

CONTRIBUTED PAPER

Conservation effectiveness of seven biosphere reserves in the Western Amazon

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Abstract

Over the past three decades, a complex network of area-based conservation strategies and Indigenous territories has expanded across the Amazon, driven by global efforts to curb biodiversity loss and regional initiatives to uphold Indigenous Peoples' rights. Yet, deforestation continues to threaten the region's ecological integrity and the sociocultural resilience of its diverse mosaic of Indigenous, rural, and traditional communities. Biosphere reserves—spanning protected areas, Indigenous territories, and rural–urban settlements—epitomize the governance challenges and environmental pressures facing the broader Amazon Basin. Evaluating their conservation effectiveness is critical for advancing the goals of the Kunming-Montreal Global Biodiversity Framework. In this study, we apply advanced matching methods to assess deforestation avoidance in seven biosphere reserves across the western Amazon: Podocarpus-El Cóndor, Yasuní, and Sumaco in Ecuador; Manu and BIOAY in Peru; and Beni and Pilón Lajas in Bolivia. Our findings show that most biosphere reserves in the region are effective in reducing deforestation, although forest loss is still increasing across all sites. Notably, transition areas and buffer zones generally avoided more deforestation than core areas, despite their relatively less stringent protection status. These results contribute to the growing

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evidence of the vital role biosphere reserves can play in mitigating deforestation. Their hybrid governance models, institutional arrangements, and territorial strategies may provide meaningful pathways for aligning local aspirations with global biodiversity targets.

KEYWORDS

Amazon Basin, biosphere reserves, conservation effectiveness, counterfactual analysis, matching methods, Western Amazon

1 | INTRODUCTION

The window of opportunity to take action to address the interwoven climate and biodiversity crises is narrowing (Díaz et al., 2019; IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), 2019, 2024b). A substantial body of evidence shows that the cumulative, interconnected, and far-reaching drivers of biodiversity loss can only be effectively tackled through urgent and coordinated actions that promote transformative change, understood as a fundamental system-wide reconfiguration of technological, economic and social domains, including prevailing paradigms, goals and values (Visseren-Hamakers et al., 2021). Such change is urgently called upon to transform how conservation policies are designed, implemented, and enforced, across different scales, from national to international, and sectors, confronting deeply embedded structural dynamics, entrenched systemic barriers, and critical leverage points in prevailing decision-making architectures (Chan et al., 2020; Díaz et al., 2020).

Designated by UNESCO's the Man and the Biosphere (MAB) Programme, biosphere reserves' long and continuously evolving experience over the past 50 years offers guidance for putting in place the seeds for this foundational, system-wide shift away from "business as usual" models that isolate conservation from production and settlement areas (Barraclough et al., 2023; Reed, 2016; Stoll-Kleemann & Welp, 2008; Van Cuong et al., 2017). Considered as "learning places for sustainable development," biosphere reserves offer management and collaborative approaches that strive to balance biodiversity conservation and sustainable development across an array of juxtaposed territories, urban and rural, including protected areas, Indigenous lands, rural settlements, and diverse political-administrative jurisdictions (Barraclough et al., 2023; Bridgewater, 2002; Dabard et al., 2025). There are at the moment 784 UNESCO biosphere reserves in the world, spread across 142 countries, including 25 transboundary sites, that are home to more than 300 million people that compose the dynamic and interactive World Network of Biosphere Reserves (WNBR) (UNESCO, 2024, 2025). In that

regard, the WNBR might work as bridging institutions to steer collaboration and concerted actions from multiple agents, stakeholders, and organizations toward reconciling and leveraging conservation and production goals across interconnected and overlapping territories (Barraclough et al., 2022; Schultz et al., 2018). This entails a shift across sectors in how human-nature inter-relations, and their multiple expressions at the territorial level are recognized, reconsidered, and sustained in conservation policy and practice (Ellis, 2019; Kremen & Merenlender, 2018; Visseren-Hamakers et al., 2021). Actions as such are inextricably linked to global efforts to address the lattice of interdependencies between biodiversity loss and climate change (Díaz et al., 2019; IPBES, 2019, 2024a).

Transformative action for both people and nature can be enabled and accelerated through collaborative conservation approaches that prioritize justice, inclusion and plural visions of a good life (Reyes-García et al., 2022; Vucetich et al., 2018). Within this context, biosphere reserves have been widely promoted as instruments to support transformative pathways by reclaiming, restoring and reconfiguring relationships between people and nature (Ishwaran et al., 2008; Moreira-Muñoz et al., 2025; Reed & Massie, 2013). Biosphere reserves are structured around a zonation system (Table 1) that establishes a gradient of management objectives, ranging from strictly protected core areas to zones dedicated to sustainable development and livelihood activities (UNESCO, 2022).

The role of biosphere reserves in conservation policy and practice has gained increasing recognition (Barraclough et al., 2023; Bridgewater, 2002; Stoll-Kleemann & Welp, 2008), particularly in debates concerning their potential contribution to national and global commitments to conserve 30% of the planet's lands and waters by 2030 under the Kunming-Montreal Global Biodiversity Framework (Barraclough & Måren, 2022). As part of UNESCO's MAB Programme, biosphere reserves are explicitly designed to integrate conservation with participatory governance, positioning them as potentially well aligned with global biodiversity and sustainability agendas as well as National Biodiversity Strategies and Action Plans (NBSAPs), including several Kunming-Montreal targets (Barraclough

TABLE 1 Biosphere reserves zonation system and correspondence to IUCN categories based on protected areas management objectives and protection levels (Bridgewater et al., 1996; UNESCO, 2022).

Biosphere reserve zonation	Description
Core areas	Areas strictly protected and fully dedicated to long-term biodiversity conservation, with a focus on informing about ecosystemic processes and functions. Very often, these represent territories officially recognized in the countries' national protected areas system. These areas fall within I to IV and sometimes VI categories of the IUCN management scale.
Buffer zones	Areas surrounding and protecting the core area in which land uses are limited to research monitoring, education, and ecologically sound activities in accordance to conservation objectives. These spaces allow for the development and exploration of management techniques with the aim of preserving semi-natural ecosystems that take biodiversity into account. These areas fall within IV to VI categories of the IUCN management scale.
Transition areas	Areas with fewer restrictions where diverse land uses are allowed with particular emphasis on developing, supporting, and encouraging sustainable productive activities. These areas focus on supporting and encouraging local communities and enterprises with an aim to maintain sustainable land use and socio-economic systems. These areas can fall within V or VI of the IUCN management scale.

Note: UNESCO's zonation system for biosphere reserves also correlates with the management and conservation categories defined in each country's national official protected areas system.

et al., 2025). The WNBR therefore holds considerable potential to contribute to area-based conservation targets across diverse countries and biomes (Barraclough et al., 2025; Wu et al., 2025). Despite this promise, biosphere reserves face a range of persistent challenges, underscoring the need for careful evaluation of their conservation effectiveness and outcomes (Gohr et al., 2025; Sandström et al., 2025). To date, however, empirical assessments of biosphere reserve effectiveness remain scarce, with only a limited number of studies addressing this question (e.g., Auliz-Ortiz et al., 2024; Coetzer et al., 2014; Ferreira et al., 2020; Gohr et al., 2025).

In this study, we aim to fill this gap by using matching methods to evaluate the ability to deter deforestation of seven biosphere reserves in the Western Amazon. Increasingly, matching methods have gained prominence among non-experimental approaches for assessing conservation effectiveness (Schleicher et al., 2020; Sze et al., 2021). By reducing treatment effect estimation bias, they become particularly useful in protected area impact assessment where analysts face a situation in which treatment placement and outcome could be related (Stuart, 2010). Carefully designed, evaluated, and context-based estimations and interpretations of the matching results are essential to bring this methodological framework forward within conservation studies (Eklund et al., 2019; Schleicher et al., 2020). Within our study, we aim to advance on the practice of this methodology by implementing matching assessments independently (for each study case), and by integrating both match quality and pressure estimates into the assessment process. To our best knowledge, this is the first study to systematically examine the conservation effectiveness of the Amazon's biosphere reserves.

More specifically, we aimed to understand the following questions:

1. Are the biosphere reserves effective at avoiding deforestation?
2. Does conservation effectiveness in avoiding deforestation vary according to the conservation and management zonation system of biosphere reserves?
3. How does the matching analysis performance vary in quality and reliability across biosphere reserves and their zonation system?

The findings of this study are relevant to advance current scientific knowledge about the extent to which biosphere reserves can effectively contribute to advancing toward the targets of the Kunming-Montreal Global Biodiversity Framework.

2 | RESEARCH DESIGN AND METHODS

2.1 | Study area

We analyzed seven biosphere reserves across Bolivia, Ecuador, and Peru included in the scope of UNESCO's Amazon Biosphere Reserves Project. Despite their shared location in the western Amazon River basin (Figure 1), these biosphere reserves vary in terms of designation year, extension, and demography (Table 2). They also feature diverse biophysical attributes, such as

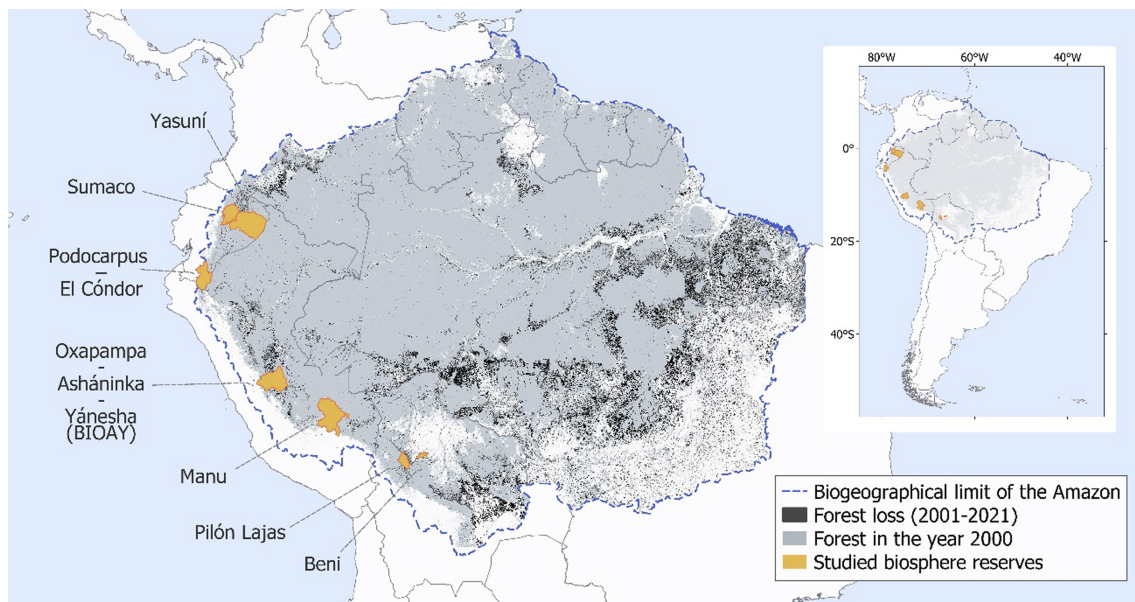


FIGURE 1 Map of the study area.

TABLE 2 Biosphere reserves under study.

Country	Biosphere reserve	Designation year	Area (km ²)	Percentage of land cover			Population ^a
				Core area	Buffer zone	Transition area	
Bolivia	Beni Biosphere Reserve	1987	1348.1	25.9	46.75	27.3	653
Bolivia	Pilón Lajas Biosphere Reserve	1992	4050.9	42.74	26.54	30.72	11,007
Ecuador	Podocarpus—El Cónдор Biosphere Reserve	2007	11398.4	19.05	29.8	51.15	388,784
Ecuador	Sumaco Biosphere Reserve	2000	9950.6	20.66	21.88	57.47	197,723
Ecuador	Yasuní Biosphere Reserve	1989	27406.0	29.88	56.16	13.95	77,167
Peru	Manu Biosphere Reserve	1977	24205.8	70.18	11.12	18.7	36,487
Peru	Oxapampa-Asháninka-Yánesha Biosphere Reserve (BIOAY)	2010	18032.2	6.14	59.09	34.76	98,369

^aSims, K., Reith, A., Bright, E., McKee, J., & Rose, A. (2022). *LandScan Global 2021* [Data set]. Oak Ridge National Laboratory. <https://doi.org/10.48690/1527702>.

topography and vegetation typologies, and differ in terms of the socioeconomic context in which they are embedded both at national and local levels, featuring a diverse population, urban and rural, whose livelihoods and economies experience varying connection degrees to markets and infrastructure. As such, these biosphere reserves encompass the broader Amazonian challenges of reconciling sustainable development and conservation across a mosaic of social and environmental realities in rapid transformation. At the national level, biosphere reserves in Bolivia, Ecuador, and Peru are officially recognized in their respective national systems of protected territories.

2.2 | Matching methods

Matching methods (Figure 2) are a powerful non-parametric data pre-treatment that has been increasingly used to study the impact of conservation strategies, such as protected areas (Ribas et al., 2021; Schleicher et al., 2020). These methodologies take Rubin's Causal Inference theory as the basis for reducing bias in the distribution of control and treatment (Sekhon, 2011; Stuart, 2010). In assessing deforestation, for instance, matching methods compare forest loss inside protected areas to external areas potentially subject to similar deforestation pressures, representing one of the most

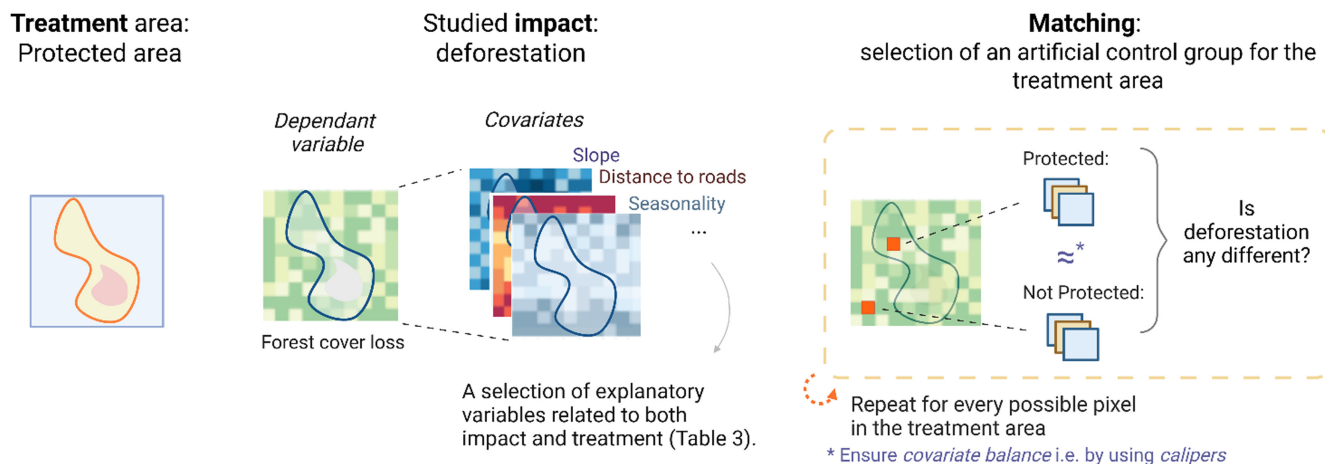


FIGURE 2 Methodological framework of the single cell (i.e., pixel-level) matching to assess the deforestation avoidance capacity in the biosphere reserves.

rigorous methodologies to empirically quantify the role of protected areas in avoiding deforestation (Ribas et al., 2021; Schleicher et al., 2020; Stuart, 2010). Thus, it has extensive use in assessing the conservation effectiveness of protected areas and Indigenous territories, particularly with regard to deforestation avoidance (Carranza et al., 2014; Hernandez et al., 2021; Nelson & Chomitz, 2011; Simkins et al., 2024).

2.3 | Operationalizing the matching methods analysis in the biosphere reserves

Usually matching methods tend to examine protected areas as a single entity, yet biosphere reserves integrate different area-based management strategies, which brings additional challenges to the effectiveness evaluation process. Given our aim to assess effectiveness both at the biosphere reserve level and also at each of the areas of the zoning system (see Table 1 for information on the zoning system), we performed a single cell Mahalanobis-distance matching with replacement for each biosphere reserve, as well as for each of the three zones in the reserve. The Average Treatment effect on the Treated (ATT) was used as the effectiveness estimate (Equation 1).

$$ATT = E(\partial | T = 1) = [E(\gamma_i - \gamma_o | T = 1)] \quad (1)$$

where E = the expected value, ∂ = the potential causal effect at a population (i.e., of units) level if the studied intervention is present, T = the presence ($T = 1$) or absence ($T = 0$) of the studied intervention, γ_i = the potential outcome of the treated units, and γ_o = the potential outcome of the untreated units. We aimed to artificially create two big control and treatment groups

that are as similar as possible in the balance of covariate values, considering the variables that affect the impact (i.e., deforestation). We establish our study area by limiting the matching to forested areas inside the biogeographical limits of the Amazon region (see “Forest cover and yearly forest cover loss” section for details on the forest cover base), including biome, river basin and administrative regions (RAISG (Red Amazónica de Información Socioambiental Georreferenciada), 2020) within each of the three studied countries, and excluding other protected areas and Indigenous territories. We then randomly selected 10% of the defined forest pixels within the study area. All pixels falling inside of the treatment areas are then matched with those outside the treatment area by measuring their similarity regarding the values for those variables known to affect the impact (Ribas et al., 2021). We assessed the similarity of the pairs by using the Mahalanobis distance as an equal percent bias reducing distance measurement (Figure 2).

We assessed the quality of the matching process by examining the representativeness of the procedure for each reserve (i.e., what proportion of the treatment data points were matched; see Table 2), and by measuring the correspondence of deforestation before and after of the matching process (i.e., by looking for differences in deforestation rates in the treatment and control groups before and after the matching process; see [Supporting Information: Complete results](#)). Additionally, we provided three Match Balance evaluation metrics and a visualization of the covariates before and after the matching ([Supporting Information: Match balance](#)). Balanced control and treatment groups are achieved by dropping matches under a given standard deviation distance by applying calipers. We applied a 0.25 caliper as a filtering tool for the quality of the matches and additionally,

performed fast-matching analyses at eight different caliper levels (ranging from 0.025 to 2) to understand how the balance and representativity of the matches responded to the constraints of the matching process (Supporting Information: Caliper analysis). The matching code was developed with R 4.1.3 (R Core Team, 2022) following the study by (Ribas et al., 2021). We used the Match and MatchBalance functions of the Matching R package (Sekhon & Saarinen, 2022).

2.4 | Measuring conservation effectiveness through deforestation avoidance

Within this study, conservation effectiveness is understood as a measure of the capacity of a protected territory to avoid deforestation when compared to its counterfactual scenario without protection. We used the Average Treatment effect on the Treated (ATT) (Equation 1) to estimate the outcome of the matching analysis. The ATT represents a counterfactual measure of the fraction of treatment units that would have been affected by deforestation in the absence of the treatment. ATT has not only been used as a proxy to calculate deforestation avoidance, but can also be used to estimate the amount of forest loss that is avoided by protection (Hernandez et al., 2021; Jusys, 2018; Ribas et al., 2021; Simmons et al., 2018). We use a relatively narrow measure of effectiveness, as it only focuses on one single conservation outcome (i.e., deforestation avoidance) and does not consider impacts on other conservation dimensions (e.g., dissuading illegal wildlife hunting). In spite of that, avoided deforestation represents a key conservation effectiveness measure in the regional context since forest cover loss is one of the most pressing drivers threatening Amazonian social-ecological systems.

To estimate the avoided deforestation area of each zone, we used the ATT as a fraction to obtain a non-conservative estimate of the area that would have been deforested in the counterfactual scenario (Equation 2).

$$\text{Avoided deforestation (area)} = \text{ATT} \times \text{treatment area} \quad (2)$$

Negative ATT values indicate that forest cover loss within the reserve (i.e., the treated area) is smaller than in the control areas outside the reserve without the same treatment (i.e., protection in the form of biosphere reserve designation). On the contrary, positive ATT values suggest that the treatment does not contribute to avoiding deforestation. To understand the scope of this effect, we also calculated the percentage of deforestation that we estimate was avoided in the counterfactual scenario (Equation 3).

$$\begin{aligned} \text{Avoided deforestation}(\%) = & \left[\frac{\text{deforestation avoidance}(\text{km}^2)}{\left[\text{forest loss 2000 - 2021}(\text{km}^2) \right.} \right. \\ & \left. \left. + \text{deforestation avoidance}(\text{km}^2) \right] \right] \times 100 \end{aligned} \quad (3)$$

2.5 | Deforestation avoidance: Accounting for context-specific deforestation pressure

When evaluating counterfactual effectiveness, deforestation pressure is one major factor that must be considered in relation to forest loss. Avoided deforestation measures alone can be misleading. It has been demonstrated that a protected area with no deforestation that happens to be in a landscape with limited deforestation pressure will generally result in low avoided deforestation values, disregarding its possible conservation effectiveness in a changing scenario in which deforestation pressure were to rise (Eklund et al., 2019). Accordingly, one must contrast avoided deforestation to absolute and relative deforestation pressure to which the sites of interest are subject.

We measured counterfactual deforestation pressure rates as the forest loss rate in the matched control group for each studied case. We then calculated relative deforestation pressure as the difference between the counterfactual deforestation pressure and the national forest loss rate, divided by the national forest loss rate (Equation 4), that is, how many times more or less deforestation is there within each matched control group relative to the whole country.

$$\begin{aligned} \text{Relative deforestation pressure} \\ = & (\text{counterfactual deforestation pressure} - \\ & \text{national deforestation rate}) / \text{national deforestation rate} \end{aligned} \quad (4)$$

2.6 | Data collection and analysis covariates selection

A crucial step in the matching process is the selection of all possible covariates related to the outcome and (Ribas et al., 2021; Schleicher et al., 2020). Informed by the scholarly literature analyzing the factors and processes that have driven forest cover loss in the Amazon River basin, we selected a set of biophysical and social economic variables to use in our model. The dataset consists of 12 vector and raster-based geospatial data layers (Table 3). We considered the biosphere reserves' distance to roads and populated sites as proxies for infrastructure and market access, critical

TABLE 3 Geospatial dataset used in this study.

Variable	Description	Time frame (year/s)	Resolution	Source
Forest loss	Yearly forest cover loss detected in the years 2001–2021	2001–2021	3 arc-seconds	Hansen database v 1.9
Tree cover 2000	Baseline forest coverage on the year 2000.	2000	3 arc-seconds	Hansen database v 1.9
Annual Mean Temperature	Mean annual temperature in (C° decimals)	1970–2000	30 arc-seconds	WorldClim1
Annual Precipitation	Annual precipitation (mm)	1970–2000	30 arc-seconds	WorldClim1
Precipitation Seasonality	Precipitation seasonality (Coefficient of Variation)	1970–2000	30 arc-seconds	WorldClim1
Elevation	Digital Elevation Model from the Shuttle Radar Topography Mission (SRTM) 1.4	2000	3 arc-seconds	International Centre for Tropical Agriculture (CIAT)
Slope	Slope percentage values extracted from the elevation data by using the Slope GDAL tool on QGIS (GDAL/OGR contributors, 2020)	—	3 arc-seconds	Slope percentage values extracted from the elevation data by using the Slope GDAL tool on QGIS (GDAL/OGR contributors, 2020)
Distance to roads	Distance in pixels to the closest road	2022	3 arc-seconds	Open Street Map
Distance to cities	Distance in pixels to the closest city	— / 2017 / 2022	3 arc-seconds	<ul style="list-style-type: none"> Ecuador: Instituto Geografico Militar (IGM) poblados layer Peru: INEI 2017 population centers census (Geogpsperu). Selected for places over 5000 inhabitants Bolivia: Open Street Map (Humanitarian data exchange)
Biosphere reserve limit and zonation		2022	Vector	UNESCO-MAB project internal database
Protected areas	Areas under diverse land protection strategies	2022	Vector	WDPA
Indigenous Territories	Titled and Untitled Indigenous territories	2022	Vector	RAISG

Note: GIS data used for the matching analysis.

factors underlying tropical deforestation patterns (Barber et al., 2014; Pfaff et al., 2007; Sader & Joyce, 1988). Data on population sites was obtained from different sources for each country (Table 3). Further, we selected five biophysical and climatic variables (elevation, slope, annual precipitation, annual temperature, and seasonality) correlated to agricultural land suitability (Amin et al., 2019; Armenteras et al., 2011; Laurance et al., 2002; Sader & Joyce, 1988; Silva Junior et al., 2021) and climatic variation (Dávalos et al., 2011) considered determinant factors underlying

forest cover conversion to agricultural land uses (Armenteras et al., 2011).

Lastly, we included additional information on land governance in order to exclude other protected areas and Indigenous territories from the matching analysis. Given that these areas are known to significantly impact deforestation pressure (Baragwanath & Bayi, 2020; Blackman et al., 2017; Nolte et al., 2013; Pacheco & Meyer, 2022), they could act as hidden confounders, and we therefore removed them from the calculation of deforestation pressure.

2.7 | Forest cover and yearly forest cover loss

We calculated yearly forest cover loss for the period 2000–2021 using the Global Forest Change (GFC) project database (Hansen et al., 2013). GFC builds on Landsat satellite imagery to track tree cover changes globally at 30×30 m spatial resolution (Elias et al., 2013; Galiatsatos et al., 2020; Milodowski et al., 2017; Steininger et al., 2001). We followed the designated Earth Engine analysis protocol (Google Developers, 2021) to quantify yearly loss in the studied areas. The dataset maps tree cover considering all vegetation taller than five meters height and encodes it as the percentage of tree canopy closure. The tree cover extent baseline is set in the year 2000, and tree cover loss is detected annually for the subsequent years.

In order to limit uncertainty in the spatial layer acting as the base for our analysis, and to keep the focus on forest ecosystems, we removed from the analysis, areas where the canopy is less dense, such as those interspersed in grasslands (e.g., forest fragments), wetlands, and some degraded areas. We defined the forest cover baseline by selecting pixels with a canopy cover of at least 80%, and then performing a juxtaposition analysis (Greenlee, 1987; Johnston, 1998) before aggregating to a 3 arc-second resolution, to select forest patches of at least six contiguous pixels (i.e., at least 0.5 ha) in the year 2000. Yearly forest cover loss was subsequently marked over that forest cover baseline.

We then also calculated the percentage of avoided deforestation (Equation 2) in the biosphere reserves to provide a comparative measure of the conservation effectiveness in deterring deforestation across the biosphere reserves.

QGIS (QGIS Development Team, 2021) was used to process most of the GIS files used in the matching analysis. All georeferenced data were standardized to a WGS 84 (EPSG 4326) CRS and maximum 3 arc-second resolution for the raster. Open Street Map data were downloaded at the Geofabrik server, and the bioclimatic data were retrieved through the WorldClim server when performing the data extraction. Distance rasters were created by rasterizing the data over a blank raster which was created from the forest layer, and then by using the GDAL proximity tool (GDAL/OGR contributors, 2020) on QGIS.

2.8 | Matching quality assessment

We evaluated the quality of the matching in the following ways. First, we assessed the matching representativity (i.e., the fraction of data-points that could be matched in each control group). Second, we calculated the Abadie

Imbens Standard Error (AI SE) (Abadie & Imbens, 2006). Third, we assessed the covariate balance improvement of the matching process through a Kolmogorov–Smirnov test, a measurement of the standard difference in means, and the variable ratio before and after the matching. We also interpreted visually the variable density curves for the matched and un-matched treatment and control groups (Supporting Information: Match balance). Fourth, we assessed how forest cover loss corresponded within the matching procedure, that is, the fraction of forest cover loss on treatment and control areas before and after the matching (Supporting Information: Complete match results). Fifth, we visualized the Region of Common support of the original dataset to assess the covariate distribution overlap between control and possible treatment units, to detect possible singularities of the treatment areas (Details and results available in the data repository: Region of Common support). Finally, compiled contextual information was gathered on land tenure and overlapping threats (Supporting Information: Additional Information on the studied biosphere reserves), and differences in zonation were assessed using Kruskal–Wallis test due to the small sample size, as it is not possible to ensure distribution normality.

3 | RESULTS

3.1 | Conservation effectiveness of Amazonian biosphere reserves in deterring deforestation

Regarding our first research question, “Are the biosphere reserves effective at avoiding deforestation?”, our results showed that five out of the seven biosphere reserves studied were effective in buffering deforestation at the biosphere reserve level (i.e., negative ATT estimate values) during the period from 2000 to 2021. Two biosphere reserves (Podocarpus-El C ndor in Ecuador and Beni BR in Bolivia) experienced more deforestation than what was measured in the matched control groups. These results are notable since deforestation avoidance in four biosphere reserves was effective under deforestation pressure conditions higher than what we found at the national level (Table 4).

Avoided deforestation during these two decades ranged from 176.54 km² to 4744.72 km². These results indicate that the designation of biosphere reserves contributed to reducing deforestation rates between 9.11% and 79.77% relative to what would be expected in their absence (i.e., the deforestation detected in counterfactual scenarios). On the other hand, the results showed that total deforestation in one of the biosphere reserves (i.e., Podocarpus—El C ndor BR in Ecuador) was about

TABLE 4 Summarized results of the general matching results for the whole biosphere reserve areas.

	Sumaco	Yasuní	Podocarpus	BIOAY	Manu	Beni	Pilón Lajas
ATT	−0.0177	−0.1731	0.0146	−0.061	−0.0538	0.0152	−0.0799
Number of matches	72,317	50,822	44,435	168,642	183,100	18,271	25,515
Representativity	0.5200	0.1324	0.3606	0.6910	0.5750	0.9992	0.4293
AI SE	0.0050	0.0022	0.0014	0.0040	0.0045	0.0242	0.0067
Match balance	Very good	Good	Very good	Very good	Very good	Fair	Good
Forest loss 2001–2021 km ²	1759.42	1203.07	853.29	3058.36	753.79	233.40	391.21
Deforestation pressure relative to national forest loss	0.283	0.616	−0.608	1.288	−0.322	−0.285	0.007
Avoided deforestation km ²	−176.536	−4744.715	166.224	−1100.69	−1301.638	20.517	−323.561
Percentage of avoided deforestation	−9.119	−79.773	24.193	−26.465	−63.327	9.638	−45.268

Note: Main effectiveness results, ATT and Avoided deforestation area, are marked in bold. The information provided in the row “Match Balance” represents our own assessment of the match quality after analyzing the Kolmogorov–Smirnov test, standard mean differences, variable ratios, and variable curb densities before and after the matching (see Section 2). Since all of the balance evaluation metrics are relative, we automatically classified each match's measurements into 4 quality categories using Fishers natural breaks and outliers, depending on its position in relation to all matches (for detailed information and visualization see [Supporting Information: Match evaluation](#)). For the ATT, avoided deforestation and percentage of deforestation avoided, negative numbers indicate that the reserve has less deforestation than its counterfactual scenario, while positive numbers indicate the opposite. For the deforestation pressure relative to the national average, numbers between −1 and 0 indicate that the pressure is less than the national average (as a fraction); positive numbers indicate the opposite.

24% higher (166.22 km²) than what we estimated in a counterfactual scenario, even under lower deforestation pressure (Table 4).

Following up with our second research question “Does conservation effectiveness vary according to the conservation and management zonation system of biosphere reserves?”, the results did not show significant differences in the mean ATT values among the different conservation and management zones when applying the Kruskal–Wallis test. Despite this, at the biosphere reserve level, we found that the transition areas were generally more effective (i.e., showed negative ATT values, indicating a stronger reduction in deforestation) in deterring deforestation in five out of seven biosphere reserves, followed by buffer zones (four out of seven) and core areas (three out of seven), respectively (Figure 3a; Table 5). The percentage of avoided deforestation was relatively higher in the buffer zones, which reduced forest cover loss by approximately 50%–93% compared with counterfactual scenarios without protection (Table 5).

We found that the overall range of results obtained from core areas showed less effectiveness, with ATT values closer to zero and the lowest effectiveness mean (highest mean ATT value) compared to the other conservation and management zones, although individual tendencies for each reserve were diverse (Figure 3 and [Supporting Information: Additional information for Figure 3](#)). Buffer zones had the highest mean effectiveness but estimates in both buffer zones and transition areas varied significantly. We performed the Kruskal–

Wallis test to understand the differences between the studied groups and found: No significant differences between groups in ATT values (Chi square = 0.67532, $p = 0.7134$, $df = 2$), No significant differences in the fraction of Matched/treated control groups (Chi square = 3.9481, $p = 0.1389$, $df = 2$), No significant differences for the Abadie–Imbens standard error (Chi square = 3.9481, $p = 0.1389$, $df = 2$) and significant differences for the forest loss in the matched control group relative to the national average (Chi square = 11.451, $p = 0.003262$, $df = 2$). In the last case, a subsequent pairwise comparisons using Wilcoxon rank sum exact test showed that core zones differed significantly from both buffer and transition zones ($p = 0.0166$, and $p = 0.0035$, respectively).

3.2 | Deforestation avoidance relative to deforestation pressures

The measurement of deforestation pressure, as an additional outcome from the matching that is calculated independently from the conservation effectiveness, helps us to understand and contextualize the results. Relative pressure was significantly lower in the core areas than in buffer zones and transition areas, respectively (Kruskal–Wallis, Chi square = 11, $p = 0.003$, Figure 3d). Mean relative pressure was the highest in transition zones but fell within a wider range of results (Figure 3d and [Supporting Information: Additional material for Figure S3](#)).

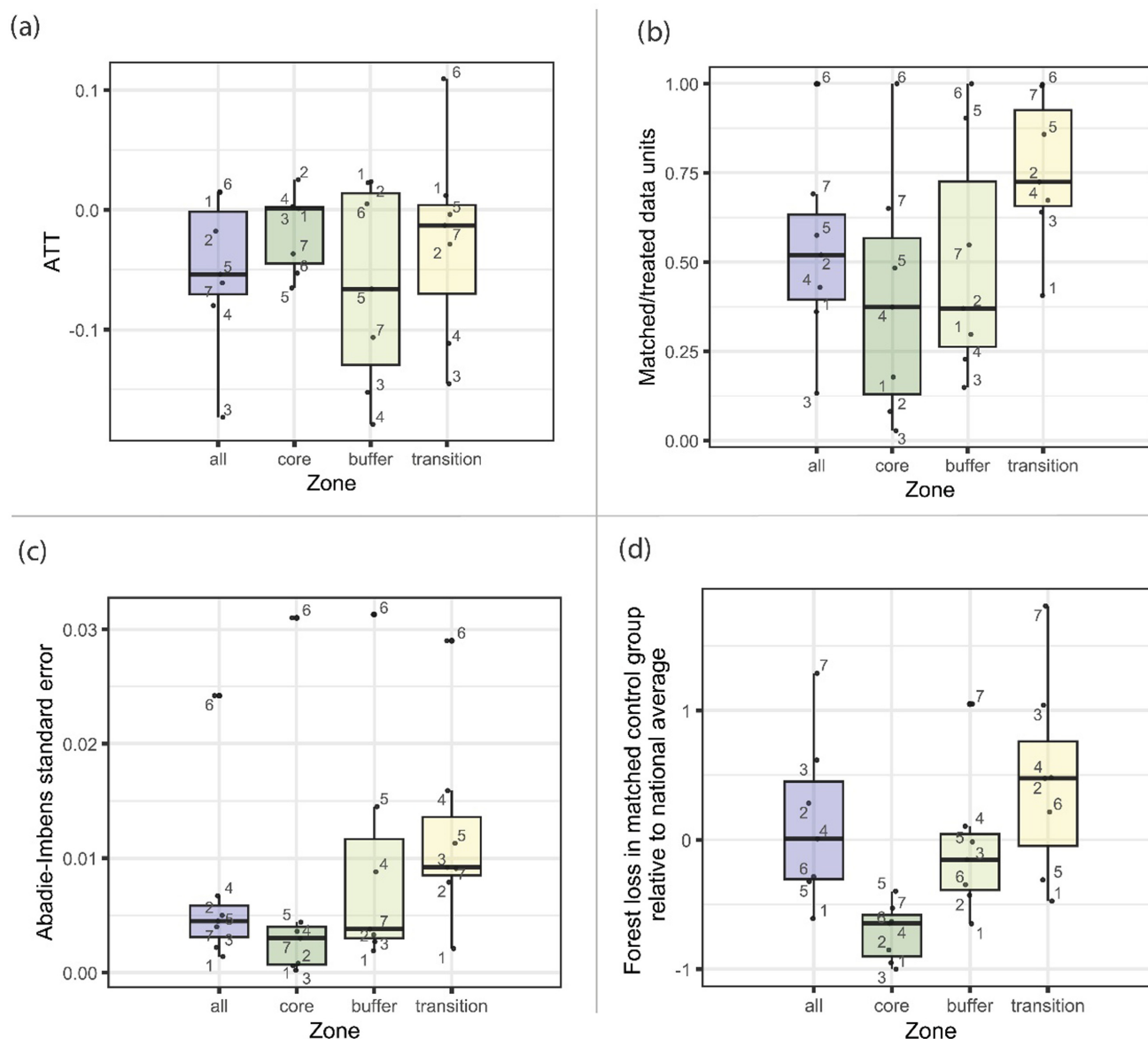


FIGURE 3 Main results of the matching. (a) ATT, (b) representativity, (c) Abadie-Imbens standard error (AI-SE), and (d) Relative pressure of the studied reserves and zones. Biosphere reserves are marked as such: 1-Podocarpus—El Cóndor, 2-Sumaco, 3-Yasuni, 4-Pilón Lajas, 5-Manu, 6-Beni, 7-Bioay. (a) The Average effect of the Treatment on the treated (ATT) is a direct result of the matching process and estimates the effectiveness of the reserves and zones in avoiding deforestation, lower (more negative numbers) indicate higher effectiveness. (b and c) Representativity and the Abadie Imbens Standard error are both metrics that refer to the matching process quality in different ways. Higher representativity numbers indicate that we were able to match a higher percentage of treatment data points in each case; lower Abadie Imbens standard error values indicate a higher quality of the matching of each case. (d) Relative deforestation pressure is an estimate that we derive from the counterfactual pressure (the deforestation in the matched control group) calculated in the matching process. Positive numbers indicate how many times more deforestation is there, and negative numbers indicate how many times less deforestation is there in the matched control group relative to the national average.

3.3 | Matching method performance

We obtained 943.221 data points for Ecuador, 3.153.107 data points for Bolivia and 5.468.132 for Peru. After testing and adjusting parameters by running preliminary iterations, 28 final matching runs with standard 0.25 caliper values were processed (four for each of the reserves:

one for the whole biosphere reserve area and three for each of the studied zones). Additionally, we performed 20 runs with alternative caliper values. Full results and evaluation metrics can be found in the [Supporting Information](#).

Within our third research question we aimed at understanding “How the matching analysis performances

TABLE 5 Summarized results of the matching of the internal zoning areas of the biosphere reserves.

	Sumaco	Yasuní	Podocarpus	BIOAY	Manu	Beni	Pilón Lajas
<i>Core area</i>							
ATT	0.0252	0.0012	0.0023	−0.0367	−0.0652	−0.0528	0.0027
Number of matches	2380	3213	4681	9915	119,092	5059	9534
Representativity	0.0815	0.0273	0.1777	0.6502	0.4835	1	0.3738
AI SE	0.0008	0.0002	0.0006	0.0030	0.0044	0.0310	0.0036
Match balance	Good	Fair	Very good	Very good	Good	Fair	Good
Forest loss 2001–2021 km ²	2055.0	31.22	30.44	26.93	327.19	10.13	85.11
Deforestation pressure relative to national forest loss	−0.851	−1	−0.950	−0.529	−0.398	−0.632	−0.645
Avoided deforestation km ²	32.99	10.196	5.102	−40.655	−1107.244	−18.469	4.721
Percentage of deforestation avoided	NA	48.497	20.136	−60.154	−77.190	−64.579	5.873
<i>Buffer zone</i>							
ATT	0.0227	−0.1523	0.0236	−0.1065	−0.066	0.005	−0.1791
Number of matches	11,338	32,798	11,516	84,500	30,565	8627	3651
Representativity	0.3700	0.1485	0.2974	0.5481	0.9034	0.9997	0.2278
AI SE	0.0033	0.0027	0.0019	0.0038	0.0145	0.0313	0.0088
Match balance	Very good	Good	Very good	Good	Very good	Fair	Good
Forest loss 2001–2021 km ²	229.29	170.90	220.28	1095.49	154.40	95.49	69.80
Deforestation pressure relative to national forest loss	−0.428	−0.154	−0.649	1.050	−0.016	−0.347	0.105
Avoided deforestation km ²	49.342	−2343.761	80.239	−1135.209	−177.621	3.179	−192.605
Percentage of deforestation avoided	27.420	−93.204	57.297	−50.890	−53.497	3.444	−73.400
<i>Transition area</i>							
ATT	−0.0286	−0.1452	0.0119	−0.013	−0.0038	0.1095	−0.1114
Number of matches	57,380	28,910	23,628	74,153	32,860	4585	12,052
Representativity	0.7244	0.6399	0.4062	0.9936	0.8578	0.9976	0.6732
AI SE	0.0079	0.0092	0.0021	0.0091	0.0113	0.0290	0.0159
Match balance	Very good	Good	Very good	Good	Very good	Fair	Good
Forest loss 2001–2021 km ²	1497.18	1000.93	602.56	1935.90	272.18	127.77	236.27
Deforestation pressure relative to national forest loss	0.475	1.041	−0.473	1.807	−0.310	0.215	0.480
Avoided deforestation km ²	−163.478	−555.327	69.581	−81.368	−17.014	40.297	−138.665
Percentage of avoided deforestation	−9.844	−35.684	13.055	−4.034	−5.883	46.068	−36.984

Note: Main effectiveness results, ATT and Avoided deforestation area, are marked in bold.

vary in quality and reliability across biosphere reserves and their zonation system?”. We found that match quality was generally good for Podocarpus-El Cónor, Sumaco, Manu and BIOAY, which makes the effectiveness estimates we obtained for these reserves sounder. On the other hand, Yasuní, Beni and Pilon Lajas obtained poorer matches.

Different indicators of match quality point to issues for different studied sites (Figure 3). To allow us to interpret these results, quality indicators for Beni were considered as outliers. Representativity is an indicator that varied largely at the standard caliper level. Core areas were more difficult to match, on average, with Yasuni and Sumaco having the lowest percentage of matched

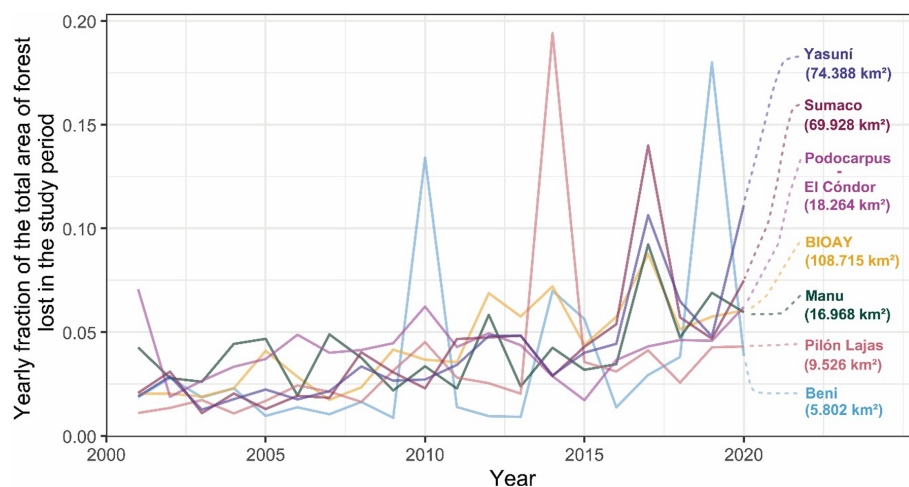


FIGURE 4 Forest loss trends across the Biosphere Reserves in the 2001–2020 time period (data source: Hansen database v 1.9). The total amount of forest loss is quantified below the names of the reserves. Note that the scale of the y axis is logarithmic.

treatment points (2.71 and 8.15%, respectively), while values got up to 65% for BIOAY and transition zones overall ranged from 40% to 99%.

4 | DISCUSSION

Our results reveal a heterogeneous pattern of conservation effectiveness across the seven biosphere reserves analyzed in the western Amazon. Five of the seven biosphere reserves deterred deforestation compared to counterfactual scenarios—Sumaco and Yasuní in Ecuador, BIOAY and Manu in Peru, and Pilon Lajas in Bolivia. In other words, our results showed that forest cover loss in these biosphere reserves was 9%–79% lower than expected without UNESCO's Biosphere Reserve designation. These findings indicate that, under certain conditions, biosphere reserve designation can contribute meaningfully to forest conservation outcomes, even in contexts characterized by high deforestation pressure. Similar mixed but generally positive conservation outcomes have been reported for other biosphere reserves worldwide, particularly in relation to biodiversity protection and ecosystem service provision (Auliz-Ortiz et al., 2024; Gohr et al., 2025).

However, the effectiveness of biosphere reserves was far from uniform. In one biosphere reserve (Beni in Bolivia), designation did not result in measurable deforestation avoidance relative to counterfactual areas. Finally, one biosphere reserve (Podocarpus—El Cónдор in Ecuador) exhibited higher deforestation rates than expected. This outcome should be interpreted cautiously, as the biosphere reserve was designated in 2007, whereas our analysis covers the period 2002–2021. This is also the case for the BIOAY biosphere reserve which was designated in 2010. Temporal mismatches between designation and evaluation periods have been shown to

influence estimates of protected area effectiveness, particularly when pre-designation dynamics are not fully accounted for (Li et al., 2024).

However, the effectiveness of biosphere reserves was far from uniform. In one biosphere reserve (Beni BR in Bolivia), designation did not result in measurable deforestation avoidance relative to counterfactual areas. Finally, one biosphere reserve (Podocarpus—El Cónдор BR in Ecuador) exhibited higher deforestation rates than expected. We find this outcome to be coherent within our study context in which the selected biosphere reserves act as a representation of a broader mosaic of Amazonian socio-environmental realities, where we can find diverse implementations of conservation and sustainable development approaches in the face of rapid transformation. Specifically, the Yasuní (Ecuador), Sumaco (Ecuador), Pilon Lajas (Bolivia), BIOAY (Peru), and Manu (Peru) biosphere reserves were the most effective in deterring forest loss, although the latter in a context of low deforestation pressure. BIOAY and Manu Biosphere Reserves in Peru consistently showed lower deforestation rates across their conservation and management zones than counterfactual scenarios. Beni Biosphere Reserve (Bolivia) presented mixed outcomes. At the biosphere reserve level, even under relatively low deforestation pressure, deforestation was higher than in counterfactual scenarios (Table 4). That is likely the result of the high deforestation rates detected in the transition area, which showed the least conservation effectiveness among the biosphere reserves under study (Figure 3a). The matching did not perform well in this case. A more in-depth exploration of the situation of this specific reserve can be found in the [Supporting Information](#) (Additional information and reflections on the studied areas).

The relatively strong performance of several biosphere reserves aligns with a growing body of evidence showing that conservation outcomes are often enhanced

in territories where Indigenous Peoples play a central role in land stewardship and governance (Fa et al., 2020; Garnett et al., 2018; Pratzner et al., 2023). Indigenous territories across the tropics have repeatedly been shown to experience lower deforestation rates and better conservation outcomes than areas outside of them, particularly where land tenure rights are recognized and governance institutions are strong (Baragwanath & Bayi, 2020; Schleicher et al., 2020; Sze et al., 2021). In the Amazon basin specifically, Indigenous territories have been highlighted as critical contributors to regional forest conservation, climate change mitigation and biodiversity conservation (Barreto et al., 2025; Fernández-Llamazares et al., 2020). Biosphere reserves that effectively integrate Indigenous governance systems and participatory decision-making processes may therefore be better positioned to deliver positive conservation outcomes (Barracough et al., 2025; Moreira-Muñoz et al., 2025), although this potential is highly context dependent.

Despite these encouraging results, deforestation rates across all biosphere reserves remained substantial and, in most cases, increased over time (Figure 4). This finding reflects broader regional trends in Amazonian deforestation driven by agricultural expansion, infrastructure development, and extractive activities (Science Panel for the Amazon, 2025), and highlights the limitations of area-based conservation instruments when operating within unfavorable political and economic contexts. In the next subsections, we explain how the context-specific socio-ecological characteristics of each biosphere reserve and the matching method affect both the effectiveness evaluation process and the deforestation avoidance results.

4.1 | Context-specific conditions underlie conservation effectiveness

This study assessed conservation effectiveness of seven biosphere reserves in three countries in the Western Amazon, using a matching approach that accounts for key biophysical and accessibility-related drivers of deforestation. While this allowed for a systematic comparison of observed and counterfactual forest cover loss, our results also highlight the importance of interpreting conservation outcomes in light of context-specific conditions that fall beyond the scope of the matching analysis. Differences in biophysical settings, land-use histories, governance arrangements, and socio-political contexts are likely to play a central role in explaining the heterogeneous effectiveness observed across the

biosphere reserves studied (den Braber et al., 2024; Li et al., 2024; Mouillot et al., 2024; Sinare et al., 2025). Located in the western Amazon River basin, the seven biosphere reserves encompass a topographic gradient ranging from roughly 160 to 4500+ meters above sea level in the transition between the Amazonian lowlands and the Andean montane region. These biophysical conditions go together with local climate variation that shapes a vast diversity of ecosystems representative of biomes ranging from tropical and subtropical grasslands, shrublands and savannas (e.g., Beni savanna), moist broadleaf forests (e.g., Bolivian and Peruvian Yungas, Ecuadorian Napo moist forests), to montane grasslands and shrublands (e.g., Andean wet punas and páramos). These conditions mediate historical processes linked to the occupation and colonization of these territories (Roosevelt, 2013). Whether the customary land of Indigenous groups inhabiting remote areas in the Amazonian lowlands (e.g., Manu National Park and Yasuni National Park in the respective Peruvian and Ecuadorian Biosphere Reserves) or the expanding frontiers where a diverse group of agents (e.g., farmers, goldminers, mestizos) have settled following market connections and opportunities boosted by official government programs and incentives (e.g., roadways extension, oil and mining concessions, hydropower dams, agricultural credits, land reform and land titling), the reality is that the seven biosphere reserves own geographic and biophysical features that mediate dominant economic activities and land use and cover dynamics that ultimately impact conservation effectiveness and forest cover loss in these territories.

Data variance in our results may explain the mixed outcomes of our study (Figure 3), highlighting the need of considering context-specific conditions in interpreting and assessing the conservation effectiveness of each biosphere reserve in deterring deforestation. Biosphere reserves are a conservation instrument that remains comparatively under-studied and not yet fully understood, often overlapping with pre-existing protected areas and a variety of territorial designations, situating them in a diverse landscape of governance arrangements, land use zoning, and different capacities for monitoring and implementation of environmental laws and regulations. Limited visibility and recognition of the reserves as area-based conservation institutions, even at the local scale, can hinder the implementation of their functions. Consequently, the implementation of management plans varies significantly across biosphere reserves due to limited or lack of financial resources and capacities. Indigenous Peoples and local rural communities play a critical role

in shaping land-use outcomes within many biosphere reserves, and often act as de facto stewards of forested landscapes even when situated outside of the scope of protected areas strategies (den Braber et al., 2024; Nolte et al., 2013; Virtanen et al., 2025). However, their capacity to do so is unevenly supported by national government and external organizations (Tran et al., 2020). Similarly, most biosphere reserves do not receive the same political-administrative support and economic resources that national governments usually concede to state protected areas (Sandström et al., 2025). These institutional asymmetries likely contribute to the mixed conservation outcomes observed.

Political, economic, and institutional factors across different countries mediate how each country prioritizes, funds, and supports local agencies overseeing national protected territories and Indigenous lands (covariates included in this study), impacting stakeholders' ability and conditions to manage these territories and govern the biosphere reserves. More, these factors also connect to governmental programs (e.g., infrastructure expansion, oil and mining concessions, agricultural credits) and market incentives (e.g., commodities demand) associated with activities driving forest cover loss expansion that threatens the biosphere reserves.

Social and cultural factors related to demography and the diversity of groups and populations in the biosphere reserves may mediate conflicting interests and trust-building among local stakeholders, impacting opportunities for inclusion and participation in decision-making arenas and their ability to collectively govern these territories (Sandström et al., 2025; Sinare et al., 2025). That is particularly relevant considering the extensive territories characterizing Amazonian biosphere reserves, where large distances and remote locations limit connectivity. Our covariates (i.e., distance to roads and distance to cities) capture the connectivity factor as direct drivers of forest cover loss, not as proxies explaining the degree with which diverse groups and people living in the biosphere reserves can meet and interact, essential for building trust and crafting strong governance arrangements (Ostrom, 1990).

4.2 | The effectiveness of biosphere reserve zoning in avoiding deforestation is context-specific

The effectiveness of biosphere reserves in avoiding deforestation is consistent with those in the literature, which show similar negative ATT values across other protected areas in Amazonia (Andam et al., 2008; Hernandez

et al., 2021; Nolte et al., 2013). On average, the most strictly protected zones in the biosphere reserves (i.e., core areas) were subject to significantly low deforestation pressure (Figure 3d), which may also explain their aggregated smaller deforestation avoidance values (i.e., higher mean ATT value) relative to the buffer zones and transition areas (Figure 3a). The core areas in the biosphere reserves usually correspond to and overlap with protected territories and Indigenous lands designated at the national level and recognized by each country's official protected areas system. Despite the relatively lower deforestation pressure upon these areas (Figure 3d), conservation effectiveness in deterring deforestation varied across the core areas in the biosphere reserves and did not show a clear pattern. Whereas the core areas in Yasuní (i.e., national park), Sumaco (i.e., national park), and Podocarpus—El Cóndor (i.e., national parks and biological reserve) Biosphere Reserves in Ecuador were ineffective in deterring deforestation, those in the Manu (i.e., national park and a natural World Heritage site) and BIOAY (i.e., national park) Biosphere Reserves in Peru were effective. In Bolivia, although not significant, deforestation avoidance in the core areas showed opposite results, being effective in Pilon Lajas (i.e., Indigenous territory) but not in the Beni Biosphere Reserve (i.e., biological station).

Conversely, the buffer zones and transition areas of the biosphere reserves follow a different pattern. When overlapping exists, notably with buffer zones, locally designated protected territories belong to categories that are not strictly protected, allowing the sustainable use and management of these territories (e.g., protective forests and communal areas in Peru). Accordingly, deforestation pressure in the buffer zones and transition areas is higher than in the core areas. That highlights their roles in avoiding forest cover loss in the biosphere reserves as a whole and in protecting the core areas and the strictly protected areas of each country, in particular. These results highlight the particular role that transition areas play, even under higher deforestation pressure conditions, in deterring deforestation drivers from expanding into the inner and strictly more protected areas of biosphere reserves, and their importance in promoting more sustainable pathways.

It is important to note that avoided deforestation is as much a measure of effectiveness as it is of the threat level upon the biosphere reserves since both characteristics are deeply intertwined (Eklund et al., 2019). Our results confirmed this pattern, and conservation effectiveness in deterring deforestation (estimate ATT values) was higher as deforestation pressure on the biosphere reserves increased (Figure 5).

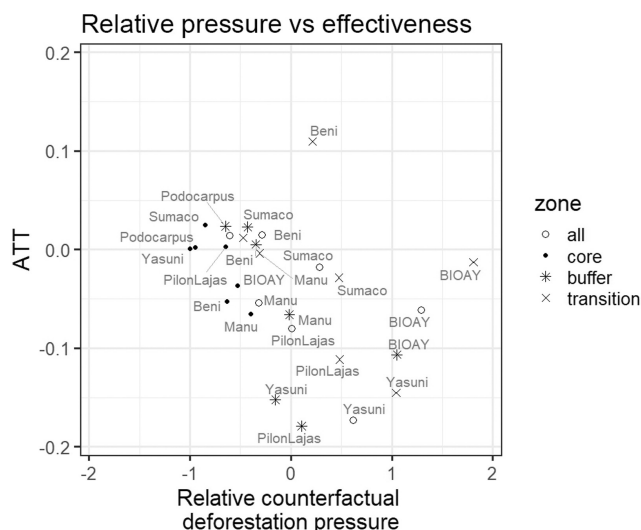


FIGURE 5 How do the effectiveness estimate, represented by the Average Treatment effect on the Treated (ATT), and the relative pressure estimate relate to each other within our matching results? Studied areas under lower deforestation pressure (relative to their national average) tend to obtain less effective effectiveness results (more positive ATT values).

4.3 | Applying matching methods across a strikingly diverse region

Analyzing the match balance is a key aspect of the matching assessment (Ribas et al., 2021; Schleicher et al., 2020; Stuart, 2010). It is nearly impossible to obtain a perfect covariate balance, but understanding the relative differences between matches was essential in helping us to establish quality criteria. Too many covariates to analyze at the same time is a common challenge when matching, which is referred to as the curse of dimensionality. We aimed at avoiding this by using the Mahalanobis distance metric to measure match similarity, which is a robust metric that can work well with a large number of data points under eight covariates (Zhao, 2004). In most cases, the matching process was successful resulting in reliable results. Despite that, we faced critical challenges in finding adequate matches for some of the conservation and protection zones of the biosphere reserves and selected covariates. For instance, it was challenging to navigate the trade-off between representativeness and matching quality in Yasuni buffer zone and core area, while bioclimatic variables (notably seasonality and annual precipitation) constrained the matching between treatment and control groups throughout the analysis, particularly in Pilon Lajas and Beni ([Supporting Information: Match evaluation](#)). When considering both balance and representativity results the biosphere reserves in Peru had the best match quality outcome.

4.4 | Limitations

Matching methods are effective at bias reduction when comparing between data groups, yet they have inherent methodological limitations (Schleicher et al., 2020) and should not be considered as the ultimate bullet-proof method to assess conservation effectiveness (Eklund et al., 2019; Ribas et al., 2021; Schleicher et al., 2020; Stuart, 2010). When doing a matching analysis, the reliability and quality that we can achieve is always limited to the original data and the study question. We based our covariate selection on scientific literature. However, we cannot rule out the possibility that missing covariates (i.e., distance to rivers, which was left out due to data inaccuracies) might have introduced biases in our results. Hidden confounders could also introduce additional biases (Schleicher et al., 2020). We met the assumption of the region of common support, by which the possibility of creating matches depends on a pre-existing overlap of the covariate distribution between the possible treatment and control units (Schleicher et al., 2020; Stuart, 2010), in most of our data (Data repository: Region of Common Support). Yet, some of the studied reserves and zonation were strikingly different to match all the controls or achieve acceptable covariate balance (i.e., the Yasuni core area and most of the Beni biosphere reserve).

It is also important to note that the quality of the matching results is also context dependent. Therefore, in order to get a reliable assessment of deforestation avoidance it is necessary to contextualize the matching and understand the variables being evaluated, the social-ecological contexts of the studied areas and their existing threat levels (Baragwanath & Bayi, 2020; Eklund et al., 2019; Nolte et al., 2013). This context dependency, which inevitably affects quality (e.g. the balance of the matches, the fraction of treatment points that can be matched to controls, the dissimilarity between treatment and matched pairs...) is not usually assessed or even reported in conservation assessments (see Schleicher et al., 2020). We also acknowledge that our measure of conservation effectiveness only focuses on deforestation avoidance and does not reflect many other ways in which the reserves under study could be effective (e.g., combatting overharvesting of particular species and illegal activities, restoring previously deforested areas).

As for the way in which we measure deforestation, the Hansen GFC database is based on a set of Landsat satellite images that are processed to fit certain deforestation criteria on a global scale (Hansen et al., 2013). This creates a globally consistent map but can limit the accuracy of the measurements at the local scale, which can be largely dependent on the types of disturbances and can change depending on location (Milodowski et al., 2017).

and the vegetation typology. For the first half of our study period, this dataset has been found to possibly underestimate deforestation within our study area (Vancutsem et al., 2021) and on the contrary it could overestimate deforestation to some extent toward the second end of the 2001–2021 period (Kinnebrew et al., 2022; Vancutsem et al., 2021). Uncertainty in classification is bigger in transition and regrowth areas, river banks, or areas close to the classification criteria thresholds (Elias et al., 2013; Kinnebrew et al., 2022). This uncertainty could be affecting some of our case studies (e.g., Beni biosphere reserve) more than others due to their intrinsic landscape characteristics. The dataset is also not intended to distinguish forest types or whether forests are being managed sustainably or unsustainably (Elias et al., 2013). Uncertainties can also arise from spatial data sources, taking particular relevance for the definition of the covariate values in our study. As an example, the distance to cities layer was created by using the latest census data for each of the studied countries. This allowed us to create a relatively standardized layer for each of the countries, but was limiting the covariate information to the available data at the given time, which didn't cover the whole study period and might introduce uncertainties from the original census data.

Conservation effectiveness estimates in our study should be interpreted cautiously: by design our analysis covers a temporal period of 2000–2021 for the studied impact, however, the BIOAY (Peru) and Sumaco (Ecuador) Biosphere Reserves were designated in 2010 and 2007, respectively. We might be underestimating their potential in avoiding deforestation by examining forest cover loss during years in which these biosphere reserves (i.e., the treatment group) had not been in place. It is worth noting that effectiveness has been found to appear within a time lag after designation of a protected area, particularly in non-strict protection zones (Li et al., 2024). Something to point out from our methodological design is that pre and post designation dynamics, which can be particularly influential due to the complex nature of Biosphere reserve designation and implementation dynamics, were not accounted for in our study. Rather, we approached the presented counterfactual analysis as a way to study the potential of these reserves as institutions, in a much broader time-scale, by studying an alternate scenario without designation.

Finally, regarding the selection of covariates mediating Amazonian forest cover loss, we suggest future research to consider fire events as a key covariate in the matching analysis. Increasingly fire events in Amazonia have gone in tandem with forest cover loss. Therefore, considering fire occurrence as a covariate in matching treatment and control groups may improve the study

capacity to match points subjected to similar conditions underlying our dependent variable (i.e., forest cover loss).

While matching approaches must be applied with careful attention to their assumptions and limitations, they remain a powerful means of implementing counterfactual causal frameworks for impact assessment in conservation science (Schleicher et al., 2020). When applied at large spatial scales, such approaches offer a valuable opportunity to generate comparable, relative estimates of conservation effectiveness across multiple biosphere reserves and management contexts. This makes them particularly well suited for informing assessments of the global biosphere reserve network, as well as for comparative analyses of protected area systems operating under similar conditions or facing analogous pressures.

At the same time, the application of large-scale matching analyses requires substantial effort to appropriately contextualize results, including careful consideration of social, ecological, and governance conditions, as well as pre and post-designation dynamics that observed outcomes (Schleicher et al., 2020; Sharkey et al., 2025). When combined with such contextual interpretation, these methods can support policy-relevant evaluations of biosphere reserve performance at broad scales. In this sense, large-scale counterfactual analyses can contribute to evidence-based reflection on protected area strategies and offer actionable insights relevant to the monitoring and implementation of international conservation frameworks, including the Global Biodiversity Framework (Wu et al., 2025).

5 | CONCLUSIONS

The international community can draw important lessons from the fact that biosphere reserves were found to be generally effective at avoiding deforestation across the Western Amazon. Although these biosphere reserves are by no means immune to deforestation and biodiversity loss, this article shows that they generally offer greater effectiveness than comparable areas while aiming at reconciling diverse land use systems and safeguarding natural and managed ecosystems in the Amazon. The ability of transition areas and buffer zones within biosphere reserves to effectively prevent deforestation—even under higher deforestation pressures and despite their relatively less strict protection status compared to core areas—highlights their potential and critical role in reconciling biodiversity conservation with the sustainable use and management of landscapes. That adds to growing evidence of the importance of putting people at the center of conservation action within and across landscapes (Reed, 2016, 2019), and how pluralistic and multi-level

governance models such as the ones in place in biosphere reserves can support biodiversity conservation and its sustainable use (Dabard et al., 2025). Supporting Indigenous land tenure in the Amazon is essential for safeguarding Indigenous peoples' lives and livelihoods, upholding their rights, and contributing to more just, pluralistic, and participatory approaches to biodiversity conservation (Fernández-Llamazares et al., 2020; Moreira-Muñoz et al., 2025). A corollary to this is that biosphere reserves should receive increasing recognition in national and international biodiversity science, policy, and practice. Accordingly, biosphere reserves can make a significant contribution towards achieving the targets set out in the Kunming-Montreal Global Biodiversity Framework (Barracough et al., 2025; Wu et al., 2025). The Convention on Biological Diversity could build upon the experiences and the regional roles and models that biosphere reserves have provided in supporting the transformative change so widely called upon.

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DATA AVAILABILITY STATEMENT

All geospatial information used in the analysis can be accessed using the public geospatial databases and open sources cited in this article. Authors do not claim ownership of the used data. Information on sampled data as well as the codes and additional information to replicate the analysis and the presented results, evaluations and figures is available at: <https://doi.org/10.6084/m9.figshare.29822081>.

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SUPPORTING INFORMATION

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