UNIVERSIDAD NACIONAL AUTÓNOMA DE MÉXICO

POSGRADO EN CIENCIAS BIOLÓGICAS

INSTITUTO DE INVESTIGACIONES EN ECOSISTEMAS Y SUSTENTABILIDAD

ÁREAS NATURALES PROTEGIDAS EN MÉXICO: DISYUNTIVAS ENTRE LA CONSERVACIÓN Y EL DESARROLLO SOCIAL LOCAL

TESIS

QUE PARA OPTAR POR EL GRADO DE:

DOCTOR EN CIENCIAS BIOLÓGICAS

PRESENTA:

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MORELIA, MICHOACÁN, 2023

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UNAM POSGRADO

POSGRADO CIENCIAS BIOLÓGICAS

COORDINACIÓN DEL POSGRADO EN CIENCIAS BIOLÓGICAS

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ASUNTO: Oficio de Jurado

M. en C. Ivonne Ramírez Wence
Directora General de Administración Escolar, UNAM
P r e s e n t e

Me permito informar a usted que, en la reunión ordinaria del Comité Académico del Posgrado en Ciencias Biológicas, celebrada el día **15 de agosto de 2022,** se aprobó el siguiente jurado para el examen de grado de **DOCTOR EN CIENCIAS,** del estudiante **AULIZ ORTIZ DANIEL MARTÍN,** con número de cuenta **408009023,** con la tesis titulada, "**ÁREAS NATURALES PROTEGIDAS EN MÉXICO: DISYUNTIVAS ENTRE LA CONSERVACIÓN Y EL DESARROLLO SOCIAL LOCAL",** realizada bajo la dirección del **DR.MIGUEL MARTÍNEZ RAMOS**, quedando integrado de la siguiente manera:

Presidenta:	DRA. CLAUDIA ELIZABETH MORENO ORTEGA
Vocal:	DRA. MARTHA BONILLA MOHENO
Secretario:	DR. VÍCTOR ARROYO RODRÍGUEZ
Vocal:	DR. ENRIQUE MARTÍNEZ MEYER
Vocal:	DRA. MARÍA ISABEL RAMÍREZ RAMÍREZ

Sin otro particular, me es grato enviarle un cordial saludo.

A T E N T A M E N T E
"POR MI RAZA HABLARÁ EL ESPÍRITU"
Ciudad Universitaria, Cd. Mx., a 10 de noviembre de 2022

COORDINADOR DEL PROGRAMA

POSGRADO EN CIENCIAS BIOLÓGICAS COORDINACIÓN

DR. ADOLFO GERARDO NAVARRO SIGÜENZA

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Resumen:

Las áreas naturales protegidas (ANPs) son una de las políticas más efectivas para conservar la biodiversidad. En América Latina, las ANPs suelen estar en contextos rurales rodeados de comunidades que dependen de los recursos naturales para subsistir, lo cual plantea una disyuntiva entre conservación y desarrollo muy importante. Por un lado, a través de acciones legales las ANPs reducen el tipo de actividades humanas permitidas con el objetivo de preservar la cobertura forestal y mantener la riqueza y abundancia de las especies que albergan. Sin embargo, algunos factores socioeconómicos (por ejemplo de carácter demográfico y económico) amenazan la biodiversidad en las ANPs pues promueven cambios en términos de la cobertura forestal y su fragmentación. Por otro lado, algunos autores sugieren que el poco acceso a los recursos naturales puede reforzar las condiciones de pobreza de las comunidades locales vecinas a las ANP. Existen algunos esfuerzos por analizar estas disyuntivas, pero aún existen vacíos de información que son necesarios atender especialmente en el contexto nacional.

Por ejemplo, poco es sabido sobre qué factores socioeconómicos determinan los cambios en de la cobertura y fragmentación del bosque en las ANPs y más aún de cómo esto se conecta con los cambios en biodiversidad observados. Asimismo, la información sobre si las ANPs mexicanas refuerzan o no la pobreza en las comunidades locales y si ello depende de su esquema de manejo es muy escaza. Más aún, existe aún necesidad por evaluar los posibles compromisos entre conservación y desarrollo existentes en algunas ANPs mexicanas que permitan encontrar escenarios favorables para ambas metas. Esta investigación aborda todos estos aspectos de la compleja relación entre ANPs y sociedad considerando los más recientes avances metodológicos para reducir el efecto de factores de confusión creados por un sesgo en el establecimiento de las ANPs (e.g., grado de aislamiento, clima, elevación, pendiente, etc.).

En el capítulo 1 se describen los principios teóricos que guían la investigación y se plantean sus objetivos. El capítulo 2 aborda la pregunta de si las ANPs mexicanas refuerzan la pobreza de las comunidades locales vecinas. Asimismo, en este capítulo se evalúan los posibles compromisos (*trade-off*) entre conservación y desarrollo que ocurren en dos esquemas de manejo diferentes: las reservas de la biosfera (que permiten el uso sustentable de recursos) y los parques nacionales (un esquema más restrictivo). Los resultados muestran que las ANPs no refuerzan la pobreza y que además por sí mismos los diferentes esquemas de manejo no crean disyuntivas entre conservación y desarrollo; pero que bajo condiciones más adversas (lugares aislados, con suelos pobres y poca aptitud agrícola), los parques nacionales se asocian a lugares más marginados.

En el capítulo 3 evalúa la efectividad de las reservas de la biosfera tropicales mexicanas para reducir la pérdida y fragmentación del bosque y promover su regeneración. Asimismo, en este capítulo también se exploran los promotores socioeconómicos de los cambios espaciales

de la estructura forestal. Los resultados confirman el rol de las ANPs para reducir la pérdida de bosque. Además, en un aspecto menos documentado en la literatura, se encontró que las ANPs también tienen estructuras forestales menos fragmentadas, pero que no son efectivas para reducir la tasa de fragmentación ni promover la regeneración del bosque. Los factores más importantes que promueven el deterioro de la estructura forestal en la ANPs fueron la densidad de asentamientos rurales y la poca disponibilidad de empleos fuera del sector agrícola.

En el capítulo 4 con la colaboración de más de 60 expertos en biodiversidad de 14 reservas de la biosfera mexicanas se documentan los cambios en biodiversidad ocurridos durante los últimos 30 años en las reservas, se exploran sus tendencias y los factores socioeconómicos detrás de éstas. Los resultados muestran una tendencia a disminución de la biodiversidad general en las reservas, pero más acentuada en gremios sensibles a as perturbaciones (animales de talla grande o alto nivel trófico y plantas tolerantes a la sobra). El capítulo propone un modelo multivariado que explica satisfactoriamente los cambios de biodiversidad y que incluye como promotores subyacentes a la densidad poblacional y la poca disponibilidad de empleos fuera del sector agrícola, como promotores proximales a la extensión de redes carreteras y de la frontera agrícola y como variables de cambio de la estructura forestal a la tasa de pérdida de bosque y la tasa de cambio de aislamiento entre parches de bosque.

Finalmente, en el capítulo 5 se discuten los principales hallazgos de esta tesis y su importancia para las ANPs mexicanas. En su conjunto los resultados muestran que existen escenarios favorables tanto para la conservación como el desarrollo de las comunidades y otros donde alguna de las dos metas puede verse comprometida. De esta forma, las ANPs en contextos donde la presión demográfica alrededor de las reservas es menor (i.e., menor densidad poblacional y densidad de asentamientos rurales) y en donde existe una mayor oferta de trabajos en sectores no agrícolas guardan mejores condiciones en términos de la cobertura forestal, fragmentación del bosque y mayor riqueza y abundancia de gremios sensibles a perturbaciones antropogénicas. Asimismo, los resultados muestran que *per se* el tipo de esquema de manejo de las ANPs no crea disyuntivas entre conservación y desarrollo pero que, no obstante, en condiciones que en un principio pueden ser adversas para el desarrollo de comunidades rurales (i.e., lugares aislados, con poca conectividad con mercados para el intercambio de productos, suelos pobres de pendiente pronunciada que dificultan el cultivo de la tierra) establecer medidas más estrictas en las ANPs puede dificultar el desarrollo de las comunidades locales vecinas. Por tanto, los resultados sugieren que, en el contexto nacional, las mejores condiciones en donde escenarios “ganar-ganar” en términos de conservación y desarrollo están en reservas de la biosfera situadas en regiones donde la disponibilidad de oportunidades laborales en sectores no agrícolas, como turismo e industria, son importantes. Esta investigación resalta la necesidad de ver a las ANPs de manera integral con las comunidades que la rodean de tal manera que la conservación no quede restringida dentro de los límites de las reservas, ni que, por otro lado, se excluyan de estas zonas las posibilidades de desarrollo de las comunidades.

Abstract

Protected areas (PAs) are one of the most effective policies for biological conservation. In Latin America, PAs are usually established in rural contexts surrounded by communities that depend on natural resources for their livelihoods. This represents an important trade-off between conservation and development. On the one hand, through legal actions, PAs reduce the type of human activities allowed in order to preserve forest cover and maintain the richness and abundance of the species they harbor. However, some socioeconomic factors (e.g., demographic and economic) threaten biodiversity in PAs by promoting changes in forest cover and fragmentation. On the other hand, some authors suggest that the restriction on access to natural resources can reinforce the poverty of local communities neighboring PAs. There are some efforts to analyze these trade-offs, but there are still information gaps that need to be addressed, especially in the national context.

For example, little is known about the socioeconomic factors that drive changes in forest cover and fragmentation in PAs and even less about how they affect the changes in biodiversity. Furthermore, information on whether Mexican PAs reinforce poverty in local communities and whether this depends on their management scheme is very scarce. Moreover, it remains a need to evaluate the trade-offs between conservation and development in Mexican PAs and to find synergic scenarios for both goals. This research addresses all these aspects of the complex relationship between PAs and society considering the most recent methodological advances that reduce the effect of some confounding factors created by the PAs establishment bias (e.g., degree of isolation, climate, elevation, slope, etc.).

Chapter 1 describes the theoretical principles guiding the research and sets out its objectives. Chapter 2 addresses the question of whether Mexican PAs reinforce the poverty of neighboring local communities. This chapter also evaluates the possible trade-offs between conservation and development that occur in two different management schemes: biosphere reserves (which allow sustainable resource use) and national parks (a more restrictive scheme). The results show that PAs do not reinforce poverty and that the different management schemes themselves do not create trade-offs between conservation and development; but that under more adverse conditions (isolated places, with poor soils and low agricultural suitability), national parks are associated with more marginalized communities.

Chapter 3 evaluates the effectiveness of Mexican tropical biosphere reserves in reducing forest loss and fragmentation and promoting forest regeneration. This chapter also explores the socioeconomic drivers of spatial changes in forest structure. The results confirm the role

of PAs in reducing forest loss. Furthermore, in an aspect less documented in the literature, I found that PAs also have less fragmented forest structures, but that is not effective in reducing the fragmentation rate nor promoting forest regeneration. In addition, the most important factors driving forest structure deterioration in PAs were the density of rural settlements and the low availability of non-farm jobs.

In Chapter 4, with the collaboration of more than 60 biodiversity experts from 14 Mexican biosphere reserves, we document the changes in biodiversity that have occurred over the last 30 years in the reserves and explore their trends and the socioeconomic drivers behind them. The results show a trend towards a general decline in biodiversity in the reserves, but more accentuated in the disturbance-sensitive guilds (large animals or high trophic level and plants tolerant to disturbances). In this chapter, I propose a multivariate model that satisfactorily explains biodiversity changes and that includes as underlying drivers the population density and the low availability of non-farm occupation; as proximal drivers the extension of road networks and the agricultural frontier; and as variables of forest, structure change the rate of forest loss and the rate of change of isolation between forest patches.

Finally, Chapter 5 discusses the main findings of this thesis and their relevance for Mexican PAs. Overall, the results show that there are favorable scenarios for both, conservation and community development, and others where either of the two goals may be compromised. Thus, PAs in contexts where demographic pressure around the reserves is lower (i.e., lower population density and density of rural settlements) and where there is a greater supply of non-farm occupations have better conditions in terms of forest cover, lower forest fragmentation, and greater richness and abundance of guilds sensitive to anthropogenic disturbances. Likewise, the results show that per se the type of PA management scheme does not create trade-offs between conservation and development but that, nevertheless, in conditions that initially are adverse for the development of rural communities (i.e., isolated places, with little access to trade markets, poor soils and steeper slopes that make it difficult to cultivate the land) establishing stricter measures in PAs may hinder the development of neighboring local communities. Therefore, the results suggest that, in the national context, the best conditions for "win-win" scenarios in terms of conservation and development are in biosphere reserves located in regions where the availability of job opportunities in non-farm activities, such as tourism and industry, are important. This research highlights the need to consider the PAs as integral systems along with their surrounding communities in such a way that conservation is not restricted within the limits of the reserves, and on the other hand, nor that the development possibilities of the communities are excluded from these zones.

Capítulo 1: Introducción general

1.1 La crisis de biodiversidad

En las últimas décadas se ha registrado un decremento sostenido de la riqueza y abundancia de grupos biológicos enteros a nivel global (Butchart et al., 2010). De hecho, la tasa de extinción en los últimos 50 años es varios ordenes de magnitud superior a la tasa promedio de los últimos 100,000 años (Barnosky et al., 2011; Pimm et al., 2014), y la mayor ocurrencia de extinciones se encuentra en las regiones tropicales (Dirzo & Raven, 2003). Asimismo, la Unión Internacional para la Conservación de la Naturaleza (IUCN por sus siglas en inglés) señala que en la actualidad cerca de 27,159 especies de vertebrados, invertebrados, plantas, hongos y protistas se encuentran bajo amenaza de extinción (IUCN, 2019b). Aunado a esto, datos recientes señalan que cerca de un millón de especies enfrentarán la extinción durante las siguientes décadas si no se toman acciones para impedirlo (IPBES, 2019).

Estos cambios en la biodiversidad dependen de los rasgos funcionales de las especies. Características como la talla (Newbold et al., 2015; Smith et al., 2020) o el tipo de alimentación son importantes para determinar el riesgo de extinción de las especies (Newbold et al., 2020). Por ejemplo, las especies de animales de talla grande suelen tener requerimientos energéticos mayores a los de las especies de talla menor, lo cual se ve reflejado en la necesidad de una mayor superficie de hábitat (Biedermann, 2003). Debido a esto, las especies de talla grande y altos requerimientos energéticos son más sensibles a la pérdida de la cobertura forestal (Cardillo et al., 2005). En contraste, existen especies que por sus características (talla pequeña, ciclos de vida cortos, de hábitos generalistas) son más tolerantes a estos cambios y, por tanto, su riqueza y abundancia se ve menos perjudicada, o incluso, favorecida por la disminución de cobertura forestal (Filgueiras et al., 2021). Esta diferencia puede tener severas consecuencias sobre la función misma del ecosistema, pues el papel que desempeñan las especies sensibles y las tolerantes es diferente (Magioli et al., 2021). Las especies sensibles, por sus características biológicas, suelen ser las más abundantes en biomas con menor grado de deterioro, por ello, su impacto en funciones ecosistémicas como la productividad primaria es mayor en comparación con las especies tolerantes que son menos abundantes (Smith et al., 2020). De este modo, una reducción en la riqueza de especies sensibles puede no representar un cambio mayor en términos de la riqueza general de especies en el ecosistema (Dornelas et al., 2014), y no obstante tener gran impacto sobre las funciones (Clavel et al., 2011) y los servicios ecosistémicos (Cardinale et al., 2012).

1.2 La pérdida de bosque y sus promotores

La mayor causa de pérdida de biodiversidad es la pérdida de hábitat (IPBES, 2019; Newbold et al., 2015), provocada por la severa intervención humana en los sistemas naturales. A este hecho, se suma una crisis de los biomas del planeta (Hoekstra et al., 2005), en la que las regiones tropicales han sido las más perturbadas en las últimas décadas (Hansen et al., 2013ª). Muestra de eso son los 179, 000 km2 de cobertura forestal que ha perdido el bosque tropical seco y húmedo desde la década de los ochenta(Song et al., 2018). Además, los bosques tropicales remanentes están mayormente compuestos por fragmentos de tamaño pequeño (Taubert et al., 2018), que son más susceptibles a desaparecer (Hansen et al., 2020).

Los cambios en la cobertura forestal son causados por dos tipos de factores de origen antropogénico: directos e indirectos. Los promotores directos son aquellas actividades antrópicas que modifican directamente la cobertura forestal (Curtis et al., 2018). Algunas de las principales causas directas de la pérdida de bosque son la extensión de infraestructura (e.g. la red de carreteras), la explotación maderera y la expansión de la frontera agrícola,(Geist & Lambin, 2002), sobre todo esta última es reconocida como la actividad con mayor efecto sobre la cobertura boscosa en las regiones tropicales (Gibbs et al., 2010).

Por otro lado, los promotores indirectos (también conocidos como subyacentes) son aquellos factores demográficos, económicos, tecnológicos, políticos o culturales, que modulan la acción de los promotores directos de cambio (Geist & Lambin, 2002). La acción de los promotores subyacentes puede ser difusa y muy compleja, pues varios factores actúan de forma simultánea y su acción depende de las escalas de tiempo y espacio (Lambin & Meyfroidt, 2011); sin embargo, es posible distinguir algunos patrones. Los factores demográficos son reconocidos como unos de los promotores subyacentes más importantes (Aide et al., 2013), pues una concentración alta de personas demanda una gran cantidad de recursos e impone una mayor presión sobre los ecosistemas (Ehrlich & Holdren, 1971). De hecho, las altas tasas de crecimiento poblacional están asociadas a una mayor pérdida de bosque, tanto a escala regional (Wittemyer et al., 2008) como global (Defries et al., 2010). Otro tipo de promotores con gran relevancia sobre la cobertura forestal son los económicos.

La demanda de productos agrícolas en países desarrollados impulsa la realización de actividades agrícolas a gran escala en países en vías de desarrollo, provocando altas tasas de deforestación (Hoang & Kanemoto, 2021; Lambin & Meyfroidt, 2011). Contrario a esto, en las regiones en las que ha habido un cambio en las actividades económicas, de unas con mayor demanda de superficie (actividades agrícolas) a otras orientadas al sector industrial o de servicios (actividades no agrícolas), han mostrado una reducción de las tasas de pérdida de bosque e, incluso, su regeneración (Lambin & Meyfroidt, 2010; Meyfroidt & Lambin, 2011).

1.3 Las áreas naturales protegidas y la conservación

Ante el escenario de deterioro de los ecosistemas y la disminución de la riqueza de especies, las áreas naturales protegidas (ANPs) representan una de las mejores herramientas para la conservación de la biodiversidad (Watson et al., 2014). Las ANPs han demostrado ser efectivas para reducir la pérdida de bosque (Andam et al., 2008; Joppa & Pfaff, 2011) y para preservar especies (Geldmann et al., 2013; Gray et al., 2016);además, algunos estudios sugieren que son importantes para reducir la fragmentación de bosque (Hansen et al., 2020; Sims, 2014) y favorecer su regeneración (Borda-Niño et al., 2020). No obstante, estas aseveraciones aún son debatibles pues existen factores biofísicos — como el grado de aislamiento, la altitud y la pendiente— que pueden generar confusión y atribuir un mayor efectividad a las ANPs del que realmente tienen (Joppa & Pfaff, 2009), por lo que el efecto de las ANPs sobre la fragmentación y regeneración de bosque es aún un punto a esclarecer.

La IUCN clasifica a las ANPs en seis diferentes categorías de manejo según el tipo de actividades que son permitidas dentro de sus fronteras (IUCN, 2019a). Las categorías I-IV son reconocidas como ANPs de protección estricta pues su vocación está en la conservación de la biodiversidad y por ello prohíben la realización de actividades extractivas (Tabla 1.1). En contraste, las categorías V-VI son ANPs que promueven la conservación biológica pero que también reconocen la necesidad de permitir el uso sustentable de sus recursos por parte de la población (Tabla 1.1). Aunado a esto, con base en estos criterios internacionales, cada país establece sus propios esquemas de manejo y regula el tipo de actividades que son permitidas. En general, las ANPs de protección estricta son más efectivas para prevenir la pérdida de bosque que las de manejo sustentable de recursos (Wade et al., 2020), aunque a nivel regional, en países como México, Guatemala, Costa Rica y Tailandia el segundo tipo de ANPs es al menos tan efectivo como el primero (Blackman, 2015; Blackman et al., 2015; Ferraro et al., 2013; Figueroa et al., 2011).

Tabla 1.1: Categorías de manejo de las ANPs de acuerdo con la IUCN y su equivalente en México.

Esquemas de manejo de IUCN	Categorías IUCN	Esquema equivalente en México	Descripción breve
Reserva natural estricta	IUCN 1a	ZN-RB, S	Las ANPs protegen la biodiversidad y también, posiblemente, las características geológicas/geomórficas. Los visitantes, el uso de recursos y el impacto humano están estrictamente controlados y limitados para garantizar la protección de los valores de conservación.
Área silvestre	IUCN 1b	ADVC	Las ANPs suelen ser grandes zonas no modificadas o ligeramente modificadas, que conservan su carácter e influencia natural sin presencia humana permanente o significativa, y que se protegen y gestionan para preservar su estado natural.
Parque Nacional	IUCN II	PN	Las ANPs son grandes áreas reservadas para proteger los procesos ecológicos a gran escala, junto con las especies y ecosistemas característicos de la zona, que también proporcionan una base para oportunidades ambientales y culturales compatibles, espirituales, científicas, educativas, recreativas y de turismo.
Monumento Natural	IUCN III	MN, PN arqueológicos	Las ANPs protegen un monumento natural concreto, que puede ser un relieve, un monte marino, una caverna submarina, un elemento geológico como una cueva o incluso un elemento vivo como una arboleda antigua. Suelen ser zonas protegidas bastante pequeñas y a menudo tienen un gran valor para el turismo.
Área de manejo de hábitat/especies	IUCN IV	---------	Las ANPs tienen como objetivo la protección de especies o hábitats concretos y la gestión refleja esta prioridad. Estas ANPs necesitan intervenciones regulares y activas para atender las necesidades de determinadas especies o para mantener los hábitats.
Paisaje protegido	IUCN V	---------	Son ANPs en las que la interacción entre las personas y la naturaleza a lo largo del tiempo ha dado lugar a una zona de carácter distintivo con un valor ecológico, biológico, cultural y paisajístico significativo, y en las que la salvaguardia de la integridad de esta interacción es vital proteger.
Área protegida con uso sustentable de recursos naturales	IUCN VI	ZB-RB, APFF, APRN	Son ANPs que conservan ecosistemas y hábitats, junto con los valores culturales asociados y los sistemas tradicionales de gestión de los recursos naturales. Suelen ser extensas, con la mayor parte de la superficie en estado natural, en las que una parte está sometida a una gestión sostenible de los recursos naturales y en las que el uso no industrial y de bajo impacto de los recursos naturales es compatible con la conservación de la naturaleza.
Sin categoría asignada		NP degradados	

* ZN-RB: zona núcleo de las reservas de la biosfera, ZB-RB: zona de amortiguamiento (*buffer*) de las reservas de la biosfera, RB: reservas de la biosfera, PN: parques nacionales, MN: monumentos naturales, APRN: áreas de protección de recursos naturales, APFF: áreas de protección de flora y fauna, ADVC: áreas dedicadas voluntariamente a la conservación. Adaptado de Bezaury-Creel and Gutiérrez-Carbonell (2009) con información de (IUCN, 2019a).

A pesar de su gran valor en términos de conservación, las ANPs están sometidas a presiones antropogénicas que ponen en riesgo su conservación (Schulze et al., 2018). La caza furtiva, la contaminación, la introducción de especies exóticas, la extracción ilegal de recursos y la realización de actividades no permitidas dentro de las ANPs son algunos de los factores que las amenazan (Laurance et al., 2012b; Schulze et al., 2018; Wade et al., 2020). De esta forma, y no obstante el estado legal de protección, las ANPs sufren deforestación dentro de sus fronteras (Spracklen et al., 2015). Además, se estima que cerca del 30% de las ANPs a nivel global son inefectivas (Yang et al., 2021) para reducir la pérdida de bosque, y que alrededor de la tercera parte se encuentra altamente amenazada por presiones antropogénicas (Jones et al., 2018). Por ello, conocer los factores que favorecen la conservación de las ANPs es clave para asegurar su éxito en el futuro.

1.4 Posibles disyuntivas (*trade-off)* entre las ANPs y la sociedad

La interacción entre las ANP como políticas de conservación biológica y la sociedad es compleja. Por un lado, la conservación de las ANPs se ve afectada por factores socioeconómicos, como se explica en la sección anterior, y, por otro lado, el establecimiento de las ANPs tiene repercusiones de diferente índole sobre la población que habita en las zonas adyacentes a éstas, las cuales pueden ser positivas o negativas (Oldekop et al., 2016).

Las ANPs otorgan una serie de beneficios a las comunidades aledañas. De manera directa, contribuyen a la alimentación de la población por medio de los recursos naturales (Gardner & Davies, 2014; Goñi et al., 2008). Además, las ANPs proveen una serie de servicios ecosistémicos que mejoran el bienestar y la salud de las personas, a partir de la regulación hídrica, el secuestro de carbono, el sustento de suelos, el reciclaje de nutrientes y el control regional del clima (Daw et al., 2011; Dudley & Stolton, 2010). Asimismo, las ANPs contribuyen a la economía local, pues proveen fuentes de empleo en sectores como el turístico. Algunas estimaciones sobre el impacto económico que tiene el sistema de áreas protegidas de la República Democrática del Congo demostraron que éste puede contribuir a tres cuartas partes del PIB, el 90% del empleo y casi el 60% de las exportaciones de ese país (Scherl & Emerton, 2008).

Sin embargo, las ANPs también pueden tener un impacto negativo sobre la población, debido a que su establecimiento impone nuevas condiciones al uso que la población local hace de los recursos naturales, ya que, en mayor o menor medida, se restringen las actividades económicas extractivas dentro de éstas. Así, las políticas de protección de las ANPs pueden desplazar de su territorio a las poblaciones (West et al., 2006), o bien, criminalizar el uso tradicional de los recursos por parte de ellas (Freedman, 2002). Por esta razón, algunos autores señalan que las políticas restriccioncitas de las ANPs pueden reforzar la pobreza de las comunidades locales (Adams et al., 2004; Roe, 2008). No obstante, la evidencia al respecto de esta idea es mixta.

Por un lado, algunos trabajos documentan que las ANPs no tienen un efecto positivo sobre el reforzamiento de la pobreza sobre las comunidades locales, y que, por el contrario, pueden ayudar a combatirla (Naidoo et al., 2019; Oldekop et al., 2016). Por otro lado, algunos trabajos sugieren que bajo ciertas condiciones las ANPs pueden tener efecto positivo sobre el reforzamiento de la pobreza en comunidades locales (Ferraro & Hanauer, 2011; Naidoo et al., 2019). Así, factores como el tipo de manejo del ANP, las condiciones biofísicas en que se establece y los factores socioeconómicos cobran importancia. Además, poco se sabe sobre el efecto que puede tener el tipo de esquema de manejo de las ANPs en la capacidad de las comunidades locales para reducir sus niveles de pobreza.

En las ANPs existe una interacción mutuamente dependiente entre lo que ocurre en términos socioeconómicos y ecológicos (Fig. 1.1). Por un lado, los factores socioeconómicos, a través de la implementación de las políticas que rigen las ANPs, aportan suministros en términos de personal, recursos y materiales con los que las ANPs operan. Además, en el contexto socioeconómico se incluye la serie de promotores proximales y subyacentes, cuya interacción define el grado de impacto sobre las ANPs. Por su parte, las ANPs proveen de recursos, servicios ecosistémicos y oportunidades laborales a las comunidades locales, en función de sus características intrínsecas (i.e., tipo de bioma, factores biofísicos, tipo de manejo, presupuesto y gobernanza) y del grado de impacto al que están sometidas. La interacción entre las características intrínsecas al ANP y los factores socioeconómicos puede arrojar dos tipos de respuesta que son de interés para este estudio: de desarrollo económico, y de conservación de la cobertura forestal y la biodiversidad.

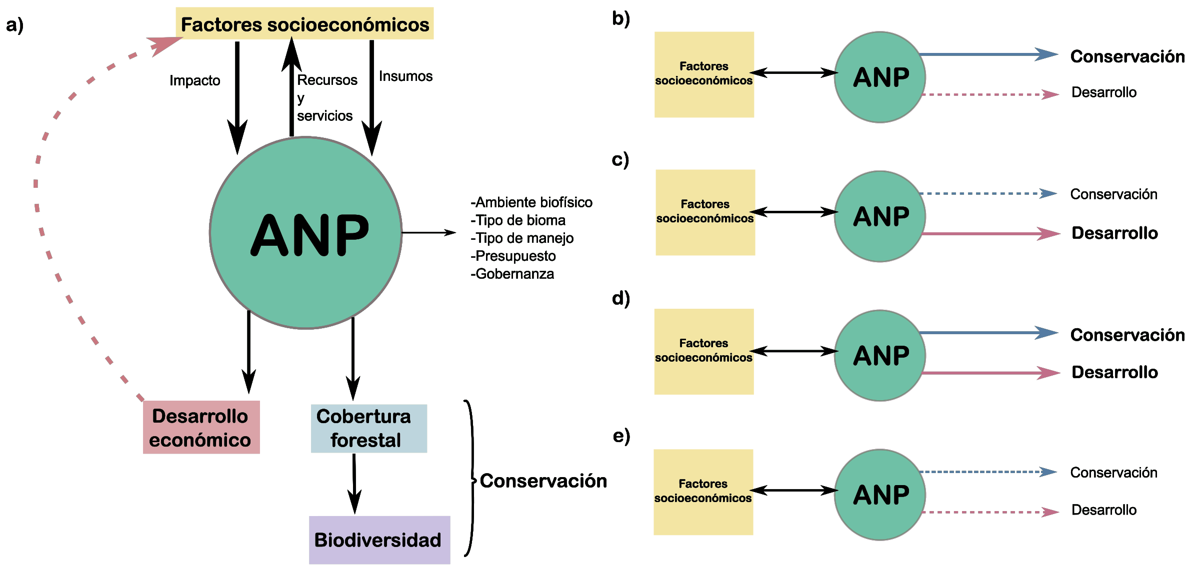

Figura 1.1: Marco conceptual propuesto en este trabajo sobre la interacción de las ANPs y los factores socioeconómicos ligados a las comunidades locales adyacentes, y los cuatro escenarios identificados en esta interacción. En el panel a) se muestra el marco conceptual general. Las ANPs poseen características intrínsecas denotadas por la flecha a la derecha. Los factores socioeconómicos definen el grado de impacto antrópico al que las ANPs son sometidas, pero también proveen los insumos necesarios y el marco político en el que éstas operan. Como resultado de esta interacción se producen dos tipos de respuesta que son de interés para este estudio: de desarrollo económico y de conservación. La línea discontinua entre desarrollo económico y factores sociales implica el reconocimiento de una posible retroalimentación de este sistema, pero que escapa a los alcances del estudio. Se plantean cuatro escenarios posibles de acuerdo con si se pondera una de estas dos metas (b y c), las dos (d) o ninguna (e). Las flechas discontinuas en b-e denotan respuestas desfavorecidas, mientras que aquellas que son continuas son respuestas favorecidas.

Las interacciones entre las ANPs y los factores socioeconómicos pueden clasificarse en cuatro escenarios diferentes que representan disyuntivas entre desarrollo y conservación. En el primero, como lo sugiere Adams et al. (2004), las ANP llevan a cabo eficientemente las labores de conservación, sin embargo, esto dificulta el desarrollo económico de las comunidades locales relacionadas con el ANP debido al poco acceso a los recursos naturales (Fig. 1.1b). En un segundo escenario (Fig. 1.1c), las ANPs implementan medidas que permiten el desarrollo económico de las comunidades locales (e.g., permitiendo el uso sustentable de recursos), pero en detrimento de su conservación biológica. En el tercer escenario, el más deseable, las ANPs logran buenos resultados en términos de conservación biológica sin comprometer el desarrollo económico de las comunidades locales (Fig. 1.1d). Por último, el cuarto escenario es el peor de los posibles pues en él las ANPs tienen subcomponentes ecológicos tan deteriorados que comprometen el desarrollo económico de las comunidades, lo cual a su vez refuerza la presión sobre las ANPs (Fig. 1.1e). Esto se ha visto en sistemas donde se llevan a cabo actividades económicas con bajo nivel de retorno económico pero un impacto ambiental alto y donde para tratar de conseguir insumos suficientes las actividades se mantienen en detrimento de los recursos naturales (Kovacic & Viteri Salazar, 2017).

Ante este panorama, es necesario comprender mejor, por un lado, los factores socioeconómicos que promueven la pérdida de bosque en las ANPs y sus consecuencias sobre la biodiversidad; pero también aquellos factores que comprometen el desarrollo económico de las comunidades aledañas a las ANPs, especialmente en el caso mexicano donde la evidencia al respecto de ambas situaciones es aún escaza. A continuación, se exponen las preguntas que guían esta investigación.

1.5 Preguntas de investigación

- ¿Cómo impacta el tipo de manejo de las ANPs en su efectividad para reducir la pérdida de bosque y a la capacidad de las comunidades humanas locales cercanas a las ANPs de reducir sus niveles de pobreza?
- ¿Qué tan efectivas son algunas ANPs mexicanas para reducir el impacto en los biomas que protegen, en términos de atributos espaciales de la cobertura de bosque (perdida, fragmentación y regeneración de bosque)?
- ¿Qué factores socioeconómicos influyen en la pérdida, fragmentación y regeneración de bosque en algunas ANPs mexicanas y que, por tanto, actúen como promotores de estos cambios?
- ¿Cómo ha cambiado la diversidad de especies en algunas ANPs mexicanas durante los últimos 30 años?
- ¿Cuáles son los promotores de los cambios en biodiversidad documentados en estas ANPs?

1.6 Objetivo general

Evaluar los compromisos de la interacción entre algunas ANPs mexicanas y las comunidades locales vecinas, en términos de conservación de la cobertura forestal y las especies que protegen, así como en el desarrollo económico de las comunidades.

1.7 Objetivos particulares

1. Evaluar las posibles disyuntivas (*trade-offs*) entre la efectividad de ANPs con esquemas de manejo contrastantes para prevenir la pérdida de bosque y la capacidad de las poblaciones adyacentes a éstas para reducir sus niveles de pobreza.
2. Documentar los patrones de cambio en la cobertura forestal de algunas ANPs mexicanas y evaluar su efectividad para reducir la pérdida y la fragmentación de bosque, así como para promover su regeneración.
3. Evaluar el efecto de algunos factores demográficos, económicos y políticos como promotores de pérdida, fragmentación y regeneración de bosque en algunas ANPs mexicanas.
4. Documentar los cambios en la riqueza y abundancia que han sufrido algunos gremios biológicos funcionales en algunas ANPs mexicanas durante los últimos 30 años.
5. Evaluar la relación entre los promotores subyacentes, promotores proximales y los cambios en la estructura forestal que explican los cambios en la biodiversidad de las reservas estudiadas.

Estos objetivos se desarrollan en los capítulos que componen esta tesis y que se describen brevemente a continuación. El primer capítulo es introductorio y establece las bases teóricas sobre las que se asienta el trabajo. Asimismo, se establece la problemática y los objetivos que persigue la tesis.

En el capítulo 2 se desarrolla el primer objetivo específico. Se utiliza el sistema mexicano de ANPs federales para evaluar su capacidad para reducir la pérdida de bosque y la reducción de la pobreza en los municipios que se intersecan territorialmente con las ANPs. Se pone a prueba la hipótesis de que los esquemas de manejo que permiten el uso sustentable de recursos (e.g., reservas de la biósfera) son efectivas para reducir la pérdida de bosque, a la vez que permiten la reducción de la pobreza, en comparación con esquemas más restrictivos (e.g., parques nacionales). Se utilizaron técnicas contrafactuales para controlar el efecto de factores de confusión que pueden interferir tanto en las estimaciones de pérdida de bosque y como en la reducción de pobreza.

En el capítulo 3 se desarrollan los objetivos específicos dos y tres. El capítulo se centra en analizar las reservas de la biósfera de la región tropical en México (menos de 27° de latitud N). En primera instancia, se utilizaron técnicas contrafactuales para poner a prueba la hipótesis de que las reservas reducen la pérdida y fragmentación del bosque, así como promover su regeneración. En segunda instancia, se explora el efecto de una serie de factores socioeconómicos como promotores subyacentes de la pérdida de bosque, su fragmentación y su regeneración. En particular, se pone a prueba la hipótesis de que una mayor disponibilidad de trabajos no agrícolas en torno a las reservas, así como una baja presión demográfica tienen una menor pérdida y fragmentación de bosque, además de una mayor regeneración.

El capítulo 4 desarrolla los objetivos específicos cuatro y cinco. Para ello, se utiliza una aproximación empleada en otras investigaciones y se aplicó un cuestionario a investigadores y funcionarios públicos con amplia experiencia en alguna de las catorce reservas de la biósfera que se seleccionaron para documentar los cambios en riqueza y abundancia de diferentes gremios biológicos durante los últimos 30 años. Además, se propone un modelo multivariado sobre las relaciones entre factores subyacentes, factores proximales y cambios espaciales en la estructura del bosque que pueden explicar los cambios en biodiversidad observados. A través de modelos de ecuaciones estructurales se pone a prueba un modelo integral para explicar los cambios en biodiversidad observados. Este capítulo compila información de más de 60 colaboradores, lo que representa un esfuerzo por documentar los cambios en biodiversidad en ANPs sin precedente en el país.

Finalmente, el capítulo 5 hace un análisis de los capítulos previos y resalta los mensajes principales encontrados en ellos. En este capítulo se ofrecen conclusiones generales al respecto de la tesis.

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Capítulo 2: El efecto de las ANPs sobre las comunidades locales: ¿refuerzan las condiciones de pobreza?

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Manuscrito en revisión en *Land Use Policy*

Abstract: Protected areas (PAs) are essential for biodiversity conservation, but their restrictive policies could accentuate poverty. Such a possibility may occur with the more restrictive PAs (e.g., national parks), which prioritize conservation while limiting the use of natural resources. However, less restrictive PAs, such as biosphere reserves, which allow the sustainable use of natural resources, may be better at alleviating poverty. However, such permissibility may reduce the effectiveness of preventing deforestation. Here, we assessed this conservation-development tradeoff by testing changes in marginalization (an indicator of poverty) and forest loss between two contrasting PAs management scheme types (MST, national parks and biosphere reserves) in Mexico. We quantified forest loss inside PAs and unprotected areas during the 2000-2019 period. Also, we contrasted marginalization changes during the 2000-2020 period between municipalities included in PAs (n = 288) and municipalities not directly influenced by PAs (n =1615). Using a matching analysis approach, we tested for differences between protected and unprotected areas and between MST, controlling for the potential effects of confounding factors (e.g., slope, altitude, distance to cities, economic sector). We also evaluated potential conservation-development trade-offs resulting from the interaction of MST with the biophysical-socioeconomic context. PAs did not accentuate marginalization comparing unprotected areas. After matching, both national parks and biosphere reserves showed similar evolution of marginalization and forest loss probability. However, national parks showed higher marginalization than biosphere reserves in areas far from cities and sites with poor agriculture suitability. Also, national parks showed higher forest loss than biosphere reserves in areas suitable for agriculture. Our results suggest that, in the Mexican protected areas system, MST does not cause a conservation-development tradeoff by itself. Nonetheless, the more restrictive MST does not provide greater protection to the forest than the less restrictive MST and, under certain biophysical conditions, may reduce the capability of communities to cope with poverty.

Keywords: protected areas, deforestation, poverty, trade-offs, biosphere reserves.

2.1 Introduction

Protected areas (PAs) are geographical spaces to conserve biodiversity by reducing anthropogenic threats, such as deforestation (Dudley, 2008). However, socioeconomic factors such as population density and productive activities in the communities living adjacent to PAs can affect their effectiveness in conservation (Auliz-Ortiz et al., 2022). Likewise, the presence of PAs also affects these neighboring communities in different ways. For example, PAs provide benefits such as ecosystem services, natural resources, and aesthetic and cultural values (Dudley et al., 2010). Also, PAs can attract governmental programs, resources from non-governmental organizations, and economic activities such as ecotourism (Ferraro & Hanauer, 2014; Mariyam et al., 2021). However, PAs can also negatively affect neighboring communities because, in general, they restrict most extractive activities within PAs (Adams et al., 2004). Some authors suggest that PAs can reinforce poverty in neighboring communities (Sanderson & Redford, 2004).

These complex relationships challenge human development and biodiversity conservation goals in PAs. On the one hand, development implies using natural resources and, consequently, a more significant impact on ecosystems (Sanderson & Redford, 2003). On the other hand, the restrictive policies of PAs can accentuate the condition of poverty in the communities living adjacent to them (Adams et al., 2004). However, poverty and high biodiversity coexist since the highest biodiversity regions in the world are in the poorest countries (Fisher & Christopher, 2007). Because of that, the Convention on Biological Diversity highlights that to achieve sustainable development, PAs should contribute to reducing poverty, besides contributing to biological conservation. Moreover, the International Union for Conservation of Nature (IUCN) recognizes that PAs must contribute to solving problems related to human development, such as poverty alleviation (IUCN, 2021). Therefore, besides increasing the number of PAs worldwide (Maxwell et al., 2020), we should also look for strategies to promote win-win scenarios between conservation and poverty alleviation.

Although all PAs are, to some extent, restrictive, they notably differ in their management objectives (Table 2.1). These management scheme types (MST) cause different relationships between PAs and the neighboring communities. More restrictive MST, such as national parks in the Mexican protected areas system, forbid most human activities (including the use of natural resources). In contrast, less restrictive MST, such as biosphere reserves, recognize the importance of human communities in conservation and allow some activities compatible with the sustainable use of natural resources in a buffer zone but maintain higher restrictions in a core zone. Nevertheless, less restrictive MST may not be as effective as more restrictive ones in preventing deforestation (Wade et al., 2020), which can thus limit their conservation value.

In addition, the biophysical context where PAs are embedded can also influence their ability to prevent forest loss (Joppa & Pfaff, 2009) and their effects on the poverty of local communities (Andam et al., 2010a). Thus, under specific biophysical contexts, PAs might reinforce forest loss and or poverty (Ferraro et al., 2011; Hanauer & Canavire-Bacarreza, 2015), but this issue requires more research for clarification.

This study used the Mexican protected areas system to assess whether PAs accentuate poverty. We also evaluated the conservation-development tradeoff by assessing how marginalization (an indicator of poverty) and forest loss differ between two MST: national parks (a more restrictive PA) and biosphere reserves (sustainable resources use PA). We quantified forest loss in the 2000-2019 period, inside and outside the PAs, and marginalization changes from 2000 to 2020 in 288 municipalities included in 46 Mexican PAs, and 1615 municipalities outside them (unprotected areas). High poverty near PAs could be simply related to PAs being located in marginalized communities and not to a PA effect *per se*. First, we tested whether marginalization in 2000 (i.e., our marginalization baseline year) differs between PAs and unprotected areas. Second, we compared the evolution of marginalization between protected and unprotected areas. Third, using a quasi-experimental approach (matching analysis), we assessed a possible conservation-development trade-off caused by the MST *per se* (after controlling biophysical and societal covariates), comparing temporal changes in marginalization and forest loss between national parks and biosphere reserves. Finally, we explored whether the MST promotes a conservation-development trade-off depending on biophysical and socioeconomic contexts.

2.2 Materials and methods

2.2.1 Study system

We used two contrasting MST of the Mexican protected areas system: national parks and biosphere reserves. National parks (IUCN category II) are the earliest MST in Mexico that emerged at the beginning of the XX century to protect nature. Nowadays, national parks are the most frequent MST in Mexico, and nature conservation remains their main objective (Table 2.1). On the other hand, biosphere reserves emerged in the late 1970s from Man and Biosphere program as tools for improving human livelihoods and safeguarding the natural and managed ecosystem. In the Mexican legislation, biosphere reserves correspond to natural protected areas having two zones: the core zone (IUCN category I), dedicated to strict protection, and the buffer zone (IUCN category VI), where different activities are allowed (Table 2.1). In Mexico, biosphere reserves are the second most frequent MST.

Table 2.1: Comparison between national parks and biosphere reserves. We show their corresponding IUCN category and the type of human activities allow in the Mexican legislation (DOF, 1988).

Management scheme types	IUCN category	Allowed activities
National parks	II	Ecosystem preservation, research, recreation, tourism, and education.
Biosphere reserves	Ia (core zone) and VI (buffer zone)	Core zone: ecosystem preservation, environmental education Buffer zone: Productive activities developed by local communities (e.g., agriculture, agroforestry, ecotourism, sustainable forest resource use)

Considering these differences in the origin of national parks and biosphere reserves, we only selected those PAs established before the year 2000 and used a time window (2000-2020) that allows the comparison of these MST. We used all terrestrial non-insular national parks and biosphere reserves with an area higher than 5 km^2. We selected this area size as some small PAs in big cities may cause a significant bias in our results. Thus, we assessed 46 PAs: 28 biosphere reserves and 18 national parks (Fig. 2.1, and Appendix 1A). We obtained PAs' polygon data from the National Commission on Protected Areas (CONANP, 2019).

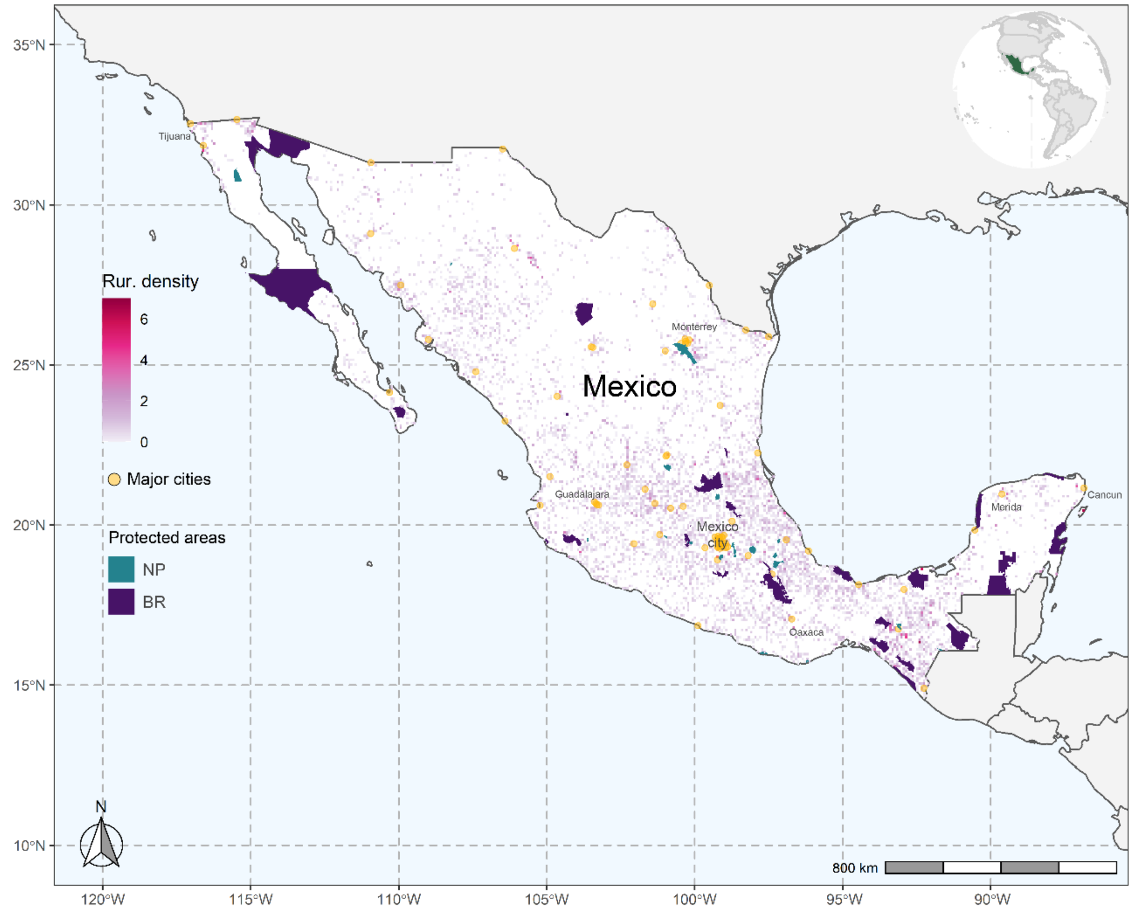

Fig. 2.1. Map showing the distribution of the 46 Protected Areas (PAs) used in this study. The colors in the map indicate the management scheme types (MST). BR: Biosphere Reserves, NP: National Parks. Names of major cities of the country (population over 100000 people) are indicated and the density of rural settlements (localities of 5000 people or less per square kilometer) is shown by the scale of color (Rur. density).

2.2.2 Marginalization data

Marginalization is a concept closely related to poverty that refers to the lack of development

Marginalization is a concept closely related to poverty that refers to the lack of development opportunities in society. A marginalized society cannot overcome its deficiencies because of structural problems (CONAPO, 2010). The marginalization concept includes four dimensions of precariousness: education, dwelling, rural population, and income. We calculate the Absolute Marginalization Index (AMI), which enables us to compare temporal changes in marginalization at the municipality level. Since AMI is not directly available from the Mexican institutional open data repositories, we estimated this index for the years 2000 and 2020 using the following formula (CONAPO, 2013b):

$$AMI = \frac{\sum_{j}^{9} I_{ij}}{9} \qquad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots(1)$$

where I_{ij} is the value of the indicator i for the municipality j. The marginalization index is composed of nine indicators: 1) percentage of the illiterate population of 15 years or more, 2) the percentage of the population of 15 years or more without elementary school, 3) percentage of people in dwellings without drainage service, 4) percentage of people in dwellings without electric energy, 5) percentage of people living in dwellings without piped water, 6) percentage of dwellings with some level of overcrowding, 7) percentage of people in dwellings with earth floor, 8) percentage of the people living in localities with less than 5,000 inhabitants, 9) percentage of people with an income equal or less than two Mexican minimum wages (~9 dollars per day). We calculated the difference in the marginalization index between 2020 and 2000 per municipality (see details in Appendix 2B). We classified municipalities into two groups: those included in one of the 46 PAs in Figure 2.1 (i.e., municipalities intersected with a PA, n = 288) and those not included in any PA (n = 1615).

2.2.3 Deforestation data

To assess the effect of PAs on forest cover, we gathered deforestation data for the period 2000-2019 from the Global Forest Change data (Hansen et al., 2013b), which provided data on global forest loss occurrences. For the entire Mexican territory, we obtained raster data at a 30-m spatial resolution on forest loss occurrences (a binary layer of forest loss/no loss) and the percentage of tree cover in 2000.

2.2.4 Matching analysis and covariates data

Often, PAs are established in isolated rural regions with hard access, including isolation, high elevations, and steeper slopes (Fig. 2.1, Joppa and Pfaff, 2009). Therefore, some authors point out the need to control these factors to determine the effect of PAs on poverty (Andam et al., 2010a) and deforestation (Yang et al., 2021). Because of that, our work adopts a matching analysis methodology. Matching is a quasi-experimental approach that statistically allows comparing sampling units with similar covariates and, therefore, testing the effect of a predictor variable without the effect of covariates (Ho et al., 2011). Our design allows comparing the effect of a treatment (e.g., PAs presence or strict protection PAs) against a control (e.g., unprotected areas or sustainable use of PAs) during the same period (not before-after control/impact design, dos Santos Ribas et al., 2020). It was not possible obtaining a reference point before PAs establishment (a before-after control/impact design) due to the low availability of data before 1980 (the average foundation year of the PAs is 1975, see Appendix 2A).

Different variables can influence marginalization index changes, such as the distance to cities, agriculture suitability, and non-farm occupation. Distance to cities is considered a proxy of access to markets that influence poverty, affecting product trade prices and revenues (Ferraro et al., 2011). Agriculture suitability is an index that identifies the environmental conditions in which agricultural activities are carried out based on climate, soil, and orographic variables. Poor agriculture suitability conditions accentuate poverty in communities that depend on agriculture activities (Ferraro et al., 2011). The non-farm occupation is the proportion of the population in a municipality working in industrial, professional, or service activities (then values near 0 indicate most labor is allocated in the agricultural sector while the value near 1 indicates less dependence on agricultural activities; Auliz-Ortiz et al., 2022). In Mexico, the higher the non-farm occupation, the lower the marginalized values in municipalities (Appendix 2A). We calculated the mean value of all these covariables for each municipality (see Appendix 2C). Previous studies have found significant relationships between variables assessed at the municipality level and PAs situation (Auliz-Ortiz et al., 2022; Figueroa et al., 2009). We classified the municipalities into two groups: municipalities included in PAs (those that intersect with a PA but distinguish between national parks and biosphere reserves); and municipalities with no direct influence of PAs (those that do not meet the previous condition). We also assessed the percentage of municipality territory overlapping polygon of each PA.

As described above for the marginalization index, the distance to cities, distance to roads, and agriculture suitability can also affect deforestation patterns. For example, deforestation can be higher far away from the cities due to an increase in the cost of agricultural production, which implies that farmer needs a greater arable surface to get profitable revenues (Angelsen, 2010). Moreover, distance to roads represents a proxy of accessibility, a key factor promoting forest loss (Kolb et al., 2013). Agricultural suitability is critical because PAs occur in places where conditions for agricultural activities are often precarious (i.e., steeper slopes, high elevation, poor soils, etc.). Thus, low deforestation rates in PAs may be due to environmental conditions rather than to protection itself (Joppa & Pfaff, 2009).

We divided the Mexican territory into a 1x1-km grid composed of 1,956,098 cells in which centroid we seeded a sampling point. In each sampling point, we extracted the information on forest loss occurrence, tree cover in 2000, distance to roads, distance to cities, agriculture suitability, biome, protection state (inside a PA or not), where was pertinent, the type of PA management, and the IUCN category in the specific location of each sampling point. Then, to compare forest loss that occurred during the study period (2000-2019) only in areas covered by forest in 2000, we filtered those sampling points with >10% tree cover. We chose this threshold because it allows discerning forests from non-forest with higher accuracy than others (overall accuracy = 0.87, Appendix 2A, Appendix 2D). We include the sampling points included inside biosphere reserves (n=26,411), national parks (n=3,201), and unprotected areas (n=574,298) obtaining a total of 603,910 sampling units.

We used a propensity scores matching approach (Austin, 2011) to balance the covariates in the marginalization analysis. We performed two different matching analyses: the first to compare municipalities influenced by PAs (treatment) and those with no direct influence of PAs (control); and the second to compare national parks (treatment) and biosphere reserves (control). We estimated the propensity scores using a model with binomial error distribution with logit function and the nearest neighbor covariate matching algorithm allowing replacement. We use the treatment as the response variable. After matching all the standardized mean covariates, differences were below 0.2, indicating a good covariate balance between the control and treatment municipalities (see Appendix 2E). We did the matching analysis using the *MatchIt* package (Ho et al., 2011; R Core Team, 2021).

We also used the propensity scores matching to balance the covariates in forest loss analysis. In a similar way we did with marginalization data, we performed two matching analyses with forest loss data. The first analysis compared protected (treatment) and unprotected (control) areas, while the second one compared national parks (treatment) and biosphere reserves (control). We used generalized linear models with binomial error distribution with logit function and a 1:1 nearest neighbor matching algorithm allowing replacement where the response variable was the treatment. All standardized mean differences for the covariates were below 0.2, indicating a good balance (Appendix 2E).

2.2.5 Testing the effect of MST on forest loss and marginalization

We used linear models to test whether the baseline marginalization year (2000) differed between municipalities included in PAs and those without the direct influence of PAs (unprotected areas). We did this analysis with pre-matching and matched data. We also used linear models to test for differences between municipalities in the temporal change (2000-2020) of the marginalization index, separately for pre-matching and matched data. Furthermore, we tested differences between national parks and biosphere reserves in the temporal marginalization change using a linear mixed effects model (LMEM) and the matched data, including the economic region of each PAs as a random variable (Appendix 2A). Economic regions encompass territories with similar resource availability and financial activities (Bassols Batalla, 2006; CONABIO, 2010) but the regions differ in biophysical characteristics (Appendix 2A). We tested for differences in forest loss between MST using a generalized linear mixed effects model (GLMEM) with binomial error distribution (we included the biome as a random effect) and matching data. Since biosphere reserves have two zones with different IUCN categories (Table 2.1, and Appendix 2A), we performed these models using biosphere reserves as a whole (core and buffer areas together) and separated each zone. Furthermore, in our models we included the age (years since decree) and size (area in hectares) of PAs, and the percentage of municipality territory overlapping the polygon of each PA as covariables. On one hand, populations in the vicinity of younger PAs may suffer a higher marginalization than those near older PAs, because the first ones are still to be adapted to the PAs' policies that limit the access to resources. On the other hand, smaller PAs suffer more deforestation than large ones because they are more likely to be disturbed by human activity (Blackman et al., 2015). Moreover, overlapping PAs in municipality territory can affect marginalization since local communities with higher overlap can be more restricted by PAs' policies (Job et al., 2021). We used the size and age of PAs as predictors in the forest loss and marginalization models and the overlapping of PAs only in this last one.

We weighted all the models to account for the differences in the number of sampling units of the control and treatment using the weights estimated by the matching model. Finally, we calculated the odds ratio from the forest loss model. Odds ratios are used in fields such as medicine (Ong et al., 2018) and ecology (Rita et al., 2008) to compare the direction of the effect of a response variable in a logistic model (in our case, whether national parks avoid or promote forest loss regarding biosphere reserves) and its magnitude.

2.2.6 Interaction between MST and biophysical-socioeconomic context.

We assessed potential biophysical-socioeconomic scenarios where marginalization and or forest loss accentuates. We used linear models with the marginalization index in 2020 as a response variable and the interaction between MST and distance to cities, agriculture suitability index, and non-farm occupation as predictors. Similarly, we used generalized linear models with binomial error distribution and forest loss incidence as a response variable to explore potential biophysical scenarios where forest loss accentuates. In this case, we used the interactions between MST and distance to cities, agriculture suitability index, and distance to roads as predictor variables. In all these analyses, we used matched data.

2.3 Results

2.3.1 Baseline marginalization

Before matching, the municipalities included in PAs showed a lower mean marginalization index in 2000 than the unprotected areas. However, after matching, we found no differences between municipalities (Fig. 2.2a).

2.3.2 Changes in marginalization in protected and unprotected areas

The municipalities included in PAs had a significantly lower reduction of marginalization than unprotected areas, but this difference disappeared when we did control for covariates (i.e., after matching; Fig. 2.2b).

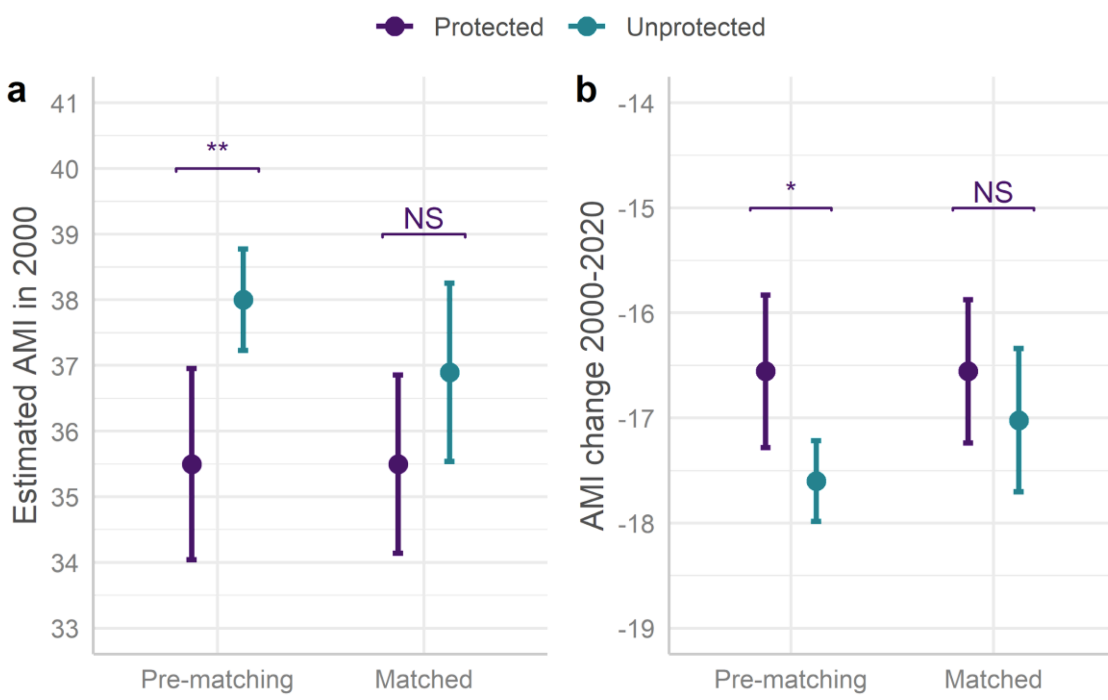

Fig. 2.2. Comparison of estimated mean absolute marginalization index (AMI) for the 2000 year (a) and mean relative AMI change (b) before and after the matching analysis. Error bars correspond to the 95% confidence intervals. ns: non-significant difference, * p<0.05, **p<0.01, ***p<0.001.

2.3.3 Marginalization and forest loss trade-offs in biosphere reserves and national parks

We found that the MST, size, and age of the PA as well as the percentage of municipality territory overlapping with PAs did not affect marginalization change when comparing treatment (national parks) and control (biosphere reserves) units and controlling for covariate differences (2.3a). MST and the age of the PA also did not affect forest loss probability (2.3b), even comparing buffer zones of biosphere reserves (IUCN category VI) and national parks

(IUCN category II, see Fig. 2.4). However, we found that larger PAs had less forest loss probability (3b).

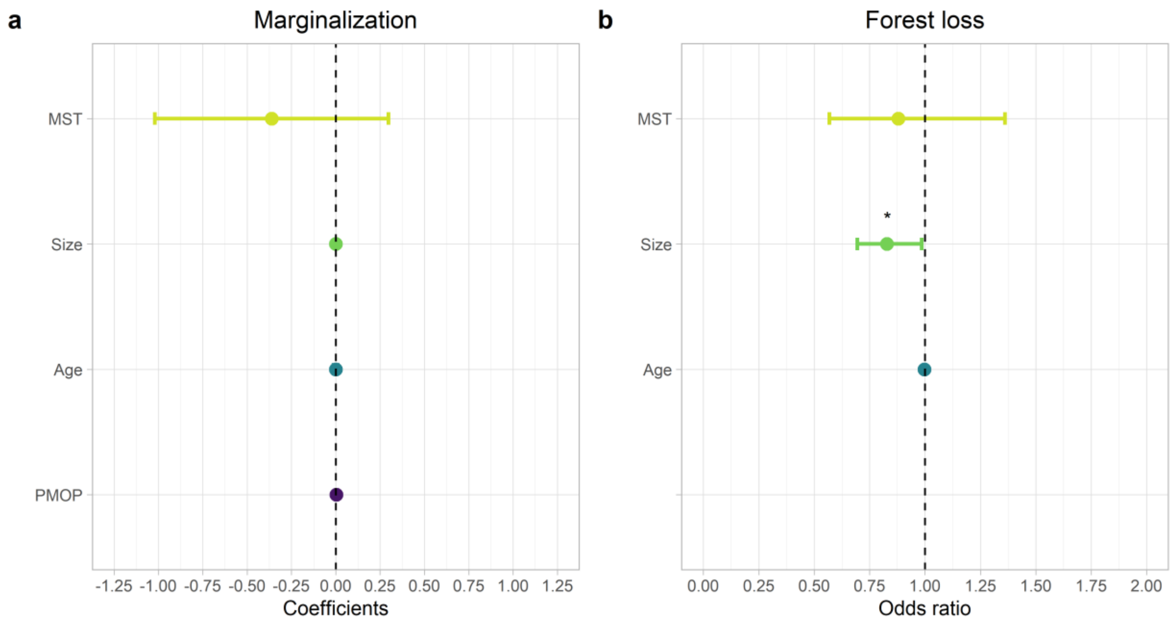

Fig. 2.3: Effect of the management scheme type (MST), the size (area in hectares) and age (years since decree) of the protected areas (PAs), and the percentage of municipality territory overlapping PAs (PMOP) on the change of marginalization (a) and forest loss (b) in the models that control for covariates differences (matched data). Since forest loss is not assessed at the municipality level PMOP was not included in this model. Points in panels indicate the mean coefficient and the mean odds ratio, respectively, while error bars correspond to the 95% confidence intervals. In panel b, values less than 1 indicate that the predictor has a protective effect against forest loss, while values higher than 1 indicate that the predictor promotes forest loss. The dashed line in each panel denotes no effect. *$p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

2.3.4 Interaction of MST and biophysical-socioeconomic context

Distance to cities showed significant interaction with the MST in marginalization models. Marginalization was similar in national parks and biosphere reserves near cities; however, marginalization was higher in national parks far from cities (at a distance higher than 40 km, Fig. 2.5a). We found no significant interaction between MST and distance to cities on forest loss data (Fig. 2.5d).

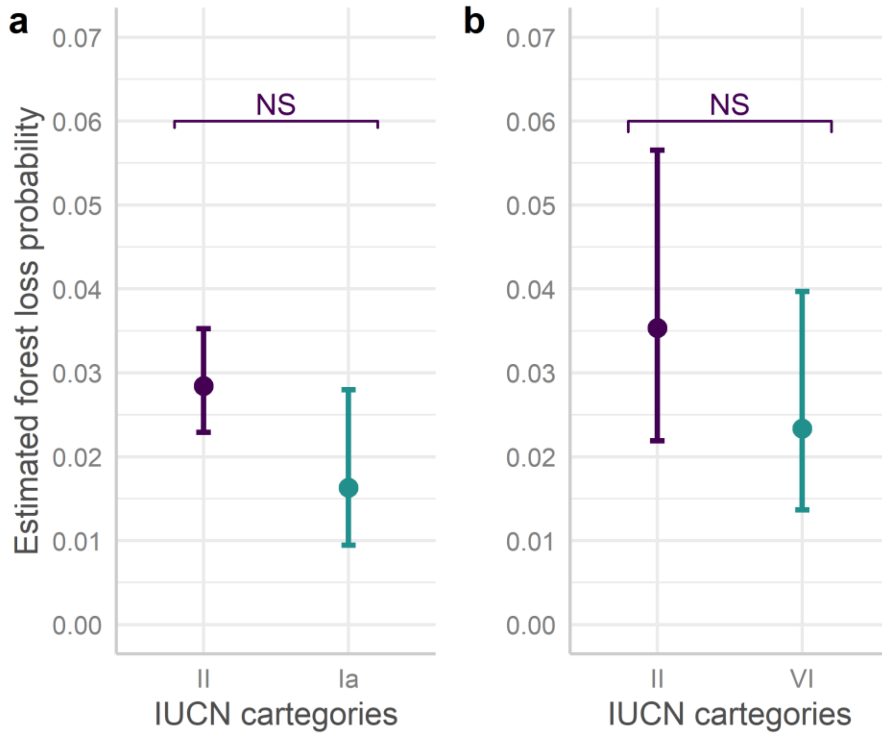

Fig. 2.4. Comparison of forest loss probability in different IUCN categories: (a) national parks (category II) vs. core zone of biosphere reserves (category Ia), (b) and national parks vs. buffer zone of biosphere reserves (category VI). Points indicate the estimated mean value, while error bars are 95% confidence intervals.

We found a significant interaction between the agriculture suitability index and MST in the marginalization model. While marginalization in biosphere reserves was similar along an agriculture suitability gradient, it was higher in national parks where agriculture activities are less suitable and lower in areas with more suitable conditions for agriculture (Fig. 2.5b). Forest loss was similar in national parks and biosphere reserves under low agriculture suitability but increased in national parks where agriculture suitability was high (Fig. 2.5e). We did not find significant interactions between MST, non-farm occupation, and distance to roads (Fig. 2.5c and 2.5d).

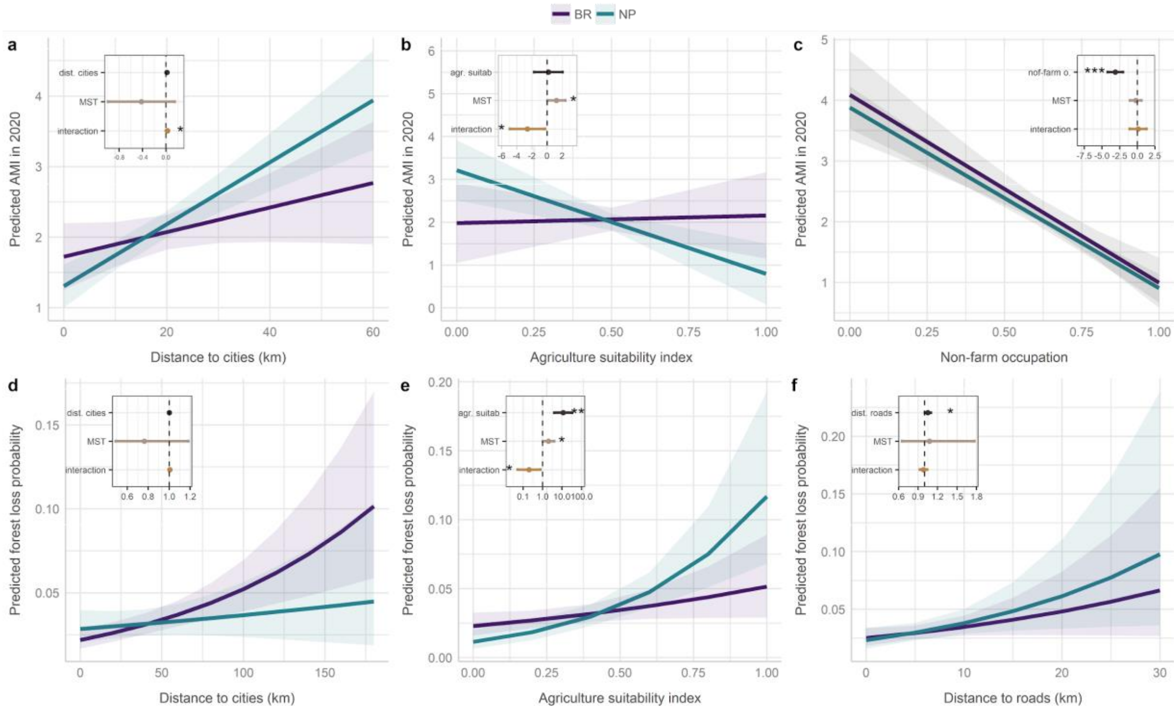

Fig. 2.5. Plots of the interaction between management scheme types (biosphere reserves in purple and national parks in green) and biophysical-socioeconomic variables (distance to cities, agriculture suitability, distance to roads, and non-farm occupation) as predictors of absolute marginalization index in 2020 and forest loss probability in the period 2000-2019. Lines are predictions of linear (a, b, c) and logistic (d, e, f) models. Shadows represent the 95% confidence intervals. The inset plots in each panel show the mean coefficients (and 95% confidence intervals) resulting from the models. In the case of forest loss, coefficients represent odds ratios. Asterisks denote a significative effect of the variable (* $p<0.05$, ** $p<0.01$, *** $p<0.001$) while variables with no asterisks are not significative ($p>0.05$).

2.4 Discussion

Our study analyzes the effect of PAs management scheme types on marginalization and forest loss reduction. Overall, our results do not support the idea that the Mexican PAs system reinforces marginalization compared to unprotected zones. After controlling for biophysical and socio-economical covariates, marginalization reduced similarly in municipalities influenced by restrictive PAs (national parks) and PAs enabling sustainable resource use (biosphere reserves). However, we found differential effects of MST on marginalization depending on biophysical and socio-economical contexts. Marginalization in national parks varied across contexts to a larger extent than in biosphere reserves, exhibiting a potential conservation-development trade-off. We also find higher forest loss in more restrictive MST, especially under suitable agricultural areas. As discussed below, our results highlight the importance of evaluating conservation and development goals and have important implications on PA sustainability.

2.4.1 Does PAs are located in places with higher baseline marginalization?

Protected areas were not established in more marginalized municipalities in 2000 than unprotected zones. Previous studies have assessed the effects of PAs on poverty using indicators measured at a single moment (Andam et al., 2010a; Naidoo et al., 2019). This approach, however, does not enable us to distinguish whether the poverty of societies near PAs emerges from restrictions imposed by PAs or because PAs establish in the areas already enduring poverty (i.e., different baseline poverty). Because of that, it is crucial to clarify this point by determining the baseline marginalization of municipalities near the PAs. Thus, our results indicate that marginalization was not particularly acute in the municipalities near the PAs in 2000 regarding unprotected areas.

2.4.2 Does PAs accentuate marginalization in neighboring communities?

Our results show that PAs, as a whole, do not reinforce marginalization in their neighboring communities. Without controlling the biophysical context, a comparison of marginalization changes between municipalities within PAs and unprotected areas may lead to the wrong conclusion that PAs limit the development of neighboring communities, as suggested elsewhere (e.g., Sanderson and Redford, 2003). However, in concordance with other studies (Naidoo et al., 2019; Sims & Alix-Garcia, 2017), our results show that PAs have no significant influence on poverty. PAs may help prevent society from falling into poverty through different mechanisms. PAs provide a series of ecosystem services to the population, reducing its vulnerability to poverty (Dudley et al., 2010). In addition, tourism activities associated with PAs help increase revenues obtained by the people (Ferraro & Hanauer, 2014). Moreover, governments and civil society promote conservation projects in PAs and their surrounding areas with national and international capital that provide labor opportunities for the population (e.g., the conservation program for sustainable development of the Mexican government, DOF, 2020).

2.4.3 Does MST promote a tradeoff between marginalization and deforestation reduction?

Biosphere reserves and national parks similarly reduced marginalization once controlling for covariate differences. This result is in line with previous studies in Latin American PAs showing that contrasting MST delivery similar results on poverty once controlling for confounding factors (Corral et al., 2016). Our results show that at a regional level, the MST of PAs does not represent a factor that determines the capacity of the communities to deal with poverty, and other variables are more important as determinants of poverty. According to the literature, poverty is a systemic problem that is associated with low levels of economic growth, low infrastructure development, and low investment in developing countries (Brady, 2019) and rural regions of middle-income countries such as Mexico (CONEVAL, 2019; Garza-Rodriguez, 2018). The increase in non-farm revenues derivate from governmental assets has helped communities deal with poverty (Scott, 2007). This increase seems to be one of the most critical factors in the study region since we find that, regardless of PAs, marginalization reduces significantly with non-farm occupation (Fig 2.5c, $R^2 = 0.58$, $p < 0.001$).

Once controlling for confusing factors, biosphere reserves were as effective as national parks in preventing forest loss. This result contrast with global patterns of forest loss, which point out that less strict protection (PAs in IUCN categories V and VI) have more significant forest loss than more rigid protection (PAs in IUCN categories I and II, Leberger et al., 2020; Wade et al., 2020). Nonetheless, there exists some evidence at a regional scale (e.g. Costa Rica, Thailand, Guatemala, and Mexico) that indicate that the less strict MST can be at least as effective as stricter schemes to avoid forest loss (Blackman, 2015; Blackman et al., 2015; Ferraro et al., 2013). Since biosphere reserves have two zones (Table 2.1), one possibility is that the high protection zone (the core zone, IUCN category Ia) is more effective. In contrast, the sustainable resource use zone (buffer zone, IUCN category VI) is less effective. Nonetheless, we find no evidence to support this idea (Fig. 2.4). Previous studies have highlighted the advantages of including communities in the management and conservation of protected areas (Porter-Bolland et al., 2012; Schultz et al., 2011). Involving communities in PAs management allow them to create a sense of ownership of the territory (Durand & Jímenez, 2010), which may impacts positively the way people relate to nature and the decisions that they make concerning it (Durand & Lazos, 2008; West & Brockington, 2006).

Protected areas' age and the percentage of municipality territory overlapping PAs had no important effect on our results. Since our study is not experimental, we are not exempt from the effect of not-controlled variables that may affect the phenomena we study. Because of that, we have made efforts to take into consideration some covariables intrinsic to the PA (e.g., age and size) and relatives to the spatial scale of analysis. The results suggest our study design is sufficiently robust in terms of the spatial (municipality) and temporal scale (the period 2000-2020) to analyze the possible trade-offs between conservation and development in Mexican PAs.

The finding that PAs with a larger size have less forest loss probability than smaller PAs parallels the results of previous studies (Blackman et al., 2015; L. N. Joppa et al., 2008). According to these studies, PAs' larger size allows deforestation near their edges, protecting their inner zone. In the context of the single large or several small (SLOSS) conservation debate (Mccarthy et al., 2011), our results suggest that, in terms of forest loss, the amount of area should be considered in future PAs planning. Nonetheless, in terms of biodiversity, other features such as connectivity, emerge as important variables in a proper PAs management requirement (Brennan et al., 2022).

In our studied PAs, the MST *per se* (i.e., controlling for covariate differences) does not create a trade-off between conservation and development. This finding concurs with the study of Corral et al., (2016) in Peru and contributes to the conservation/development compatibility in the PAs debate (Adams et al., 2004). Also, our results indicate that PAs, both, reduces forest loss and do not obstacle marginalization reduction in local communities regardless of the MST. However, there is the possibility that a potential trade-off exists due to the interaction of different biophysical and socioeconomic contexts with PAs.

2.4.4 Is there a conservation-development trade-off resulting from PAs interaction with the biophysical-socioeconomic context?

Although our results indicate that the MST did not affect poverty, we found more marginalization in PAs with higher restrictions for using resources under specific contexts. It has been documented that rural communities living far from cities face problems obtaining profitable revenues from agricultural products since a higher distance to cities carries higher trade prices (Sims, 2010). On the other hand, biophysical factors such as climate, slope, elevation, and soil quality are vital in determining whether agriculture activities are profitable or not (Zabel et al., 2014). Steeper slopes, poor soils, and drier climates require higher labor and capital investment to produce agricultural products, making it difficult to obtain profitable revenues (Angelsen, 2010; Pfaff et al., 2014). Our results show that under such circumstances, Mexican restrictive national park policies may represent a factor against the capability of local communities to deal with poverty (Fig. 2.5a and 2.5b). This finding aligns with a study conducted in Costa Rica showing that PAs with a more restrictive MST reinforce poverty in areas with a poor agricultural aptitude and far from cities (Ferraro & Hanauer, 2011). In contrast, in a global analysis, Naidoo et al., (2019) found that PAs enabling multiple uses of resources (IUCN categories V and VI) provide higher benefits to households than PAs with strict protection policies. Likewise, previous works document that the higher involvement of local communities in some PAs delivers more significant benefits to the population (Bray et al., 2008). Thus, our study suggests that PA managers should focus on involving neighboring communities in participatory conservation activities. An example of this idea is the conservation program for the Mexican government's sustainable development (CONANP, 2021). Moreover, the involvement of local stakeholders in governance and tourism activities in Vizcaino Biosphere Reserve in Mexico has also outcome positive results both in ecosystem conservation and the economy of local people (Mayer et al., 2018). Thus, involving local people in mechanisms promoting their sustainable development should be especially relevant where socioeconomic and biophysical conditions challenge human development.

More restrictive management schemes (national parks) were less effective in preventing forest loss than PAs enabling sustainable use of resources (biosphere reserves) in Mexican regions with higher agriculture suitability. This finding suggests that additional restrictions in PAs do not necessarily outcome better results in preventing forest loss, just as previous studies show (Blackman, 2015; Ferraro et al., 2013). Some authors indicate that stricter protection of PAs may reduce forest loss (Wade et al., 2020). However, some of this protection is due to PAs' biophysical conditions limiting human activities (Joppa & Pfaff, 2009). Thus, one possible explanation of our results is that national parks are more frequently located in places with poor agriculture suitability than biosphere reserves, conferring to national parks an additional protective effect from forest loss. If this is true, national parks might perform less effectively in the absence of such extra protective effects. We found evidence that national parks are more frequently established in areas less suitable for agriculture than biosphere reserves (mean suitability index in national parks 0.30 ± 0.01 CI 95%, biosphere reserves 0.50 ± 0.01, $p < 0.001$). When we controlled the effect of biophysical covariates with matching technics, the extra protective effect of national parks disappeared (see Appendix 2A). Also, Sims and Alix-Garcia (2017) documented a higher effect of biosphere reserves than stricter PAs in Mexico once controlling for covariates. However, it is important to remark that, in practice, PAs are not isolated from their biophysical and socioeconomic context, which may influence forest cover spatial changes (Auliz-Ortiz et al., 2022). Therefore, stakeholders involved in the management of PAs should consider the possible interactions between PAs, their intrinsic characteristics (e.g., MST, budget, personnel, governance, among others), and their biophysical and socioeconomic contexts. Such comprehensive planning could help find scenarios where local communities' development and conservation do not conflict with each other.

2.5 Concluding remarks

Although PAs are not specially designed as tools to alleviate poverty, they can be adapted to facilitate the implementation of policies designated by each country to overcome poverty. According to our study, the management scheme types of Mexican PAs do not represent a factor that per se creates a trade-off between conservation and development. Instead, biophysical and socioeconomic conditions that make difficult the action of poverty alleviation politics (e.g., low access to trade markets and poor conditions for agricultural development) may interact with MST and create potential trade-offs. Thus, it is necessary to have an appropriate diagnosis of PAs situation that include the possible effects of socioeconomic context besides further research to identify the key factors involved in PAs conservation-development trade-offs.

As Naughton-Treves et al. (2005) recognized, we need to promote the environmental agenda beyond PAs boundaries and promote the economic development of neighboring communities. The long-term viability of the PAs could depend on the capability of the stakeholders to integrate efforts in sustainable development plans that reduce the impacts on ecosystems and foster the improvement of the livelihood of the population.

Conflict of interest

The authors have no conflicts of interest to declare.

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Capítulo 3: Promotores subyacentes de cambios de la cobertura forestal en reservas de la biosfera mexicanas

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Abstract

Protected areas can prevent forest loss, but their effects on forest fragmentation and forest regrowth are poorly understood. Furthermore, the importance of protected areas in shaping these forest spatial changes may depend on different socioeconomic drivers (e.g. population size, distance to cities, proportion of local people working in non-farm occupation), but the empirical evidence on such dependence is very scarce. Here, we used contra factual technics to assess whether biosphere reserves (n = 19) in the Mesoamerican biodiversity hotspot can reduce forest loss and fragmentation and promote forest regrowth during the period 2000-2020. We used satellite imagery and governmental data to assess the socioeconomic factors driving these changes. Particularly, using multimodel inference analysis, we tested whether higher non-farm occupation, combined with low demographic pressures, reduces forest loss and fragmentation and promotes forest regrowth. We found that reserves reduce forest loss and preserve less-fragmented configurations, however, they neither reduce fragmentation rate nor promote forest regrowth. Forest loss rate inside the reserves decreased as non-farm occupation enhanced and the density of rural settlements decreased. Therefore, promoting higher opportunities in non-farm economic activities and planning rural settlements distribution around reserves could help to increase the effectiveness of reserves for forest conservation.

Keywords: fragmentation, matching analysis, Mesoamerican biological corridor, non-farm activities, protected areas, underlying drivers.

3.1 Introduction

Forest loss is considered a major driver of biodiversity loss (Watling et al., 2020). This process is particularly acute in tropical forests (Hansen et al., 2013a). Forest loss is also causing the fragmentation of tropical forests worldwide (Taubert et al., 2018), which can increase the susceptibility of remaining patches to post-fragmentation threats such as negative edge effects, logging, hunting, and fires (Malhi et al., 2014). To prevent this global pattern of tropical forest degradation, protected areas (PAs) are frequently considered a major strategy for biodiversity conservation since they can reduce deforestation (Spracklen et al., 2015) and protect species.

Despite their conservation value, PAs face a growing threat due to human activities (Jones et al., 2018), which result in forest cover changes inside and outside them (Wade et al., 2020) impacting their conservation. Thus, forest cover changes may affect PAs conservation in several ways. On the one hand, forest loss is identified as the main cause of biodiversity loss (Newbold et al., 2015). On the other hand, despite that forest fragmentation may have negative effects (Ewers & Didham, 2006), most of them are neutral or positive (Fahrig, 2017), nonetheless this is currently under debate (Fletcher et al., 2018). Furthermore, forest regrowth in the tropical forest promotes the recovery of biodiversity (Rozendaal et al., 2019). Thus, evaluating the pattern of forest cover changes in PAs is critical to improving management and conservation strategies.

Understanding the causes of forest cover changes in PAs is also of paramount importance. Overall, two types of drivers can be distinguished: proximate and underlying. The proximate drivers correspond to human activities that directly modify the forest cover, such as infrastructure extension, agricultural expansion (both subsistence and large scale), and wood extraction (Curtis et al., 2018). Environmental factors, such as hurricanes, fires, landslides, severe droughts can also act as proximate drivers of forest cover changes (Geist & Lambin, 2001). The underlying drivers are related to factors that indirectly modify the forest cover by modulating proximate drivers, such as demographic, economic, technological, political-institutional, and cultural factors (Geist & Lambin, 2001). For example, demographic-related factors, such as human population growth and population density, are recognized as major drivers of deforestation (Aide et al., 2013) as they are linked to forest resource consumption. Economic factors also play a critical role in forest cover dynamics. Access to markets is a key factor influencing trade prices for agricultural products, which in turn affects deforestation (Angelsen, 2010).

Protected areas in the tropics are usually inhabited by rural communities that subsist from agriculture activities. These communities derive low income from agriculture due to the high costs of product transportation (Angelsen, 2010), which may increase deforestation to expand the cultivated area seeking higher income (Ferraro et al., 2011). Also, in tropical regions, a shift from economies based on agriculture to non-agricultural economies has been indicated as one of the drivers producing a decrease in forest loss rates and an increase in forest regrowth, a process known as forest transition (Meyfroidt & Lambin, 2011). According to the forest transition theory, non-agricultural economies reduce the pressure on forests because local people obtain their income from activities representing less change in land use (Rudel et al., 2005). For example, rural communities having labor opportunities on tourism (Hoang et al., 2014) and industrial activities (Wunder, 2003) carry out less deforestation than communities depending only on agriculture (Kovacic & Viteri Salazar, 2017). However, the effect of deforestation reduction, associated with the emergence of non-farm occupation in tropical regions, on PAs conservation function is still an open issue, and empirical evidence is very scarce.

Here, we first apply contrafactual matching methods to test the hypothesis that PAs not only prevent forest loss but also fragmentation and promote forest regrowth during the period 2000-2020. Second, we determined the effect of some underlying socioeconomic drivers on forest cover changes within some reserves located in Mesoamerica. We hypothesize that economies based on non-farm occupation, combined with low demographic pressures, reduce forest loss and fragmentation, and promote forest regrowth inside reserves.

3.2 Methods

3.2.1 Study system

We selected all Mexican MAB-UNESCO Biosphere reserves (hereafter reserves) located in the Mesoamerican hotspot, which were established before the year 2000 (Fig. 3.1, and Appendix 3A). We exclude from our analysis the biosphere reserves La Encrucijada and Pantanos de Centla because of the low performance of forest/no forest identification (see below), therefore we include 19 reserves. The reserves show a gradient of forest loss, with the remaining forest cover within the reserves ranging from 36% to 98%. We combined the Lacan-Tun/Montes Azules and Los Petenes/Ria Celestún reserves into two reserve complexes called Lacandona and LPRC, respectively, because they shared borders, same biome, and management.

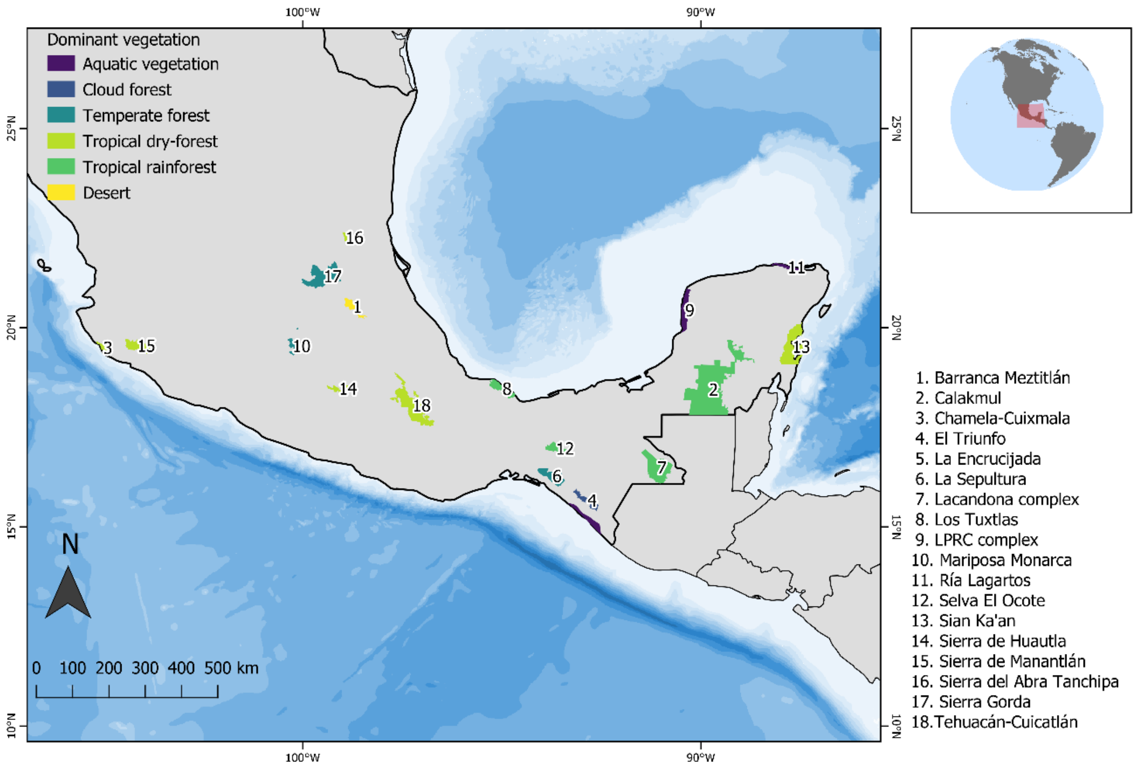

Figure 3.1. Location of the studied biosphere reserves in Mexico. Colors show the dominating vegetation type in each reserve. All the Mexican territory belove the Tropic of Cancer not included in any protected area was considered in our study as unprotected zones.

3.2.2 Land cover classification

To characterize forest changes inside reserves and unprotected areas (areas in Mexican tropic non included in any PA, see Fig. 3.1), we performed a supervised classification of Landsat images for the years 2000 and 2020. We used three major classes: forest, no-forest, and water. We evaluated the accuracy of this classification using a set of independent class cover validation points and determined that the mean (± SD) value of the overall satellite image classification accuracy was 94.4% ± 0.02 for the 2000 year and 94.4% ± 0.04 in 2020 (Appendix 3B). We conducted a cover change analysis to identify the areas with forest loss, forest regrowth, and no change for a period of 20 years, from 2000 to 2020 in the whole studied area.

We also calculated forest loss rate, forest regrowth rate, and forest fragmentation rate within reserves. To calculate forest loss rate (FLR) and forest regrowth rate (FRR) we use the next formula:

$$FLR|FRR = \left(1 - \left(1 - \frac{FLA\ |FRA}{A_0}\right)^{1/t_1 - t_0}\right) * 100 \qquad (1)$$

where A_0 is the area covered by forest at 2000 (t_0) and t_1 represents the year 2020. FLA is the area covered by forest at t_0 but no forest at time t_1 while FRA represents the area with the opposite transition. We quantified the number of forest patches in the year 2000 (NP_{2000}) and the year 2020 (NP_{2020}) and calculated forest fragmentation rate (FFR) with the formula:

$$FFR = \left(1 - \left(1 - \frac{NP_{2020} - NP_{2000}}{NP_{2000}}\right)^{1/t_1 - t_0}\right) * 100 \qquad (2)$$

3.2.3 Matching analysis for forest loss, forest regrowth, and fragmentation data

Protected areas are often located in isolated places with steeper slopes, higher elevations, and poorer soils than unprotected places (Joppa & Pfaff, 2009). Thus, a simple comparison of forest changes between protected and unprotected regions may overestimate the effect of legal protection (dos Santos Ribas et al., 2020). Matching analysis is a quasi-experimental approach that statistically allows comparing sampling units with similar covariates and, therefore, to test the effect of treatment without the effect of covariates (Ho et al., 2011). This technic matches sampling units of treatment and control groups with similar covariates values to test the effect of a treatment. Matching technics have been used previously to assess the effect of PAs on forest loss (Yang et al., 2021), but its use regarding forest regrowth and forest fragmentation in PAs is less frequent (but see Sims, 2014).

We used two different sampling strategies for the matching analysis: the first with one million points uniformly seeded in the entire studied area (both inside and outside reserves) at 1km of separation, and the second with 1500 circular microlandscapes with 3km of radius randomly seeded in the same area (see Appendix 3A). Each sampling point has information regarding whether they are located inside reserves or in unprotected areas, and about the changes in forest cover that occurred during 2000-2020 in its specific location (i.e., forest loss, forest gain, or no change). We used the microlandscapes to document forest fragmentation inside reserves and in unprotected areas using formula 2. Furthermore, each sampling unit had the value of the three covariates that may act as confusing factors on forest cover changes: distance to cities, distance to roads, and agriculture suitability (see Appendix 3A). To properly compare forest cover changes in reserves and unprotected areas we only include those samples located within forest cover in the year 2000 for the forest loss data (n=423,900), and within no forest cover in 2000 for the forest regrowth data (n=331,623).

We estimated propensity scores for forest loss, forest regrowth, and fragmentation data using binomial error distribution with a logit function where the response variable was the treatment (protected or unprotected). After matching, all standardized mean differences for the covariates were below 0.1 which indicates a good balance between control and treatment samples (see Appendix 3A). The matching analysis was performed using the *MatchIt* package (Ho et al., 2011) of R (R Core Team, 2021).

3.2.4 Underlying drivers of forest cover change

As underlying drivers of these forest changes, we considered two economic indicators: (i) the distance to major cities (localities with a population larger than 15,000 people) as a proxy of access to markets, and (ii) the non-farm occupation (i.e. the mean proportion of people, of the municipalities surrounding the biosphere reserves, working in the professional, services, and industrial sectors; Appendix 3A) in the year 2000. Population growth rate (during 2000-2020), population density in the year 2000, and rural settlement density in the year 2000 were also used as indicators of demographic factors. Using a correlation and a collinearity analysis, we proved that these indicators were not correlated (see Appendix 3A). During the first stage of our study, we considered including other indicators, such as unemployment rate, marginalization, human development, and agricultural subsidies. However, they were not included in our analysis because of the high correlation with the selected indicators ($r > 0.5$, $p < 0.05$, Variance Inflation Factor (VIF) > 2). Most selected indicators of underlying drivers are common in other studies (Aide et al., 2013; Figueroa et al., 2009). The theoretical relationship with forest loss and details of the calculation of these indicators can be found in Appendix 3A and Appendix 3B, respectively.

3.2.5 Data analysis

We tested for the effect of reserves (i.e., protection) on forest loss, forest regrowth, and forest fragmentation. For the first two variables, we used generalized mixed models with binomial error distribution for each one. We included forest change as binary response variables (forest loss/no change and forest regrowth/no change) and the protection condition (inside reserves or unprotected) as a fixed effect factor. To test for the effect of the protection condition on forest fragmentation rate and the number of patches in the year 2020, we used a generalized linear mixed model with a Gaussian error and a Poisson error, respectively. We included the biome type as a random effect. We performed the models with pre-matching and matching data. Matched data models were weighted using the weights automatically calculated by the matching algorithm that accounts for the potential differences in the number of samples in each treatment.

To determine the effect of the indicators of socioeconomic drivers on the forest loss, fragmentation, and forest regrowth rates, we used a multimodel inference approach with linear models (Burnham & Anderson, 2002). Because of the small sample size (n = 17), we limited to 3 the maximum number of terms in the models to avoid overfitting. Therefore, for each response variable, we constructed all models that represent all possible model combinations with 1, 2, and 3 predictors and their pair interaction (i.e., approximately 600 models). To calculate standardized parameter estimates (i.e., slopes), we applied a z-score transformation for all predictor variables. We calculated the Akaike Information Criterion corrected for small samples (AICc), and the model with the lowest AICc value was considered the best model. The percentage of deviance explained by the best model was calculated with square-R (goodness-of-fit). The effect size of each variable was estimated with the model-averaged standardized parameter estimates.

3.3 Results

The study biosphere reserves had significantly less probability of forest loss than unprotected places, even when controlling for any bias caused by differences in environmental conditions. Our models indicated that reserves reduced forest loss by about 54% compared to unprotected places (Fig. 3.2a). Regarding fragmentation, we found that reserves have a significantly smaller number of patches in 2020 than unprotected places (Fig. 3.2b). Nonetheless, reserves did not reduce the fragmentation rate (Fig. 3.2c). Reserves had a higher forest regrowth probability than unprotected places without considering the differences in environmental conditions. However, reserves had a similar forest regrowth probability than unprotected places when we controlled for covariates (Fig. 3.2d).

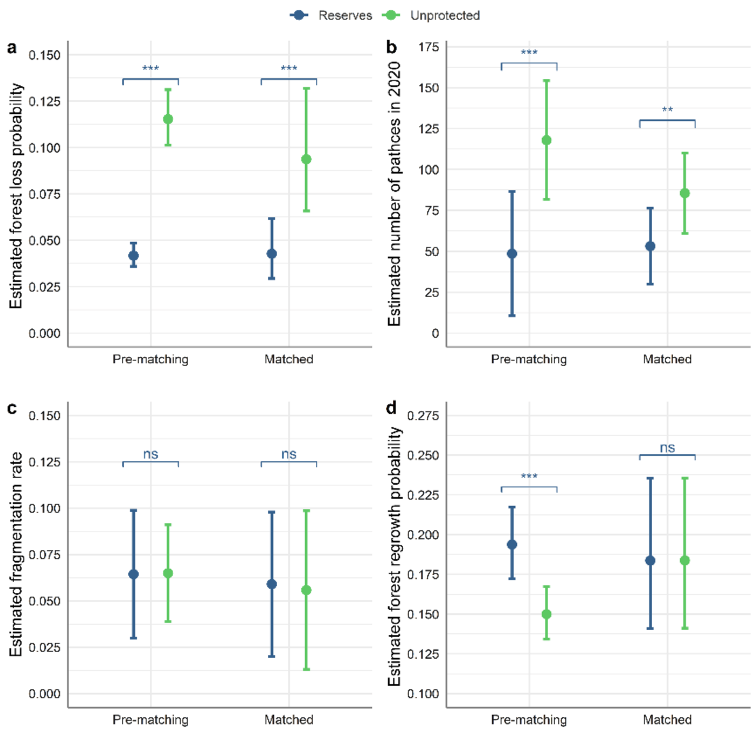

Figure 3.2. Mean (±SE) estimated forest loss probability (the probability that a pixel transit from forest cover to no forest cover in the period 2000-2020, a), the estimated number of patches in the year 2020 (b), estimated fragmentation rate (annual relative change in the number of patches) in the period 2000-2020 (c), and estimated forest regrowth probability (the probability that a pixel transit from no forest cover to forest cover in the period 2000-2020, d) in the reserves and unprotected zones before and after the matching analysis. ns: non-significative, *p < 0.05, **p < 0.01, ***p < 0.001.

The forest loss rate inside the reserves was strongly and negatively related to non-farm occupation and positively to the density of rural settlements (Fig. 3.3a). In addition, the fragmentation rate in the reserves was strongly and positively associated with the population growth rate (Fig. 3.3b). Finally, the forest regrowth rate was mostly and positively influenced by the density of rural settlements and negatively by the population growth rate (Fig. 3.3c).

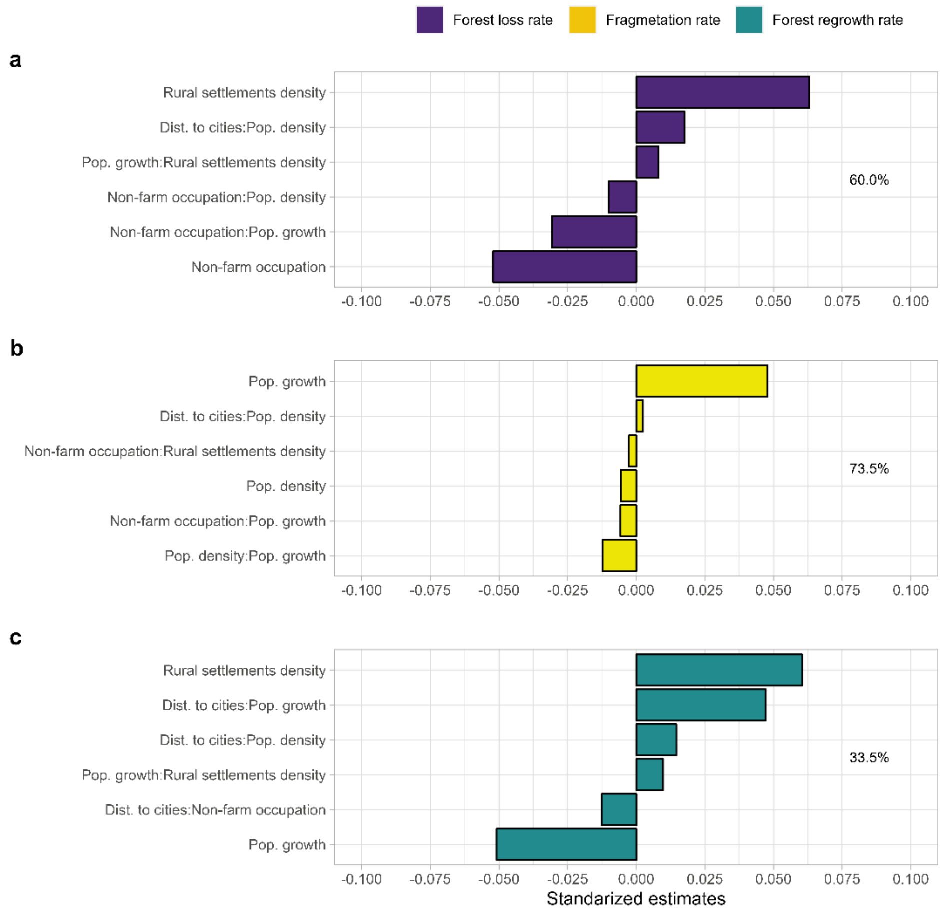

Figure 3.3. Response of forest spatial changes assessed inside the biosphere reserves to different underlying drivers (showed in different fill colors). For practical reasons we only show the six best models according to the AICc. The vertical axis shows the underlying driver indicators, and the horizontal axis shows the model-averaged standardized parameter estimates, a proxy of effect size. Bars oriented to the right represent positive responses, while bars oriented to the left are negative responses. In all the cases the explanatory variable has a higher model-averaged standardized parameter estimates value than mean unconditional variance, and therefore statistical confidence. The percentages in each panel indicate the goodness-of-fit of the best model.

3.4 Discussion

This study assesses the patterns and underlying drivers of land-use change in 19 reserves within the Mesoamerican biodiversity hotspot. Our findings confirm the key role played by protected areas in preventing forest loss within reserves. However, we go further than previous studies (Joppa & Pfaff, 2011; Spracklen et al., 2015; Yang et al., 2021) by showing that reserves had less fragmented landscapes, although the rate of fragmentation was not different from unprotected places. Overall, our results indicate that a high non-farm occupation, and a low demographic pressure, increase the effectiveness of reserves in preventing forest loss. As argued below, these findings have important applied implications for promoting the effectiveness of protected areas in the Mesoamerican biodiversity hotspot.

3.4.1 Do reserves prevent forest loss and fragmentation and promote forest regrowth?

Our findings confirm the importance of reserves as key tools to prevent forest loss and to preserve landscapes with a low fragmentation degree. Nevertheless, we did not find evidence supporting the idea that reserves also reduce the forest fragmentation rate. In other words, biodiversity within reserves copes with a lower fragmentation degree than in unprotected zones, but fragmentation increases at similar rates within reserves and unprotected zones. Since forest loss has higher deleterious effects on biodiversity than fragmentation (Fahrig, 2017), our findings represent less concern news for biodiversity inside reserves, without neglecting the potential negative effects of fragmentation (Ewers & Didham, 2006).

Surprisingly, we did not find support to the hypothesis that reserves promote forest regrowth. This contrasts with other works that suggest a positive effect of protected areas on forest recovery (Borda-Niño et al., 2020). This apparent contradiction may be clarified by our matching analysis. In our study, a simple comparison of protected and unprotected areas, without controlling for environmental covariables (pre-matching condition), supports the expected higher forest regrowth within reserves. However, controlling for environmental covariables obliterated such differences. Therefore, the forest regrowth inside and outside reserves does not seem to depend on the establishment of the reserves per se, but on the distance to cities and agriculture suitability (Appendix 3A). Thus, forest regrowth occurs in areas where agricultural activities are not favored because of the biophysical environment (i.e., steeper slopes, high elevations, and poor soils) or because of economic causes (a large distance to markets promote land abandoning and forest regrowth).

3.4.2 Underlying drivers of forest cover changes

Our findings indicate that the forest loss rate inside the reserves is mainly related to two main underlying drivers, positively to the density of rural settlements and negatively non-farm occupation. These results suggest that the lack of job opportunities in rural areas, beyond those offered by the agricultural sector, accentuates deforestation as the demand for agricultural land increases in the absence of other livelihood options. As reported by previous studies, reduction of forest loss rates is associated with the increase of non-farm labor activities (Hoang et al., 2014; Wunder, 2003), since these allow local people to obtain revenues without the need of causing more deforestation for arable land (Curtis et al., 2018; Kovacic & Viteri Salazar, 2017; Vedeld et al., 2007). In parallel with these studies, we found that reserves embedded in municipalities with greater labor opportunities in non-farm sectors, especially in the industrial activities (e.g., manufactures, electricity, or construction, see Appendix 3A), endure lower forest loss rates (Goers & Lawson, 2009). Also, a higher human development index was associated with a greater non-farm occupation (Fig. 3.4). This finding concurs with studies pointing out that non-farm activities allow local people to obtain higher profits and deal with poverty (Haggblade et al., 2007). According to our study, the emerging non-farm occupation may promote a "win-win" scenario with a positive balance between conservation and development, especially considering that an incipient forest transition process in southern Mexico has been identified (Vaca et al., 2012). However, it is important to note that this "win-win" scenario is likely limited to regions dominated by subsistence agriculture, such as the study reserves in Mexico (Bonilla-Moheno & Aide, 2020; Curtis et al., 2018). Additional studies are required in places where large-scale agriculture oriented to the production of commodities occurs, as the diversification of labor activities could have a relatively weak effect on deforestation rates in contexts where agriculture is typically supported by large capitals.

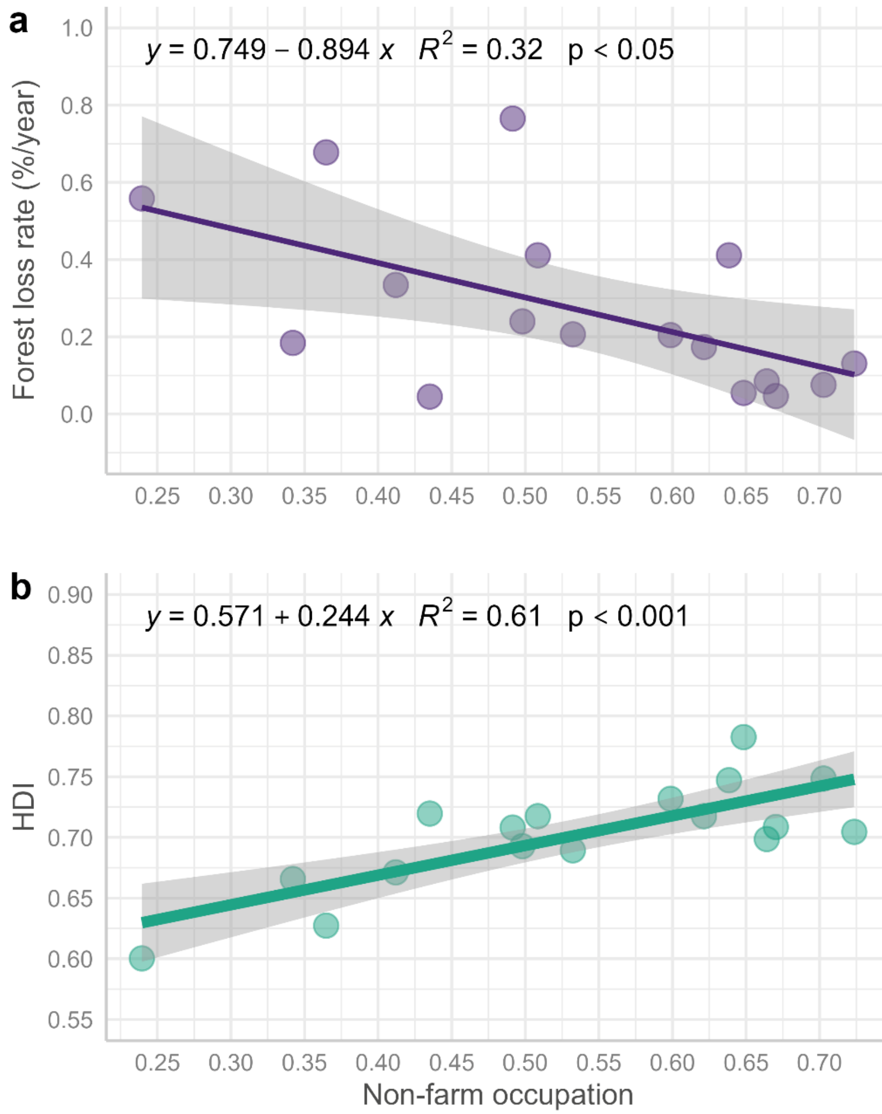

Figure 3.4. Relationship between the non-farm occupation, forest loss rate (a), and human development index (HDI, b). Each point represents a single reserve, lines represent the generalized linear model, and the grey shadow is the 95% confidence interval.

As expected, the population growth rate was a strong predictor of forest cover changes, being positively related to forest fragmentation rate, and negatively associated with forest regrowth. Human population size is considered a major driver of forest loss (Aide et al., 2013) and fragmentation (Li et al., 2010). Human pressures such as logging and agriculture are expected to increase with increasing the population size (Ehrlich & Holdren, 1971), which in turn can promote the loss and fragmentation of the remaining forest, and limit the recovery of the forest in degraded lands. Our findings contrast with those of Borda-Niño et al., (2020), which suggest that the forest regrowth rate can increase with increasing (not decreasing) demographic factors such as the density of rural settlements. This counterintuitive finding can be related to the environmental policies after the establishment of the reserves. In Mexico, for example, the creation of reserves often requires the relocation of rural communities that were within the limits established for the reserve, which leads to farmers abandoning their cropland allowing the forest to regrow (Appendix 3A). We infer that a higher forest regrowth occurred especially in reserves that had a higher density of rural settlements in their surroundings. Indeed, previously to the year 2000, when some reserves have not been legally established, the amount of forest loss rate inside reserves was significantly and positively related to the density of rural settlements ($R^2 = 0.4$, $P < 0.01$, Appendix 3A). Although difficult to support with empiric data, another possibility is that there exist active restoration efforts, as occur in some regions (Juan-Baeza et al., 2015), potentially associated positively with demographic factors (the larger the population, the greater the possibility of restoration).

Finally, we have some caveats about our work to enunciate. Our work assessed the influence of socioeconomic variables measured at the municipality on forest spatial changes, however, there exists the possibility that a finer scale (e.g., locality) outcome more reliable results, nonetheless, this is still a challenge since some variables are not available at fine resolution. In addition, there exist other underlying and proximate drivers (e.g., environmental factors) non-assessed in our work that influence forest spatial changes in protected areas that may need specific hypothesis tests in future studies. On the other hand, since here we assessed forest spatial changes in 20 years, our work may not identify properly early stages of forest regrowth (mostly given by small trees and shrubs) and therefore, underestimates forest regrowth and forest fragmentation.

3.5 Concluding remarks

Our results indicate that promoting some underlying drivers, specifically increasing labor opportunities in non-farm sectors (i.e., reducing local people's dependence on agricultural activities) and reducing demographic pressures, could strengthen the conservation function of biosphere reserves by reducing forest loss inside reserves. However, reducing the number of people in an area is still a major challenge in terms of public policies. Therefore, allocating resources to provide more economic opportunities seems to be a better option. In our view, in the absence of further job opportunities, farmers are forced to continuously expand the agricultural frontier. Finally, it is important to note that increasing non-farm jobs opportunities should be considered in the management plans of reserves, as a way to mitigate negative anthropogenic effects on them.

Conflict of interest

The authors have no conflicts of interest to declare.

Data and code availability

Data of spatial changes inside the reserves, methodological workflow, and matching algorithm code are available online https://github.com/aulizdaniel/Biosphere-resereves-data.

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Appendix A. Supplementary material

3.7 References

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Capítulo 4: Promotores de cambios en biodiversidad en reservas de la biosfera mexicanas

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Abstract

Protected areas (PAs) are paramount to preserving biomes, species, and critical ecosystem contributions to people. Yet, the conservation success of PAs is increasingly threatened by human activities such as habitat loss and disturbance. Therefore, understanding the underlying and proximate drivers of such threats is urgently needed for improving the effectiveness of PAs, especially in the tropics. We addressed this topic by gathering data on biodiversity trends during the last three decades provided by experts in 14 Mexican biosphere reserves across the Mesoamerican biodiversity hotspot. We related such trends to major socioeconomic drivers (demographic, economic, and political factors), spatial indicators of human activities (agriculture expansion and road extension), and forest spatial changes to assess the underlying and proximate causes of biodiversity changes within reserves using multivariate analyses and structural equation models. As expected, the significant proliferation of disturbance-tolerant guilds and the loss of disturbance-sensitive guilds within reserves, caused a 'winner and loser' species replacement over time. This process was directly caused by forest loss and interpatch isolation distance, which were promoted by the expansion of agriculture and roads in reserves with high population density and low non-farm occupation as the main underlying forces. Therefore, to prevent forest loss, and their negative effects on biodiversity, we should increase non-farm occupation and plan population density around biosphere reserves.

Keywords: anthropogenic disturbances, conservation success, deforestation, protected areas, species loss, biological guilds

4.1 Introduction

Despite their conservation value, PAs face a growing threat due to human activities (Jones et al., 2018), which result in forest cover changes inside and outside them (Wade et al., 2020) impacting their conservation. Thus, forest cover changes may affect PAs conservation in several ways. On the one hand, forest loss is identified as the main cause of biodiversity loss (Newbold et al., 2015). On the other hand, despite that forest fragmentation may have negative effects (Ewers & Didham, 2006), most of them are neutral or positive (Fahrig, 2017), nonetheless this is currently under debate (Fletcher et al., 2018). Furthermore, forest regrowth in the tropical forest promotes the recovery of biodiversity (Rozendaal et al., 2019). Thus, evaluating the pattern of forest cover changes in PAs is critical to improving management and conservation strategies.

Understanding the causes of forest cover changes in PAs is also of paramount importance. Overall, two types of drivers can be distinguished: proximate and underlying. The proximate drivers correspond to human activities that directly modify the forest cover, such as infrastructure extension, agricultural expansion (both subsistence and large scale), and wood extraction (Curtis et al., 2018). Environmental factors, such as hurricanes, fires, landslides, severe droughts can also act as proximate drivers of forest cover changes (Geist & Lambin, 2001). The underlying drivers are related to factors that indirectly modify the forest cover by modulating proximate drivers, such as demographic, economic, technological, political-institutional, and cultural factors (Geist & Lambin, 2001). For example, demographic-related factors, such as human population growth and population density, are recognized as major drivers of deforestation (Aide et al., 2013) as they are linked to forest resource consumption. Economic factors also play a critical role in forest cover dynamics. Access to markets is a key factor influencing trade prices for agricultural products, which in turn affects deforestation (Angelsen, 2010).

Protected areas in the tropics are usually inhabited by rural communities that subsist from agriculture activities. These communities derive low income from agriculture due to the high costs of product transportation (Angelsen, 2010), which may increase deforestation to expand the cultivated area seeking higher income (Ferraro et al., 2011). Also, in tropical regions, a shift from economies based on agriculture to non-agricultural economies has been indicated as one of the drivers producing a decrease in forest loss rates and an increase in forest regrowth, a process known as forest transition (Meyfroidt & Lambin, 2011). According to the forest transition theory, non-agricultural economies reduce the pressure on forests because local people obtain their income from activities representing less change in land use (Rudel et al., 2005). For example, rural communities having labor opportunities on tourism (Hoang et al., 2014) and industrial activities (Wunder, 2003) carry out less deforestation than communities depending only on agriculture (Kovacic & Viteri Salazar, 2017). However, the effect of deforestation reduction, associated with the emergence of non-farm occupation in tropical regions, on PAs conservation function is still an open issue, and empirical evidence is very scarce.

Here, we first apply contrafactual matching methods to test the hypothesis that PAs not only prevent forest loss but also fragmentation and promote forest regrowth during the period 2000-2020. Second, we determined the effect of some underlying socioeconomic drivers on forest cover changes within some reserves located in Mesoamerica. We hypothesize that economies based on non-farm occupation, combined with low demographic pressures, reduce forest loss and fragmentation, and promote forest regrowth inside reserves.

4.2 Methods

4.2.1 Study system

We selected fourteen biosphere reserves located within the highly biodiverse Mesoamerican region (Fig. 4.2, and Appendix 4A). The reserves are composed of different vegetation types, but principally tropical rainforest, tropical dry-forest, temperate forest, cloud forest, and mangrove. Mean annual temperature ranges from 10 to 27 °C and mean annual precipitation is 1289 mm. The studied reserves represent a gradient of human disturbance, with the remaining forest cover currently ranging from 44% to 98%.

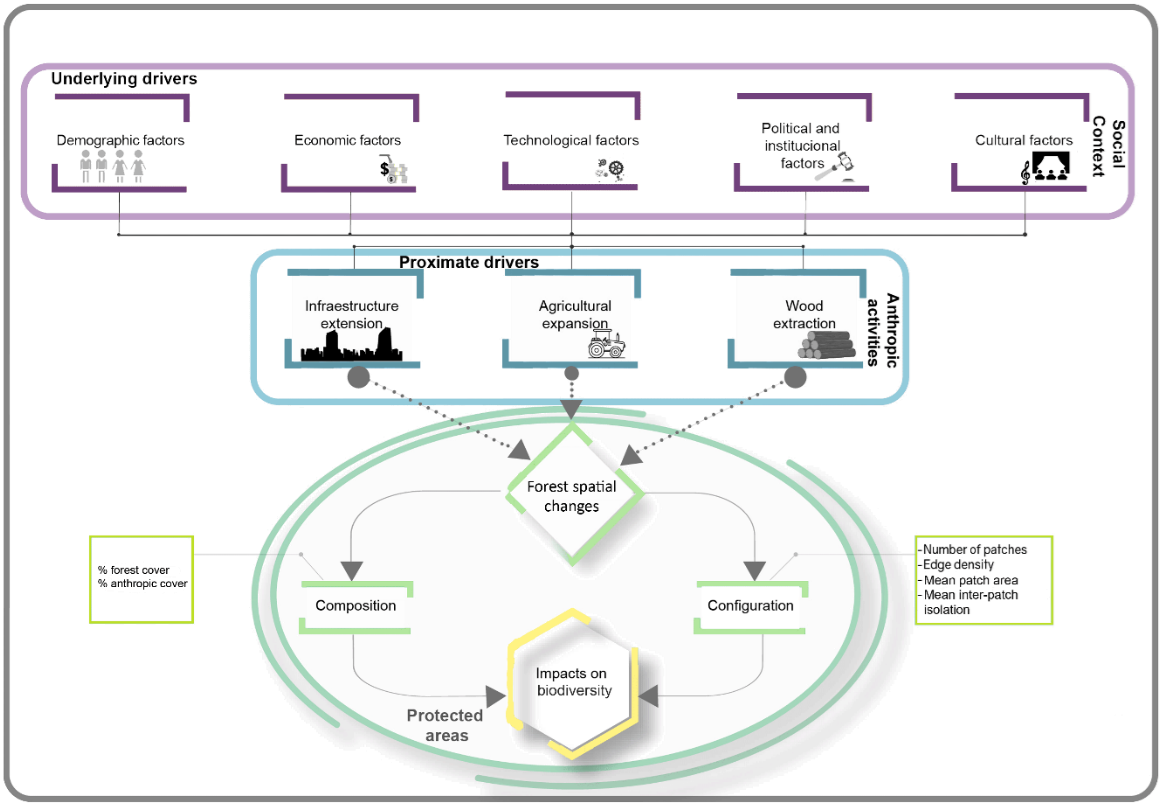

Figure 4.1: Hypothetical relationships between underlying drivers, proximate drivers, forest spatial changes, and their impact on biodiversity in protected areas. The underlying drivers affect different proximate forces, such as wood extraction, infrastructure extension, and agricultural expansion, which can directly determine forest spatial changes (Geist & Lambin, 2002). Such spatial changes can affect both the composition (e.g., forest cover) and configuration (e.g., number of patches) of the landscape surrounding each protected area, ultimately shaping biodiversity trends over time.

4.2.2 Assessing biodiversity changes

4.2.2.1 Biodiversity data

We compiled information on biodiversity changes in the studied reserves during the last three decades replicating the methods used by Laurance et al. (2012); yet, a brief overview is given here. We applied electronic surveys using the Google forms tool to gather information on biodiversity trends (Appendix 4B) separately assessing the abundance of individuals and species richness of 31 biological groups (hereafter guilds, i.e., a group of organisms with taxonomic or functional convergences). Sixty-four experts (researchers, technicians, and government officials), from twenty institutions and four countries, who have lengthy experience (21 years on average) working in the studied reserves, responded to these surveys (Appendix 4C). Each expert provided information about the level and direction (positive/negative) of change in richness and abundance of each guild: no change (less than 5% of change, either increase or decrease), small change (from 5% to 25% of change), high change (from 26% to 50% of change) and strong change (>50% of change). We assigned numeric values to these categories in a scale that ranges from 0 (no change) to 3 (strong changes). The values were positive when the abundance and richness of the guilds increased and negative when it decreased. Each expert also provided a level of certainty for each answer. If the expert had direct expertise documenting biodiversity changes, the level was identified as "high", but if the expert knew about these changes through literature, the level was identified as "good". Instead, if the expert had no evidence of these changes, but suspected them, the level was defined as "speculation". To increase data certainty, we used the same criteria as Laurance et al. (2012) and only included the information of 10 biosphere reserves for which there were surveys from at least three different experts and excluded data where researchers declared speculation about a particular change (see Appendix 4B). Following Laurance et al. (2012), we classified the more representative guilds (i.e., guilds with a greater number of surveys, n >18) into two groups depending on their characteristics and expected response to forest loss: disturbance-sensitive (i.e., large-sized animals and/or of high trophic level, and shade-tolerant plants), and disturbance-tolerant (i.e., small-sized and habitat generalist animals, and light-demanding plants) (Table 4.1).

4.2.2.2 Determining significant biodiversity trends

To identify significant temporal changes in the richness and/or abundance of general biodiversity (i.e., all guilds regardless of their response to disturbance) we used a bootstrap resampling with 10,000 iterations. Bootstrap allows estimating a mean value and 95% confidence interval that theoretically approximate the parameters of the statistical population (Efron & Tibshirani, 1985). We tested the null hypothesis that no significant changes in the abundance and richness occurred along the study period (i.e., that 95% CIs overlap zero) using a bootstrapping p-values estimator. We used the same procedure to test whether the abundance and richness of disturbance-sensitive guilds decreased, and disturbance-tolerant guilds increased during the study period. Finally, we tested for significant changes in the abundance and richness of each guild. These analyses were performed with the *boot* R package (Canty & Ripley, 2021).

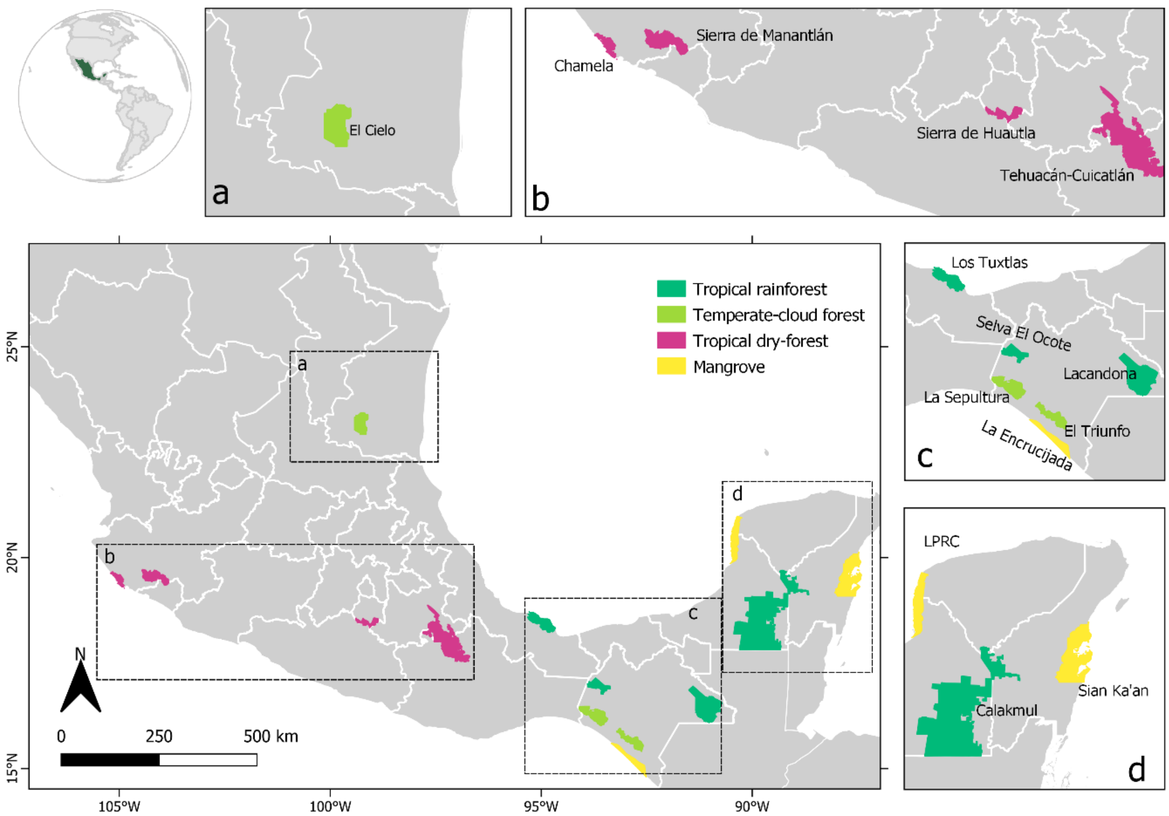

Figure 4.2: Study system: sixteen biosphere reserves located in the Mesoamerican region. Colors show the predominant biome of each reserve while the letters show details of the location of the reserves. LPRC and Lacandona are complex of reserves resulted from the combination of reserves Los Petenes – Ría Celestún and Montes Azules – Lacan-tun respectively.

Table 4.1: Biodiversity guilds evaluated in the present study. We show the total number of surveys (n) that documented changes in abundance and richness in the studied reserves

Taxa	Guild	n	Disturbance response*
Mammals	Bats	18	
	Large, non-predatory mammals	41	Sensitive
	Omnivorous/opportunistic mammals	39	Tolerant
	Primates	23	
	Rodents	18	Tolerant
	Top predators	38	Sensitive
Birds	Large frugivorous birds	19	
	Large game birds	25	Sensitive
	Raptors	20	
	Small nectarivores birds	11	
	Understory birds	18	
Amphibians and reptiles	Lizards and large reptiles	19	
	Non-venomous snakes	16	
	Stream-dwelling amphibians	15	
	Terrestrial amphibians	17	
	Venomous snakes	18	
Insects	Army ants	9	
	Disease-vectoring invertebrates	11	
	Dung beetles	8	
	Leaf-cutter ants	8	
	Light-loving butterflies	11	
Plants	Epiphytic plants	22	Sensitive
	Large-seeded species (shade-tolerant trees, climax species)	36	Sensitive
	Lianas/Climbing vines	20	Tolerant
	Pioneer species	32	Tolerant
General groups	Ecological specialists	10	
	Exotic animals (non-native)	17	
	Exotic plants (non-native)	22	Tolerant
	Human diseases	7	
	Migratory species	11	
	Species dependent on tree cavities	16	

* The most frequent guilds were classified according to their hypothesized response to disturbance (i.e., disturbance-sensitive and disturbance-tolerant guilds).

4.2.3 Testing the causal model of biodiversity changes

4.2.3.1 Underlying drivers

Previous studies have documented the importance of human population growth and density (Aide et al., 2013; Wittemyer et al., 2008), access to markets (Angelsen, 2010), non-farm occupation (Auliz-Ortiz et al., 2022), and governmental subsidies (Klepeis & Vance, 2009) as major drivers of forest loss. Therefore, we selected three types of underlying drivers of change: demographic, political, and economic. We gathered information on these indicators as close as possible for the study period (1990-2020) depending on the availability of data of enough quality from the National Institute of Statistics and Geography (INEGI in Spanish). As demographic factors, we selected population growth rate (1990-2020), population density (in 1990), and density of rural settlements (in the year 2000 because locality data in 1990 had gaps of information). As political factors, we used governmental subsidies for agriculture (period 2013-2018). Finally, as economic factors, we included distance to cities (i.e., human settlements with ≥15,000 people in 1990), unemployment rate in 1990, and non-farm occupation (i.e., the proportion of the population in a municipality who worked in industrial, professional or services activities in 1990). All variables were calculated at the municipality level by averaging the value of each indicator for all municipalities with at least 10% of its territory located inside the reserves (Appendix 4D).

4.2.3.2 Proximate drivers

We considered both road and agriculture expansion – two factors that have demonstrated to be important proximate drivers of forest loss in previous studies (Curtis et al., 2018; Laurance et al., 2009). In particular, we assessed the road density change in the period 2008-2019. To this end, we gathered information on the road network from the national network of roads for the years 2008 and 2019, and calculated kernel density for each year using a search radius of 5 km and a cell size of 100 m. We calculated the mean road density for each year inside each reserve, and finally subtracted the road density of 2008 from that of 2019 to obtain a single value per reserve. As an indicator of agriculture expansion, we used the agriculture cover change (r) in the period 1990-2019, calculated with the formula:

$$r = \left(1 - \left(1 - \frac{A_{2019} - A_{1990}}{A_0}\right)^{1/t_1 - t_0}\right) * 100 \quad (1)$$

where A_{2019} and A_{1990} are the area covered by agriculture and pasture inside a reserve in 2019(t_1) and 1990(t_0) respectively, and A_0 is a normalizing factor corresponding to the area of each reserve. The agricultural area for each year was obtained from the supervised classification of Landsat images that is described below.

4.2.3.3 Forest spatial changes

To characterize forest and anthropic cover inside the studied reserves, we gathered Landsat images around the years 1990 and 2019. We performed a supervised classification of these images to identify the next land cover classes for each year: tropical rainforest, tropical dry forest, cloud forest, temperate forest, mangrove, shrublands, agriculture lands, pasture, urban zones (cities and roads), and water bodies (Appendix 4D). We merged these classes to create three major land-cover categories: forest (all-natural vegetation composed of trees and shrubs), anthropic areas (agriculture lands, pasturelands, and urban zones), and water bodies. We estimated the land-cover classification accuracy using a confusion matrix and land cover truth points derived from ancillary data (Appendix 4D). We estimated that overall accuracy was 92.2% (±0.04 SD) for 1990 and 91.63% (±0.04) for 2019, which suggests that our forest spatial changes estimations are reasonably reliable.

Within each reserve, we calculated five metrics of forest spatial pattern. We first estimated the forest loss area as the area covered by forest in 1990 that was lost in 2019 (i.e., forest loss area = A_{2019}-A_{1990}). Then, we calculated the forest loss rate using equation 1, but in this case, the normalized factor was A_{1990}. To assess the landscape attributes that have stronger effects on the abundance and richness of plants and animals, we also calculated four class-level (i.e., forest) metrics: the number of patches (i.e., degree of forest fragmentation), mean patch size, mean interpatch isolation distance, and edge density for the years 1990 and 2019. We then calculated the relative change (RC) in each metric using the equation:

$$\mathrm{RC}_i = \frac{A_{2019} - A_{1990}}{A_{1990}} \quad (2)$$

where A_{2019} and A_{1990} correspond to the value of attribute i for the years 2019 and 1990, respectively.

4.2.3.4 Biodiversity indicators

We evaluated the effect of underlying and proximate factors on general biodiversity (i.e., by merging all guilds), and separately for guilds with different sensitivity to disturbance (i.e., disturbance-sensitive, and disturbance-tolerant species). To this end, we measured the mean change in abundance and species richness reported by all experts for each guild in each reserve. To synthesize the information, we performed principal component analyses as detailed in Appendix 4D. The scores of the first principal component of general biodiversity data were used as an indicator of general changes in abundance and richness. Moreover, scores of the first and second principal components of disturbance response data were then used as indicators of changes in abundance and richness of disturbance-sensitive and disturbance-tolerant guilds, respectively. Scores of each component were rescaled to range from 0 to 1, with positive values indicating positive changes in diversity (i.e., both abundance and richness).

4.2.3.5 Statistical analyses

We used a multimodel inference approach (Burnham & Anderson, 2002) to select the most relevant variables and simplify the relationships (paths) involved in the framework presented in Fig. 4.1. With this approach, we assessed the relative importance of all predictors involved in each path to select the most plausible and parsimonious relationships in the structural equation model (SEM) that is described below. In particular, we performed linear models for each pathway in Fig. 4.1 using the *glmulti* package for R (Calcagno & de Mazancourt, 2010) and selected the model with the lowest Akaike Information Criterion corrected for small samples (AICc). Regarding the relationship between underlying and proximate drivers, we performed a linear model with each proximate driver as a response variable, and all underlying indicators as predictor variables to select the most important predictors of each proximate driver. For example, as agriculture expansion was best predicted by a model that includes only the non-farm occupation (Appendix 4E), we only included this underlying driver in the SEM model. Also, road density change was best predicted by population density (Appendix 4E), so we selected population density and diversity of non-farm occupation as underlying factors, and road density change and agricultural expansion as proximate ones in the SEM model. Biodiversity trends were better predicted by forest loss rate and interpatch isolation (Appendix 4E), so we included these two forest spatial changes in the SEM model. We used the *picewiseSEM* package of R (Lefcheck, 2016) to perform SEM models to assess the cascading effects of socioeconomic drivers on biodiversity changes. We built SEM models for general biodiversity and for disturbance-sensitive and disturbance-tolerant guilds separately. All relationships were expressed as equations using linear models (see Appendix 4E). We evaluated the overall fit of the models and missing paths through Shipley's test of d-separation, which includes Fisher's C statistic and AIC.

4.3 Results

4.3.1 Biodiversity changes

The most representative taxa in our dataset were terrestrial mammals followed by plants and birds, with 177, 133, and 93 surveys, respectively. For insects, we compiled 50 surveys. The most representative guilds were large non-predatory mammals, omnivorous/opportunistic mammals, and top predators, with nearly 40 surveys each, while army ants, leaf-cutter ants, and dung beetles were poorly covered, with less than 9 surveys each (Table 4.1). In general, the abundance of individuals tended to decrease over time (mean value = -0.33, 95%CI = -0.45 to -0.20, $p < 0.001$, Fig. 4.3). This trend was particularly shaped by a significant decrease in the abundance of disturbance-sensitive species (-0.88, -1.14 to -0.63, $p<0.001$), as the abundance of disturbance-tolerant ones showed the opposite trend (0.68, 0.43 to 0.93, $p<0.001$, Fig. 4.3, details in Appendix 4F). When assessing each guild separately, we found that the abundance of large non-predatory mammals, primates, top predators, large game birds, raptor birds, epiphytic plants, ecological specialists (foraging specialists and species with complex mutualisms), and large-seeded trees tended to decrease over time (Fig. 4.4, details in Appendix 4F). Interestingly, the abundance of most amphibian and reptile guilds, except venomous snakes, decreased, whereas the abundance of disease-vectoring invertebrates, lianas/climbing vines, pioneer species, exotic plants, and exotic animals increased over time (Fig. 4.4). In all these cases, species richness followed a similar pattern to abundance (see Appendix 4F).

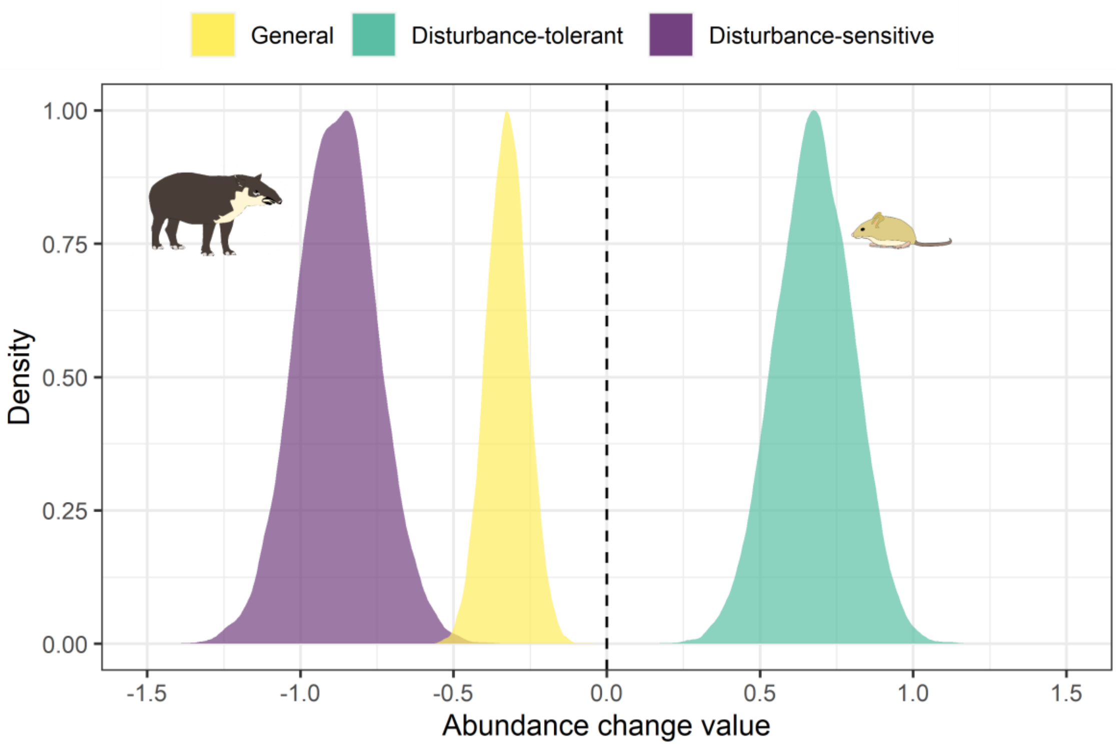

Figure 4.3: Distribution of mean changes in abundance of all biodiversity guilds, disturbance-sensitive guilds, and disturbance-tolerant ones during the last three decades in the studied biosphere reserves. The density plots resulted from bootstrap replacing with 10,000 iterations. The dashed line marks no change. In all cases, we found significant changes over time (estimated $\bar{x} \neq 0$, $p<0.05$), not only in abundance, but also in species richness (see details in Appendix 4F).

4.3.2 Drivers of biodiversity change

Our SEM models fitted well the data (i.e., 40% to 81% of explained variance) indicating that our conceptual framework explains the observed biodiversity changes in the studied biosphere reserves. The reserves with lower non-farm occupation experienced a higher agriculture expansion, which caused a higher forest loss rate, ultimately decreasing biodiversity in general (Appendix 4F), and the abundance and richness of disturbance-sensitive species, in particular (Fig. 4.6a). However, the abundance and richness of disturbance-tolerant species were not related to forest loss rate, but to changes in interpatch isolation distance over time. In particular, these species proliferated in biosphere reserves where interpatch isolation decreased, with such a decrease being associated with an increase in road density in reserves with higher population density (Fig. 4.6b). To understand the causes of such a proliferation of disturbance-tolerant species, it is important to note that isolation distance decreased principally in less forested but extremely fragmented biosphere reserves (Appendix 4C).

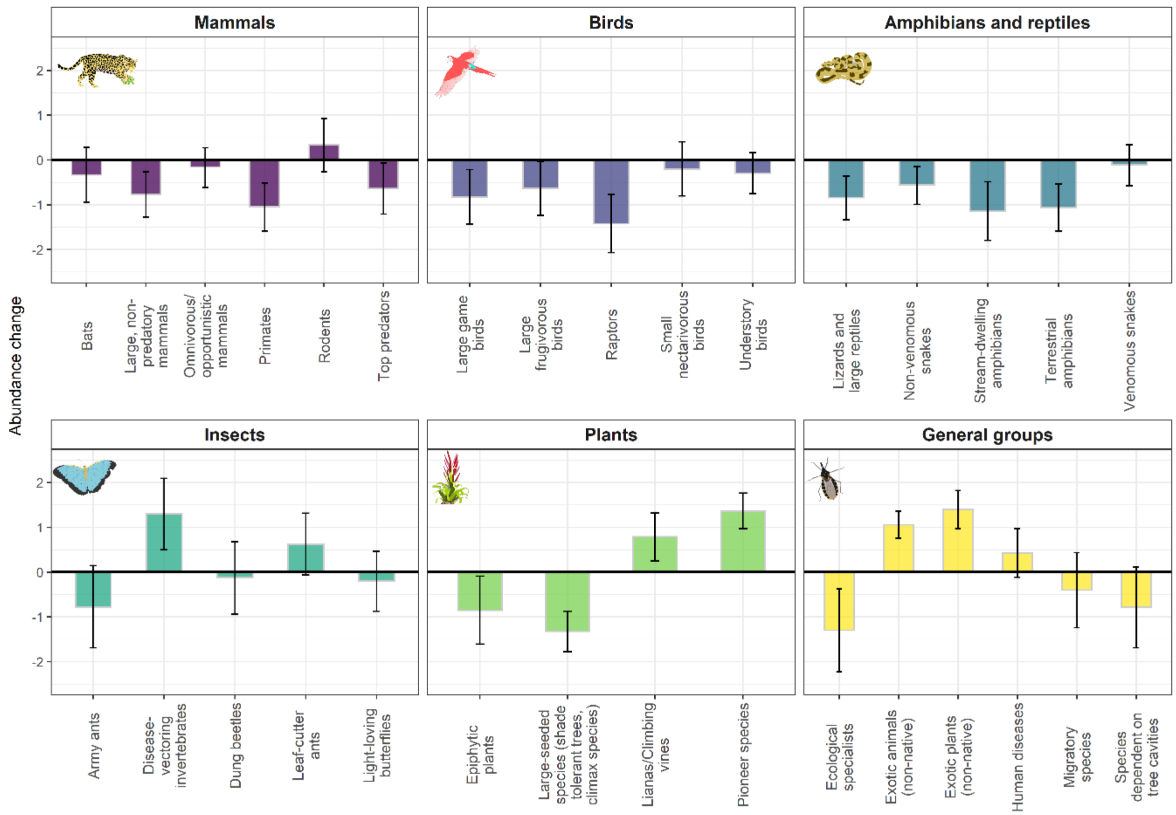

Figure 4.5 Mean changes in abundance (± 95% CI) of 31 biological guilds over the last 30 years in the studied biosphere reserves. We used a bootstrap resampling with 10,000 iterations to estimate mean values and 95% CI. We considered that a change was significant if its 95% CI did not overlap zero (for details see Appendix 4F).

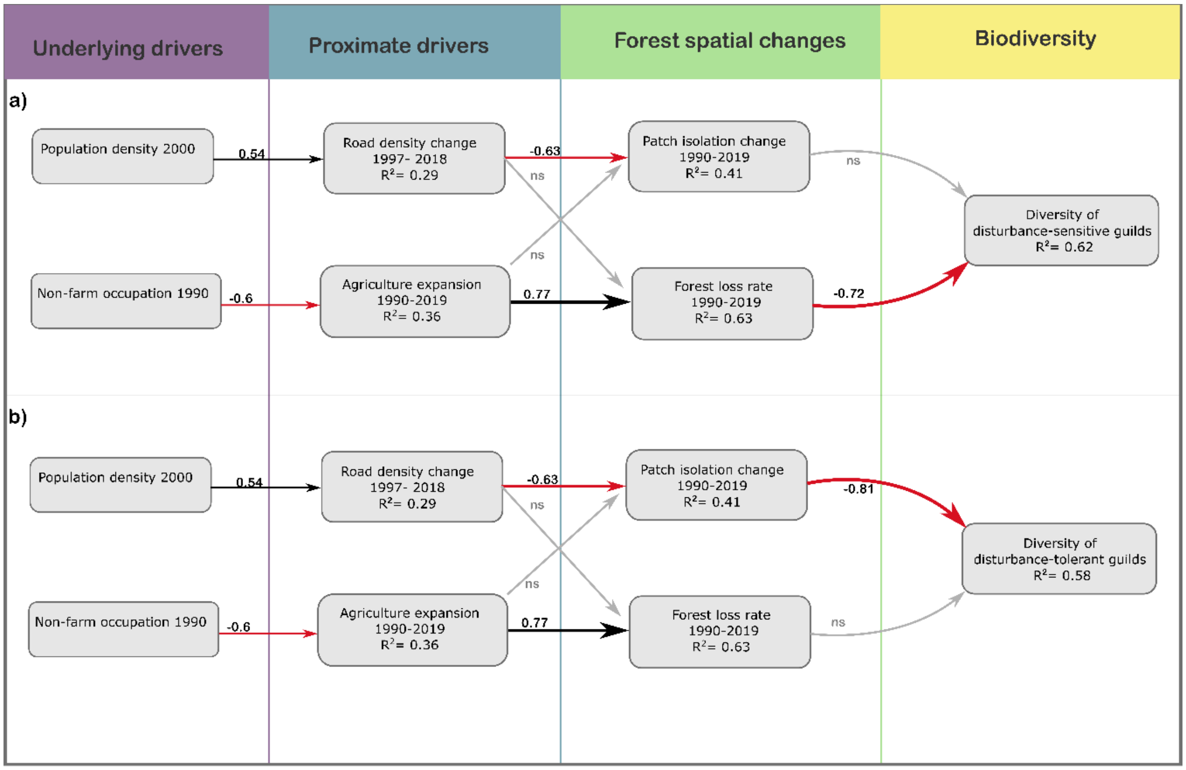

Figure 4.6: Structural equation models (SEM) of the relationships between underlying and proximate drivers of forest spatial changes and their effects on the diversity (PCA scores of mean richness and abundance) of disturbance-sensitive (a) and disturbance-tolerant guilds (b). Significant ($p < 0.05$) positive and negative paths are indicated with black and red arrows, respectively, whereas gray arrows indicate non-significant relationships ($p>0.05$). Values near the arrows correspond to standardized coefficients and indicate the direction (positive/negative) and strength of each path. Note that an increase in interpatch isolation change indicates that isolation decreased though time, so the negative effect of this variable on the diversity of disturbance-tolerant guilds (b) implies that these guilds proliferated in biosphere reserves where interpatch isolation decreased. Within the box of each response variable, we also show the R^2 value. The models fitted the data well (model a: Fisher's C=17.42, p=0.83; model b: Fisher's C=16.64, p=0.86).

4.4 Discussion

Our findings demonstrate that in the last three decades there has been a generalized biodiversity impoverishment in 14 Mesoamerican biosphere reserves. This is consistent with the global pattern of biodiversity loss (Newbold et al., 2015; Powers & Jetz, 2019), which is particularly acute in tropical forest reserves (e.g., Laurance et al., 2012). As a novel contribution, we found that, as predicted, disturbance-tolerant species are replacing disturbance-sensitive species in Mesoamerican biosphere reserves, thus contributing to the understanding of the winners-losers paradigm (sensu Dornelas et al., 2019; Filgueiras et al., 2021). Another novel and valuable contribution is that our multivariate models highlight the critical role of human demographic factors (e.g., population density) and economic factors (e.g. non-farm occupation) as major underlying drivers of biodiversity loss and species replacement within reserves. As discussed below, this is not only critical for understanding the causes of biodiversity changes within PAs over time but to improve management and conservation strategies.

The biodiversity impoverishment is principally caused by a significant decrease in the abundance and richness of disturbance-sensitive species in more deforested biosphere reserves. This was the case with large-sized mammals and birds, apex predators, large-seeded trees, and epiphytic plants. However, we also found that primates and all amphibian and reptile guilds but one (venomous snakes) are also decreasing in the studied reserves. This is not surprising as all these guilds are composed of highly-threatened forest-dependent species (e.g., Nori et al., 2015; Estrada et al., 2017; Doherty et al., 2020). In fact, forest loss is known to eliminate tree species and associated plants (epiphytes, lianas) that constitute the structural and functional basis of old-growth forests (Cudney-Valenzuela et al., 2022; Pinho et al., 2020). Logically, the removal of trees, especially the biggest ones, has negative impacts on arboreal mammals, including primates (Cudney-Valenzuela et al., 2022). Similarly, ground mammals and birds highly dependent on forest resources, such as seed predators, frugivorous, herbivorous, and insectivorous are also highly vulnerable to forest loss (Newbold et al., 2013; Dirzo et al., 2014; Watling et al., 2020; Arce-Peña et al., 2022). Large-sized vertebrates are particularly impacted because they require larger forest areas to maintain viable populations (Pe'er et al., 2014) due to their high energy requirements (Brown et al., 2004). This can explain why top predators and their relatively large-sized prey are declining in most biomes worldwide (Estes et al., 2011; Magioli & Ferraz, 2021).

Importantly, these biodiversity changes are mainly driven by agriculture expansion within biosphere reserves inhabited by populations whose economy depends on agriculture. The importance of agriculture in driving tropical forest loss has been documented in previous studies (Gibbs et al., 2010), and subsistence agriculture is particularly important in our study region (Curtis et al., 2018). What is not so well understood is the role of the local economy in driving forest spatial changes, and our findings indicate that promoting non-farm occupation can prevent forest loss in biosphere reserves. This is in agreement with some previous studies. For example, forest loss rates seem to be lower in regions where economies are oriented to industrial (Wunder, 2003) or touristic activities (Bluffstone, 1995; Hoang et al., 2014). Similarly, there is evidence that in the absence of labor opportunities, local communities can develop agriculture activities with low revenues (mostly subsistence agriculture, Curtis et al., 2018). Under such circumstances, farmers are forced to increase the cultivated area to obtain profitable revenues, which promotes forest loss (Angelsen, 2010). In our study region, local communities with higher non-farm occupation not only exert lower forest loss pressure on reserves but also have higher human welfare (Auliz-Ortiz et al., 2022). Therefore, allocating higher resources to increasing non-farm labor opportunities would potentially prevent forest loss and the extirpation of disturbance-sensitive species in forest reserves.

In contrast to disturbance-sensitive guilds, disturbance-tolerant ones are increasing their abundance and richness over time. This was the case of disease-vectoring invertebrates, lianas and climbing vines, pioneer tree species, and exotic plants and animals – guilds that not only tolerate but can take advantage of the conditions prevailing in human-modified landscapes (Arroyo-Rodríguez et al., 2009; Bradley et al., 2010; Burkett-Cadena & Vittor, 2018; Laurance et al., 2001). Interestingly, these ‘winner’ guilds proliferated principally in reserves where interpatch isolation distance decreased through time, and this configurational trend was promoted by the expansion of roads in reserves with higher population density. To understand the landscape context in which interpatch isolation decreased, it is important to note that isolation decreased principally in less forested reserves composed of a high number of small patches, i.e. the higher the number of patches the lower the mean distance among them. Therefore, the proliferation of disturbance-tolerant species is likely not only related to a decrease in isolation, which obviously can favor interpatch movements (e.g., seed dispersal), facilitating the colonization and recruitment of these species in different patches of the reserve (Arroyo-Rodríguez et al., 2013). It could also be related to the availability of suitable conditions in these reserves, such as a large number of small edge-affected patches, as these are optimal conditions for the recruitment of light-demanding pioneer plants (Arasa-

Gisbert et al., 2021; Arroyo-Rodríguez et al., 2009; Laurance et al., 2006; Santos et al., 2008), and other 'winner' species (Filgueiras et al., 2021; Magioli et al., 2021). Likewise, forest edges facilitate the invasion of exotic species into the landscapes, which may cause additional negative effects on native biodiversity (Brook et al., 2008).

Our results, therefore, indicate that there is an ongoing species replacement process in the study reserves, which can potentially impact ecosystem functioning. In particular, disturbance-tolerant species appear to be replacing disturbance-sensitive ones, and as these two guilds can play different roles in the ecosystem (reviewed by Filgueiras et al., 2021), such a species replacement could have strong ecological impacts. For example, long-lived, hard-wood, large-seeded, shade-tolerant tree species, typical of old-growth forests, have a much more important contribution to the aboveground biomass of the ecosystem than short-lived, soft-wood, light-demanding pioneer tree species (Chazdon et al., 2016). As carbon in tropical forests is mainly stored in the aboveground biomass (Schlesinger & Bernhardt, 2020), the replacement of shade-tolerant tree species by light-demanding pioneer species can significantly limit global carbon storage (Houghton, 2005). Also, shade-tolerant tree species provide important resources for feeding and shelter to a plethora of invertebrate and vertebrate species (Cudney-Valenzuela et al., 2021; Dirzo et al., 2014), which play critical ecological functions as pollinators, seed dispersers, primary and secondary consumers, and even as biological controls for small and medium-sized animal species associated with human-disturbed habitats (Magioli et al., 2021). In addition, the depletion of top predators species (disturbance-sensitive species) triggers important cascading effects (Dirzo et al., 2014; Estes et al., 2011), such as the increase of small-sized herbivore prey that in turn reduces the abundance of seedlings and juvenile trees (Terborgh et al., 2001). Therefore, the disappearance of disturbance-sensitive species can have strong negative consequences on the functionality of forest ecosystems (Barnes et al., 2014; Clavel et al., 2011), and on a variety of contributions these ecosystems provide to people (Soliveres et al., 2016).

Biosphere reserves are known to be effective tools for preventing forest loss (Auliz-Ortiz et al., 2022). However, our findings indicate that in Mesoamerica, such a conservation outcome has not prevented the impoverishment of biotic assemblages within biosphere reserves. In fact, disturbance-sensitive guilds have decreased significantly over the last three decades, whereas disturbance-tolerant guilds have increased, leading to a process of disturbance-sensitive/disturbance-tolerant species replacement within reserves. This loser/winner replacement (sensu Filgueiras et al., 2021) seems to be caused by forest loss (proximate cause), which was promoted by the expansion of agriculture and roads in reserves with high population density and low non-farm occupation (underlying drivers). Therefore, to increase the conservation success of biosphere reserves we should conceive them as integral systems embedded into a socio-economic context, so to prevent forest loss and their negative effects on biodiversity, we need to increase non-farm occupation and plan population density around biosphere reserves. In Mexico, there are programs that promote employment in non-agricultural sectors in PAs encouraged by both governmental and non-governmental institutions (e.g., the conservation program for sustainable development and the program for temporal employment of the Mexican commission on Protected Areas). Furthermore, touristic activities in PAs represent economic alternatives for the population that in other studies have shown to have positive effects on the livelihoods of local communities (Ferraro & Hanauer, 2014) and on reducing forest loss when it is associated with PAs (Brandt & Buckley, 2018). Therefore, this research suggests that supporting and reinforcing this type of program and economic activities should be necessary to mitigate anthropogenic impacts on PAs.

4.5 Supporting Information

Additional information is available online in the Supporting Information section at the end of the online article. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Capítulo 5: Discusión y conclusiones generales

5.1 Discusión general

En este trabajo se analizan los posibles compromisos (*trade-off*) entre la conservación biológica y el desarrollo dados por la interacción entre las ANPs mexicanas y las comunidades locales aledañas. A lo largo de las siguientes líneas se ofrecen los principales mensajes derivados del trabajo y se destacan las implicaciones de éstos. La figura 5.1 resume los hallazgos más importantes de la tesis por lo que servirá de guía durante la discusión.

Las ANPs mexicanas no causan ni refuerzan la pobreza en las comunidades locales adyacentes. Los resultados encontrados en el capítulo 2 de este trabajo concuerdan con otros estudios realizados tanto en Latinoamérica como a escala global (Andam et al., 2010; Naidoo et al., 2019). Esto representa un área de oportunidad, pues los resultados de esta tesis demuestran que las ANPs no deben ser vistas necesariamente como factores contrapuestos con el desarrollo de las comunidades locales. De acuerdo con la bibliografía, las ANPs contribuyen de diferente manera al bienestar de las personas (Dudley & Stolton, 2010). Los recursos naturales que yacen en las ANPs aportan todo tipo de servicios ecosistémicos a las poblaciones que permiten su sostén y desarrollo (Dudley et al., 2010; Secretariat of the CBD, 2008). Asimismo, las ANPs pueden ser fuente de empleo en sectores como el ecoturismo, lo que ha ayudado a reducir las condiciones de pobreza en algunas regiones (Ferraro & Hanauer, 2014). Además, en las ANPs se reciben programas de inversión por parte del gobierno y de organizaciones no gubernamentales para incentivar empleos, un ejemplo de esto es el programa de conservación para el desarrollo sostenible (CONANP, 2021).

De acuerdo con los resultados de este estudio, las ANPs con un manejo estricto, que no permite el uso sustentable de recursos, pueden influir en el nivel de pobreza de las poblaciones adyacentes bajo ciertas condiciones biofísicas (Fig. 5.1A-B). Los parques nacionales están asociados a un mayor nivel de pobreza que las reservas de la biósfera, sobre todo en zonas alejadas de la ciudad y en zonas con pendiente pronunciada, gran altitud y un suelo pobre. Las zonas rurales alejadas de ciudades importantes tienen menos acceso a mercados para intercambiar productos, lo cual reduce las posibilidades de obtener ganancias redituables debido a los altos costos de transporte (Ferraro et al., 2011; Partridge & Rickman, 2008). Por otro lado, bajo condiciones poco aptas para el desarrollo de actividades agrícolas (e.g. pendiente pronunciada, zonas de gran altitud, con suelos pobres) los agricultores necesitan invertir más recursos y trabajo para obtener ganancias redituables (Angelsen, 2010; Pfaff et al., 2014).

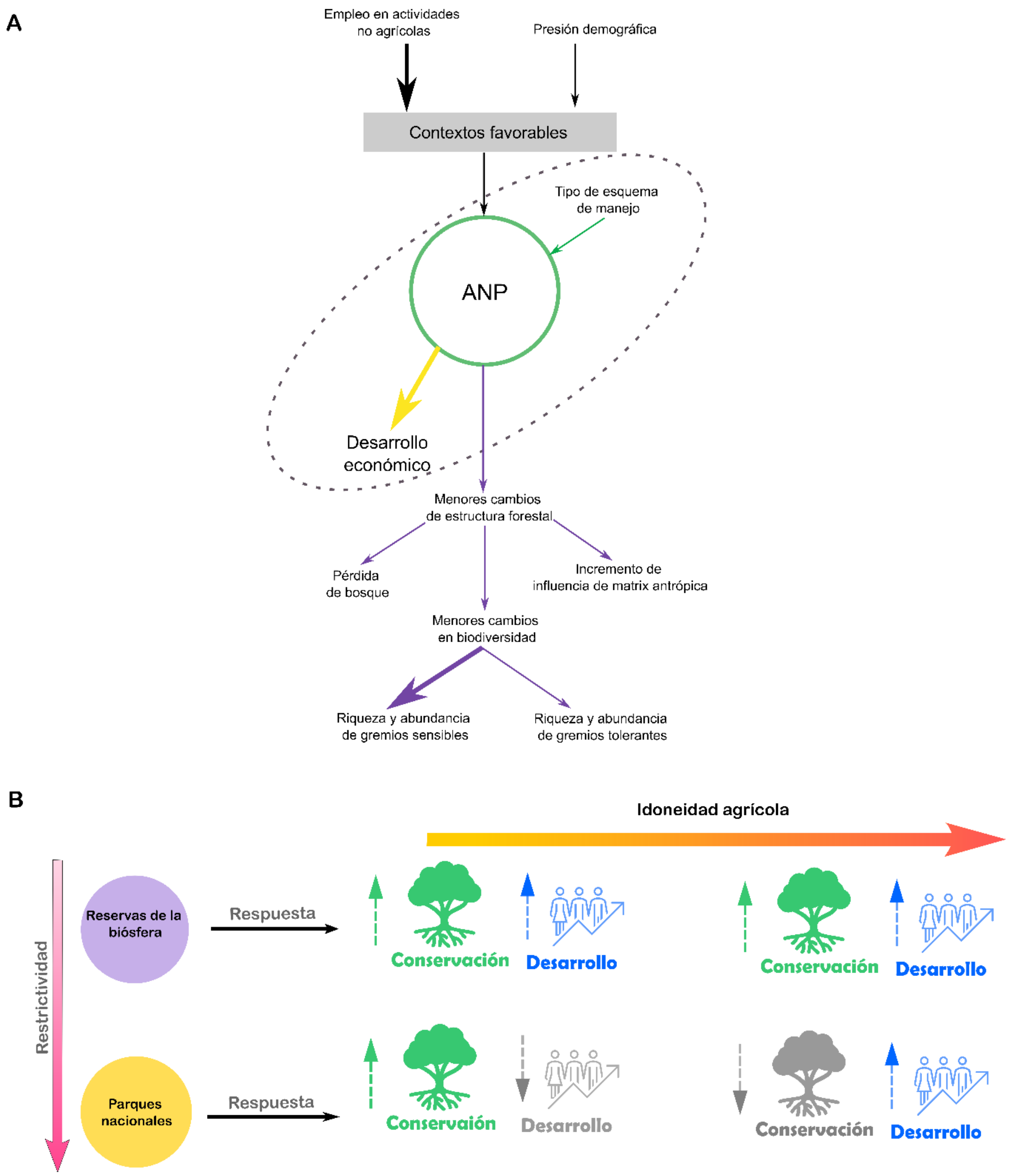

Figura 5.1: Resumen gráfico de los resultados de la investigación doctoral. En el panel A el ancho de las flechas denota la intensidad de la relación que existe entre las variables conectadas. El círculo gris punteado denota la relación entre ANPs y el desarrollo económico, que se explica a mayor profundidad en el panel B. Los contextos más favorables para la conservación de las ANPs son aquellos donde existe menor presión demográfica (i.e., densidad y crecimiento poblacional) y mayor ocupación de empleos no agrícolas. Estos contextos provocan menores cambios en la estructura forestal en términos de pérdida de

bosque e incremento de matriz antropogénica. Es así que, ANPs con menores cambios tienen mayor riqueza y abundancia de gremios sensibles a las perturbaciones y menor de gremios tolerantes (ver texto para detalles). En el panel B, las flechas hacia arriba y logos en color vivo denotan relaciones favorecidas, mientras que logos grises y flechas hacia abajo son relaciones no favorecidas.

Los resultados del capítulo 2 advierten que cuando las condiciones biofísicas dificultan el desarrollo socioeconómico de las personas, el restringir el acceso a los recursos naturales en las ANPs puede ser un obstáculo para el desarrollo de las comunidades adyacentes. Algo similar ha sido reportado en Costa Rica (Ferraro et al., 2011). Por tanto, este estudio sugiere que sería necesario plantear modos más flexibles de manejo en ANPs, especialmente en contextos biofísicos desfavorables como los aquí planteados. En este sentido el involucramiento participativo de las comunidades en labores de conservación es clave (Bray et al., 2008; Stoll-Kleemann et al., 2010), pues no solo permite una mayor integración y concientización de éstas en las ANPs sino que les ofrece opciones de desarrollo.

Las reservas de la biósfera mexicanas favorecen en mayor medida escenarios “ganar-ganar” en términos de conservación y desarrollo que los parques nacionales en los escenarios aquí explorados (Fig. 5.1B). Este es uno de los puntos más relevantes del trabajo, pues a nivel global, los esquemas que permiten el uso sustentable de recursos (por tanto menos restrictivos) son vistos como menos efectivos para reducir la pérdida de bosque (Wade et al., 2020). No obstante, a nivel nacional este patrón no se repite y son varios estudios, además de éste mismo, que confirman este punto (Blackman et al., 2015; Figueroa et al., 2011; Sims & Alix-Garcia, 2017). Esto abre un área de oportunidad, pues se reconoce que los esquemas de manejo que permiten el usos de recursos de manera sustentable otorgan mayores beneficios a las poblaciones (Naidoo et al., 2019) y que no representan un riesgo de reforzamiento de pobreza (Capítulo 2). Futuros estudios podrían enfocarse en mejorar el entendimiento de los factores involucrados en las disyuntivas entre conservación y desarrollo en ANPs, pues hacer esto permitiría localizar áreas donde las ANPs no sean solamente ecológicamente relevantes sino también socioeconómicamente aptas para su adecuado funcionamiento. Son pocos los estudios que con metodología tan robusta como la empleada en esta investigación que han evaluado simultáneamente metas de conservación y desarrollo en ANPs (pero ver Corral et al., 2016; Sims and Alix-Garcia, 2017) por lo que este trabajo abona a tener un mejor entendimiento de los posibles costo-beneficios asociados a estas metas.

Las reservas de la biósfera son herramientas efectivas para reducir la pérdida de bosque y además conservan paisajes menos fragmentados, pero no mostraron reducir la tasa de fragmentación ni promover mayor rebrote de bosque en comparación con sitios no protegidos. La mayor parte de los estudios se centra en analizar el efecto de las ANPs en la pérdida de bosque y los resultados coinciden con los hallazgos del capítulo 3 en el sentido de reafirmar el rol de las ANPs en prevenir la deforestación (Andam et al., 2008; Ferraro et al., 2013; L. Joppa & Pfaff, 2010; Yang et al., 2021). Sin embargo, son muy escasos los trabajos que analizan el efecto de las ANPs sobre la fragmentación y la regeneración del bosque. Dentro de estos trabajos destaca el de Sims (2014), quen usa una aproximación muy parecida a la de esta investigación y encuentra configuraciones con menor fragmentación en zonas protegidas que no protegidas, aunque no documenta un cambio en el número de fragmentos como sí se hace en el capítulo 3. A su vez, en una revisión sistemática, Borda-Niño et al. (2020) documentan que las ANPs están asociadas a un mayor rebrote de bosque, no obstante, los trabajos que recopilan no toman en cuenta las diferencias en el contexto biofísico de las ANPs, lo cual puede explicar las diferencias con respecto a los hallazgos del presente estudio. De este modo, esta investigación confirma la importancia de las ANPs como protectoras de la cobertura forestal, pero también abre áreas de investigación sobre las implicaciones de los resultados en temas de fragmentación y regeneración del bosque sobre la conservación de las especies.

Los principales promotores de cambio en la cobertura forestal y la biodiversidad son los factores demográficos, como la densidad poblacional, y los factores económicos, como la baja disponibilidad de empleos en sectores no agrícolas (Fig. 5.1A). Estudios previos han demostrado que los factores demográficos son de los más importantes en lamodificación de la cobertura forestal, tanto en ANPs (Figueroa et al., 2009; Wittemyer et al., 2008) como en regiones enteras (Aide et al., 2013). La concentración de personas en una región demanda mayores recursos y promueve la creación de infraestructura para satisfacer las demandas de la población (Ehrlich & Holdren, 1971). Por otro lado, la ocupación en empleos no agrícolas es un *proxy* del proceso de incipiente transición forestal que se empieza a documentar en la zona de estudio (Vaca et al., 2012). En estos procesos existe un vire en el tipo de oportunidades laborales disponibles en el que se pasa de economías más apegadas a actividades agrícolas hacia economías dependientes de sectores como el industrial y el de servicios. Este cambio económico trae consigo una reducción de las tasas de deforestación (Meyfroidt & Lambin, 2011), pues en los trópicos las actividades agrícolas son el mayor determinante de la deforestación (Curtis et al., 2018). En este sentido este trabajo aporta evidencia empírica a las ciencias de la conservación sobre la importancia de factores económicos ligados a la transición forestal y sus repercusiones sobre la biodiversidad de las ANPs. Asimismo, este trabajo subraya la necesidad de hacer un manejo integral de las ANPs tomando en cuenta el contexto en que se desenvuelven. De este modo, los resultados sugieren que proveer oportunidades laborales a las poblaciones adyacentes y planificar los asentamientos humanos permitiría reducir el impacto sobre las ANPs. No obstante, es necesario evaluar la pertinencia de estas alternativas en las comunidades y no proceder de manera impositiva, pues para lograr los mejores escenarios, es necesario el involucramiento participativo de todos los actores (Stoll-Kleemann et al., 2010). Por último, es importante mencionar que existen otros promotores no evaluados, por ejemplo, la demanda de productos comerciales a nivel internacional (Hoang & Kanemoto, 2021), que han demostrado ser importantes a escala de país y cuya acción sobre la conservación de las ANPs debe ser esclarecida en futuros trabajos.

Las reservas de la biósfera mexicanas afrontan un proceso de sustitución de especies, en donde especies de talla grande, nivel trófico alto, de hábitos especialistas, plantas de ciclo de vida largo y que son tolerantes a sombra tienen reducciones de su riqueza y abundancia. En contraste, las especies de talla pequeña, nivel trófico bajo, de hábitos generalistas y plantas de ciclo de vida corto adaptadas a claros han incrementado en su riqueza y abundancia. Este proceso ha sido documentado previamente en sitios no protegidos (Filgueiras et al., 2021) y el capítulo 4 lo documenta para ANPs. Esto indica que pese su estado de protección, las ANPs no son ajenas al proceso generalizado de pérdida de biodiversidad que encara el mundo (Newbold et al., 2015) aunque la magnitud es menor en comparación con sitios no protegidos (Gray et al., 2016). A partir de los resultados del capítulo 4 se abren oportunidades de investigación tanto de las causas de estos cambios como de sus consecuencias. Todo ello en virtud de que se sabe que la sustitución de especies trae consigo cambios en la función de ecosistemas (Barnes et al., 2014) y de los servicios ecosistémicos otorgados (Soliveres et al., 2016).

Las características demográficas y económicas de las comunidades locales alrededor de las reservas de la biósfera estudiadas explican el cambio en biodiversidad observado en ellas durante los últimos 30 años vía las modificaciones de la estructura forestal (i.e., la pérdida de bosque y la emergencia de matriz antropogénica, Fig. 5.1A). Es sabido que la principal causa de pérdida de biodiversidad es la pérdida de hábitat por acciones antropogénicas (IPBES, 2019; Newbold et al., 2015, 2016); sin embargo, son pocos los trabajos que aportan evidencia empírica sobre la serie de relaciones causales existentes entre promotores subyacentes, promotores proximales, y modificaciones de la estructura forestal que guían estos cambios. Destaca el trabajo de Laurance et al. (2012) que documenta los cambios en biodiversidad ocurridos en 60 ANPs a lo largo del mundo con ayuda de un grupo nutrido de expertos y explora sobre sus posibles causas.

El trabajo expuesto en el capítulo 4 da un paso más allá pues, si bien se parte de la propuesta de Laurance et al. (2012), en el presente estudio se recopila también información socioeconómica y espacial sobre cambios en la estructura de bosque para proponer un modelo integral que explique los cambios en biodiversidad observados en las reservas estudiadas. Así, este trabajo representa un esfuerzo sin precedente en el país donde a través de la colaboración de más de 60 expertas y expertos se documenta los cambios en biodiversidad y sus posibles causas. La información recopilada pretende ayudar a los tomadores de decisiones a identificar los puntos clave donde se puede incidir para reducir la presión sobre los biomas y especies que las ANPs protegen. Asimismo, el trabajo destaca la importancia de la colaboración de personas con distintas experiencias y de integrar información de distinto tipo para ayudar encontrar patrones generales y responder preguntas en las ciencias de la conservación. Al empezar a integrar conocimiento podemos dar un paso más hacia el entendimiento de las complejas relaciones entre los sistemas sociales y ambientales.

En su conjunto, este trabajo destaca la estrecha interdependencia que existe entre las ANPs como política y su contexto socioeconómico (Fig. 5.1). Es por ello que estudiar los efectos bidireccionales de la relación entre las ANPs y su contexto social permite identificar escenarios favorables en donde la conservación biológica y el desarrollo de las comunidades aledañas a las ANPs encuentren mejores resultados y al mismo tiempo desafiar los paradigmas donde el desarrollo social se ve como un obstáculo para la conservación biológica y viceversa sin diferencias matices. Por ello, es necesario identificar prácticas que permitan extender la agenda de conservación más allá de los límites mismos de las ANPs y al mismo tiempo que promuevan el desarrollo económico de manera sustentable de las comunidades adyacentes a éstas.

5.2 Aportaciones teóricas

A continuación, se mencionan algunos puntos sobre los cuales el presente trabajo aporta evidencia para enriquecer algunos debates relacionados a la relación entre la conservación y el desarrollo económico en torno de las ANPs.

1. El debate conservación-desarrollo en las ANPs. Algunos autores han cuestionado la viabilidad de compaginar la conservación y el desarrollo de comunidades en las ANPs porque intuitivamente son metas contrarias (Adams et al., 2004). No obstante, la evidencia más reciente sugiere que las ANPs por sí mismas no generan mayor pobreza (Ferraro & Hanauer, 2011; Naidoo et al., 2019; Sims & Alix-Garcia, 2017). Los resultados del capítulo 2 van en concordancia con esta evidencia, pues se demuestra que cuando se toman en cuenta las diferencias en el contexto biofísico y socioeconómicos de las ANPs, el cambio de marginación de las comunidades aledañas a éstas es similar al ocurrido en sitios no protegidos. El trabajo representa uno de los pocos esfuerzos en el país por documentar el rol de las ANPs en la reducción de pobreza, con metodologías que controlen el efecto de factores de confusión como el contexto biofísico y socioeconómico (pero ver Sims and Alix-Garcia 2017).
2. Debate sobre disyuntivas provocadas por el tipo de manejo en ANPs. Existe evidencia previa de que las ANPs con esquemas de manejo que permiten el uso de recursos naturales (i.e., esquemas menos restrictivitos) otorgan más beneficios a las comunidades locales que esquemas más restrictivos (Oldekop et al., 2016). Sin embargo, los esquemas menos estrictos sufren más deforestación dentro de sus fronteras que esquemas más restrictivos, presuntamente por permitir actividades extractivas (Wade et al., 2020). Esto plantea una disyuntiva entre conservación y desarrollo relacionada con el tipo de manejo del ANP. El capítulo 2 muestra que por sí mismos los esquemas de manejo en las ANPs mexicanas no promueven estas disyuntivas. No obstante, el contexto biofísico y socioeconómico de las comunidades

cercanas a las ANPs sí puede influir en su grado de marginación dependiendo del tipo de manejo del ANP bajo cuya influencia están. En condiciones que dificultan el desarrollo económico de las comunidades (poco acceso al mercado, lugares asilados, de pendiente pronunciada y con suelos pobres) los esquemas restrictivos están asociados a lugares con mayor marginación.

3. La efectividad de las ANPs para prevenir la transformación de los paisajes forestales., El capítulo 3 reporta que las ANPs son efectivas para reducir la pérdida de bosque en comparación con lugares no protegidos, tal como se había documentado en investigaciones previas (Andam et al., 2008; Ferraro et al., 2013; Joppa & Pfaff, 2010; Yang et al., 2021). Sin embargo, este trabajo va más allá, pues también evalúa la efectividad de las ANPs para mantener paisajes menos fragmentados. Los resultados muestran que las ANPs no son efectivas para reducir el ritmo de fragmentación ni tampoco para promover la regeneración de bosque en comparación con sitios no protegidos. Estos resultados son importantes pues a nivel global son muy escasos los trabajos que documentan el rol de las ANPs sobre la fragmentación y la regeneración de hábitat, sobre todo con el uso de métodos quasi-experimentales como las técnicas de emparejamiento (*matching analysis*, pero ver Sims, 2014).
4. Promotores de cambio en ANPs. El capítulo 3 ahonda en la discusión sobre los promotores de pérdida de bosque en ANPs pues, además de identificar a los factores demográficos como promotores de deforestación como ya lo han hecho otros trabajos (Figueroa et al., 2009; Kleemann et al., 2022; Phiri et al., 2022; Wittemyer et al., 2008), destaca el rol de la ocupación en empleos no agrícolas (un proxy de la transición forestal) como uno de los principales factores de deforestación. Este punto es importante pues, para las ANPs estudiadas, proporcionar mayor fuente de empleos en sectores no agrícolas permitiría reducir la pérdida de bosque, al disminuir la dependencia de actividades agrícolas que demanden áreas forestales para funcionar. Asimismo, en este capítulo se denota que los lugares con mayor ocupación no agrícola no solo tienen una menor tasa de deforestación sino también un mayor índice de desarrollo humano, lo cual abre posibilidades de encontrar escenarios “ganar-ganar” en términos de conservación y desarrollo. Es importante advertir que la evidencia no es directamente generalizable a otras regiones del país o del mundo ya

que sería necesario hacer más investigación para dilucidar el rol de la ocupación no agrícola en otras regiones. En el contexto mexicano, el trabajo representa el segundo aporte al estudio de promotores de deforestación en ANPs (ver Figueroa et al., 2009) y da un paso más hacia entender cuáles son los promotores de fragmentación y de regeneración de bosque.

5. Pérdida de biodiversidad en reservas. Los resultados del capítulo 4 representan un esfuerzo único en el contexto mexicano por compilar información sobre los cambios en biodiversidad en algunas ANPs. En este capítulo se ofrece evidencia empírica que indica que en las reservas existe una tendencia general a la pérdida de biodiversidad. Los resultados coinciden con los patrones generarles documentados por Laurance et al (2012) en el estudio de 60 ANPs a nivel global (incluyendo 2 mexicanas).
6. Sustitución de especies. Hasta donde es de nuestro entendimiento, los resultados del capítulo 4 representan también el primer estudio que documenta un patrón generalizado de sustitución de especies sensibles a las perturbaciones humanas (i.e. deforestación) por otras tolerantes a la misma en ANPs mexicanas. No obstante, respuestas diferenciales a los cambios de cobertura forestal han sido ya registrados en estudios a escala local en diferentes sitios en el país (Arasa-Gisbert et al., 2021; Arce-Peña et al., 2019; Carrara et al., 2015). Los resultados del capítulo 4 enfatizan también la necesidad de incluir la respuesta funcional de las especies en las estimaciones de impacto de la transformación del paisaje forestal, pues esto permitirá visualizar respuestas diferenciadas de los organismos que pueden escapar a la vista si sólo se usan estimaciones generales de biodiversidad.
7. Discusión sobre promotores de cambio en la biodiversidad. La principal causa de la pérdida de biodiversidad es la disminución del hábitat, a su vez ocasionadas por actividades humanas (IPBES, 2019; Newbold et al., 2015). Asimismo, las actividades humanas que causan la deforestación dependen de factores socioeconómicos (demográficos, económicos, políticos, etc., Geist and Lambin, 2001). Son varios las investigaciones que documentan alguna parte de esta compleja relación. Por ejemplo, los estudios de ciencias del uso de suelo que documentan los promotores del cambio de cobertura forestal (Aide et al., 2013; Armenteras et al., 2017; Kleinschroth & Healey, 2017; Lambin et al., 2003; Schulz et al., 2011); los estudios de la ecología

del paisaje documentan la respuesta de las especies a los cambios de la estructura forestal (Arroyo-Rodríguez et al., 2020; Ewers & Didham, 2006; Fahrig, 2013; Pfeifer et al., 2017; Wies et al., 2021); y también hay estudios que documentan los cambios mismos de biodiversidad en el tiempo (Laurance et al., 2012). Sin embargo, en general existe la carencia de modelos integrales que den cuenta de cómo todos estos factores se conjuntan para explicar los cambios en biodiversidad. En este sentido, el trabajo expuesto en el capítulo 4 es un esfuerzo por llenar este vacío. Los resultados subrayan la influencia que tienen factores demográficos y económicos para modificar indirectamente la riqueza y abundancia de organismos a través de los cambios en la estructura forestal.

5.3 Referencias

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Apéndices

Apéndice Capítulo 2

Apéndice 2A. Resultados suplementarios

Table S1: The protected areas used in our study

no	ID	Name	MST	Area (ha)	Year of establishment
1	2.1.01.104	Alto Golfo de California y Delta del Río Colorado	BR	934756.25	1993
2	7.3.03.025	Cañón del Río Blanco	NP	48799.77547	1938
3	8.3.04.060	Cañón del Sumidero	NP	21789.419	1981
4	9.1.02.095	Calakmul	BR	723185.125	1989
5	3.3.02.062	Cascada de Bassaseachic	NP	5802.8513	1981
6	5.1.02.106	Chamela-Cuixmala	BR	13141.69245	1994
7	7.3.02.017	Cofre de Perote o Nauhcampatépetl	NP	11530.73275	1937
8	2.1.02.105	El Pinacate y Gran Desierto de Altar	BR	714556.5	1993
9	6.3.09.015	El Tepozteco	NP	23258.7	1937
10	1.1.02.094	El Vizcaíno	BR	2546790.25	1988
11	4.3.02.010	Gogorrón	NP	38010.04111	1936
12	8.3.06.129	Huatulco	NP	11890.98	1998
13	5.3.04.034	Insurgente José María Morelos	NP	7191.769293	1939
14	6.3.02.002	Iztaccíhuatl-Popocatépetl	NP	39819.086	1935
15	3.1.01.054	La Michilía	BR	35000	1979
16	8.1.04.101	Lacan-Tun	BR	61873.96025	1992
17	8.3.01.020	Lagunas de Chacahua	NP	14896.0734	1937
18	8.3.03.046	Lagunas de Montebello	NP	6425.4927	1959
19	6.3.04.007	Los Mármoles	NP	23150	1936
20	6.3.18.031	La Montaña Malinche o Matlalcuéyatl	NP	46112.24142	1938
21	4.1.01.055	Mapimí	BR	342387.9917	2000
22	8.1.01.051	Montes Azules	BR	331200	1978
23	7.3.01.014	Pico de Orizaba	NP	19750.005	1937
24	9.1.05.134	Ría Lagartos	BR	60347.8271	1999
25	6.1.03.137	Sierra de Huautla	BR	59030.94159	1999
26	1.3.01.044	Sierra de San Pedro Mártir	NP	72910.68	1947
27	4.1.02.108	Sierra del Abra Tanchipa	BR	21464.4425	1994
28	1.1.04.109	Sierra La Laguna	BR	112437.0725	1994
29	5.3.02.006	Volcán Nevado de Colima	NP	6554.75	1936
30	6.1.01.125	Sierra Gorda	BR	383567.4488	1997
31	6.1.04.138	Mariposa Monarca	BR	56259.05073	2000
32	6.3.20.040	Bosencheve	NP	14599.61686	1940
33	4.3.05.139	Cumbres de Monterrey	NP	177395.9546	2000
34	8.1.03.096	El Triunfo	BR	119177.29	1990
35	8.1.05.118	La Encrucijada	BR	144868.1588	1995
36	8.1.06.119	La Sepultura	BR	167309.8625	1995

37	9.1.07.143	Ría Celestún	BR	81482.33446	2000
38	9.1.06.136	Los Petenes	BR	282857.6271	1999
39	7.1.02.133	Los Tuxtlas	BR	155122.469	1998
40	6.1.05.144	Barranca de Metztitlán	BR	96042.94702	2000
41	7.1.01.098	Pantanos de Centla	BR	302706.625	1992
42	8.1.02.072	Selva El Ocote	BR	101288.1513	1982
43	9.1.01.073	Sian Ka'an	BR	528147.668	1986
44	5.1.01.090	Sierra de Manantlán	BR	139577.125	1987
45	6.1.02.130	Tehuacán-Cuicatlán	BR	490186.8755	1998
46	1.1.01.049	Complejo Lagunar Ojo de Liebre	BR	79328.97663	1972

*BR: biosphere reserves, NP: national parks.

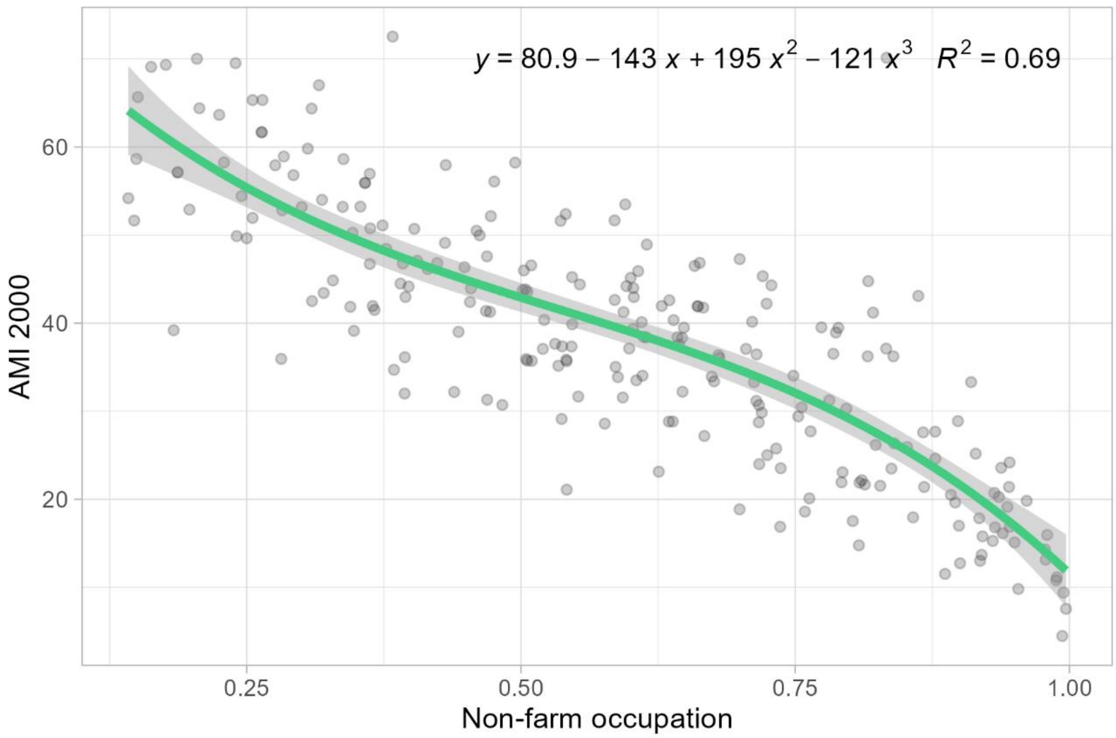

Fig. S1: Relationship between non-farm occupation and the Absolute Marginalization Index (AMI) in the year 2000. Gray points represent a single municipality while the green line corresponds to a third-order polynomial regression fit.

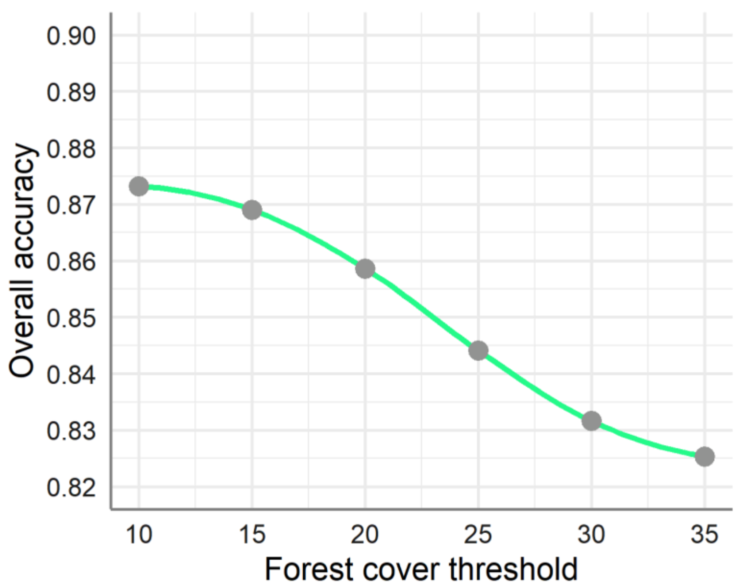

Fig. S2: Relationship between forest cover threshold and forest cover classification accuracy.

Fig. S3: The economical regions of Mexico. Each region groups municipalities with similar resources, and economic activities (CONABIO, 2010d). Each region provides different economic opportunities for investment (Bassols Batalla, 2006).

Table S2: Description of the different economic regions in terms of the number of PAs, the distance to cities, the distance to roads, and the agriculture suitability index. The values correspond to the mean value (± standard error).

Economic zone	Number of PAs	Distance to cities	Distance to roads	Agriculture suitability
General	75	59.5 ± 0.15	5.29 ± 0.02	0.47 ± 0.0006
Centralwest	11	29.94 ± 0.24	4.21 ± 0.04	0.45 ± 0.0018
Southcentral	17	20.75 ± 0.2	3.29 ± 0.04	0.46 ± 0.0027
Gulf of Mexico	7	24.71 ± 0.22	6.01 ± 0.13	0.71 ± 0.0024
Northeast	3	47.07 ± 0.29	4.61 ± 0.05	0.47 ± 0.0018
Northwest	14	87.66 ± 0.5	6.24 ± 0.05	0.42 ± 0.0011
North	19	72.36 ± 0.21	4.68 ± 0.02	0.46 ± 0.0009
South Pacific	15	40.35 ± 0.22	6.92 ± 0.08	0.48 ± 0.0022
Yucatan Peninsula	10	52.43 ± 0.36	5.97 ± 0.07	0.57 ± 0.001

Table S3: Description of protected (inside PAs) and unprotected (outside PAs) zones within the different economic regions in terms of the number of PAs, the distance to cities, the distance to roads, and the agriculture suitability index. The values correspond to the mean value (± standard error).

	Distance to cities		Distance to roads		Agriculture suitability	
	Protected	Unprotected	Protected	Unprotected	Protected	Unprotected
General	110 ± 0.91	54.7 ± 0.13	8.44 ± 0.08	4.99 ± 0.02	0.41 ± 0.002	0.48 ± 0.0006
Centralwest	44.01 ± 1.47	29.16 ± 0.24	8.37 ± 0.3	3.98 ± 0.04	0.37 ± 0.0084	0.45 ± 0.0019
Southcentral	32.19 ± 0.9	19.18 ± 0.17	4.19 ± 0.14	3.17 ± 0.04	0.34 ± 0.0081	0.48 ± 0.0028
Gulf of Mexico	23.99 ± 0.87	24.76 ± 0.23	8.3 ± 0.38	5.87 ± 0.14	0.65 ± 0.0116	0.72 ± 0.0024
Northeast	20.81 ± 0.92	47.55 ± 0.29	7.89 ± 0.5	4.55 ± 0.05	0.15 ± 0.0126	0.48 ± 0.0017
Northwest	179.2 ± 1.52	68.6 ± 0.36	10.28 ± 0.15	5.4 ± 0.04	0.39 ± 0.0025	0.42 ± 0.0012
North	78.98 ± 0.76	71.9 ± 0.22	7.14 ± 0.13	4.5 ± 0.02	0.36 ± 0.0038	0.46 ± 0.0009
South Pacific	59.83 ± 1.35	39.16 ± 0.22	8.75 ± 0.27	6.81 ± 0.08	0.51 ± 0.0109	0.48 ± 0.0023
Yucatan Peninsula	79.26 ± 1.26	48.39 ± 0.34	7.42 ± 0.19	5.75 ± 0.08	0.55 ± 0.0032	0.58 ± 0.0011

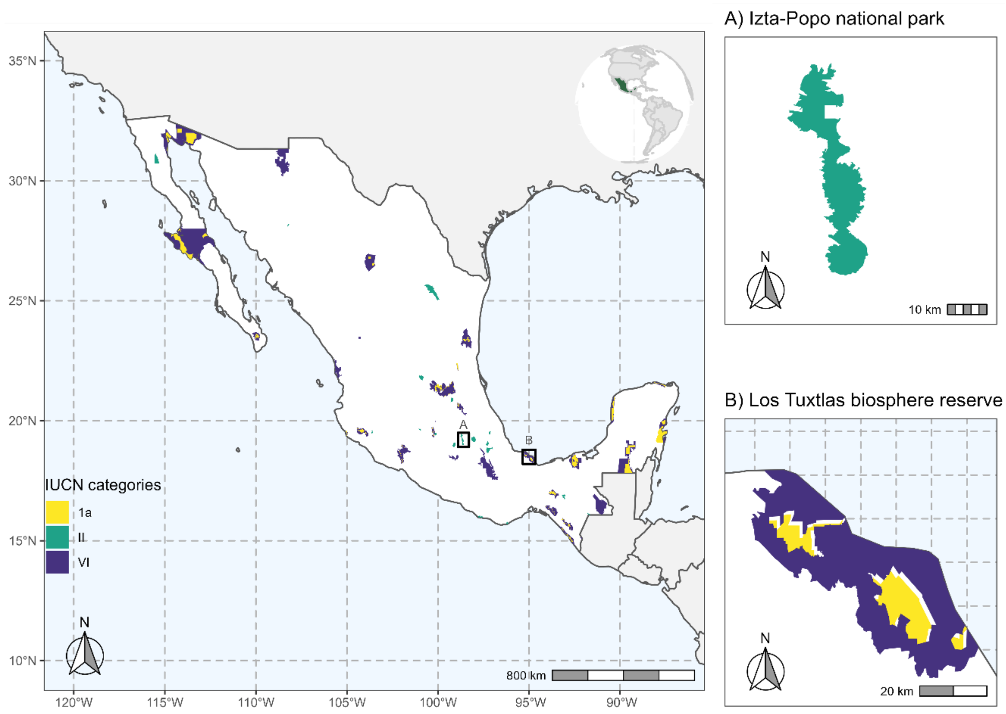

Fig. S4: IUCN categories of the studied PAs. National parks (as in A) are identified with category II. Biosphere reserves have two zones: the core area identified with category Ia, and the buffer area identified with category VI (as shown in B). These data were obtained from the national commission on protected areas (CONANP, 2019).

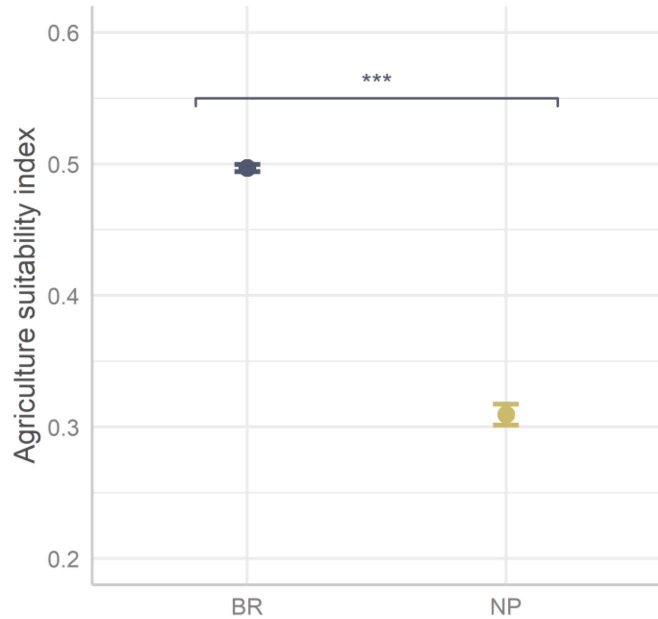

Fig. S5: Differences in agriculture suitability between biosphere reserves (BR) and national parks (NP). Note that NP are located in places with poorer conditions for agriculture development (i.e., high elevation, stepper slopes, poor soils, etc). *p < 0.05, ** p < 0.01, *** p < 0.001

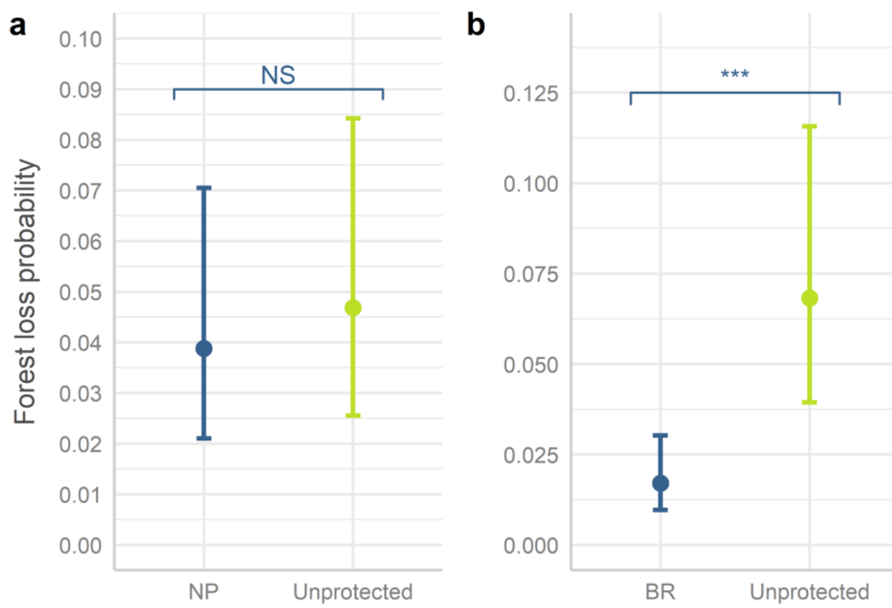

Fig. S6: Comparison of forest loss probabilities occurred in national parks (NP) (a), and biosphere reserves (BR) (b) regarding unprotected areas. We applied matching technics to reduce the effect of biophysical differences between PAs and unprotected areas, and then we used generalized linear models to test for differences. *p < 0.05, ** p < 0.01, *** p < 0.001.

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Apéndice 2B. Cálculo del índice de marginación

To calculate the Absolute Marginalization Index (AMI) we follow the methodology of the Mexican Council of population. Details of the theory and calculation can be found in the official documentation of this institution (CONAPO 2013), but here, we summarize the calculation.

AMI is composed of nine indicators on a municipality scale: 1) illiterate population of 15 years or more, 2)population of 15 years or more without elementary school, 3) the percentage of people in dwellings without drainage service, 4) percentage of people in dwellings without electric energy, 5) percentage of people living in dwellings without piped water, 6) percentage of dwellings with some level of overcrowding, 7) the percentage of people in dwellings with earth floor, 8) percentage of the people living in localities with less than 5,000 inhabitants, 9) the percentage of people with an income equal or less than two Mexican minimum wages (~9 dollars per day).

We used the data of the official population census of 2000 (INEGI, 2000) and 2020 (INEGI, 2020)to determine AMI for these years. Illiterate population of 15 years or more (I_{i1}) was calculated as:

$$I_{i1} = \frac{P_i^{ill}}{P_i^{+15} - Na_i} \quad \text{(Eq. B1)}$$

Where P_i^{ill} corresponds to the illiterate population in municipality *i* and P_i^{+15} are the total population higher than 15 years old and Na is the number of people that did not answer this specific question in municipality *i*.

The population of 15 years or more without elementary school at the municipality *i* (I_{i2}) was calculated in two steps. First, we used the next formula

$$PP_i^{1-5} = P_i^{1-5} + \left[\frac{P_i^{1-5}}{P_i^{1-5} + P_i^{6}} \times Na_i^{g}\right] \quad \text{(Eq. B2)}$$

Where PP_i^{1-5} is the population of the municipality *i* of 15 years or more with elementary studies between the 1th and the 5th grade, P_i^{1-5} is the number of people that declare to successfully passed between the 1th and the 5th grade of the elementary school in the municipality *i*, P_i^6 is the number of people that finished their elementary studies in the municipality *i* but did not continue studying, and Na_i^g is the number of people that did not finish the elementary school but that did not specify the last grade of studies in the municipality *i*.

Then we calculate Ii_2 as follows:

$$I_{i2} = \frac{P_{ws} + PP_i^{1-5}}{P_i^{+15} - Na_i} \times 100 \quad \text{(Eq. B3)}$$

Where P_{ws} is the population without studies, P_i^{+15} is the total population higher than 15 years old and Na is the number of people that did not answer this specific question.

The percentage of people in dwellings without drainage service in the municipality *i* (I_{i3}) was calculated with:

$$I_{i3} = \frac{O_i^{ds}}{O_i^t - Na_i} \times 100 \quad \text{(Eq.B4)}$$

Where O_i^{ds} is the number of dwelling occupants with no drainage service in the municipality *i*, O_i^t is the total number of dwelling occupants in municipality *i*, and Na is the number of dwelling occupants from which there is no information on this specific answer in the municipality *i*.

The percentage of people at municipality *i* in dwellings without electric energy (I_{i4}) was calculated with the next formula:

$$I_{i4} = \frac{O_i^{nel}}{O_i^t - Na_i} \times 100 \quad \text{(Eq. B5)}$$

Where O_i^{nel} is the number of dwelling occupants with no electric energy, O_i^t is the total number of dwelling occupants at municipality *i*, and Na is the number of dwelling occupants from which there is no information on this specific answer.

The percentage of people living in dwellings without piped water at municipality *i* (I_{i5}) was calculated as follows:

$$I_{i5} = \frac{O_i^{npw}}{O_i^t - Na_i} \times 100 \quad \text{(Eq. B6)}$$

Where O_i^{npw} is the number of dwelling occupants with no piped water service in the municipality *i*, O_i^t is the total number of dwelling occupants in the municipality i, and Na is the number of dwelling occupants from which there is no information on this specific answer in the municipality *i*.

The percentage of dwellings with some level of overcrowding in the municipality *i* (I_{i6}) was calculated as follows:

$$I_{i6} = \frac{D_i^{o}}{D_i^t - Na_i} \times 100 \quad \text{(Eq. B7)}$$

Where D_i^o is the number of dwellings with some level of overcrowding in the municipality *i*, D_i^t is the total number of dwellings in the municipality *i*, and Na is the number of dwellings from which there is no information on this specific answer in the municipality *i*. We say that a dwelling is overcrowded when there exist: dwellings with one restroom but three or more occupants, dwellings with two restrooms but with five or more occupants, dwellings with three restrooms but with seven or more occupants, and finally, dwellings with four restrooms but with nine or more occupants.

The percentage of people in dwellings with earth floors at municipality *i* (I_{i7}) was calculated with the next formula:

$$I_{i7} = \frac{O_i^{ef}}{O_i^t - Na_i} \times 100 \quad \text{(Eq. B8)}$$

Where O_i^{ef} is the number of dwelling occupants with earth floor in the municipality *i*, O_i^t is the total number of dwelling occupants in the municipality *i*, and Na is the number of dwelling occupants from which there is no information on this specific answer in the municipality *i*.

The percentage of the people living in localities with less than 5,000 inhabitants in the municipality *i* (I_{i8}) was calculated with the next formula:

$$I_{i8} = \frac{P_i^{l<5000}}{P_i^t} \times 100 \quad \text{(Eq. B9)}$$

Where $P_i^{l<5000}$ is the number of people in localities with a population of fewer than 5000 people in municipality *i*, and P_i^t is the total number of people in municipality *i*.

The percentage of people with an income equal to or less than two Mexican minimum wages in the municipality *i* (I_{i9}) was calculated as follows:

$$I_{i9} = \frac{P_i^{mw<2}}{P_i^w} \times 100 \quad \text{(Eq. B10)}$$

Where $P_i^{mw<2}$ is the number of people with no monetary income or with revenues that represent less than 2 minimum wages in municipality *i*, and P_i^w is the total number of people with employment in municipality *i*.

Finally, we applied the formula:

$$AMI = \frac{\sum_j^9 I_{ij}}{9} \quad \text{(Eq. B11)}$$

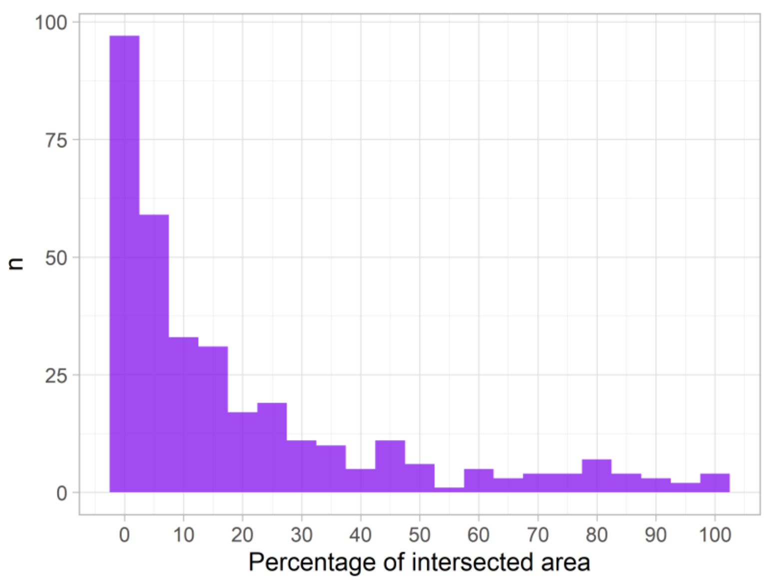

Fig. S1: Histogram on the percentage of the municipality area covered by a PA as a proxy of the degree of protection of the municipalities.

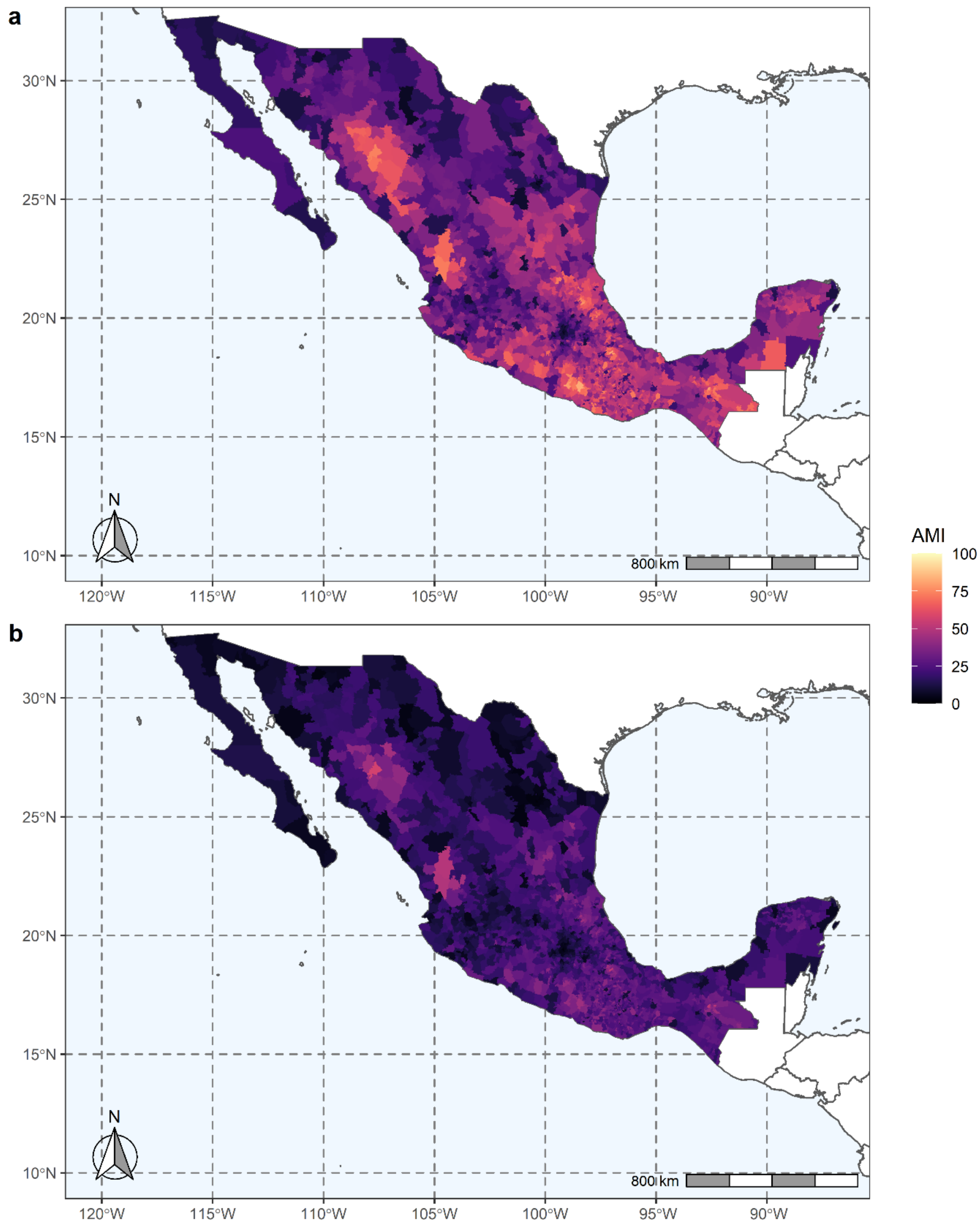

Fig. S2: AMI calculation for each Mexican municipality for the years 2000 (a) and 2020 (b).

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Apéndice 2C. Cálculo de covariables

Covariates for the forest loss data

We used the distance to cities, distance to roads, and agriculture suitability index as covariates that may act as confounding factors in the forest loss dynamic.

We calculated the distance to cities, defined as localities with a population of 15,000 or higher, through a geographic information system (GIS). To do that, we use vectorial data of Mexican localities for the year 2000 (CONABIO, 2002). We generated a raster dataset of 1km of cell size that contains the information.

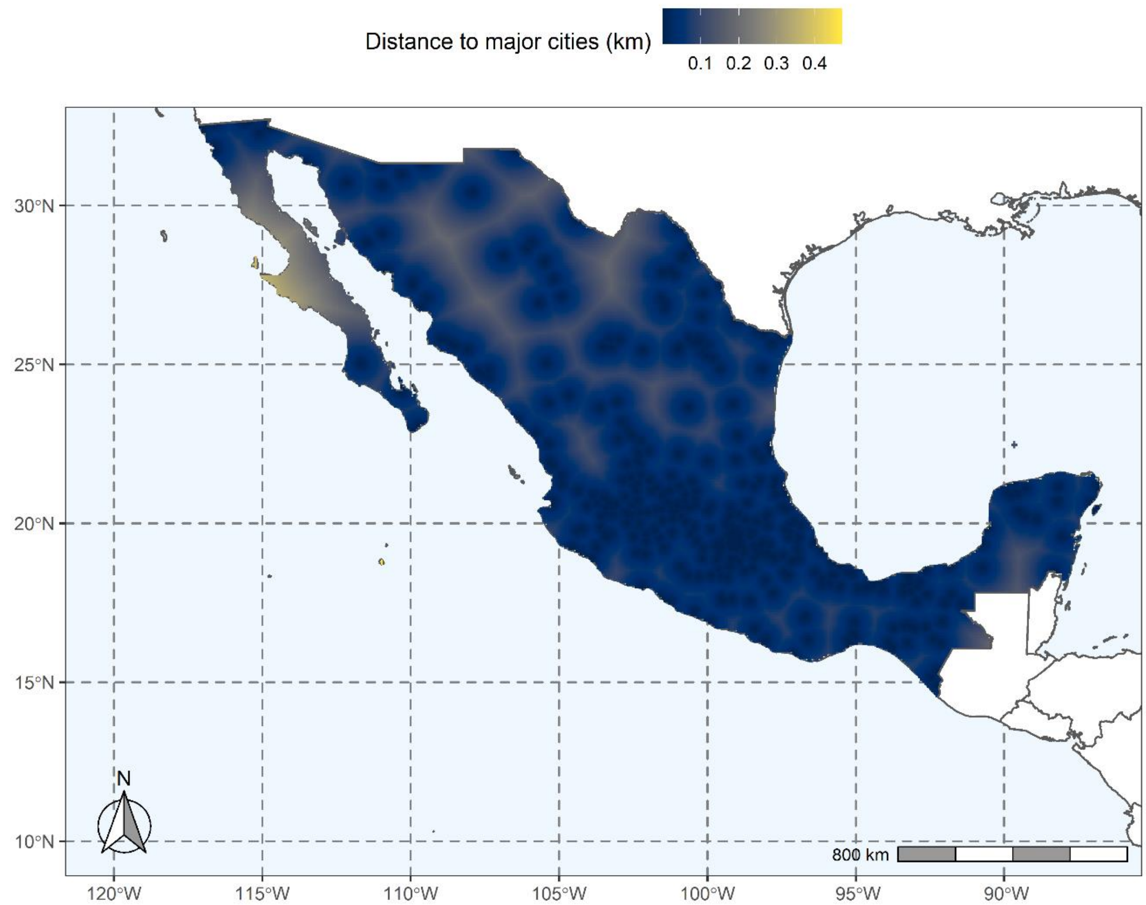

Fig. S1: Map of the distribution of distance to major cities in the Mexican territory.

To calculate the distance to roads we used vectorial data from the national roads network (INEGI, 2018). We assessed the distance to roads through the GIS and generate a raster dataset of 1km of cell size.

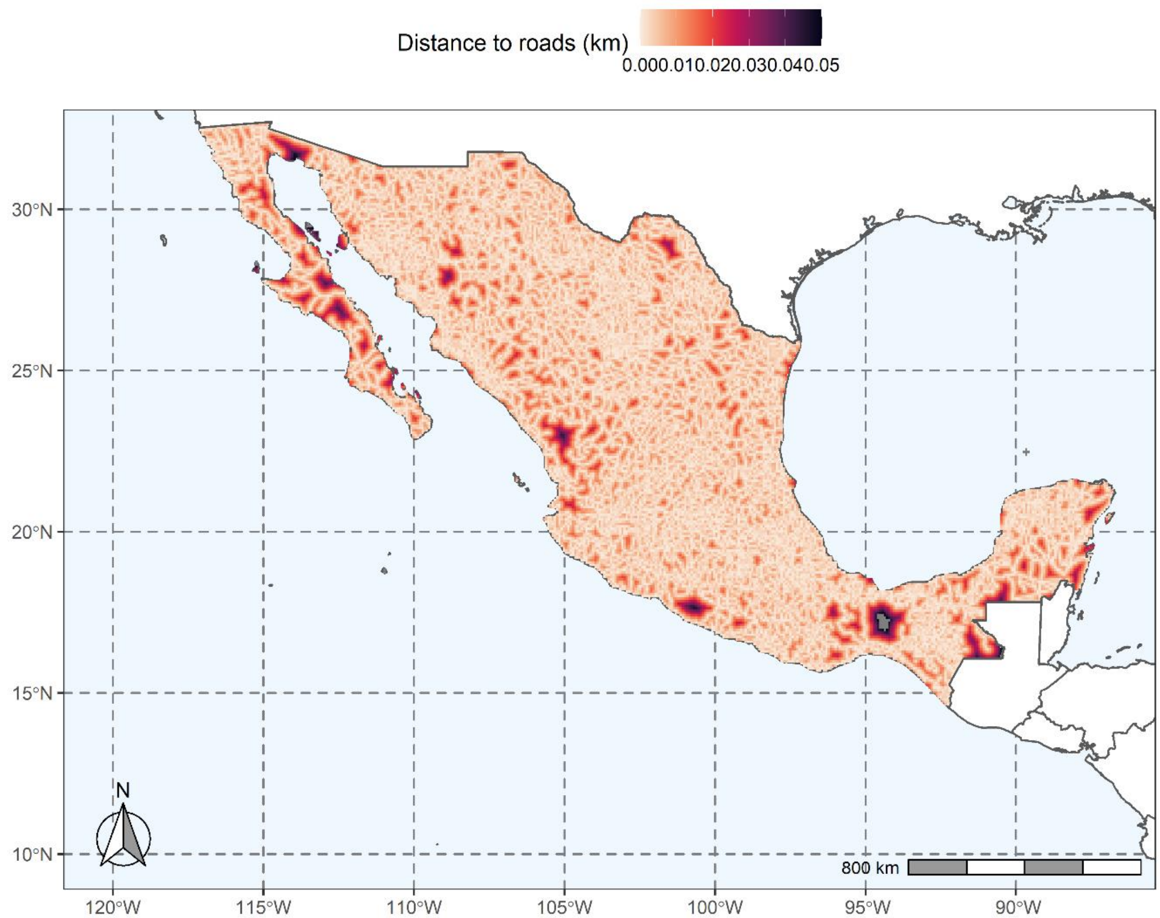

Fig. S2: Map of the distribution of distance to roads in the Mexican territory.

We calculated the agriculture suitability index using data on climate, soil, and orography. As climate variables, we considered the mean annual temperature, temperature annual range, mean annual precipitation, and precipitation of the driest quarter (Fick & Hijmans, 2017). As soil variables, we used the concentration of Ca, Na, organic carbon, and pH (INEGI, 2013a). And as orographic variables, we use elevation and slope (INEGI, 2013b). We obtained data on the presence or absence of agricultural lands from the Mad-Mex land use/cover classification. Mad-Mex provides information on 17 land cover classes at 30m of resolution. We reclassified these classes in a binary raster of agriculture/non-agriculture.

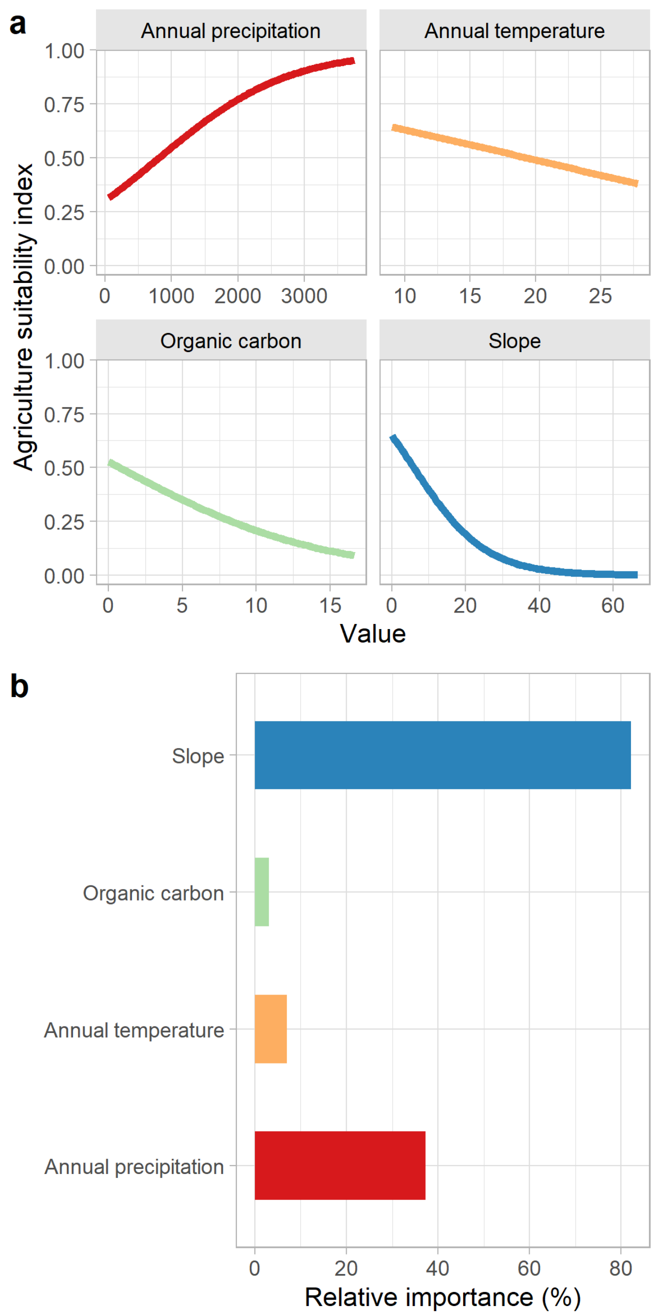

Fig. S3: a) Predicted response curves of the explanatory variables as predictors of agriculture suitability index according to binomial model. b) Relative importance of each explanatory variable determined by Pearson correlation permutation.

To calculate the agriculture suitability index, we used binomial generalized linear models (GLM). As a response variable, we use the presence-absence of agriculture and we train the model with all the set of climates, soil, and, orographic variables previously described. We used 1600 random points isolated at a distance of at least 1km from each other and seeded in the entire Mexican territory to sample the variables. Through this approach, we describe the combination of conditions where agriculture activities are developing in the country with a low level of spatial autocorrelation (Moran's I= 0.0012, p=0.79). To select a candidate set of explanatory variables, we tested for their collinearity and their relative importance through correlation and area under the curve metrics. We selected the most parsimonious model based on AIC criteria. Finally, the selected explanatory variables were: slope, organic carbon, mean annual temperature, and mean annual precipitation (Fig. C3).

We predicted agriculture suitability for the entire Mexican territory and we evaluated the accuracy of the model through the ROC curve. All the analysis was performed using the *sdm* package of R (Naimi & Araújo, 2016).

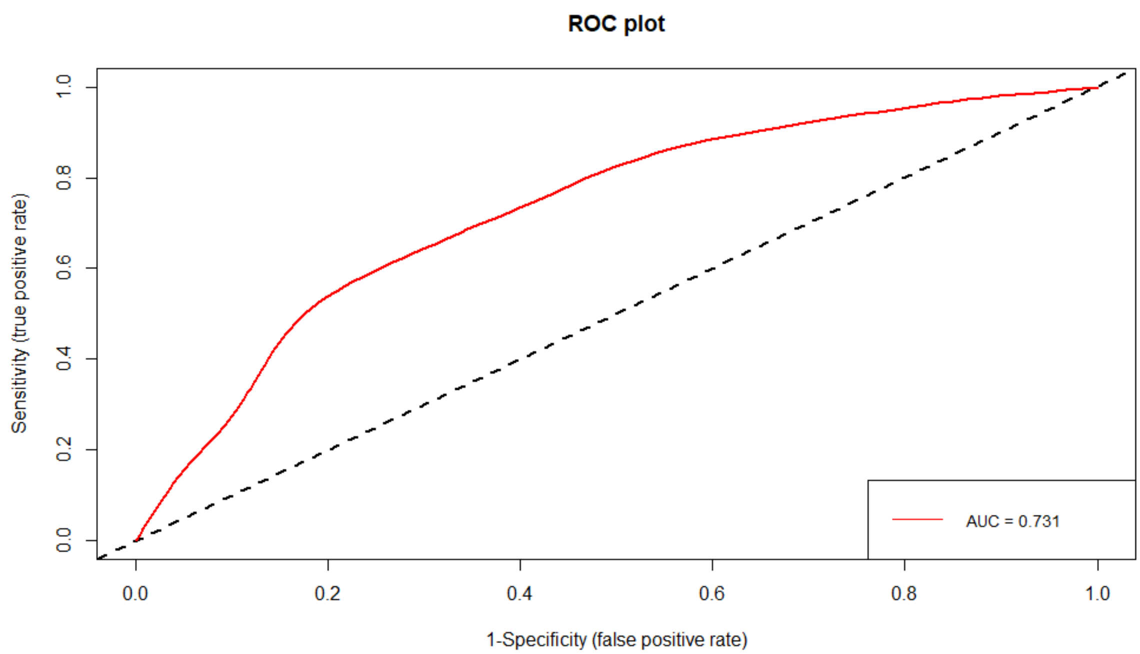

Fig. S4: Receiver operating curve (ROC) of GLM model. It is shown the area under the curve (AUC) that represents a medium performance level.

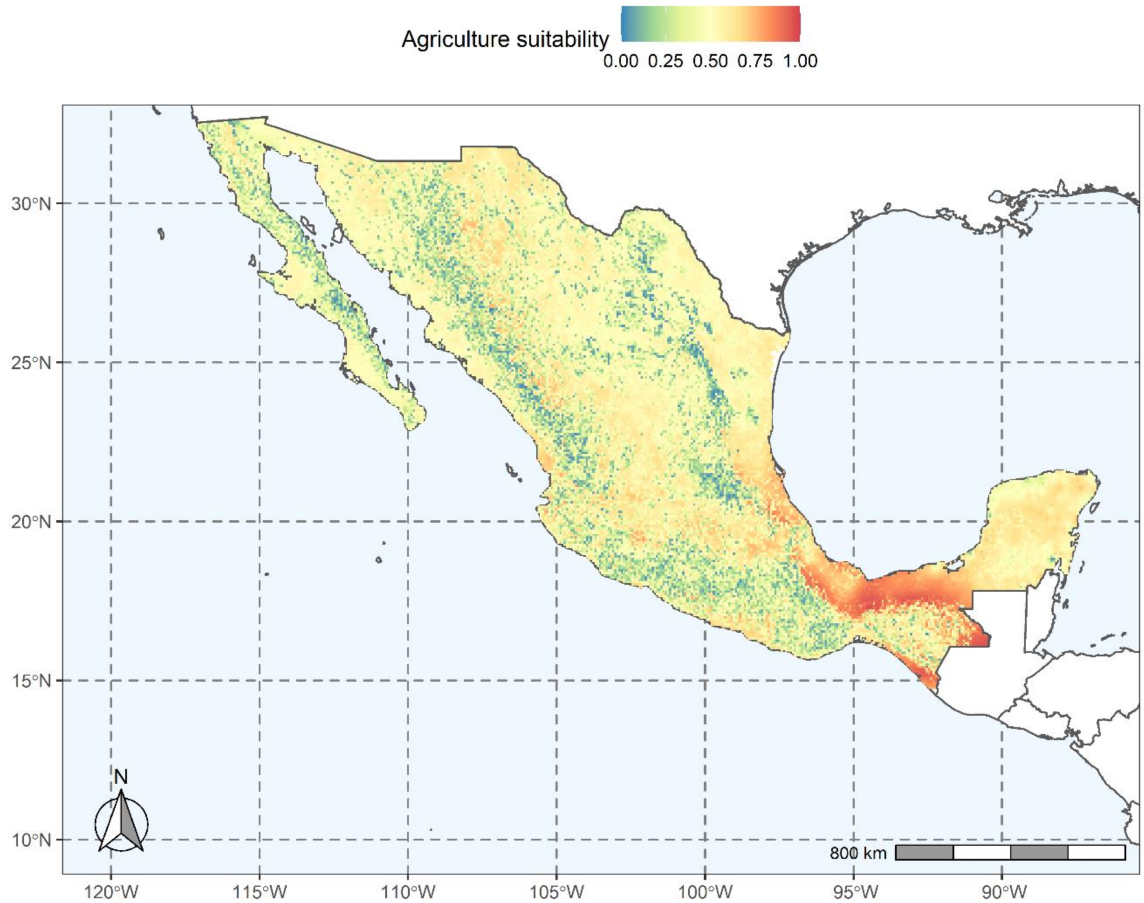

Fig. S5: Map of the distribution of agriculture suitability index in the Mexican territory.

Covariates for the marginalization data

We used the distance to cities, agriculture suitability index, and non-farm occupation as covariates that may act as confounding factors in the forest loss dynamic.

Non-farm occupation assessed the proportion of the population in a municipality working in the industrial, professional, and services sectors. Details of this indicator are provided in Auliz-Ortiz et al., (2022).

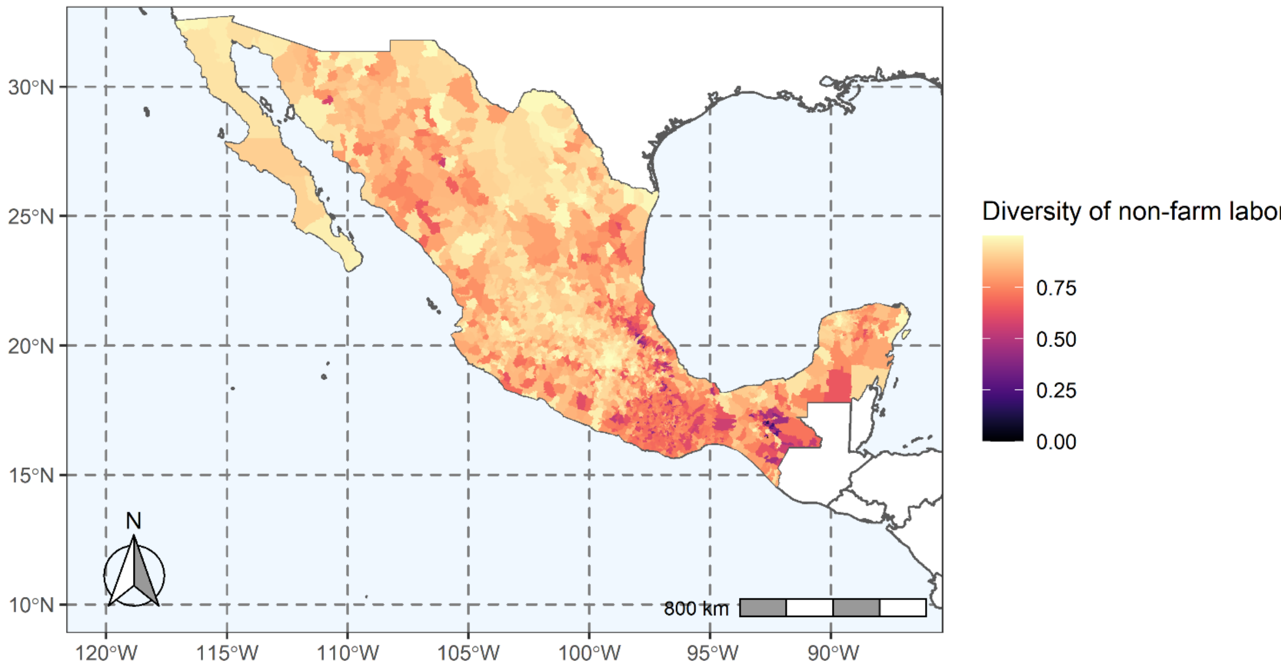

Fig. S6: Map of the distribution of the diversity of non-farm labor calculated in each municipality for 2000.

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Apéndice 2D. Precisión de datos de cobertura forestal

In order to compare property forest changes in the period 2000-2019, we discriminate between areas covered by forest and no forest in the year 2000. Using a set of five hundred sampling points seeded randomly in the Mexican territory we subtracted the data of tree cover in the year 2000 from the corresponding layer of Global Forest Change (Hansen et al., 2013) which provide the percentage of tree cover per pixel at a resolution of 30m (Fig. D1). Then, we used the forest cover classification for the year 2001 produced by the National Institute on Statistics and Geography (INEGI, 2003) as well as cloud-free Landsat composite images for the year 2000 in the Google Earth Engine platform as ancillary data to determine the actual cover (forest or no forest) in that year.

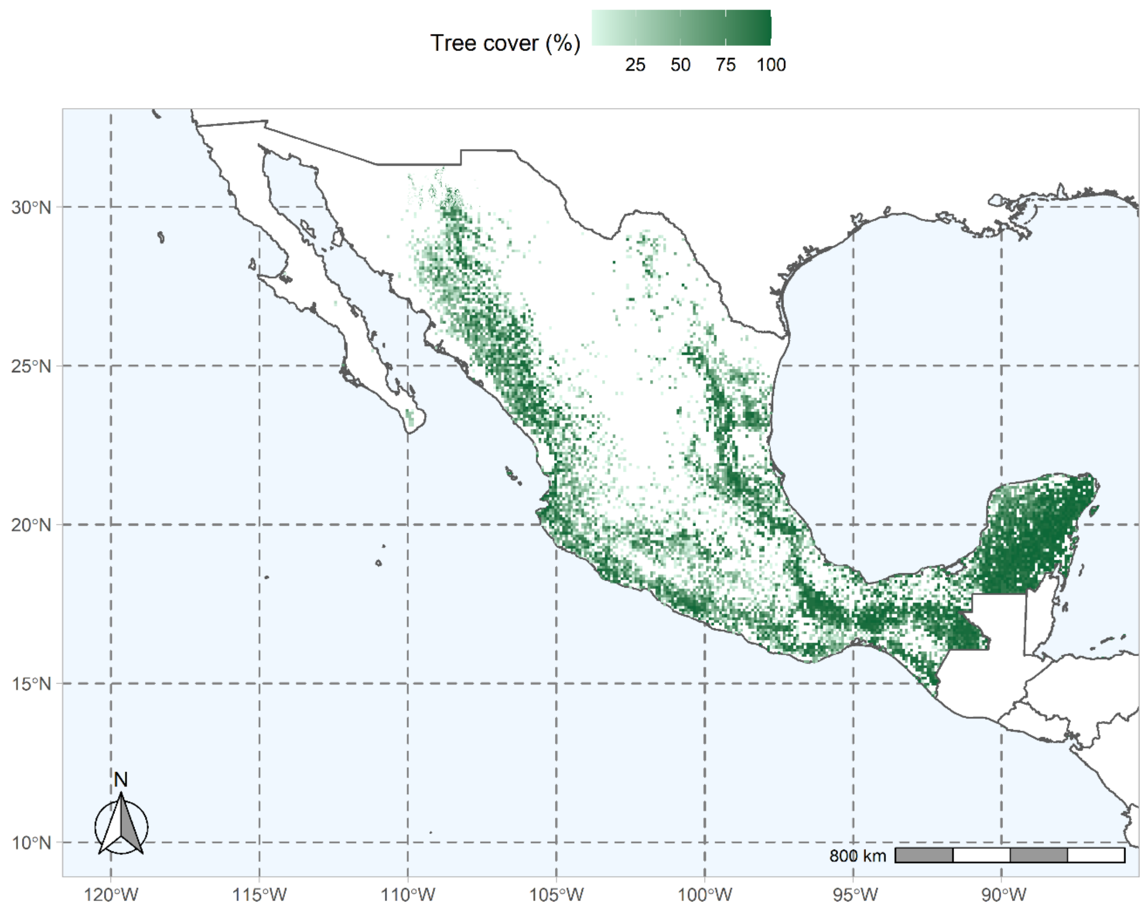

Fig. S1: Map of the tree cover distribution in the Mexican territory.

Subsequently, we define six tree cover thresholds to discriminate forest/no forest cover: 10%, 15%, 20%, 25%, 30%, and 35%. We evaluate the accuracy of forest cover identification in each threshold through the confusionMatrix function of the caret package in R. Finally, we select the threshold with the higher overall accuracy that corresponds to 10% (see Fig. A2, Appendix 2A).

References

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Apéndice 2E. Balance de covariables y análisis de matching

Here we describe the additional matching analysis performed to assess the effect of biosphere reserves, national parks, and unprotected places on marginalization evolution and forest loss. In the same way, we did in the matching analysis described in the main text, we use distance to cities, distance to roads, agriculture suitability index, and diversity of non-farm occupation as covariates that influence forest loss and marginalization that may act as confounding factors.

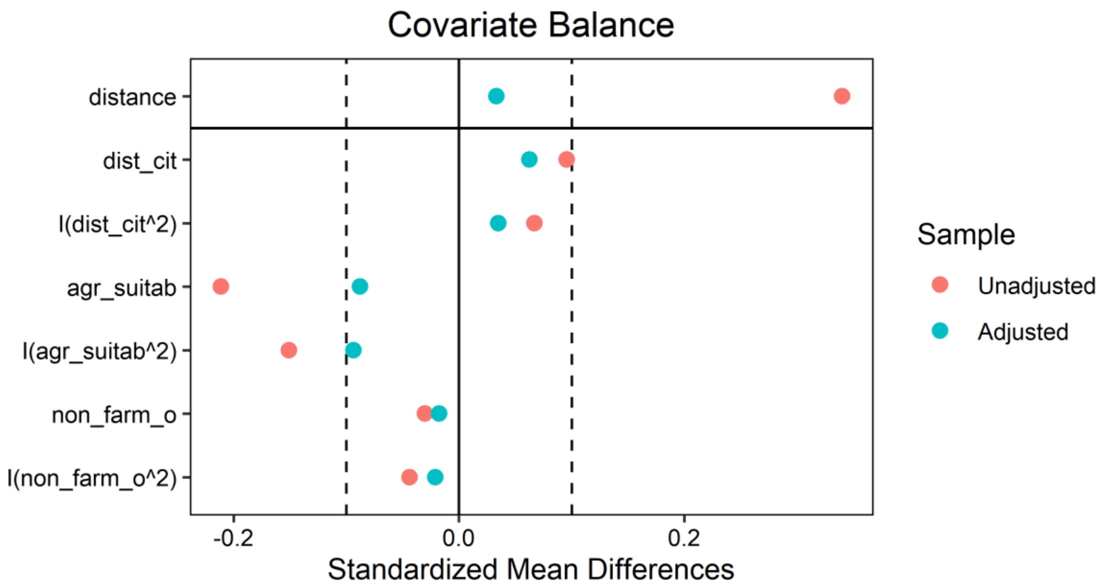

Fig. S1: Covariate balance for the marginalization data using a generalized linear model and selecting municipalities influenced directly by PAs (treatment) and with no direct influence of them (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.1 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, agr_suitab: agriculture suitability, non_farm_o: non farm occupation.

Table S1: Summary of balance for matched data for the forest loss model that includes municipalities influenced directly by PAs (treatment) and with no direct influence of them (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.1778	0.1747	0.0332	1.2781	0.0008	0.0192	0.0349
dist_cit	29.47	27.867	0.0625	1.0842	0.0284	0.0654	0.9393
I(dist_cit^2)	1523.235	1380.4946	0.035	1.8474	0.0284	0.0654	0.5174
agr_suitab	0.4856	0.4998	-0.0876	0.9394	0.025	0.0692	0.6472
I(agr_suitab^2)	0.2619	0.2775	-0.0934	0.8642	0.025	0.0692	0.7232
non_farm_o	0.5927	0.5966	-0.0176	0.983	0.0117	0.0423	1.0359
I(non_farm_o^2)	0.4012	0.4068	-0.0207	0.9702	0.0117	0.0423	1.0395

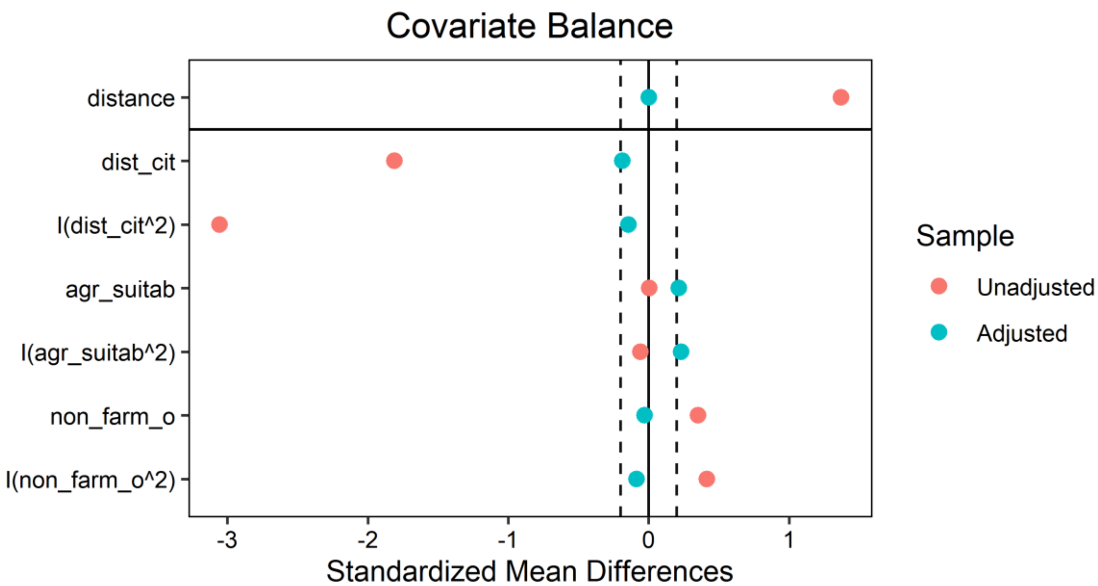

Fig. S2: Covariate balance for the forest loss data using a generalized linear model and selecting municipalities influenced by national parks (treatment) and biosphere reserves (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.25 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, agr_suitab: agriculture suitability, non_farm_o: non farm occupation.

Table S2: Summary of balance for matched data for the forest loss model that includes municipalities influenced directly by national parks (treatment) and biosphere reserves (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.4967	0.4961	0.0026	0.9815	0.0063	0.0758	0.0205
dist_cit	17.571	19.747	-0.1879	0.9731	0.058	0.1667	0.6021
I(dist_cit^2)	446.4794	528.8471	-0.1439	0.9331	0.058	0.1667	0.5282
agr_suitab	0.4852	0.4549	0.2143	1.0965	0.1155	0.2576	1.2865
I(agr_suitab^2)	0.2567	0.226	0.2308	0.8251	0.1155	0.2576	1.313
non_farm_o	0.5861	0.5936	-0.0292	0.7184	0.0466	0.1364	0.9856
I(non_farm_o^2)	0.3929	0.4199	-0.0849	0.7575	0.0466	0.1364	0.8579

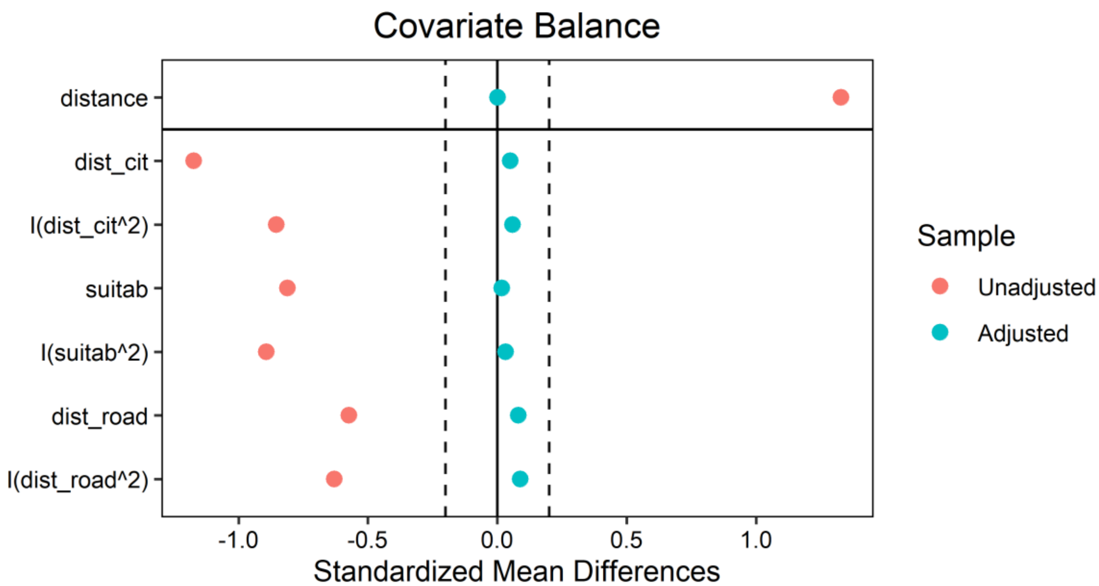

Fig. S3: Covariate balance for the forest loss data using a generalized linear model and selecting only national parks (treatment) and biosphere reserves (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.2 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, suitab: agriculture suitability, dist_road: distance to roads.

Table S3: Summary of balance for matched data for the forest loss model that includes only national parks (treatment) and biosphere reserves (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
Distance	0.4015	0.4012	0.0011	1.0036	0.0001	0.0144	0.0025
dist_cit	29.2245	27.4051	0.0503	1.1965	0.0168	0.0525	0.5073
I(dist_cit^2)	2161.4943	1842.9653	0.0592	1.2877	0.0168	0.0525	0.4617
suitab	0.3094	0.3054	0.0174	1.0577	0.0364	0.089	0.9109
I(suitab^2)	0.1493	0.1438	0.0321	0.8436	0.0364	0.089	0.8935
dist_road	5.1966	4.8116	0.0808	1.1935	0.0224	0.0506	0.9608
I(dist_road^2)	49.6728	42.1314	0.0893	1.0712	0.0224	0.0506	0.7563

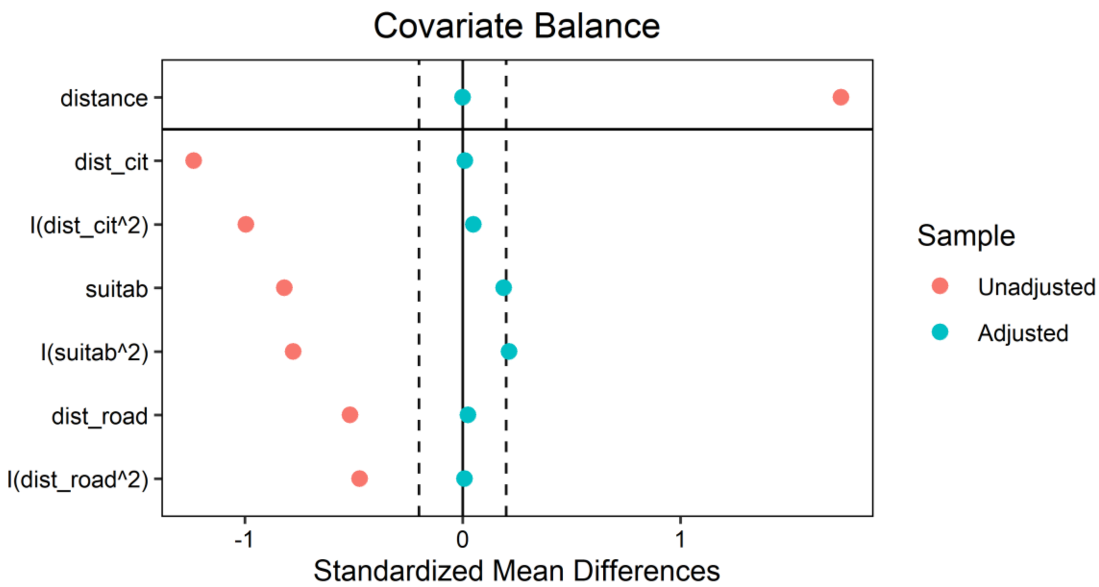

Fig. S4: Covariate balance for the forest loss data using a generalized linear model and selecting only national parks (treatment) and core zone of biosphere reserves (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.2 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, suitab: agriculture suitability, dist_road: distance to roads.

Table S4: Summary of balance for matched data for the forest loss model that includes only national parks (treatment) and core zone of biosphere reserves (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.6556	0.6556	-0.0001	0.9964	0.0005	0.0231	0.002
dist_cit	29.9049	29.4863	0.0116	1.215	0.036	0.1356	0.4617
I(dist_cit^2)	2221.6572	1958.1267	0.049	1.2361	0.036	0.1356	0.4083
suitab	0.316	0.2725	0.188	1.24	0.071	0.134	0.8324
I(suitab^2)	0.1532	0.1171	0.2132	1.1587	0.071	0.134	0.7939
dist_road	5.3083	5.187	0.0255	0.9738	0.0088	0.0466	0.9691
I(dist_road^2)	51.0088	50.268	0.0088	0.8864	0.0088	0.0466	0.8014

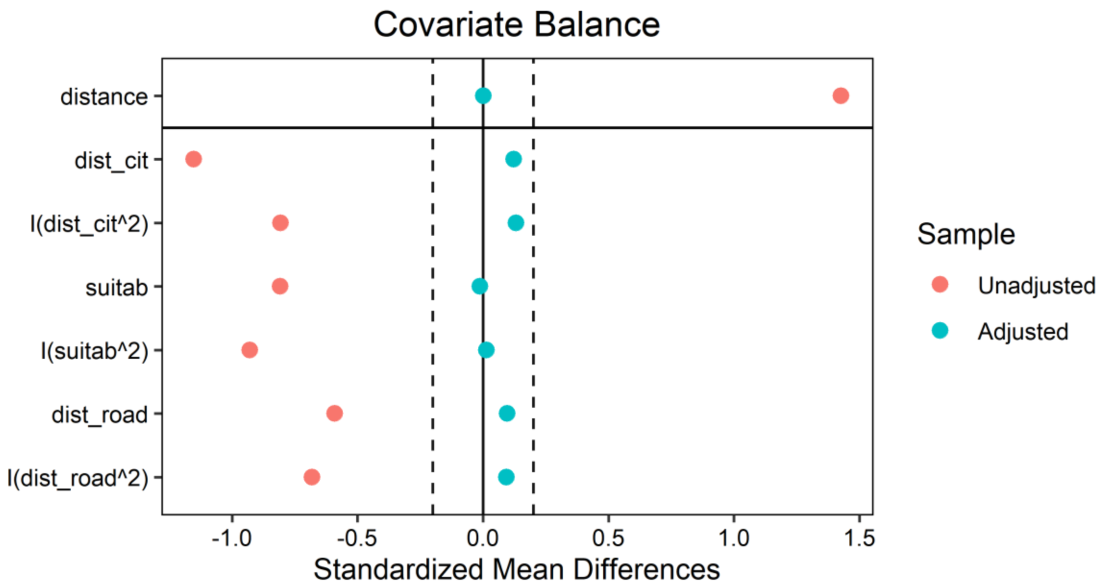

Fig. S5: Covariate balance for the forest loss data using a generalized linear model and selecting only national parks (treatment) and buffer zone of biosphere reserves (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.2 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, suitab: agriculture suitability, dist_road: distance to roads.

Table S5: Summary of balance for matched data for the forest loss model that includes only national parks (treatment) and buffer zone of biosphere reserves (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.4498	0.4494	0.0016	1.0052	0.0001	0.0178	0.0024
dist_cit	29.2245	24.7768	0.123	1.5496	0.0313	0.0734	0.4965
I(dist_cit^2)	2161.4943	1456.8412	0.1309	1.9767	0.0313	0.0734	0.4365
suitab	0.3094	0.3124	-0.0128	1.085	0.0412	0.1147	0.9474
I(suitab^2)	0.1493	0.1469	0.0142	0.8301	0.0412	0.1147	0.9162
dist_road	5.1966	4.7379	0.0963	1.1705	0.027	0.0528	0.9724
I(dist_road^2)	49.6728	41.7967	0.0932	0.923	0.027	0.0528	0.7747

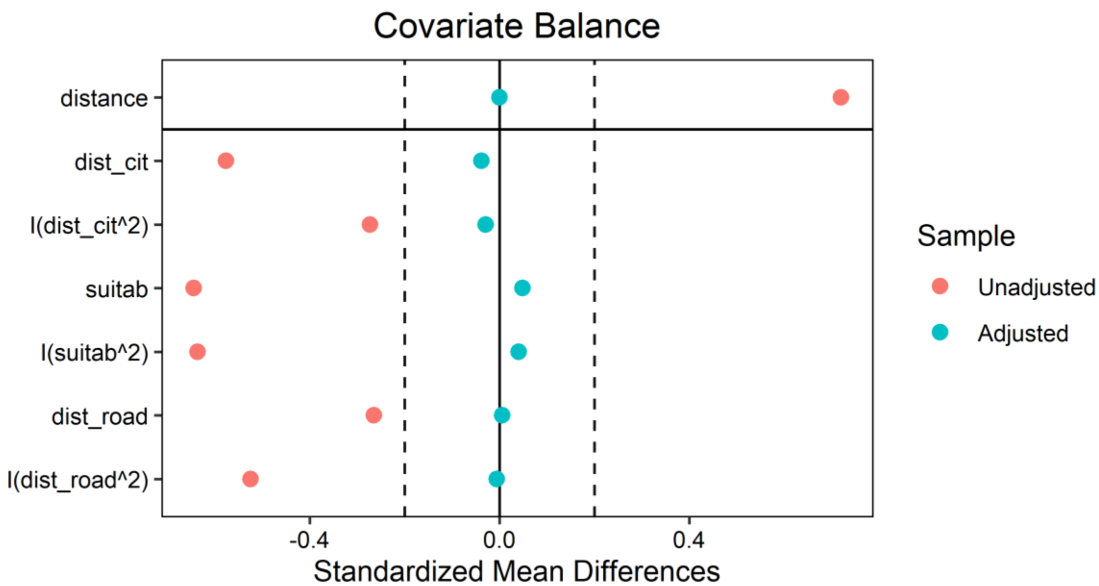

Fig. S6: Covariate balance for the forest loss data using a generalized linear model and selecting only national parks (treatment) and unprotected areas (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.2 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, suitab: agriculture suitability, dist_road: distance to roads.

Table S6: Summary of balance for matched data for the forest loss model that includes only national parks (treatment) and unprotected areas (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.036	0.036	0	0.9997	0	0.0009	0.0002
dist_cit	29.2245	30.5846	-0.0376	0.947	0.0275	0.1081	0.6222
I(dist_cit^2)	2161.4943	2315.9515	-0.0287	0.7384	0.0275	0.1081	0.5744
suitab	0.3094	0.2982	0.0485	0.9998	0.0323	0.0675	0.69
I(suitab^2)	0.1493	0.1425	0.0402	0.8313	0.0323	0.0675	0.7055
dist_road	5.1966	5.1686	0.0059	0.9687	0.0109	0.0515	1.015
I(dist_road^2)	49.6728	50.1155	-0.0052	0.4659	0.0109	0.0515	0.8544

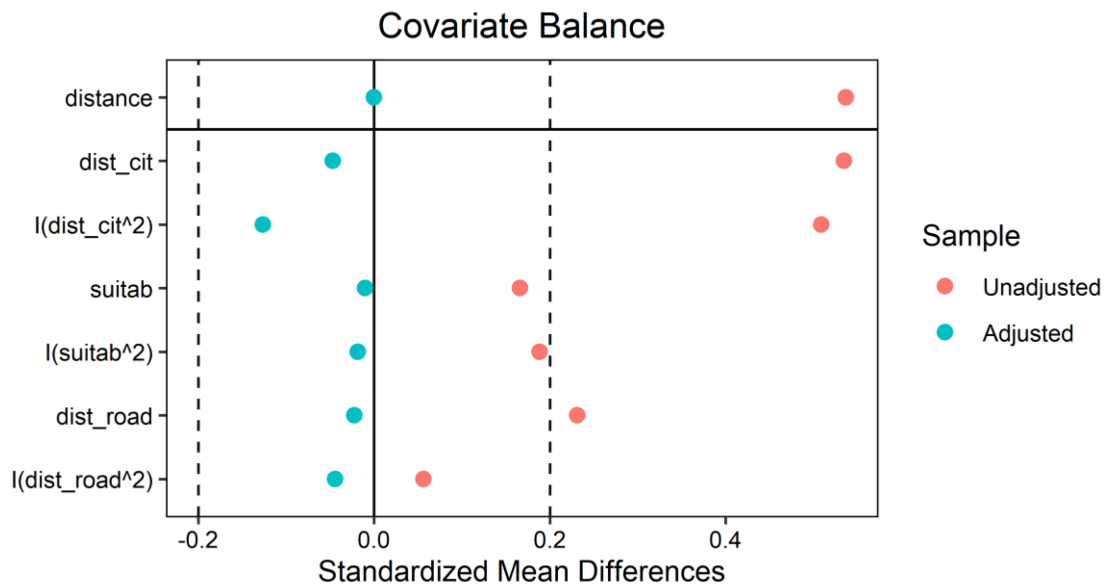

Fig. S7: Covariate balance for the forest loss data using a generalized linear model and selecting only biosphere reserves (treatment) and unprotected areas (control). The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.2 (dashed line) indicate a good covariate balance. dist_cit: distance to cities, suitab: agriculture suitability, dist_road: distance to roads.

Table S7: Summary of balance for matched data for the forest loss model that includes only biosphere reserves (treatment) and unprotected areas (control).

Covariates	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.0817	0.0817	0	1	0	0.0004	0.0001
dist_cit	71.6455	73.5322	-0.0468	0.763	0.0271	0.1126	0.6399
I(dist_cit^2)	6757.6125	7536.059	-0.1267	0.5802	0.0271	0.1126	0.6865
suitab	0.4969	0.4993	-0.01	0.9642	0.0145	0.0421	0.9803
I(suitab^2)	0.3002	0.3044	-0.0187	0.942	0.0145	0.0421	0.9174
dist_road	7.923	8.065	-0.0224	0.8956	0.005	0.0136	0.9201
I(dist_road^2)	102.8591	109.8034	-0.0443	0.7904	0.005	0.0136	0.7893

Table S8: Summary of the number of sampling units (municipalities) allocated to control and treatment in matching analyses conducted on marginalization data

	Protected vs unprotected		**National parks vs biosphere reserves**	
	Control	Treatment	Control	Treatment
All	1463	260	147	86
Matched (ESS)			29.84	66
Matched	260	260	41	66
Unmatched	1203	0	85	0
Discarded	0	0	21	20

Table S9: Summary of the number of sampling units (sampling points) allocated to control and treatment in matching analyses conducted on forest loss data

	National parks vs biosphere reserves		**IUCN category II vs IUCN category Ia**		**IUCN category II vs IUCN category VI**	
	Control	Treatment	Control	Treatment	Control	Treatment
All	26411	3201	6604	3201	19807	3201
Matched (ESS)	952.36	3201	261.28	3113	812.5	3201
Matched	2018	3201	1070	3113	1877	3201
Unmatched	24393	0	5342	0	17234	0
Discarded	0	0	192	88	696	0

Table S10: Summary of the number of sampling units (sampling points) allocated to control and treatment in matching analyses conducted on forest loss data

	Protected vs unprotected areas		**National parks vs unprotected areas**		**Biosphere reserves vs unprotected areas**	
	Control	Treated	Control	Treatment	Control	Treatment
All	574298	29612	574298	3201	574298	26411
Matched (ESS)	25835.9	29612	2984.68	3201	20955.36	26411
Matched	27733	29612	3089	3201	24064	26411
Unmatched	539733	0	571209	0	550234	0
Discarded	6832	0	0	0	0	0

Apéndice Capítulo 3

Apéndice 3A. Resultados suplementarios

Table S1: General information on the Mexican biosphere reserves.

no	Name	Year of creation	IUCN category	Area (ha)	Dominant vegetation
1	Barranca de Metztitlán	2000	I and VI	96,117.76	Shrubland
2	Calakmul	1989	I and VI	1,370,590.73	Tropical forest
3	Chamela-Cuixmala	1993	I and VI	43,176.12	Tropical dry-forest
4	El Triunfo	1990	I and VI	119,276.54	Cloud forest
5	La Sepultura	1995	I and VI	178,639.22	Temperate forest
6	Lacan-Tun	1992	I and VI	61,873.96	Tropical forest
7	Los Petenes	1999	I and VI	282,857.63	Aquatic vegetation
8	Los Tuxtlas	1998	I and VI	154,884.85	Tropical forest
9	Mariposa Monarca	2000	I and VI	56,277.67	Temperate forest
10	Montes Azules	1978	I and VI	331,200.00	Tropical forest
11	Ría Celestún	2000	I and VI	81,482.33	Aquatic vegetation
12	Ría Lagartos	1999	I and VI	60,096.87	Aquatic vegetation
13	Selva El Ocote	1982	I and VI	101,352.24	Tropical forest
14	Sian Ka'an	1986	I and VI	375,062.90	Tropical dry-forest
15	Sierra de Huautla	1999	I and VI	60,697.05	Tropical dry-forest
16	Sierra de Manantlán	1987	I and VI	139,652.98	Tropical dry-forest
17	Sierra del Abra Tanchipa	1994	I and VI	21,483.33	Tropical dry-forest
18	Sierra Gorda	1997	I and VI	383,567.45	Temperate forest
19	Tehuacán-Cuicatlán	1998	I and VI	490,645.28	Tropical dry-forest

*IUCN category I correspond to the core zone of the biosphere reserve, while *IUCN category VI corresponds to their buffer zone.

Table S2: Description of labor composition, in the year 2000, in the municipalities surrounding the studied biosphere reserves, as indicated by the mean proportion of the population working in the municipalities in different sector labors. The agriculture sector includes: crop production and cattle ranching; the industrial sector: manufactures, electricity, and construction; the business & services sector: trade, transportation, financial, administration, cultural, hotels, property, business support, education and health; and the professional: technician, directors, civil officials. The non-farm occupation corresponds to the sum of industrial, professional, and business & services sectors.

Reserve	Agricultural sector	Industrial sector	Business & services sector	Professional sector	Non-farm occupation	no labor information
Barranca de Metztitlán	0.38	0.29	0.31	0.00	0.62	0.02
Calakmul	0.56	0.12	0.30	0.00	0.44	0.02
Chamela-Cuixmala	0.36	0.19	0.42	0.01	0.64	0.02
El Triunfo	0.66	0.09	0.23	0.00	0.34	0.02
La Sepultura	0.49	0.13	0.35	0.01	0.51	0.02
Lacandona	0.76	0.06	0.16	0.00	0.24	0.02
Los Tuxtlas	0.64	0.09	0.25	0.00	0.36	0.02
LPRC	0.33	0.27	0.38	0.00	0.67	0.01
Mariposa Monarca	0.34	0.31	0.32	0.00	0.66	0.03
Ría Lagartos	0.47	0.16	0.35	0.01	0.53	0.02
Selva El Ocote	0.51	0.16	0.31	0.01	0.49	0.02
Sian Ka'an	0.28	0.16	0.54	0.01	0.72	0.02
Sierra de Huautla	0.30	0.27	0.40	0.01	0.70	0.02
Sierra de Manantlán	0.40	0.20	0.37	0.01	0.60	0.02
Sierra del Abra Tanchipa	0.35	0.19	0.43	0.01	0.65	0.02
Sierra Gorda	0.50	0.20	0.26	0.00	0.50	0.04
Tehuacán-Cuicatlán	0.59	0.22	0.17	0.00	0.41	0.01

Table S3: The predictor variables and their relationship with forest loss according to the bibliography.

Predictor variable	Type of predictor	Hypothetical effect on forest loss	Mechanisms	Source
Distance to cities	Economic	Directly proportional	Longer distances result in higher transportation costs for agricultural products, increasing their trade price. To compensate for these lower costs, some producers increase the cultivated area promoting deforestation	Angelsen et al. 2010, Ferraro et al. 2011, Pfaff and Robalino 2012
Marginalization	Political	Directly proportional	Marginalization can be considered as a proxy of poverty. Poverty can increase deforestation in the case that population subsist from agriculture and there exist few other opportunities to fulfill their needs so that they increase the cultivated area to increase revenues which promote forest loss	Gesit and Lambin 2003
Non-farm occupation	Economic	Inversely proportional	Non-farm occupation can be considered a proxy of forest transition. Regions where job opportunities in the industrial and services sectors exert less pressure on the forest since reducing the demand for the forest for cultivation, which is the main direct cause of deforestation in tropical regions	Wunder et al 2003, Angelsen and Kaimowitz 1999,Bluffstone 1995, Hoang et al. 2014, Klooster 2003, Stem 2003
Population density	Demographic	Directly proportional	Population density imposes higher pressure on ecosystems. Higher concentration of people demands higher resources, land surface for agriculture activities and induce technological and institutional change (e.g. higher infrastructure development)	Gesit and Lambin 2002, Aide 2013, Laurance et al. 2002
Population growth	Demographic	Directly proportional	Population growth imposes higher pressure on ecosystems. The increase in population over time increase the demands of resources, the land surface for agriculture activities and induce technological and institutional change (e.g. higher infrastructure development)	Gesit and Lambin 2002, Erlich and Holdren 1971, Wittemyer et al 2008
Rural settlement density	Demographic	Directly proportional	Each settlement exerts pressure on the ecosystem. Higher density of settlements increase deforestation	Mas and Cuevas 2015, Tritsch et al. 2016
Subsidies for agriculture	Political	Directly proportional	Higher economic incentives for agriculture activities may promote deforestation because, in order to increase their revenues, the population transform forest areas into agricultural fields	Klepeis and Vance 2009, Schmook and Vance 2009
Unemployment rate	Economic	Directly proportional	The unemployed population may choose to make use of forest resources to compensate for their shortages, thereby increasing deforestation rates.	Call et al. 2017, Tariq et al. 2014

To compare the effect of the different labor sectors on forest loss rate inside the studied biosphere reserves, we performed simple linear models for each non-farm sector and the non-farm occupation indicator. We found that industrial activities account for the higher effect followed by business & services (Table S4).

Table S4: Results of linear models used to test the relationship between forest loss rate inside the studied biosphere reserves and different labor sector variables (Predictor), expressed as the proportion of people in the municipalities surrounding the studied reserves working in each sector. We order the predictors by Akaike Information Criterion (AIC) ascending values. Note that the industrial sector explains a higher proportion of the variance(R^2) in forest loss rate than business & services and professional activities. Also, note that the proportion of local people to non-farm occupation was the best predictor (higher R^2) of forest loss rate. The significance level of each predictor is indicated by the p.value.

Predictor	AIC	R^2	p.value
Non-farm occupation	-4.10	0.32	0.019
Industrial	-2.46	0.25	0.043
Business & Services	-0.99	0.18	0.091
Professional	1.40	0.05	0.371

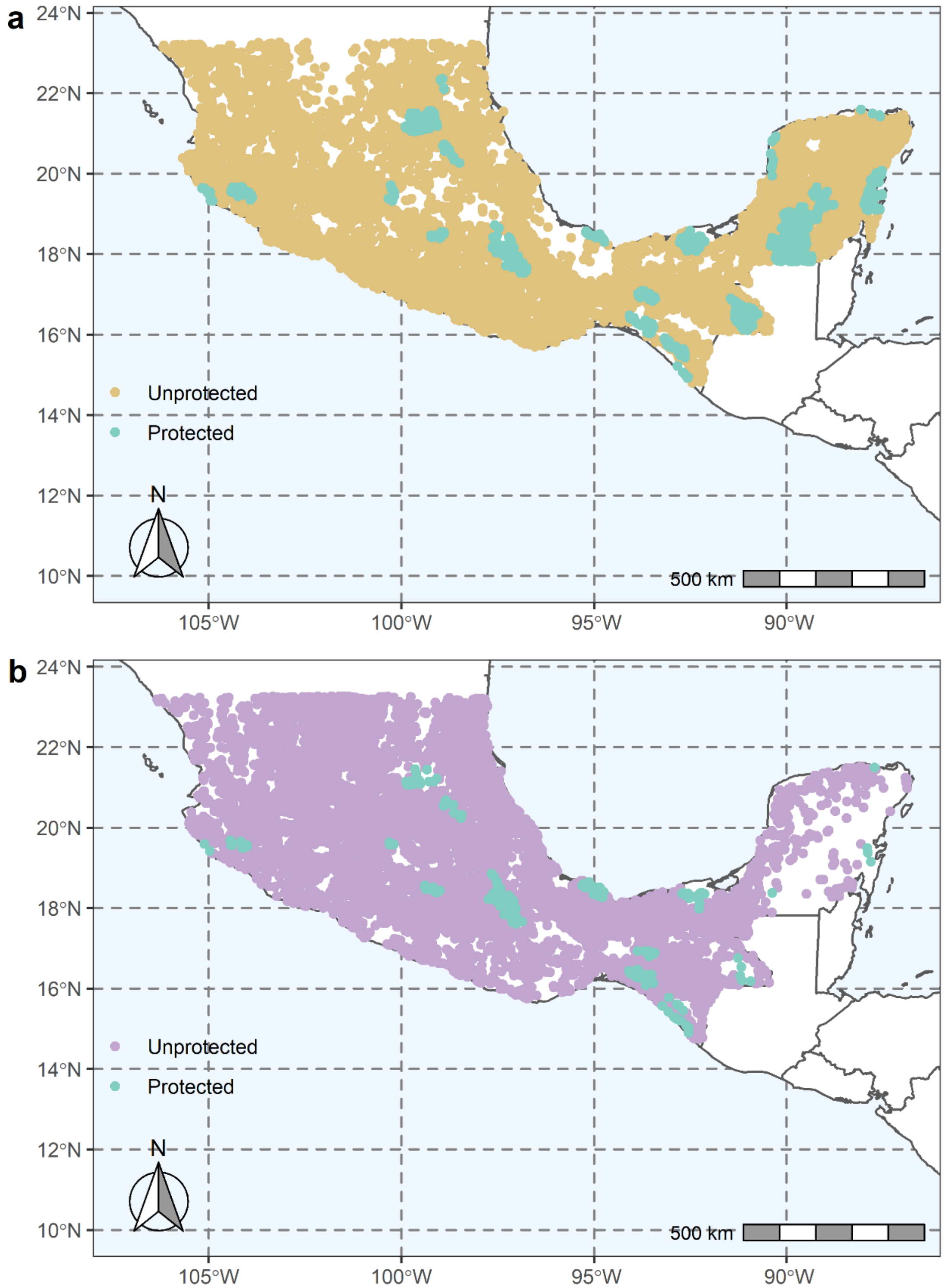

Figure S1: The distribution of the point sample units for the forest loss data (a) and the forest regrowth data (b). Colors indicate the treatment of the sample unit: protected (inside reserves) or unprotected (area not included in any Mexican protected area). Forest loss points (n=423,900) are located in areas covered by forest in 2000 and forest regrowth points (n=331,623) in areas covered by no forest in the same year.

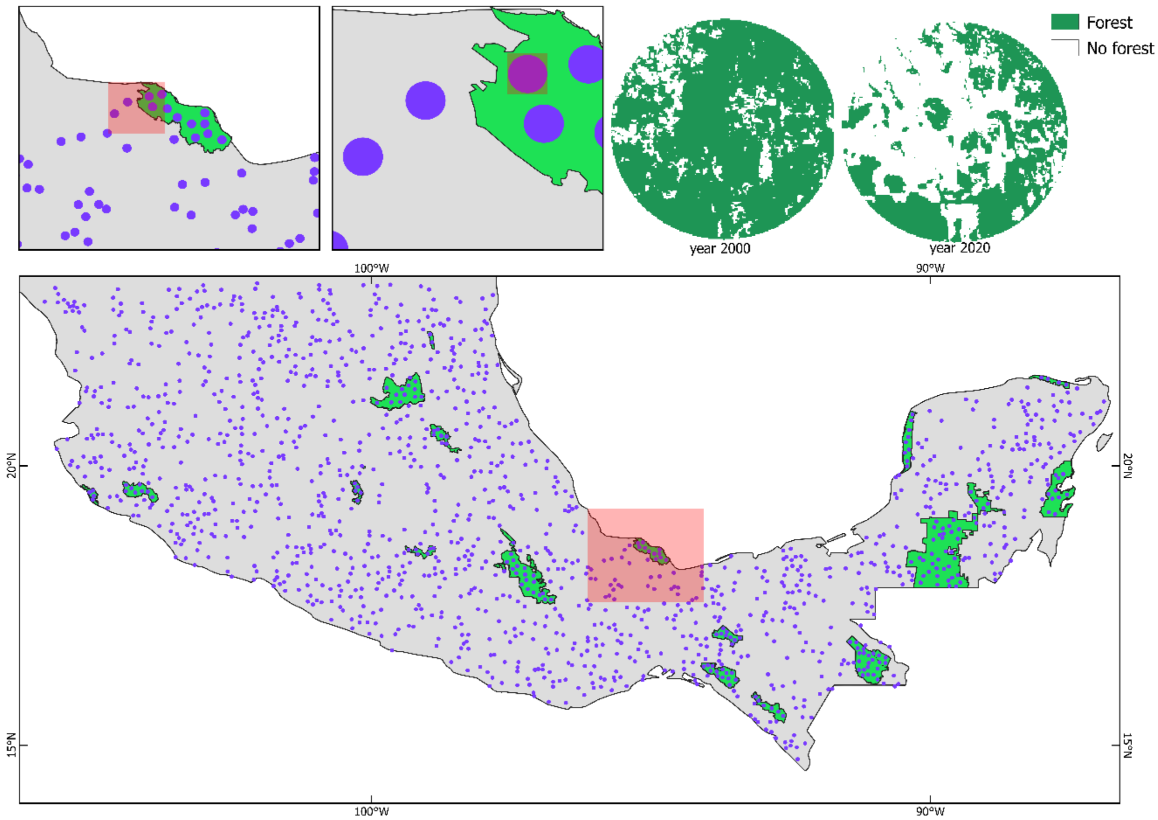

Figure S2: Distribution of the microlandscapes used to evaluate forest fragmentation rate. In the bottom panel, the reserves are represented by green polygons while the purple circles represent microlandscapes (n=1,500). In the upper-right part of the plot, we show an example of a microlandscape and its spatial configuration of forest patches for the years 2000 and 2020. The red square inside the map indicates the location of this microlandscape.

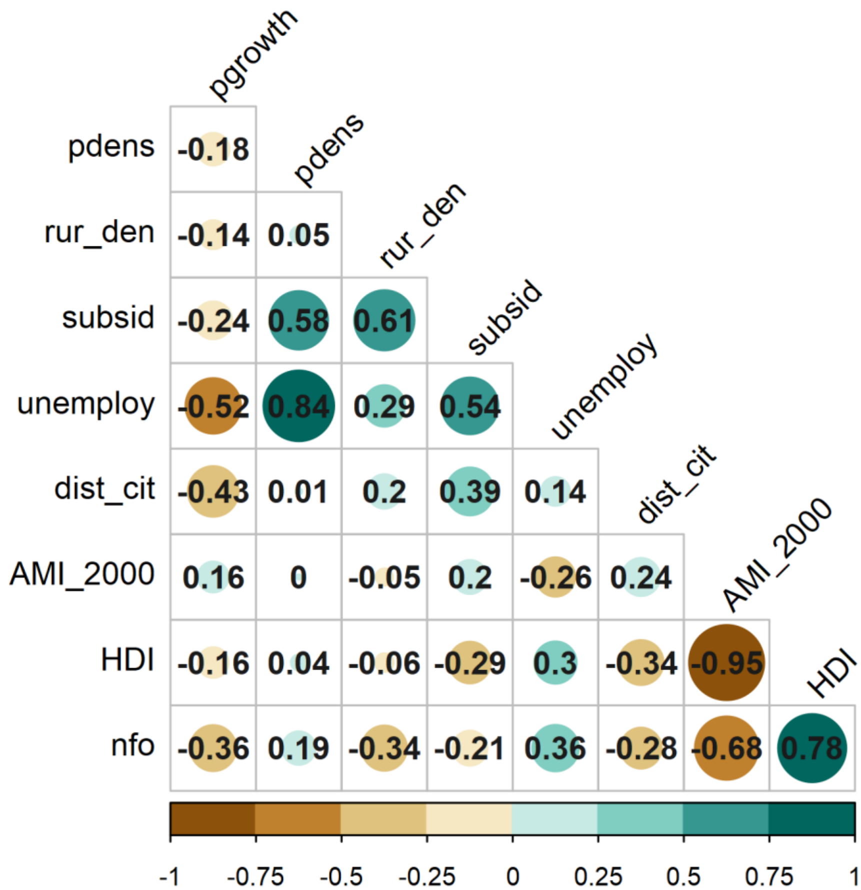

Figure S3: Relationship between socioeconomic variables. The size of the circles represents the magnitude of correlation. Green color denotes positive correlations while orange color negatives. Numbers in the plot represent Pearson correlation coefficients. Pgrwoth: population growth rate, pdens: population density, subsid: governmental subsidies for agriculture, unemploy: unemployment rate, dist_cit: distance to cities, AMI_2000: absolute marginalization index in the year 2000, HDI: human development index, nfo: non-farm occupation.

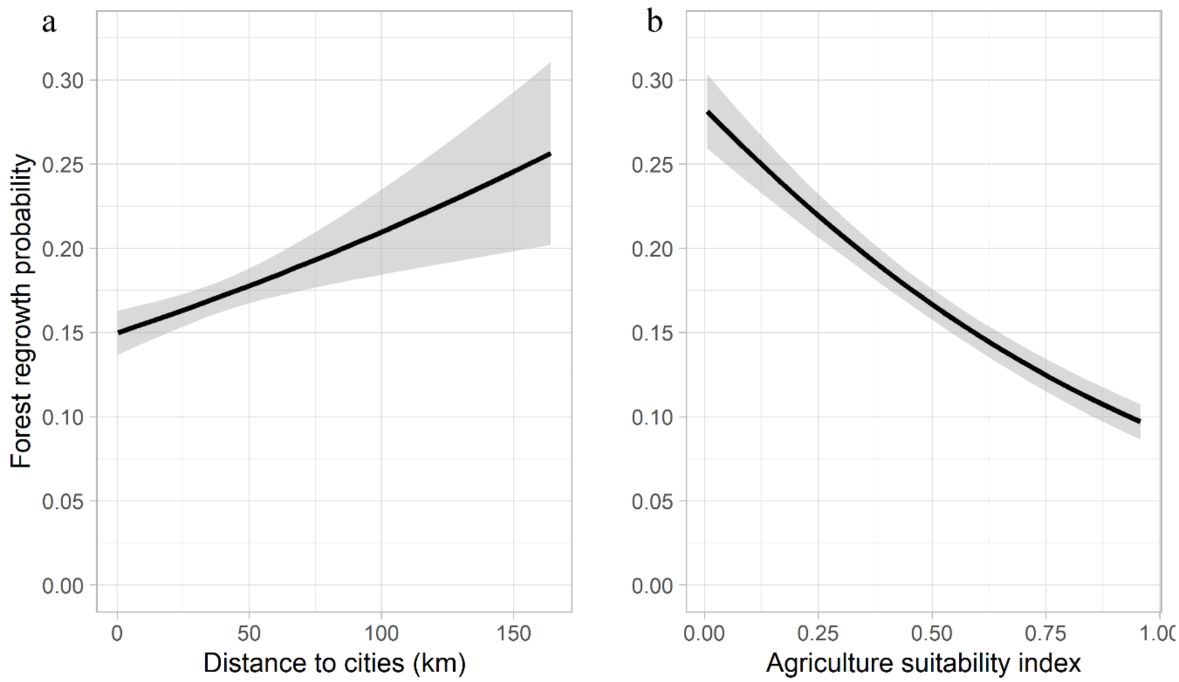

Figure S4: Predicted response of the distance to cities (a) and the agriculture suitability index (b) on forest regrowth rate. Forest regrowth is higher in localities far from cities with poor conditions for agricultural activities (steeper slopes, high elevations, poor soils).

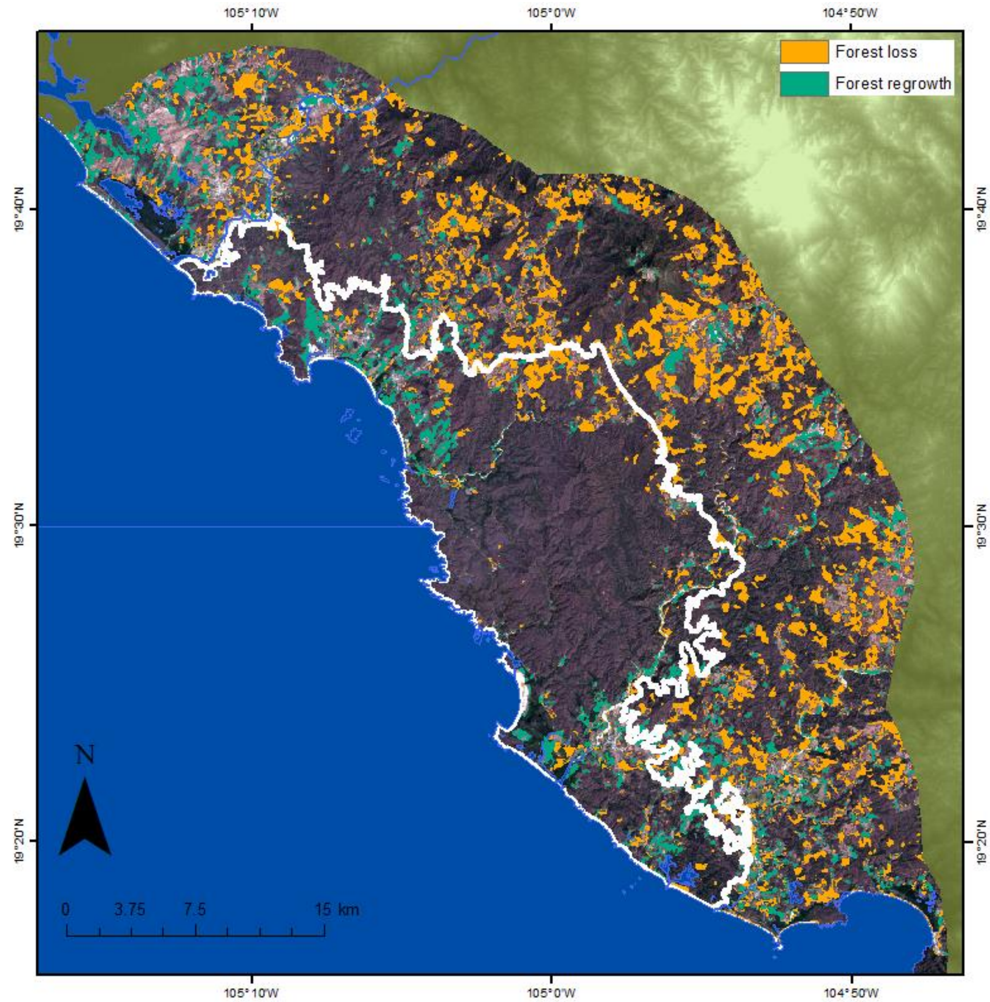

Figure S5: Pattern of forest loss and forest regrowth inside Chamela-Cuixmala Biosphere Reserve (delimited by the white line) and its surrounding (10 km of buffer distance) area during 2000-2020. The formal establishment of the reserve caused the prohibition of some extractive human activities inside its boundaries which promote the regrowth (green areas) of the forest but also increase the agricultural activities outside which is reflected in forest loss (orange areas).

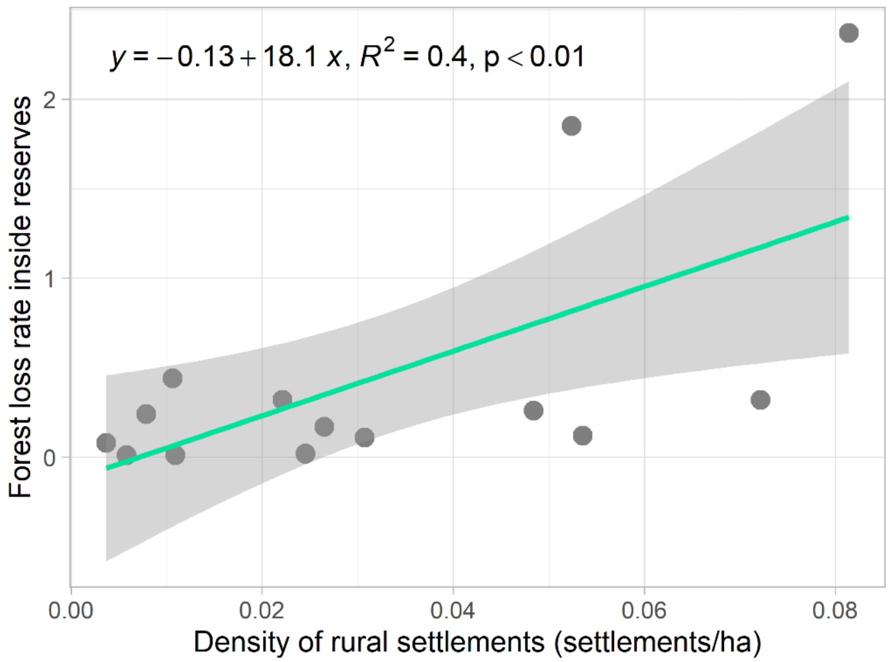

Figure S6: Relationship between the density of rural settlements and forest loss rate (during the period 1985-2000, before the formal establishment of most reserves) inside studied biosphere reserves. Gray points correspond to reserves, the green line corresponds to a linear regression fit.

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Apéndice 3B. Información metodológica suplementaria

Land cover classification methodology

We performed a supervised classification of Landsat images for the years 2000 and 2020. We used the Google Earth Engine platform (Gorelick et al., 2017) because is a cloud online platform that allows processing massive remote sensing data effectively.

We used LANDSAT/LT05/C01/T1_SR image collection for the year 2000 and LANDSAT/LC08/C01/T1_SR for the year 2020. Therefore, our source images consist of surface reflectance satellite images. We applied a cloud mask algorithm for image collections using the median value of data per pixel using images from January to December. We filter images from 2000 to 2002 year to compose the image of 2000 and from 2016 to 2020 to compose the image of 2020.

Since we are working with a large region composed of several biomes, we decide to divide the study region into five subregions delimited by polygons (Fig. S7). The polygons were located in a latitudinal gradient covering relatively homogeneous areas within each polygon based on the authors' knowledge of the study area.

We used a supervised classification for each polygon for the years 2000 and 2020. We used a total of 350 points per polygon to perform the classification form which we used 85% for training and 15% for test. We used a set of geographic products as ancillary data to confirm that both, training and test points correspond to one of three classes: forest, no- forest, and water bodies. This ancillary data was the official land use land cover products of Mexican government edition three and six (INEGI, 2003, 2017), the Madmex land cover product for the year 2018, and the Google Earth desktop app. We do not distinguish between different types of forest (e.g. tropical rainforest, tropical dry-forest, temperate forest, etc.) nor succession stage (old forest or secondary forest) since our main objective was to assess changes in forest cover in general. We used the Random Forest classification algorithm because, in a prospective exercise, this yields the best performance in comparison to Classification and Regression Trees (CART), Support Vector Machines (SVM), and the Hansen cover product (Hansen et al., 2013a).

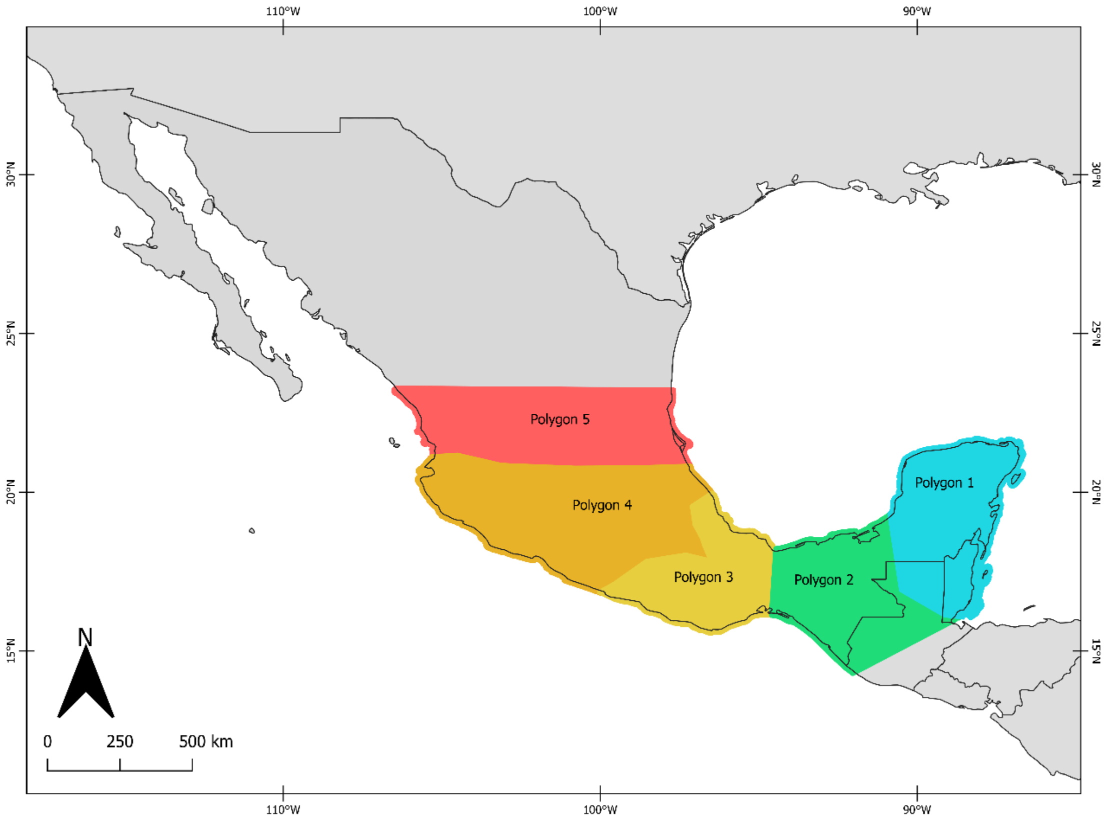

Figure S7: The five polygons in which the study region was divided.

We calculated the classification accuracy for each polygon for years 2000 and 2020 (Table S5) using the caret package in R studio. We also assessed the classification accuracy inside the eighteen studied biosphere reserves (Table S6). To do that, we used 200 random points seeded inside each reserve. We used ancillary data to verify the actual land cover in the location of each point for the year 2020.

Table S5: Classification accuracy indicators for training and test data for the five polygons in which the study region was divided for the years 2000 and 2020.

Polygon	Year	Train overall accuracy	Train kappa	Test overall accuracy	Test kappa
p01	2000	0.9930	0.9770	0.9600	0.8344
p02	2000	0.9900	0.9790	0.9400	0.8800
p03	2000	0.9800	0.9599	0.9400	0.8682
p04	2000	0.9130	0.8264	0.9600	0.9168
p05	2000	0.9300	0.8598	0.9200	0.8379
Average		**0.9612**	**0.9204**	**0.9440**	**0.8675**
p01	2020	0.9967	0.9892	0.9800	0.9117
p02	2020	0.9900	0.9791	0.9000	0.7899
p03	2020	0.9967	0.9933	0.9800	0.9596
p04	2020	0.9700	0.9397	0.9200	0.8390
p05	2020	0.9091	0.8197	0.9400	0.8796
Average		**0.9725**	**0.9442**	**0.9440**	**0.8760**

Table S6: Classification accuracy indicators for 18 biosphere reserves units. The classification corresponds to the year 2020.

No	Reserve	Overall accuracy	Kappa
1	Barranca de Metztitlán	0.8800	0.7200
2	Calakmul	0.9750	0.7869
3	Chamela-Cuixmala	0.9100	0.7162
4	El Triunfo	0.9450	0.8084
5	La Encrucijada	0.7990	0.5986
6	La Sepultura	0.8250	0.5750
7	Lacandona complex	0.9300	0.7122
8	Los Tuxtlas	0.9450	0.8684
9	LPRC complex	0.9343	0.7487
10	Mariposa Monarca	0.9450	0.8078
11	Ría Lagartos	0.9594	0.9114
12	Selva El Ocote	0.9250	0.7727
13	Sian Ka'an	0.9444	0.7263
14	Sierra de Huautla	0.9700	0.8961
15	Sierra de Manantlán	0.9200	0.6396
16	Sierra del Abra Tanchipa	0.9350	0.4118
17	Sierra Gorda	0.8900	0.5908
18	Tehuacán-Cuicatlán	0.8000	0.5035
	Average	**0.9129**	**0.7108**

Covariates calculation

We used the distance to cities, distance to roads, and agriculture suitability index as covariates for matching analysis. We calculated the distance to cities, defined as localities with a population of 15,000 or higher, through a geographic information system (GIS). To do that, we used vectorial data of Mexican localities for the year 2000 (CONABIO, 2002). We generated a raster dataset of 1 km of cell size that contains the information.

To calculate the distance to roads we used vectorial data of the national roads network (INEGI, 2018). We assessed the distance to roads through the GIS and generated a raster dataset of 1 km of cell size. We calculated the agriculture suitability index using data of climate, soil, and orography. As climate variables, we considered the mean annual temperature, temperature annual range, mean annual precipitation, and precipitation of the driest quarter (Fick & Hijmans, 2017). As soil variables, we used the concentration of Ca, Na, organic carbon, and pH (INEGI, 2013a). And as orographic variables, we use elevation and slope (INEGI, 2013b). We obtained data on the presence or absence of agricultural lands from the Mad-Mex land use/cover classification. Mad-Mex provides information on 17 land cover classes at 30m of resolution. We reclassified these classes in a binary raster of agriculture/non-agriculture.

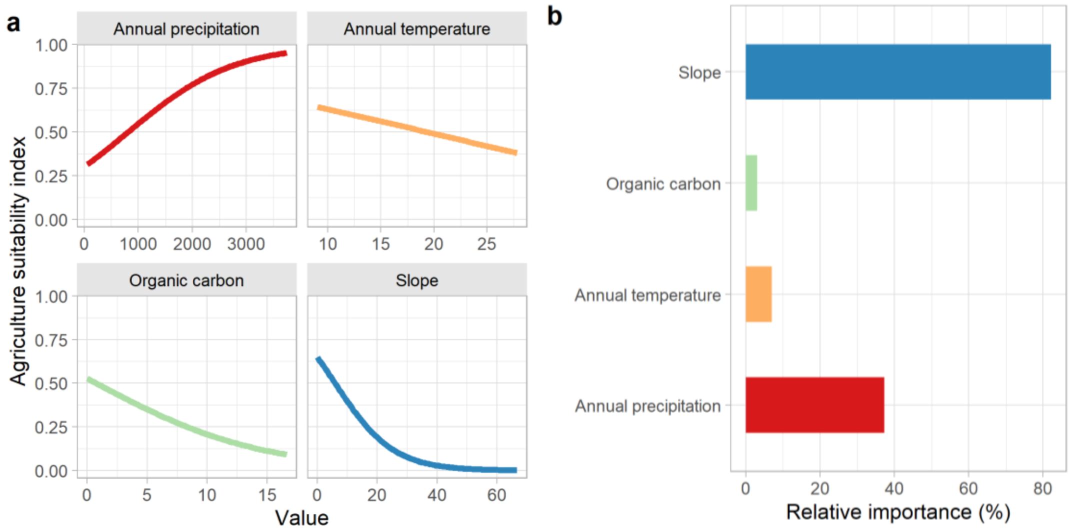

Figure S8: a) Predicted response curves of the explanatory variables as predictors of agriculture suitability index according to binomial model. b) Relative importance of each explanatory variable determined by Pearson correlation permutation.

To calculate the agriculture suitability index, we used binomial generalized linear models. As response variable, we used presences-absence of agriculture and we trained the model with all the set of climates, soil, and, orographic variables previously described. We used 1600 random points isolated at a distance of at least 1 km each other and seeded in the entire Mexican territory to sample the variables. Through this approach, we described the combination of conditions where agriculture activities were developing in the country with a low level of spatial autocorrelation (Moran's I= 0.0012, p=0.79). To select a set of candidate explanatory variables, we tested for their collinearity and their relative importance through correlation and area under the curve metrics. We selected the most parsimonious model based on AIC criteria. Finally, the selected explanatory variables were slope, organic carbon, mean annual temperature, and mean annual precipitation (Fig. S8). We predicted agriculture suitability for the entire Mexican territory. We evaluated the accuracy of the model through the ROC curve (Fig. S9). All the analysis was performed using the *sdm* package of R (Naimi & Araújo, 2016).

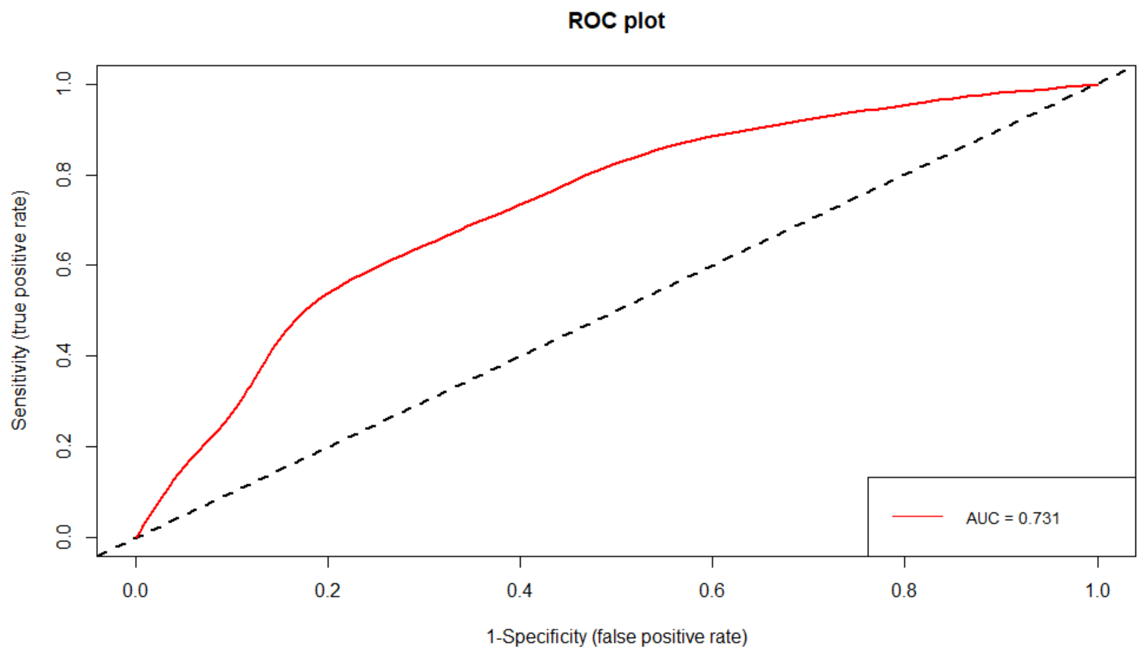

Figure S9: Receiver operating curve (ROC) of the agriculture suitability model. The area under the curve (AUC) value is shown in the bottom right part of the plot. The magnitude of AUC indicates a medium level of the model prediction.

Matching analysis methodology

Two different strategies were used to carry out the matching analysis. On the one hand, for the data on forest loss and regeneration, a total of one million sample points uniformly seeded in the Mesoamerican region were used. Each point had the information about the covariates, whether there was forest loss, forest regrowth, or no cover change in the specific location of the point during the period 2000 2020, and on one of the two conditions: the treatment (i.e. reserves) and the control (unprotected areas). On the other hand, a total of 1500 microlandscapes were randomly seeded in the Mesoamerican region. Each landscape was defined by a circle of 3km of radius which is the maximum distance that perfectly fits inside the smallest reserve (Fig. S2, in Appendix 3A). This approach is similar to the used in previous studies to evaluate fragmentation through matching analysis (Sims 2014). We quantified the number of forest patches in the year 2000 (NP_{2000}) and in the year 2020 (NP_{2020}) for each landscape using the *landscapemetrics* package of R (Hesselbarth et al., 2019), then we calculated the forest fragmentation rate (FFR) as follows:

$$FFR = \left(1 - \left(1 - \frac{NP_{2020} - NP_{2000}}{NP_{2000}}\right)^{1/20}\right) * 100$$

We also calculated the mean value of the covariates for each landscape. Finally, each landscape was assigned to control or treatment according to whether they were located inside a reserve or in unprotected zones. We avoided that the landscapes had an intersection of protected and unprotected zones by moving them a few meters. Data with information from any other type of PAs that are not biosphere reserves were not included in the analysis.

To achieve covariate balance, we used the nearest neighbor covariate matching algorithm with sampling replacement for forest loss and forest regrowth data, and the genetic covariate matching algorithm with sampling replacement and 150 generations for fragmentation data. The propensity score was estimated using binomial error distribution with logit function where the response variable was the treatment (protected or unprotected). After matching, all standardized mean differences for the covariates were below 0.1 which indicates a good balance between control and treatment samples. The matching analysis was performed using the *MatchIt* package (Ho et al., 2011) of R(R Core Team, 2021).

Indicators of underlying drivers

The population density was calculated by dividing the number of people in a municipality in the year 2000 by the municipality area. The population growth rate was calculated as $r = \ln(N_{2020}/N_{2000})/20$, which corresponds to the intrinsic population increase rate per year. Here, N_{2000} is the total population from all municipalities that intersect a given reserve in 2000, N_{2020} is the total population from the same municipalities ten years later (2020). Rural settlement density was estimated from vector data of Mexican rural localities in 2000 (CONABIO, 2002). We calculated Kernel density in QGIS, with a searching radius of 5 km and a cell size of 100 m. We calculated line density in QGIS using a searching radius of 5km and a cell size of 100m and we determine the mean value of road density by each reserve. Government subsidies for agriculture were calculated from PROAGRO program data (SAGARPA, 2018), which lists the amount of money given to each municipality per agriculture cycle (two cycles per year) for the period 2013-2018. We then calculated the total amount of money given for this period per municipality per unit area (Mexican pesos invested per square kilometer). To estimate non-farm occupation, we calculated the proportion of the population in a municipality working in the industrial or services sector. To calculate the distance to major cities, which here was used as a proxy to access markets, we used vector data of localities in the year 2000 with a population >15,000 (CONABIO, 2002). Then we calculate vector point distance in QGIS as raster data at a resolution of 100m. Finally, we calculate the mean distance to major cities for each reserve. Indicators such as marginalization index, human development index, and unemployment index were obtained from the different sources in a format that did not need any calculation.

Since most socioeconomic indicators were gathered from the municipality scale, we calculated the mean value of those that influence directly each reserve. To do that, we defined a minimum intersection area threshold of 10%. Thus, a municipality with an area higher than 10% covered by a reserve was considered to have a direct influence (Table S7). In this form, from the original 297 municipalities that intersect at some level any reserve, we obtained 220 that meet the 10% coverage threshold.

We calculated Pearson correlation coefficients of indicators of underlying drivers of forest spatial changes (forest loss, fragmentation, and regrowth). We discarded variables highly correlated (correlation value higher than 0.5). We also tested for multicollinearity through the variance inflation factor (VIF) and we exclude variables with values higher than 2. Thus, we discarded the subsidies for agriculture, unemployment rate, human development index, and marginalization index (Figure S3, Appendix 3A). In this way, for our analysis, we selected unemployment rate, rural density, population density, population growth, non-farm occupation, and distance to major cities.

Table S7: The number of municipalities included to calculate the mean value of socioeconomic indicators per reserve.

no	Reserves	Municipalities
1	Barranca de Metztitlán	17
2	Calakmul	4
3	Chamela-Cuixmala	1
4	El Triunfo	13
5	La Sepultura	8
6	Lacandona	4
7	Los Tuxtlas	9
8	LPRC	8
9	Mariposa Monarca	15
10	Ría Lagartos	4
11	Selva El Ocote	5
12	Sian Ka'an	2
13	Sierra de Huautla	13
14	Sierra de Manantlán	11
15	Sierra del Abra Tanchipa	3
16	Sierra Gorda	18
17	Tehuacán-Cuicatlán	85

Table S8: Socioeconomic indicators of underlying drivers of forest spatial changes.

No	Indicator	Type of driver	Source	Period	Scale	Resolution	Range	Units	Brief description
1	Marginalization	Economic	CONABIO (2006) with information of CONAPO	2000	Municipality	-	0 to 100	Dimensionless	It is an indicator that assesses the intensity of privations suffered by the population. It takes into account education, households, population distribution, and income. The higher the value the more the deprivation.
2	Human development index (HDI)	Politic	CONABIO (2014) with information of PNUD Mexico	2000	Municipality	-	0 to 1	Dimensionless	It is an indicator of the effectiveness of the state to provide adequate conditions for the proper development of people's lives, taking into account income, health, and education.
3	Unemployment rate	Economic	Derived from CONABIO (2010a) with information of INEGI	2000	Municipality	-	0 to100	Percentage	It is the percentage of the economically active population with no job
4	Non-farm occupation	Economic	Derived from CONABIO (2010a) with information of INEGI	2000	Municipality	-	0 to 1	Proportion	The proportion of the population of a municipality that works in a non-farm sector (i.e. services or industrial).
5	Population growth rate	Demographic	CONABIO (2010b, 2010c) with information from INEGI	2000-2020	Municipality	-	$-\infty$ to ∞	Individuals/year	It is an indicator of the increase or decrease of the population in the period studied.
6	Population density	Demographic	Derived from CONABIO (2010b) with information of INEGI	2000	Municipality	-	0 to ∞	Individuals/km^2	Number of individuals per km^2

7	Government subsidies for agriculture	Politic	Derived from SAGARPA (2018)	2013-2018	Municipality	-	0 to ∞	Pesos/km^2	Amount of money per km^2 invested in the municipality in the period.
8	Distance to cities	Economic	Derived from CONABIO (2002) with information of INEGI	2000	Pixel	100m	0 to ∞	km	Distance to localities with 15,000 people or more. Distance to cities where there is a major market for trade in goods and services
9	Rural settlement density	Demographic	Derived from CONABIO (2002) with information of INEGI	2000	Pixel	100m	0 to ∞	Settlements/km^2	Number of settlements per km^2

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Apéndice 3C. Balance de covariables y análisis de matching

Table S9: Summary of sampling balance between control (unprotected zones) and treatment(reserves) sampling units for forest loss data. To achieve these results, we used a 1:1 nearest neighbor matching with replacement.

	Control	Treated
All	387888	36012
Matched (ESS)	17470.42	36012
Matched	28176	36012
Unmatched	359712	0
Discarded	0	0

* Matched (ESS): Effective Sampling Size

Table S10: Summary of sampling balance between control (unprotected zones) and treatment(reserves) sampling units for forest regrowth data. To achieve these results, we used a 1:1 nearest neighbor matching without replacement.

	Control	Treated
All	325065	6558
Matched	6558	6558
Unmatched	318507	0
Discarded	0	0

Table S11: Summary of sampling balance between control (unprotected zones) and treatment(reserves) sampling units for forest fragmentation data. To achieve these results, we used a 1:1 genetic matching with replacement.

	Control	Treated
All	1258	214
Matched (ESS)	73.39	214
Matched	154	214
Unmatched	1104	0
Discarded	0	0

* Matched (ESS): Effective Sampling Size

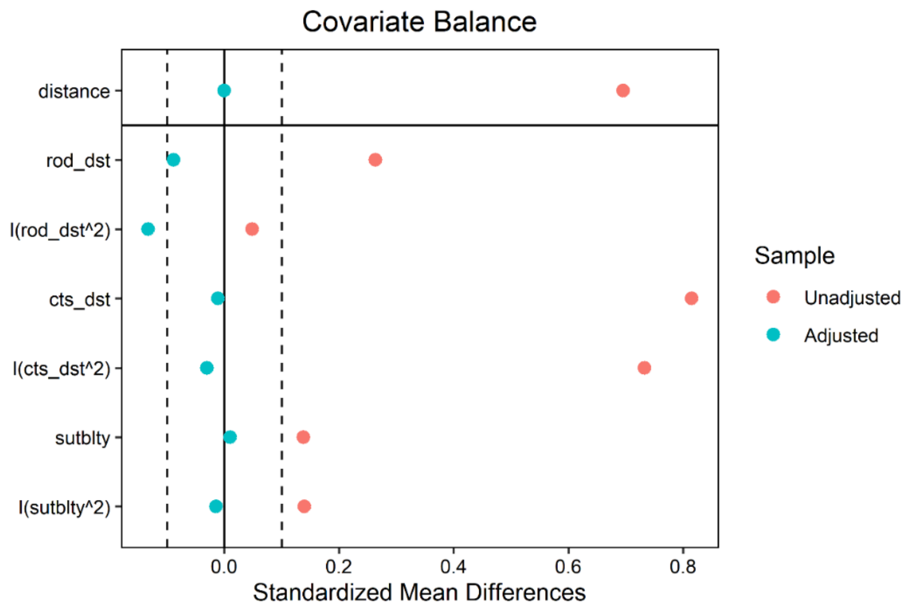

Figure S10: Covariate balance for the forest loss data using the nearest neighbor algorithm. The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.1 (dashed line) indicate a good covariate balance. rod_dst: mean road distance, cts_dst: mean distance to cities, suitability: agriculture suitability.

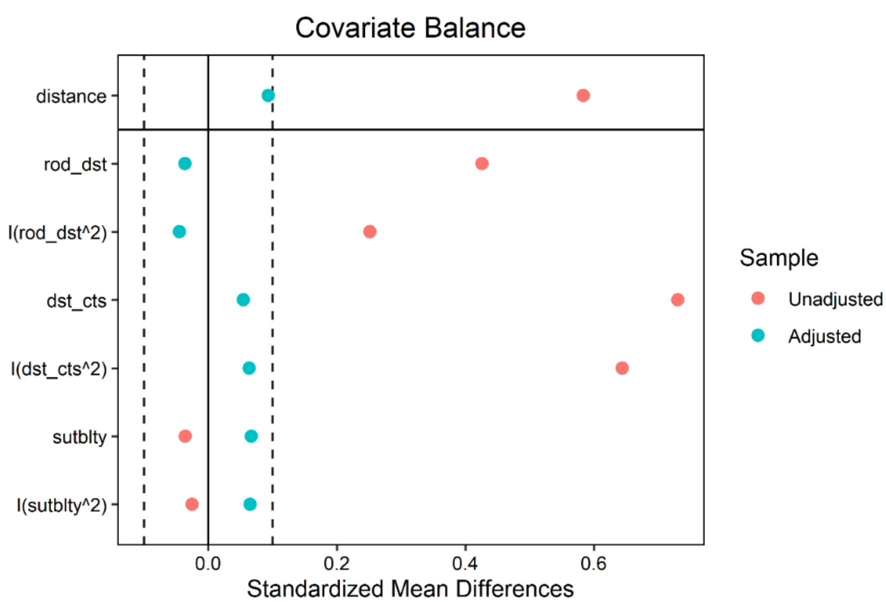

Figure S11: Covariate balance for the forest regrowth data using the nearest neighbor algorithm. The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.1 (dashed line) indicate a good covariate balance. rod_dst: mean road distance, cts_dst: mean distance to cities, suitability: agriculture suitability.

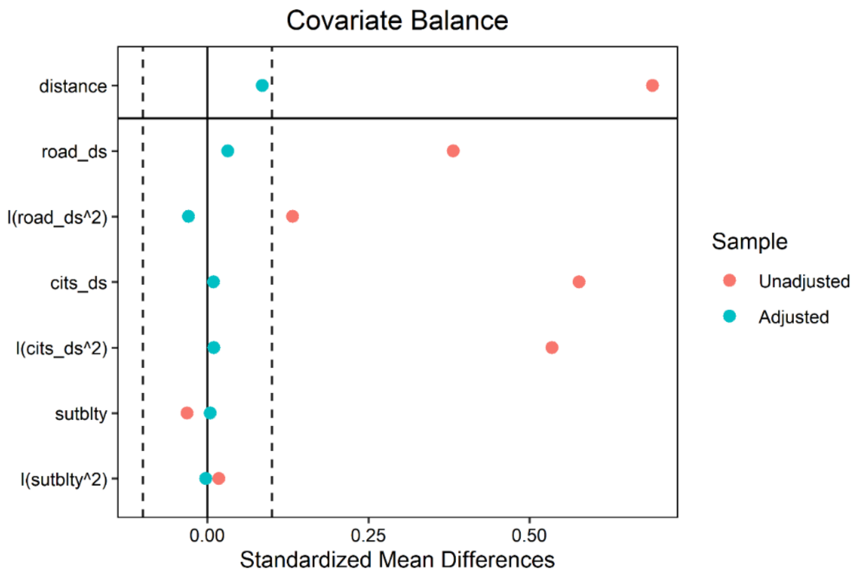

Figure S12: Covariate balance for the forest fragmentation data using the nearest neighbor algorithm. The panel shows the standardized mean difference for each covariate previous (unadjusted) and after (adjusted) the matching analysis. Mean differences less than 0.1 (dashed line) indicate a good covariate balance. rod_dst: mean road distance, cts_dst: mean distance to cities, suitablty: agriculture suitability.

Table S12: Summary of balance for matched data for the forest loss data. All standard mean differences were below 0.1

Covariate	Mean Treated	Mean Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
Distance	0.0325	0.0325	0	1	0	0.0003	0
road.dist	6.4336	6.5826	-0.0246	1.1095	0.0188	0.0977	0.7097
road.dist^2	78.2068	76.5115	0.012	0.818	0.0188	0.0977	0.6088
cities.dist	34.8597	35.646	-0.0376	1.0425	0.0241	0.0698	1.0294
cities.dist ^2	1653.0833	1690.6608	-0.0186	1.0266	0.0241	0.0698	0.8765
suitability	0.4821	0.4932	-0.0455	1.0789	0.0324	0.0668	0.9549
suitability ^2	0.2921	0.2986	-0.0267	1.0653	0.0324	0.0668	0.9887

Table S13: Summary of balance for matched data for the forest regrowth data. All standard mean differences were below 0.15

Covariate	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
Distance	0.2307	0.2307	0	1	0	0.0007	0.0001
road.dist	7.9584	8.5226	-0.0884	0.7829	0.0139	0.0408	1.0126
road.dist^2	104.0745	124.6694	-0.1326	0.5079	0.0139	0.0408	0.9495
cities.dist	70.0068	70.4093	-0.011	0.9242	0.0098	0.0318	0.3506
cities.dist ^2	6248.7251	6415.77	-0.0302	0.8317	0.0098	0.0318	0.3088
suitability	0.5008	0.4987	0.0101	0.8952	0.0189	0.0443	1.0883
suitability ^2	0.2937	0.2967	-0.0144	0.8168	0.0189	0.0443	1.0719

Table S14: Summary of balance for matched data for the forest fragmentation data. All standard mean differences were below 0.1

Covariate	Means Treated	Means Control	Std. Mean Diff.	Var. Ratio	eCDF Mean	eCDF Max	Std. Pair Dist.
distance	0.2804	0.2609	0.0852	1.1189	0.018	0.0701	0.161
road.dist	8.0225	7.8248	0.0318	0.8281	0.03	0.0841	0.6797
road.dist^2	102.7291	107.1441	-0.0289	0.7053	0.03	0.0841	0.7008
cities.dist	57.5375	57.1893	0.0095	1.0006	0.0088	0.0561	0.1157
cities.dist ^2	4657.1201	4604.216	0.0101	0.9943	0.0088	0.0561	0.1452
suitability	0.5059	0.5049	0.0046	0.957	0.0189	0.0841	0.1071
suitability ^2	0.2957	0.2962	-0.0023	0.8913	0.0189	0.0841	0.1198

Apéndice 3D. Marco conceptual

In figure S13 we show the conceptual framework that guides our study. As discussed by (Lambin et al., 2003), land cover changes are caused by several interacting factors, however, in a general way, we can distinguish direct and indirect causes of change.

Direct causes, also known as proximate drivers of change, are activities that modify directly forest cover. Thus, infrastructure extension, agricultural expansion, and wood extraction are classified as proximate drivers of forest change (Geist & Lambin, 2002). In turn, the action of these activities depends on other factors that operate at higher scales and in a more diffuse way (Lambin & Meyfroidt, 2011), the so-called underlying drivers of change. Underlying drivers correspond to demographic, economic, technological, political, and cultural factors that modulate proximate drivers and therefore, modify indirectly forest cover (Geist & Lambin, 2001). For example, demographic factors, such as population growth, might promote deforestation by increasing the demand for land for infrastructure-

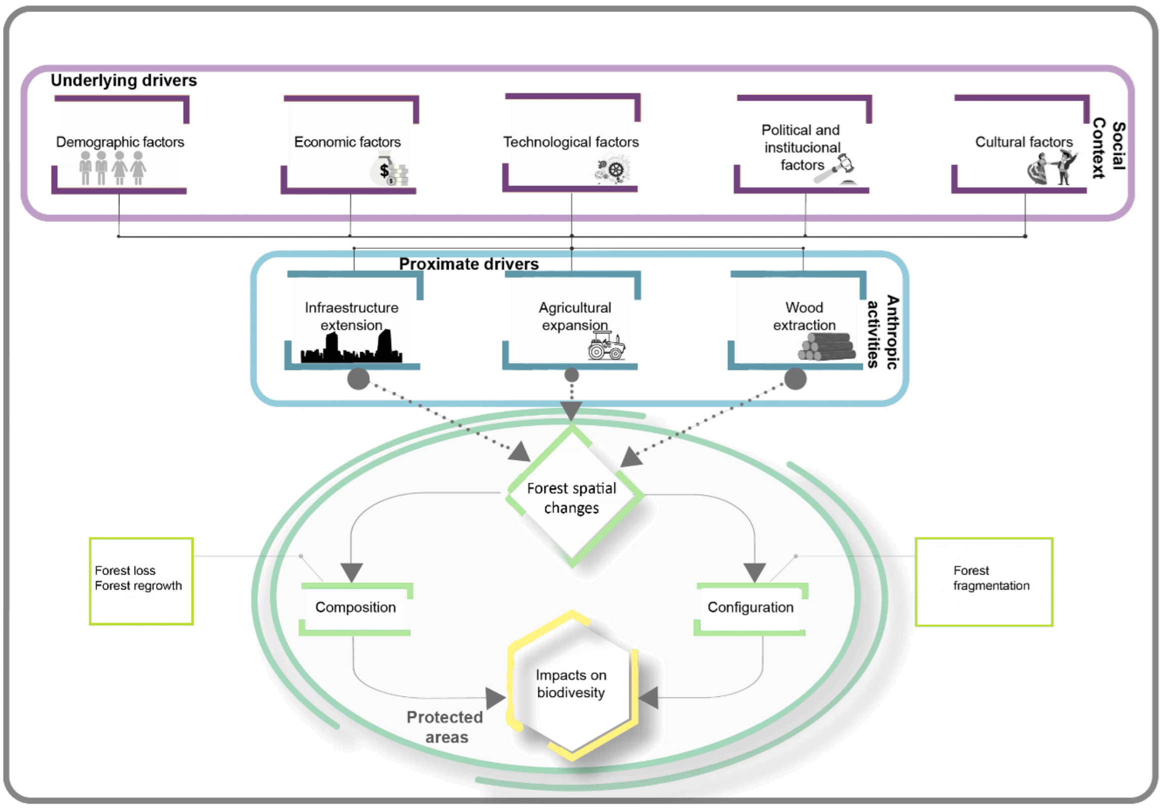

Figure S13: The relationship between underlying and proximate drivers of forest spatial changes and their potential implications to biodiversity. Underlying drivers act on proximate drivers by fostering or inhibiting the execution of the different human activities. In turn, proximate drivers modify directly forest cover which causes the forest spatial changes: forest loss, forest regrowth, and forest fragmentation. Protected areas, whose main purpose is to preserve biomes and biodiversity, are influenced by the underlying and proximate drivers and therefore, their conservation outcomes are affected by the socio-economic context where they exist. Our conceptual framework is based on Geist and Lambin (2001).

development or by demand a higher amount of resources for feeding. On the other hand, international trade promotes deforestation in the Amazonian for producing commodities. Furthermore, a shift in the economic opportunities associated with forest transition dynamics has consequences in lowering the deforestation rates (Rudel et al., 2005). Policies that promote the development of agricultural activities might promote deforestation (Klepeis & Vance, 2009). Examples of studies that document the effect of underlying drivers on forest loss (Aide et al., 2013), forest regrowth (Borda-Niño et al., 2020), and fragmentation (Liu et al., 2016) can be found in the literature.

The conservation status of protected areas is affected by intrinsic factors such as limited budget or failure in management implementation (Leverington et al., 2010); and also by extrinsic factors such as the socioeconomic context where they are located (Laurance et al., 2012b). For example, protected areas in countries with higher human welfare have experienced a lower increase in anthropogenic pressure (Geldmann et al., 2019). Besides that, higher deforestation pressure has been recorded in protected areas under the influence of high population growth (Wittemyer et al., 2008) high population density (Figueroa et al., 2009) and high influence of road network (Barber et al., 2014). In contrast, protected areas with the development of ecotourism activities (Vuohelainen et al., 2012) and with a high presence of indigenous people (Figueroa et al., 2009) have experienced low deforestation pressures. If we consider that almost the third part of global protected areas are influenced by high anthropogenic pressure (Jones et al., 2018), then, understanding the causes that drive forest spatial changes is paramount to provide management alternatives that help to improve the conservation status of protected areas.

It is important to remark that despite that biophysical variables can modify forest cover (e.g. storms, droughts, landslides, etc.) we do not include these variables in our conceptual framework because we consider that the effect of anthropogenic variables has a higher extension, frequency, and impact (Curtis et al., 2018).

Our study explores the effect of some demographic and economical underlying drivers on forest spatial changes (i.e. forest loss, forest regrowth, and forest fragmentation) in nineteen biosphere reserves located in the Mesoamerican territory. Particularly, we evaluate the effect of population growth rate, population density, rural settlements density, distance to cities (a proxy of the access to markets), and non-farm occupation.

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Apéndice Capítulo 4

Apéndice 4A. Información suplementaria sobre las reservas estudiadas

Table S1. General information of the studied biosphere reserves in Mexico. Two pairs of reserves were merged into reserves complex because they share limits. These were Los Petenes -Ria Celestún and Montes Azules - Lacan-Tun (see map below).

No	Name	Mexican state	Area (ha)	Year of creation	Dominant vegetation cover
1	Calakmul	Campeche	1,371,766	1989	Tropical rainforest, tropical dry forest, cattle pasture
2	Chamela-Cuixmala	Jalisco	63,950	1993	Tropical dry forest, agriculture, cattle pasture
3	El Cielo	Tamaulipas	144,530	1985	Temperate mountain forest, tropical dry forest, cloud tropical forest
4	El Triunfo	Chiapas	119,177	1990	Cloud forest, tropical rainforest, agriculture
5	La Encrucijada	Chiapas	144,868	1995	Pasture, aquatic vegetation, agriculture
6	La Sepultura	Chiapas	167,310	1995	Temperate forest, cloud mountain forest, cattle pasture
7	Lacan-Tun	Chiapas	61,838	1992	Tropical rainforest, aquatic vegetation, agriculture
8	Los Petenes	Campeche	282,858	1999	Aquatic vegetation, bare, tropical rainforest
9	Los Tuxtlas	Veracruz	155,123	1998	Pasture, tropical rainforest, agriculture
10	Montes Azules	Chiapas	331,200	1978	Tropical rainforest, cattle pasture, cloud forest
11	Ria Celestún	Campeche-Yucatán	81,482	2000	Aquatic vegetation, bare, tropical rainforest
12	Selva El Ocote	Chiapas	101,288	2000	Tropical rainforest, cattle pasture, temperate mountain forest
13	Sian Ka'an	Quintana Roo	528,148	1986	Aquatic vegetation, tropical rainforest
14	Sierra de Huautla	Morelos	59,031	1999	Tropical dry forest, temperate mountain forest, agriculture
15	Sierra de Manantlán	Jalisco y Colima	139,577	1988	Temperate mountain forest, tropical dry forest, cloud tropical forest
16	Tehuacán-Cuicatlán	Puebla y Oaxaca	490,187	1998	Tropical dry forest, desert vegetation, temperate mountain forest

Apéndice 4B. **Cuestionario sobre los cambios en biodiversidad**

Objective: Document the changes in biological groups relevant to the ecology of tropical forests in Mexican Biosphere Reserves.

This survey is part of a doctoral project that analyzes the conservation situation of some relevant Mesoamerican biosphere reserves, its relationship with landscape variables, and the possible socioeconomic promoters involved in the observed changes.

Our study follows the methodology of Laurance and collaborators (2012, Nature 489) with some modifications. This tool aims to take advantage of the expertise of scientists who have worked with different biological groups over several years to identify the trends of change in biodiversity in the reserves.

People who provide detailed information for this questionnaire will be offered co-authorship in at least one publication that arises from this work.

One of the main assumptions in our study is that the data collected is reliable, therefore, we kindly require the participants to refrain from answering questions in which there is not, at least, moderate indirect or direct knowledge on the subject of any question.

We appreciate your collaboration and are remain at your disposal for any questions or comments.

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The original survey (in Spanish) can be consulted online https://forms.gle/GBG41VM8WuDDsyVS7

Researcher information

Name:

1. Academic degree:
2. Expertise area:
3. Biological sex:
4. Work address:
5. Email:

6. Phone number:
7. Beginning year of work at the reserve:
8. Please qualify your knowledge about the reserve from 1 to 10 (being 1 the lowest and 10 highest):
9. How long have you worked at the reserve?

Information about the reserve

10. Name of the reserve:
11. Name of the research station in the reserve:
12. Is your study area inside the reserve?
13. If you answered **yes** to the previous question please skip to question number 18, otherwise follow to question 14
14. Please, describe the specific locality of your study area and its position regarding the reserve (i.e., northward, southward, eastward, westward, etc.)
15. What is the nearest distance between your study area and the limit of the reserve?
16. What is the size (in ha) of your study area?
17. Please, provide the elevation range of your study area.
18. If we draw a radius of 3km from your study area, would it mostly fall within the reserve?
19. In your opinion, how good are the protective actions in the reserve?
20. Has the protection status of the reserve changed during your study period? how?
21. If you have an additional comment please write it here:

Part I: Changes in plant and animal communities

We kindly reminded the participants that they are free not to answer any questions for which they believe to have insufficient knowledge on the subject.

Please provide details regarding the change in each section as well as its possible promoters.

In the last 3 decades, have any of the following groups changed in their total abundance or species richness values (native only) in their study area associated with the corresponding reserve?

Participants could choose among seven options about the degree of changes in species abundance and richness:

Descriptors	Range of change observed
Strong decrease	More than 50% reduction
High decrease	Between 26-50% reduction
Small decrease	Between 6-25% reduction
No change	Changes between -5 a +5%
Small increase	Between 6-25% increase
High increase	Between 26-50% increase
Strong increase	More than 50% increase

Additionally, the participants could choose among three levels of certainty to rate their knowledge in each question: speculation, good certainty, and high certainty.

MAMMALS

1. Top predators (e.g., jaguars, cougars, etc.)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

2. Large, non-predatory mammals (e.g., tapir, deer, etc.)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

3. Primates

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

4. Omnivorous/opportunistic mammals (e.g., peccaries, opossums, coatis, all> 1kg body weight)

Abundance:	Richness:	Certainty level:
Details on changes:		

Potential drivers:

5. Rodents (<1kg)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

6. Bats

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

BIRDS

7. Understory birds (e.g. insectivorous birds dwellers of forest interior)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

8. Large game birds (cracids, guans, curassows, pheasants)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

9. Large frugivorous birds (e.g., toucans)

Abundance:	Richness:	Certainty level:
Details on changes:		

Potential drivers:

10. Raptors (e.g., eagles, hawks, falcons, owls)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

11. Small nectarivorous birds (e.g., hummingbirds)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

AMPHIBIANS AND REPTILES

12. Stream-dwelling amphibians

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

13. Terrestrial amphibians

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

14. Lizards and large reptiles

Abundance:	Richness:	Certainty level:
Details on changes:		

Potential drivers:

15. Venomous snakes

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

16. Non-venomous snakes

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

TERRESTRIAL INVERTEBRATES

17. Light-loving butterflies

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

18. Army ants (driver ants)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

19. Leaf-cutter ants

Abundance:	Richness:	Certainty level:
Details on changes:		

Potential drivers:

20. Dung beetles

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

21. Disease-vectoring invertebrates (e.g., mosquitoes, sandflies, ticks)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

22. Other groups

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

PLANTS

23. Large-seeded species (shade-tolerant trees, climax species)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

24. Pioneer species

Abundance:	Richness:	Certainty level:
Details on changes:		

Potential drivers:

25. Lianas/Climbing vines (predominantly light-loving)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

26. Epiphytic plants (e.g., orchids, bromeliads, ferns)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

27. Other groups

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

GENERAL GROUPS

28. Migratory species (e.g., birds and mammals, frugivores or nectarivores)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

29. Ecological specialists (e.g., foraging specialists, species with complex mutualism)

Abundance:	Richness:	Certainty level:
Details on changes:		

Potential drivers:

30. Species dependent on tree cavities (e.g., parrots, possums, bats)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

31. Exotic animals (not native)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

32. Exotic plants (not native)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

33. Human diseases (e.g., dengue, malaria, leishmaniasis, Chagas, Chikungunya)

Abundance:	Richness:	Certainty level:
Details on changes:		
Potential drivers:		

Part II: Additional information

In your opinion, what are the main threats in your study area? Could you identify a possible solution?

34. First biggest threat:

    Possible solution:

35. Second biggest threat:

    Possible solution:

36. Third biggest threat:

    Possible solution:

37. Could you please recommend someone else to participate in this survey?

38. Are you interested in continuing to be involved in this study and participate as a co-author of the publication

    Comment:

39. Please provide significant references on the described changes documented in the survey

Authors names	Publication title	Journal	Year

Your collaboration is appreciated, we remain at your disposal for any comment.

Apéndice 4C. Información suplementaria de los métodos

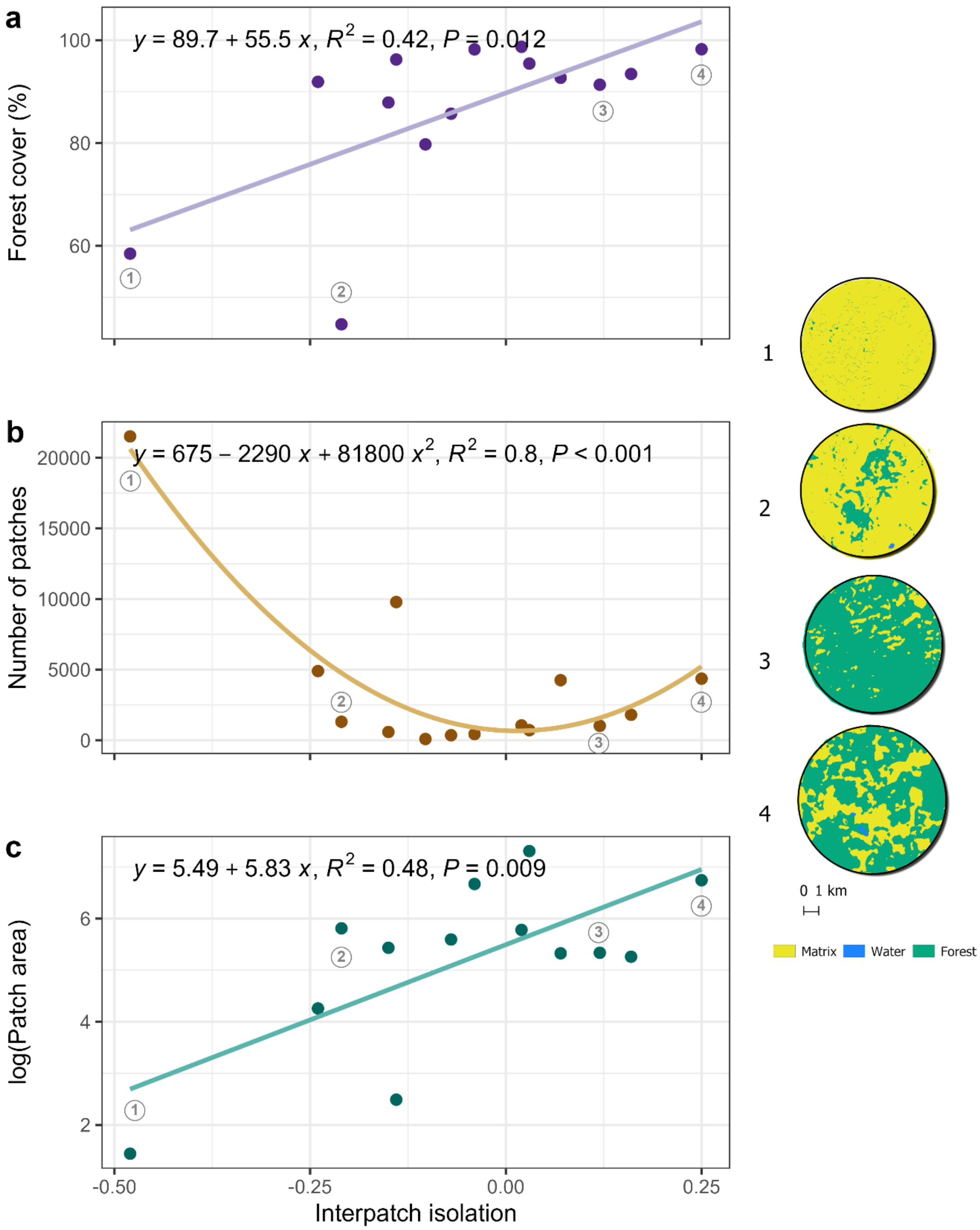

Figure S1. Relationship between interpatch isolation change (1990-2019), and (a) forest cover in 2019, (b) number of forest patches in 2019, and (c) forest patch area in 2019. Each point represents a biosphere reserve. We also show with different numbers the location of four reserves within each scatter plot, and the structure of their surrounding (2.5-km radius) landscapes: (1) La Encrucijada, (2) Los Tuxtlas, (3) Selva El Ocote, and (4) Calakmul. Note that positive isolation changes indicate that interpatch isolation was higher in 2019 than 1990, so there was an increase in isolation over time (negative values indicate the opposite).

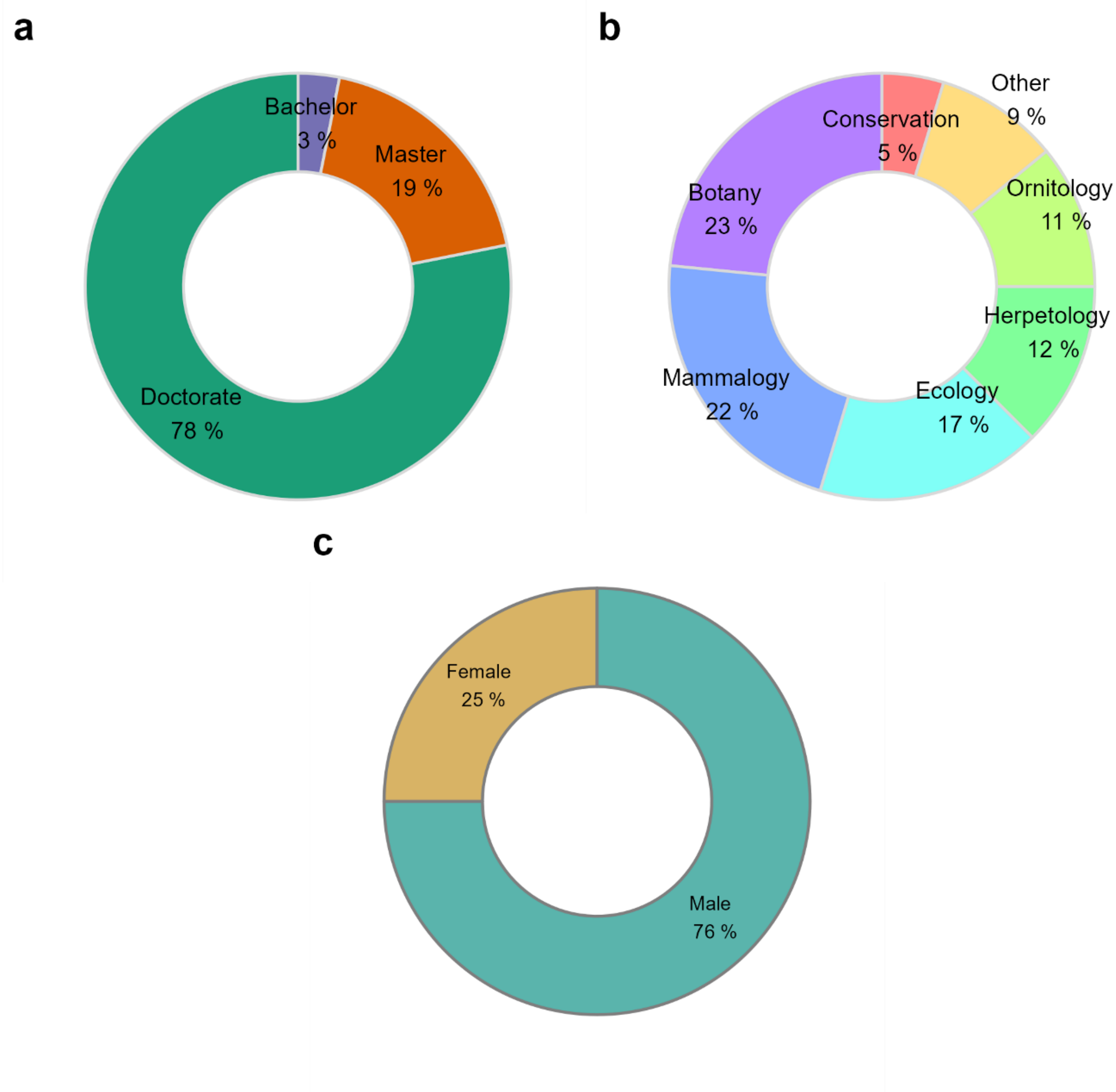

Figure S2. Percentage of experts that answered the survey by their academic degree (a), discipline (b), and biological sex (c). A total of 64 experts provided information on biodiversity changes in the studied reserves. Most of them have doctoral studies. Regarding disciplines, the most common in our group of experts were botany (23%), mammalogy (22%), and ecology (17%). The percentage of experts that self-identify as male was 75% and 25% as female.

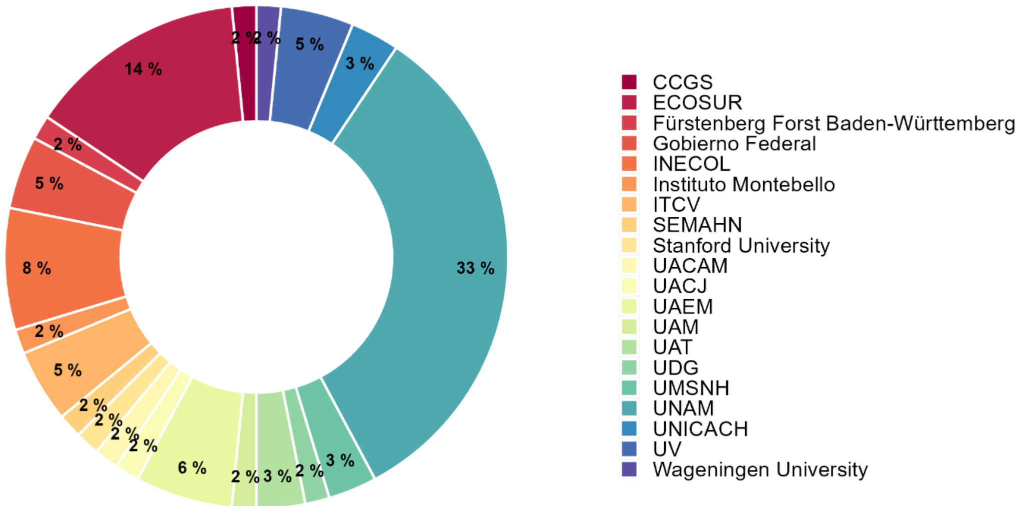

Figure S3. The experts that answered our survey come from a total of 20 different institutions, but most of them come from UNAM and ECOSUR. Academic Institutions = CCGS: Centro de Cambio Global y Sustentabilidad, ECOSUR: El Colegio de la Frontera Sur, INECOL: Instituto Nacional de Ecología, ITCV: Instituto Tecnológico de Ciudad Victoria, SEMAHN: Secretaría de Medio Ambiente e Historia Natural, UACAM: Universidad Autónoma de Campeche, UACJ: Universidad Autónoma de Ciudad Juárez, UAEM: Universidad Autónoma del Estado de Morelos, UAM: Universidad Autónoma Metropolitana, UAT: Universidad Autónoma de Tamaulipas, UDG: Universidad de Guadalajara, UMSNH: Universidad Michoacana de San Nicolás de Hidalgo, UNAM: Universidad Nacional Autónoma de México, UNICACH: Universidad de Ciencias y Artes de Chiapas, UV: Universidad de Veracruzana.

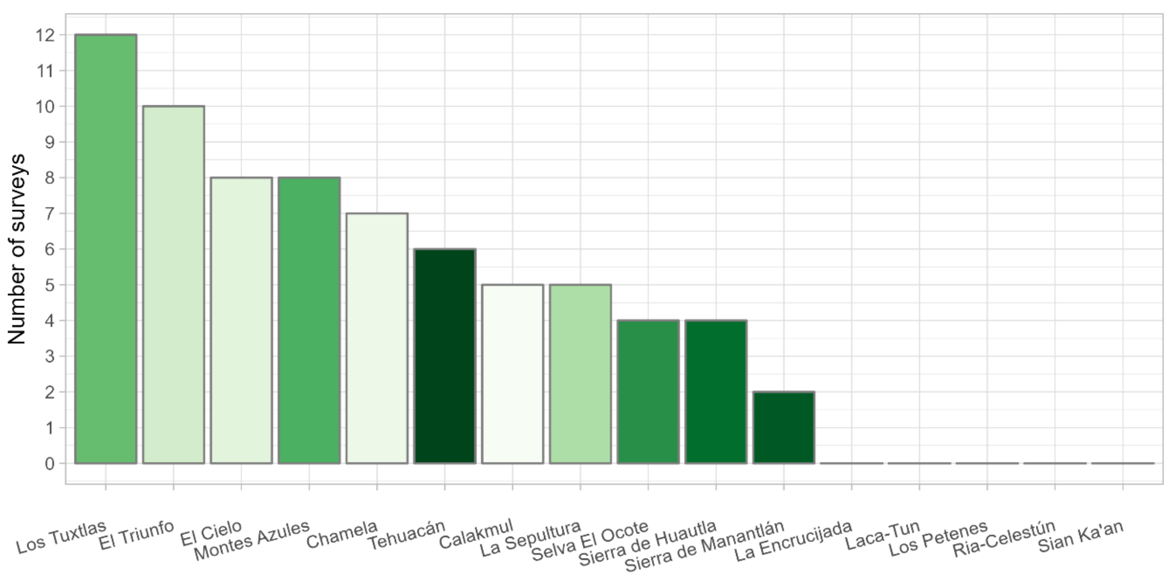

Figure S4. The number of surveys applied in each studied reserve. Due to the low number of surveys (n = 2), the Sierra de Manantlán biosphere reserve was not included in the analysis in the main text.

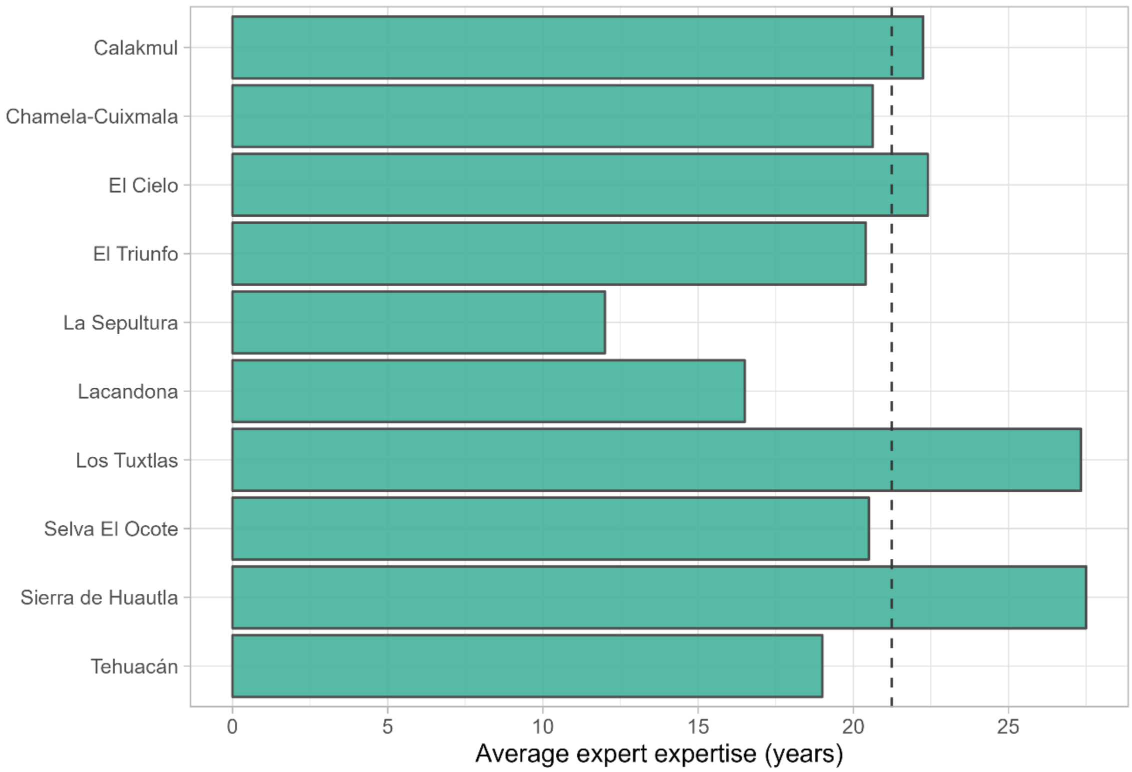

Figure S5. Average expertise (in years) of the group of experts documenting biodiversity changes in the studied reserves. The dark dashed line represents the mean value considering all reserves. On average, the group of experts have a work expertise of 21 year.

Apéndice 4D. **Indicadores usados en las ecuaciones estructurales**

Indicators of underlying drivers

All variables were calculated at the municipality level by averaging the value of each indicator for all municipalities with at least 10% of its territory located inside the reserves. The population growth rate was calculated as $r = \ln(N_{2020}/N_{1990})/20$, which corresponds to the intrinsic population increase rate per year. N_{2000} corresponds to the number of people inhabiting the adjacent municipalities of each reserve in the year 1990, while N_{2020} pertains to the same but in the year 2020. The population density was calculated by dividing N_{1990} by the area of the municipality. Rural settlement density was estimated in QGIS from vector data of Mexican rural localities in 2000 (localities with less than 5,000 people; CONABIO, 2002) using a Kernel density algorithm with a searching radius of 5 km and a cell size of 100 m. In the case of rural settlements density, we could not use data of 1990 because there were important gaps of information, thus, we used data from the year 2000.

Table S1. Number of municipalities used to calculate the indicators of underlying drivers of change per reserve

Reserve	Number of municipalities
Calakmul	4
Chamela	1
El Cielo	2
El Triunfo	8
La Encrucijada	6
La Sepultura	6
Lacandona	2
Los Tuxtlas	9
LPRC	5
Selva El Ocote	3
Sian Ka'an	1
Sierra de Huautla	4
Sierra de Manantlán	7
Tehuacán	44

Government subsidies for agriculture were calculated from PROAGRO program data (SAGARPA, 2018), which lists the money provided to each municipality per agriculture cycle (two cycles per year) for the period 2013-2018. We then calculated the total money provided for this period per municipality per unit of area (Mexican pesos invested per square kilometer). To calculate the distance to major cities, which we used as a proxy to access markets, we used vector data of localities in 2000 with a population >15,000 (CONABIO, 2002). Then we calculated vector point distance in QGIS as raster data at a resolution of 100 m. To calculate the unemployment rate, we divided the number of unemployed people in a municipality by the number of adult (>18 years old) residents and then multiplied this result by 100. Non-farm occupation corresponds to the proportion of people in a municipality working in a non-farm sector (i.e., industrial, professional, or services). Additional details can be found in Auliz-Ortiz et al. (2022).

Proximate drivers

To estimate changes in road density we gathered information on the road network in 2019 (INEGI, 2019) and 2008 (Digital Chart of the world, 2008), as data from previous years was unavailable. We applied a line density algorithm in a Geographic Information System (GIS) to calculate a road density value per studied reserve and year. Finally, we calculated the temporal change (2008-2019) in road density.

Land cover classification

We used Landsat images to characterize land use and forest cover temporal changes inside, and outside reserves. Images were gathered from the US Geological Service with a level 2 processing, i.e., with surface reflectance units and an atmospheric correction (Vermote et al., 2016). We selected images recorded in the dry season (December to May) to avoid cloud obstruction, from the years 1990 and 2019. The supervised classification was carried out using vegetation and land use cover data from the Instituto Nacional de Estadística y Geografía (INEGI, 2003, 2017) and ENVI 5.3 selecting the maximum likelihood algorithm. We considered 14 major land cover classes: agriculture (permanent and seasonal), aquatic vegetation, bare, cleared vegetation, cloud forest, desert vegetation, pasture, savanna, temperate forest, tropical dry forest, tropical rainforest, urban areas, and water bodies. For each class, we randomly seeded 150 points inside the polygons of each reserve and adjacent buffer area and randomly split them into two groups. The first group contained 75% of the points and was used to perform the supervised classification, and the second group contained 25% of the points and was used to evaluate the classification's accuracy. We applied a 7 × 7 majority filter to reduce the so-called "salt and pepper" effect and combined the cover classes into the three following major classes which were the subjects of the rest of the analysis: anthropic land cover (agriculture, bare, cleared vegetation, pasture, and urban zones), forest (all types of forest, savanna, and aquatic vegetation), and water bodies. We evaluated the accuracy of this classification through a confusion matrix. Accuracy was reasonably high in all reserves (Table S2-S3). Several satellite images were needed to cover larger reserves. In these cases, we classified images individually and merged them by each reserve.

Table S2. Accuracy value of supervised classification in each reserve for 1990.

Reserves	Overall accuracy	Kappa coefficient
Calakmul	88.8%	0.775
Chamela-Cuixmala	94.2%	0.9114
El Cielo	90.88%	0.8588
El Triunfo	88.0%	0.76
La Encrucijada	89.8%	0.845
La Sepultura	93.4%	0.9006
Lacandona	99.3%	0.989
Petenes-Celestún	82.4%	0.7303
Los Tuxtlas	87.4%	0.8559
Selva El Ocote	94.9%	0.9227
Sian Ka’an	97.8%	0.9667
Sierra de Huautla	93.3%	0.8989
Sierra de Manantlán	98.5%	0.9775
Tehuacán-Cuicatlán	90.6%	0.8567
Mean	92.2%	0.8763
Standard deviation	0.05	0.0798

Table S3. Accuracy value of supervised classification per reserve for 2019.

Reserves	Overall accuracy	Kappa coefficient
Calakmul	90.5%	0.8559
Chamela-Cuixmala	94.9%	0.9225
El Cielo	88.6%	0.8263
El Triunfo	99.0%	0.9804
La Encrucijada	84.7%	0.768
La Sepultura	88.4%	0.8243
Lacandona	97.1%	0.9558
Petenes-Celestún	87.0%	0.8007
Los Tuxtlas	92.0%	0.8785
Selva El Ocote	94.2%	0.9116
Sian Ka’an	90.3%	0.8065
Sierra de Huautla	91.9%	0.8768
Sierra de Manantlán	92.6%	0.8768
Tehuacán-Cuicatlán	95.7%	0.9347
Mean	91.6%	0.868
Standard deviation	0.04	0.0631

Biodiversity changes indicators

For each reserve, we calculated the mean change in abundance and richness of disturbance-sensitive and disturbance-tolerant guilds, which included 5 guilds each (see Table 4.1 in the main text). Then, to obtain a single value per disturbance group, we performed a principal component analysis with these data and retained the two first components (Fig. S1, Appendix 4D). The first component was mostly associated with changes in disturbance-sensitive guilds and accounted for 69.5% of data variance. The second component was mainly related to changes in disturbance-tolerant guilds and accounted for 24.8% of variance.

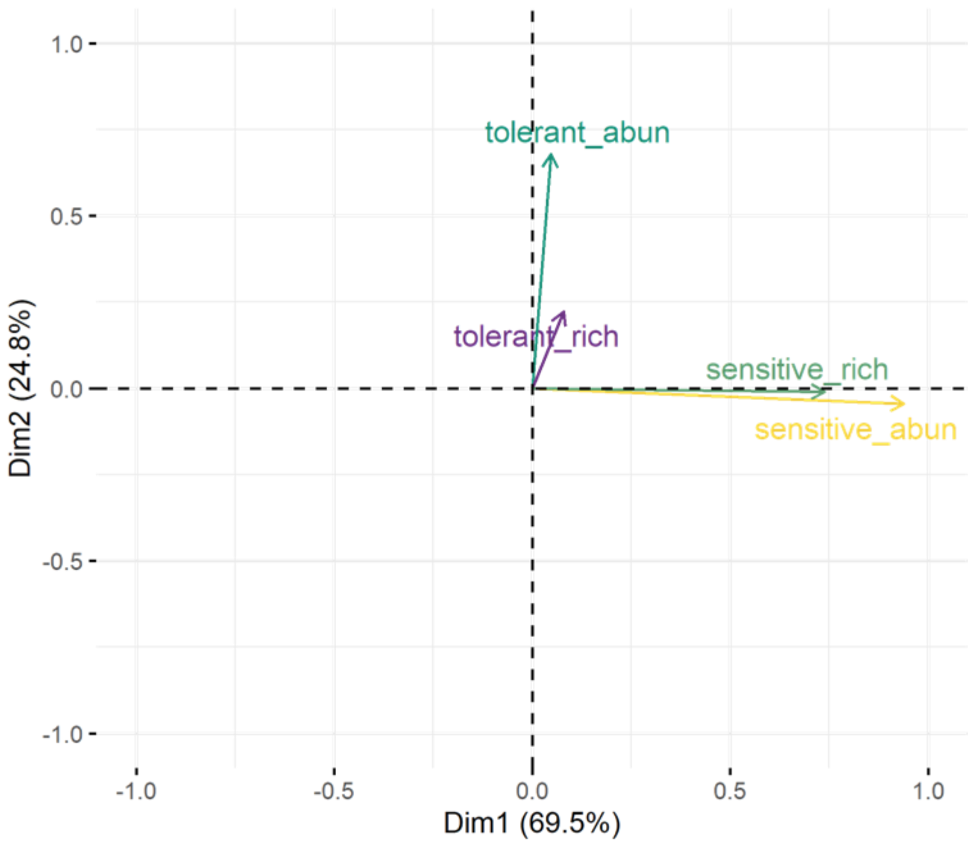

Figure S1. Ordination of disturbance-sensitive (n = 5) and disturbance-tolerant guilds (n = 5) using a principal component analysis. The first two components explained 94.3% of the variance in the temporal change of richness and abundance of these two groups of guilds.

Literature cited

Auliz-Ortiz DM, Arroyo-Rodríguez V, Mendoza E, Martínez-Ramos M. 2022. Conservation of forest cover in Mesoamerican biosphere reserves is associated with the increase of local non-farm occupation. Perspectives in Ecology and Conservation 20:286–293. Available from https://linkinghub.elsevier.com/retrieve/pii/S2530064422000268.

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Digital Chart of the world. 2008. Red de carreteras. Available from http://www.conabio.gob.mx/informacion/gis/ (accessed March 5, 2020).

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INEGI. 2017. Conjunto de datos vectoriales de la carta de Uso del suelo y vegetación serie VI. Conjunto Nacional. Available from https://www.inegi.org.mx/app/biblioteca/ficha.html?upc=889463598459 (accessed September 1, 2017).

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SAGARPA. 2018. Listado de Beneficiarios PROAGRO Productivo. Available from http://www.sagarpa.mx/agricultura/Programas/proagro/Beneficiarios/Paginas/Beneficiarios.aspx (accessed April 5, 2019).

Vermote E, Justice C, Claverie M, Franch B. 2016. Preliminary analysis of the performance of the Landsat 8/OLI land surface reflectance product. Remote Sensing of Environment 185:46–56. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.rse.2016.04.008.

Apéndice 4E. Estadísticas de selección de modelo

To avoid multicollinearity problems, we excluded predictor variables with a variance inflation factor (VIF) > 2 (Fig. S1, Appendix E). We used a multimodel inference approach with generalized linear models (Burnham & Anderson, 2002) to select the most parsimonious models in each hypothetical causal chain within our conceptual framework (Fig. S2 in this appendix, and Fig. 1 in the main text). We then used the *glmulti* package for R (Calcagno & Mazancourt, 2010) to calculate the Akaike Information Criterion corrected for small samples (AICc) of each possible model after combining the retained (non-collinear) variables and the null model. Because of the small sample size, we limited to three the maximum number of terms in the models to avoid overfitting. Therefore, for each response variable, we constructed all models that represent all possible model combinations with one, two, and three predictors and their pairwise interactions. We then ranked the models according to their AICc, from the lowest AICc (the best-supported model) to the highest AICc (the least-supported model), separately for each response variable (Tables S1-S7, Appendix E).

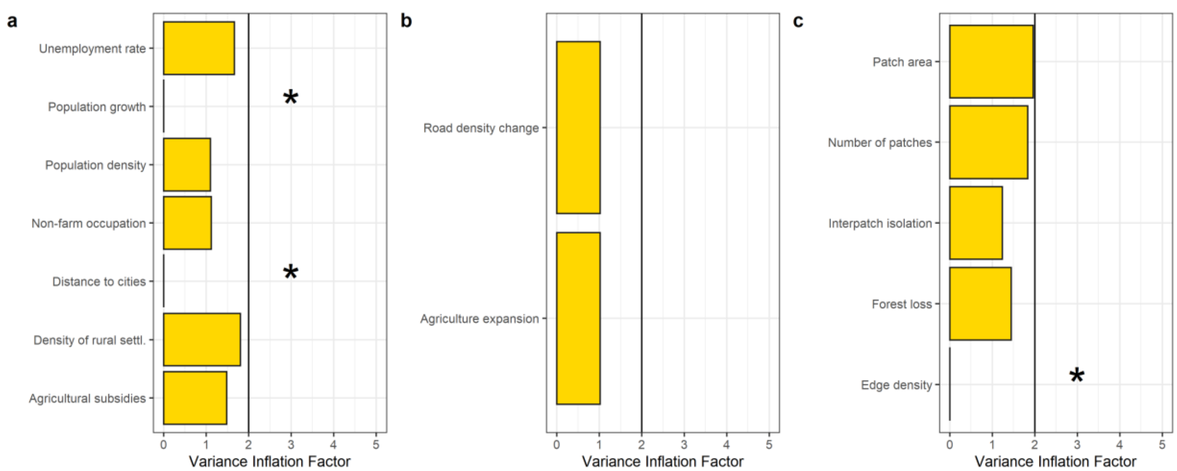

Figure S1. Variance inflation factor (VIF) of predictor variables involved in each theoretical path in the structural equation model (SEM), a) underlying drivers, b) proximate drivers, and c) forest spatial changes. To avoid multicollinearity problems in the models, we excluded predictors with VIF > 2 (indicated with asterisks).

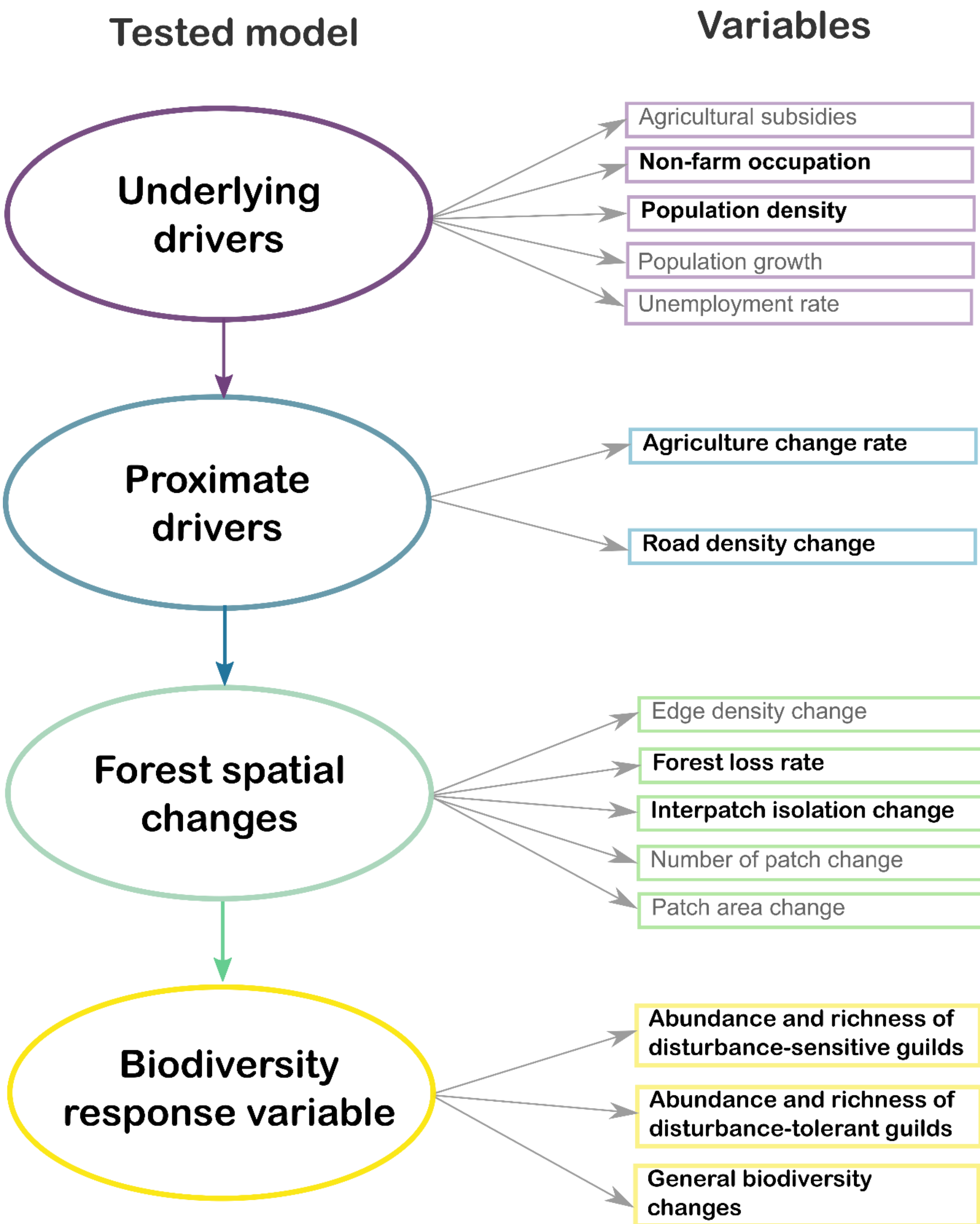

Figure S2. Operational variables assessed within our multivariate model. The structural equation models (SEM) included only the variables highlighted in bold, which were selected using a multimodel inference approach. Note that we conduced one SEM model per biodiversity variable (in yellow).

Table S1. Generalized linear models of the effect of forest spatial changes on general biodiversity (first PCA scores of mean richness and abundance of all the guilds without grouping by their response to disturbance). The models are ordered from the lowest to highest AICc value.

Model	AICc	weights
biodiversity ~ 1 + rloss_for + isolation	18.37	0.59
biodiversity ~ 1 + rloss_for	19.70	0.30
biodiversity ~ 1 + rloss_for + r_pa	24.68	0.03
biodiversity ~ 1	24.83	0.02
biodiversity ~ 1 + r_np + rloss_for	25.15	0.02
biodiversity ~ 1 + rloss_for + r_pa + isolation	25.58	0.02
biodiversity ~ 1 + r_np	26.48	0.01
biodiversity ~ 1 + isolation	27.69	0.01
biodiversity ~ 1 + r_pa	28.68	0.00
biodiversity ~ 1 + r_np + rloss_for + isolation	32.59	0.00
biodiversity ~ 1 + r_pa + isolation	33.00	0.00
biodiversity ~ 1 + r_np + r_pa	33.48	0.00
biodiversity ~ 1 + r_np + isolation	33.50	0.00
biodiversity ~ 1 + r_np + rloss_for + r_pa	35.94	0.00
biodiversity ~ 1 + r_np + r_pa + isolation	45.18	0.00

* rloss_for= forest loss rate, isolation = interpatch isolation, r_pa= patch area change, r_np=change in the number of patches

Table S2. Linear models of the effect of forest spatial changes on changes in disturbance-sensitive diversity (first PCA scores of the mean richness and abundance of the guilds classified by their response to disturbance). The models are ordered from the best to the worst performance according to the AICc value and their relative importance (weights).

Model	AICc	weights
sensitive~ 1 + rloss_for	2.91	0.80
sensitive~ 1	8.25	0.06
sensitive~ 1 + rloss_for + isolation	8.80	0.04
sensitive~ 1 + rloss_for + r_pa	8.92	0.04
sensitive~ 1 + r_np + rloss_for	10.11	0.02
sensitive~ 1 + r_pa	10.93	0.01
sensitive~ 1 + r_np	11.83	0.01
sensitive~ 1 + isolation	12.31	0.01
sensitive~ 1 + r_np + isolation	15.86	0.00
sensitive~ 1 + r_pa + isolation	16.77	0.00
sensitive~ 1 + r_np + r_pa	17.04	0.00
sensitive~ 1 + rloss_for + r_pa + isolation	17.80	0.00
sensitive~ 1 + r_np + rloss_for + isolation	19.17	0.00
sensitive~ 1 + r_np + rloss_for + r_pa	21.84	0.00
sensitive~ 1 + r_np + r_pa + isolation	26.37	0.00

* rloss_for= forest loss rate, isolation = interpatch isolation, r_pa= patch area change, r_np=change in the number of patches

Table S3. Linear models of the effect of forest spatial changes on changes in disturbance tolerant species diversity (second PCA scores of the mean richness and abundance of the guilds classified by their response to disturbance). The models are ordered from the best to the worst performance according to the AICc value and their relative importance (weights).

Model	AICc	weights
tolerant ~ 1 + isolation	5.50	0.43
tolerant ~ 1	6.39	0.28
tolerant ~ 1 + rloss_for + isolation	8.07	0.12
tolerant ~ 1 + rloss_for	9.40	0.06
tolerant ~ 1 + r_pa	10.62	0.03
tolerant ~ 1 + r_pa + isolation	11.20	0.03
tolerant ~ 1 + r_np + rloss_for + isolation	11.52	0.02
tolerant ~ 1 + r_np	12.34	0.01
tolerant ~ 1 + r_np + isolation	12.75	0.01
tolerant ~ 1 + rloss_for + r_pa	15.27	0.00
tolerant ~ 1 + rloss_for + r_pa + isolation	16.93	0.00
tolerant ~ 1 + r_np + rloss_for	18.32	0.00
tolerant ~ 1 + r_np + r_pa	19.45	0.00
tolerant ~ 1 + r_np + r_pa + isolation	23.22	0.00
tolerant ~ 1 + r_np + rloss_for + r_pa	30.27	0.00

* rloss_for= forest loss rate, isolantion= interpatch isolation distance change, r_pa= patch area change, r_np=change in the number of patches

Table S4. Linear models of the effect of proximate drivers on the forest loss rate. The models are ordered from the best to the worst performance according to the AICc value and their relative importance (weights).

Model	AICc	weights
rloss_for ~ 1 + agriculture	16.62	0.62
rloss_for ~ 1 + road_change + agriculture	17.74	0.36
rloss_for ~ 1	24.11	0.01
rloss_for ~ 1 + road_change	26.89	0.00

* rloss_for= forest loss rate, agriculture= agriculture expansion, road_change= road density change

Table S5. Linear models of the effect of proximate drivers on interpatch isolation distance change. The models are ordered from the best to the worst performance according to the AICc value and their relative importance (weights).

Model	AICc	weights
isolation ~ 1 + road_change	28.50	0.78
isolation ~ 1 + road_change + agriculture	32.49	0.11
isolation ~ 1	32.66	0.10
isolation ~ 1 + agriculture	35.72	0.02

* isolation= interpatch isolation distance change, agriculture= agriculture expansion, road_change= road density change

Table S6. Linear models of the effect of underlying drivers on agriculture expansion. The models are ordered from the best to the worst performance according to the AICc value and their relative importance (weights).

Model	AIC	weights
agriculture ~ 1 + nfo	60.75	0.42
agriculture ~ 1 + sub_agri + nfo	63.23	0.12
agriculture ~ 1	63.73	0.09
agriculture ~ 1 + nfo + pdens	64.37	0.07
agriculture ~ 1 + unemploy + nfo	64.38	0.07
agriculture ~ 1 + dens_rur + nfo	64.67	0.06
agriculture ~ 1 + unemploy + sub_agri + nfo	66.19	0.03
agriculture ~ 1 + sub_agri	66.72	0.02
agriculture ~ 1 + pdens	66.82	0.02
agriculture ~ 1 + unemploy	66.87	0.02
agriculture ~ 1 + sub_agri + nfo + pdens	66.89	0.02
agriculture ~ 1 + dens_rur	66.99	0.02
agriculture ~ 1 + sub_agri + dens_rur + nfo	68.20	0.01
agriculture ~ 1 + unemploy + dens_rur + nfo	68.56	0.01
agriculture ~ 1 + dens_rur + nfo + pdens	68.78	0.01
agriculture ~ 1 + unemploy + nfo + pdens	69.35	0.01
agriculture ~ 1 + unemploy + sub_agri	70.24	0.00
agriculture ~ 1 + sub_agri + pdens	70.29	0.00
agriculture ~ 1 + sub_agri + dens_rur	70.41	0.00
agriculture ~ 1 + unemploy + pdens	70.85	0.00
agriculture ~ 1 + dens_rur + pdens	70.86	0.00
agriculture ~ 1 + unemploy + dens_rur	70.92	0.00
agriculture ~ 1 + unemploy + sub_agri + dens_rur	75.20	0.00
agriculture ~ 1 + unemploy + sub_agri + pdens	75.22	0.00
agriculture ~ 1 + sub_agri + dens_rur + pdens	75.22	0.00
agriculture ~ 1 + unemploy + dens_rur + pdens	75.90	0.00

*agriculture= agriculture expansion, dens_rur= density of rural settlments,nfo= non-farm occupation, sub_agri= subsidies to agriculture activities pdens= population density, unemploy= unemployment rate

Table S7. Linear models of the effect of underlying drivers on the road density change. The models are ordered from the best to the worst performance according to the AICc value and their relative importance (weights).

Model	AIC	weights
road_change ~ 1 + pdens	-31.95	0.48
road_change ~ 1 + sub_agri + pdens	-28.68	0.09
road_change ~ 1 + nfo + pdens	-28.53	0.09
road_change ~ 1 + dens_rur + pdens	-28.15	0.07
road_change ~ 1 + unemploy + pdens	-27.97	0.07
road_change ~ 1	-27.13	0.04
road_change ~ 1 + unemploy	-26.61	0.03
road_change ~ 1 + dens_rur	-26.33	0.03
road_change ~ 1 + sub_agri + dens_rur + pdens	-24.64	0.01
road_change ~ 1 + nfo	-24.23	0.01
road_change ~ 1 + sub_agri + nfo + pdens	-24.04	0.01
road_change ~ 1 + dens_rur + nfo + pdens	-24.02	0.01
road_change ~ 1 + sub_agri	-23.84	0.01
road_change ~ 1 + unemploy + sub_agri + pdens	-23.62	0.01
road_change ~ 1 + unemploy + nfo + pdens	-23.56	0.01
road_change ~ 1 + dens_rur + nfo	-23.52	0.01
road_change ~ 1 + unemploy + dens_rur + pdens	-23.29	0.01
road_change ~ 1 + unemploy + dens_rur	-23.28	0.01
road_change ~ 1 + unemploy + sub_agri	-22.98	0.01
road_change ~ 1 + unemploy + nfo	-22.95	0.01
road_change ~ 1 + sub_agri + dens_rur	-22.95	0.01
road_change ~ 1 + sub_agri + nfo	-20.26	0.00
road_change ~ 1 + unemploy + sub_agri + dens_rur	-19.38	0.00
road_change ~ 1 + unemploy + dens_rur + nfo	-19.12	0.00
road_change ~ 1 + sub_agri + dens_rur + nfo	-19.05	0.00
road_change ~ 1 + unemploy + sub_agri + nfo	-18.17	0.00

* road_change= road density change, dens_rur= density of rural settlments, nfo= non-farm occupation, sub_agri= subsidies to agriculture activities pdens= population density, unemploy= unemployment rate

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Apéndice 4F. Resultados suplementarios

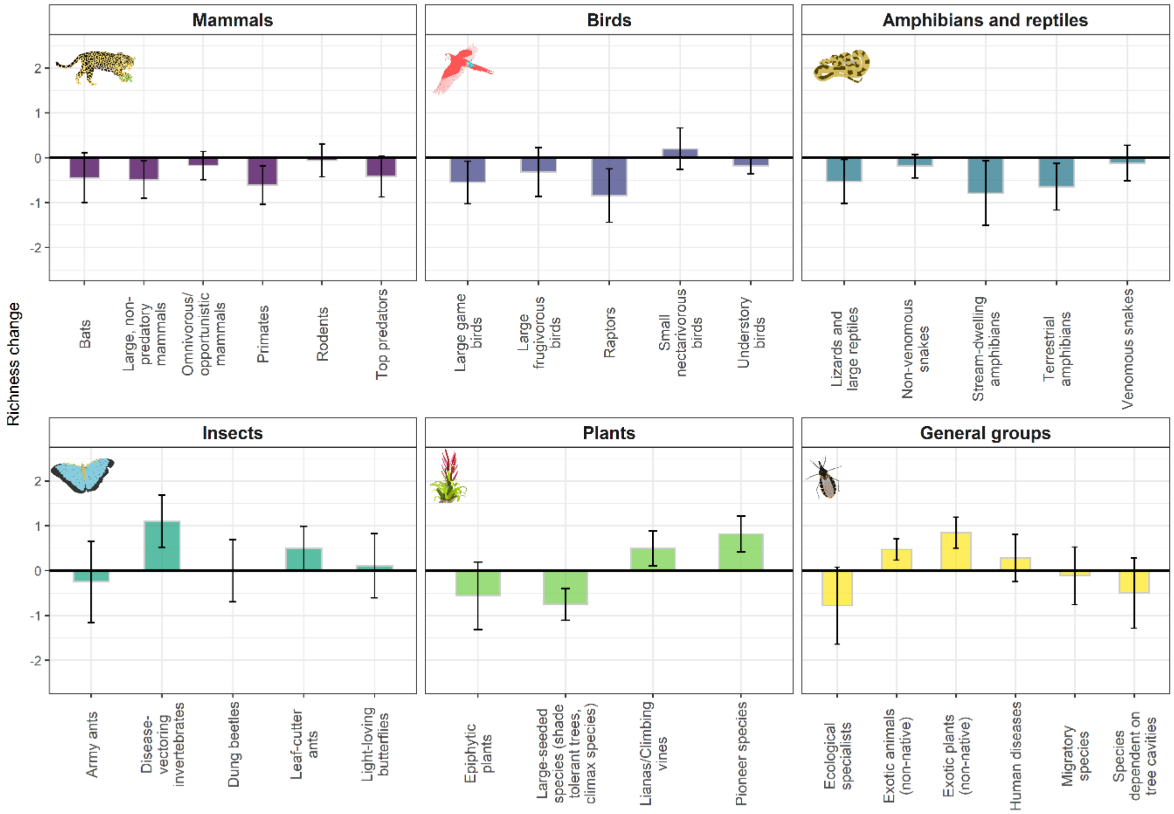

Figure S1. Mean change (± 95% confidence interval) of species richness of 31 biological guilds over the last 30 years in the studied reserves. Both mean values and 95% CI were estimated using bootstrapping with 10,000 iterations. We considered that a change was significant if 95% CIs did not overlap zero.

Table S1. Changes in abundance and species richness of all biological guilds (n = 31), and separately for disturbance-sensitive and disturbance-tolerant guilds in 14 Mesoamerican biosphere reserves over the last 30 years. The mean value, 95% CI, and p-value were estimated by bootstrapping (10,000 iterations).

Guild	variable	mean	CI-down	CI-up	p-value
General biodiversity	abundance	-0.33	-0.45	-0.20	0.0001
	richness	-0.20	-0.30	0.10	0.0001
Disturbance-sensitive species	abundance	-0.88	-1.14	-0.63	0.0001
	richness	-0.55	-0.76	-0.33	0.0001
Disturbance-tolerant species	abundance	0.68	0.43	0.93	0.0001
	richness	0.35	0.17	0.53	0.0002

Table S2. Trends in abundance changes of thirty-one biological guilds in the studied biosphere reserves located in the Mesoamerican biodiversity hotspot. The mean value, CI, and p-value were estimated by bootstrapping (10,000 iterations). The p-value denotes the probability that the register change is not statistically significant ($\bar{x}=0$, $\alpha=0.05$). Significant p-values are indicated with boldface.

Group	Guild	mean	CI-down	CI-up	p-value
Mammals	Bats	-0.33	-0.94	0.28	0.2847
	Large, non-predatory mammals	-0.77	-1.28	-0.27	**0.0028**
	Omnivorous/opportunistic mammals	-0.16	-0.61	0.28	0.4615
	Primates	-1.04	-1.58	-0.51	**0.0002**
	Rodents	0.34	-0.27	0.92	0.2809
	Top predators	-0.64	-1.21	-0.07	**0.0285**
Birds	Large frugivorous birds	-0.63	-1.24	-0.03	**0.0401**
	Large game birds	-0.83	-1.44	-0.21	**0.0082**
	Raptors	-1.42	-2.07	-0.77	**0.0001**
	Small nectarivorous birds	-0.20	-0.80	0.41	0.5253
	Understory birds	-0.30	-0.75	0.16	0.2088
Amphibians and reptiles	Lizards and large reptiles	-0.84	-1.33	-0.36	**0.0008**
	Non-venomous snakes	-0.56	-0.99	-0.14	**0.0088**
	Stream-dwelling amphibians	-1.15	-1.79	-0.49	**0.0007**
	Terrestrial amphibians	-1.06	-1.59	-0.53	**0.0001**
	Venomous snakes	-0.11	-0.58	0.35	0.6258
Insects	Army ants	-0.78	-1.69	0.14	0.0985
	Disease-vectoring invertebrates	1.30	0.50	2.09	**0.0015**
	Dung beetles	-0.12	-0.94	0.68	0.7573
	Leaf-cutter ants	0.62	-0.06	1.32	0.0736
	Light-loving butterflies	-0.20	-0.87	0.47	0.5560
Plants	Epiphytic plants	-0.85	-1.61	-0.09	**0.0286**
	Large-seeded species (shade tolerant trees, climax species)	-1.32	-1.78	-0.87	**0.0001**
	Lianas/Climbing vines	0.79	0.25	1.33	**0.0039**
	Pioneer species	1.36	0.97	1.77	**0.0001**
General groups	Ecological specialists	-1.30	-2.22	-0.38	**0.0058**
	Exotic animals (non-native)	1.06	0.76	1.36	**0.0001**
	Exotic plants (non-native)	1.40	0.98	1.82	**0.0001**
	Human diseases	0.43	-0.11	0.97	0.1210
	Migratory species	-0.40	-1.24	0.43	0.3457
	Species dependent on tree cavities	-0.78	-1.69	0.12	0.0882

Table S3. Trends in richness changes of thirty-one biological guilds in the studied biosphere reserves located in the Mesoamerican biodiversity hotspot. The mean value, confidence intervals, and p-value were estimated by bootstrapping (10,000 iterations). The p-value denotes the probability that the register change is not statistically significant ($\bar{x}=0$, $\alpha=0.05$). Significant p-values are indicated with boldface.

Group	Guild	mean	CI-down	CI-up	p-value
Mammals	Bats	-0.45	-1.00	0.12	0.1207
	Large, non-predatory mammals	-0.49	-0.90	-0.07	**0.0221**
	Omnivorous/opportunistic mammals	-0.17	-0.49	0.14	0.2776
	Primates	-0.61	-1.04	-0.19	**0.005**
	Rodents	-0.05	-0.42	0.31	0.7568
	Top predators	-0.42	-0.87	0.04	0.0748
Birds	Large frugivorous birds	-0.32	-0.86	0.23	0.258
	Large game birds	-0.54	-1.02	-0.08	**0.0229**
	Raptors	-0.84	-1.44	-0.25	**0.0056**
	Small nectarivorous birds	0.20	-0.26	0.67	0.3896
	Understory birds	-0.18	-0.36	0.01	0.0591
Amphibians and reptiles	Lizards and large reptiles	-0.52	-1.02	-0.04	**0.0351**
	Non-venomous snakes	-0.19	-0.45	0.07	0.1476
	Stream-dwelling amphibians	-0.79	-1.50	-0.06	**0.0339**
	Terrestrial amphibians	-0.65	-1.17	-0.13	**0.0148**
	Venomous snakes	-0.12	-0.51	0.28	0.5628
Insects	Army ants	-0.25	-1.16	0.65	0.584
	Disease-vectoring invertebrates	1.10	0.52	1.69	**0.0003**
	Dung beetles	0.00	-0.70	0.69	0.9974
	Leaf-cutter ants	0.50	0.01	0.99	**0.0455**
	Light-loving butterflies	0.11	-0.60	0.83	0.7566
Plants	Epiphytic plants	-0.55	-1.32	0.19	0.1457
	Large-seeded species (shade tolerant trees, climax species)	-0.75	-1.11	-0.39	**0.0001**
	Lianas/Climbing vines	0.50	0.11	0.88	**0.0112**
	Pioneer species	0.82	0.42	1.22	**0.0001**
General groups	Ecological specialists	-0.78	-1.64	0.08	0.0759
	Exotic animals (non-native)	0.47	0.23	0.71	**0.0001**
	Exotic plants (non-native)	0.85	0.50	1.19	**0.0001**
	Human diseases	0.29	-0.24	0.81	0.2822
	Migratory species	-0.11	-0.75	0.53	0.7269
	Species dependent on tree cavities	-0.50	-1.28	0.28	0.2091

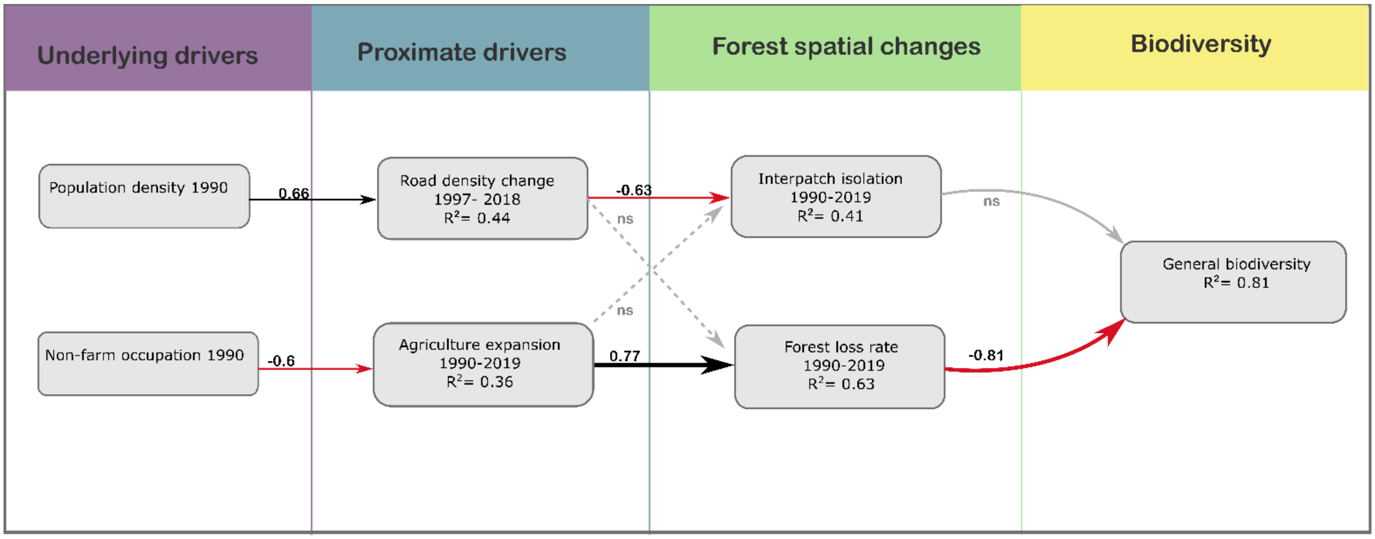

Figure S2. Structural equation model (SEM) of the relationships between underlying drivers, proximate drivers, forest cover, and general biodiversity changes (i.e., considering all guilds). Significant ($p < 0.05$) positive and negative paths are indicated with black and red arrows, respectively, whereas gray arrows indicate non-significant relationships ($p>0.05$). Values near the arrows correspond to standardized coefficients and indicate the direction (positive/negative) and strength of each path. Note that an increase in interpatch isolation change indicates that isolation decreased though time. Within the box of each response variable, we also show the R^2 value. The model fitted the data well (Fisher's C: 18.66, p = 0.76).

Apéndice 5. Publicación en el boletín de la SCME

El siguiente texto es de difusión y forma parte del la edición de noviembre del boletín de la Sociedad Científica Mexicana de Ecología.

Parque Nacional Cascadas de Agua Azul, Chiapas. Fotografía: Daniel M. Auliz-Ortiz.

Bosque tropical y Áreas Naturales Protegidas: conservación y amenazas

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Resumen: Los bosques tropicales son biomas muy importantes en términos de su biodiversidad, por lo que implementar estrategias de conservación como las Áreas Naturales Protegidas (ANPs) son de vital importancia para asegurar su permanencia. En este trabajo se hace una descripción general sobre el estado de conservación de las ANPs de bosques tropicales en México; las amenazas para su conservación; y sus posibles consecuencias sobre la biodiversidad. A lo largo de las últimas décadas, la superficie de bosques tropicales bajo protección ha incrementado debido a los esfuerzos por establecer más ANPs. No obstante, las ANPs están sujetas a actividades humanas que promueven la deforestación dentro y alrededor de ellas. Como consecuencia, en las ANPs se han registrado cambios en la riqueza y abundancia de distintas especies dependiendo de sus características funcionales. Por tanto, es necesario identificar prácticas de manejo que mitiguen los efectos adversos de origen antropogénico sobre los bosques tropicales, localizados tanto en las ANPs como en las zonas no protegidas.

Palabras clave

biodiversidad, deforestación, conservación, factores de cambio, reservas naturales.

Los bosques tropicales pueden ser clasificados por su fenología en dos tipos: bosque tropical perennifolio (i.e., precipitación promedio superior a 1600 mm anuales y al menos 60 mm en el mes más seco) y bosque tropical caducifolio (i.e., precipitación promedio cercana a 800 mm anuales con temporada de secas de hasta 8 meses). En su conjunto, estos bosques representan los biomas terrestres más importantes en términos de diversidad de especies (Dinerstein *et al.* 2017). Tan solo en México, este bioma alberga cerca de 11,000 especies de plantas y aproximadamente 1,500 especies de vertebrados.

En nuestro país, los bosques tropicales cubren alrededor de 30 millones de hectáreas y se localizan principalmente en el sureste del país y la vertiente del Pacífico. No obstante, solamente una pequeña porción se encuentra protegida por alguna de las 146 Áreas Naturales

Protegidas (ANPs) terrestres federales. En concreto, únicamente 10% de la superficie de bosque tropical cumple con esta condición; y esta cifra es del 15% considerando únicamente el bosque perennifolio y 7% si consideramos solo el bosque caducifolio (Figura 1a).

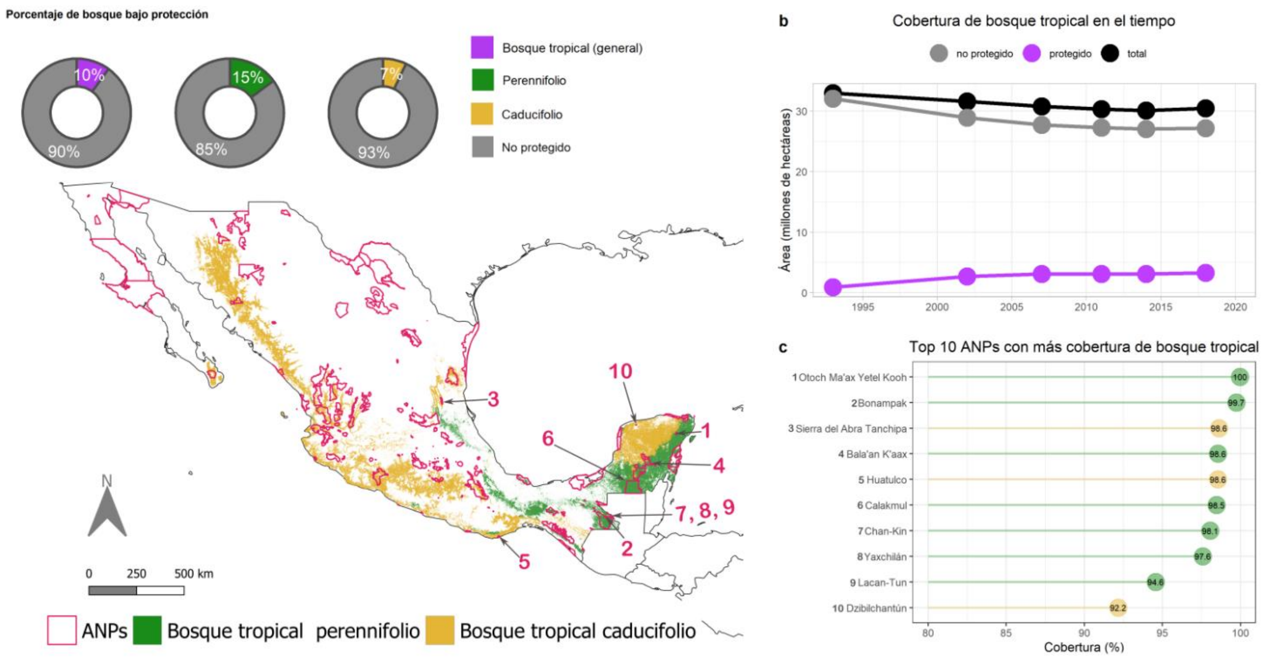

Figura 1. Descripción del bosque tropical en México en términos de su cobertura y protección: a) el mapa muestra la distribución del bosque tropical perennifolio y caducifolio, así como la localización de las ANPs federales. Los gráficos de dona muestran el porcentaje de bosque bajo protección. b) Cambio en la cobertura de bosque a lo largo del tiempo. c) Se muestran las ANPs con mayor cobertura relativa de bosque tropical. Los diferentes números en “a” corresponden a las ANPs denotadas con ellos en “c”. Elaboración: Daniel M. Auliz-Ortiz con datos de la serie VII de uso de suelo y vegetación del INEGI y Madmex de CONABIO.

Pese a su importancia y a los esfuerzos de conservación, la cobertura de bosque tropical se redujo como ningún otro bioma en el país. De acuerdo con datos del Instituto Nacional de Estadística y Geografía (INEGI), tan sólo en tres décadas y media, disminuyó casi 7.6 % (Figura 1b). No obstante, la mayor parte de la pérdida de cobertura ocurrió en sitios no protegidos. En contraste, la superficie bajo protección incrementó en el mismo periodo de tiempo debido al establecimiento de un mayor número de ANPs, pasando de 21 en 1993 a 82 en 2022.

Las ANPs son uno de los mejores instrumentos para la conservación de la biodiversidad, pues g reducen significativamente la deforestación dentro de sus fronteras. De las 82 ANPs, destacan Otoch Ma'ax Yetel Kooh (Quintana Roo), Bonampak (Chiapas) y Bala'an K'aax (Quintana Roo) por tener mayor cobertura de bosque tropical perennifolio. Mientras que, las ANPs Sierra del Abra Tanchipa (San Luis Potosí/Tamaulipas), Huatulco (Oaxaca) y Dzibilchaltún (Yucatán) poseen la mayor cobertura de bosque tropical caducifolio (Figura 1c). A pesar de ser una herramienta poderosa para la conservación, también pueden ser afectadas por la deforestación.Por ejemplo, el 15% de estas perdió más de 5% de la cobertura boscosa entre el 2000 y el 2019. Asimismo, alrededor del 15% de las ANPs se encuentra sometidas a una fuerte presión antropogénica como lo denotan los valores altos del índice de huella humana, un indicador de impacto humano que considera los tipos de coberturas antropogénicas (Figura 2).

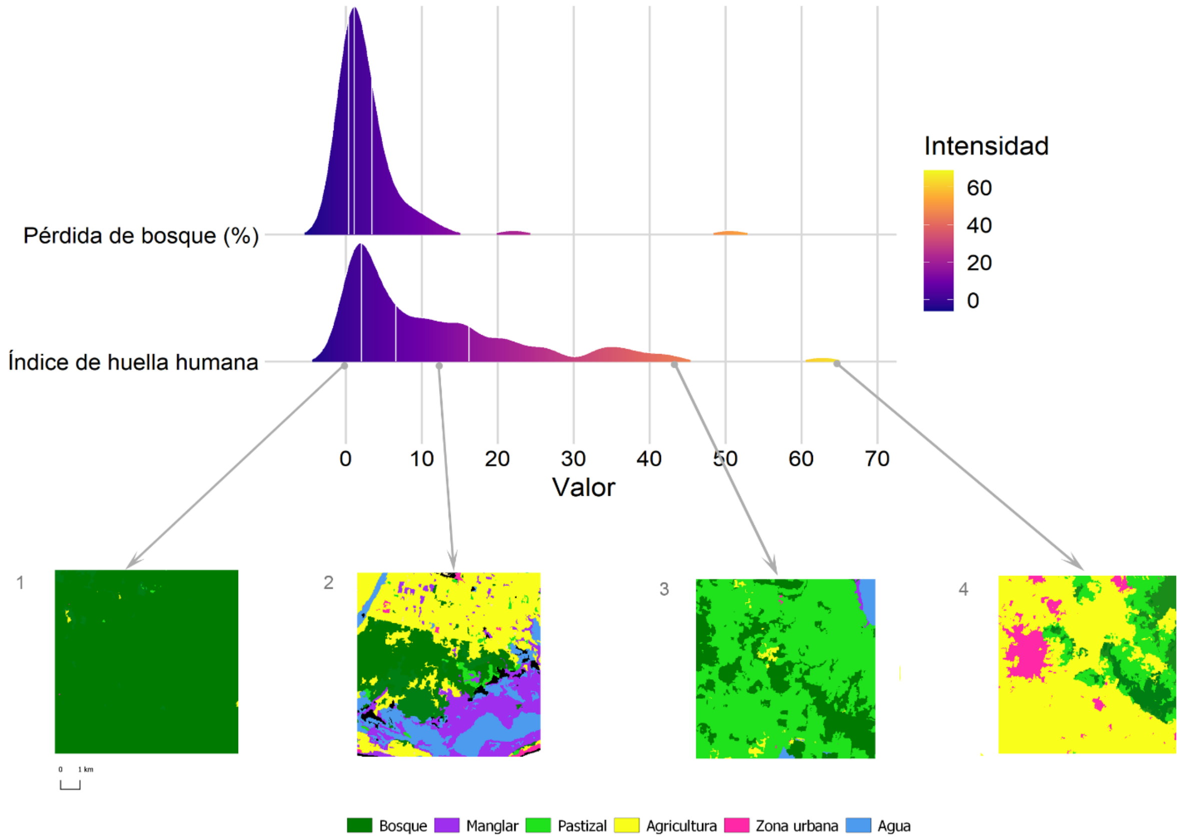

Figura 2. Estado de las ANPs en el bosque tropical (n=85), descrito en términos del porcentaje de bosque perdido entre 2000-2019 y el índice de huella humana (indicador de impacto). Las curvas corresponden a la densidad de los datos (área bajo la curva = 1) y su altura indica la proporción de ANPs con los valores de las variables indicadas en el eje "y". El color de las curvas indica incremento en el valor de cada variable (i.e., intensidad) y las líneas blancas indican percentiles (25%, 50% y 75%). Abajo se observan cuatro ejemplos de paisajes que ocurren en las ANPs y representan un gradiente con distintos valores del índice de huella humana. 1) Área de Protección de Flora y Fauna Bala'an K'aax, 2) Parque Nacional Lagunas de Chacahua, 3) Reserva de la Biosfera Los Tuxtlas, 4) Monumento Nacional Yagul. Elaboración: Daniel M. Auliz-Ortiz con datos de González-Abraham *et al.* 2015 y Hansen *et al.* 2013.

La pérdida de hábitat causada por actividades antropogénicas es una de las mayores amenazas a la conservación de las ANPs. Dentro de los factores que causan la pérdida de hábitat se pueden distinguir dos tipos dependiendo de su acción (directa o indirecta) y se conocen con el nombre de factores proximales y subyacentes (Figura 3). Los factores proximales son aquellos que directamente causan la remoción de la cobertura forestal (Curtis *et al.* 2018). Algunos de los más importantes son la extensión de la frontera agrícola, la construcción de infraestructura (por ejemplo, redes de carreteras y asentamientos humanos) y la extracción de madera. Por otro lado, los factores subyacentes son aquellos que indirectamente modifican la cobertura del suelo al favorecer o inhibir la acción de los factores proximales. Estos pueden ser de tipo demográfico, económico, político, tecnológico o cultural (Geist y Lambin 2002). Por ejemplo, la demanda de aceite de palma en países de primer mundo incentiva la deforestación para el cultivo de estas plantas en Asia y América por sus bajos costos de producción. Por otra parte, el crecimiento de la población incrementa la demanda de recursos (comida, energía, trabajos, espacios para vivir y servicios de salud). Ante esto, los gobiernos llevan a cabo programas de desarrollo de infraestructura (como la construcción de carreteras, hospitales, viviendas, etc.) que favorecen la inmigración y consecuentemente la deforestación.

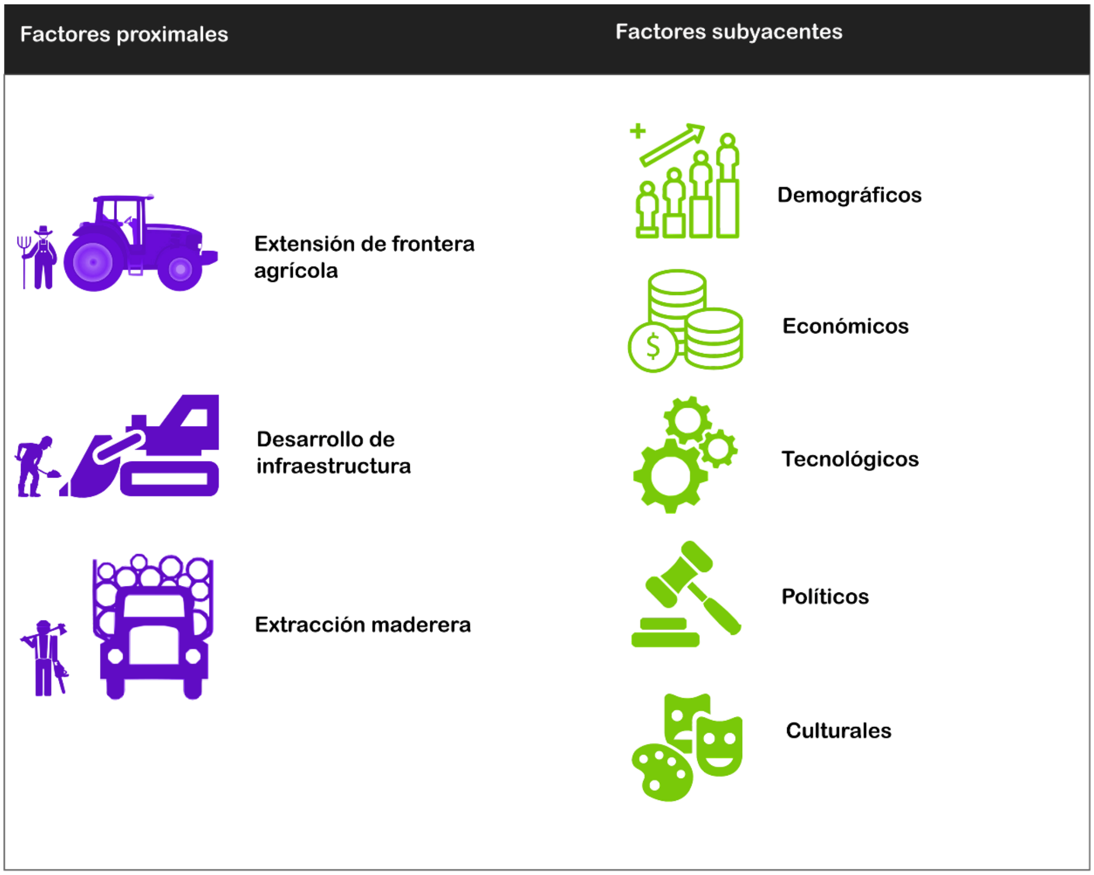

Figura 3. Los promotores proximales y subyacentes de los cambios de cobertura de suelo. Elaboración: Daniel M. Auliz-Ortiz, adaptado de Geist y Lambin 2002.

Además de los factores antes mencionados, las ANPs son susceptibles a experimentar cambios como resultado de procesos de origen natural. Los incendios forestales, los deslizamientos de tierra y las tormentas son factores que pueden promover la pérdida de cobertura forestal tanto en sitios protegidos como no protegidos.

Para lograr la protección de los biomas, las ANPs imponen restricciones al uso de recursos naturales y limitan el tipo de actividades permitidas. Si bien esto puede reducir la deforestación dentro de sus fronteras, también puede ocasionar la desvinculación de las comunidades con las ANPs y una presión sobre los recursos naturales que las rodean (Durand y Lazos 2008). En algunos casos, tal situación lleva a un proceso conocido como “derrame” (*leakage* o *spillover* en inglés), en donde la deforestación que se evita dentro del ANP se traslada a las zonas adyacentes no protegidas, por lo que el ANP en realidad magnifica la deforestación a su alrededor.

Las transformaciones de la cobertura forestal en las ANPs y sus alrededores pueden tener severos impactos sobre la biodiversidad que protegen. Por un lado, en las ANPs donde la cobertura forestal se ha reducido drásticamente en las últimas décadas (por ejemplo, en la Reserva de la Biosfera de los Tuxtlas), las especies altamente dependientes del bosque como los animales de talla grande (por ejemplo, tapir, jaguar, águila harpía y puma) y los árboles de crecimiento lento adaptadas a la sombra (por ejemplo, la caoba) han sido prácticamente eliminadas de la zona (Figura 4). En contraste, los hábitats emergentes en las ANPs tales como pastizales, campos agrícolas, bordes de bosque o pequeños asentamientos humanos pueden ser aprovechados por especies de talla pequeña y plantas de rápido crecimiento y demandantes de luz, incrementando su abundancia (Filgueiras *et al.* 2021). Además, la deforestación fuera de las ANPs puede aislar a las poblaciones de organismos que habitan dentro de ellas debido a la disminución en la conectividad con otras reservas o parches de hábitat.

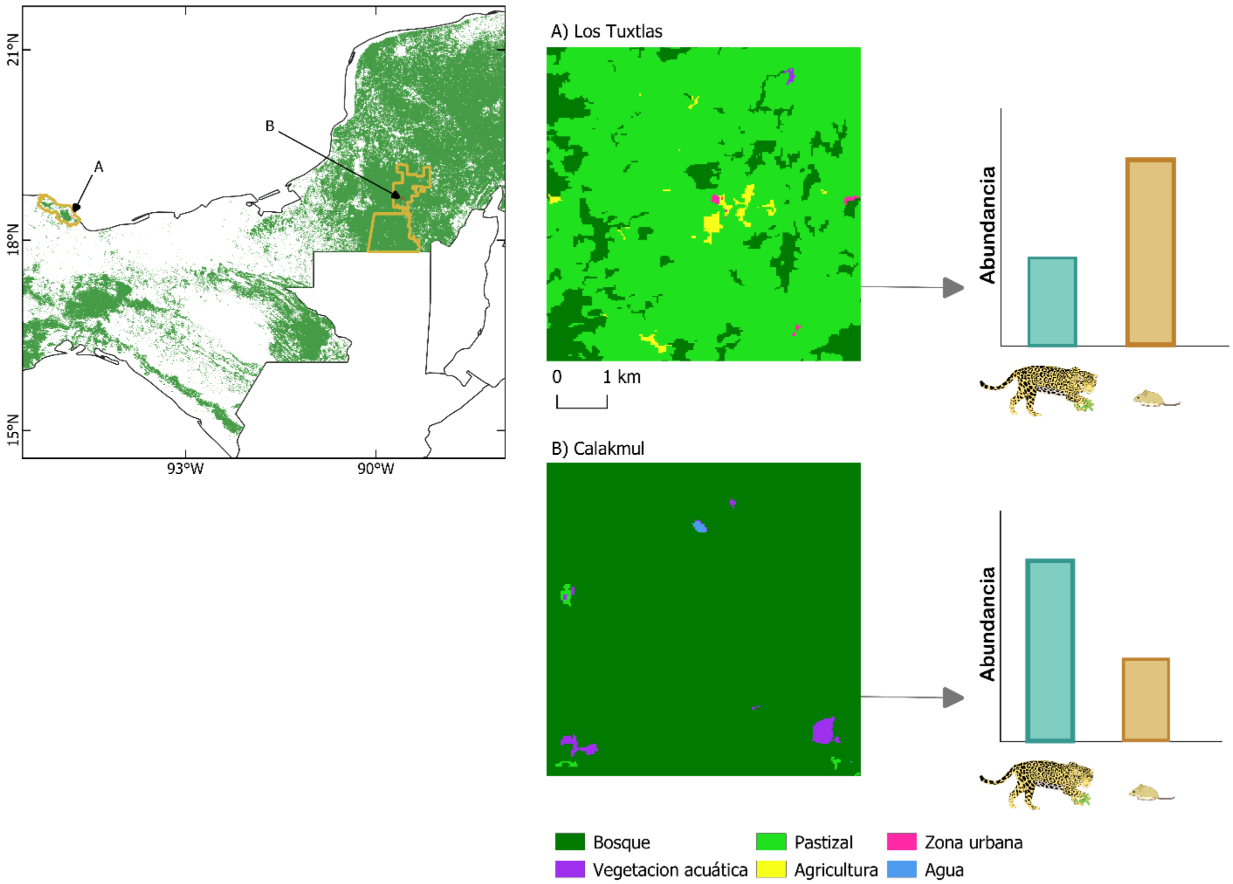

Figura 4. Ejemplo de dos reservas con estructuras forestales contrastantes: Los Tuxtlas y Calakmul. La baja cobertura de bosque en los Tuxtlas favorece el incremento en abundancia de especies de talla pequeña y hábitos generalistas. La alta cobertura de bosque en Calakmul favorece la existencia de especies de talla grande de hábitos más especializados. Elaboración: Daniel M. Auliz-Ortiz con base en manuscrito en preparación.

Ante estas circunstancias existe la necesidad de concebir la protección de los bosques tropicales de manera más integral. Para ello, es necesario identificar en qué contextos socioeconómicos se reducen los impactos antropogénicos sobre las ANPs. Los trabajos más recientes en torno a los promotores de cambio en ANPs mexicanas sugieren dos cosas: planificar los asentamientos humanos en torno a las ANPs y diversificar las oportunidades laborales más allá de las actividades agrícolas (Auliz-Ortiz *et al.* 2022). Por un lado, la planificación territorial y demográfica en torno a las ANPs permitiría atenuar los efectos negativos sobre la cobertura forestal. No obstante, esto representa todo un reto en términos de inclusión y organización social. En este sentido, la diversificación laboral parece ser una opción más viable. Las poblaciones al diversificar sus fuentes de ingresos y mejorar las prácticas de manejo pueden demandar menos tierra para actividades agrícolas y así disminuir la deforestación causada por la extensión de la frontera agrícola. No obstante, esta alternativa no representa una panacea ya que no existen soluciones sencillas ante problemáticas tan complejas. Por el contrario, deben contemplarse factores locales para poder implementar mejores medidas de manejo que permitan mitigar los impactos antropogénicos sobre los bosques tropicales.

Conclusiones

En el contexto nacional los bosques tropicales son biomas muy importantes pues, además de ser los más biodiversos, son de los más amenazados pese a los esfuerzos para su conservación.

Para asegurar la conservación de bosques tropicales en las ANPs de nuestro país, es necesario entender los factores que promueven su degradación, además de diseñar e implementar estrategias de manejo que permitan reducir los impactos negativos generados por las actividades humanas. Esto abre todo un campo para la cooperación transdisciplinaria.

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