculative land grabbing for cattle ranching (Bowman et al., 2012). This phenomenon could be partially explained in our study areas by the arrival of investors interlinking the cattle and land speculation processes (see Dávalos et al., 2014).

Our analysis offers a comprehensive interpretation of deforestation, land cover change, and structural connectivity changes in the Colombian Amazon. Notably, deforestation in the conflict-prone zones intensified following the peace process in 2016 (Prem et al., 2020). Investigating the spatio-temporal dynamics of deforestation and effects on spatial morphology of the landscape caused by the transformation of forests, correlates with decreasing species population trends where habitat loss and degradation are significant

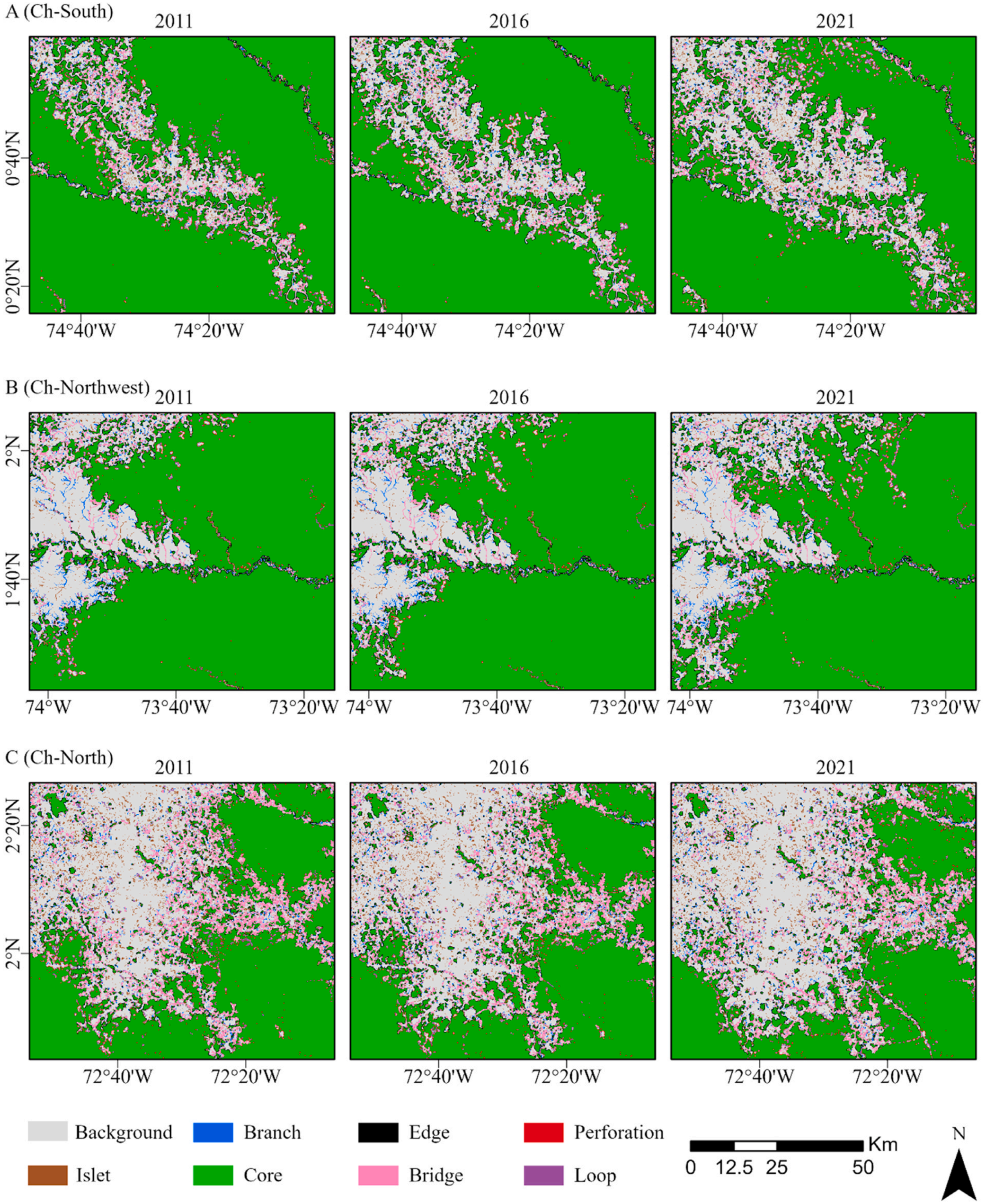

Fig. 7. MSPA classes in study regions A (Ch-South), B (Ch-Northwest), C (Ch-North) in 2011, 2016 and 2021.

factors (Braga-Pereira et al., 2020; Mendiratta et al., 2021), allowing us to assert that following the peace agreement the reduction of structural connectivity in the subsets occurred. The implications of deforestation are manifold. From an environmental perspective, the loss of forests leads to a significant loss of biodiversity, disruption of ecosystem services (e.g., nutrient cycling, water regulation, and pollination), and potential ecological imbalances. In the case of land cover change, the expansion of pastures, particularly driven by cattle ranching, affects soil, carbon sequestration potential, and increased greenhouse gas emissions (Pütz et al., 2014; Silva et al., 2020). The introduction of nearly 1 million heads of cattle (Uribe, 2019; Vilela et al., 2020) (FCDS, 2021a) indicates a shift in land cover change from tropical forests to extensive agricultural activities. The continuous expansion of irregular road construction might have an influence in this fast land transformation, threatening protected areas like the Serranía de Chiribiquete national park (Finer and Ariñez, 2024).

Throughout the analysis period, land cover classes underwent significant transformations as the collective forest area within the three subset study areas decreased by 24.19 % from 2011 to 2021, compared to the pasture area, which increased by 23.58 %. Study area A (Ch-South) experienced the most substantial decrease in forest area and the largest increase in pasture area, followed by study regions C (Ch-Northwest) and B (Ch-North). Notably, this change was more pronounced during the post-conflict period of 2016–2021, compared to the conflict period of 2011–2016, suggesting a well-known link between socio-political factors and land use change.

Other studies have confirmed a complex interplay between socio-political factors, land cover change and environmental degradation. Evidence suggests that regions in Colombia cultivated with coca experienced an expansion during 2006–2015 with declining armed conflict-related incidents, following the peace process, the withdrawal of the FARC led to a swift and significant increase in deforested areas (Landholm et al., 2019). Moreover, recent temporal trends indicate an escalation in deforestation across Colombia, implying a potential adverse impact of the peace accord on forest conservation. In alignment with this observation, our empirical findings corroborate earlier research asserting that the peace treaty inadvertently exacerbated the deforestation crisis in Colombia, even in regions previously experiencing a decline in such environmental degradation (Ganzenmüller et al., 2022; Prem et al., 2020).

Additionally, as observed in Fig. 3, concentrated deforestation at higher levels can be observed in all selected subset study areas from 2011 to 2021. Patches of illegal deforestation are evident in various locations, including inside protected areas and along river basins, showed in Fig. 4. This is where cattle are transported across the rivers to be sold in cattle markets, such as in the municipality of Cartagena del Chairá. The maps and the satellite images indicate deforestation in the southern area of Caño los Esteros in Guaviare towards the border of the Chiribiquete protected area in Ch Northwest, and towards the Yaguará II indigenous reserve in our study area Ch-south. This can significantly impact the livelihoods of campesinos and other communities due to extensive land grabs. A process that has been brought forward in the connection with deforestation and expansion of cattle ranching is land grabbing, which has been responsible for 61 % of 2021's total deforestation (MADS and IDEAM, 2022), and leads to increased road construction (Jones and Ramírez 2021; Lewis 2005; Li et al., 2014; Ministerio de Medio Amabiente, 2023; Moraes et al., 2023; NICFI,2021; Nobre and Borma, 2009; Pasquarella et al., 2022; Saturnino and Franco 2013; Silva Junior et al., 2020; Sistema de Monitoreo de Bosques, 2022; Staal et al., 2020; Turner and Gardner 2016; United Nations, 2022; Zhong et al., 2020).

According to our analysis using RAISG data, it is difficult to attribute a significant relation between road expansion and deforestation, as shown in Fig. 1. It is challenging to ascertain if the illegal road system could directly affect the trend, primarily due to the lack of an updated dataset on illegal road expansion, which necessitates further research. Concurrently, road infrastructure facilitates cattle transport, escalating forest fragmentation.

Indigenous communities and conservationists often oppose these activities, asserting they encroach on their ancestral lands, causing displacement and livelihood loss (De los Rios, 2022). This effectively expels them from their territories. Others report two significant cases in the study area are the indigenous reserves of Nukak in Guaviare and Yaguará II in Caquetá provinces (FIP and adelphi, 2021).

The increasing area of pasture exerts pressure on structural connectivity, thereby challenging conservation efforts at the landscape scale. Clear-cutting often accompanied by burning creates smaller patches, reducing Amazon forests' resilience and adaptability to changing climatic conditions like heat or drought. This leads to a decline in forest health and productivity (Gibson et al., 2011; Giammarese et al., 2023). These changes could increase vulnerability for key biodiversity hotspots in the Andes-Amazon transition zone (Murillo-Sandoval et al., 2022).

The change in the core class over the investigation period implied that the area of undisturbed forest has been reduced in all three study areas. The largest presence of structural corridors is in study area C (Ch-North), with a decrease from 2016 (9.83 %) to 2021 (8.42 %). Long-term consequences of the loss of core area represents a continuous hazard in the conversion of forests to pastures that can influence the rapid degradation and erosion of ecosystem functions. Other studies have shown that potential habitat loss and fragmentation have increased vulnerability to edge effects and biodiversity loss in these types of tropical ecosystems (Barlow et al., 2012; Fearnside, 2005; Laurance et al., 2002,2021).

Our findings support the detection of patterns in borders of PAs as areas of interest which can enhance decision-makers' understanding of the effects of reduced ecological permeability. This is particularly relevant where tropical Amazon forest and savanna ecosystems continue to disappear in the Andes-Amazon transition zone. The transformation of land cover classes and the resulting impact on structural connectivity underscores the need for integrative and more effective planning conservation policies. While the interlinkages between cattle mobilization, deforestation, and cattle rearing are complex (Castro-Nunez et al., 2021), monitoring key locations is vital for decision-makers planning future conservation/production policies outside Protected Areas.

Our results advocate the use of combined approaches with remote sensing tools as a way to assess the impact of several socio-economic effects on deforestation. This is given the diversity in drivers of deforestation dynamics in agricultural/forest frontiers developing in the arc of deforestation, corresponding to selected three subset study areas in this study. The complex progression of pastures for cattle ranching necessitates a detailed and systematic understanding of the underlying processes inherent in frontier dynamics (Baumann et al., 2022), as they are context-specific to land governance (Pacheco et al., 2020). This is relevant as the expanding clearing of forest to pastures for cattle remains economically attractive due to relatively low risk, minimal labor, well-established markets, and the availability of various government subsidies (Lima et al., 2021). Further research on alternative economies for locals and more production/conservation approaches to reduce deforestation in the area is needed.

A recent study suggests that without implementing control policies on deforestation and land cover change, the Amazon region could lose at least 2.1 million hectares of forests in the next two decades (Agudelo-Hz et al., 2023). Despite government efforts to promote sustainable agriculture, food, and agricultural production they act as pressuring main drivers of global biodiversity loss (Hoang et al., 2023).

While our study covers only a limited part of the Colombian Amazon region, we believe that this limitation has small impact on the generalization of results since the study areas include common patterns of land change. However, deepening the analysis with incorporation of additional data, e.g. on road expansion and other factors, will present ways for future studies to deepen the understanding of the processes.

6. Conclusions

This study addresses land transformation from deforestation and land cover change to structural connectivity of forests in key subsets in the Amazon region. The observed pasture expansion for cattle ranching in the northern Amazon study areas was found to disrupt landscape structural connectivity. Our metrics indicated a shrinking core area over time, reflecting forest reduction and an expansion of edge and background areas, mirroring pasture growth. Comprehensive analysis of structural connectivity in the northern Amazon is indispensable for informing land use policy and aids decision-makers in understanding the crucial role of maintaining ecosystem stability, emphasizing the significance of landscape connectivity. Our results confirm that rapid changes in connectivity are linked to pasture expansion. We acclaim that future conservation programs should include landscape spatial properties to enhance the approach towards addressing deforestation and ecological connectivity more effectively.

The detected patterns near the boundaries of and within protected areas raise serious concerns about the encroachment into these ecologically sensitive zones. Encroachment of pastures could compromise the conservation value of these areas and lead to a loss of biodiversity. The ongoing phenomena of habitat depletion and deforestation can persistently exacerbate the deterioration of ecosystems, posing a risk to the variety of life forms and compromising the connectivity of key transitional zones with PAs within the northwestern Colombian Amazon arc. Therefore, devising efficient strategies to mitigate these impacts and preserve these critical ecosystems for future generations is becoming imperative.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Jesica Lopez reports financial support and travel were provided by The Swedish Society for Anthropology and Geography. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Appendix 1

Table 1
CCDC parameters of Landsat images.

Input parameters	Landsat
collection	Landsat collection
breakpointBands	Green, Red, NIR, SWIR1, SWIR2
tmaskBands	Green, SWIR2
minObservations	6
chiSquareProbability	0.99
minNumOfYearsScaler	1.33
dateFormat	1
lambda	0.002
maxIterations	25000

Table 2
List of available variables from CCDC results.

Variables type	Options	Description
Spectral bands	BLUE, GREEN, RED, NIR, SWIR1, SWIR2	Spectral bands used for land cover classification
Coefficients	INTP, SLP, SIN,COS,SIN2,COS2,SIN3,COS3,RMSE	Model parameters for each spectral band
Derivatives	AMPLITUDE, PHASE, AMPLITUDE2, PHASE2,AMPLITUDE3, PHASE3	Seasonal metrics generated from coefficients

Table 3
Allocation of training data in study regions.

Land cover types	Study region 1 (Ch-South)	Study region 2 (Ch-Northwest)	Study region 3 (Ch-North)
Forests	700	650	650
Pastures	200	100	300
Grasslands	0	200	0
Water bodies	100	50	50

Table 4a
Accuracy assessment of the land cover map in 2021 obtained using the CCDC algorithm.

		Reference data					
		Forests	Pastures	Water bodies	Total	User's	
CCDC result (Ch-South)	*Forests*	**130**	0	0	130	100.0 %	
	Pastures	15	**65**	0	80	81.3 %	
	Water bodies	0	0	**40**	40	100.0 %	
	Total	145	65	40	250		
	Producer's	89.7 %	100.0 %	100.0 %	*Overall*	94.0 %	
		Reference data					
		Forests	*Pastures*	*Grasslands*	*Water bodies*	*Total*	*User's*
CCDC result (Ch-Northwest)	*Forests*	**110**	0	0	0	110	100.0 %
	Pastures	12	**31**	7	0	50	62.0 %
	Grasslands	1	1	**68**	0	70	97.1 %
	Water bodies	0	0	0	**20**	20	100.0 %
	Total	123	32	75	20	250	
	Producer's	89.4 %	96.9 %	90.7 %	100.0 %	*Overall*	91.6 %
		Reference data					
		Forests	*Pastures*	*Water bodies*	*Total*	*User's*	
CCDC result (Ch-North)	*Forests*	**119**	1	0	120	99.2 %	
	Pastures	1	**109**	0	110	99.1 %	
	Water bodies	0	0	**20**	20	100.0 %	
	Total	120	110	20	250		
	Producer's	99.2 %	99.1 %	100.0 %	*Overall*	99.2 %	

Table 5
Proportion of area for the core, edge, bridge and background classes.

Study regions	MSPA classes	2011 (%)	2016 (%)	2021 (%)
Study region A	Core	76.67	71.39	63.30
	Edge	3.43	4.12	4.69
	Bridge	3.88	3.48	3.88
	Background	11.13	16.01	21.23
Study region B	Core	72.91	70.84	65.11
	Edge	2.90	3.17	3.76
	Bridge	2.18	2.42	3.08
	Background	18.14	19.43	22.80
Study region C	Core	46.62	44.23	37.14
	Edge	3.66	4.16	4.31
	Bridge	9.83	8.85	8.42
	Background	32.61	35.66	42.02

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