Environmental Conservation

cambridge.org/enc

Protecting forests, losing trees: the role of community involvement

José López-García[1], Gustavo Manuel Cruz-Bello[2] and Lilia de Lourdes Manzo-Delgado[1]

[1]Instituto de Geografía, Universidad Nacional Autónoma de México, Ciudad Universitaria, Mexico City, Mexico and [2]Departamento de Ciencias Sociales, Universidad Autónoma Metropolitana, Unidad Cuajimalpa, Mexico City, Mexico

Research Paper

Cite this article: López-García J, Cruz-Bello GM, Manzo-Delgado Lde L (2025) Protecting forests, losing trees: the role of community involvement. *Environmental Conservation* page 1 of 10. doi: 10.1017/S0376892925000128

Received: 14 October 2024
Revised: 5 March 2025
Accepted: 5 March 2025

Keywords:
Forest-change processes; forest-cover density; Monarch Butterfly Biosphere Reserve; payment for environmental services; protected area performance

Corresponding author:
Gustavo Manuel Cruz-Bello;
Email: gcruz@cua.uam.mx

Summary

This study aims to evaluate the effectiveness of the Monarch Butterfly Biosphere Reserve (MBBR) in preventing forest degradation and land-use changes within communal lands in the context of limited public consultation and the transformation of communal forest governance. We analysed forest-cover changes over 50 years using a multi-temporal approach, integrating aerial photographs, orthophotographs and satellite imagery. We obtained contextual knowledge through our long-term research engagement with the region and interviews conducted during participatory fieldwork. We analysed land-use changes in a watershed within the MBBR in Mexico before and after its designation as a protected area. Despite the reserve's protected status, nearly half of the study area experienced forest-cover changes. Surprisingly, the most intense deforestation occurred after conservation decrees, as some communities engaged in pre-emptive forest clearing in response to anticipated restrictions. However, in later periods, forest recovery – driven by payment for environmental services, natural regeneration and community participation – began to outpace degradation. Nonetheless, the fir forest that is essential for monarch butterfly habitat was reduced by 43.3%, with illegal logging being one of the leading causes. This study highlights the importance of community involvement when establishing protected areas, as it can help reduce environmental impacts and ensure conservation success.

Introduction

Protected areas can reduce forest loss and help conserve biodiversity and ecosystem services (Wade et al. 2020, Liu et al. 2022), supporting national and international conservation strategies with the backing of governments and institutions such as the Convention on Biological Diversity (Hu et al. 2022). Forest extent has increased by 191 million ha globally within these areas since 1990 (FAO 2020). However, these protected areas face threats, including land-use change, invasive species, social and political issues, institutional barriers, incomplete legislation, inadequate public funding, habitat loss, pollution, diseases, climate change and conflicts with human rights, development and poverty reduction (Brenner 2006, He & Cliquet 2020, Hoffmann 2022). This situation is made worse by some protected areas being designed and managed based primarily on physical and biological factors (Stewart & Possingham 2005). However, natural resource conservation must also consider the local human population and the socioeconomic environment (López-Barrera et al. 2010, Oldekop 2015, Zhang et al. 2020, Kegamba et al. 2022). Without these considerations, conservation policies may result in restrictions on using those natural resources that are essential to the livelihoods of local inhabitants. This may provoke resentment and actions against conservation, accelerating the degradation of the natural resources they were intended to conserve (Honey 1999, Kegamba et al. 2022). Thus, particular attention must be given to improving public participation and awareness (He & Cliquet 2020) and the potential conflict with poverty reduction if local people are displaced or their access to natural resources is restricted by regulations (Hoffmann 2022). This conflict underscores the need for careful planning and the consideration of social implications in conservation efforts. Conflicts between local people and protected areas are common in developing countries (Maikhuri et al. 2000). Thus, creating new protected areas requires thorough justification and implementation, accounting for any impacts on surrounding communities (McGinlay et al. 2023). While past protected areas failed to recognize local rights, new approaches must actively promote people's participation in decisions affecting them (Iannuzzi et al. 2020).

In Mexico, protected areas for biodiversity grew by 8.1% annually from 1990 to 2015, adding 962 000 ha annually. Forest cover increased from 3.8 to 8.8 million ha between 2000 and 2015 (FAO 2015). As in other Latin American or Sub-Saharan countries (Lockwood 2010, Antoni et al. 2019), these protected areas were established on communal lands owned and managed

CAMBRIDGE UNIVERSITY PRESS

collectively by local communities. This communal land tenure situation introduces additional complexity, as protected areas on these lands restrict traditional resource use and alter local governance and community participation in forest management (Elías 2012).

In the Monarch Butterfly Biosphere Reserve (MBBR) in Mexico, the creation decree and subsequent modifications did not change the communal land ownership model (*ejidos* and communities). However, these decrees did limit the use of communal lands and forests and restricted traditional agricultural practices, reducing local control over the land and impacting local livelihoods (Brenner 2009, Honey-Rosés 2009, Merino-Pérez & González-Duarte 2021). Although the *ejido* is a solid land tenure system, deforestation can still occur in these communally owned areas due to disputes over land rights, illegal logging and decisions made by the *ejidal* assembly, which controls the use of forest resources (Duran et al. 2011, Brenner & San German 2012, Morett-Sánchez 2017).

The decrees establishing the monarch butterfly overwintering and reproduction areas significantly transformed the region's socioeconomic and forestry dynamics. The 1980 decree designated these areas as a reserve and wildlife refuge but did not specify the area to be protected or activities to be restricted (DOF 1980). The 1986 decree declared protected areas for butterfly migration, overwintering and reproduction, covering 16 110 ha in Estado de México and Michoacán. The core zone enforced a complete and indefinite ban on logging and exploiting flora and fauna, while the buffer zone enforced only temporary restrictions (DOF 1986). The third decree of 2000 expanded the protected area to 56 269.05 ha, designating it as a Biosphere Reserve. The core zone permitted only preservation, research and environmental education, prohibiting the exploitation of forests and other flora and fauna. The buffer zone allowed logging and harvesting with authorization from the environmental agency (DOF 2000).

Between 1971 and 1999, before the area was designated as a Biosphere Reserve, the forest experienced severe fragmentation and degradation (Brower et al. 2002). This was particularly pronounced when some agrarian communities cleared their forests upon learning of the second decree (WWF 2004). A field visit in 1987 revealed illegal logging of the fir trees favoured by the butterflies; in defiance of the decree, the communities responded by stating, 'When they come, they will find nothing left.' In response to the third decree, illegal logging intensified (López-García et al. 2022), with communities burning and logging their lands within the common property or even sabotaging their neighbours' lands in protest against the restrictions (Carbale et al. 1997). This pre-emptive behaviour has also been observed in northern Mexico (Blackman et al. 2015) and other countries such as Madagascar (Llopis et al. 2019), Argentina (Nolte et al. 2018) and Brazil (Brown et al. 2016). In this context, 'pre-emptive behaviour' describes an intensification of resource extraction activities undertaken in anticipation of the enforcement of stricter land-use regulations (Nolte et al. 2018, Llopis et al. 2019).

The 2000 proposal had included part of the Francisco Serrato *ejido* in the new core zone; however, after negotiations, the entire *ejido* was left in the buffer zone. In response, the *ejido* logged almost all of its land between 2001 and 2003 (WWF 2004). By 2012, 1254 ha of the core zone had been deforested, 925 ha had been degraded and 122 ha had been affected by climatic events (Vidal et al. 2013). After 2012, logging was minimal, but in November 2015, 10 ha of well-preserved forest were cleared (Brower et al. 2016). Between 1994 and 2017, 10.5% of the core zone was degraded and 10% was deforested, while in the buffer zone 8.2% was degraded and 5.1% was deforested (López-García et al. 2022).

The limitations and prohibition of productive activities generated changes in forest-cover density, including the accumulation of leaves, branches, grasses and dry trunks, which became potential fuel for forest fires (Tucker 2004). Fire prevention and suppression programmes have been implemented since the Reserve was created (CONANP 2001), and a fire management plan has also been created (Pérez-Salicrub et al. 2017).

Along with the MBBR, a conservation fund was established in 2000 in Mexico through payments for environmental services (PES) to incentivize forest protection (Honey-Rosés 2005, Missrie & Nelson 2005). Participation in the PES programme increased over time, with 78.5% of the core zone enrolled between 2000 and 2006, rising to 89% by 2009 and to 96.5% by 2018. Currently, 91.3% of the area receives PES, while 5.2% is government-owned and so is not included (MBF 2021). In addition, the National Forestry Commission and Protectora de Bosques implemented hydrological PES programmes in the region in 2004 and 2007, respectively (Alix-Garcia et al. 2009, Probosque 2024), significantly contributing to forest recovery. Enrolment in the PES programme has been a success, possibly because it has been reported that it strengthens cooperation and collective action within and between communities, and communities consider the payments to be fair despite the programme having only a small positive impact on poverty reduction (Nieratka et al. 2015).

This study aims to determine how well the MBBR functions as a conservation mechanism to prevent land-use change and degradation of forest cover in the context of communal lands and a lack of public consultation and involvement. We conducted a multi-temporal analysis using aerial photographs, orthophotographs and satellite imagery to assess forest-cover changes over 50 years, starting 5 years before the protected area was first decreed. The analysis considers possible interrelationships between community actions and conservation outcomes.

Materials and methods

Study area

The La Hacienda watershed is in the central portion of the MBBR, spanning the border between Estado de México and Michoacán. It covers an area of 1818.64 ha, with 33% located in the core zone and 67% in the buffer zone (Fig. 1). This mountainous region has an altitude ranging from 2740 to 3640 m. Communal land tenure comprises seven *ejidos* and two Indigenous communities, with four *ejidos* occupying 95% of the watershed (El Rosario, La Mesa, Francisco Serrato and El Paso). The *ejido* El Rosario covers 53.4% of the watershed, with 5.4% in the core zone and 48% in the buffer zone. The regional economy is based on agriculture, forestry, aquaculture and tourism. The watershed's headwaters include the *ejidos* Francisco Serrato (12%) to the south and La Mesa (23%), located within the Reserve's core zone to the west.

Image preprocessing

Panchromatic aerial photographs from 1971 and 1984, orthophotographs from 1994, SPOT satellite images from 2004 and 2015 and Google Earth imagery (Google Earth ©) from 2021 were used. All of these materials were from January to March. Orthomosaics were constructed from the 1971 and 1984 aerial photographs, with spatial resolutions of 1.7 and 1.6 m/pixel and mean square errors of

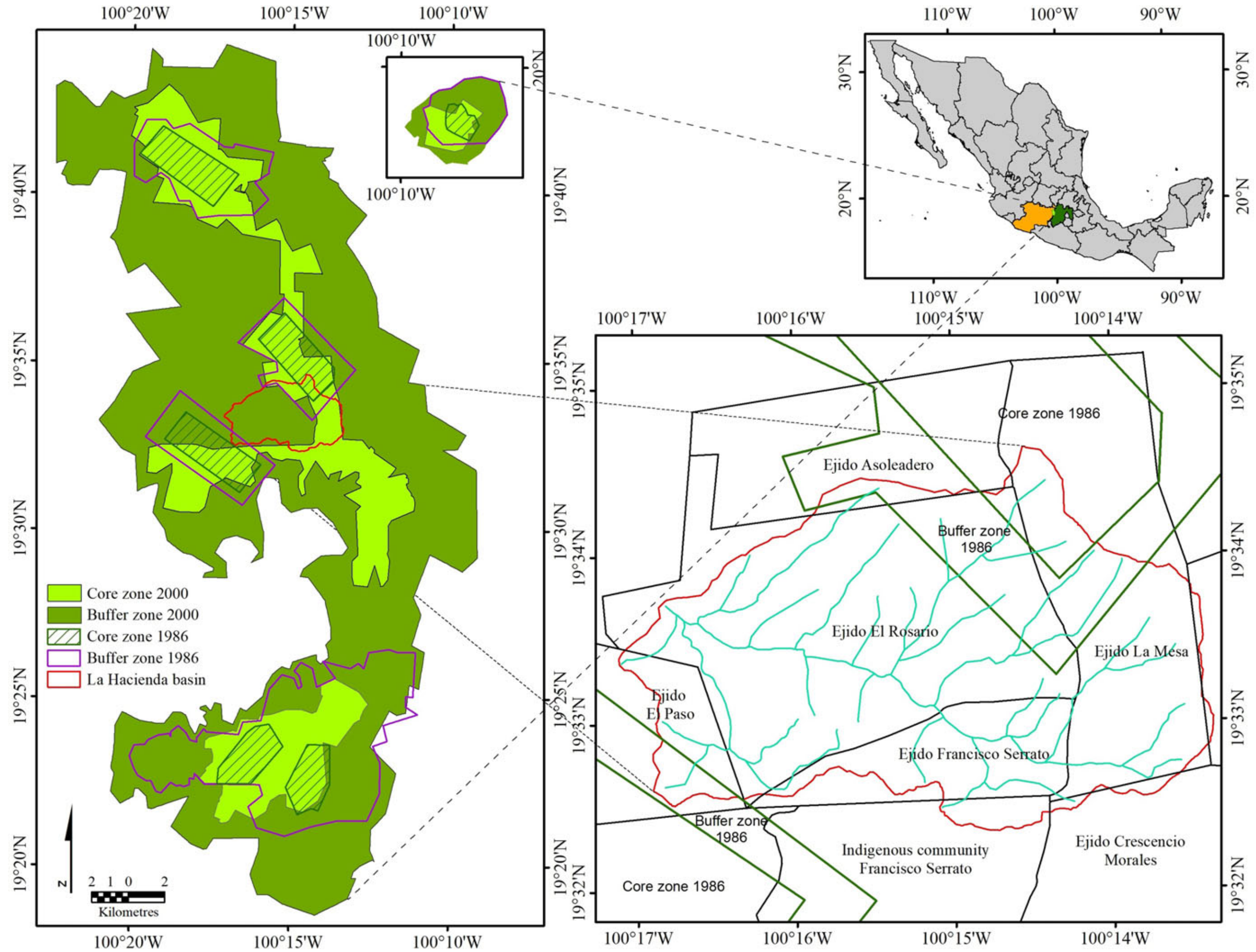

Figure 1. Location of La Hacienda watershed within the Monarch Butterfly Biosphere Reserve.

2.5 and 2.8 m, respectively. A digital elevation model with 20-m contour lines and 1994 digital orthophotographs with 2-m spatial resolution were used as supplementary data for image georeferencing. The SPOT images were radiometrically and geometrically corrected, and the multispectral image was combined with the panchromatic image to improve spatial resolution. The spatial resolutions of these materials ranged from 1.5 to 2.5 m (scale 1:5000), which enabled an adequate comparative analysis with a minimum mappable area of 625 m^2 (5 × 5 mm) for deforested and non-forest areas (Salichtchev 1979) and more than 5000 m^2 for forests (FRA 2000).

Image interpretation

Five levels of forest-cover density (separation of the treetops) were identified through image interpretation: areas with <10% were classified as deforested (<50 trees/ha; FAO 2010), those between 11% and 25% were classified as open forest (50–200 trees/ha), those between 26% and 50% were classified as semi-open (200–350 trees/ha), those between 51% and 75% were classified as semi-closed (350–500 trees/ha) and those >75% were classified as closed (in general, forests in a good state of conservation, with >500 trees/ha on average; López-García 2011). Cropland and pasture were considered non-forest (López-García 2011, López-García et al. 2016).

A map of the 1971 forest-cover density and vegetation types was generated by photointerpretation (scale of 1:5000) from elements such as the texture, tone, shade of the trees and shape using panchromatic aerial photographs, where the stereoscopic model of conifers has different characteristics (pines have a light grey tone and a semi-rounded shape; firs has a dark tone and a conical shape; oaks have a dark tone, rounded crowns and low height). The combination of these shapes and tones allowed us to identify the vegetation associations and their density; the above cover density criteria were used (López-García 2009, 2011).

Once the 1971 map was complete, it was overlaid onto the 1984 orthophotograph to modify only the changed polygons through photointerpretation. The resulting 1984 coverage density map was used similarly to determine the coverage density in 1994. The 1994 coverage was overlaid onto the 2004 SPOT image, and, by visual interpretation, the polygons that changed were modified successively for 2015 and 2021. This method reduces digitization errors by not modifying polygons that do not change over time (FAO 1996). Field trips and vegetation sampling complemented the 2015 interpretation. All images were georeferenced to the UTM coordinate system with a WGS84 reference datum.

Accuracy assessment

For 1971, 1984 and 1994, no other data sources of similar spatial resolution were available to conduct a verification. However, the maps of these years were produced by an expert photo interpreter to ensure the accuracy of the analysis. The 2004, 2015 and 2021 maps were verified by a stratified random sampling using Planet images from close dates, and the accuracy was determined using the Kappa index. Additionally, for 2015, the vegetation types and density of forest cover were verified through field trips between November 2015 and March 2016. Six circular samples of 1000 m^2 were established in previously marked sites in homogeneous areas that differed in their density of forest cover Appendix S1 & Fig. S1). In each sample, the density of forest cover and the vegetation type were determined, height and diameter at breast height were measured and dendrochronological samples were taken from five to seven trees to determine the age of the trees (Appendix S1 & Fig. S2). These data were georeferenced using a global positioning system (GPS) to validate the supervised classification of the SPOT image and obtain the overall accuracy. According to the Kappa index, the polygons derived from the 2004 SPOT imagery had an accuracy level of 0.84. In 2015, the accuracy increased to 0.88, while the 2021 Google Earth-based maps achieved a Kappa index of 0.85. The variability in omission and commission errors was primarily due to differences in texture and shading between the SPOT and Google Earth data, which affected the assignment of cover density. The validation of the 2015 vegetation types resulted in an overall accuracy of 0.86.

Analysis of change processes

Change matrices were generated to distinguish the change processes. The matrix cells above the diagonal show the disturbance processes, forest degradation, deforestation and land-use change to agriculture; the cells below the diagonal show the recovery processes, densification, reforestation and afforestation. Forest disturbance processes are forest degradation, which is the decrease in tree density (change of cover from closed to semi-closed, from closed to semi-open, from closed to open, from semi-closed to semi-open, from semi-closed to open or from semi-open to open), and densification means the opposite. Deforestation is the reduction of the density of forest cover to <10% (the change of closed, semi-closed, semi-open or open cover to deforested), and reforestation means the opposite. Land-use change to agriculture is the transition from forest (closed, semi-closed, semi-open or open) to agricultural land, and afforestation means the opposite.

Participatory fieldwork

The experience accumulated over more than two decades working in this region allowed us to get closer to the communities and *ejidos* in order to learn their views on the importance of conserving the forest and aquifer recharge to develop productive activities such as trout farming (López et al. 2014, Manzo et al. 2014). Living alongside these communities also provided insights into the conflicts and disputes generated by the selective support from government programmes. In 2010, we conducted semi-structured interviews (Appendix S1) with local stakeholders (*ejido* and community representatives, local people, non-governmental organizations and government agencies) to gather qualitative information on forest-change processes. In particular, in May 2010, we conducted participatory fieldwork and interviews with 18 inhabitants of the El Rosario *ejido* to verify the impacts of the heavy rains and mudslides of February 2010, which had destroyed more than half of the local trout farms (Mr José Felix Moreno Argueta, personal communication 2010). Interviewees also shared information on the Procampo and CONAFOR programmes that supported the afforestation of agricultural land and reforestation of deforested areas in the region.

Results

Changes in forest cover

Between 1971 and 2021, the La Hacienda watershed constantly shifted between forest density categories. Some areas were deforested, while others became forested. The net loss in the closed density area was only 24 ha (of 843.9 ha), representing a loss of 2.8% at 0.48 ha/year; 21.7% of the area increased its cover density, and 23.7% decreased its cover density (Table 1). Meanwhile, the deforested area increased by 110.7 ha (2.2 ha/year, equivalent to 13.1% of the total), while 25 ha were converted to forest from agriculture. The forest- cover category that changed the most was the semi-closed forest, which reduced from 104 to 68 ha in these 50 years, equivalent to a 35% net loss at 0.7 ha/year, where 59.8% changed to closed density and 36.8% to the other types of densities, including semi-open and open (3.3%), deforested (26.2%) and non-forested (7.3%) areas. Semi-open forest decreased very slightly (Table 1). Despite these changes, after 50 years, closed density remained the most prominent area type, followed by non-forest, deforested and the other types of densities.

All categories had negative net annual rates of change of less than 1 ha, except for the deforested category, which increased by more than 2 ha/year (Table 2). Closed cover decreased in 1984–1994, before becoming more drastic in 1994–2004, with an overall net annual rate of –0.49 ha/year. The semi-closed category decreased overall, but not during the second period, registering an overall net annual rate of –0.73 ha/year. The deforested category increased in 1984–1994 and 1994–2004, with a net change rate of 2.30 ha/year. The non-forested category increased in the second period, after which it began to decrease, registering 28.7 ha less than the initial year, achieving a net change rate of –0.57 ha/year (Table 2).

Change processes

Over the 50 years assessed, the net change between recovery and disturbance processes was a 3% decline, with a quarter of the changes attributable to deforestation. Deforestation was the dominant disturbance factor between 1971 and 2021, followed by forest degradation. The most significant recovery process was forest densification. The final outcome showed deforestation exceeding reforestation by 6%, with 46.4% of the watershed undergoing some change. Before the monarch butterfly's overwintering areas were identified, from 1971 to 1984, forest densification exceeded degradation by 8.9%, reforestation exceeded deforestation by 6.1% and agricultural land use increased slightly by 2.1% (Figs 2 & 3 & Table 3). Between 1984 and 1994, disturbance processes exceeded recovery by 8.6%, mainly due to logging throughout almost the entire Francisco Serrato *ejido*. Forest degradation was the dominant disturbance type, followed by agricultural land-use change and deforestation, mostly because the Francisco Serrato *ejido* did not participate in the PES programme. From 1994 to 2004, the watershed experienced significant land-use changes (34.7%), 29.4% of which were disturbances and 5.3% represented recovery. This period saw the greatest deforestation

https://doi.org/10.1017/S0376892925000128 Published online by Cambridge University Press

Table 1. Changes in the areas of the forest density categories between 1971 and 2021. On the diagonal are areas that did not change; above the diagonal are disturbances, and below the diagonal are recoveries. Bold numbers highlight the most significant changes.

		2021 (ha)						
	Categories	Closed	Semi-closed	Semi-open	Open	Deforested	No forest	Total 1971
1971 (ha)	Closed	482.3	47.7	35.4	25.7	**171.1**	**82.4**	844.6
	Semi-closed	62.5	3.5	2.2	1.2	27.3	7.6	104.4
	Semi-open	40.8	0.9	2.8	2.3	16.3	9.4	72.6
	Open	31.2		8.7	5.0	16.7	4.5	66.2
	Deforested	96.8	6.4	11.0	6.5	70.4	–	191.1
	No forest	**106.9**	9.4	6.7	6.0	–	410.7	539.7
	Total 2021	820.6	68.0	66.9	46.8	301.8	514.6	1818.6

Table 2. Estimated land-cover area for the La Hacienda watershed and land-cover changes by period. Bold numbers highlight the most significant changes.

	Area (ha)						Annual change rate (ha/year)					Net rate
Category	1971	1984	1994	2004	2015	2021	1971–1984	1984–1994	1994–2004	2004–2015	2015–2021	1971–2021
Closed	845	1037	880	490	770	820	16.1	−14.3	**−39.1**	**25.4**	8.4	−0.5
Semi-closed	104	72	102	89	77	68	−2.7	2.8	−1.3	−1.1	−1.4	−0.7
Semi-open	72.6	58	59	100	70	67	−1.2	0.1	4.1	−2.7	−0.5	−0.1
Open	66.2	35	50	141	37	47	−2.6	1.3	9.1	−9.4	1.6	−0.4
Deforested	191	80	130	**405**	**317**	**306**	−9.3	4.6	**27.5**	−7.9	−1.9	**2.3**
No forest	540	537	598	595	548	511	−0.2	5.5	−0.3	−4.3	−6.2	−0.6

(18%) and the most significant decrease in forest-cover density (10.4%). There was a high rate of change in the closed forest cover, primarily towards deforestation. Deforestation outpaced reforestation by 15.4%, and forest degradation outweighed densification by 9.1%. Forest recovery was negligible compared to forest disturbance. Between 2004 and 2015, forest recovery dominated forest disturbance. Densification exceeded degradation by 11.5%, and recovery outweighed disturbance by 18.9%. In the 2015–2021 interval, recovery – although less intense – continued to predominate over disturbance. From 1971 to 2021, deforestation only exceeded reforestation by 6.1%. The balance between forest densification and forest degradation was positive (1.6%), as was the balance between land-use change to agriculture and afforestation (1.4%; Figs 2 & 3 & Table 3).

Changes in vegetation

From 1971 to 2015, the temperate forests in the watershed experienced significant changes. In 1971, temperate forests covered 1087.8 ha, dominated by pure fir forests associated with pine and oak (96.9%) and pine forests (3.1%; Appendix S1 & Table S1). Over the subsequent 44 years, the conifer forest decreased to 952.4 ha, and the fir forest and its associations decreased to 56.7%, while the fir–pine association increased by 65% (to 95.5 ha). During this time, pine forests increased by 83% (163.5 ha), and new associations of pine forests with fir and oak (29.9%) and small forests of alder and cedar developed (1%; Appendix S1 & Table S1). Logging activities affected the forests from 1984 to 1994; 210 ha were cut, 49.0% corresponding to forest degradation, 26.4% to deforestation and 24.6% to conversion to agriculture. The period 1994–2004 saw even more extensive disturbances, with 457 ha affected, of which 36.6% was forest degradation, 60.3% was deforestation and 3.1% was conversion to agriculture. These changes in the pure fir stands reduced the possibility of monarch butterfly overwintering.

The *ejidos* El Rosario and Francisco Serrato were the most disturbed in the 1984–1994 and 1994–2004 periods. In El Rosario, 306.9 ha were disturbed in the first period and 164 ha were disturbed in the second period. In the Francisco Serrato *ejido*, 6.7 ha were disturbed in the first period and 206.2 ha were disturbed in the second period. The largest deforested area within the watershed corresponded to the *ejido* Francisco Serrato, with a fir forest loss of 177 ha (12.1%). Between 1971 and 1984, 80 ha were recovered, and between 1984 and 2004, 206 ha of fir forest were disturbed (184 ha deforested and 22 ha degraded), leaving most of the property deforested between 2001 and 2003. In the subsequent period of 2004–2015, 47 ha were recovered. Over the 50-year period, 163 ha were disturbed, 15 ha recovered and the deforested and agricultural lands were reduced (Appendix S1 & Table S1).

Discussion

Almost half of the La Hacienda watershed within the MBBR experienced significant changes in forest cover between 1971 and 2021, including 24.7% forest recovery and 21.7% disturbances such as deforestation and forest degradation. Crucially, 43.3% of the essential fir forest, which is critical for the monarch butterfly's habitat, was lost. The MBBR design's failure to account for the communities' locations or opinions can partially explain these results. For example, El Rosario *ejido* has the largest and most permanent monarch butterfly hibernation area in the Reserve, attracting tourists. By contrast, the Francisco Serrato and La Mesa *ejidos*, whose forests are in the watershed's uplands, did not benefit from such tourism, and consequently these communities cut down their forests after 2000 (Homero Gómez, personal communication 2005).

Four *ejidatarios* from Francisco Serrato agreed that, during the first proposal to expand the MBBR in 1999, they were invited to participate in the core zone because they had well-conserved fir forests. However, for reasons unknown to them, the entire area of the Francisco Serrato *ejido* was excluded from the core zone and included in the buffer zone. In response, they said they had logged almost all of their *ejido* forest between 2000 and 2004.

The analysis over five periods allowed us to identify the forest loss and recovery processes. The final balance shows that the closed

https://doi.org/10.1017/S0376892925000128 Published online by Cambridge University Press

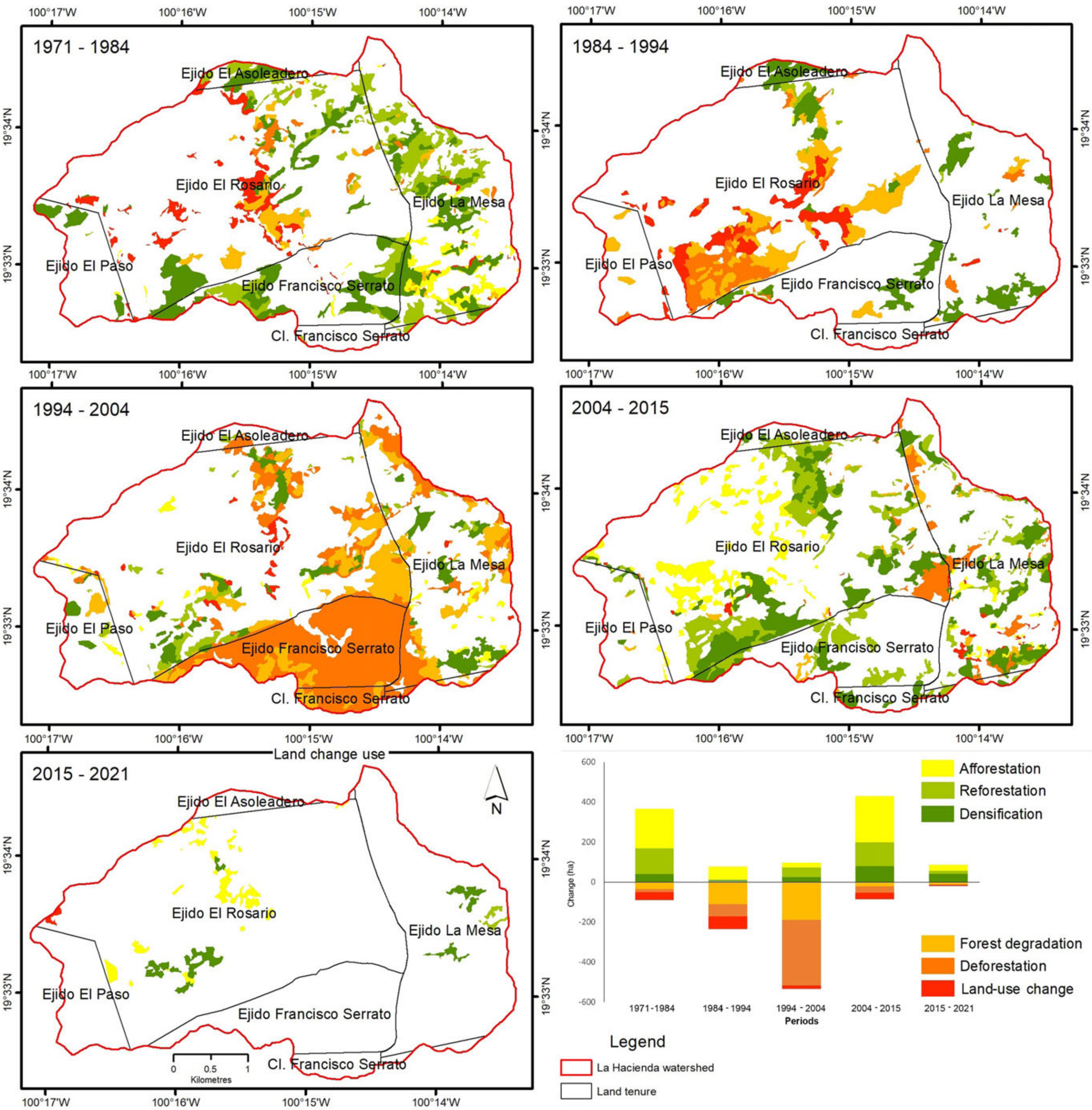

Figure 2. Processes of change in La Hacienda watershed.

forest cover has remained almost unchanged, with a net loss of 24 ha over 50 years. In 1973, a forest ban was declared in Michoacán (Merino-Pérez & Hernández 2004); nonetheless, forest cuts and the expansion of the agricultural frontier occurred (Ibarra 2011). After the ban ended, the period from 1971 to 1984 saw recovery processes outpace disturbance occurring after the first protected area decree of 1980 and before the second PA decree of 1986.

Contrary to expectations, between 1984 and 1994, degradation exceeded recovery. The second decree in 1986 contributed to this forest degradation. This pre-emptive behaviour, whereby landowners clear their forests to avoid potential restrictions, has also been observed in protected areas in African and Latin American countries (Andam et al. 2008, Brown et al. 2016, Nolte et al. 2018, Llopis et al. 2019). Additionally, the lack of consultation with local communities regarding the logging ban in the core zone and harvesting restrictions in the Reserve's buffer zone prompted retaliatory forest degradation by the El Rosario community within this watershed (WWF 2004). The situation was further exacerbated by the closure of major local industries – a flower production industry, a gold mine and wood-processing plants – which were primary sources of employment in the region (Merino-Pérez 1999).

When the protected area was expanded and designated as a Biosphere Reserve in 2000, this further restricted the economic activities of the local *ejido* and community inhabitants, generating more conflicts due to the transformation of communally regulated

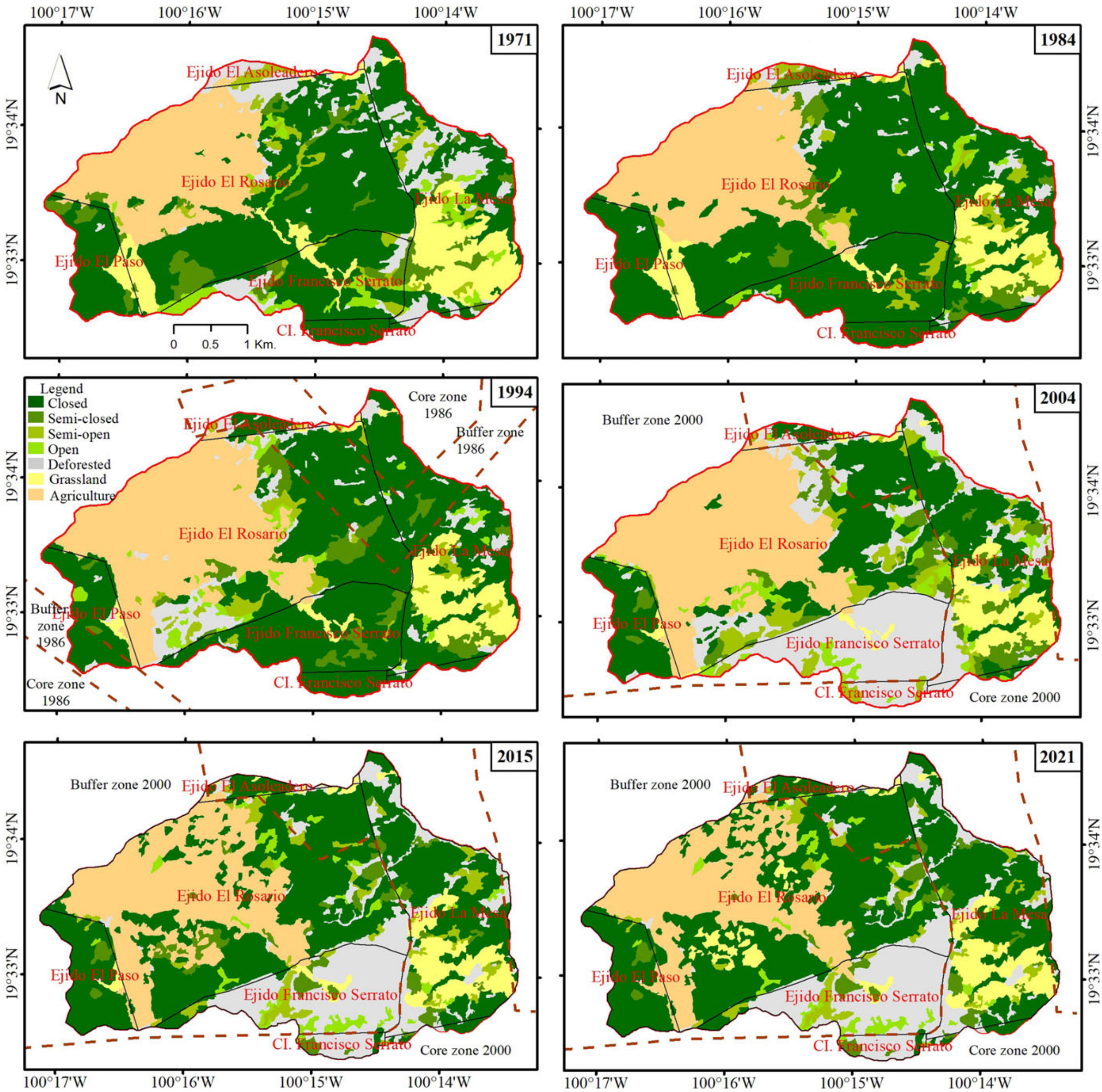

Figure 3. Sequence of forest cover in La Hacienda watershed from 1971 to 2021.

resources (Merino-Pérez & Hernández 2004, Brenner 2009, Honey-Roses 2009). This contributed to a doubling of the forest disturbance already being caused by extreme weather, large-scale illegal logging by organized criminal groups and small-scale illegal logging by individuals from the local communities (Vidal et al. 2013, Flores Martínez et al. 2019). Additionally, authorized logging in the study area from 1993 to 2006 (Navarrete et al. 2011) may have decreased forest-cover density.

Forest fires have facilitated regeneration in some MBBR regions (Honey-Rosés et al. 2018). In 2012, low-severity and superficial forest fires covering less than 3 ha in the south-western portion of the La Hacienda watershed did not influence the decrease in forest density. Nevertheless, they did influence regeneration (Cantú 2013).

After logging in this watershed, extensive reforestation with fir, pine and cedar was instigated in 1994. However, only the pine trees had high survival rates, unlike the fir trees, which required more humidity and were less successful in their development (Amado Fernandez, director of MBBR, personal communication 2023). The dense, though short, pines should soon allow this area to be classified as a forest. Before protection, the main activities in this region were logging, resin extraction and cattle ranching, which were restricted by the protected area decrees. Consequently, the local population felt impacted by this protection process, and they responded by cutting and burning the forests, and organized groups also cleared the forests (WWF 2004). This is significant because 81% of the watershed is above 2900 m, being characterized

https://doi.org/10.1017/S0376892925000128 Published online by Cambridge University Press

Table 3. Percentage contributions of various change processes by period.

Change process	1971–1984	1984–1994	1994–2004 (%)	2004–2015	2015–2021	1971–2021
Forest degradation	1.9	6.1	10.4	1.2	0.7	6.3
Deforestation	0.9	3.3	18.0	1.7	0.1	12.7
Land-use change to agriculture	2.1	3.4	1.0	1.8	0.3	5.7
Total disturbance	4.9	12.9	29.4	4.7	1.0	24.7
Forest densification	10.8	3.6	1.3	12.7	1.7	7.9
Reforestation	7.0	0.6	2.6	6.5	0.7	6.6
Afforestation	2.3	0.1	1.4	4.4	2.3	7.1
Total recovery	20.1	4.3	5.3	23.6	4.7	21.7
No change	75.0	82.9	65.3	71.6	94.2	53.7

by the fir forests that are essential for monarch butterfly migration. This fir forest community was the most affected, mainly due to illegal logging. However, global warming may have also contributed to the loss of fir trees, leading pine trees to expand into these previously fir tree-dominated areas (Zhang et al. in press).

Despite initial anger from local communities regarding the lack of consultation, which led them to cut down these forests, the overall trend has been more towards recovery than degradation. Factors like REDD+, PES, community actions, natural regeneration after fires (Skutsch et al. 2015) and abandoned farmland have contributed to this (Špirić et al. 2023), in combination with government programmes such as Procampo, which supported the reforestation of idle or abandoned lands.

We show how the performance of protected areas can be affected if management programmes do not contemplate their socioeconomic context. However, poor performance must also be analysed with the understanding that, in some countries, newly decreed protected areas lack the financial, human and technical resources necessary for their effective management (Blackman et al. 2015). Deforestation in the 1990s could have been avoided through the inclusion of more local consultation and earlier alternatives such as PES. Evidence of conservation and human benefits helps justify place-based conservation (Elías 2012, Robinson et al. 2018). Public participation is needed to align protected area planning with local knowledge, livelihoods, needs and values by establishing flexible objectives in order to avoid the occurrence of conflicts between people and conservation (Maikhuri et al. 2000, Becker 2002, He & Cliquet 2020).

We highlight the need for conservation efforts to address environmental and socioeconomic factors. On-site conservation alone is insufficient, as it can restrict local access to resources without offering viable alternatives. Instead, conservation policies should incorporate community needs and engage with local populations, thereby increasing the chances of successful forest recovery and long-term sustainability. This has broader implications for global conservation strategies, particularly in areas with complex land tenure systems and facing significant socioeconomic challenges.

Conclusion

The multi-temporal analysis from 1971 to 2021 of the La Hacienda watershed in the MBBR showed that nearly half of its area underwent land-use change during this period, despite its protected status. Examining different periods revealed the dynamics of forest loss and recovery relative to protection measures. Before the first decree in 1971–1984, closed forest cover increased, with recovery processes dominating over disturbances. However, in the periods 1984–1994 and 1994–2004, when the second and third monarch protection decrees were implemented, disturbances were more prevalent than recovery. This is attributable at least in part to pre-emptive tree-cutting in response to the anticipated restrictions. In the 2004–2021 period, with the introduction of PES, reforestation, surveillance and community engagement through productive activities and tourism, a rise in closed forest cover and a dominance of recovery processes were observed. Surprisingly, the overall closed forest cover remained nearly unchanged over these 50 years. Notably, the area of high-density fir forest, although reduced by almost half, remained the most prominent. Protected area planning must consider land tenure and involve local communities in decision-making, as doing so would help reduce forest degradation and land-use change and integrate conservation with community needs to a much greater extent.

Supplementary material. The supplementary material for this article can be found at https://doi.org/10.1017/S0376892925000128.

Acknowledgements. The Estación Receptora México, Nueva Generación provided the SPOT 6 images, and INEGI supplied the orthophotographs. The authors thank Marco Antonio López for his help with database management.

Financial support. DGAPA supported this research through the PAPIIT IN304123 'Análisis de las causas que determinan los procesos de cambio en áreas protegidas del eje neovolcánico, México'.

Competing interests. The authors declare none.

Ethical standards. Not applicable.

References

Alix-Garcia J, de Janvry A, Sadoulet E, Torres J (2009) Lessons Learned from Mexico's payment for environmental services program. In: L Lipper, T Sakuyama, R Stringer, D Zilberman (eds), *Payment for Environmental Services in Agricultural Landscapes* (pp. 163–188). Natural Resource Management and Policy 31. Berlin, Germany: Springer.

Andam KS, Ferraro PJ., Pfaff A, Robalino JA (2008) Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences of the United States of America* 105: 16089–16094.

Antoni C, Huber-Sannwald E, Reyes H, van't Hooft A, Schoon M (2019) Socio-ecological dynamics of a tropical agricultural region: historical analysis of system change and opportunities. *Land Use Policy* 81: 346–359.

Becker CD (2002) Grassroots to grassroots: why forest preservation was rapid at Loma Alta, Ecuador. *World Development* 31: 163–176.

Blackman A, Pfaff A, Robalino J (2015) Paper park performance: Mexico's natural protected areas in the 1990s. *Global Environmental Change* 31: 50–61.

Brenner L (2006) Áreas Naturales Protegidas y ecoturismo: el caso de la reserva de la Biosfera Mariposa Monarca. *Relaciones. Estudios de Historia y Sociedad* 27: 237–268.

Brenner L (2009) Aceptación de políticas de conservación ambiental: el caso de la Reserva de la Biosfera Mariposa Monarca. *Economía, Sociedad y Territorio* 9: 259–295.

Brenner L, San German S (2012) Gobernanza local para el 'ecoturismo' en la Reserva de la Biosfera Mariposa Monarca, México. *Alteridades* 22: 131–146.

Brower LP, Castilleja G, Peralta A, López-García J, Bojórquez-Tapia L, Díaz S, et al. (2002) Quantitative changes in forest quality in a principal overwintering area of the monarch butterfly in México, 1971–1999. *Conservation Biology* 2: 346–359.

Brower LP, Slayback DA, Jaramillo-López P, Ramirez MI, Oberhauser KS, Williams EH, Fink LS (2016) Illegal logging of 10 hectares of forest in the Sierra Chincua monarch butterfly overwintering area in Mexico. *American Entomologist* 62: 92–97.

Brown DS, Brown JC, Brown C (2016) Land occupations and deforestation in the Brazilian Amazon. *Land Use Policy* 54: 331–338.

Cantú MX (2013) *Incendios del 2012 en la Monarch Butterfly Biosphere Reserve: caracterización y respuesta institucional. Tesis Licenciatura en Ciencias Ambientales*. Morelia, Mexico: Universidad Nacional Autónoma de Mexico.

Carbale B, Chapela F, Madrid S (1997) El manejo forestal comunitario y la certificación. In L Merino (ed.), *El manejo forestal comunitario en México y sus perspectivas de sustentabilidad* (pp. 17–33). Cuernavaca, Mexico: Centro Regional de Investigaciones Multidisciplinarias, UNAM, SEMARNAP, CMSS, WRI.

CONANP (2001) *Programa de manejo Monarch Butterfly Biosphere Reserve*. Morelia, Mexico: CONANP-SEMARNAT.

DOF (1980) Decreto [www document]. URL https://www.dof.gob.mx/nota_detalle.php?codigo=4851272&fecha=09/04/1980#gsc.tab=0

DOF (1986) Decreto [www document]. URL https://www.dof.gob.mx/nota_detalle.php?codigo=4813548&fecha=09/10/1986#gsc.tab=0

DOF (2000) Decreto [www document]. URL https://simec.conanp.gob.mx/pdf_decretos/40_decreto.pdf

Durán E, Bray DB, Velázquez A, Larrazábal A (2011) Multi-scale forest governance, deforestation, and violence in two regions of Guerrero, Mexico. *World Development* 39: 611–619.

Durán E, Bray DB, Velázquez A, Larrazábal A (2011). Multi-scale forest governance, deforestation, and violence in two regions of Guerrero, Mexico. *World Development* 39, 611–619.

Elías S (2012) From communal forests to protected areas: the implications of tenure changes in natural resource management in Guatemala. *Conservation and Society* 10: 151–160.

FAO (1996) Forest Resource Assessment Programme 1990. Survey of tropical forest cover and study of change processes. Forestry Paper 130. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO) [www document]. URL https://www.fao.org/4/w0015e/W0015E00.htm#TOC

FAO (2010) Global Forest Resources Assessment 2010. Forestry Paper 163 [www document]. URL https://www.fao.org/3/i1757e/i1757e.pdf

FAO (2015) Global Forest Resources Assessment 2015. Desk reference [www document]. URL https://www.fao.org/3/i4808e/i4808e.pdf

FAO (2020) *Global Forest Resources Assessment 2020: Main Report*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Flores-Martínez JJ, Martínez-Pacheco A, Rendón-Salinas E, Rickards J, Sarkar S, Sánchez-Cordero V (2019) Recent forest cover loss in the core zones of the Monarch Butterfly Biosphere Reserve in Mexico. *Frontiers in Environmental Science* 7: 167.

FRA (2000) On definitions of forest and forest change. Forest Resources Assessment Programme. Working Paper 33 [www document]. URL https://www.fao.org/docrep/006/ad665e/ad665e00.htm

González-Duarte C (2021) Butterflies, organized crime, and 'sad trees': a critique of the Monarch Butterfly Biosphere Reserve program in a context of rural violence. *World Development* 142: 105420.

He M, Cliquet A (2020) Challenges for protected areas management in China. *Sustainability* 12: 5879.

Hoffmann S (2022) Challenges and opportunities of area-based conservation in reaching biodiversity and sustainability goals. *Biodiversity and Conservation* 31: 325–352.

Honey M (1999) *Ecotourism and Sustainable Development: Who Owns Paradise?* Washington, DC, USA: Island Press.

Honey-Rosés J (2005) The monarch butterfly conservation fund. In: T LeQuesne, R McNally (eds), *The Green Buck: Using Economic Tools to Deliver Conservation Goals* (pp. 16–18). London, UK: WWF United Kingdom.

Honey-Rosés J (2009) Illegal logging in common property forests. *Society and Natural Resources* 22: 916–930.

Honey-Rosés J, Maurer M, Ramírez MI, Corbera E (2018) Quantifying active and passive restoration in central Mexico from 1986–2012: assessing the evidence of a forest transition. *Restoration Ecology* 26: 1180–1189.

Hu Y, Wang M, Ma T, Huang M, Huang G, Zhou W, et al. (2022) Integrated index-based assessment reveals long-term conservation progress in implementation of Convention on Biological Diversity. *Science Advances* 8: eabj8093.

Iannuzzi G, Santos R, Mourato JM (2020) The involvement of non-state actors in the creation and management of protected areas: insights from the Portuguese case. *Journal of Environmental Planning and Management* 63: 1674–1694.

Ibarra MV (2011) Conformación del espacio social de los bosques del ejido del Rosario, Michoacán, 1938–2010. *Investigaciones Geográficas* 75: 75–87.

Kegamba JJ, Sangha KK, Wurm P, Garnett ST (2022) A review of conservation-related benefit-sharing mechanisms in Tanzania. *Global Ecology and Conservation* 33: e01955.

Liu Y, Ziegler AD, Wu J, Liang S, Wang D, Xu R, et al. (2022) Effectiveness of protected areas in preventing forest loss in a tropical mountain region. *Ecological Indicators* 136: 108697.

Llopis JC, Harimalala PC, Bär R, Heinimann A, Rabemananjara ZH, Zaehringer JG (2019) Effects of protected area establishment and cash crop price dynamics on land use transitions 1990–2017 in north-eastern Madagascar. *Journal of Land Use Science* 14: 52–80.

Lockwood M (2010) Good governance for terrestrial protected areas: a framework, principles and performance outcomes. *Journal of Environmental Management* 91: 754–766.

López F, Velázquez A, Merino L (2010) Explorando los determinantes del buen manejo forestal comunitario. *Interciencia* 35: 560–567.

López-García J (2009) Análisis de cambio en la cobertura forestal en la zona núcleo de la Reserva de la Biosfera Mariposa Monarca: 2003–2006. In: R Aguirre (ed.), *Conceptos y aplicaciones de la geomática en México* (pp. 195–210). Mexico City, Mexico: Instituto de Geografía, UNAM.

López-García J (2011) Deforestation and forest degradation in the Monarch Butterfly Biosphere Reserve, Mexico, 2003–2009. *Journal of Maps* 2011: 665–673.

López-García J, Manzo-Delgado LL, Alcántara-Ayala I (2014) Rural aquaculture as a sustainable alternative for forest conservation in the Monarch Butterfly Biosphere Reserve, Mexico. *Journal of Environment Management* 138: 43–54.

López-García J, Navarro-Cerrillo RM, Manzo-Delgado LL (2022) Forest land cover trends in the Monarch Butterfly Biosphere Reserve in Mexico 1994–2017. *Environmental Conservation* 49: 244–254.

López-García J, Prado-Molina J, Manzo-Delgado LL, Peralta-Higuera A (2016) Monitoring changes of forest canopy density in a temperate forest using high-resolution aerial digital photography. *Investigaciones Geográficas* 2016: 59–74.

Maikhuri RK, Nautiyal S, Rao KS, Chandrasekhar K, Gavali R, Saxena KG (2000) Analysis and resolution of protected area–people conflicts in Nanda Devi Biosphere Reserve, India. *Environmental Conservation* 27: 43–53.

Manzo-Delgado L, López-García J, Alcántara-Ayala I (2014) The role of forest conservation in lessening land degradation in a temperate region: the Monarch Butterfly Biosphere Reserve, Mexico. *Journal of Environment Management* 138: 55–66.

Martinez-Torres HL, Castillo A, Ramirez MI, Pérez-Salicrp DR (2016) The importance of the traditional fire knowledge system in a subtropical montane socio-ecosystem in a protected natural area. *International Journal of Wildland Fire* 25: 911–921.

Martínez-Torres HL, Pérez-Salicrup DR, Castillo A, Ramirez MI (2018) Fire management in a natural protected area: what do key local actors say? *Human Ecology* 46: 515–528.

MBF (2021) Pago de Servicios Ambientales en la Reserva de la Biosfera de la Mariposa Monarca. Unpublished internal report. Monarch Butterfly Fund (MBF).

McGinlay J, Jones N, Malesios C, Dimitrakopoulos PG, Begley A, Berzborn S, et al. (2023) Exploring local public support for protected areas: what social factors influence stated and active support among local people? *Environmental Science and Policy* 145: 250–261.

Merino L (1997) Reserva Especial de la Biosfera Mariposa Monarca: problemática general de la región [www document]. URL https://dlc.dlib.indiana.edu/dlc/bitstream/handle/10535/8220/Reserva%20Especial%20de%20la%20Biosfera%20Mariposa%20Monarca.pdf?sequence=1.

Merino-Pérez L (1999) La gestión colectiva de los recursos forestales. *Revista de Comercio Exterior* 49: 1121–1133.

Merino-Pérez L, Hernández-Apolinar M (2004) Destrucción de instituciones comunitarias y deterioro de los bosques de la Reserva de la Biosfera Mariposa Monarca, Michoacán, México. *Revista Mexicana de Sociología* 66: 261–309.

Missrie M, Nelson K (2005) *Direct Payments for Conservation: Lessons from the Monarch Butterfly Conservation Fund*. St Paul, MN: University of Minnesota Press.

Miteva DA, Ellis PW, Ellis EA, Griscom BW (2019) The role of property rights in shaping the effectiveness of protected areas and resisting forest loss in the Yucatan Peninsula. *PLoS ONE* 14: e0215820.

Morett-Sánchez JC, Cosío-Ruiz C (2017) Panorama de los ejidos y comunidades agrarias en México. *Agricultura, Sociedad y Desarrollo* 14: 125–152.

Navarrete J, Ramírez MI, Pérez-Salicrup DR (2011) Logging within protected areas: spatial evaluation of the Monarch Butterfly Biosphere Reserve, Mexico. *Forest Ecology and Management* 262: 646–654.

Nieratka LR, Bray DB, Mozumder P (2015) Can payments for environmental services strengthen social capital, encourage distributional equity, and reduce poverty? *Conservation and Society* 13: 345–355.

Nolte C, Gobbi B, le Polain de Waroux Y, Piquer-Rodríguez M, Butsic V, Lambin EF (2018) Challenges in attributing avoided deforestation to policies and actors: lessons from provincial forest zoning in the Argentine Dry Chaco. *Ecological Economics* 150: 346–352.

Oldekop JA, Holmes G, Harris WE, Evans KL (2015) A global assessment of the social and conservation outcomes of protected areas. *Conservation Biology* 30: 133–141.

Pacheco A, Meyer C (2022) Land tenure drives Brazil's deforestation rates across socio-environmental contexts. *Nature Communications* 13: 5759.

Pérez-Salicrup DR, Martínez-Torres L, Del Río G, Garduño-Mendoza E (2017). *Plan Integral del Manejo del Fuego en la Reserva de la Biósfera Mariposa Monarca: acción e investigación participativa y adaptable*. Mexico City, Mexico: FMCN, CONANP, ALTERNARE AC, IIES-UNAM.

Probosque (2024) Servicios Ambientales Hidrológicos (PSAHEM) [www document]. URL https://probosque.edomex.gob.mx/fipasahem.

Robinson BE, Masuda YJ, Kelly A, Holland MB, Bedford C, Childress M, et al. (2018) Incorporating land tenure security into conservation. *Conservation Letters* 11: 1–12.

Salichtchev KA (1979) *Cartografía*. Havana, Cuba: Editorial Pueblo y Educación.

Skutsch M, Borrego A, Morales-Barquero L, Paneque-Gálvez J, Salinas-Melgoza M, Ramirez MI, et al. (2015) Opportunities, constraints and perceptions of rural communities regarding their potential to contribute to forest landscape transitions under REDD+: case studies from Mexico. *International Forestry Review* 17: 65–84.

Špirić J, Salinas-Melgoza MA, Merlo-Reyes A, Ramírez MI (2023) Estimating the causal effect of forestry programs on forest recovery in a REDD+ priority area in Michoacán, Mexico. *Forest Policy and Economics* 146: 102879.

Stewart RR, Possingham HP (2005) Efficiency, costs and trade-offs in marine reserve system design. *Environmental Modeling & Assessment* 10: 203–213.

Tucker CM (2004) Community institutions and forest management in Mexico's monarch butterfly reserve. *Society and Natural Resources* 17: 568–587.

Vidal O, López-García J, Rendón-Salinas E (2013) Trends in deforestation and forest degradation after a decade of monitoring in the Monarch Butterfly Biosphere Reserve in Mexico. *Conservation Biology* 28: 177–186.

Wade CM, Austin KG, Cajka J, Lapidus D, Everett KH, Galperin D, et al. (2020) What is threatening forests in protected areas? A global assessment of deforestation in protected areas, 2001–2018. *Forests* 11: 530.

WWF (2004) La tala ilegal y su impacto en la Reserva de la Biosfera Mariposa Monarca [www document]. URL https://www.wwf. org.mx/wwfmex/descargas/010604_Informe_Tala_Reserva.pdf.

Zhang L, Xiao-Ming L, Hua-Zhong Z, Shan G, Jian S, Hai-Feng Z, et al. (in press) A rapid transition from spruce–fir to pine–broadleaf forests in response to disturbances and climate warming on the southeastern QinghaiTibet Plateau. *Plant Diversity* 2023 (in press).

Zhang Y, Xiao X, Cao R, Zheng C, Guo Y, Gong W, Wei Z (2020) How important is community participation to eco-environmental conservation in protected areas? From the perspective of predicting locals' pro-environmental behaviours. *Science of the Total Environment* 739: 139889.

Zhao X (2020) Land and labor allocation under communal tenure: theory and evidence from China. *Journal of Development Economics* 147: 102526.