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Long-Term Changes in Forest Cover in Central Veracruz, Mexico (1993–2014)

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Abstract

Deforestation and fragmentation are threats to the conservation of species and have consequences for ecosystem functions. The focus of this study was to elucidate forest cover change in the period of 1993 to 2014. Our study area is in the central region of Veracruz, Mexico. Land cover and land use classes for the Years 1993, 2000, and 2014 were derived from Landsat images applying supervised classification. Then, we quantified the net change in forest area, the loss of original forest area, and evaluated forest fragmentation using landscape metrics. Our results showed that the area covered by remnant forests decreased 57%. The annual net forest cover change rate for 1993 to 2000 was −0.44%; since then forest cover increased at a rate of 0.11% from 2000 to 2014. The decreasing total edge density and the mean proximity index during the entire period of the study indicate decreasing irregularity in the shape of remnant forest patches and a slight decrease of vulnerability to edge effects. Forest patches augmented in 2000 and decreased in 2014 demonstrating an 18% decrease in relation to the number of fragments existing in 1993. According to our study, this area demands an urgent attention on preservation initiatives because only 2% of the surface extent is below federal protection and 0.8% is under State protection. It is important to protect the larger forest areas left in the pine-oak and humid montane forest belt because of their importance to plant diversity conservation and particularly, as these are threatened by urban and agricultural expansion.

Keywords

vegetation belts, forest cover change, remote sensing, landscape conservation, neotropics

Introduction

Two critical components of land cover change are deforestation and forest fragmentation (Tapia-Armijos, Homeier, Espinosa, Leuschner, & de la Cruz, 2015). The loss of forest cover represents the main threat to biodiversity, with a negative impact on α-diversity (Carrara et al., 2015). Some of the negative consequences of deforestation are an increase in greenhouse gas emissions (Crutzen, 2006) and species extinctions (Whitmore & Sayer, 1992). Forest loss not only reduces forest cover but also modifies the landscape arrangement (Laurance, Vasconcelos, Stouffer, & Laurance, 2002). In the case of fragmentation, it can have negative effects on different taxa (Carrara et al., 2015). Also, the negative effects can be related to the loss of landscape connectivity and the increase in forest edges (Laurance et al., 2002). Fragmentation decreases the dimension of forest cover, intensifying edge effects, and separation of environments (Tapia-Armijos et al., 2015). The reduction in forest cover can provide obstacles that some species are incapable of crossing and reduce the accessible resources required to preserve native populations (Tapia-Armijos et al., 2015). More than 12% of Mexico's natural forest cover has been lost in the past 50 years (Velázquez et al., 2002). The state of Veracruz had one of the highest rates

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of deforestation in the country and ongoing deforestation has been reported for some areas in the central region (Muñiz-Castro, Williams-Linera, & Benítez-Malvido, 2015).

Forest cover gain has been reported in several studies throughout Mexico. There is a debate if Mexico is really undergoing a classically defined forest transition (Corona, Galicia, Palacio-Prieto, Bürgi, & Hersperger, 2016; García-Barrios et al., 2009; Hosonuma et al., 2012). This has led to a slowing down and eventually reversing of the ongoing land degradation (Belay et al., 2015). At the landscape scale, this could result in forest transition, a shift from net deforestation to forest regeneration (Lambin & Meyfroidt, 2011). The decrease of deforestation rate and reforestation or secondary vegetation growth are indicators of forest transition driven by less environmental degradation caused by land use change, emigration, or population migration to urban areas and policy interventions (Belay et al., 2015). Increasing job opportunities in other activities than agriculture have resulted in the abandonment of agricultural land; consequently, the forest has recovered (Corona et al., 2016).

Veracruz is an important state in terms of biodiversity preservation (Sarukhan et al., 2014), showing a high degree of plant endemism, that varies from the rest of the country (Gómez-Pompa, Krömer, & Castro-Cortés, 2010; Villaseñor & Ortíz, 2014). The mountainous region of Central Veracruz hosts important ecosystems, such as humid montane and pine-oak forests, which are rich in biodiversity and offer a variety of ecological services (Williams-Linera, Guillén-Servent, Gómez-García, & Lorea-Hernández, 2007). However, species richness and floristic composition have been found to be affected by the effects of deforestation and fragmentation (Carvajal-Hernández, Krömer, López-Acosta, Gómez-Díaz, & Kessler, 2017; Gómez-Díaz, Krömer, Carvajal-Hernández, Gerold, & Heitkamp, 2017).

Understanding the relative effects of landscape composition and configuration on biodiversity is needed to design effective conservation strategies. Therefore, the aim of this study is to describe forest cover change and fragmentation in the highly diverse montane forest area of Central Veracruz from 1993 to 2014 by (a) determining forest cover change rates in the study area during two periods (1993–2000 and 2000–2014) and (b) assessing the dynamics of forest cover (fragmentation) during the study period by applying landscape metrics.

Methods

Study Area

Our study area is located at 96° to 97° W and 19° to 20° N; it covers approximately 6,987 km^2 mainly in the central region of the state of Veracruz, whereas a small fraction (ca. 7%) belongs to the adjacent state of Puebla (Figure 1). This particular area was selected due to the inclusion of the major subwatersheds found within (Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, 1970), which are part of the Antigua River watershed and is considered a national priority for restoration to maintain ecosystem services as water supply and water quality (Cotler Avalos & Garrido Pérez, 2010). Additionally, our study area was chosen due to the outstanding species richness of angiosperms within Veracruz (García-Franco, Castillo-Campos, Mehltreter, Martínez, & Vázquez, 2008; Palacios-Wassenaar, Castillo-Campos, Vázquez-Torres, & Del Amo-Rodríguez, 2014; Villaseñor & Ortíz, 2014).

However, more than 80% of Veracruz' primary vegetation had been converted to pastures, plantations, and secondary vegetation and the remaining forest or vegetation is highly fragmented, especially in the central part of the state (Ellis, Martínez-Bello, & Monroy-Ibarra, 2011). The study area is therefore recognized as a priority region for conservation within Mexico (Ellis et al., 2011). The elevation of Central Veracruz ranges from 0 to 4,282 m, leading to a vertical gradient that encloses several altitudinal climate zones (Lauer, 1973) and vegetation belts (Diaz Castellón, Carrasco Núñez, & Álvarez-Manilla Aceves, 2008; Gómez-Díaz, Krömer, Carvajal-Hernández, et al., 2017; Figure 1, Online Appendix 1).

Land Cover and Land Use Classification

Historic land cover and land use data for a period of 21 years, specifically for the Years 1993, 2000, and 2014, were obtained from Landsat 5, 7, and 8 images, respectively (Online Appendix 2). We used Landsat Level 1 T products that were previously topographically and geometrically corrected and were suitable for multidate comparison of scenes. The Landsat scenes used in our classifications are provided in georeferenced form by U.S. Pat. Geological Survey (USGS) in the reference system WGS 1984 and in the UTM zone 14 and 15 projection (Online Appendix 2). A geometric correction was performed using the *Cubic Convolution (CC)* resampling method (U.S. Geological Survey, 2014). Furthermore, all scenes were radiometrically corrected using the Radiometric calibration function of ENVI 5.3 (Exelis Visual Information Solutions, Boulder, Colorado) to convert Digital Number (DN) values to surface reflectance values.

We applied a supervised classification of the corrected Landsat images using maximum likelihood estimation (de Lange, 2013). Our own field experience in the study area allowed us to define training samples based on the visual identification of easily distinguishable land cover and land use classes in each landscape. For the land cover

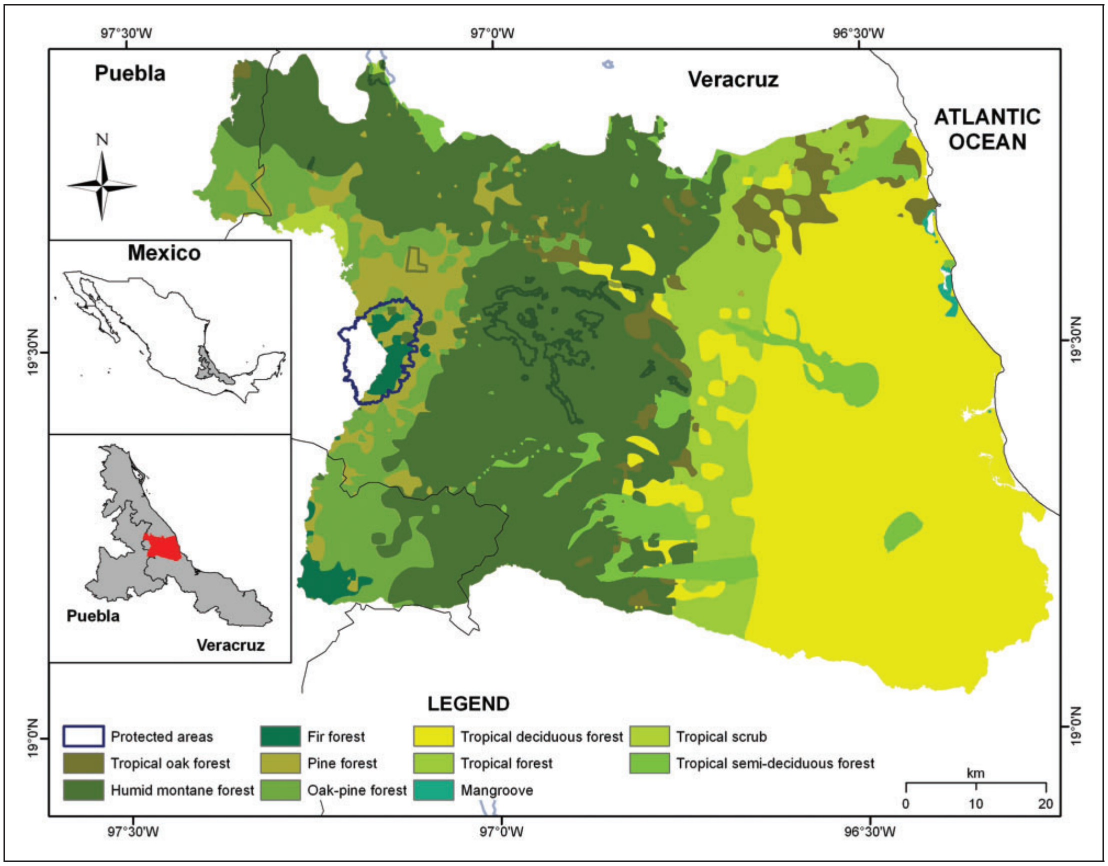

Figure 1. Study area in Central Veracruz, Mexico. Overview on the potential natural vegetation belts in the states of Veracruz and Puebla (Cruz Cárdenas, Villaseñor, López Mata, & Ortiz, 2012; Ellis et al., 2011; Gómez-Díaz, Krömer, Carvajal-Hernández, et al., 2017; INEGI, 2013; Rzedowski, 2006).

and land use classifications in the scenes of 1993, we used 80 training samples collected from the land cover and land use vector data (1:250,000) of 1997 from the Mexican National Institute of Statistic and Geography (Instituto Nacional de Estadística y Geografía, 2000). For the land cover and land use classifications of the 2000 scenes, we used 80 training pixels, using the land use and land cover map of the upper drainage basin of "La Antigua" by Muñoz-Villers and López-Blanco (2008) and the 2003 vegetation and land use classification of Instituto Nacional de Estadística y Geografía (2000) (1:250,000). The scenes for the Year 2014 were classified using 140 ground truth data, which were obtained from vegetation surveys (Gómez-Díaz, Krömer, Carvajal-Hernández, et al., 2017). These vegetation surveys were conducted in different forest stands throughout the study area from 2012 to 2014. For comparison and selection of appropriate training data from the scenes for the Year 2014, we used the classification from Ellis and Martínez (2010), for the total land cover of Veracruz (1:75,000).

For our change detection analyses, the images were classified into three land cover classes: (a) "forest," which are the original dense continuous or fragments of conserved tree cover; (b) "secondary vegetation," which is a degraded forest cover, including old fruit and shaded coffee plantations, as well as scrubby structures of pasture; (c) "coastal and hydro-vegetation," which are vegetation forms adapted to the coastal habitat; and three land use classes: (d) "agriculture," which includes rainfed agriculture (potatoes, maize, beans, etc.), irrigated agriculture (sugar crane, citrus, etc.), and pastureland for cattle raising; (e) "no vegetation or use" refers to unproductive and bare soil; and (f) "other," which includes waters, clouds, and built-up areas.

Once the images of each year were classified using the previous criteria, we focused on the forest class for a subsequent classification where: (a) all the pixels classified as forest were assigned to a vegetation type using Figure 1 (tropical oak forest, humid montane forest, fir forest, pine forest, pine-oak forest, tropical deciduous forest, tropical rainforest, mangrove, tropical scrub, and tropical semi-deciduous forest); then, we simplified the categories according to the classification proposed by Gómez-Díaz, Krömer, Kreft, et al. (2017); and (b) the pixels classified as forest in 1993 were used as the reference base for "remnant" forest, while for the Years 2000 and 2014, the forest pixels that have not changed since 1993 were classified as remnant forests.

Accuracy Assessment

We conducted a thematic accuracy assessment for the 2014 classification using an average of 250 reference points per image in ArcGIS. We used reference points collected in the field described earlier and made sure that the reference points did not coincide with those used to create training areas to avoid bias. To determine the classification accuracy of the image, we created a confusion matrix of the reference point classes with corresponding image classes. The reference pixels were randomly matched with pixels of the grid. According to de Lange (2013), the user's accuracy results from the probability that the classified land cover and land use is also the actual land cover and land use at the site.

Forest Cover Change Assessment

We considered remnant forest as those forest areas that did not change since 1993. We conducted two levels of deforestation analyses: (a) the changes in the forest cover class were used to calculate the mean annual deforestation rates for each period (1993–2000; 2000–2014); and (b) the changes in the remnant forest were used to calculate the yearly gross-forest loss in the area (Tapia-Armijos et al., 2015). Both analyses were applied to the total area and on each forest cover based on our natural vegetation belts data (e.g., fir forest, pine forest, pine-oak forest, humid montane forest, tropical oak forest, and tropical semideciduous forest; Figure 1). We used the formula (Equation (1)) proposed by Puyravaud (2003) to obtain annual net forest change and annual gross forest cover loss, defined by Hansen, Stehman, and Potapov (2010) as any "conversion of primary forest cover to non-forest cover."

$$P = \frac{100}{t_2 - t_1} \ln \frac{A_2}{A_1} \tag{1}$$

where A_1 and A_2 are the area cover by natural forest at time t_1 and t_2, respectively, and P is the annual deforestation rate.

We analyzed the transitions based on the postclassification analysis through an extended transition matrix. The method was designed to account for persistence to obtain the net change of each land-use type in the transition. Transition matrices were calculated based on postclassification across maps (Soares-Filho, Coutinho Cerqueira, & Lopes Pennachin, 2002).

Fragmentation Analysis

We used the landscape metrics proposed by Tapia-Armijos et al. (2015) in order to analyze the spatial configuration of the fragments of the main vegetation types listed earlier. We used the program FRAGSTATS 4.2 (McGarigal, Cushman, & Ene, 2012) to calculate the following parameters as proposed by Tapia-Armijos et al. (2015): (a) patch area, (b) number of patches, (c) patch density, (d) largest patch index, (e) edge density, (f) mean patch size, (g) total core area, and (h) mean proximity index.

Results

A confusion matrix was used for the corroboration of the land cover or land use maps (Online Appendix 3). Our results showed an overall accuracy of 82.5% for the Year 1993, 88.9% for the Year 2000, and 84.3% for the Year 2014, which means that the pixels with land cover and land use were correctly distinguished among them.

Overall Forest Cover Change Patterns

The land cover or land use maps of 1993, 2000, and 2014 (Figure 2) were used to derive variations in the land cover and land use (Online Appendix 4). Throughout the entire study period, the zone covered by remnant vegetation (vegetation without change since 1993) was reduced by approximately 57%. For the whole study period (21 years), the mean net-forest cover change rate was −0.17%.

The patch density decreased from 1,102 to 906 patches per 100 km^2 (Table 1). The decreasing total edge density (m/ha) and the mean proximity index during the entire period of the study indicate an increase in roads and this can influence edge effects negatively. Also, a decrease in the proximity index means that the patches are moving further apart.

Forest Cover Change Patterns at Individual Time Periods

During the first period (1993–2000), the annual net forest change rate in Central Veracruz was −0.44%; and conversely, there was a regrowth of vegetation (+0.11%) during the second period (2000–2014). For the study area, the annual gross forest loss rate was −1.17% during the first period, which was reduced to −0.14% during the second period (Figure 3).

Forest cover was 37% of the study area for 1993, 30% in 2000, and 37% in 2014. Forest cover decreased from 2,774 km^2 in 1993 to 2,076 km^2 in 2000 (approximately 19% decrease during the period) at 0.4% per year, but by 2014, it had partially recovered to 2,562 km^2, an increase of 0.11% per year (2000–2014). In the first

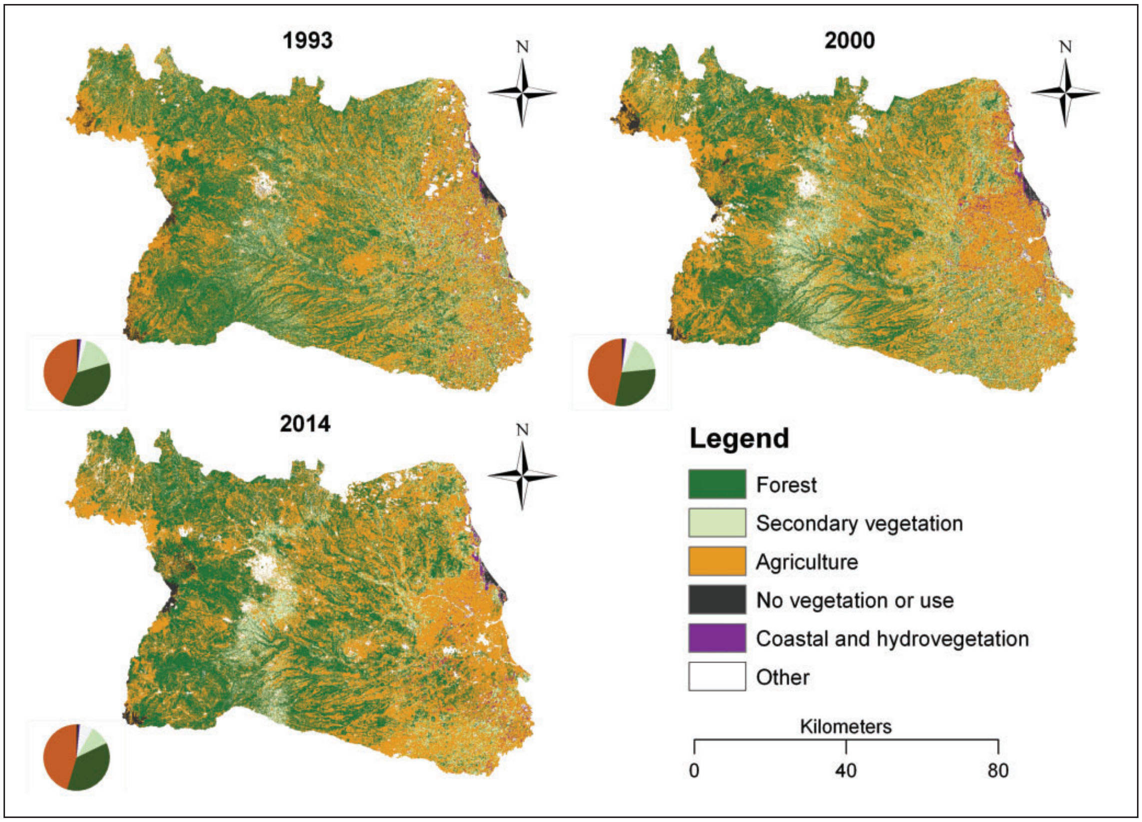

Figure 2. Land cover or land use maps for the studied years (1993, 2000, and 2014). The spatial distribution arrangements of the land cover or land use categories in Central Veracruz, Mexico, are shown on the maps. The limits of the protected area are displayed in black lines.

period, other land use or land cover types increased mainly at the expense of forest cover, which experienced high deforestation rates (Table 2, Figure 3); in fact, agricultural activities account for >22% of the deforestation. Nevertheless, the contribution of agriculture decreased in the second period, whereas the contribution of other land uses (build-up mainly) increased (Table 2). In 1993 to 2000, the area covered by agricultural land increased at the expense of forest cover (8% of the area), whereas during 2000 to 2014, deforestation represented 5% of the total area (Table 2). Forest proved to be the most dynamic land cover type or class since all the other land use and land cover types or classes for both periods affected previously forested areas. However, secondary vegetation and agriculture were the main land cover and land uses associated with forest regeneration. The highest levels of forest regeneration were observed during the second period and were related mainly with the shift from agriculture to the forest in over 9% of the landscape, followed by secondary vegetation with 7% (Table 2).

Forest patches augmented from 76,983 to 10,0831 in 2000 and decreased to 63,273 in 2014 demonstrating an 18% decrease in relation to the number of fragments existing in 1993 (Table 1). Mean forest fragment size increased from 0.03 km^2 in 1993 to 0.04 km^2 in 2014.

Forest Cover and Land Use Change Patterns With Respect to Different Forest Types

From the total forest cover in 1993, 30% represents disturbed forest area fraction (forest cover—remnant forest), 34% in 2000, and 60% in 2014. Throughout the studied period of 21 years, 271 km^2 of the initial 2,586 km^2 of remnant forest have been transformed to secondary vegetation, another 592 km^2 have been transformed to agriculture, and additional 135 km^2 to no vegetation or land use (Table 2). The most important changes in forest cover were the transformations of the tropical oak forest, humid montane forest, tropical rainforest, and mangrove (Online Appendix 4).

Remnant Versus No Remnant Forest

Considering a buffer region of 300 m, the entire core area of lasting remnant forest increased from 3.1 km^2 in 1993

Table 1. Variations in Spatial Conformation of Main Vegetation Types in Central Veracruz, Mexico Throughout the Period From 1993 to 2014.

Forest	Year	PA	NP	PD	LPI	ED	AREA_MN	TCA	PROX_MN
FF	**1993**	21	494	24	40	245	4.16	0.00	263
	2000	13	463	36	28	310	2.77	0.00	125
	2014	20	397	20	41	238	4.99	0.01	404
PF	**1993**	121	1,748	14	53	187	6.93	1.09	1,598
	2000	91	2,082	23	16	240	4.35	0.38	367
	2014	144	2,144	15	34	186	6.74	3.06	1,692
POF	**1993**	263	5,310	20	15	258	4.96	0.20	946
	2000	223	4,894	22	15	238	4.56	0.92	466
	2014	282	4,839	17	14	218	5.82	2.03	973
HMF	**1993**	852	10,872	13	30	306	7.84	0.02	14,140
	2000	714	17,013	24	24	282	4.19	0.79	3,169
	2014	816	13,767	17	29	245	5.93	2.99	7,924
TOF	**1993**	510	14,819	29	22	427	3.44	0.00	3,950
	2000	348	24,644	71	5	488	1.41	0.00	215
	2014	472	17,466	37	13	319	2.70	1.38	1,264
TSF	**1993**	815	47,524	58	8	457	1.71	0.00	620
	2000	695	55,517	80	8	481	1.25	0.00	320
	2014	836	28,027	34	9	291	2.98	1.97	995

Note. FF = fir forest; PF = pine forest; POF = pine-oak forest; HMF = humid montane forest; TOF = tropical oak forest; TSF = tropical semi-deciduous forest; PA = total area of forest (km^2); NP = number of patches; PD = patch density (number of patches/km^2); LPI = largest patch index (%); ED = edge density (m/ha); AREA_MN = mean patch size (ha); TCA = total forest core area[a] (km^2); PROX_MN = mean proximity index.[b]

[a] We measured a buffer region of 300 m length for the calculation of forest core area.

[b] For the calculation of the mean proximity index, we considered a search radius of 300 m width.

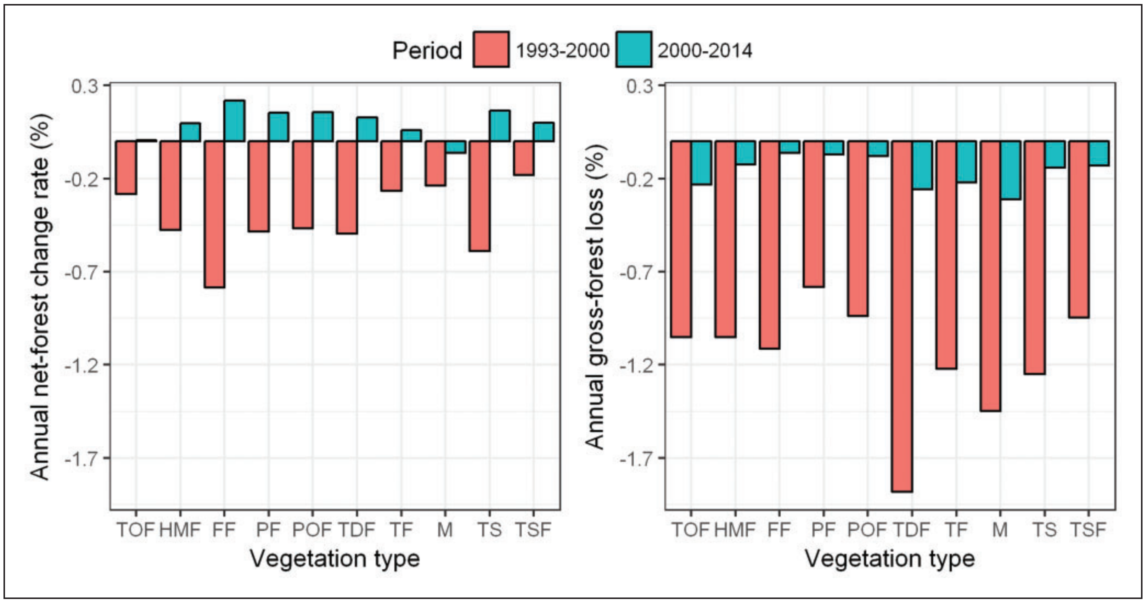

Figure 3. Annual net-forest change and gross forest loss rates (%) in the natural vegetation types. TOF = tropical oak forest; HMF = humid montane forest; FF = fir forest; PF = pine forest; POF = pine-oak forest; TDF = tropical deciduous forest; TF = tropical forest; M = mangrove; TS = tropical scrub; TSF = tropical semideciduous forest in Central Veracruz, Mexico, for the periods 1993 to 2000 and 2000 to 2013.

Table 2. Transitional Matrixes for Central Veracruz, Mexico.

	2000							
1993	FO	SV	AG	NV	CH	OT	Total 1993	Loss
(a) First period 1993–2000								
FO	1,449	481	558	7	6	73	2,574	1,125
SV	294	368	401	0	10	18	1,091	723
AG	303	318	2,163	29	43	114	2,970	807
NV	2	0	21	31	1	12	67	36
CH	3	6	38	1	12	7	67	55
OT	25	21	71	3	6	69	195	126
Total 2000	2,076	1,193	3,252	71	77	294	6,963	
Gains	627	825	1,089	40	65	225		
	2014							
2000	FO	SV	AG	NV	CH	OT	Total 2000	Loss
(b) Second period 2000–2014								
FO	1,432	199	355	5	3	81	2,076	644
SV	463	276	396	0	5	52	1,193	917
AG	596	187	2,236	18	27	188	3,252	1,016
NV	6	0	29	27	2	6	71	44
CH	3	6	50	0	9	8	77	68
OT	62	8	89	12	3	119	294	175
Total 2014	2,562	677	3,156	62	50	456	6,963	
Gains	1,130	401	920	36	40	337		

Note. The matrixes express the changes between land use and land cover in km^2 for each period as well as the total area per year. The total losses and gains for each period are also reported.

to 18 km^2 in 2014 (Table 1). In the first period (1993–2000), the mean proximity index, which is more sensitive to patch distribution, was reduced. That means that the landscape had dispersed patches (Table 1).

We identified 1,100 km^2 of forest that had not changed since 1993. These remnant forests are located mostly at mid-elevations in the transition belt of humid montane and pine-oak forest (Online Appendix 5 and Figure 4). There are also small corridors in the tropical oak forest belt located at the ravines. At the extremes of the altitudinal gradient, there are almost no patches of remnant forest left (Online Appendix 5 and Figure 4). The area of the regenerating forest after 2000 is 1,474 km^2, which represents 21% of the total study area and 57% of the forest area in 2014 (Figures 2 and 4).

Discussion

Deforestation Patterns and Fragmentation Configurations

Mexican land use policies have contributed to the transformation of natural forest to other land uses. In the period between 1970 and the 1990s, deforestation in Mexico was related to the implementation of federal agricultural and livestock programs by the Secretariat of Agriculture (López-Barrera, Manson, & Landgrave, 2014).

Data on population growth indicate that urban expansion soared from the 1960s, which have resulted in the decline of forest cover and the disappearance of natural vegetation (López-Barrera et al., 2014). Although in some areas of the region, such as the upper watershed of the Pixquiac, forest regeneration was precisely due to the abandonment of agricultural activities by the population-seeking employment in the surrounding urban centers, which is a driver of increasing secondary vegetation (Gerez, Fuentes, Vidriales-Chan, Toledo-Aceves, & Pérez, 2012).

We found that fragmentation patterns varied considerably between the two different time periods analyzed. In our study area, deforestation and forest fragmentation were concentrated in the vegetation types at lower elevations and flat areas (Figures 2 and 3). Since the Spanish conquest, these regions were extensively used and the remnant vegetation was almost completely lost due to overexploitation (Gómez-Pompa & Castillo-Campos,

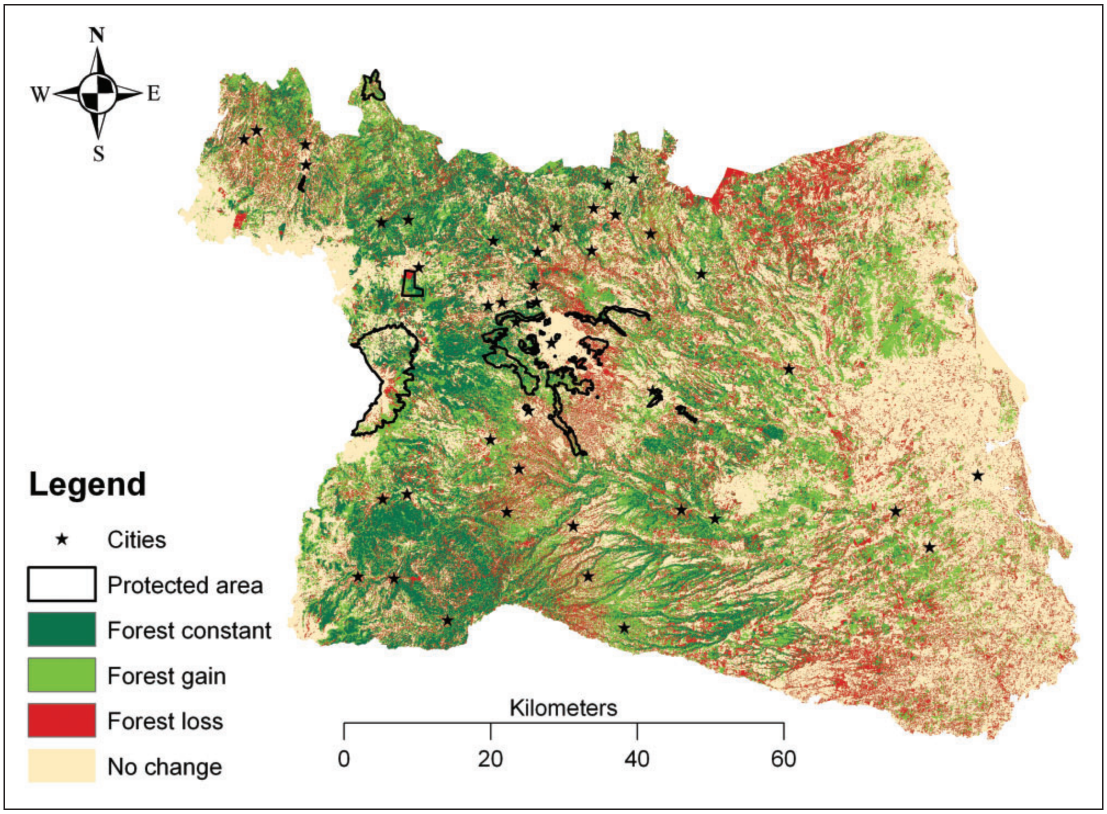

Figure 4. Forest change use analysis. Fragments of forests without change since 1993 (forest constant), forest gain since 1993, forest loss since 1993, and no change.

2010). Nowadays, those regions are still important for agriculture, which has been reported as the main deforestation driver in Latin America (Hosonuma et al., 2012). In addition, conserved forest cover was mostly replaced by agriculture, followed by secondary vegetation and finally, with other land use types (mainly built-up; Table 2, Online Appendix 4).

Forest Transition

We found a recovery of forest cover during the second period (2000–2014). One factor that can explain this recovery is the abandonment of agricultural land due to high rates of emigration to the USA (López-Barrera et al., 2014; Tuirán, 2002). Many farmers have abandoned their agrarian lifestyle in the study area due to the implementation of NAFTA free-trade arrangements over the last decade (López-Barrera et al., 2014), which augmented rivalry with the U.S. producers.

As a consequence, the less profitable agricultural land was abandoned, resulting in an exodus of the rural population (García-Barrios et al., 2009). There is a relation between the processes of abandonment, secondary forest growth, and the recovery of forest cover that is reflected in the decrease in the rate of loss of forest cover. These trends of forest recovery, particularly notable in temperate forests and higher elevations, are being observed and are attributed mostly to agricultural abandonment, urbanization, and change in productive activities.

López-Barrera et al. (2014) studied the vegetation cover of seasonally dry tropical forests 10 km south of our study are and found that most deforestation happened more than a century ago. However, among shorter and different time periods, rates of forest loss varied with reforestation during the period of 1973 to 1990, followed by a noticeable deforestation in the period of 1990 to 2000 mostly transformed into open forest, and finally a rise in forest cover in the period of 2000 to 2007 that matches with tendencies in population migration and modifications in public policy toward conservation and reducing deforestation.

In a forest cover change study of the tropical dry forest in two municipalities of southern Oaxaca, Mexico, it was found that the municipality with a larger agriculture area and a higher proportion of rural population had higher deforestation rates through the period (1985–2006; Corona et al., 2016). The main deforestation drivers were rainfed agriculture and pastureland for cattle. Also, they found a recovery in forest cover (~0.5% annually) between 1995 and 2006 and that the introduction of an economic activity unrelated to agriculture (tourism) promoted the internal migration, which has lead to the abandonment of croplands.

The forest transition (FT) framework considers land abandonment as an opportunity for biodiversity conservation (Rudel et al., 2005). According to Hosonuma et al. (2012), Mexico is in Phase 3 (late transition) of the FT model and not yet experiencing net forest renewal. A large proportion of forests in countries at the late FT phase remains degraded (Hosonuma et al., 2012), which is true in our study area. Timber extraction, logging, and uncontrolled fires are the main drivers of forest degradation in Latin America (Hosonuma et al., 2012), which are also found in our study area.

However, there is evidence that forest losses exceed forest gains in Mexico; García-Barrios et al. (2009) reported that in low and mid-altitudes, there is an expansion of livestock and grassland. In our study, we have found that this area has the highest proportion of converted forest (70% in low and 52% in mid altitudes). Moreover, in our study area, there are still deforestation processes occurring in the tropical forest (dry and humid).

For the whole study area, the annual gross forest loss rate was −1.17% during the first period (1993–2000), which was reduced to −0.14% during the second period (2000–2014). However, this is not a signal that forest transition is occurring in our study area, as this value is about the national average deforestation rate of the country according to Food and Agriculture Organization (2015). García-Barrios et al. (2009) claim that Mexico is not really by definition experiencing a forest transition since it is not particularly related to a process of economic development and changes in agricultural practices, intensification, production, and growth of the industrial and nonagricultural sector. Rather they claim that globalization, rural out-migration, poverty, or rather economic hardships are responsible for forest renewal in some areas, although deforestation in the tropics seems to be the business as usual.

Value of Remnant Forests

It is necessary to distinguish between the value of the remnant forest and secondary vegetation due to the ecological importance of the former (Gibson et al., 2011). In our study area, we found that deforestation and land use change has reduced the size of remnant forests mainly in humid montane forest (−637 km^2), tropical deciduous forest (−395 km^2), and tropical rainforest (−135 km^2; Figures 2 and 4). In the second period, there was a regrowth of forests; however, these secondary forests of 15 to 20 years derived from humid montane forests showed a reduction in fern species of 37% to 50% and marked changes in species composition (Carvajal-Hernández & Krömer, 2015). On the other hand, according to Williams-Linera, Álvarez-Aquino, Muñiz-Castro, and Pedraza (2016), there is a potential that richness and diversity of trees in the secondary forests are similar to the remnant forest of 15 and 25 years. Respectively, the average canopy height recovers after 35 years, and maximum height, basal area, and density of trees after 80 years from logging.

There is only one federal protected area in our study area (National park "Cofre de Perote") that comprises only about 2% of the total surface (about 117 km^2; Figure 2). However, there has been an enlargement of protected areas in Central Veracruz, especially where remnants of humid montane forests can be found; in 2015, the *Archipiélago de bosques y selvas de la región capital del Estado de Veracruz* Natural Protected Area of 55.8 km^2 was decreed. Nevertheless, preservation actions are an urgent task in the study area as indicated by our main results of forest gross loss.

Moreover, an extra issue to be considered is the extreme fragmentation among patches of remnant forests, especially because a high beta diversity has been detected on those fragments (Gómez-Díaz, Krömer, Kreft, et al., 2017). The remaining remnant natural forest areas constitute minor vegetation islands detached from each other (Online Appendix 5), a phenomenon that may have important consequences on biodiversity. This condition is mainly perturbing since Veracruz is one of the most important states in Mexico regarding total plant species diversity and its flora is characterized by a high level of endemism (Villaseñor, 2016; Villaseñor & Ortíz, 2014).

Implications for Conservation

In biologically diverse regions like Central Veracruz, a better understanding of the patterns, dynamics, and spatial degree of deforestation and forest fragmentation is desirable. According to our study, this region demands urgent attention in implementing preservation initiatives because only 2% of the surface extent is below federal protection and 0.8% is under State protection due to the new archipelago reserve. It is important to protect the larger remnant forest areas left in the pine-oak and humid montane forest belts because of their high contribution to plant diversity conservation, ecosystem services (water and carbon storage), and particularly, as these are still threatened by urban and agricultural expansion.

Finally, it is important to emphasize the need to control and restrict the causes of deforestation at the regional level and in terms of the different vegetation types that are most affected in Veracruz. On the one hand, it is necessary to stop the expansion of the agricultural and livestock land uses by consolidating Sustainable Agricultural Intensification, and on the other hand, it is also necessary to evaluate the economic and social viability of implementing environmental services programs, which promote the permanence and maintenance of natural ecosystems. Our findings have important potential implications for the management and conservation of tropical forests in Central Veracruz, as we have identified those sites where conservation should be prioritized.

As an outlook, we can expect in the future for Veracruz, a recovery and regrowth of forest in some areas if the current low deforestation rates continue. However, it is important to plan and design corridors that connect the remaining remnant forests with these young forests to promote the protection and development of biodiversity, although it takes at least 80 years for a secondary forest to recover from disturbance (Williams-Linera et al., 2016).

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