

STATE OF THE LANDSCAPE AND DYNAMICS OF LOSS AND FRAGMENTATION OF FOREST CRITICALLY ENDANGERED IN THE TROPICAL ANDES HOTSPOT: IMPLICATIONS FOR CONSERVATION PLANNING

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ABSTRACT

Currently, there is no precise information on the degree of transformation of Tropical Andes hotspot landscape and native ecosystems due to the intensification of agricultural and urban land-use. Proper knowledge of these changes would provide crucial information for planning conservation strategies. We evaluated the impact of the intensification of agricultural and urban land-use on the Inter-Andean Dry Forest and Tropical Montane Forest, both of which are categorized as Critically Endangered, and the state of the landscape in the High Rio Guayllabamba watershed, Ecuador, during the periods 1991–2005 and 2005–2017. The evaluation was carried out using Landsat satellite images of 30 x 30 m pixels and landscape metrics. We found an advanced state of landscape transformation. Since the 1990s, the loss of both ecosystems was largely caused by the conversion of forest to agriculture, resulting in substantial changes in the spatial configuration of these ecosystems. From 1991 to 2017, 19.8 % and 16.1 % of Inter-Andean Dry Forest and Tropical Montane Forest respectively, were converted to agriculture. The loss of Inter-Andean Dry Forest was 28 % and the number of forest patches increased by more than 150%. The loss of Tropical Montane Forest was 16.5 % and the number of forest patches increased by more than 300 %. The largest loss and fragmentation of forest cover occurred from 1991 to 2005. We suggested planning landscape-scale conservation, using the patch-corridor-matrix model. This model is appropriate given the current configuration of the landscape studied, with Inter-Andean Dry Forest and Tropical Montane Forest restricted to small patches sparsely distributed across the landscape.

Keywords: Changing landscape; Ecuador; Forest habitat; Forest landscape; Landscape-scale conservation; Rio Guayllabamba watershed.

INTRODUCTION

The intensification of agricultural and urban land-use can lead to habitat loss and fragmentation, and these are two of the greatest threats to forest landscapes (Tapia-Armijos *et al.*, 2015; Ferrer-Paris *et al.*, 2018; Rodríguez-Echeverry *et al.*, 2018a). Agricultural and urban land-use modifies the spatial patterns (composition and spatial configuration) of forest landscapes, such as the total area of forest, patch number, and connectivity (Echeverría *et al.*, 2014; Lindenmayer, 2016). Habitat loss and fragmentation may have negative effects on

biodiversity, by increasing isolation of habitats, endangering species, and modifying species' population dynamics (Rodríguez-Echeverry *et al.*, 2015). Such alteration could lead to local and global extinction of the most vulnerable species (Bennett, 2003; Lindenmayer & Fischer, 2006).

The altered forested landscapes can be a region with a low rate of deforestation and good connectivity (Echeverría *et al.*, 2012) to highly fragmented and modified landscapes. With the progressive forest loss/fragmentation the forest cover represents a reduced area consisting of scattered and isolated habitat patches, with no connectivity (Rodríguez-Echeverry *et al.*, 2018a). Therefore, the evaluation of forest landscape changes allows a better understanding of their degree of integrity, functionality, and landscape characteristics (composition and spatial configuration) (Rodríguez-Echeverry *et al.*, 2017). This evaluation is essential in landscapes with high species diversity and endemism combined with high forest habitat loss (Lindenmayer & Franklin, 2002; Rodríguez-Echeverry *et al.*, 2015).

The biodiversity hotspots are regions with a global priority for conservation due to their high degree of endemism and strong anthropic impact (Myers *et al.*, 2000). These regions have at least 1,500 species of endemic plants and have lost at least 70 % of their original extent (Mittermeier *et al.*, 2011). The Tropical Andes is a hotspot characterized by being the most diverse for holding the highest plant endemism in the world. The Tropical Andes hosts a total of 30,000 species of native plants, of which 50 % are endemic (NatureServe & EcoDecision, 2015). The Tropical Dry Forest and Tropical Montane Forest ecosystem, both of which are categorized as Critically Endangered by The International Union for Conservation of Nature (IUCN) (Ferrer-Paris *et al.*, 2018), are a part of the Tropical Andes hotspot. In Ecuador the Tropical Dry Forest is restricted to the lowlands regions across the inter-Andean valleys and has been described as an Inter-Andean Dry Forest (Aguirre *et al.*, 2006), while the Tropical Montane Forest is restricted to the highland regions and is characterized by evergreen forests, frequently enveloped in clouds and mist (IUCN, 2000).

The Inter-Andean Dry Forest and Tropical Montane Forest are in the High Rio Guayllabamba watershed (HRGW), in northern Ecuador. Despite the high value for the biodiversity conservation of this watershed, a number of anthropogenic processes have been associated with the degradation and loss of Inter-Andean Dry Forest and Tropical Montane Forest, such as intensive agriculture, extraction of firewood, and overexploitation of native species (Aguirre *et al.*, 2006, 2011; Mittermeier *et al.*, 2011; NatureServe & EcoDecision, 2015). The foregoing impacts on the native forest have been the focus of discussion among environmental entities of government and communities, which have highlighted the need to implement effective conservation strategies for native forest ecosystems (Sarmiento, 1995; Aguirre *et al.*, 2011; Ministerio del Ambiente del Ecuador, 2013; NatureServe & EcoDecision, 2015).

Although there are few specific studies on the diversity of flora and fauna and their state of conservation in the Inter-Andean Dry Forest and Tropical Montane Forest in the HRGW (Sarmiento, 1995; Aguirre *et al.*, 2006, 2011; Cadena-Ortiz *et al.*, 2015; Cisneros-Heredia *et al.*, 2017), there are no spatio-temporal studies on the transformation state of this landscape, and in particular on the changes in spatial patterns of the native forest ecosystems due to the intensification of agricultural and urban land-use. This type of information is necessary to implement actions or strategies of conservation and restoration at the landscape scale, which offers a way out to the degradation of the ecosystems while ensuring the participation of different levels of governance and the community in this process (Ministerio del Ambiente del Ecuador, 2013).

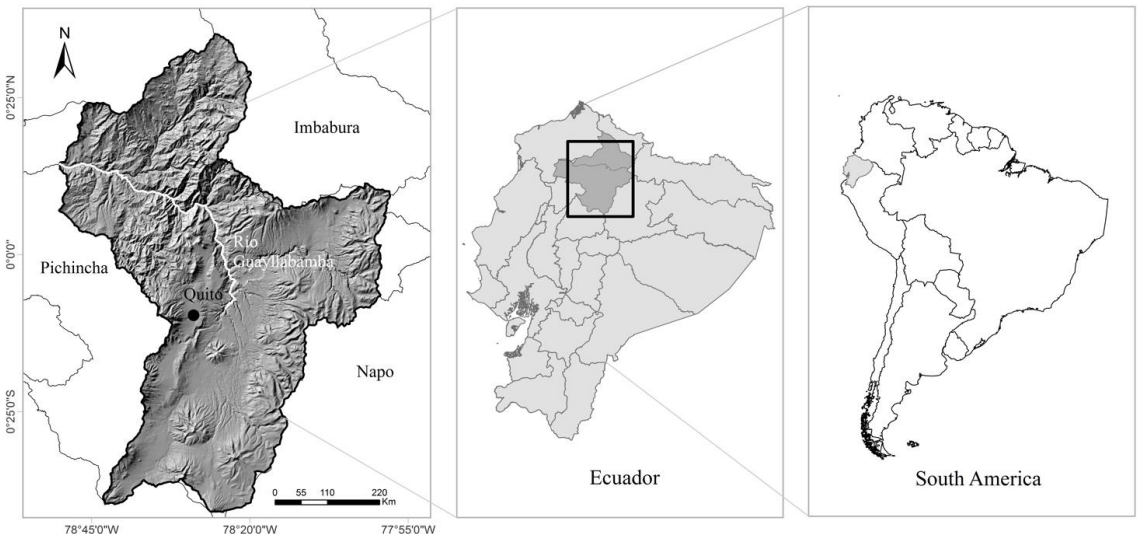
We assessed the impact of the intensification of agricultural and urban land-use on the spatial pattern of the Inter-Andean Dry Forest and Tropical Montane Forest and the state of the landscape in the HRGW by 1) determining deforestation rates during the periods 1991–2005 and 2005–2017, 2) evaluating the changes in the spatial patterns of forest cover over time and 3) determining the state of transformation of the landscape. We hypothesize that there has been a substantial loss of Inter-Andean Dry Forest and Tropical Montane Forest due to an increase in agriculture and urban areas, resulting in an advanced state of a transformed landscape. This study is the first of its kind undertaken to understand the spatio-temporal changes in the Inter-Andean Dry Forest and Tropical Montane Forest, which can contribute to the conservation planning of these threatened ecosystems.

MATERIAL AND METHODS

Study area

The study area is the HRGW located in the cordillera of the Andes ($0^{\circ}13'N$ and $0^{\circ}40'S$) in the provinces of Pichincha and Imbabura, Ecuador (Fig. 1). The watershed occupies 6,758 Km^2 , has a mean temperature of $14^{\circ}C$ and an annual rainfall of 1,200 mm. The HRGW is a topographically diverse area where the elevation ranges from 1,150 to 5,897 m a.s.l. (Instituto Nacional de Meteorología e Hidrología, 2005). Currently, the total area of watershed consists of agriculture 43 %, Montane Forest 19 %, Paramo 17 %, shrub vegetation 10 %, Inter-Andean Dry Forest 5 %, urban areas 4 % and other land-uses types (water bodies, bare land, and snow) 2 %. The watershed has a human population of approximately 3,228,233 inhabitants, with a population density of 591 people km^{-2} . The main economic activity in the watershed is agriculture (Sistema Nacional de Información, 2019).

Fig. 1: Location of the HRGW in the provinces of Pichincha and Imbabura, Ecuador



The Inter-Andean Dry Forest ecosystem, located in the study area, occupies 33,609 ha and covering from 1,400 to 2,500 m a.s.l. (Ministerio del Ambiente del Ecuador, 2013). This ecosystem is characterized by the lowland forests, with open canopies and shrubs up to 2 m growing on dry mountain slopes. The vegetation composed of *Acacia macracantha*, *Caesalpinia spinosa*, *Opuntia cylindrica*, *Opuntia soederstromiana*, *Croton wagneri* and *Dalea hufusa* (Aguirre *et al.*, 2011). The fauna is represented by *Stenocercus guentheri* and the *Lampropeltis triangulum micropholis* reptiles, the *Camptostoma obsoletum* and *Catamenia analis* birds, and the *Lycalopex culpaeus* mammal (MECN, 2009). The Tropical Montane Forest occupies 130,557 ha and covering from 3,000 to 3,400 m a.s.l. (Ministerio del Ambiente del Ecuador, 2013). In this ecosystem, tree trunks, branches, and rock surfaces are covered with moss, ferns, orchids, and other epiphytic plants (IUCN, 2000). The vegetation comprises of *Clusia alata*, *Blackea quadriflora*, *Meriania tomentosa*, *Meriania maxima*, and *Cecropia maxima*. The fauna is represented by *Anoura geoffroyi* bat, the *Riama colomaroni* lizard, the *Diglossa lafresnayii* bird, the *Prisitmantis eugeniae* amphibian, and the *Thomasomys aureus* rodent (MECN, 2009).

Land-cover classification

Landsat satellite images were used to generate the land-use thematic maps for 1991 (TM), 2005 (ETM), and for 2017 (ETM) (< 5 % cloud cover). These images were acquired for the months of July 1991 and 2005, and October 2017, from the USGS portal. Due to a prevailing cloud cover in the Andes range, it was impossible to obtain images with cloudiness < 10 % for the same month in the three years of study. To avoid major differences in phenology, all images acquired were taken during the dry season (image dates from June to November). As fluctuations in precipitation are relevant for the spectral response of biomass, all selected images represent similar drought conditions. The images used had a spatial resolution of 30 x 30 m pixels, which is a resolution appropriate for the spatial analysis (Daly, 2006). The images were corrected geometrically, atmospherically, and topographically (Chander *et al.*, 2009). The geometric correction was performed using the “full processing” module in PCI Geomatics and ENVI. This consisted of the transformation of each image using GCPs (ground control points) and a 2nd order polynomial mathematical model. The satellite images were georeferenced separately by locating approximately 120 GCPs in each image and producing corresponding reference maps. The geometric accuracy ranged from 0.11 to 0.29 pixels, corresponding to 3.3 - 8.7 m. Atmospheric correction was applied to all of the scenes, transforming the original radiance image to a reflectance image. The topographic correction was made using the IDRISI Andes hillshade control procedure (IDRISI, 2012) with a Digital elevation model (30 x 30 m) to remove differences in solar illumination influenced by relief. Through the C factor method, the topographic shadows were eliminated (Reese & Olsson, 2011). The Normalized Difference Vegetation Index (NDVI), Simple Ratio Index (SR), Soil Adjusted Vegetation Index (SAVI), and Land Surface Water Index (LSWI) were used to increase the accuracy of the classification (Huete, 1988). Supervised classification was carried out using the statistical method of maximum likelihood estimation and 950 training points taken in the field, which represented the patterns of the types of land cover. The accuracy in the classification of the images was calculated using confusion matrices from 540 validation points taken in the field. For the accuracy of 1991 and 2005 images, land cover maps generated by previous studies were used (Sistema Nacional de Información, 2019). For the three thematic maps, the following categories were identified: Inter-Andean Dry Forest, Tropical Montane Forest, Paramo, shrub vegetation, agriculture, urban areas, bare land, snow, and water bodies.

Analysis of changes in the spatial patterns

We assessed the impact of land-use change and intensification of agricultural and urban land-use on the spatial pattern of Inter-Andean Dry Forest and Tropical Montane Forest by determining the changes of area, patch edge, core area, aggregation, shape, contrast, and proximity metrics (Table 1).

Table 1: Landscape metrics used for the analysis of spatial patterns of the Inter-Andean Dry Forest and Tropical Montane Forest in the HRGW (McGarigal *et al.*, 2013)

Metrics	Description	Unit of measurement	Variation Rate
Total area	It is the sum of all patches' areas of the same forest type	Hectares	CA > 0, unlimited.
Patch area	Area of each patch of native forest in the landscape	Hectares	0 and unlimited
Metric of the largest patch	Percentage of the area of the largest native forest patch with respect to the total landscape area	Percentage	$0 < LPI \leq 100$
Total edge length	It is equal to the sum of the lengths (km) of all edge segments that involve the corresponding patch type	Kilometers	TE ≥ 0, unlimited.
Average Core Area	The core area of the native forest, specified by the depth of edge according to the proximity of the patch to other land uses	Hectares	CORE ≥ 0 and unlimited
Number of Patches	Number of patches in the landscape	None	NP ≥ 1, unlimited.
Average perimeter-area ratio	Relationship between the perimeter and the area	None	PARA > 0 and unlimited
Average edge contrast weighted by area	Degree of the contrast of the forest with respect to its density. The contrast represents the magnitude of the difference between the forest and other land uses	Percentage	$0 \leq ECON \leq 100$
Average distance to the nearest neighbor	The proximity of native forest patches, based on the weighted average distance	Meters	ENN > 0 and unlimited
Aggregation Index	The number of like adjacencies involving the native forest	Percentage	$0 \leq AI \leq 100$

The choice of the metrics used was based on the review of studies on landscape metrics that were representative of essential components of the landscape structure, such as the quality of the patches, the edge of the patch, the context of the patch in the landscape, and patch connectivity (Peng *et al.*, 2010; Rodríguez-Echeverry *et al.*, 2018b). The analysis of these metrics was carried out using FRAGSTATS (McGarigal *et al.*, 2013) and the Spatial Analyst extension for ArcGIS 10.4 (ESRI, 2016).

To calculate the edge contrast metric, contrast weights were assigned between the patches edges of the Inter-Andean Dry Forest, Tropical Montane Forest, and other types of land-use in the study area. The contrast weight of the edges was determined based on the variables of composition and structure of the vegetation. These variables were measured in 30 sampling plots of 20 × 10 m, which were established within forest patches. To calculate the core area, perpendicular distances from the edge to the center of the patch were used. These distances corresponded to the edge effect area between the forest and other types of land-use. The distances assigned were based on those reported in the works carried out by Laurance *et al.* (2002), López-Barrera (2004), and Lindenmayer & Fischer (2006), considering that the degree of contrast between habitats (soft or abrupt edges) expresses the magnitude and distance of primary and secondary responses of the structure, and the habitat's composition and processes before the edge effects.

The analysis of the transition from Inter-Andean Dry Forest cover and Tropical Montane Forest cover to other land-uses was carried out using Land Change Modeler for Ecological Sustainability extension of IDRISI Andes (IDRISI, 2012). The annual rate of change for forests, agriculture, and urban areas was analyzed using the following formula:

$$P = \left[\left(\frac{A_2}{A_1} \right)^{\left(\frac{1}{t_2 - t_1} \right)} - 1 \right] \times 100$$

A_1 and A_2 are the class areas at t_1 and t_2 time, respectively; and P is the percentage of change per year (Newton, 2007).

We identified land-use change hotspots using the maps of 1991 and 2017. Considering the surface and number of patches of forest, and the surface of other land-use types, we identified the deforestation and fragmentation hotspots that are the areas with the highest changes from forest to other land-use.

Analysis of state of the landscape

The current state of alteration of the landscape was analyzed based on the landscape change models proposed by McIntyre & Hobbs (1999) and Echeverría *et al.* (2012). McIntyre and Hobbs model, which is based on loss and fragmentation of forest habitat, and percentage of remaining forest habitat, proposes four landscape states: intact (> 90 % of remaining habitat), variegated (60-90 %), fragmented (10-60 %), and relictual (< 10 %). A functional distinction between variegated and fragmented landscapes is supported by the percolation theory that indicates that organisms are operationally unfragmented when there is > 60 % habitat retention. Between 10 % and 60 % retention (fragmented landscapes), organisms of medium and low mobility may be highly affected by fragmentation. Below 10 % retention (relictual), organisms may be severely affected by fragmentation; in this state, there appears to be an important difference in the composition of flora and fauna. The Echeverría *et al.* model proposes three phases of landscape alteration based on dominant

human processes of the landscape and changes in the matrix over the entire study period. In Phase I, the main transformation process corresponds to forest degradation. The second transformation process is the clearcutting of forests to enable farming or cattle raising. In this phase, the landscape is dominated by degraded forest cover. In Phase II, the clearance for agriculture creates a landscape dominated by disturbed forest and shrubland. In Phase III, the landscape is mostly affected by the habilitation of wooded land rather than by degradation and is dominated by agricultural land or livestock.

RESULTS

Accuracy of classification

For the validation of the 2017 land cover map, we used a confusion matrix (Table 2). According to Foody (2002), the overall accuracy shows the percentage of cases correctly allocated. Our results showed an overall accuracy of 93 %, which means that the 2017 land cover classification had a high performance in discriminating the pixels with natural cover from those with other covers (non-natural covers). The areas classified as natural cover and other covers were accurate to 95.2 % and 90.7 %, respectively.

Table 2: Confusion matrix obtained from the accuracy assessment of the 2017 land cover map of the HRGW

CLASSIFIED	REFERENCE			User's accuracy	Commission error
	Natural Cover	Other Covers	Total		
Natural Cover	258	13	271	95.20	4.80
Other Covers	25	244	269	90.70	9.30
Total	283	257	540		
Producer's accuracy	91.17	94.94			
Omission error	8.83	5.06			
Overall accuracy	0.93				

Change in agricultural and urban land use

In the study landscape, there was an area increase of 31.5 % of agriculture from 1991 to 2017 (i.e. at annual rates of 1.07 %). The largest area increase (29.4 %) occurred from 1991 to 2005, expanded at annual rates of 1.9 % (Fig. 2). There was an area increase of 119.1 % of urban areas from 1991 to 2017 (i.e. at annual rates of 3.06%). The largest area increase (51.42 %) occurred from 2005 to 2017, expanded at annual rates of 3.5 % (Fig. 2).

Deforestation and fragmentation patterns

There was a loss of 28 % of Inter-Andean Dry Forest in the study landscape from 1991 to 2017 (i.e. a mean annual loss of 1.3 %) (Fig. 2). The largest loss (19 %) occurred from 1991 to 2005 (i.e. a mean annual loss of 1.53 %). The number of patches of Inter-Andean Dry Forest increased by 300 % from 1991 to 2005 but decreased by 42 % in 2017 (Table 3 and Fig. 2). There was a loss of 16.5 % of Tropical Montane Forest from 1991 to 2017 (i.e. a mean annual loss of 0.7 %) (Fig. 2). The largest loss (11.5 %) occurred from 1991 to 2005 (i.e. a mean annual loss of 0.9 %). The number of patches of Tropical Montane Forest

increased by 665 % from 1991 to 2005 but decreased by 27 % in 2017 (Table 3 and Fig. 2). From 1991 to 2017 the major contributions to the net change were the conversion of Inter-Andean Dry Forest to agriculture (19.8 %) and Tropical Montane Forest to agriculture (16.1 %) (Fig. 3). The minor contributions to the net change were the conversion Inter-Andean Dry Forest and Tropical Montane Forest to urban areas, 9 % and 0.1 % respectively (Fig. 3). Inter-Andean Dry Forest and Tropical Montane Forest registered a similar trend of deforestation and fragmentation over the entire study period. In 2017 the hotspots of deforestation and fragmentation were located in the north and southeast parts of the Inter-Andean Dry Forest; and in the north, center, and west parts of the Tropical Montane Forest (Fig. 2). Agriculture was the dominant cover in the landscape from 1991 to 2017.

Fig. 2: Spatio-temporal variation and percentage of the area occupied by the types of land cover in 1991, 2005, and 2017 in the HRGW

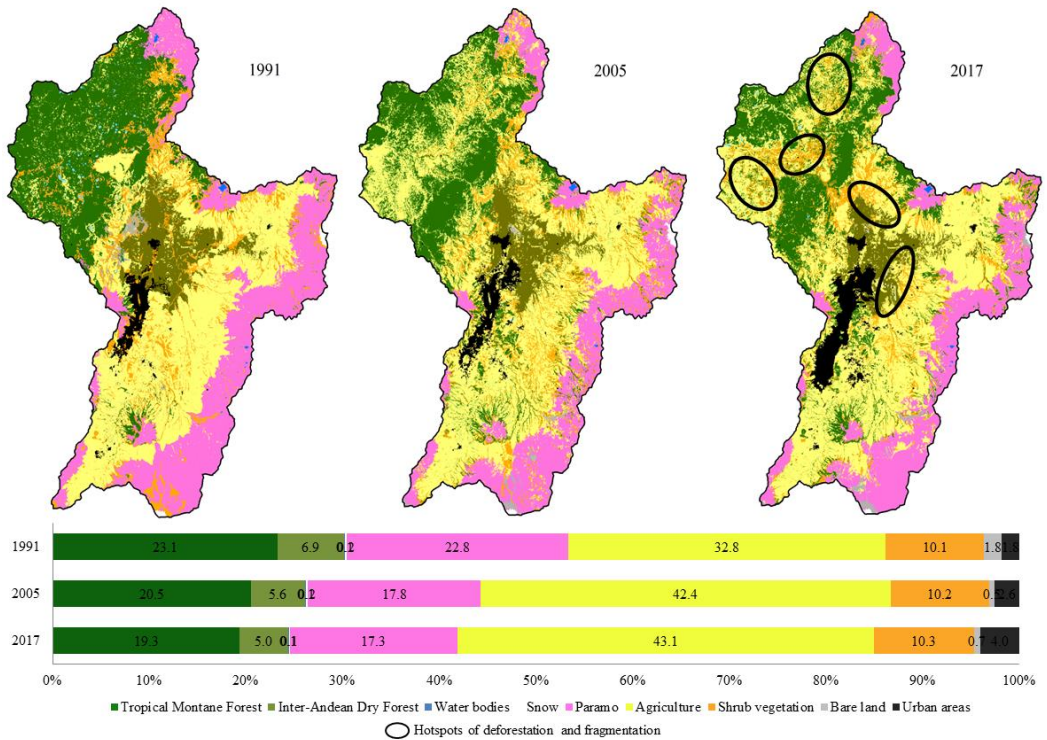
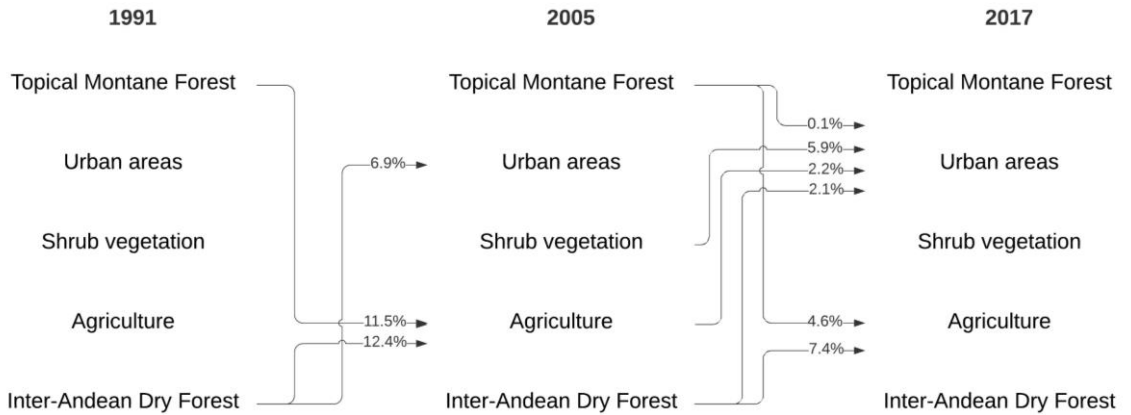


Fig. 3: Major trajectories of land cover change and their contributions to the net change in percentage of the total area of the respective land cover types**Table 3: Changes in landscape pattern metrics for the Inter-Andean Dry Forest and Tropical Montane Forest in the HRGW between 1991 and 2017**

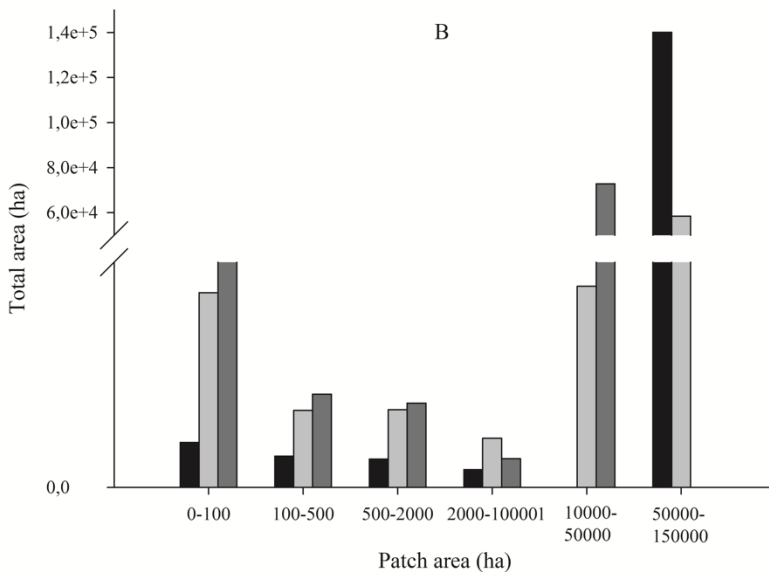
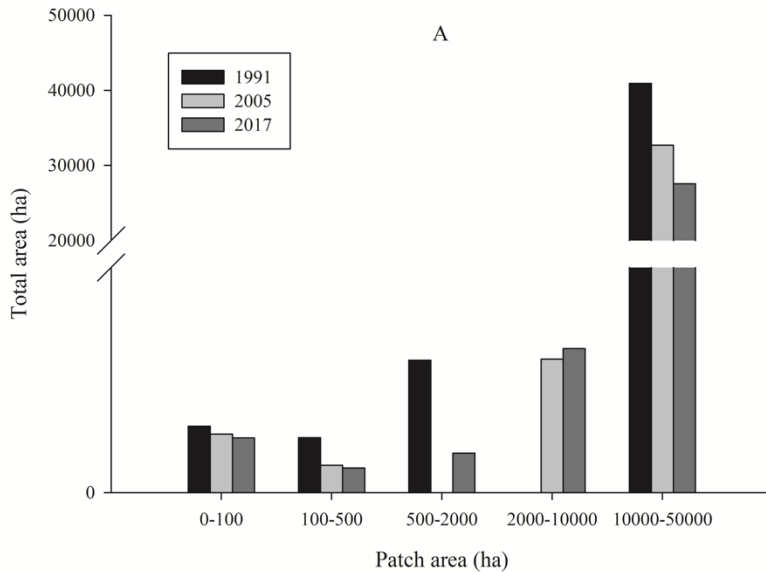
Landscape metrics	Inter-Andean Dry Forest			Tropical Montane Forest		
	1991	2005	2017	1991	2005	2017
Metric of the largest patch (%)	6.05	4.84	4.10	20.73	8.66	4.40
Total Edge (km)	2,742	4,379	3,370	8,331	22,774	21,896
Average Core Area (ha)	0.79	0.72	0.69	0.84	0.67	0.55
Number of patches	793	2,377	1,379	3,981	26,487	19,348
Average distance to the nearest neighbor (m)	85	203	241	100	163	168
Average perimeter-area ratio	888	1,207	1,070	917	1,072	1,004
Average edge contrast weighted by area (%)	40.21	73.25	72.80	42.75	75.41	74.25
Aggregation index (%)	95.70	92.61	91.90	96	87.60	87.49

Changes in the spatial patterns

In the HRGW considerable changes were found in the distribution of forest patch size from 1991 to 2017 (Fig. 4). In 1991, 88 % of the Inter-Andean Dry Forest area was concentrated in patches between 10,000 and 50,000 ha; 5 % in patches between 500 and 2,000 ha; and 7 % of the remaining forest area occurred in isolated patches of less than 500 ha (Fig. 4). In 2005, 86 % of the Inter-Andean Dry Forest area was concentrated in patches between 10,000 and 50,000 ha, 8 % in patches between 2,000 and 10,000 ha; and 6 % of the remaining forest area occurred in isolated patches of less than 500 ha. In 2017, 82 % of the Inter-Andean Dry

Forest area occurred in patches between 10,000 and 50,000 ha; 10 % in patches between 2,000 and 10,000 ha; 3 % in patches between 500 and 2,000 ha; and 5 % of the remaining forest area occurred in patches of less than 500 ha (Fig. 4).

Fig. 4: Temporal variation of the patches size of A) Inter-Andean Dry Forest and B) Tropical Montane Forest in the HRGW



In 1991, 89 % of the Tropical Montane Forest area was concentrated in patches between 50,000 and 150,000 ha; 4 % in patches between 500 and 10,000 ha; and 7 % of the remaining forest area occurred in patches of less than 500 ha (Fig. 4). In 2005, 42 % of the Tropical Montane Forest area was concentrated in patches between 50,000 and 150,000 ha; 19 % in patches between 10,000 and 50,000 ha; 10 % in patches between 500 and 10,000 ha; and 28 % of the remaining forest area occurred in isolated patches of less than 500 ha. In 2017, 56 % of the Tropical Montane Forest area occurred in patches between 10,000 and 50,000 ha; 12 % in patches between 500 and 10,000 ha; and 32 % of the remaining forest area occurred in isolated patches of less than 500 ha (Fig. 4).

The largest forest patch of Inter-Andean Dry Forest and Tropical Montane Forest decreased by 1.95 % and 16.33 %, respectively, from 1991 to 2017 (Table 3). The core area of the Inter-Andean Dry Forest and Tropical Montane Forest decreased by 0.10 and 0.29, respectively, over the entire study period (Table 3). A decrease in the aggregation index between forest patches was observed. The largest decrease in the aggregation index occurred from 1991 to 2005, which was 3.09 % and 8.51 % for Inter-Andean Dry Forest and Tropical Montane Forest, respectively. From 1991 to 2005, an increase in the shape complexity of the Inter-Andean Dry Forest and Tropical Montane Forest patches was observed (Table 3). However, from 2005 to 2017 a decrease in this index was registered for both ecosystems. The degree of edge contrast between the forests and agriculture, which was the dominant cover in the landscape, increased by 32.6 % and 31.5 % for Inter-Andean Dry Forest and Tropical Montane Forest, respectively. The largest increase in the degree of edge contrast occurred from 1991 to 2005. In the first study period, the total edge length increased by 1,637 km and 14,443 km for Inter-Andean Dry Forest and Tropical Montane Forest, respectively. In the second study period, a decrease in this index was registered for both ecosystems. The distance between patches of Inter-Andean Dry Forest and Tropical Montane Forest increased by 156 m and 68 m, respectively, over the entire study period (Table 3). The largest increase in the distance between forest patches occurred from 1991 to 2005, which was 118 km and 63 km for Inter-Andean Dry Forest and Tropical Montane Forest, respectively (Table 3).

DISCUSSION

Deforestation and fragmentation patterns

This study revealed an important increase in agriculture and urban land-use between 1991 and 2017. The urban land-use was that registered a higher increase mainly due to the conversion of shrub vegetation to urban areas. While the conversion of forest lands to agriculture was the one that generated the greatest impact on Inter-Andean Dry Forest and Tropical Montane Forest. The largest loss of native forests occurred from 1991 to 2005. The annual loss rate of forests reported in this study (1.0 % per year) is higher than that recorded in other hotspot landscapes that have also been significantly transformed, such as the Valdivian Temperate Forest in Chile (0.5 % per year) (Rodríguez-Echeverry *et al.*, 2018a) and the hotspot of the Eastern Arc Mountains of Tanzania (0.5 % per year) (Green *et al.*, 2013). Similar to other countries, the deforested areas in the Ecuadorian Andes are mainly used for agriculture.

Specifically, our study registered an important rate of deforestation for Inter-Andean Dry Forest (1.3 % per year) and Tropical Montane Forest (0.7 % per year) between 1991 and 2017. In the provinces of Loja and Zamora Chinchipe in Ecuador, a similar annual deforestation rate in the Inter-Andean Dry Forest (1.2 % per year), and an annual deforestation rate higher for Tropical Montane Forest (6 % per year) were registered

(Tapia-Armijos *et al.*, 2015). Forest loss in the Ecuadorian Andes is close to 80 % due to the conversion of this ecosystem to agriculture (Ferrer-Paris *et al.*, 2018). In the study landscape, forest deforestation is commonly characterized by its conversion to agriculture and is mainly concentrated in the valleys and lower slopes where it is more accessible to use the land. With an expanding population, production pressures have driven a rapid expansion of agriculture into forested areas (Sierra, 2013). The lack of effective conservation planning has inadequately exposed an area of high diversity and endemism to high rates of forest deforestation and fragmentation. Accordingly, urgent long-term conservation actions are needed in this landscape.

The loss of native forest area was associated with an increase in the number of patches during the first study period and a decrease in the subsequent study period. This trend is similar to that reported in other hotspot landscapes that have been significantly transformed, such as the Chilean Temperate Forest in the Nahuelbuta Mountain between 1986 and 2011 (Otavo & Echeverría, 2017) and in the Chilean winter rainfall–Valdivian forests between 1970 and 2010 (Miranda *et al.*, 2017). Our results show that the HRGW is a highly fragmented forest landscape. However, it is important to highlight that over the entire study period, and between 1991 and 2005, the main process that modified the Inter-Andean Dry Forest and Tropical Montane Forest was deforestation. The deforestation and fragmentation hotspots in the HRGW are located in areas where land is accessible, which is similar to that reported in the Colombian Andes (Rodríguez *et al.*, 2013). If the current trajectory of loss and fragmentation of Inter-Andean Dry Forest and Tropical Montane Forest continues, further losses of both ecosystems can be expected.

State of the landscape

In 2017, the landscape registered a high rate of deforestation, a decrease in the number of forest patches, an increase in the distance between forest patches, a decrease in patch edges, and an increase in the matrix dominated by agriculture. Our results show that the landscape studied corresponds to an advanced state of a transformed landscape (McIntyre & Hobbs, 1999; Echeverría *et al.*, 2012). In this state, the loss of forest patches predominates over the division of them. Moreover, forest loss usually occurs after intense processes of forest fragmentation (McIntyre & Hobbs, 1999; Echeverría *et al.*, 2012). The state of the landscape studied is more critical than that observed in India between 1975 and 2005 (Reddy *et al.*, 2013); and in southern Ecuador, between 1976 and 2008 (Sierra *et al.*, 2013), where the reduction of large patches of native forest, and an increase in the number of patches occurred.

Changes in the spatial pattern of forest cover

The changes in the number of patches and total edge length were associated with deforestation. These metrics increased in the earliest stage of forest loss and fragmentation and decreased during the later stages of deforestation. Echeverría *et al.* (2006), also observed that the constant action of deforestation led to a decline in the number of patches and total edge length in central Chile. The decrease of edge length indicates that forest patches on average became less geometrically complex over time. Habitat patch shape has a strong effect on the presence of species, with fewer species in less geometrically complex patches (Haddad *et al.*, 2015). In the study landscape, the patches of Inter-Andean Dry Forest and Tropical Montane Forest are less geometrically complex and sparsely distributed across the landscape, which can affect organisms with low mobility (McIntyre & Hobbs, 1999).

In our study, patch size declined over time, which differs substantially from the situation recorded for the scrubby forest in the Nahuelbuta Mountain (Otavo & Echeverría, 2017). Haddad *et al.* (2015), stated that progressive reduction in the size of forest habitats is a key

component of ecosystem fragmentation. Therefore, the size reduction of forest patches in landscapes with high conservation priority may affect the availability of forest habitats (Moreno *et al.*, 2014). In addition, several studies carried out in different forest ecosystems have shown that progressive reduction in the size of forest patches alters the structure of the forest habitat (Torrella *et al.*, 2013; Rodríguez-Echeverry *et al.*, 2015). Consequently, as the structural diversity of the forest decreases, the use of the habitat also decreases, which can produce a severe fragmentation and decline of the endangered populations (Lindenmayer & Fischer, 2006).

High-contrast edges between Inter-Andean Dry Forest and Tropical Montane Forest with the agriculture were recorded in the landscape studied. Previous studies have also reported that monocultures may exhibit high-contrast edges with the natural habitat due to a more simple composition and structure (Reino *et al.*, 2009; Echeverría *et al.*, 2013). The edges play critical roles in the ability of species to adapt and move between disturbed habitats and in the introduction of edge effects that cause a deterioration in the qualities of habitats (Peyras *et al.*, 2013). Therefore, a marked contrast between native forest ecosystems and human-related land can increase the risk of extinction for endangered species (Laurance *et al.*, 2002; Peyras *et al.*, 2013; Torrella *et al.*, 2013). In the study landscape, the trajectories of non-forest cover changes were useful to explain the changes in the forest cover since the 1990s. While agriculture and urban areas cover increased, Inter-Andean Dry Forest and Tropical Montane Forest became gradually more fragmented and decreased their aggregation.

The results of the edge contrast metric were derived from field measurements, which were spatially analyzed by using remote sensors. The analysis of edge contrast in different spatial scales has been highlighted as a very important factor in detecting edge influence (Granados *et al.*, 2014). Therefore, the edge contrast metric can be used to analyze the impact of human-induced land-use change on the spatial pattern of the forest landscape and thus to propose general guidelines for management and restoration in fragmented forest landscapes (Sano *et al.*, 2009). This recommendation can contribute to recovering ecological integrity, especially in landscapes where the amount, size, and spatial configuration of suitable forest habitats are determinant for edge-sensitive species.

We observed that the highest loss of forest cover was associated with the highest loss of core areas over the entire study period. The reduction trend in the core area reported in this study is similar to that observed in native forests of other landscapes that are also areas of high priority for world conservation (Miranda *et al.*, 2017; Otavo & Echeverría, 2017). The core areas provide the optimal biotic and abiotic conditions to maintain biodiversity and ecological processes (Isaac-Cubides & Ariza, 2016). Therefore, as the forest patches are smaller, the contribution of their core area will be less, as well as the integrity of the cover. If the loss of forest cover continues in the studied landscape, further losses of core area can be expected, which could impact the biodiversity and ecological processes of both ecosystems.

The landscape studied has been affected over the last 26 years by progressive isolation of the forest patches due to a significant reduction of forest cover by conversion to agriculture and urban areas. The main change in the average distance to the nearest neighbor was recorded between 1991 and 2005. During this period, the neighborhood of forest patches rapidly became occupied by areas of a different land cover type, as native forest patches became spatially separated and less contiguous in distribution. Some researchers have stated that the disruption of landscape connectivity may have important consequences for the distribution and persistence of population, and therefore with the maintenance of species diversity (Bennet, 2003; Pacioni *et al.*, 2015; Thompson *et al.*, 2017). Also, the disruption of landscape connectivity may lead to an increased risk of local extinctions, making species

more vulnerable to stochastic processes, natural catastrophes, human threats, and loss of genetic variation (Paviolo *et al.*, 2016; Horváth *et al.*, 2019; Clerici *et al.*, 2020). According to the findings in this study, a discontinuous spatial pattern of forest patches indicates low functional connectivity of the Inter-Andean Dry Forest and Tropical Montane Forest. This low functional connectivity can affect to move of organisms between habitat patches, the colonization of species, and gene exchange between populations (Echeverría *et al.*, 2014). Therefore, the guidelines for management and conservation for both native forest ecosystems should consider the implementation of corridors to reduce the negative effects of fragmentation.

Implications for conservation

The HRGW should be considered as a premium conservation landscape, as it still concentrates high levels of biodiversity and endemism, and is simultaneously in an advanced state of landscape transformation. This premium condition justifies the urgency of safeguarding the Inter-Andean Dry Forest and Tropical Montane Forest whose risk of extinction may increase if the current trajectory of loss and fragmentation continues. We suggest carrying out conservation planning at a landscape scale based on the corridor-patch-matrix model. This model is an alternative to reduce and avoid the degradation of ecosystems, contributes to biodiversity conservation, and guarantees the participation of the Government and the community (Rodríguez-Echeverry & Leiton, 2019).

The corridor-patch-matrix model is appropriate given the current configuration of the landscape of the HRGW, with Inter-Andean Dry Forest and Tropical Montane Forest restricted to small patches sparsely distributed across the landscape. The main objective of the corridor-patch-matrix model is to maintain the quality and quantity of the forest patches through the management of the matrix. Therefore, the management of the matrix should focus on the sensitive buffer areas that improve the connectivity between forest patches and increase the ability of the matrix to maintain biodiversity (Lindenmayer & Fischer, 2006). Consequently, the management of the matrix should include the restoration of the Inter-Andean Dry Forest and Tropical Montane Forest on the sensitive buffer areas, which in turn would improve the connectivity of this ecosystem and increase the availability of habitats for species, such as *S. guentheri*, *C. obsoletum*, *L. triangulum micropholis*, *A. geoffroyi*, *R. colomaronani*, *D. lafresnayii*, among others. The management of the matrix should also include the implementation of sustainable production practices, such as agroforestry on the sensitive buffer areas. These practices could reduce the edge contrast and, thus, lowers the impacts to the patches of Inter-Andean Dry Forest and Tropical Montane Forest. The design and implementation of this strategy require the evaluation and identification of the sensitive buffer areas and their connectivity. Furthermore, this strategy must be complemented with land use planning and supported within a framework of environmental policies that guarantee the commitment and active participation of the Ecuadorian Government and the community in general. Finally, we suggested that future studies address the causes of the loss of Inter-Andean Dry Forest and Tropical Montane Forest and landscape transformation of the HRGW.

CONCLUSIONS

Our study constitutes the first analysis carried out on the spatio-temporal changes of the Inter-Andean Dry Forest and Montane Tropical Forest, in the HRGW, due to the intensification of agricultural and urban land-use. Accordingly, our study contributes to

a better understanding of the impacts of increased agriculture and urban areas on these native ecosystems. Our results demonstrated that Inter-Andean Dry Forest and Tropical Montane Forest loss were largely caused by the conversion of forest lands to agriculture. In addition, the loss of both native ecosystems was related to its fragmentation during the first study period. Both ecosystems registered a similar trend of deforestation and fragmentation over the entire study period. According to our findings, urgent conservation efforts are required for the Inter-Andean Dry Forest and Tropical Montane Forest, in order to maintain high levels of biodiversity and endemism, and the provision of the ecosystem services.

The HRGW should be considered as a premium conservation landscape. Based on our findings, we suggest the patch-corridor-matrix model as a strategy for conservation planning of the Dry Inter-Andean Forest and Montane Tropical Forest on a landscape scale. This model is important because allowing maintaining the production lands. At small spatial scales, management decisions should be focused on particular plots of land, be they for commodity production or otherwise. But at landscape-scales, conservation decisions should be focused on the choice between mosaics with contrasting patches of production and conservation lands. In this way, socio-economic development can be in concordance with conservation. Therefore, the patch-corridor-matrix model can ensure the provision of multiple ecosystem services that would provide human well-being to the inhabitants of the HRGW. Furthermore, this model must be supported by a framework of environmental policies, which must be based on the 17 UN sustainability criteria and generated by the Ecuadorian government. We want to emphasize that the successful conservation of both ecosystems requires the engagement of multidisciplinary teams and related stakeholders.

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CONFLICTS OF INTEREST

The authors declare no conflict of interest.

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