Ecological Applications, 28(8), 2018, pp. 1982–1997

Payments for ecosystem services in Mexico reduce forest fragmentation

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Abstract. Forest fragmentation can lead to habitat reduction, edge increase, and exposure to disturbances. A key emerging policy to protect forests is payments for ecosystem services (PES), which offers compensation to landowners for environmental stewardship. Mexico was one of the first countries to implement a broad-scale PES program, enrolling over 2.3 Mha by 2010. However, Mexico's PES did not completely eliminate deforestation in enrolled parcels and could have increased incentives to hide deforestation in ways that increased fragmentation. We studied whether Mexican forests enrolled in the PES program had less forest fragmentation than those not enrolled, and whether the PES effects varied among forest types, among socioeconomic zones, or compared to the protected areas system. We analyzed forest cover maps from 2000 to 2012 to calculate forest fragmentation. We summarized fragmentation for different forest types and in four socioeconomic zones. We then used matching analysis to investigate the possible causal impacts of the PES on forests across Mexico and compared the effects of the PES program with that of protected areas. We found that the area covered by forest in Mexico decreased by 3.4% from 2000 to 2012, but there was 9.3% less forest core area. Change in forest cover was highest in the southern part of Mexico, and high-stature evergreen tropical forest lost the most core areas (−17%), while oak forest lost the least (−2%). Our matching analysis found that the PES program reduced both forest cover loss and forest fragmentation. Low-PES areas increased twice as much of the number of forest patches, forest edge, forest islets, and largest area of forest lost compared to high-PES areas. Compared to the protected areas system in Mexico, high-PES areas performed similarly in preventing fragmentation, but not as well as biosphere reserve core zones. We conclude that the PES was successful in slowing forest fragmentation at the regional and country level. However, the program could be improved by targeting areas where forest changes are more frequent, especially in southern Mexico. Fragmentation analyses should be implemented in other areas to monitor the outcomes of protection programs such as REDD+ and PES.

Key words: deforestation; forest loss; habitat loss; image morphology; land use change; Morphological Spatial Pattern Analysis program; protected areas.

Introduction

Human activities modify landscapes around the globe, leading to increased fragmentation of ecosystems (Wade et al. 2003). Fragmentation (i.e., the breaking apart of habitat, not the reduction of habitat area) results in an increase in the number of patches, decrease in patch sizes, and increasing isolation of patches (Fahrig 2003). Although forest fragmentation *per se* may have limited negative or even some positive effects on ecosystems (Fahrig 2017), it is often coupled with habitat loss, which has adverse effects on ecosystems such as loss of biodiversity (Brooks et al. 2002), less habitat connectivity (Dixo et al. 2009), and the promotion of nonnative species (With 2004). Furthermore, while forest fragmentation can be reversed through forest regeneration (del Castillo 2015), its impacts can be long lasting (Ferraz et al. 2003, Vellend et al. 2006, Gibson et al. 2013). In order to prevent these negative effects, multiple conservation efforts have been promoted, including the creation of protected areas. However, despite a growing number of protected areas, deforestation rates are still high in many forests outside of the protection boundaries (DeFries et al. 2005) and new parks are politically difficult to create. This is why other protection programs have been proposed to limit tropical forest loss outside of protected areas including Payment for Ecosystems Services (PES), which are part of efforts to reduce emissions from deforestation and forest degradation, and foster conservation, sustainable management of forests, and enhancement of forest carbon stocks (REDD+; Hosonuma et al. 2012). These policies have become important instruments to limit deforestation across the world (Adhikari and Agrawal 2013, Grima et al. 2016), and although studies on PES schemes have rapidly increased in the past decade (Schomers and Matzdorf 2013), they have been studied far less than protected areas.

Global forest cover change assessments indicate that although forest loss has slowed from 2010 to 2015 (3.3 Mha/yr), compared to the 1990s (7.3 Mha/yr), there are still hotspots of forest loss (Hansen et al. 2013, Keenan et al. 2015, Tyukavina et al. 2016), and this is where fragmentation patterns are also most likely prevalent (Wade et al. 2003). Indeed, while only 3.2% of all forests were disturbed globally between 2000 and 2012, a full 9.9% of interior forests were lost (Riitters et al. 2016), and the global forest wildlands shrunk by 7.2% since the 2000 (Potapov et al. 2017). This highlights why fragmentation analyses are

Manuscript received 8 January 2017; revised 25 March 2018; accepted 8 May 2018.
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crucial to identify whether forest loss is also reducing forest core areas, which are important for wide-ranging species and interior specialists (Laurance 2005, Peres 2005, Vogt et al. 2007a), or increasing forest edge areas (Harper et al. 2005). Moreover, recent global forest loss assessments could help monitoring policies that aim to prevent forest fragmentation in sensitive areas.

Among all forest types, tropical forests have lost the most area (Hansen et al. 2013). This has been the case for several decades, and particularly between the 1990s and 2010s, when tropical forest increased their forest loss rates (Kim et al. 2015). Tropical forest loss is mostly due to the conversion of forests to agricultural and pasture lands (Geist and Lambin 2002, Gibbs et al. 2010). Globally, this pattern is common in lowlands with low population densities, which experience conversion due to agroindustrial plantations (DeFries et al. 2010, Aide et al. 2013). This transformation has large effects on biodiversity, because it occurs in the tropics where species richness is highest (Myers et al. 2000). Moreover, preventing forest loss in the tropics could mitigate changes in the hydrological regime and emission of greenhouse gases (Fearnside 2005, Geissen et al. 2009).

Mexico has experienced high deforestation rates threatening its ecosystems (Velazquez et al. 2002, Mas et al. 2004). Some factors that increase deforestation risk are land suitability, increments in crop prices, slope, and vicinity to roads or population centers (Lopez-Feldman 2012). As a response to the threat of forest loss, Mexico initiated a major PES program in 2003, one of the first in the world. By the year 2010, over 3,300 properties were enrolled in the program, covering >2.3 Mha. This PES program is managed by Mexico's National Forest Commission, CONAFOR, and pays rural landowners in areas with water problems and high deforestation risk to maintain existing forest cover (Muñoz-Piña et al. 2008). Landowners receive five-year contracts and are required to carry out management plans to preserve existing land cover. The Mexican PES program has been evaluated at the local level, reporting a range of outcomes such as noncompliance with PES rules (Costedoat et al. 2015, Le Velly et al. 2015) and difficulties targeting the program to high-risk areas (Muñoz-Piña et al. 2008), but also successes and positive perception of the PES program (Rico García-Amado et al. 2013, Manzo-Delgado et al. 2014, Caro-Borrero et al. 2015). Nationwide assessments have found that the PES program successfully targeted high conservation priority areas and provided neutral or small economic benefits for landowners (Sims et al. 2014, Alix-Garcia et al. 2015, Sims and Alix-Garcia 2017). Prior research also indicates that PES slowed the rate of forest loss but did not completely eliminate deforestation in enrolled parcels (Alix-Garcia et al. 2012, 2015). However, it is unclear whether the PES program also prevented forest fragmentation, which forest types are more affected, and whether there are regional differences in the effectiveness of the program. PES may reduce patterns of forest fragmentation, because the program specifies that landowners must maintain forest cover on enrolled parcels, which are generally contiguous within property and may be close to each other due to the program targeting. At the same time, PES could increase incentives to hide deforestation, possibly driving more clearing of forest interior areas where enforcement is harder to achieve or driving multiple small areas of clearing, leaving more fragmented forests (for models of clearing behavior illustrating similar results; see, e.g., Albers 2010, Sims et al. 2014). In addition, PES usually does not provide complete landscape coverage, so slippage or leakage to areas around the submitted parcel as a result of the program could increase forest fragmentation at the landscape level due to noncontiguous patterns of clearing (for details on slippage, see Alix-Garcia et al. 2012).

Given these reasons why the policy could either decrease or increase forest fragmentation, our goal was to determine whether landscapes with a large share of area enrolled in Mexico's PES program have different forest fragmentation than landscapes with few enrolled areas, after accounting for potentially confounding factors such as baseline forest cover, access to roads, and topography. Our null hypothesis is thus the neutral view that the policy had no effect: high-PES areas would exhibit the same forest fragmentation as low-PES areas (two-sided tests). Our objectives were to (1) calculate the differences in forest fragmentation in Mexico between 2000 and 2012, (2) identify the forest types that were most affected by fragmentation, (3) assess the performance of the PES to prevent fragmentation, and 4) compare the performance of the PES program with that of protected areas to diminish forest fragmentation.

Methods

Study area

We studied forest fragmentation in Mexico, which is varied in topography, located in the transition zone of the Neotropical and Neoartic realms, and has precipitation patterns that are influenced by two oceans. These environmental conditions result in multiple vegetation types across the country, including evergreen tropical forest, deciduous forest, temperate forest, and semideserts (Rzedowski 2006). We divided the country into four regions, that is, north, central, southwest, and southeast, based on their socioeconomic and vegetation similarities, to identify regional variation in fragmentation patterns (Fig. 1). This division was made by grouping neighboring states with similar ecological and socioeconomic characteristics (Table 1).

Data

Forest cover.—We analyzed forest fragmentation from 2000 to 2012 based on the Global Forest Change data set, GFC, which provides wall-to-wall forest canopy cover, gross forest canopy loss, and gross forest gain based on Landsat satellite images (Hansen et al. 2013). We analyzed the 30-m resolution forest cover layer for 2000 as the baseline for our analysis. In order to define a forest vs. nonforest map, we considered any pixel with ≥30% tree cover as forested. We chose this threshold because previous broad-scale forest fragmentation studies used the same value thereby ensuring comparability of our results (Lira et al. 2012, Haddad et al. 2015). In order to obtain net forest cover in 2012, we added the 2000–2012 forest gain to the 2000 forest cover and then subtracted 2000–2012 forest loss. We caution that comparing the forest gain and forest loss products might not reflect

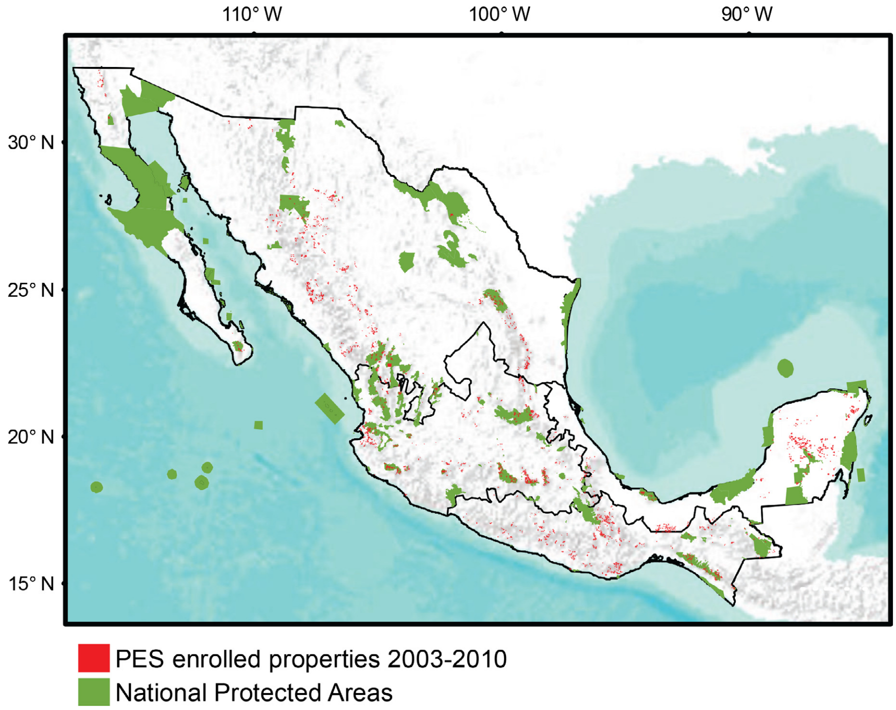

FIG. 1. Study area in Mexico, including the properties enrolled in the payments for ecosystem services between 2003 and 2010, and the north, central, southwest, and southeast zones used for our analysis.

TABLE 1. Summary of socioeconomic characteristics for each study zone.

Zone	Total population in 2010 (10^6)†	Total surface (10^3 km^2)	Total GDP (10^9 pesos) in 2014‡	GDP (%)	GDP per capita in 2014 (10^6 pesos)	Population density in 2010 (no./km^2)	Total forested area (10^3 km^2)	% land covered with forest	Principal forest types
N	26.4	1114.8	3691.1	27.5	0.14	24	138.2	12.4	Forest is confined to mountainous areas, with a mix of deciduous, pine, and oak forests.
C	59.9	368.6	6939	51.8	0.12	163	106.6	28.9	High and low elevation covered by different vegetation types, including tropical deciduous, pine, and oak forest.
SW	12.0	231.3	643.5	4.8	0.05	52	125.6	54.3	Forest vegetation ranges from evergreen and deciduous tropical forest, including oak and pine forest in the mountain ranges.
SE	14.0	233.7	2129.9	15.9	0.15	60	140	59.9	The Yucatan peninsula hosts primarily deciduous and evergreen tropical forest and mangroves, while the highlands will include pine, oak, and cloud forests.

Notes: N, north; C, central; SW, southwest; SE, southeast.
†INEGI 2017a
‡INEGI 2017b

the net forest cover change accurately given the accuracy differences between the products (Hansen and Potapov 2014). Nevertheless, we can assume that these accuracies are equal within and outside PES areas, and within and outside other protected areas, making comparisons among them valid. The GFC has been previously challenged regarding its accuracy (Lindgren 2014) and for not being able to discriminate between natural forest and plantations (Tropek et al. 2014). However, there have been responses addressing these concerns (Hansen and Potapov 2014, Hansen et al. 2014), and the GFC has also been lauded as a cost-effective tool for local and regional management studies (Burivalova et al. 2015, Linke et al. 2017), as well as being the only wall-to-wall, detailed coverage for Mexico that is available during this time period. In order to investigate the GFC's performance for Mexico, we made an accuracy assessment in three regions using deforestation polygons detected using high-resolution imagery available in Google Earth during the same period of analysis. To find these deforestation polygons, we created a systematic sampling grid that covered about 20% of three Landsat footprints located in the northern, central, and southern parts of the country. In each of the quadrants of the grid, we looked for all available high-resolution imagery and digitized forest loss between 2000 and 2012. We sampled 100 random points in the areas that we identified as intact forest and 100 random points within the polygons that we identified as deforested. We then compared these 200 points with the GFC product finding forest cover loss accuracies of 62% and 65% in the central and northern part of Mexico (where vegetation is sparse), respectively, and 83% for the southern part (with denser vegetation). Irrespective of these regional differences, we assumed that the GFC accuracy is equal in areas with PES and in areas without it (within the same geographic region), allowing us to make comparisons among them.

Properties enrolled in PES.—We analyzed all the properties that were enrolled in the Payment for Hydrological Services (PSAH) between 2003 and 2010, which was the largest component of the PES program in Mexico during this period. The property data set was obtained from CONAFOR, and it included 6,297 polygons of enrolled properties ranging in size from 2 to 7,287 ha with a mean of 362 ha. In our analysis, we considered any property that was paid for the entire time between 2003 and 2010, or for some years in this period, as treated by PES. We dissolved all polygons to avoid double-counting in the cases where properties overlapped, given that they were enrolled in different years within our period of analysis.

Forest composition and configuration

In order to quantify forest patterns, we used the Morphological Spatial Pattern Analysis program (MSPA; Soille and Vogt 2009). MSPA has been previously used to analyze forest fragmentation (Vogt et al. 2007b, Estreguil and Mouton 2009) and lauded as a cost-effective tool to monitor forest change (Bucki et al. 2012). With MSPA, we calculated several forest morphology classes, including core (forest interior area), islets (forest patches too small to be considered core), perforation (holes in core area), edge (external perimeter), bridge (corridor connecting core areas), loop (forest corridor ending in same core area), and branch (small area connected to core; Fig. 2). We obtained these forest morphology classes for the forest present in both 2000 and 2012 by applying the eight neighbor rule, in which a forest cell is connected if any of its sides or corners is in contact with another forest cell, and a one-pixel edge (30 m). This neighbor rule and edge width have been used previously to calculate forest patches and their connectivity (Sorte et al. 2004, Locke and Rissman 2012, Rogan et al. 2016). We also calculated the number of forest patches, the mean area of the forest patch, the number of forest loss patches, and the largest area of forest loss using the R package SDMTools (VanDerWal et al. 2014; Table 2).

Microlandscapes

Because the PES properties are very different in size, we defined a constant unit of analysis to compare enrolled and nonenrolled forests based on the microlandscape approach (Sims 2014). This approach divides regional landscapes into smaller units that can be compared using quasi-experimental methods, such as matching, to estimate policy effects. The goal of such methods is to ensure a comparison of actual outcomes between landscapes with similar potential outcomes, based on their region, ecological system, or topography. Specifically, we divided the country into a continuous grid of 1.98×1.98 km microlandscapes (Fig. 2). We chose this grid cell size because the mean size of PES enrolled areas is 362 ha (roughly 1.9×1.9 km). In order to ensure that the forest cover pixels (30 m) were nested within our microlandscapes, we used 1.98×1.98 km (67×67 pixels in our forest cover data) as our grid cell size.

We classified the microlandscape cells according to their protection level. We considered high-PES microlandscapes to be all grid cells with ≥60% of their area enrolled in the PES program. The microlandscapes with <60% of PES enrolled area were treated as low-PES microlandscapes. Similarly, we considered as protected areas all the microlandscapes with ≥60% of their area within the boundaries of Mexico's National Protected Areas System (CONANP 2016). Last, we identified those microlandscapes that are within core zones of Biosphere Reserves in Mexico, which have the highest level of protection because human activities are restricted to research and conservation. For that reason, we performed another filter for the microlandscapes and selected those with ≥80% within the Biosphere Reserves' core zones. This threshold was applied to ensure that most of the pixels were within the core zone.

Matching microlandscapes

We used matching analysis to select a set of microlandscapes with low PES as a control group and to compare them to areas with high PES. Matching analysis estimates policy impact by analyzing differences in outcomes between units with high and low-PES that are otherwise similar in terms of baseline characteristics, essentially mimicking a standard experimental design (Stuart 2010). In order to identify microlandscapes that were comparable, we identified covariates that may influence both deforestation and the PES status of a property, such as the distance to roads and cities, and slope (Alix-Garcia et al. 2015; full set of covariates in Table 3). Based on these covariates, we selected microlandscapes with low-PES area, but similar geographic characteristics (Stuart and Rubin 2008, Randolph et al. 2014) using the R package MatchIt (Ho et al. 2013). We did the matching separately for each of the four zones in the country and only considered microlandscapes with >50% forest cover in 2000. We used Mahalanobis distance with replacement following prior studies (Sims 2014). We calculated the standardized mean difference (SMD) for the matched microlandscapes to verify that the low and high-PES matched samples had similar covariate scores (Austin 2011; Fig. 3). Lastly, we performed a similar process to match high-PES microlandscapes with microlandscapes within protected areas.

We summarized forest morphology transitions between 2000 and 2012 for the whole country and also for each forest type according to a vegetation map for Mexico (CONABIO 1998), using the percent change formula

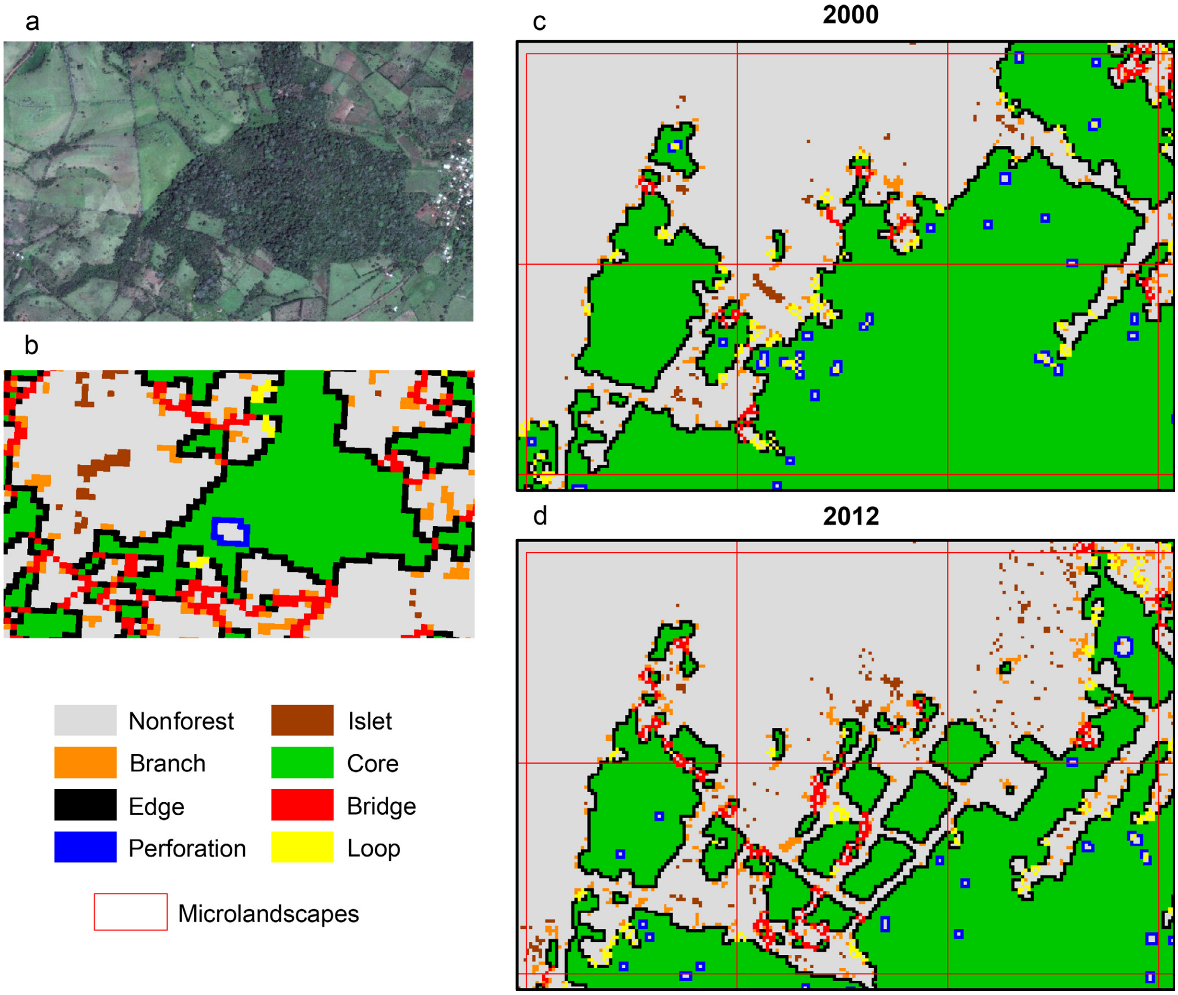

FIG. 2. Example of forest morphology analysis. (a) Aerial view of a landscape with forested (darker) and agricultural (lighter) areas. (b) For the landscape above, each forest pixel was assigned to a particular morphology category: core areas are inner pixels beyond 30 m from nonforest areas; edge are pixels between core and external nonforest areas; perforation pixels are transitions from core to internal nonforest areas; bridges are pixels connecting at least two disjoint core areas; islets are pixel patches too small to contain core; loops are pixels connecting a core area with itself (Clerici and Vogt 2013). Panels (c) and (d) show the forest morphology for six microlandscapes in the years 2000 and 2012, respectively. By year 2012, the reduction of forest core areas and the increase of other categories associated with fragmentation, such as islet, perforation, and edge, is evident.

$$\text{Percent Change} = \frac{\text{Fragmentation metric}_{2012} - \text{Fragmentation metric}_{2000}}{\text{Fragmentation metric}_{2000}} \times 100. \quad (1)$$

At the regional level, we calculated the percent change of forests for each of the main forest morphology classes (core, islets, perforation, and edge), the percent change of forest patches and mean forest patch area, the number of deforested patches, and the largest area of forest loss.

We performed generalized linear models (GLM) analyses on the matched sets of microlandscapes to detect the influence of forest protection schemes between (1) high-PES areas vs. low-PES areas, (2) high-PES areas vs. protected areas, and (3) high-PES areas vs. core zones within protected areas. Our model was

$$Y_{ij} = \beta_0 + \beta_1 \text{HIGH\_PES}_{ij} + \delta X_{ij} + \varepsilon_{ij} \quad (2)$$

where Y_{ij} is the fragmentation outcome for a given microlandscape i that is located in municipality j. HIGH_PES_{ij} represents the PES status of the microlandscapes, with this variable equal to 1 for high and 0 for low PES. $\mathbf{X}_{ij}$ represents a vector of the 10 covariate controls for each microlandscape that are used in matching (Table 3), and ε_{ij} represents the error. All specifications were run with robust standard errors and clustered at municipality level to account for possible spatial correlation. Clustering allows the errors to be correlated within municipality to take into account possible common patterns and avoid overstating the precision of estimates.

RESULTS

Forest morphology in Mexico

According to our morphology analysis for 2012, forests in Mexico were mainly core areas (67%) or edges (11%), while the other forest categories ranged only between 3% and 5%. Between 2000 and 2012, there were substantial changes in the proportion of the different forest morphology classes,

Table 2. Fragmentation metrics employed in this study.

Metric	Ecological relevance
Forest composition	
Core	Indicates the availability of forest interior, which is important for species that require large areas of forest cover (Watson et al. 2004)
Islet	Represents isolated forest patches that are small and generally with low potential to host species that require large habitat extents. They could potentially serve as stepping stone habitat (Estrada and Coates-Estrada 2002, Fischer and Lindenmayer 2002)
Perforation	Represents gaps within forest interior, which can bring edge effects inside of the forest interior areas (Leupin et al. 2004)
Edge	Border areas surrounding the forest interior and therefore exposed to external factors (Laurance 2000)
Bridge	Provide structural connectivity of interior forest habitat via corridors (Uezu et al. 2005)
Loop, Branch	These areas are generally subject to edge effects since they have a large perimeter:area ratio (Gascon et al. 2000)
Forest configuration	
Number of forest patches	Indicates the areas where forest species can exist. An increase in their number generally is an indication of forest fragmentation (Fahrig 2003)
Mean area of forest patch	Area available for forest species to live, a reduction in area could create forest patches that are not large enough to exclude edge effects (Urbina-Cardona et al. 2006)
Forest loss configuration	
Number of forest loss patches	Areas that have experienced forest canopy removal between 2000 and 2012. An increase in their number is a sign of reduction of habitat availability and the creation of edges (Broadbent et al. 2008)
Largest area of forest lost	This shows the largest patch area of forest lost between 2000 and 2012. A measure of the extent of habitat loss events

Table 3. Variables that were used to match high-PES microlandscapes vs. low-PES microlandscapes and high-PES microlandscapes vs. protected areas.

Variable	Source
Average slope	Shuttle Radar Topography Mission (USGS 2004)
Maximum slope	Shuttle Radar Topography Mission (USGS 2004)
Average elevation	Shuttle Radar Topography Mission (USGS 2004)
Maximum elevation	Shuttle Radar Topography Mission (USGS 2004)
Average distance to urban areas	Carta de localidades (INEGI 2010)
Maximum distance to major road	Carta topografica (INEGI 2000)
Maximum distance to any road	Carta topografica (INEGI 2000)
Average distance to streams	Carta topografica (INEGI 2000)
Average distance to country borders	World border map (ESRI 2015)
Percentage of forest cover in 2000	Global forest change (Hansen et al. 2013)

Notes: PES, payment for ecosystem services.

with most of the fragmentation happening in the east, central, and southern parts of the country (Fig. 4). Core areas were lost at particularly high rates in the Yucatan peninsula and along the Gulf of Mexico, while high changes in forest perforation were observed in the Yucatan peninsula. We found that a total of 21,700 km^2 of forest were lost, which accounted for 3.4% of the forest present in year 2000. The majority of this forest loss was within core forest areas (69%), while edges and branches contributed only 9% and 5%, respectively. As a consequence of this forest loss, forest core areas diminished by 9.3%. More than one-half of the forest core that was lost transitioned to other forest categories, such as perforation (24%) and edges (11%), while 40% became nonforest (Table 4). In the case of forest islets, 90% of their area remained, while 7% became deforested and the rest transitioned to other categories such as branches (1%). From the total forest classified as perforation in year 2000, 76.2% remained by 2012, and the rest was either lost (4.1%), or transitioned to forest edge (7.5%), core (5.7%), or to other categories. In the case of forest edges, 90.9% of their area remained, while 3.2% transitioned to nonforest, 1.7% to bridge, 1.4% to branch, 1.1% to core, and the rest to other categories.

Fragmentation according to the forest types

Forest fragmentation occurred at different intensities depending on the forest type. The forests that experienced the most forest loss were the mid- and high-stature evergreen and semievergreen tropical forests, which lost 7.4% and 7.3% of forest cover, respectively, followed by mid-stature deciduous and semideciduous tropical forest, which each lost 6.3%. This forest loss caused a core forest reduction of 17% for high-stature evergreen and semievergreen tropical forests, 13% for mid-stature evergreen and semievergreen tropical forests, and 8% for cloud forest. Some of this core forest that was lost transitioned to another forest category, such as islets, perforation, and edges, which increased between 20% and 94% in the same period (Fig. 5). The forest types that experienced the least forest loss were oak (−0.67%), conifers other than pine (−0.68%), mangrove (−1%), and pine (−1%). These forest types also presented the least reduction in core forest (between −2% and −4%) and the least increase of islets, perforation, and edge (<7%).

Fragmentation in PES areas

Based on the GLM results (Table 5), we generally rejected the null hypothesis that there is no difference in forest fragmentation between high-PES and low-PES microlandscapes.

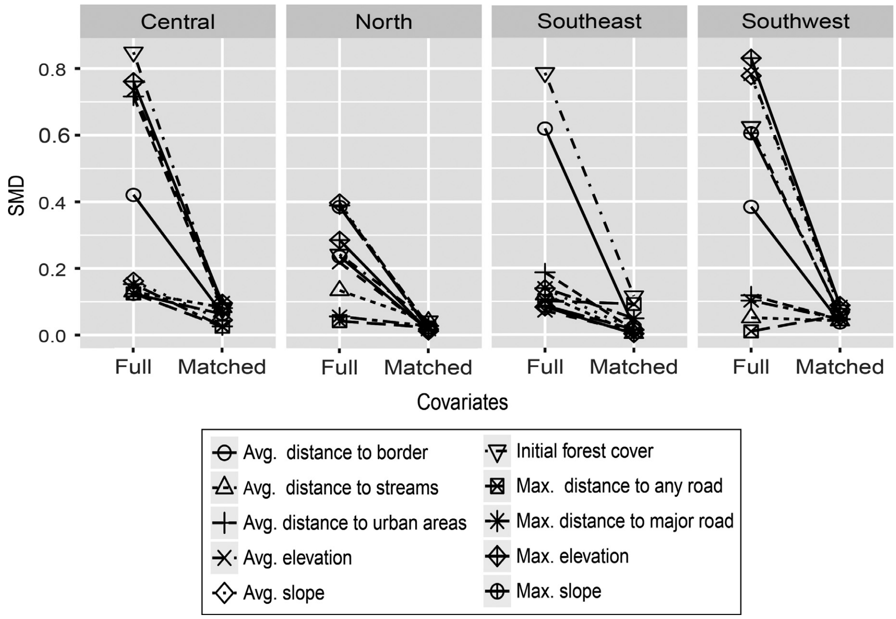

FIG. 3. Change in the standardized mean difference (SMD) before (Full) and after matching (Matched) microlandscapes. When the SDM value is below 0.1, the difference in the mean or prevalence of a covariate between treatment groups is small (Austin 2011). Avg., average; Max., maximum.

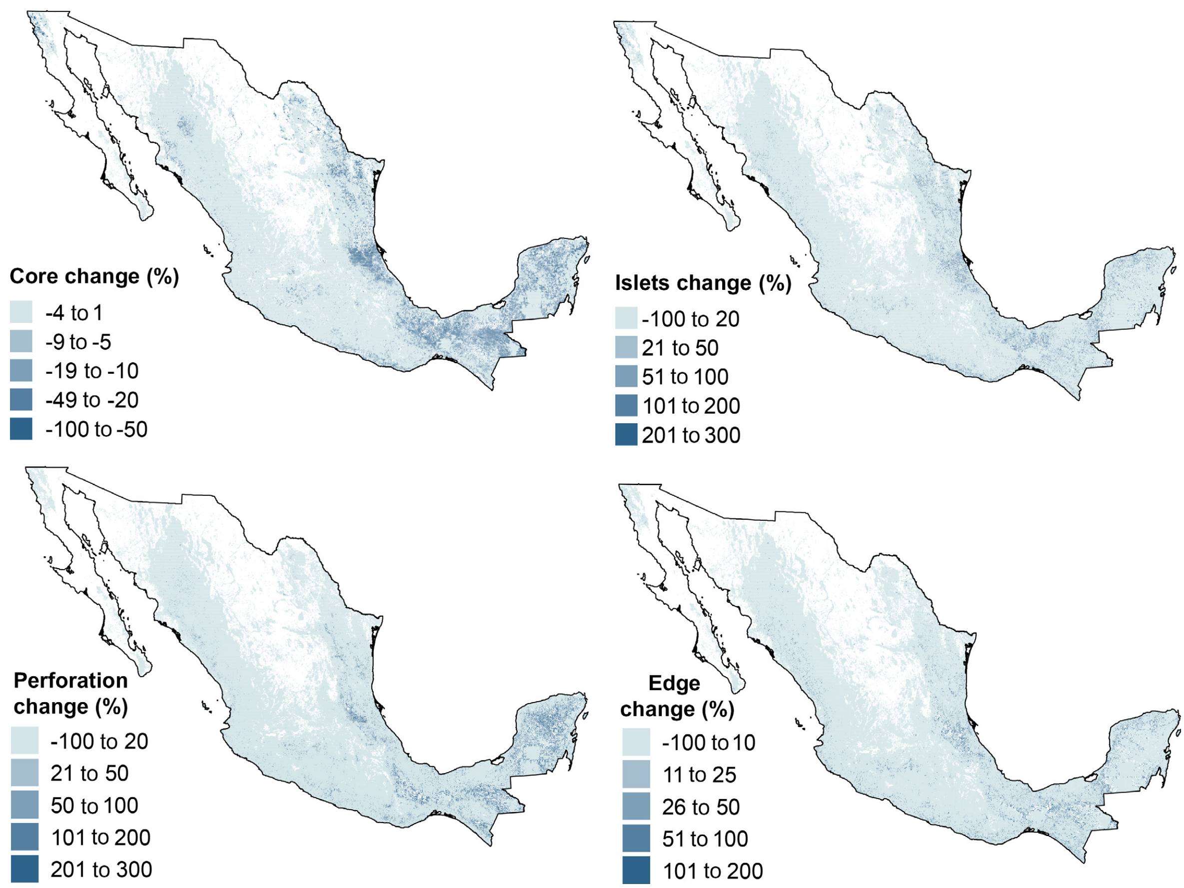

FIG. 4. Percent changes for the main forest morphology categories for all of Mexico between 2000 and 2012.

Table 4. Transitions between different forest morphology classes from 2000 to 2012 expressed as both the area in km^2 and the proportion in parentheses.

	Year 2012							
		Forest						
	Nonforest	Core	Islet	Perforation	Edge	Loop	Bridge	Branch
Year 2000								
Nonforest	1,416,297 (99.74)	1,651 (0.12)	521 (0.04)	175 (0.01)	508 (0.04)	174 (0.01)	300 (0.02)	402 (0.03)
Forest								
Core	15,028 (4.03)	338,028 (90.68)	325 (0.09)	8719 (2.34)	4311 (1.16)	2,544 (0.68)	2,424 (0.65)	1,381 (0.37)
Islet	1,042 (7.02)	47 (0.32)	13,479 (90.79)	5 (0.04)	73 (0.49)	23 (0.15)	31 (0.21)	146 (0.98)
Perforation	1,049 (4.18)	1431 (5.7)	62 (0.25)	19,138 (76.22)	1,904 (7.58)	649 (2.59)	500 (1.99)	375 (1.49)
Edge	1,883 (3.25)	674 (1.16)	280 (0.48)	151 (0.26)	52717 (90.89)	443 (0.76)	990 (1.71)	865 (1.49)
Loop	638 (5.24	490 (4.02)	90 (0.74)	167 (1.37)	269 (2.21)	9472 (77.77)	801 (6.58)	252 (2.07)
Bridge	910 (4.37)	377 (1.81)	122 (0.59)	56 (0.27)	414 (1.99)	306 (1.47)	18,173 (87.35)	447 (2.15)
Branch	1,150 (4.66)	184 (0.75)	550 (2.23)	44 (0.18)	248 (1.01)	113 (0.46)	207 (0.84)	22,195 (89.89)

At the national level, the postmatching microlandscapes with high-PES area had overall less forest fragmentation compared to those with low-PES. For instance, the number of forest patches increased more in low-PES (44.8%) compared to high-PES areas (25.8%; see Table 5 for the standard deviations of these changes, the differences in means, and the standard errors of the differences). According to the GLM model, which adjusts for potential remaining differences, in the absence of payments in currently high-PES areas, we would expect 20.6% more change in the number of patches. These differences were also observed in other fragmentation metrics, such as core forest, which decreased by 2.3% in high-PES areas while low-PES areas decreased by 3.6%. For other forest fragmentation categories, rates of changes were almost twice as high in areas with low-PES. For instance, forest islets increased on average by 13.3% for high-PES and 39.2% for low-PES, while forest edge increased by 40.8% in high-PES areas, but 80.7% in low-PES areas. The largest area of forest loss was also smaller for high-PES areas (1.8 ha) and larger in low-PES areas (3.3 ha). The average number of forest loss patches was 9.5 for high-PES and 12.6 for low-PES areas. These differences at the country level are generally similar to the ones estimated in the GLM models.

At the regional level, the effects of the PES program to prevent fragmentation in the matched microlandscapes were most clearly evident in the southwest and southeast (Fig. 6). For instance, in the southeast, forest core decreased almost twice as rapidly in low-PES (−7.1%) compared to high-PES areas (−4%). Forest islets increased four times more in low-PES (125.3%) compared to high-PES areas (33.3%). We found a similar trend where the number of forest patches increased 69% in high-PES areas while for low-PES areas the increase was 150.9%. Therefore, without PES incentives, we would expect an increase of 79% in the number of forest patches in the areas currently deemed as high PES. We would also expect larger forest loss events in these high-PES areas, as their largest area of forest loss averaged 4.9 ha, while for low-PES areas, it averaged twice this amount (10.7 ha). However, we found less fragmentation-limiting effects of the PES in the central and north regions.

Fragmentation in protected areas

There were no evident effects of PES in preventing forest fragmentation compared to areas within the National Protected Areas System in Mexico (Fig. 6). Similarly, there were no significant effects of PES in change of forest edge, perforation, and islets, between the PES and the core zones of biosphere reserves. However, core zones of biosphere reserves had on average less forest core loss (0.9%), a smaller decrease forest patch area (2.6%), and fewer deforested patches (4.8). In the case of high-PES areas, the same categories changed 2.1%, 4.6%, and 8.5, respectively.

Discussion

We found that forest became more fragmented in Mexico from 2000 to 2012 and that tropical forest in particular lost a large proportion of its core forest areas, and increased forest edges. However, high-PES areas had lower rates of forest

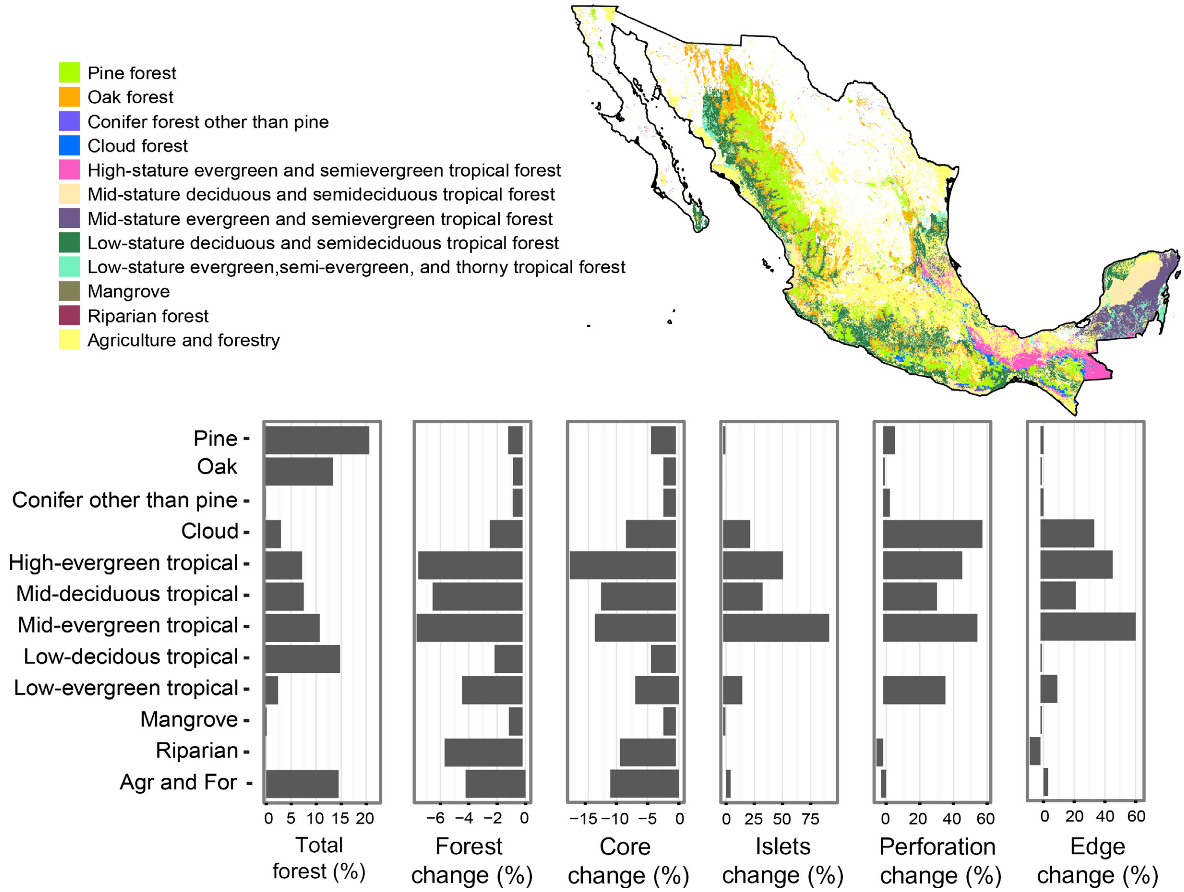

FIG. 5. Top: Map of vegetation types in Mexico, out of which we analyzed all forests types. Bottom: Forest morphology changes in each forest type (proportion of the total of forest per category in 2000). Agr and for, Agriculture and forestry.

fragmentation, particularly in the southeast and southwest regions, where most of the forest loss occurred. Compared to protected areas, high-PES performed similarly well in terms of preventing forest fragmentation. However, when compared to core zones of biosphere reserves, high-PES areas performed less well and did not match the effectiveness of protection of these most strictly protected areas.

Forest loss is widespread in Mexico, as there is constant pressure for new agriculture and housing development (Calderon-Aguilera et al. 2012). Despite that, forest loss rates have decreased in Mexico (Roth et al. 2016). We found that the majority of forests in Mexico were core forests (69%), which is also where the majority of forest loss happened (69.2% of the total area that lost forest cover). Hence, forest loss in the country occurred both in forest interior and along forest edges. However, the high proportion of core forest reflected to some extent our choice to apply a 30-m edge (Ostapowicz et al. 2008). While this is the area where edge effects are most pronounced (Tuff et al. 2016), some forest edge effects can penetrate up to 100 m into the forest, or even more (Laurance et al. 2007, Briant et al. 2010). If we had chosen a 100-m edge for our analysis, the core areas would comprise only 44% of the total forest in the country, and the observed forest composition transitions would have occurred mostly at the forest edge category. Our findings that the net forest loss was 3.4%, and the loss of core forest was 9.3% are similar to global fragmentation trends reported for the same period, in which the net forest loss was 3.2% and the forest interior was reduced in 9.9% (Riitters et al. 2016). This highlights that even when forest loss rates are low, fragmentation may proceed rapidly. In our analysis, half of the forest core that was lost transitioned to nonforest, while the other half transitioned to another forest category. As a result, forest categories associated with fragmented forests (Ostapowicz et al. 2008, Rogan et al. 2016), such as perforation, loop, bridge, and edge all increased 16–17% across the country, while the number of forest patches increased 27.9%. This additional forest fragmentation can have positive effects (Fahrig 2017), such as increasing habitat for certain generalist species (Carrara et al. 2015), but it may also have negative effects such as shifts in thermal gradients affecting species (Tuff et al. 2016), increasing carbon release to the atmosphere (Brinck et al. 2017), and promoting changes in species composition (McDonald and Urban 2006, Barlow et al. 2007, Herrerías-Diego et al. 2008, Haddad et al. 2015).

Among forest types, tropical forests experienced the most forest changes in Mexico. High- and mid-stature tropical forest had the highest rates of forest core loss (17% and 13%, respectively) and also the highest edge gain (47% and 62%). These high fragmentation rates are likely due to the close proximity of tropical forest to human disturbances in

TABLE 5. Percent change in different fragmentation metrics for high-PES vs. low-PES microlandscapes.

Fragmentation metric and zone (1,2)	High-PES change (%)†	Low-PES change (%)†	Difference (%)‡	GLM Estimated difference§	GLM Robust SE¶	GLM *P*#
PES enrolled vs. all forested areas						
Core change						
Country (2972, 2654)	−2.34 (4.9)	−3.59 (6.65)	1.25 (0.15)	1.26	0.24	<0.001
N (713, 654)	−0.85 (2.46)	−0.98 (2.55)	0.13 (0.13)	0.12	0.13	0.361
C (764, 670)	−1.98 (5.63)	−1.89 (4.43)	−0.09 (0.26)	−0.23	0.44	0.605
SW (779, 707)	−2.5 (4.17)	−4.41 (6.3)	1.91 (0.28)	1.84	0.36	<0.001
SE (716, 583)	−4.03 (5.97)	−7.14 (10.79)	3.11 (0.47)	2.93	0.53	<0.001
Islet change						
Country	13.3 (155.5)	39.2 (351)	−25.9 (7.38)	−26.03	8.24	0.002
N	3.05 (65.41)	13.15 (153.17)	−10.1 (6.47)	−9.86	5.22	0.058
C	6.74 (48.95)	8.28 (102.71)	−1.54 (4.34)	−1.64	3.83	0.669
SW	11.15 (133.99)	27.78 (318.49)	−16.63 (12.9)	−15.91	10.43	0.127
SE	33.34 (271.23)	125.36 (789.99)	−92.02 (34.28)	−83.33	32.29	0.01
Perforation change						
Country	100.98 (524.66)	115.06 (482.65)	−14.08 (13.43)	−18.27	15.43	0.236
N	14.43 (111.4)	12.87 (86.28)	1.56 (5.26)	0.88	5.45	0.871
C	72.29 (490.23)	38.4 (174.16)	33.89 (18.3)	31.17	21.33	0.144
SW	101.98 (314.83)	190.87 (689.75)	−88.89 (28.29)	−92.69	27.20	0.001
SE	216.72 (875.14)	259.92 (1071.19)	−43.2 (55.16)	−51.32	43.43	0.237
Edge change						
Country	40.88 (632.47)	80.7 (865.38)	−39.82 (20.41)	−39.14	22.79	0.086
N	3.73 (24.07)	20.73 (171.82)	−17 (6.78)	−17.33	6.51	0.008
C	32.74 (353.4)	44.34 (335.53)	−11.6 (18.21)	−7.17	25.98	0.782
SW	40.12 (331.12)	44.61 (294.61)	−4.49 (16.24)	−2.39	19.07	0.9
SE	87.39 (1185.46)	178.82 (1057.21)	−91.43 (62.33)	−82.95	51.54	0.107
No. forest patches change						
Country	25.79 (147.54)	44.81 (245.53)	−19.02 (5.41)	−20.58	6.58	0.002
N	3.16 (32.06)	4.74 (36.23)	−1.58 (1.85)	−1.61	1.99	0.418
C	16.24 (186.24)	9.53 (57.66)	6.71 (7.1)	6.53	6.45	0.311
SW	16.09 (92.14)	23.94 (96.57)	−7.85 (4.9)	−7.03	5.30	0.185
SE	69.09 (201.32)	150.97 (409.74)	−81.88 (18.57)	−79.58	20.79	<0.001
Forest patch area change						
Country	−5.26 (27.12)	−6.65 (31.74)	1.39 (0.79)	1.53	0.95	0.108
N	−0.49 (12.7)	−1.82 (11.11)	1.33 (1.04)	1.33	0.72	0.064
C	−3.76 (17.33)	−2.33 (21.73)	−1.43 (0.64)	−1.61	1.02	0.116
SW	−3.94 (26.04)	−6.94 (27.18)	3 (1.38)	2.81	1.45	0.054
SE	−13.08 (41.82)	−18.79 (57.76)	5.71 (2.85)	5.49	3.16	0.083
No. forest patches lost						
Country	9.52 (14.05)	12.69 (17.02)	−3.17 (0.41)	−3.09	0.65	<0.001
N	4.01 (6.55)	4.93 (9.02)	−0.92 (0.43)	−0.89	0.44	0.042
C	7.99 (16.09)	8.15 (13.52)	−0.16 (0.78)	0.37	1.41	0.79
SW	11.43 (12.26)	17.24 (17.79)	−5.81 (0.8)	−5.48	1.00	<0.001
SE	14.57 (16.58)	19.91 (20.47)	−5.34 (1.05)	−4.50	1.21	<0.001
Largest forest loss area						
Country	1.87 (6.8)	3.31 (11.09)	−1.44 (0.24)	−1.54	0.41	<0.001
N	0.54 (3.16)	0.5 (1.42)	0.04 (0.13)	0.04	0.16	0.77
C	0.98 (6.36)	1.1 (4.68)	−0.12 (0.29)	−0.12	0.28	0.659
SW	1.15 (2.54)	1.85 (4.16)	−0.7 (0.18)	−0.65	0.20	<0.001
SE	4.9 (10.93)	10.73 (19.88)	−5.83 (0.92)	−5.56	1.39	<0.001
PES enrolled vs. protected areas						
Core change						
All PA (1866, 1226)	−2.04 (0.11)	−2.01 (0.13)	−0.03 (0.17)	0.17	0.27	0.531
Core PA (1626, 358)	−2.11 (0.12)	−0.95 (0.16)	−1.16 (0.2)	−1.03	0.30	0.001
Islet change						
All PA	9.42 (3.29)	14.39 (4.66)	−4.97 (5.71)	−4.08	6.19	0.509
Core PA	9.19 (3.5)	5.79 (2.84)	3.4 (4.51)	7.16	7.22	0.322
Perforation change						
All PA	102.61 (14.12)	68.71 (10.74)	33.9 (17.75)	12.53	22.94	0.585
Core PA	108.37 (16.05)	60.25 (11.38)	48.12 (19.63)	43.77	28.92	0.13

TABLE 5. (Continued)

Fragmentation metric and zone (1,2)	High-PES change (%)†	Low-PES change (%)†	Difference (%)‡	GLM		
				Estimated difference§	Robust SE¶	P#
Edge change						
All PA	38.2 (16.79)	40.15 (18.20)	−1.95 (24.78)	−0.32	26.23	0.99
Core PA	39.56 (19.07)	31.62 (22.62)	7.94 (29.62)	15.54	40.76	0.703
No. forest patches change						
All PA	22.23 (3.67)	19.65 (3.15)	2.58 (4.84)	−0.68	4.42	0.877
Core PA	21.7 (2.88)	16.71 (6.81)	4.99 (7.4)	6.11	7.53	0.417
Forest patch area change						
All PA	−4.05 (0.57)	−3.85 (0.62)	−0.2 (0.85)	0.45	0.95	0.631
Core PA	−4.62 (0.58)	−2.55 (1.2)	−2.07 (1.34)	−2.13	1.16	0.068
No. forest patches lost						
All PA	8.38 (0.31)	7.92 (0.35)	0.46 (0.46)	0.18	0.83	0.882
Core PA	8.56 (0.34)	4.87 (0.41)	3.69 (0.53)	3.47	0.98	<0.001
Largest forest loss area						
All PA	1.53 (0.15)	1.86 (0.23)	−0.33 (0.28)	−0.79	0.37	0.036
Core PA	1.53 (0.13)	1.29 (0.29)	0.24 (0.32)	0.11	0.43	0.785

Note: Numbers in parentheses in first column are (1) sample size for matched high-PES microlandscapes and (2) sample size for matched low-PES microlandscapes.

†Mean percent change and standard deviations for the fragmentation metrics, except for number of forest loss patches and largest forest loss area, which reflect the absolute values of forest loss events that occurred between 2000 and 2012.

‡Values are means with SE in parentheses.

§β_1 coefficients for the high-PES treatment of the microlandscapes, following Eq. 2.

¶Robust standard errors clustered by municipalities. The number of municipalities within each matched sample is north, 399; central, 248; southwest, 272; southeast, 85; protected areas, 399; core zones in protected areas, 317.

#The Bonferroni correction for multiple tests is $P = 0.001$ ($\alpha = 0.05$).

Mexico, where only 12% of its forests are isolated enough to avoid such influences (Moreno-Sanchez et al. 2012). The high fragmentation levels that we found are a concern because tropical forest provides multiple ecosystem services and therefore are of priority for protection (Myers et al. 2000). Cloud forest is another forest type that lost a large proportion of forest core areas (11%). Although cloud forest represents only a small proportion of the total forest in Mexico (3.5%), it is a conservation priority because it is high in biodiversity and provides important ecosystem services such as water catchment (Martínez et al. 2009). On the opposite side of the spectrum, forest types that are fairly widespread, such as pine forest (20%), low-stature deciduous forest (15%), and oak forest (13%), had the least forest core loss (4%, 4%, and 2%, respectively). These forest types are relatively resilient to fragmentation, because they harbor more edge-adapted species, but they can still be affected by invasive species (Harper et al. 2005). Our observed forest fragmentation in temperate forest is consistent with trends seen in other countries, where these forest types had less forest loss, or even expanded (Chazdon 2008). In Mexico, a large proportion of pine forest (23%) is isolated from human influences, thus preventing its forest loss (Moreno-Sanchez et al. 2012).

The PES program in Mexico has reduced the expected rate of forest cover change in the enrolled properties between 40% and 51% (Alix-Garcia et al. 2015), and we found a concomitant reduction in fragmentation within high-PES areas. Our analysis indicated that the PES program had only minor effects in preserving forest in the north, perhaps because of low access and low population densities near forests, which resulted in low rates of fragmentation regardless of whether forests were enrolled in the PES program or not. In the south of Mexico, however, the PES has performed well in terms of preventing fragmentation, particularly in the southeast. Part of this reduction in fragmentation is because the Mexican authorities have increased the scope of the PES program and the targeting of high-risk deforestation areas over time (Sims et al. 2014). We had expected that the south would exhibit higher rates of fragmentation, given the high deforestation risk due to physical geographic conditions, high poverty levels, and high population density (González-Abraham et al. 2015). Southern states in Mexico have also increased small-scale corn production to compensate for income loss produced by policies such as the North American Free Trade Agreement, NAFTA, increasing deforestation in areas that are not very suitable for agriculture (Vilas-Ghiso and Liverman 2007, Keleman 2010, Soto 2012). On the other hand, agriculture in northern Mexico tends to be more industrialized, has larger yields, and is largely restricted to low-slope areas, therefore reducing forest loss pressure in mountain ranges.

Protected areas have different outcomes in preventing deforestation across the world (Naughton-Treves et al. 2005, Laurance et al. 2012), but we found positive effects in Mexico, similar to other studies (Sims and Alix-Garcia 2017). Protected areas can have conservation management and funding that facilitates forest monitoring, enforcement, and planning (Blackman et al. 2015). In our analysis, the PES program performed similarly well as protected areas in terms of preventing forest fragmentation. The effects of protection of the PES program, however, were lower compared to core zones of biosphere reserves. Because most types of human activities are restricted in core zones of biosphere reserves, and there are very few human settlements on them, we used them as a proxy to differentiate natural (e.g.,

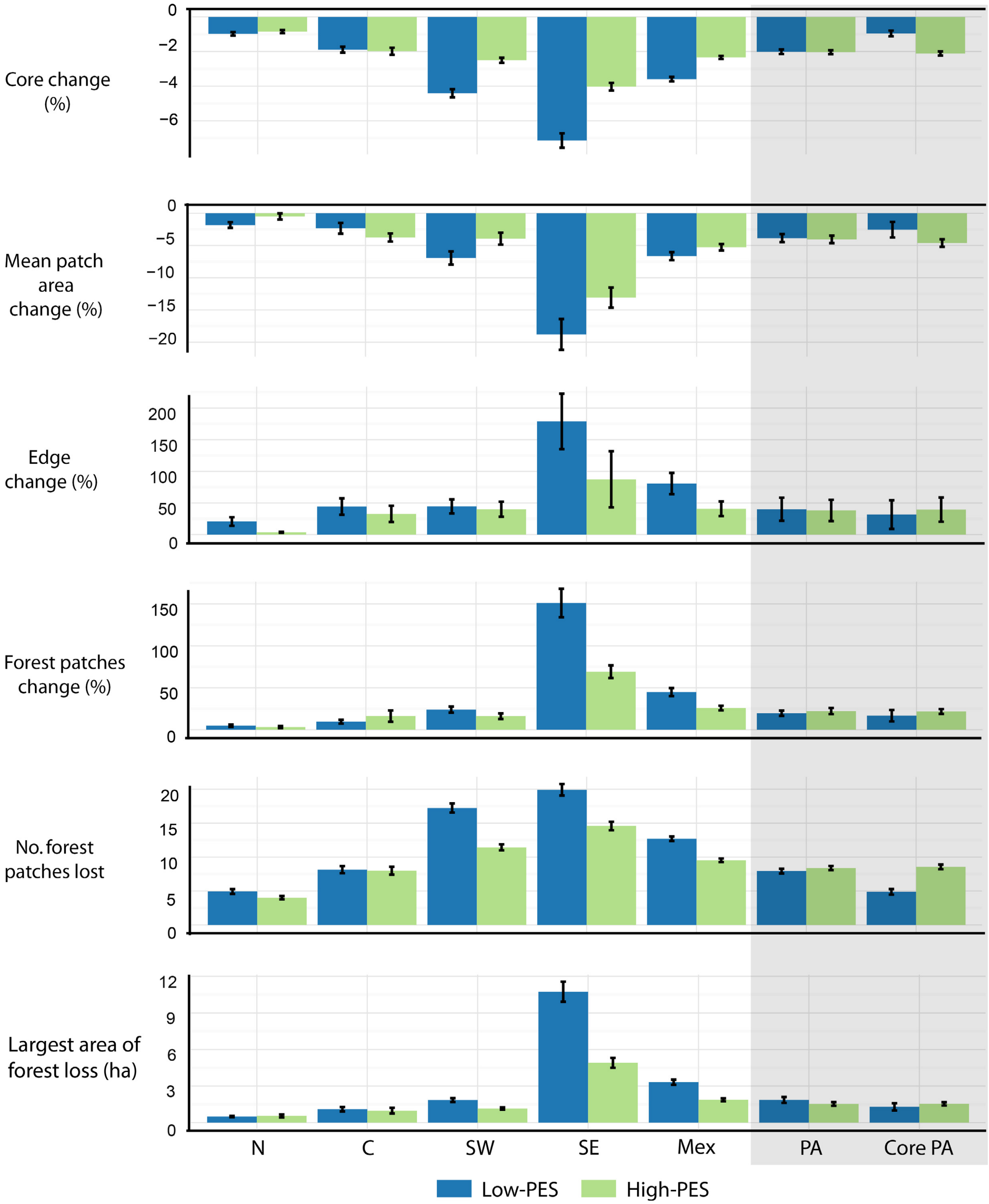

FIG. 6. Postmatching comparisons between high-PES and low-PES microlandscapes (PES, payments for ecosystem services). We calculated percent changes in fragmentation metrics from 2000 to 2012 and conducted the analysis for each zone (N, north; C, central; SW, southwest; SE, southeast; Mex, whole country; PA, protected areas; Core PA, core zones within protected areas). Shaded areas correspond to comparisons made between high-PES and low-PES within protected areas. Shown are the means and standard error bars.

hurricanes and fires) vs. human forest disturbances (e.g., logging and agriculture expansion). We assumed that the fragmentation rates observed in core zones are mostly due to natural disturbances and higher rates elsewhere would be due to human disturbances. As most of the forest fragmentation changes in PES are higher than in core zones, we assume that PES protection decreased human disturbances, but did not stop them entirely.

It is important to note that the majority of forest in Mexico are owned and managed by communities (*comunidades* and *ejidos*). This collective tenure allows shareholders to farm and manage forest on family lots or in forest held in common

(Corbera and Estrada 2010). Since the 1980s, these communities have rights to manage their local resources under state regulation, which in the past were assigned to concessionaries. Local institutional and environmental laws have also strengthened in recent decades, allowing local communities to improve their forest management. Indeed, communal property in Mexico can be an option for sustainable forest management and governance because they restrict their consumption of forest products in common properties (Bray 2013) and offer an alternative to efforts to halt deforestation that are more centralized (Merino 2016). Nevertheless, many communities in Mexico are marginalized, with low access to resources and education, increasing the risk of deforestation (Alix-Garcia 2007, Morales-Barquero et al. 2015). This highlights the importance of forest protection incentives such as PES to support conservation efforts.

Our results show that Mexico's PES is an important instrument to reduce forest fragmentation. We control for many other differences between landscapes with high- and low-PES enrollment in order to isolate the relationship between PES and fragmentation outcomes. It is still possible that other biophysical and socioeconomic variables could explain the observed relationship, or could augment or diminish the PES protection effects in particular regions (Lopez-Feldman 2012). Such variables include enrollment to government assistance programs such as agriculture incentives, poverty, or migration (Lopez et al. 2006). Given these complex relationships, policy impacts of PES and PAs cannot necessarily be generalized across space and time (Lambin et al. 2003). However, studies like ours highlight at the aggregated level how to evaluate the impacts of large-scale forest protection programs.

Other forest fragmentation assessments around the globe have not yet been related to PES, but there are studies that measure the impact of these incentives in reducing deforestation, particularly in Costa Rica, Colombia, and Mexico (Pattanayak et al. 2010, Börner et al. 2017). Measuring deforestation in relation to PES is more common in developing countries, as developed countries target PES programs to agricultural landscapes (Schomers and Matzdorf 2013). However, there are few studies that compare PES areas with controls to evaluate the impact of the payments in forest cover (Samii et al. 2014). In Costa Rica, there has been small evidence that the PES reduced deforestation compared to areas that did not receive payments (Robalino et al. 2008, Arriagada et al. 2012). In Uganda, villages enrolled in PES reduced the expected tree cover loss by half (Jayachandran et al. 2017). Similarly, the PES in Quindio, Colombia, has been successful in promoting environmentally beneficial land uses (Pagiola et al. 2016). The lack of studies on fragmentation highlights the importance of forest change analyses like ours to assess the impact of PES to prevent forest fragmentation.

Conclusion

To our knowledge, this is the first study that used Hansen et al.'s (2013) forest change data to assess forest fragmentation differences due to a forest policy intervention. We found that Mexico's PES program worked well to reduce forest fragmentation, particularly in the southwest and southeast parts of the country, where forest patches and edges have doubled or tripled in areas that were not enrolled, and where tropical forests were lost and fragmented. Mexico's forest loss affected especially forest core areas, which either transitioned to edge or to nonforest. Compared to other protection schemes such as protected areas, the PES program performed similarly well as protected areas in limiting forest fragmentation, showing that both represent similarly effective alternative approaches for forest protection. This is encouraging, because both PES and protected areas are both key strategies for achieving reduced deforestation globally. The data that we used, that is, the global analysis for deforestation is available worldwide and updated annually (Hansen et al. 2013), which means that forest fragmentation assessments could, and possibly should, be conducted elsewhere in order to monitor the effects of programs aimed to prevent deforestation under REDD+ initiatives.

Acknowledgments

We thank the Mexican National Forest Commission, CONAFOR, for allowing us to use their PES data. We also thank 3ie, NSF (grant SES-1061852), the NASA Earth System Science Fellowship Program (grant NNX12AO07H), the Carnegie Fellows Program, and our institutions for their support of this work. We also thank the valuable comments from J. Alix-Garcia and four anonymous reviewers.

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Data Availability

Data available from Figshare: https://doi.org/10.6084/m9.figshare.c.3915814.v1.