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EVALUACIÓN MULTITEMPORAL DE LA DEFORESTACIÓN Y REGENERACIÓN DE BOSQUES: IMPLICACIONES SOBRE LA ESTRUCTURA ESPACIAL, LA CONECTIVIDAD Y LA COMPOSICIÓN DE LOS BOSQUES SECUNDARIOS DEL GUAVIARE (2000-2020)

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EVALUACIÓN MULTITEMPORAL DE LA DEFORESTACIÓN Y REGENERACIÓN DE BOSQUES: IMPLICACIONES SOBRE LA ESTRUCTURA ESPACIAL, LA CONECTIVIDAD Y LA COMPOSICIÓN DE LOS BOSQUES SECUNDARIOS DEL GUAVIARE (2000-2020)

Pregunta de investigación

¿Cómo han incidido los procesos de deforestación y regeneración de bosques en la estructura espacial, la conectividad y la composición sucesional de los bosques en el área de colonización del departamento de Guaviare durante los últimos 20 años?

Objetivo general

Evaluar los efectos de los procesos de deforestación y regeneración de bosques en la estructura espacial, la conectividad y la composición sucesional de los bosques en el área de colonización del departamento de Guaviare desde el año 2000.

Objetivos específicos

1. Determinar los cambios en los patrones espaciales de los bosques resultantes de los procesos de deforestación y regeneración.
2. Establecer la composición por edades de la vegetación secundaria y su contribución a la conectividad del bosque restante.
3. Evaluar los procesos de deforestación y regeneración del bosque dentro de las áreas protegidas y las reservas autóctonas en la dinámica de transformación del paisaje.

Multitemporal Assessment of Deforestation and Forest Regeneration: Implications on Spatial Structure, Connectivity and Composition of Secondary Forests of Guaviare (2000-2020)

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Abstract

Tropical forests are vital to global climate stability and their destruction threatens biodiversity, putting their ecological functions at risk. Since the 1970s, there has been an increase in deforestation of the northwestern Colombian Amazon, affecting the structural connectivity. Guaviare department is part of the colonization front, this territory has numerous Protected Areas and Indigenous Reserves that aim to mitigate the forest loss within the region. The present study analyzes forest loss and forest regeneration dynamics for three periods (2000-2015, 2015-2017 and 2017-2020), assessing the effectiveness of Protected areas and Indigenous reserves as barriers of deforestation. The results indicate that deforestation levels are constant throughout the study period with a total loss of 170,000 ha for the area of interest and 20,000 ha within Protected areas, conversely to Indigenous reserves that showed a neutralizing effect thanks to the presence of secondary vegetation. Secondary forests are a key component of landscape regarding their increasing effect in core area (5%) and connectivity (20 %) of remnant forest for the year 2020. The age structure of secondary vegetation in 2020 consisted of three age classes (3, 5 and 20 years of age), occurring in several patches.

Key words: Secondary vegetation, Forest loss, Tropical forest, Landscape ecology

1 Introduction

The tropics harbor over 40% of the planet's forests (FAO & UNEP, 2020). Tropical forests are characterized by their high levels of biodiversity that are primarily manifest in intact forests (Potapov et al., 2017), and also provide crucial ecological functions for global climate stability through regulation of processes such as carbon storage and water balance (Dias et al., 2015; Hansen and Defries, 2004). However, forests are vulnerable to anthropic intervention (Hamunyela et al., 2020) that leads to the transformation of land use and land cover (LULC) of landscapes, affecting its structure and functions at local, regional and global levels (Etter et al., 2005; Turner et al., 1989).

The degradation, clearing and fragmentation of forests threaten biodiversity and ecological integrity of ecosystems(Grantham et al., 2020; Hamunyela et al., 2020). The trajectories that follow land clearing of forests consist mainly of two possibilities: first, the deforested areas can be permanently transformed into non-forest type of landcover (e.g. agriculture, pasture, urban, etc.) (Armenteras et al., 2013; Potapov et al., 2017) or, second, a process of secondary succession can begin after being abandoned (Etter et al., 2005; Nunes et al., 2020; Saatchi, 1994). Despite the uncertainty of vegetation regrowth leading to a mature forest (Norden et al., 2015), secondary vegetation is fundamental to restoring the spatial structure of the forest landscape, climate change mitigation (Chazdon et al., 2016; Nunes et al., 2020), functions of harboring fauna and flora (Arroyo-Rodríguez et al., 2017) and acting as a buffer against disturbance and biodiversity loss from a landscape perspective (Hurtado-M et al., 2021).

Monitoring forest cover dynamics using remote sensing is an effective and practical way to reveal the relationships between anthropogenic actions and forest landscape transformation over time (Brovelli et al., 2020; Hansen et al., 2010; Senf et al., 2020). In order to advance and enhance the scope of biodiversity conservation, spatial modelling of landscape structural and functional connectivity comes in handy (Bennett, 2003; Crooks and Sanjayan, 2010; Lira et al., 2012; Rudnick et al., 2012). Given that several studies demonstrate that landscape changes can result in either permanent transformation or temporary change of forest cover (see, for example: Armenteras et al., 2017; Etter et al., 2005; Hamunyela et al., 2020; Nunes et al., 2015) it is crucial to assess the role of secondary vegetation in landscape configuration (Etter et al., 2005; Nunes et al., 2020).

A study conducted by Etter et al., (2005) in areas of the Colombian rainforests that had suffered prologued landscape transformation, found an increasing trend in secondary forest cover. Similarly, Teixeira et al., (2009) in their study of the Brazilian rainforest documented a tendency towards younger secondary forest in areas that had been subject to high rates of both deforestation and regeneration. However, recent studies have shed new light on the different pathways taken by regrown vegetation areas once the deforestation rates decrease. This is the case of the study conducted by Carvalho et al., (2019) whose findings report that when deforestation rates decrease, the transformation of secondary vegetation into non-forest land cover increases. Some of the most common uses for the transformed secondary vegetation are livestock, agriculture and palm oil plantations (Nunes et al., 2020).

Another approach to forest monitoring is through conservation, specifically in evaluating the effectiveness of protected areas (PA), being a figure frequently used to reduce deforestation (Andam et al., 2008). Several authors (Barber et al., 2014; Cabral et al., 2018) have shown that PA act as a barriers against forest clearing, but often given by their remote location and inaccessibility. On the contrary, Bray et al. (2008) found in their study that PA do not perform well in active colonization fronts. In this sense, Armenteras et al. (2009), study found that Colombian tropical lowland forest PA have been under increasing exposure to pressures in their surroundings for the past 30 years. The same study highlighted variability in performance between PA and indigenous reserves (IR), showing that PA perform better.

The northwestern Colombian Amazon, of which the Guaviare department is part, has been one of the most dynamic regions in terms of forest loss for the past several decades due to the presence of multiple colonization fronts and lack of territory planning (Armenteras et al., 2006; Etter et al., 2008, 2006a; Etter and Andrade, 1989). Despite high deforestation rates, large areas of previously cleared and cultivated lands are being abandoned, allowing regeneration processes to arise that can derive into secondary forest growth (Meyfroidt and Lambin, 2011; Soares-Filho et al., 2013). In other words, landscape regeneration in forest mosaics where the cover is not severely fragmented can lead to regeneration of vegetation (Armenteras-Pascual et al., 2013).

The department of Guaviare is both a biodiversity and deforestation hotspot (Armenteras et al., 2019). Land clearing processes are mainly produced by the transition from forests to pastures, evidencing forest loss in greater quantity and in a more accelerated way than in other areas of Colombian forests (Meza and Armenteras, 2018). However, studies regarding biodiversity loss and landscape transformation remain largely unknown for this region (Armenteras-Pascual et al., 2013; Armenteras et al., 2006).

This paper aims to contribute to a better understanding of the spatial dynamics of clearing and regeneration of tropical rainforests, and the effects of secondary forests on the spatial structure and connectivity of the forest mosaics in the Colombian Amazon. The main objectives being pursued are: (1) to determine the changes in spatial patterns of the resulting forests from deforestation and regeneration processes, (2) to establish the age composition of secondary vegetation and its contribution to the connectivity of the remaining forest and (3) to evaluate forest loss and forest regeneration processes within protected areas and indigenous reserves in landscape transformation dynamics.

2 Materials and methods

2.1 Study area

The department of Guaviare is located in the northwestern part of the Amazon occupying an area of 53,460 km2 corresponding to 5% of the Colombian territory, it is divided in four municipalities: Calamar, El Retorno, Miraflores and San José del Guaviare (UNODC, 2014). The area of interest (AOI) of this study occupies 39,130 km2 (Figure 1) and harbor the main human settlements of the region. This region is known as a deforestation hotspot, presenting colonisation processes since the 1970s (Armenteras et al., 2019).The main economic activities associated with land clearing are livestock and illicit coca crops (*Erythroxylum coca*), the second has been a critical driver for land cover transformation within the region in the past two decades (Armenteras et al., 2013, 2019; Etter et al., 2005; Etter, McAlpine, et al., 2006). The deforestation processes associated with these economic activities are related to accessibility factors to the forest such as distance to the interior of them, studies in similar areas have shown that land clearing processes follow the course of roads and navigable rivers (Armenteras et al., 2006; Etter et al., 2006b; Marsik et al., 2011).

The AOI is covered by tropical forest biome and some prominent rocky outcrops, the average altitude is 180 m (SIAT-AC, 2020). The climate is tropical, humid and monomodal. Humidity varies between 80% and 90% throughout the year, the average annual rainfall is 2,800 mm, presenting a dry season (December-March) and a rainy season (April-November) with an average daily temperature of 25°C (SIAT-AC, 2020). Soils are acid and nutrient deficient, thus, highly susceptible to deterioration (SIAT-AC, 2020). Guaviare is crossed by numerous rivers which are major affluents of the Orinoco and the Amazon basins (UNODC, 2014). This region comprises a variety of tropical rain forest systems, characterized by having complex vegetation communities and ecological systems, reflecting in high species richness and abundance (Meyfroidt and Lambin, 2011; Souza et al., 2020; Wang et al., 2019).

The AOI (Figure 1.) comprehends a series of 5 PA (1,510,104 ha) and 28 IR (465,888 ha) to protect the biodiversity. PA and IR are both conservation strategies with different purposes contributing towards social, environmental, and cultural dimensions (Barragán Alvarado, 2008). For this study, these figures are going to be taken as conservation strategies in the broader sense. Acknowledging that deforestation processes can occur in a different way within the territorial boundaries of PA and IR, as well as social and environmental management (Andam et al., 2008; Armenteras et al., 2009).

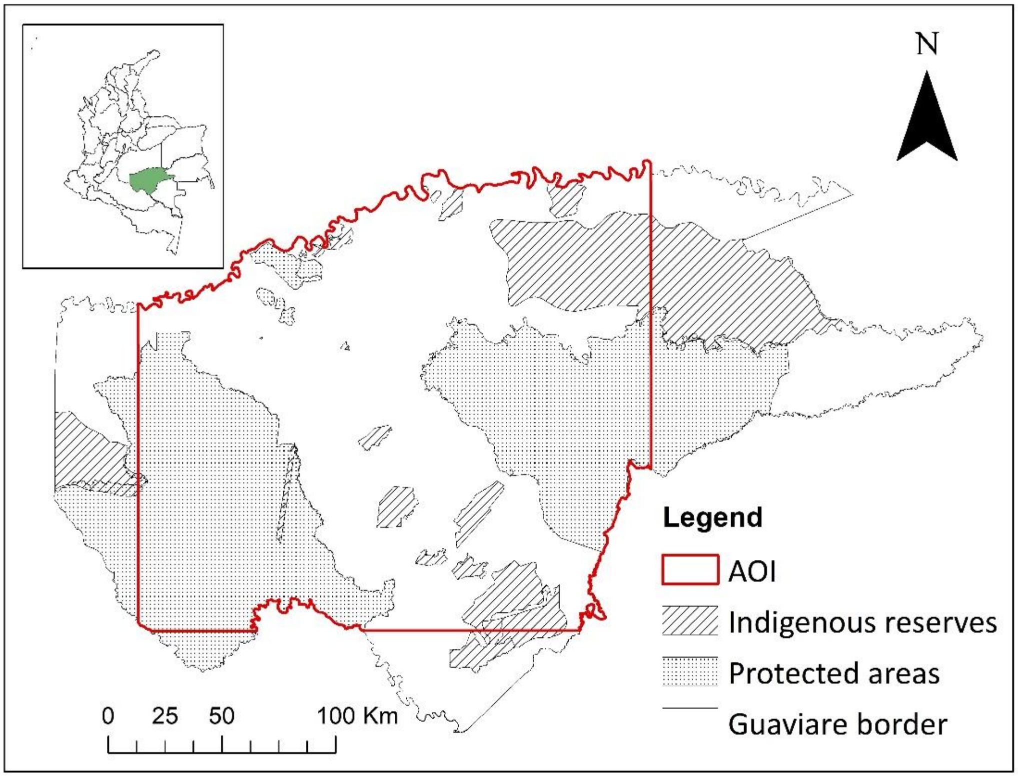

Figure 1. Study area map

2.2 Data sources and processing

Satellite images were used to generate land cover maps for the years 2000, 2015, 2017 and 2020, respectively. These years were chosen according to image quality and cloud cover percentage, between 2000 and 2015 there were no images available that met the criteria of 20% or less cloud cover percentage, explaining the 15-year gap between the first and second year analyzed. The satellite images used in this study were retrieved from Google Earth Engine GEE (Gorelick et al., 2017; Hurni et al., 2017) from Landsat 5 Thematic Mapper (TM), Landsat 7 Thematic Mapper Plus (ETM+) and Landsat 8 Operational Land Imager (OLI), using Surface Reflectance Tier 1 Collection which has bottom of atmosphere (BOA) images with corrected data that have well characterized radiometry and are inter-calibrated across the different Landsat instruments. The georegistration of Tier 1 scenes is consistent and within prescribed image-to-image tolerances of ≦ 12-meter radial root mean square error (Gorelick et al., 2017; Young et al., 2017). Geometric correction is dismissed if only L1T Landsat images are used in change detection, hence, considered most suitable for time series analysis (Hamunyela et al., 2020; Zhu, 2017).

For each year, a mosaic filtered by the AOI was generated with the median value of the image pixels to create a homogeneous composite, 54 images were processed in total (10 images for year 2000, 8 for 2015, 27 for 2017 and 9 for 2020). Then, a cloud mask function was applied using the pixel quality attributes based on the CFMASK algorithm (Hamunyela et al., 2020).

In order to analyze deforestation and regeneration dynamics through the years, land cover maps were created for each year as proposed by Etter, Mcalpine, et al.,(2006). A supervised classification through visual interpretation of the satellite images was performed (Jensen, 2015; Richards and Jia, 2006) using Random Forest algorithm in GEE (Brovelli et al., 2020). The classifier was applied to five broad land cover classes: forest, secondary vegetation (SV), cleared land, water bodies and rocky outcrops. The training sites were selected through stratified random sampling (Hamunyela et al., 2020; Jensen, 2015; Olofsson et al., 2014), more than 1,000 pixels were selected for each date. To test the model, an accuracy assessment was performed creating new validation sites, obtaining matrix of error, overall accuracy, producer’s accuracy, user´s accuracy and kappa coefficient. Errors in classification can be attributed to spectral confusion between regenerating forest and crops

(Prates-Clark et al., 2009) explaining why SV class had the lowest accuracy of classification. After the classification, a majority filter of 3X3 moving window was applied to remove the isolated pixels and reduce the image noise for each land cover map (Jensen, 2015; Wang et al., 2019) in ArcMap 10.7.1 (Esri, 2008).

2.2 Forest clearing and forest regeneration dynamics

To assess forest loss and forest regeneration for the two decades, forest-non-forest maps were produced for each year studied, obtaining forest clearing and forest regeneration maps for the 2000-2015, 2015-2017 and 2017-2020 periods. These data were also calculated within 5 Protected areas and 28 Indigenous reserves, measuring the impact of landscape dynamics inside and outside these boundaries. Information regarding PA and IR was retrieved from the Colombian Geographic Institute Agustín Codazzi (IGAC, 2020), changes in the extension of PA and IR during the study period were not taken into account for the forest dynamics calculations, therefore, the areas used for measurements correspond to the year 2020. Rates of change were quantified for each conservation figure to compare the effectiveness individually, and for the combined effect by overlaying both PA and IR.

The annual rates of class cover change were calculated following the formula proposed by Puyravaud (2003):

$$r = (1/(t2 - t1)) * \ln(A2/A1) ,$$

Where r is the rate of change per class for each year, t1 and t2 are the initial and the final time, and A1 and A2 are the class areas for the beginning and the end of the period.

A forest age map was created for the year 2020 in ArcMap 10.7.1 (Esri, 2008), in order to identify the ages of vegetation regrowth within the secondary forest mosaic. This map was based on the forest-non-forest maps obtained for the four dates studied, focusing on the changes (presence vs absence) of SV pixels from each period to the year 2020. Additionally, the age classes were reclassified as Etter et al., (2005) proposed, by overlapping the forest-non-forest maps for the four years studied. The SV pixels that persisted from years 2000 to 2020 were taken as 20 years old of secondary forest (SF), the same calculation was done for 2015-2020 (5 years of SF age) and 2017-2020 (3 years of SF age). The mature forest corresponds to the forest pixels that persisted intact within the 20-year period studied, since the exact age cannot be calculated within the study period proposed. In other words, the land cover of SV calculated between periods (2000-2015, 2015-2017 and 2017-2020) will be taken as SF, whilst mature forest refers to the pixels of forest that persisted throughout the study period.

2.3 Remnant forest connectivity

Landscape metrics adapted from (Etter et al., 2005; Zhang et al., 2020) were implemented to understand the effect of SV in the connectivity of remnant mature forest mosaics (fixed depth of 100m and connectance threshold of 100m were applied) using FRAGSTATS 4.2 (McGarigal et al., 2012). The metrics chosen for this study were : number of patches (NP), mean patch size (MPS), total core area (TCA) and connectance index (CONNECT), proven to be useful to analyze land cover dynamics (Spanowicz and Jaeger, 2019). All the proposed metrics were calculated for the secondary forest ages, whilst TCA and CONNECT were calculated for the three transition periods with and without the presence of SV and inside PA and IR.

3 Results

3.1 Land cover dynamics

Four land cover maps were generated for the years 2000, 2015, 2017 and 2020 with an overall accuracy of classification of 97%, 96%, 98% and 93%, respectively (Appendix A) (Figure 2.) using GEE at 30 m resolution. A tendency towards forest reduction and land clearing expansion is evident to the naked eye when contrasting the land cover maps. Each map exhibits forest as the dominant cover, followed by cleared land, then secondary vegetation and lastly water and rocky outcrops. Tables 1 and 2. present the temporal trend of area and annual rate of change for the main classes being monitored in this paper: forest, secondary vegetation (SV) and cleared land. The category of others comprises both water and rocky outcrops which are beyond the scope of the study.

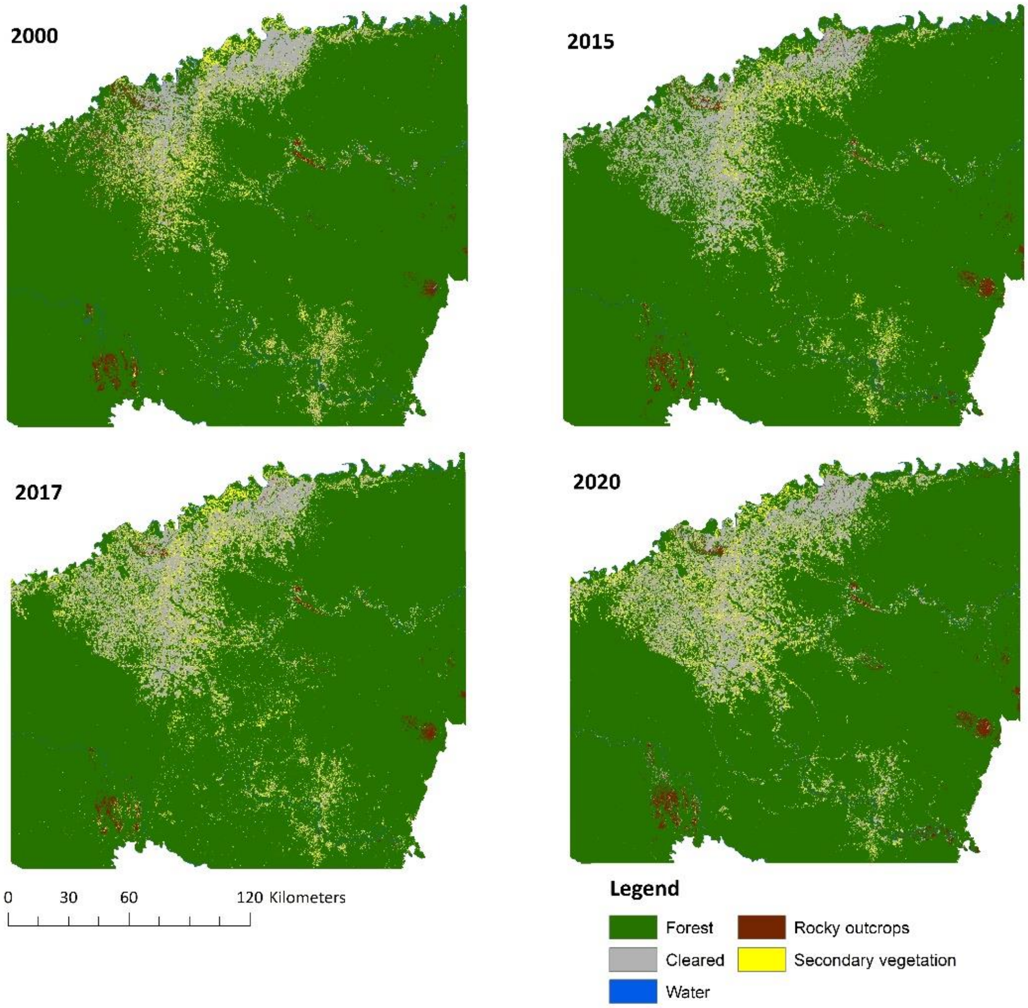

Figure 2. Land cover maps

Table 1. Land cover type, area and percentage

Land cover	2000		2015		2017		2020	
	ha	%	ha	%	ha	%	ha	%
Forest	3,423,163	87.48	3,379,289	86.36	3,305,201	84.47	3,250,278	83.06
Secondary vegetation	138,802	3.55	120,630	3.08	199,090	5.09	169,799	4.34
Cleared	264,290	6.75	346,756	8.86	360,409	9.21	422,876	10.81
Others	86,782	2.22	66,363	1.70	48,338	1.24	70,084	1.79

Table 2. Annual Rate of Change for the study period

Land cover	Annual Rate of Change (2000-2015) %	Annual Rate of Change (2015-2017) %	Annual Rate of Change (2017-2020) %
Forest	-0.001	-0.011	-0.006
Secondary vegetation	-0.009	0.251	-0.053
Cleared	0.018	0.019	0.053
Others	-0.018	-0.158	0.124

Forest cover was reduced by nearly 175,000 ha for the study period, with the period 2015-2017 with an annual rate of change of 0.011, almost double than 2017-2020 (0.006%), and ten times higher than the 2000-2015 average (Table 2). On the other hand, cleared areas showed an increase in its area for each year with an annual rate of change ranging from 0.018% to 0.053 %. Secondary vegetation was the most dynamic cover with areas increasing and decreasing through the periods, ranging in cover between 3.08% to 5.09% of the AOI. This class presented the greatest annual rate of change for the 2015-2017 (0.251%), while a reduced annual rate of change was observed for the other two periods.

3.2 Age structure and spatial metrics of the 2020 forest remnants

The mosaic of SF is composed by several patches of ages 3 (12,685ha), 5 (7,723 ha) and 20 (68,609 ha) years of regrowth as shown in Figure 3. which illustrates the synergetic effect of the SF to restore the mature forest structural connectivity through a closeup in a random chosen window. Table 3. and Figure 3. show the effect of different SF age patches individually contributing to increase the connectivity of the remnant forest (21.5%). The pattern of regrowth happens in several patches of different mean sizes ranging from 0.29 ha to 1 ha, while MPS of mature forest is considerably bigger (118 ha). The TCA of year 20 (449.82 ha) had the greater effect for total input of SF in 2020 forest cover by representing 30% of it.

Table 3. Number of patches, mean size of patches, total core area and connectance index of secondary forests in 2020

Secondary forest age (yr)	NP	MPS (ha)	TCA (ha)	CONNECT %
3	35712	0.3552	315.00	0.0016
5	26141	0.2954	4.95	0.002
20	66951	1.0248	449.82	0.0013
mature	26161	118.1993	2,595,578.40	0.004

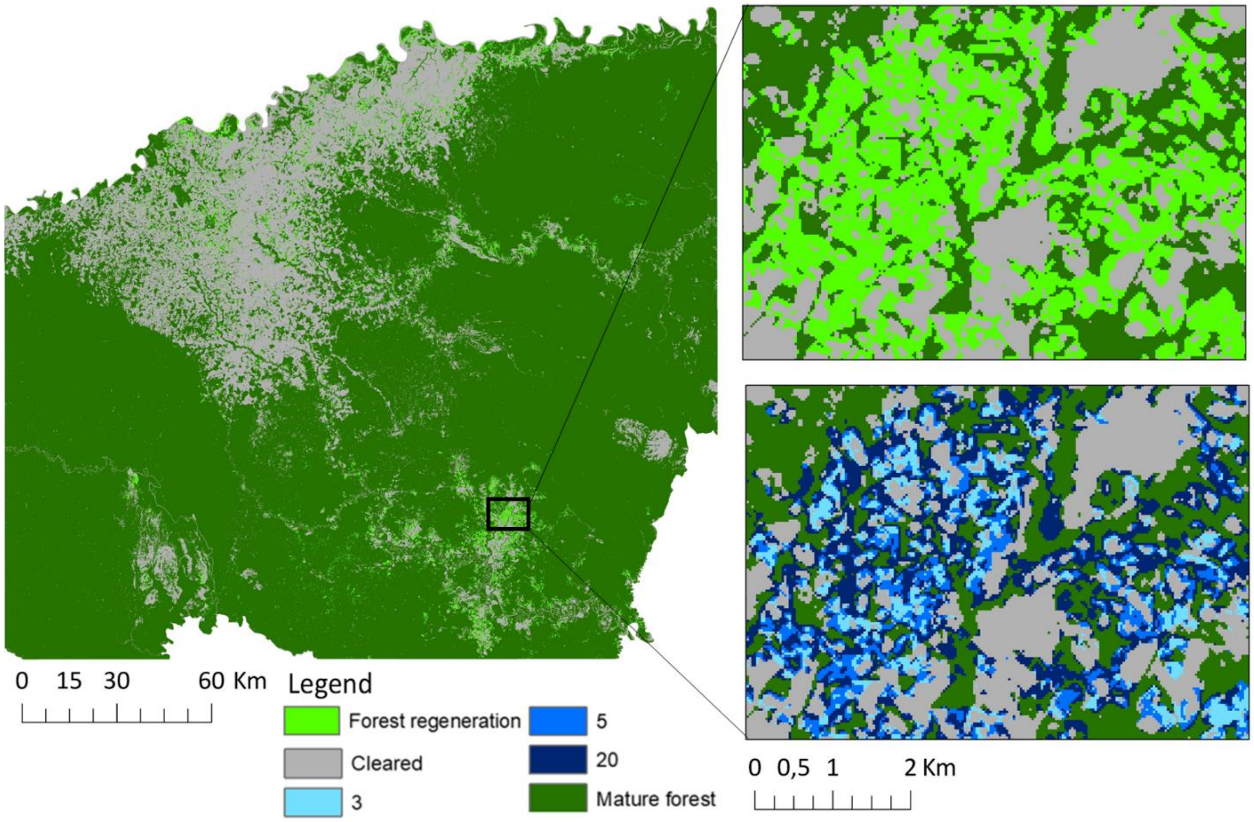

Figure 3. Mature and secondary forests, and approximate ages of secondary forest in 2020

3.3 Deforestation, regeneration and forest mosaic age in relation to protected areas and indigenous reserves

The results presented in Figure 4. show a similar pattern of forest clearing and forest regeneration dynamics within the three periods. There was an increasing trend in forest loss which was, nevertheless, mitigated by the growth of SF. However, this still resulted in forest loss for all three periods. The forest loss peaked in 2015-2017 (37,000 ha/yr) representing 42.85% of the total forest loss (172,885 ha) during the study period and decreased (19,166 ha) between 2017-2020.

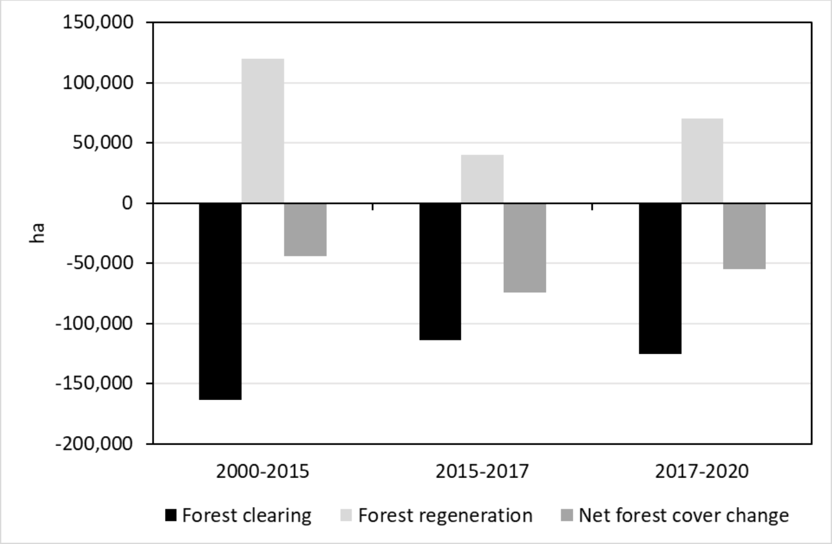

Figure 4. Forest cover change due to forest clearing and forest regeneration for the three periods

Forest regeneration process contributes to the generation of SF cover, which constitutes a part of the total forest cover (Table 4). 2015 was the year with more SF extent within the three years, contributing by 4% the total forest cover. Similarly, in 2017 the SF represented 1% of the forest cover, while in 2020 accounted for 2% of the forest.

Table 4. Forest cover composition, area and percentage

Year	Mature forest		Secondary forest		Total forest cover	
	ha	%	ha	%	ha	%
2015	3,259,605	96	119,684	4	3,379,289	100
2017	3,265,126	99	40,074	1	3,305,201	100
2020	3,180,002	98	70,275	2	3,250,278	100

Protected Areas cover 38.59 % of the AOI while Indigenous Reserves occupy 11.9%. Taken together these areas represent 1,975,992 ha (50%) of the AOI. The results given in Figure 5. reveal that forest dynamics follow an inverse trend in both figures of conservation for the 2000-2015 and 2015-2017 periods, as opposed to 2017-2020 where both forest dynamics follow a similar pattern. Specifically, PA presented deforestation rates of 0.00029% and 0.004% for the first and last periods, while 0.0008% rate of regeneration for the second period. Conversely, IR revealed 0.0009% rate of regeneration in the first period and 0.004% and 0.002% rates of deforestation for the second and third periods. The joint effect of PA and IR shows a compensation of forest loss and forest regeneration in 2000-2015, and a replication of the general pattern of deforestation for the other two periods.

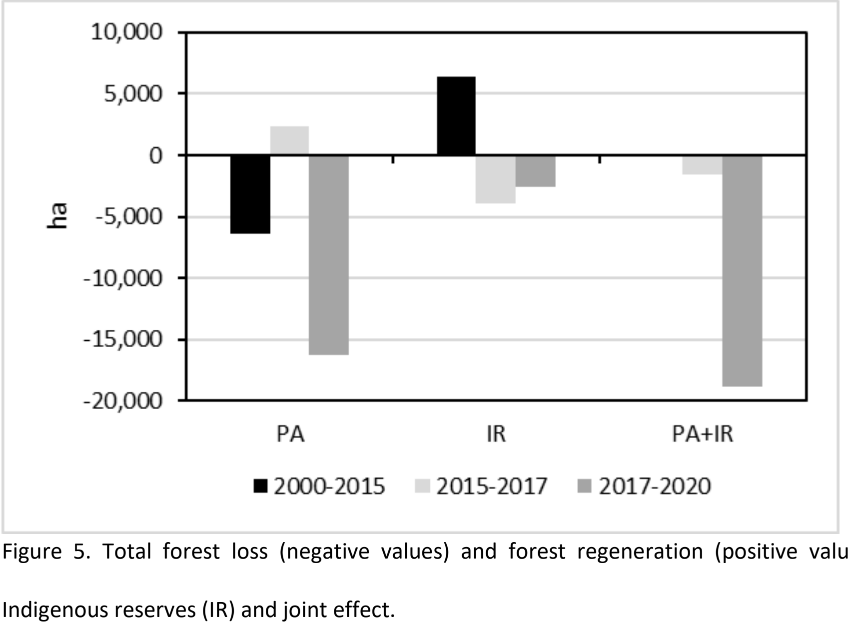

Figure 5. Total forest loss (negative values) and forest regeneration (positive values) within Protected areas (PA), Indigenous reserves (IR) and joint effect.

3.4 Effects of regenerating forests on the remnant forest spatial structure metrics

The cumulative effect of secondary vegetation the landscape metrics TCA (total core area) and CONNECT (connectance index) of remnant forests in 2020 (Figure 6a) and for the years 2015, 2017 and 2020, individually, indicate the positive effect of the regenerating forests in terms of overall forest core area and connectivity in the forest mosaic. In 2020 approximately 249,800 ha of SV had accumulated over the 2000-2020 period, increasing the forest connectivity by 21%. The positive effect of SV increased TCA by 2 to 6% and increased the connectivity over the study period by 27% in 2015, 4% in 2017 and 8% in 2020.

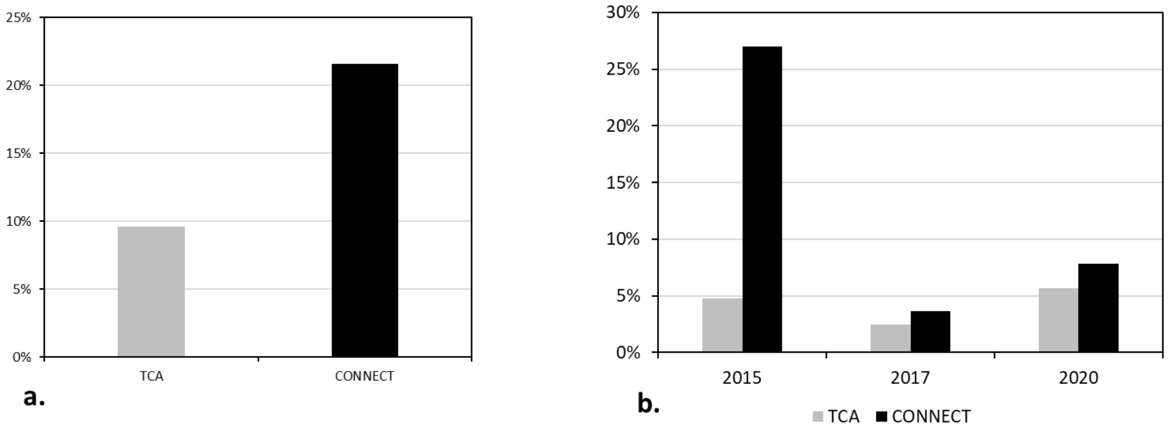

Figure 6. (a) Cumulative effect of secondary vegetation on TCA (100m) and CONNECT in 2020; (b) Effect of secondary vegetation on TCA (100m) and CONNECT for the three periods.

4 Discussion

Contemporary forests have accumulated an increasing footprint of the cumulative effects of land use over decades and even centuries, visible in the high proportion of secondary forests (Arroyo-Rodríguez et al., 2017; Rudnick et al., 2012), which implies a rethinking of the management and conservation of forest landscapes (Gullison and Hardner, 2018). This study exemplifies this phenomenon for the northern Amazon in Colombia. Although time series analysis of LULCC in tropical forest demonstrate a tendency towards forest loss (Figure 3.), at the same time remnant forests increasingly make-up a larger proportion of remnant forests. Various studies (Armenteras et al., 2006; Etter, Mcalpine, et al., 2006; Meyfroidt & Lambin, 2011; Soler et al., 2014) have documented the drivers behind deforestation and found a direct relationship between this process and socio-economic factors such as agriculture, infrastructure projects (roads), livestock (pasture creation) and illicit crops, among others. This multitemporal approach has been helpful in monitoring forest cover change over the last 20 years in this colonization front. Revealing the importance of secondary vegetation in the connectivity of remnant forest must include a consideration regarding variation between forest clearing and regeneration processes (Etter et al., 2005).

Deforestation and land use are a constant threat towards intact forests (Potapov et al., 2017), yet the contribution of vegetation regrowth is key to countervail forest loss in addition to stabilizing secondary forest. In particular, secondary vegetation is known for its potential to become a source of habitat for fauna and flora (Arroyo-Rodríguez et al., 2017; Meyfroidt and Lambin, 2011). In addition, the first decade of succession in tropical forests endorse pronounced transformations in the vegetation structure and composition and can generate a canopy closure (Chazdon, 2008). Therefore, it is important to deepen the research emphasizing in vegetation regrowth by exploring the patterns and configurations at the landscape level of the ages and mosaic distribution of vegetation regrowth (Etter et al., 2005; Prates-Clark et al., 2009).

Contributing to the focus of previous literature on the effects of secondary vegetation on landscape configuration of tropical forests, this study highlights its effect on core area and connectivity of remnant forest. Specifically, this study diverts from the previous interest in the extent, distribution, and drivers of change (Sloan et al., 2016) and instead emphasizes on the emerging patches of different SF ages (Nunes et al., 2020) which configure a complex mosaic. The findings suggest an overall increase over the core area and connectivity of remnant forest in the presence of SF. Similar findings were reported by Etter et al., (2005) in that improving the spatial structure of the remaining mature forests is directly related to the presence of SF. Is important to highlight that the 20-year window proposed in this study is unable to descry a trend regarding the potential of SF to become a mature forest within the AOI, because an effective regeneration process takes between 60 to 80 years (Brown and Lugo, 1990).

Studies in the Brazilian rainforests vary depending on the area where the data was taken (Atlantic vs Amazon). Where Atlantic forests are dominated by young secondary forest resulting from high dynamics of both deforestation and regeneration (Teixeira et al., 2009). Conversely, in the Amazon forests, the extent of SV is affected by increasing rates of deforestation in these areas which in turn compromise forest regeneration (Carvalho et al., 2019). The explanation rests on policies that promote concentrated forest clearing in SV areas to prevent the agricultural frontier from expanding (Soler et al., 2014). These findings add to the trend that this study follows concerning SF after deforestation peaks (2015-2017) depicting a positive relation between forest loss and SV clearing, as well as a parallel decrease in rates of change.

The results give an insight into the performance of both PA and IR as conservation figures. The cumulative results shed some light on the overall superior performance of IR in terms of mitigating deforestation for the time period studied. During the 20 years studied IR showed a neutralizing effect between forest clearing and forest regeneration (Figure 5.), opposing to the overall trend within PA that followed the general trend of the AOI, where deforestation stood out. Refuting the results of Armenteras et al., (2009) given that PA performed better than IR, where the forest loss within IR was approximately six times greater than within PA during 1985 to 2002.

The peak of forest loss coincides with the peace accord in 2016, paradoxically increasing deforestation inside PA, changing the paradigm that raised them as great barriers against forest loss (Barber et al., 2014). This result agrees with the statistics exposed by Clerici et al., (2020) postulating that approximately 79% of Colombian PA experienced increased forest loss in post-conflict years. The tendency of forest loss applies similarly for IR between 2017 and 2020, but in a smaller extent. Deforestation inside PA is alarming because of the extent and the acceleration of the process, challenging this type of conservation measures. Gonzáles-Gonzáles et al., (2021) suggest implementing complementary top-down governance measures for conservation with regionally adapted policy.

The different trajectories for PA and IR concerning deforestation rates can relate to the finding that indigenous territories may be less likely to be affected by external drivers and more likely to be affected by internal factors (Nolte et al., 2013). The case of Brazilian amazon, highlights that all types of conservation figures have been successful to avoid deforestation, mainly by their remoteness (Gullison and Hardner, 2018; Nepstad et al., 2006; Nolte et al., 2013). Nonetheless, forest loss in IR was related towards invasions from nonindigenous populations increasing natural resource exploitation.

Furthermore, deforestation within PA and IR was lower than outside of both figures, proving that legal protection is not always an effective way to mitigate the future forest loss (Potapov et al., 2017). Given that the access to them may change over time, making them available for the expansion of colonization front, hence research concerning socio-economic drivers at a finer scale is recommended.

5 Conclusions

Secondary forests are becoming more and more prominent, therefore, reevaluating the actual conservation and management strategies is important. In order to favor the development of secondary vegetation and ensure the conservation of intact forests. Over the 20- year period cover by this study, there has been changes in land covers manifested primarily by a reduction of the forest cover that parallelly increased the cleared area. The overall forest loss within the AOI represented 5% of the forest cover, the same tendency was evidenced within PA, while IR had a neutralizing effect and presenting a better performance. The peak of deforestation occurred between 2015-2017 triggered by the offset of the peace accord in 2016.

Forest clearing and forest regeneration dynamics were influenced by the persistence of secondary vegetation occupying an intermediary stage. Secondary vegetation plays an important role in forest change, by increasing the core area and connectivity of remnant forest. Studies concerning the quality of SV are necessary to compliment the results presented in this research. The secondary forest in 2020 consists of a mosaic configurated by patches of SV of different ages (3, 5, 20 years of age).

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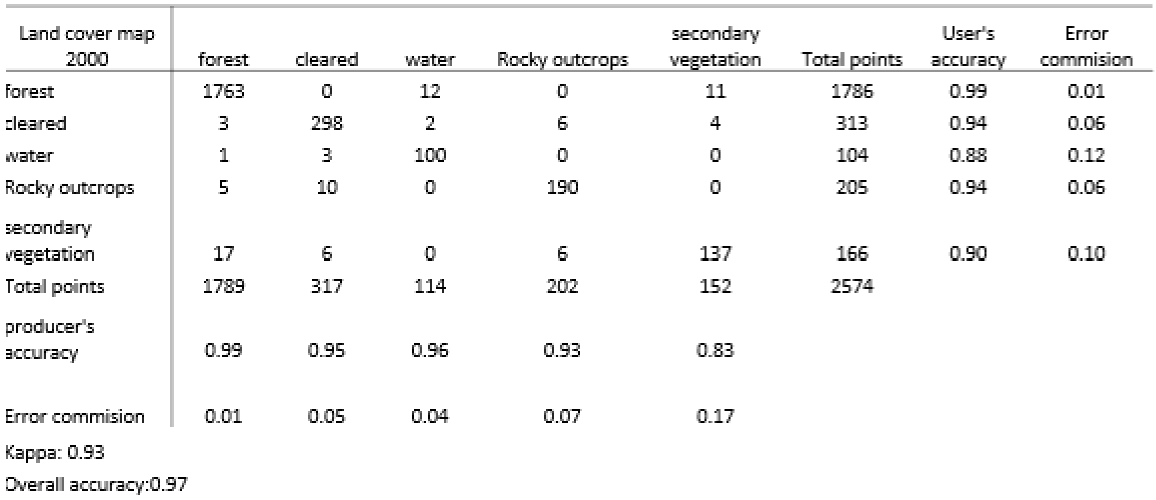
Appendix A

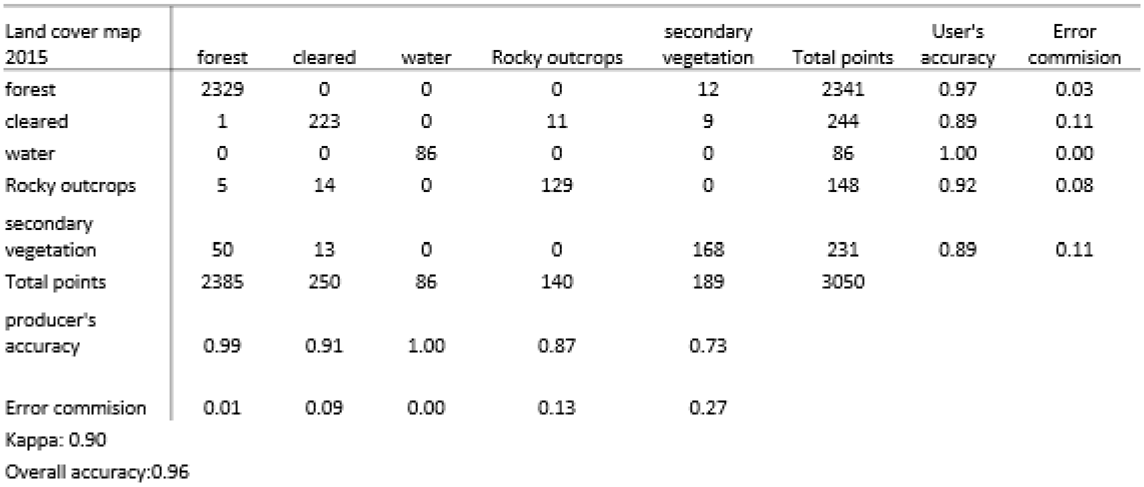
Confusion matrixes for the land cover maps.



Land cover map 2000	forest	cleared	water	Rocky outcrops	secondary vegetation	Total points	User's accuracy	Error commision
forest	1763	0	12	0	11	1786	0.99	0.01
cleared	3	298	2	6	4	313	0.94	0.06
water	1	3	100	0	0	104	0.88	0.12
Rocky outcrops	5	10	0	190	0	205	0.94	0.06
secondary vegetation	17	6	0	6	137	166	0.90	0.10
Total points	1789	317	114	202	152	2574		
producer's accuracy	0.99	0.95	0.96	0.93	0.83			
Error commision	0.01	0.05	0.04	0.07	0.17			

Kappa: 0.93

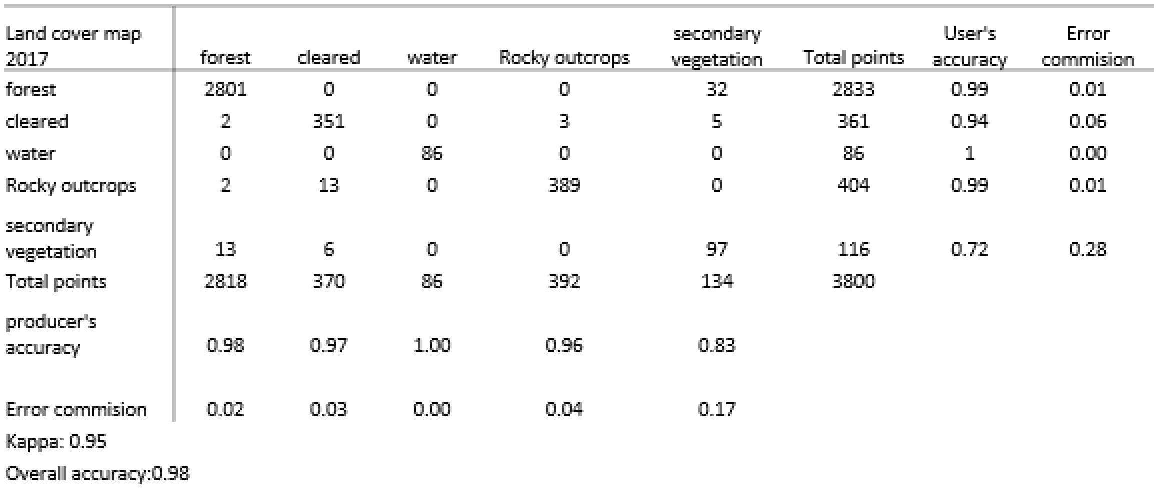
Overall accuracy:0.97



Land cover map 2015	forest	cleared	water	Rocky outcrops	secondary vegetation	Total points	User's accuracy	Error commision
forest	2329	0	0	0	12	2341	0.97	0.03
cleared	1	223	0	11	9	244	0.89	0.11
water	0	0	86	0	0	86	1.00	0.00
Rocky outcrops	5	14	0	129	0	148	0.92	0.08
secondary vegetation	50	13	0	0	168	231	0.89	0.11
Total points	2385	250	86	140	189	3050		
producer's accuracy	0.99	0.91	1.00	0.87	0.73			
Error commision	0.01	0.09	0.00	0.13	0.27			

Kappa: 0.90

Overall accuracy:0.96



Land cover map 2017	forest	cleared	water	Rocky outcrops	secondary vegetation	Total points	User's accuracy	Error commision
forest	2801	0	0	0	32	2833	0.99	0.01
cleared	2	351	0	3	5	361	0.94	0.06
water	0	0	86	0	0	86	1	0.00
Rocky outcrops	2	13	0	389	0	404	0.99	0.01
secondary vegetation	13	6	0	0	97	116	0.72	0.28
Total points	2818	370	86	392	134	3800		
producer's accuracy	0.98	0.97	1.00	0.96	0.83			
Error commision	0.02	0.03	0.00	0.04	0.17			

Kappa: 0.95

Overall accuracy:0.98

Land cover map 2020	forest	cleared	water	Rocky outcrops	secondary vegetation	Total points	User's accuracy	Error commision
forest	178	0	0	0	1	179	0.97	0.03
cleared	0	216	0	5	8	229	0.86	0.14
water	0	0	61	0	0	61	1	0.00
Rocky outcrops	0	10	0	60	0	70	0.92	0.08
secondary vegetation	4	15	0	0	69	88	0.89	0.11
Total points	182	241	61	65	78	627		
producer's accuracy	0.99	0.94	1.00	0.86	0.78			
Error commision	0.01	0.06	0.00	0.14	0.22			

Kappa: 0.91
Overall accuracy:0.93

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ANEXO A. Guía para autores de la revista “Forest Ecology and Management”

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ANEXO B. Marco Teórico extendido

El marco teórico que fundamenta este trabajo se compone de las teorías y los conceptos de referencia presentados a continuación. Con la intención de presentar una base teórica para el mejor entendimiento de las dinámicas espaciales que se desarrollaron en el trabajo. La síntesis de relaciones entre los conceptos de referencia, se presentan en el diagrama conceptual de la Figura A. 1, que representa el problema de investigación.

Teoría del cambio de cobertura y uso del suelo (LULCC)

Los paisajes altamente dinámicos dan la oportunidad de estudiar el cambio de cobertura y uso del suelo LCLUC por sus siglas en inglés (Wu and Hobbs, 2002), una de las formas más destacadas de cambio global ambiental (Hersperger et al., 2010), ya que permite caracterizar y analizar los cambios multitemporales en el paisaje. En primer lugar, las coberturas terrestres hacen referencia a las características biofísicas de la superficie terrestre, incluyendo la distribución de la vegetación, fuentes de agua, suelo, entre otras (Liping et al., 2018). Las coberturas pueden ser determinadas por el análisis de imágenes satelitales diferenciando diversos tipos y su extensión (NOAA, n.d.).

Por otro lado, el uso del suelo es el proceso de la interacción entre factores biofísicos y socioeconómicos, que resulta en la manera en que los humanos administran su hábitat con fines funcionales vinculados al desarrollo de actividades económicas, este no puede ser visto por imágenes satelitales (Etter, McAlpine, et al., 2006; Liping et al. 2018; NOAA, 2020). En otras palabras, el uso del suelo es el reflejo de las decisiones individuales de las personas y se evidencia en una escala local (Geist and Lambin, 2001).

Los motores de cambio o “drivers” son fuerzas que generan cambios observables en los paisajes, guiando sus diferentes trayectorias (Bürgi et al., 2004). Estas fuerzas impulsoras están inmersas dentro de un sistema complejo de interacciones y dependencias, afectando varios niveles temporales y espaciales (Bürgi et al., 2004; Hersperger et al., 2010).

Ecología del paisaje

La ecología del paisaje es una subdisciplina de la ecología, enfocada en amplias escalas espaciales, estudia cómo la estructura del paisaje afecta la abundancia y la distribución de los organismos, a través del análisis de la influencia de los patrones espaciales sobre los procesos ecológicos (Fahrig, 2005; Turner et al., 1989). Este enfoque teórico se compone por dos dimensiones: espacial descriptiva (cuantitativa) y funcional (cualitativa), abordadas por tres aspectos: la estructura, el funcionamiento y la temporalidad (Etter, 1991).

La estructura y los patrones presuponen heterogeneidad espacial, a través la composición entendida como la cantidad de diferentes entidades posibles y la configuración siendo el arreglo espacial (Fahrig, 2005). Mientras que el funcionamiento y los procesos hacen referencia a la interacción entre los elementos espaciales expresados en flujos de materia y energía entre los ecosistemas (Turner et al., 1989).

La fragmentación es una variable que altera la configuración de las coberturas espaciales (Etter, 1991; Fahrig, 2005). El proceso de fragmentación ocurre con la aparición de parches dentro de una matriz, las características más importantes para analizarlos son el tamaño, la forma, el tipo, la cantidad y la configuración (Forman et Gordon, 1981). La conectividad o el grado de aislamiento de los fragmentos incide en la función ecosistémica de los mismos y en una escala más amplia puede llegar a afectar en diferentes niveles la configuración y composición de la matriz en la que están inmersos (Etter, 1991). Un paisaje se considera en estado fragmentado con menos del 60% de la vegetación nativa restante (Etter et al., 2008).

Para el caso concreto de la deforestación de los bosques tropicales Geist & Lambin (2001, pp. 5–13) plantean tres factores directos de cambio producidos por el uso del suelo en un nivel local: expansión agrícola, extracción de madera y expansión de infraestructura. También señalan cinco categorías de fuerzas conductoras subyacentes (o procesos sociales) que actúan en nivel local y secundariamente desde nivel nacional o global: factores demográficos, económicos, tecnológicos, políticos/ institucionales y culturales o sociopolíticos.

La deforestación es uno de los principales problemas ambientales globales asociado a la destrucción de los bosques (Armenteras et al., 2013). Este proceso ha llevado a la fragmentación de los ecosistemas naturales resultando en mayor pérdida de los bosques y disminución de la conectividad de los fragmentos (Etter et al., 2006b). Puede ser permanente o transitoria, además, tiene consecuencias deletéreas para los nutrientes de los suelos tropicales siendo eliminados de los suelos después de dos años (Rodríguez Becerra, 2019). La deforestación en los trópicos está impulsada por la acción conjunta de factores económicos, demográficos, institucionales, naturales y políticos (Armenteras et al., 2013).

La deforestación se puede abordar desde el concepto de disturbio, ya que este proceso nace de un evento momentáneo o acumulativo que cambia el bosque e inicia nuevas trayectorias sucesionales, además, el disturbio puede ser natural o antropogénico. De acuerdo con Johnson and Miyanishi (2020) el disturbio esta intrínsecamente relacionado con los procesos de sucesión del bosque haciendo posible el crecimiento de vegetación secundaria.

Los procesos de regeneración son la consecuencia de la transición de la sucesión secundaría a un bosque y no se da directamente de un claro de bosque (Etter et al., 2006b). De acuerdo con Soares-Filho et al., (2013), tras la deforestación algunos sitios serán abandonados, dando pie al proceso de regeneración, los factores más importantes para que ocurra son la distancia al bosque y la zoocoria por medio de animales tales como aves y murciélagos (entre otros mamíferos), estos vectores necesitan estar cerca a los bosques, ya que no pueden dispersar semillas a ciertas distancias. El proceso de regeneración puede ser detenido por disturbios como consecuencia de prácticas de mantenimiento de las pasturas e incendios.

Chazdon (2008) plantea que el proceso de sucesión secundaria que genera un bosque secundario se divide en tres fases. La primera etapa o etapa de iniciación ocurre después del disturbio y tarda de 0 a 10 años en el cual empiezan a llegar especies primarias. La segunda etapa va de los 10 a 25 años de crecimiento secundario, en este periodo ocurren los cambios más drásticos de la composición y estructura de las especies vegetales, además se llega a el cierre de los doseles de los árboles. La tercera etapa va de 25 a 200 años donde se complejizan los procesos del bosque secundario, por otro lado Brown y Lugo (1990) plantean que el bosque secundario se consolida desde los 60 a 80 años de crecimiento.

Ecología de bosque húmedo tropical

Los bosques son sistemas complejos que inciden en la integridad del planeta por su alta biodiversidad y gran biomasa que albergan (Potapov et al., 2017). El ecosistema de bosque húmedo tropical se encuentra entre las latitudes 10°N y 10°S, la precipitación anual sobrepasa la suma de agua pérdida mediante la evaporación y la transpiración (Ofosu-Asiedu, 2008). En el caso del bosque amazónico se registra un sistema de lluvias unimodal influenciado por el paso de la Zona de Convergencia Intertropical (ZCIT), una precipitación media multianual de 3.307mm, temperatura media de 25.3 °C, cuenta con una de las cuencas más grandes del mundo, alimentándose de algunos de los ríos que nacen en la cordillera de los Andes (SINCHI, 2007). Otras característica relevante son que el suelo amazónico es mayormente ácido, pobre en nutrientes (Rodríguez Becerra, 2019) y la dispersión de semillas está a cargo de los animales, principalmente aves y mamíferos, influyendo en la regeneración de la vegetación (Soares-Filho et al., 2013).

Figuras de conservación

Las áreas protegidas AP nacen como una apuesta a la conservación de la biodiversidad, al tiempo contribuyen a los medios de vida de las personas, especialmente a nivel local (IUCN, 2008). La red mundial de AP está compuesta por redes nacionales de obras que tienen diferentes historias, incluyendo diversas motivaciones por las cuales se estableció su conservación (Joppa and Pfaff, 2011). La eficacia de la gestión de las áreas protegidas y los territorios indígenas de la Amazonía depende de su contexto y vulnerabilidad (Gullison and Hardner, 2018). Se ha probado que son útiles a la hora de mitigar los procesos de pérdida de bosque en regiones tropicales aunque este no sea un objetivo central, de igual forma las AP están distribuidas de manera no aleatoria, en lugares remotos usualmente alejados de vías de acceso como carreteras o ríos (Clerici et al., 2020).

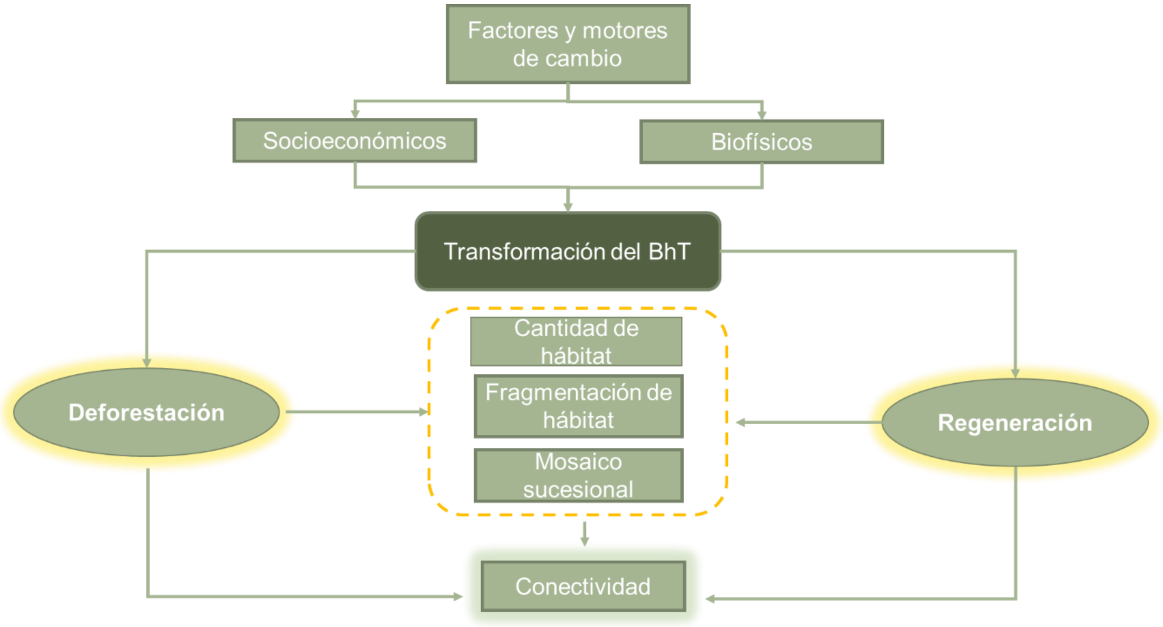

Figura A.1 Diagrama conceptual

Antecedentes

El monitoreo de las dinámicas que transforman los bosques tropicales por medio de sensores remotos (SR) es una forma eficaz y práctica de revelar la correlación entre las acciones antropogénicas y la transformación forestal a lo largo del (Brovelli et al., 2020; Hansen et al., 2010; Senf et al., 2020). Con el fin de avanzar y mejorar el alcance de la conservación de la biodiversidad, la modelización espacial de la conectividad estructural y funcional del paisaje resulta (Bennett, 2003; Crooks and Sanjayan, 2010; Lira et al., 2012; Rudnick et al., 2012). Dado que varios estudios demuestran que los cambios del paisaje pueden dar lugar a una transformación permanente o a un cambio temporal de la cobertura boscosa (véase, por ejemplo: D. Armenteras et al., 2017; Etter et al., 2005; Hamunyela et al., 2020; Nunes et al., 2015) es fundamental valorar el papel de la vegetación secundaria en la configuración del paisaje (Etter et al., 2005; Nunes et al., 2020).

Un estudio realizado Etter et al., (2005) en áreas del bosque húmedo colombiano que habían sufrido una transformación del paisaje, encontró una tendencia creciente en la cobertura de bosque secundario. Del mismo modo, Teixeira et al., (2009) en su estudio de la selva brasileña se documentó una tendencia hacia un bosque secundario más joven en áreas que habían estado sujetas a altas tasas de deforestación y regeneración. Sin embargo, estudios recientes han arrojado nueva luz sobre las diferentes vías que siguen las zonas de vegetación reagrupadas una vez que disminuyen las tasas de deforestación. Es el caso del estudio de Carvalho et al., (2019), cuyos hallazgos reportan que cuando las tasas de deforestación disminuyen, aumenta la transformación de la vegetación secundaria en cubierta terrestre no forestal. Algunos de los usos más comunes de la vegetación secundaria transformada son la ganadería, la agricultura y las plantaciones de aceite de palma (Nunes et al., 2020).

Otro enfoque del monitoreo forestal es la conservación, específicamente en la evaluación de la eficacia de las áreas protegidas (AP), siendo una figura utilizada frecuentemente para reducir la deforestación (Andam et al., 2008). Varios autores (Barber et al., 2014; Cabral et al., 2018) han demostrado que las AP actúan como una barrera contra la pérdida de bosque, pero a menudo se debe a su ubicación remota e inaccesibilidad. Por el contrario, Bray et al. (2008) encontraron en su estudio que las AP no se desempeñan bien en los frentes de colonización activa. En este sentido, Armenteras et al. (2009), un estudio encontró que los bosques tropicales de tierras bajas de Colombia han estado sometidos a presiones crecientes en su entorno durante los últimos 30 años. El mismo estudio destacó la variabilidad en el desempeño entre AP y reservas indígenas (IR), mostrando que AP funcionan mejor.

En la Amazonia colombiana los procesos de deforestación no son aleatorios pero sí espontáneos, impulsados por diversos factores de accesibilidad al bosque como los ríos, además, la destrucción del bosque está asociada a actividades económicas como la ganadería y los cultivos de coca (Armenteras et al., 2006; Etter et al., 2006b). Otro aspecto relevante sobre esta región es que los núcleos de biodiversidad pueden coincidir con los centros de deforestación (Etter et al., 2006b).

Gracias a la colonización, la región noroccidental de la Amazonia presenta dinámicas aceleradas de intervención humana, expresadas en incendios y cicatrices, estratos de intervención, frontera agrícola y rondas hídricas (zonas prioritarias de restauración) (SIAT-AC, 2020). Desde la década de los 2000 se registran conflictos socioambientales alrededor de la tenencia y acaparamiento del territorio, por lo cual en la región es común encontrar la figura de latifundios, haciendo referencia a grandes áreas de tierra bajo la administración de pocas personas (SINCHI, 1999).

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ANEXO C. Metodología extendida

Esta sección presenta la versión extendida de la metodología aplicada en el trabajo.

Área de estudio

El departamento del Guaviare está ubicado en el noroeste de la Amazonia ocupando una superficie de 53. 460 km2 que corresponde al 5% del territorio colombiano, y está dividido en cuatro municipios: Calamar, El Retorno, Miraflores y San José del Guaviare (United Nations Office for Drug Control UNODC, 2014). El área de interés (AOI) de este estudio ocupa 39. 130 km2. Este territorio es conocido como un foco de deforestación, presentando procesos de colonización desde 1970 (Armenteras et al., 2019). Las principales actividades económicas asociadas al desmonte de tierras son la ganadería y los cultivos ilícitos de coca (*Erythroxylum coca*), el segundo ha sido un motor fundamental para la transformación de la cubierta terrestre en la región en las últimas dos décadas (Armenteras et al., 2013, 2019; Etter et al., 2005; Etter, McAlpine, et al., 2006). Esta región comprende un amplio sistema de bosques tropicales, caracterizados por tener sistemas complejos de comunidades de fauna y flora, lo que se refleja en una gran riqueza y abundancia de especies (Meyfroidt and Lambin, 2011; Nunes et al., 2020; Wang et al., 2019). El área de interés comprende una serie de 5 áreas protegidas (1. 510. 104 ha) y 28 resguardos indígenas (465. 888 ha) constituidos con el fin de proteger la biodiversidad.

Guaviare es el departamento amazónico que presenta menor precipitación media multianual con 2.500 mm., presentando una temporada seca (diciembre-marzo) y una de lluvias (abril-noviembre). Este departamento es recorrido por numerosos ríos, quebradas y caños que se dividen en dos grandes cuencas: la del Orinoco y la del Amazonas (United Nations Office for Drug Control UNODC, 2014).La temperatura media en la región es de 25.3 °C y la humedad varía entre el 80% y 90% a lo largo del año (SIAT-AC, 2020). Los suelos se caracterizan por ser muy pobres en nutrientes y altamente susceptibles al deterioro, su capa orgánica es delgada, compuesta por hojarasca en distintos grados de descomposición (SIAT-AC, 2020).

Hace parte de la reserva forestal de la Amazonia (Ley 2 de 1959), del área de protección regional Ariari-Guayabero y de la Zona de Reserva Campesina del Guaviare (ZRCG) (INCODER & COOAGROGUAVIARE, 2012). En el Registro Único de Áreas Protegidas (RUNAP, n.d.) se encuentran inscritas nueve áreas protegidas ocupando 1,958,868.53 ha del departamento, las más extensas son: el Parque Nacional Natural Serranía de Chiribiquete con 1,063,019 ha, la Reserva Nacional Natural Nukak (I Reserva natural estricta) 875,650.56 ha, la Reserva Forestal Protectora Nacional Serranía La Lindosa-Angosturas II (VI Área protegida con recursos administrados) 28,224.00 ha. Adicionalmente, el departamento comprende cerca de 30 asentamientos indígenas con diferentes culturas (Sinic, n.d.)

Preprocesamiento de imágenes satelitales

El preprocesamiento de los datos espaciales fue hecho en Google Earth Engine GEE, se eligió esta plataforma debido a que es de libre acceso y online (Gorelick et al., 2017), además, se conecta con el servicio de nube de Google facilitando el almacenamiento de los productos. Por otro lado, es una herramienta de procesamiento muy rápida.

Se utilizaron imágenes satelitales para elaborar mapas coberturas de los años 2000, 2015, 2017 y 2020, respectivamente. Estos años se eligieron de acuerdo con la calidad de las imágenes y el porcentaje de nubes (menor que 20%). Entre 2000 y 2015 no se encontraron imágenes que cumplieran con los criterios propuestos y que a su vez cubrieran toda el área de interés, lo que explica la diferencia de 15 años entre el primer y el segundo año analizado. Las imágenes satelitales utilizadas en este estudio se obtuvieron vía GEE (Gorelick et al., 2017; Hurni et al., 2017), específicamente utilizando Landsat 5 Thematic Mapper (TM) para el año 2015, Landsat 7 Thematic Mapper Plus (ETM+) para el año 2000 y Landsat 8 Operational Land Imager (OLI) para 2017 y 2020. Se utilizó la colección de reflectancia de superficie Tier 1, que tiene imágenes del fondo de la atmosfera (BOA por sus siglas en inglés) con datos corregidos que tienen radiometría bien caracterizada y están Inter calibrados entre los diferentes instrumentos del Landsat. El geo-registro de escenas del nivel 1 es coherente y se encuentra dentro de las tolerancias prescritas de imagen a imagen de ≦ 12-error cuadrático medio (Gorelick et al., 2017; Young et al., 2017). Consecuentemente, la corrección geométrica se descarta para la detección de cambios, en la medida en que se utilicen solamente imágenes L1T de Landsat, por lo que se consideran las más adecuadas para el análisis multitemporal (Hamunyela et al., 2020; Zhu, 2017).

Para cada año se generó un mosaico filtrado por el área de interés utilizando el valor de la mediana de los píxeles de cada imagen para crear un compuesto homogéneo, se procesaron 54 imágenes en total, 10 para el año 2000, 8 para el año 2015, 27 para el año 2017 y 9 para el año 2020. Igualmente, se aplicó una función de máscara de nubes utilizando los atributos de calidad de píxeles basados en el algoritmo CFMASK (Hamunyela et al., 2020).

Con el fin de analizar las dinámicas de deforestación y regeneración a lo largo de los años, se crearon mapas de coberturas para cada año, según lo propuesto por Etter, Mcalpine, et al.,(2006). Se realizó una clasificación supervisada mediante interpretación visual de las imágenes satelitales (Jensen, 2015; Richards and Jia, 2006) utilizando el algoritmo Random Forest RF en GEE (Brovelli et al., 2020). Este algoritmo es ideal para clasificar más de dos clases a diferencia del clasificador CART por ejemplo, que se desempeña mejor en clasificaciones binarias (Ang et al., 2021). RF es ideal para este trabajo debido a su sistema basado en árboles de decisión que acelera y afina el proceso de selección de pixeles (Wang et al., 2019), además, minimiza la redundancia mediante la selección efectiva de propiedades espectrales relevantes de una clase al clasificar características de interés (Ang et al., 2021).

El clasificador se aplicó a cinco amplias clases de coberturas: bosque, vegetación secundaria, tierra despejada, cuerpos de agua y afloramientos rocosos. Los sitios de entrenamiento se seleccionaron mediante muestreo aleatorio (Hamunyela et al., 2020; Jensen, 2015; Olofsson et al., 2014), se seleccionaron más de 1. 000 píxeles para cada fecha. Para probar el modelo, se realizó una evaluación de precisión creando nuevos sitios de validación, obteniendo matriz de error, precisión global, precisión del productor, precisión del usuario y coeficiente kappa. Los errores de clasificación pueden atribuirse a la confusión espectral entre bosques regenerados y cultivos (Prates-Clark et al., 2009) explicando por qué la clase de vegetación secundaria obtuvo la menor precisión de clasificación. Después de la clasificación, se aplicó un filtro mayoritario por medio de una ventana móvil 3X3 para eliminar los píxeles aislados y reducir el ruido de los mapas de cobertura (Jensen, 2015; Wang et al., 2019). Todos los mapas fueron proyectados bajo el mismo sistema de proyección, fijados a la extensión del departamento del Guaviare con la resolución de 30m en ArcMap 10.7.1 (Esri, 2008).

Modelamiento espacial

Para determinar la pérdida y regeneración de bosque durante las dos décadas estudiadas, se elaboraron mapas de bosque-no-bosque (1,0) para cada año. Obteniendo mapas de cambio (Figura A.2.) para los períodos 2000-2015, 2015-2017 y 2017-2020, posteriormente fueron reclasificados para identificar la deforestación y la vegetación regenerada durante cada periodo. Estos datos también se calcularon dentro de 5 áreas protegidas y 28 reservas (Armenteras et al., 2009; Nepstad et al., 2006; Soares-Filho et al., 2010),midiendo el impacto de las dinámicas del paisaje dentro y fuera de estos límites. La información sobre áreas protegidas y resguardos indígenas se obtuvo del Instituto Geográfico Colombiano Agustín Codazzi (IGAC, 2020). Se cuantificaron las tasas de cambio para cada figura de conservación con el fin de comparar la eficacia de cada figura individualmente, al igual que determinar su efecto conjunto.

Las tasas anuales de cambio de las coberturas se calcularon según la fórmula propuesta por Puyravaud (2003):

$$r = \left(\frac{1}{t2-t1}\right) X \ln \left(\frac{A2}{A1}\right),$$

Donde r equivale a la tasa de cambio por clase para cada año, t1 y t2 son el tiempo inicial y el último, y A1 y A2 son las áreas de clase para el inicio y el final del período.

Se creó un mapa de edades del bosque para el año 2020 en ArcMap 10. 7. 1 (Esri, 2008), con el fin de identificar las edades de regeneración de la vegetación dentro del mosaico de bosque secundario. Este mapa se basó en los mapas de bosque-no-bosque obtenidos para las cuatro fechas estudiadas, centrándose en los cambios (presencia vs ausencia) de píxeles de vegetación secundaria desde cada período hasta el año 2020. Adicionalmente, se reclasificaron las clases de edad como propusieron Etter et al., (2005), superponiendo los mapas de bosque-no-bosque de los cuatro años estudiados. Los píxeles de vegetación secundaria que persistieron de los años 2000 a 2020 se tomaron como 20 años de regeneración, el mismo cálculo se hizo para 2015-2020 (5 años de VS) y 2017-2020 (3 años de VS). El bosque maduro corresponde a los píxeles forestales que persistieron intactos a lo largo del período estudiado.

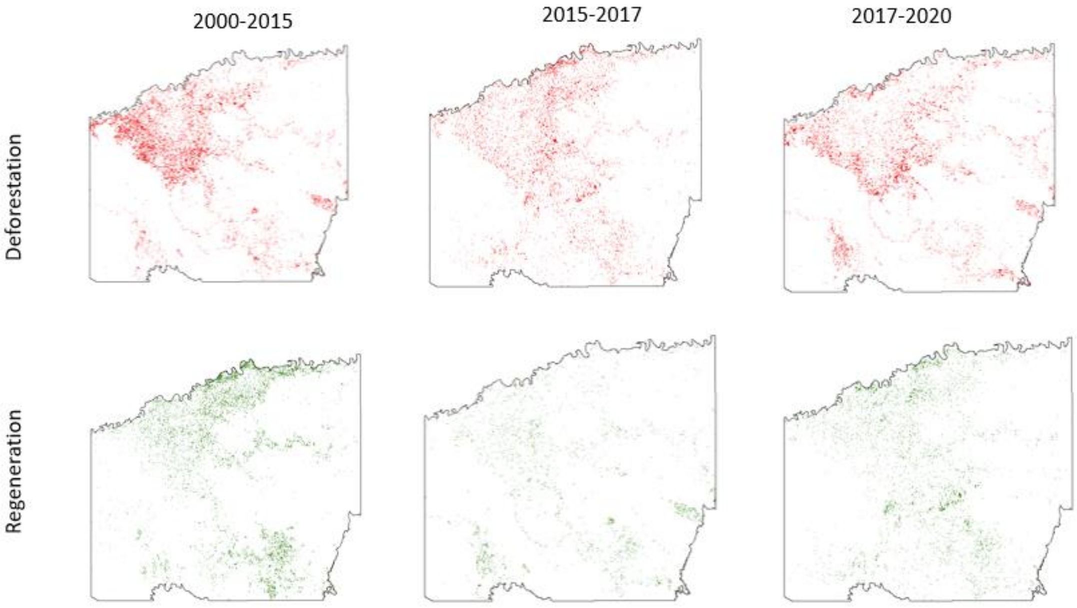

Figura A.2. Mapas deforestación y regeneración

Conectividad

Se implementaron métricas de paisaje adaptadas de (Etter et al., 2005; Zhang et al., 2020) para entender el efecto del bosque secundario sobre la conectividad de los mosaicos de bosques remanentes, se aplicaron profundidad fija de 100m y umbral de conexión de 100m utilizando FRAGSTATS 4. 2 (McGarigal et al., 2012). Los valores de 100m fueron elegidos bajo criterio experto, teniendo en cuenta que se busca evaluar la conectividad entre clases del paisaje. Las métricas elegidas para este estudio fueron: número de parches (NP), Tamaño medio del parche (MPS), área total del núcleo (TCA) e índice de conectancia (CONNECT), su utilidad para analizar dinámicas de coberturas terrestres ha sido reiterada (Spanowicz and Jaeger, 2019). Todas las métricas propuestas se calcularon para las diferentes edades del bosque secundario, mientras que TCA y CONNECT se calcularon para los tres periodos de transición con y sin presencia de VS.

El efecto de los bosques secundarios sobre la conectividad y área núcleo total de la cobertura boscosa se midió en primer lugar para los años 2015, 2017 y 2020 individualmente, y en segundo lugar se calculo su efecto acumulativo hasta el año 2020, entendiendo el efecto acumulativo como los pixeles que eran no-bosque en el año 2000 y pasaron a ser bosque del 2015 en adelante. En este sentido se calcularon inicialmente las métricas TCA y CONNECT para los mapas binarios compuestos por bosque (bosque maduro y bosque secundario) y no bosque (cuerpos de agua, afloramientos rocosos, áreas despejadas), luego se hizo el mismo calculo para mapas binarios de los mismos años con la reclasificación de las categorías de bosque (sólo bosque maduro) y no bosque (vegetación secundaria, cuerpos de agua, afloramientos rocosos, áreas despejadas). Una limitación de FRAGSTATS es que al no utilizar el 100% del potencial de la memoria del computador los cálculos pueden tardar mucho tiempo, más de una hora por mapa, para el caso de este trabajo.

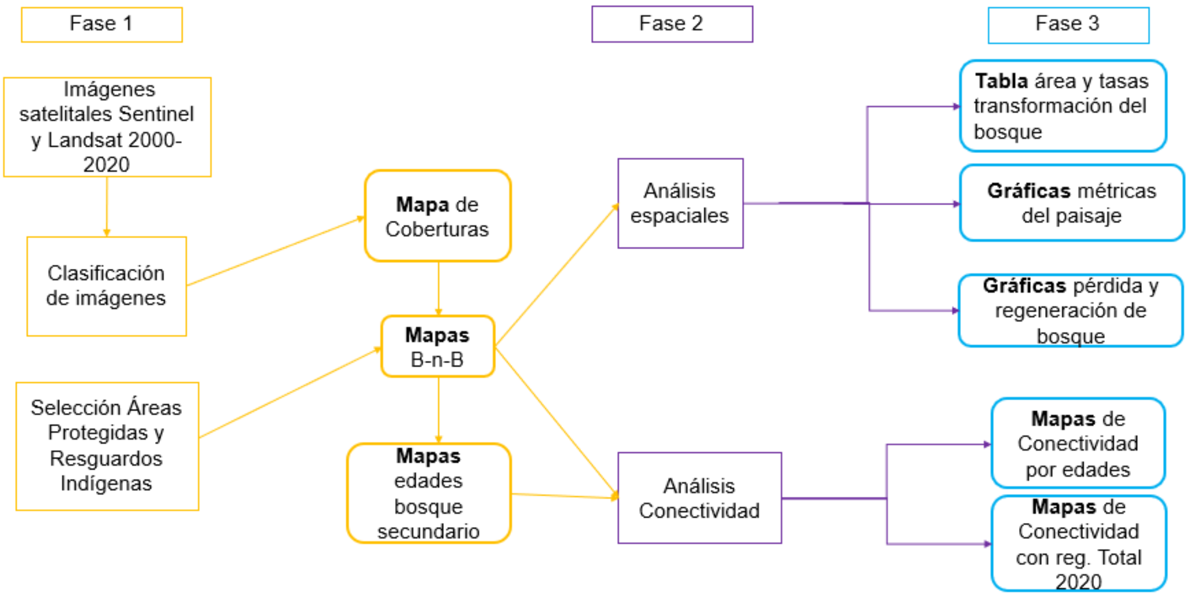

Figura A.3. Diagrama metodológico

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